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BOOK OF PAPERS

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- Contents
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C O N T E N T S

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RADIATION PROTECTION ASPECTS IN DECOMMISSIONING OF A FUEL REPROCESSING PLANT

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INTRODUCTION

The Fuel Reprocessing Plant at Trombay (1) was designed as a pilot plant to develop the technology of chemical processing of irradiated nuclear fuel. The chemical process employed was a Purex type, solvent extraction process. The aluminium clad natural uranium fuel from the CIRUS reactor was successfully processed in this Plant (2).

After a long period of operation, it was decided to carry out a partial decommissioning of this plant with a view to modifying the plant to increase its capacity, to improve the safety features and to extend its useful life. Any major modifications of a nuclear chemical plant require massive decontamination and decommissioning efforts to permit easy and unrestricted entry of personnel for the new installation work.

In the case of the Trombay Fuel Reprocessing Plant, these jobs have been successfully carried out. Salient features of these efforts have already been described (3). The purpose of the present paper is to provide information on the radiation protection aspects of such an operation.

DISMANTLING AND REMOVAL OF PROCESS EQUIPMENT

The recoverable nuclear materials from the vessels and columns were removed by repeated rinsings. The internal decontamination campaign for these was then started.

Initial radiation surveys were carried out by lowering wide range (5 rems) self reading dosimeters attached to long thin rods or strings. After several washings, the radiation levels on the equipments were reduced to levels below 250 mr/hr. There were spots of much higher levels which could be located only after entering the cells. These latter surveys were carried out by using wide range dose measureing instruments with extendable arms (upto 4 meters). The dismantling of the equipments and piping was done by electric arc cutting. The heavy equipments were lifted by a crane through the cell top plug openings and stored in a previously unused cell.

When the dissolver off-gas filter was disconnected from the suction side, the gamma radiation levels in front of the flanged openings were unexpectedly higher (nearly 10 times) than the gamma dose rates on the body of the S.S. container of the filter. This was due to bremstrahlung low energy X-rays, which could be easily shielded by a metal cap. If not anticipated or measured such high radiation

through the flanged opening can cause excessive exposures.

Beta radiation dose rates were also contributing to the workers exposures. Due to spills of hold up liquids from cut pipes or contaminated stagings and walls, high beta dose rates 30 to 50 times the gamma dose rates were observed. The beta dose rate levels were measured with a small ionization chamber type instrument. Its response was calibrated using an extrapolation chamber. For the immediate estimation of the beta doses received by the workers, use was made of a pair of dosimeters, one of which was shielded from beta rays by means of a G.E. tube. The difference in the readings of the two dosimeters was used to estimate the beta ray doses. This procedure was also calibrated for dosimetry of beta rays from aged fission products, using an extrapolation chamber.

Each worker was provided with a full set of protective equipments consisting of undergarments, coverall, ventilated one piece plastic suit covering the body from the toes to the neck, shell cap, transparent plastic hood covering the entire head and face, two pairs of rubber hand gloves and ankle high rubber gum boots. Each worker used a supplied air positive pressure respirator connected to a compressed air supply station by a 10 to 20 meters long rubber pressure hose. With this protective gear, a worker could carry out his assigned jobs only upto 30 minutes. The heat stress and excessive sweating caused fatigue and exhaustion. In the initial stages the high radiation levels set limits on working time to less than 5 minutes.

DECONTAMINATION OF CELLS AND OTHER AREAS

A large effort was spent on the decontamination of cell interiors and adjacent areas such as dilution cells, rod storage bay, tunnels in the sampling gallery, the cubicles housing the service pumps and the plutonium processing areas.

By mopping and hosing withhigh pressure water much of the removable contamination was removed. Fixed contamination on the walls and floors had to be removed by pneumatic drilling. To prevent spread of contamination due to airborne dust or by liquids, only small areas were fully decontaminated and reclamied at a time. During drilling of concrete surfaces water spraying of the spot being removed, helped in limiting the amount of airborne dust. All the drilled surfaces were refilled with fresh concrete and all surfaces were given a fresh coat of paint.

MANAGEMENT OF RADIOACTIVE WASTES

The liquid wastes generated during the internal decontamination of the process equipments were concentrated by evaporation and transferred to the underground storage tanks.

During the dismantling operations, the unusable items and a large quantity of metal scrap produced were put in drums and sent as active waste for burial. Large unwieldy items were decontaminated to an extent permitting their cutting and handling. The drums were moved by crane and put in the shielded transport containers.

During the decontamination work of the cells and other areas,

large amounts of solid and liquid wastes were generated. The solid waste consisted of cotton mops and rags, concrete rubble, small size metallic scrap and polythene and rubber items. These were packed in plastic bags. For safe transport the bags were put in drums which were transported to the disposal site and brought back for reuse. The liquid wastes were put in plastic carboys which were again put in secondary metal containers before shipment for immobilization and burial. The organic liquid waste from one cell was sent for disposal in a S.S. tank in a single lot.

Many of the operations gave rise to airborne radioactivity. The normal plant ventilation and filtration system could adequately handle these situations. A meticulous record was maintained on the release of activity through the stack.

Table 1 gives the volumes of solid and liquid wastes and activities released through the stack.

TABLE 1. Discharges during Decommissioning

Solid Wastes						Liquid Wastes				Stack Release	
Bags(M^3)			Drums(M^3)			Inorganic(M^3)			Organic(M^3)	Alpha	Beta
L	M	H	L	M	H	L	M	H		mCi	mCi
800	5	86	278	64	170	2.2	0.3	0.03	2.7	25.4	614

L = Low active (less than 200 mr/hr); M = Middle active (200 to 1000 mr/hr); H = High active (greater than 1000 mr/hr).

CONTROL AND MEASUREMENT OF RADIATION DOSE TO WORKERS

Each day's work was planned through a system of special work permits requiring details of (i) work to be done; (ii) radiation surveys of the area; (iii) the authorised dose to the workers; and (iv) the protective wears and dosimetry systems to be worn and such other relevant information.

Normally a standard film badge, a self reading dosimeter, a self reading dosimeter with a beta shield (1.5 mm S.S. sleeve) were worn on the chest. In special cases additional film badges were used on wrists and feet. In most of the cases workers were exposed to significant beta doses. These were taken into account by adding 1/6th of the beta dose to the gamma dose and calling the sum as the whole body dose. Even though such addition was not required by the I.C.R.P. recommendation (I.C.R.P. 26) gives a factor of 1/16 as the weighting factor for skin exposure.

The total dose commitment for this work was around 3000 man-rems. Out of this nearly 1000 man-rems were due to dose from beta radiation exposure. (This means that the total beta dose was 6000 beta man-rems).

A shadow shield whole body counter was available full time for this job. Workers were scanned for internally deposited radioactivity within a week after their job. A plutonium lung counter was also

available for estimation of chest burdens of plutonium. All the workers were subjected to bioassay procedures for body burden estimation of fission products, plutonium and uranium.

CONCLUDING REMARKS

The work was completed safely by using relatively simple but fully reliable protective wears. The entire job was done by the staff normally associated with the operation of the plant. This aspect contributed significantly in the management and control of personnel exposures. Good planning and supervision of the radiation work during all the phases resulted in keeping the external doses to individuals below the annual limit of 5000 millirems. There was no case of any internal contamination resulting in a dose commitment in excess of 10% of the annual limit. General industrial safety precautions must also be followed in such jobs in which compressed air, scaffoldings and ladders, slippery surfaces, harmful chemicals, sharp objects and a host of other lurking causes may lead to serious injury.

A job of this type looks staggering in the initial stage. A large responsibility rests on the health physicists in building up the confidence of the radiation workers in the efficacy of safety measures and dose control methods. We were fortunate in having a band of dedicated workers who took to the task in a sincere and very cooperative manner. Our experience shows that decommissioning of nuclear chemical plants can safely be carried out using the presently available technical facilities and scrupulously following the principles and procedures of Health Physics.

ACKNOWLEDGEMENTS

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THE PRACTICAL APPLICATION OF ICRP RECOMMENDATIONS REGARDING DOSE-EQUIVALENT LIMITS FOR WORKERS TO STAFF IN DIAGNOSTIC X-RAY DEPARTMENTS

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In many diagnostic examinations using X-rays in hospitals, staff work in the X-ray room with the patient, and wear lead aprons to protect their bodies from the radiation. The effect of the protective apron is to reduce the dose-equivalent received by those parts of the body that the apron covers, leading to grossly non-uniform irradiation of the body as a whole. Under these circumstances we have to consider how best to monitor the exposure of the individual.

APPLICATION OF ICRP RECOMMENDATIONS

Where there is non-uniform irradiation, ICRP (1) recommends that the annual dose-equivalent limit of 50 millisieverts should be applied to the expression $\Sigma_T w_T H_T$ where w_T is a weighting factor representing the proportion of the stochastic risk resulting from tissue T to the total risk and H_T is the annual dose-equivalent in tissue T. This expression is referred to as the annual effective dose-equivalent.

Let us examine how the effective dose-equivalent is influenced by wearing a lead protective apron. We will assume that the radiation field is uniform and that the apron protects the trunk leaving the head, neck arms and legs exposed. Let us further assume that the annual dose-equivalent in protected organs is H_1 and that in unprotected organs is H_2. ICRP assigns weighting factors w_T to the gonads, breast, red bone marrow, lung, thyroid and bone surfaces and the remainder of the body.

In the situation described the gonads, breast and lung are protected while the thyroid is unprotected. The red bone marrow is distributed so that approximately 80% is protected while 20% is unprotected (2). The mean dose-equivalent in this tissue is therefore $0.8H_1 + 0.2H_2$. Data for the distribution of bone surfaces are not available but from the distribution of dry bone (2) we may assume that they are divided in approximately equal proportions. The mean dose-equivalent here is therefore $0.5H_1 + 0.5H_2$.

For the remainder of the body, ICRP recommends that a value of $w_T = 0.06$ is applicable to each of the five organs or tissues receiving the highest dose-equivalents and that all others can be neglected. There are more than five organs and tissues in the head and neck (e.g brain, salivary glands, tongue, pharynx, larynx, etc) and these are all unprotected. It follows that they will receive higher dose-equivalents than organs in the trunk and therefore the dose-equivalent to be applied to the remainder is H_2. Table 1 shows the derivation of the annual effective dose-equivalent.

TABLE 1. Annual Effective Dose Equivalent when a lead apron is worn.

Tissue	Weighting Factor (w_T)	Annual Dose-Equivalent (H_T)	Weighted Annual Dose-Equivalent ($w_T H_T$)
Gonads	0.25	H_1	$0.25H_1$
Breast	0.15	H_1	$0.15H_1$
Red Bone Marrow	0.12	$0.8H_1 + 0.2H_2$	$0.096H_1 + 0.024H_2$
Lung	0.12	H_1	$0.12H_1$
Thyroid	0.03	H_2	$0.03H_2$
Bone Surfaces	0.03	$0.5H_1 + 0.5H_2$	$0.015H_1 + 0.015H_2$
Remainder	0.30	H_2	$0.30H_2$
Annual Effective Dose-Equivalent			$0.631H_1 + 0.369H_2$

The value of the annual effective dose-equivalent obtained in this way may be rounded to give:-

$$\sum_T w_T H_T = 0.6H_1 + 0.4H_2 \text{ ---- Formula 1}$$

To make use of Formula 1 in practical situations, H_1 may be estimated by means of a dosemeter worn on the trunk under the protective apron and H_2 may be estimated by means of a dosemeter worn on the collar or forehead.

COMPARISON WITH PUBLISHED DATA.

Formula 1 is subject to a number of simplifying assumptions (notably the uniformity of the radiation field as a whole and the uniformity of irradiation of tissues within the protected and unprotected parts of the body). It is therefore important to compare the result of using the formula with that obtained by measurement of individual organ or tissue doses. Wøhni and Stranden (3) have published measurements of the absorbed dose in most of the relevant tissues in a variety of operating conditions using a phantom, fitted with a lead apron (0.25 mm lead equivalent), to represent a physician or nurse standing close to a patient undergoing X-ray examination.

In Table 2 the effective dose-equivalent has been calculated using the data of Wøhni and Stranden. Unfortunately their data do not include doses in organs of the head and neck with the exception of the eye lens and thyroid. Both organs being close to the body surface, doses in these organs will not be typical of organs such as the brain. Arbitrarily, therefore, and for want of better data, we have taken the mean absorbed dose in the gastrointestinal tract, measured <u>without</u> the lead apron, for the dose-equivalent in the "remainder". Also given in Table 2 are values for the effective dose-equivalent obtained by the use of Formula 1 (above). For this purpose the doses measured in the breast and thyroid have been assumed to be equal to the dose-equivalent as measured by a dose-

meter located on the trunk under the apron (H_1) and one on the collar outside the apron (H_2) respectively. All the values in the table are expressed in the original units, i.e rads for 1 röntgen exposure on the outside of the apron at the right hand side of the thorax, and are quoted for a female phantom.

TABLE 2. Derived values of effective dose-equivalent in X-ray work.

	Radiological Conditions								
	60 kVp		80 kVp	100 kVp		120 kVp	140 kVp		S+I**
	OC*	UC	OC	OC	UC	OC	OC	UC	
Effective dose-equivalent derived from measured organ doses	0.08	0.19	0.12	0.14	0.25	0.15	0.16	0.29	0.29
$0.6H_1 + 0.4H_2$ ***	0.21	0.25	0.23	0.24	0.29	0.25	0.26	0.32	0.29

* OC = overcouch tube, UC = undercouch tube

** S+I was a simulated stomach and intestine examination

*** H_1 taken as the dose in the breast, H_2 taken as the dose in the thyroid.

Inspection of the values given in Table 2 shows that there is good agreement between the effective dose-equivalent obtained from measured values of organ doses and that obtained by application of the formula. In all but one of the cases the valuee differ by less than a factor of 2 and in none of the cases is the value based on the organ doses higher than the value obtained using the formula.

EFFECTIVE DOSE-EQUIVALENT AND EYE LENS.

Previous authors, e.g Littleton, et al (4), have considered that in work of this nature the principle cause for concern regarding the exposure of parts of the body not protected by an apron is the dose-equivalent received by the lens of the eye. In Table 3 we have compared the effective dose-equivalent derived from measured organ doses as a percentage of the annual dose-equivalent limit for the whole body (50 millisieverts) with the eye lens dose as a percentage of the annual dose-equivalent limit for the lens (300 millisieverts), in each of the conditions of measurement of Wøhni and Stranden (3) when a protecti apron is used.

The table shows that in all cases but the first (60 kVp, overcouch tube) the effective dose-equivalent calculated from organ doses is a bigger percentage of the annual limit than is the eye lens dose. In the first case the two are approximately equal percentages of the annual limits. It can be concluded therefore that if the effective dose-equivalent is limited to 50 millisieverts per annum the annual limit for the lens is most unlikely to be exceeded, particularly if the effective dose-equivalent is assessed by means of the formula.

TABLE 3. Dose-equivalent in lens, and effective (whole body) dose-equivalent, as percentages of annual limits (for 1 röntgen exposure)

	Radiological Conditions								
	60 kVp OC	60 kVp UC	80 kVp OC	100 kVp OC	100 kVp UC	120 kVp OC	140 kVp OC	140 kVp UC	S+I
Eye lens dose	1.8	2.0	1.9	2.0	1.8	2.0	1.9	1.9	1.1
Effective dose-equivalent	1.6	3.8	2.4	2.8	5.0	3.0	3.2	5.6	5.8

PRACTICAL APPLICATION

In principle, to limit the effective dose-equivalent to 50 millisieverts per annum, one should use two personal dosemeters and apply Formula 1 to the results obtained from the dosemeters. However, in practice it will not be necessary to insist on two dosemeters being worn in all cases. If, in the light of experience, the dose recorded by the body dosemeter (H_1) is found to be very small compared with that recorded at the collar (H_2), the former may be dispensed with. This would apply to the results in the first three columns of Table 2 where H_2 contributes lees than 5% of the effective dose-equivalent. Under these conditions, an annual limit of 125 millisieverts can be applied to H_2 (i.e from the formula with H_1 taken as zero and the annual limit of effective dose-equivalent taken as 50 millisieverts). If on the other hand, despite the protective apron, there is significant exposure of the trunk, a single dosemeter worn at the collar will still suffice provided that the annual whole body limit of 50 millisieverts is applied. It is only in those few cases where such an approach would be too restrictive that two dosemeters need to be worn and the formula used.

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SOME RADIATION PROTECTION PROBLEMS IN A CANCER HOSPITAL AND ASSOCIATED RESEARCH INSTITUTE

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The Royal Marsden Hospital and the Institute of Cancer Research both derive from the original Cancer Hospital (Free) founded in 1851 by Dr. William Marsden at a site in Chelsea. From the beginning the hospital had a dual role - care of the patient and research. This has now become a large multidisciplinary organisation, comprising a postgraduate teaching hospital within the National Health Service, and an independent Research Institute which is part of the University of London. Different branches of the organisation are located in three sites many miles apart. Both Hospital and Institute make extensive use of ionizing radiation in therapy and diagnosis, and in clinical and laboratory investigations in radionuclide work. Four linear accelerators, and several telecobalt, telecaesium and orthovoltage sets are used in beam therapy and there is a wide range of X-ray diagnostic equipment, including a computed tomography whole body scanner. Brachytherapy involves an annual turnover of about 3.7 TBq (100 Ci) (Cs-137, Ir-192, Au-198) administered as small sealed sources to some 450 patients, while the annual administration of unsealed sources, in diagnosis (Tc-99m, etc) and therapy (I-131, etc) is about 3.0 TBq (80 Ci) involving some 10,000 patients. In the Institute, a range of radionuclides, notably H3, C14 and I-125, are used at tracer and higher level, and there are irradiation facilities with high activity ($\sim$50 TBq, $\sim$1.4 kCi) Co-60 sources.

In this paper we first identify some of the problems we face in ensuring that our protection organisation responds appropriately to the latest recommendations and legal requirements in radiological protection, against a background of steady increase in the scope of our radiation work. In recent years there have been sharp increases in the volume of work with unsealed sources, and we draw special attention to our experience in these areas.

RESPONSE TO RADIOLOGICAL PROTECTION RECOMMENDATIONS

For over 20 years we have imposed upon female staff of reproductive capacity, an annual whole-body dose-equivalent limit of 15mSv, to be accumulated so far as possible, at a uniform rate. This limit was imposed in the context of a wide range of brachytherapy procedures, including associated nursing of patients, the primary aim being to ensure a good standard in limitation of foetal dose if any of the exposed staff become pregnant. In our experience, with constant attention to protection procedures, including the control of dose accumulation by personnel monitoring

such limitations appeared to be readily achievable in all our radiation practice. From about 1970 we had moved towards imposing a 15 mSv limit for all staff and only a very small group have been allowed even to approach that limit. Our response to ICRP26 has been to confirm our position but to focus even more attention on the ALARA (as low as reasonably attainable) principle. For our establishment we now regard the breaking of a 15mSv limit as evidence of inadequate control rather than of need for worker classification. Nevertheless, it is administratively valuable to us, particularly against the possibility of an incident, to recognise that occasional dose accumulation up to the ICRP recommended limit (50mSv/annum) might still be regarded as acceptable.

To reduce the attention required to dose control in brachytherapy procedures, we have supported within our hospital the introduction of significant changes in radiotherapy technique over the years, such as total replacement of Ra-226 by Cs-137 and Ir-192, the development of convenient small-source dispensing equipment, the use of after-loading techniques, and the provision of a special brachytherapy nursing unit equipped with inter-bed and mobile shielding structures plus television and intercommunication systems for observation-nursing. We have also developed our teaching programmes in radiological protection, including special instruction for nursing staff within our Nursing School (1).

Despite these efforts, the pressure on our protection organisation continues to increase markedly, but only in part through our rising work-load in radiation procedures. Our attention to control of dose to patient has been much increased, especially in relation to our important commitment to mammography (2), the '10-day rule', quality control procedures in diagnostic radiology and new safety requirements in radiotherapy installations. Additionally, we are beginning to encounter new administrative requirements stemming from European and National Regulations on Health and Safety, and the demands for attention from Trades Union representatives through powers granted by such regulationa.

WORK WITH UNSEALED SOURCES

Control of external radiation exposure is exercised by our own film monitoring service which has maintained records of staff-dose within our establishment for more than thirty years. We regard the constant use of the information derived from this service as a crucial element of our protection arrangements, not only as a means of meeting our self-imposed limit on staff-dose accumulation, but also as a way of identifying at an early stage any significant changes in the collective dose-equivalents for different groups of staff. Additionally, inspection of films provides a useful check on any looseness in contamination control. Within our many departments surveillance of procedures is exercised by departmental radiological safety officers, and primary control of contamination is by routine contamination monitoring. A low background whole-body counter and a thyroid I-125 assembly are also used for countercheck measurements.

NUCLEAR MEDICINE

From somewhat primitive beginnings some 30 years ago (using open bench procedures and a spirit lamp to sterilise the air!) we have progressed to air-conditioned radiopharmaceutical clean rooms housing laminar flow cabinets, at both branches of our hospital (3). During early periods of low work-load our chief concern was contamination control. However, in the late 1960's our work-load increased sharply; in 1969 a total activity of 630 GBq (17 Ci) in a wide range of radiopharmaceuticals was administered to some 3000 patients. The accumulation of external dose by a small group of staff was then evident but still relatively small. The work-load has, however, continued to increase with the 1979 total of activity administered exceeding 3 TBq (80 Ci) chiefly Tc-99m in various forms, to some 10,000 patients for a wide range of diagnostic examinations. External dose accumulations by staff also continued to increase and by the mid-1970's the annual collective dose-equivalent for our nuclear medicine group (~20 staff - medical, radiography and nursing) had risen to some 40mSv.

In considering the problems of external dose, we note particularly the gain resulting from the use of more sensitive equipment (in reducing patient measurement time) and from attention to techniques of administration.

We have made detailed investigations into the control of dose to the hands (4), (5), (6).

RESEARCH

We have given much attention to procedures involving I-125, and have noted the problems reported elsewhere (7). In one pathology laboratory here some 20 members of staff are involved in labelling techniques, using up to 74 MBq (2 mCi) in each procedure and over 11 GBq (300 mCi) per year. A Class B laboratory with high grade fume cupboard is used for this work. Staff are instructed to self-monitor, including the thyroid, by contamination meter, and counter-check measurements are made periodically by Physics Staff using a thin crystal NaI detector. Under these conditions during two years we have only detected I-125 in about one third of these workers, at levels generally of 370 - 740 Bq (10 - 20 nCi).

One traumatic experience with tritium deserves mention. In spite of our centralised control on radionuclide acquisition, a member of our staff acquired, by an unusual method of requisition, an aqueous solution of an organic compound supposedly labelled at high specific activity with 92.5 GBq (2.5 Ci) T and supposedly free of labile tritium. In spite of our Instruction Manual, our local rules, our B-laboratory facility and our safety officer, an open bench evaporation of the solution was conducted. Later analysis showed that virtually none of the tritium had ever been associated with the compound. As a result of the evaporation procedure, the laboratory and its equipment were substantially contaminated and body burdens reached the ICRP investigation level in several staff. The decontamination procedures were lengthy and the interruption of important work programmes was severe.

SUMMARY

By giving steady attention to the design of facilities and the arrangement of procedures, and with an active personnel monitoring policy, relatively large scale radiation commitments within medical and research organisations can proceed with individual whole-body doses to staff being held well below 15mSv/annum. Such control generally relies heavily on the experience of a small group of staff within the organisation, who now face increased problematical administrative commitments from recent legislation.

In spite of detailed attention to control of radiation work, traumatic radiation incidents may still occur.

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RADIATION PROTECTION IN THE DENTAL PROFESSION.

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INTRODUCTION

Prior to 1974, factories within Great Britain were subject to Regulations made under the Factories Act. Within teaching and research establishments, radiation was, broadly speaking, subject to non-enforceable Codes of Practice. In 1974 the Health and Safety at Work etc. Act was made; this sets the regulatory framework and it is enforceable within all premises where there are employed persons.

As yet no statutory provisions which relate to dental radiography have been made under this Act; these will be enacted in the near future as part of wider legislation to meet a Euratom Directive.

The Health and Safety Executive (HSE) initiated a survey in 1977 in order to acquire detailed information on the standard of radiation protection in dental practices. This stemmed, first, from the need for HSE to meet its enforcement duties, secondly, to devise a basis for regulatory requirements and, thirdly, to evaluate the dental postal pack provided by the National Radiological Protection Board (NRPB).

METHOD OF ASSESSMENT

The postal pack method of assessing the performance of dental X-ray sets was developed by the NRPB to enable relatively large numbers of X-ray sets to be checked cheaply. Its primary aim was to make it unnecessary for an experienced surveyor to visit each dental surgery, whilst retaining the capacity to collect all the information necessary to make recommendations regarding the compliance of the equipment and procedures with the accepted radiological protection standards. The system was designed to measure the linearity of output, the cone tip exposure rate, the beam diameter and an estimation of beam quality in terms of added aluminium filtration.

The method of assessment consists of two stages. Initially the dentist is asked to complete a questionnaire regarding the equipment, its operation and the radiographic procedures used in the practice. At the same time he is asked to expose three monitoring films at the tip of the X-ray cone for timer settings of 1.0 s, 0.7 s and 0.3 s respectively. From the exposures recorded on these films the linearity of output of the X-ray set is assessed.

On the basis of a tentative estimation of the output of the X-ray set, the dentist is sent specially developed cassettes for exposure at the tip of the cone for a specific timer setting which is designed to give an exposure of approximately 8 mGy. An X-ray film in the cassette is used to determine beam diameter and a standard personal monitoring film, behind filters, is used to measure the other parameters.

Added filtration is by far the most difficult parameter to measure. However, the apparent doses recorded on the personal monitoring film behind two copper filters and an unfiltered area can be used to estimate added tube filtration to an accuracy sufficient for radiological protection purposes. If the unfiltered to 0.48 mm Cu filter apparent dose ratio is plotted against the 0.1 mm Cu filter to 0.48 mm Cu filter apparent dose ratio, for a series of exposures made at various known tube filtrations and kilovoltages, a family of curves will be produced. Each curve represents a particular tube filtration value and an interpolation between curves can be made for exposures made on equipment whose tube filtration is not known.

Previous experience had shown that cone tip exposures in general are much greater than is necessary for the production of a good quality radiograph. A series of experiments with a phantom has revealed that for a given type of radiograph, there is a direct relationship between kilovoltage and the cone tip exposure required to produce a satisfactory radiograph (1). The 0.1 mm Cu to 0.48 mm Cu apparent dose ratio gives a good indication of operating kilovoltage and this is used to assess the cone tip exposure required for the production of a satisfactory radiograph. This exposure is then used as a basis for making recommendations regarding timer settings for different radiographic techniques.

Personal monitoring films were worn by dental staff during each of three consecutive months.

GUIDANCE TO PRACTITIONERS.

Dentists were chosen at random from a list of practitioners offering treatment within the National Health Service. Participation in the survey was voluntary. The standards against which each practitioner was evaluated were those set out in a Code of Practice (2), for medical and dental uses of ionising radiation. In 1975 all dental practitioners within Great Britain were issued with a synopsis of the Code (3) specific to dental radiography.

Following each assessment, recommendations are given, in writing, to practitioners to enable them to improve, or maintain, the quality of their radiographs while minimising the radiation dose to staff and patients.

The results presented here relate to two phases of the survey; Group A consists of the first 585 practitioners surveyed. Between the end of phase A and the commencement of phase B an effort was made to give additional guidance on radiation protection to the profession as a whole, this was based on the evidence from the Group A survey. The guidance consisted of a detailed article published in the professional literature (4), and a placard listing simple rules which was sent to all practitioners at the time of publication. Results from 194 Group B dentists are available to date. No member of Group B was a member of Group A.

SURVEY RESULTS

It is possible to give only a synopsis of the results in this paper:-

		Group A	Group B
(i)	Personal Radiation Dose to Dental Staff		
	Results from film badges worn for 1 month		
	< 50 μSv	82.3%	97.2%
	≥ 50 μSv but < 100 μSv	5.8%	1.6%
	≥ 100 μSv but < 200 μSv	7.1%	1.0%
	≥ 200 μSv but < 500 μSv	3.4%	0.1%
	≥ 500 μSv but < 1000 μSv	0.5%	0.1%
	≥ 1000 μSv	0.6%	0
(ii)	Beam Diameter		
	> 7.5 cms	12%	7%
	> 6.0 cms	55%	12%
(iii)	Beam Filtration		
	< 1.5 mm Al	11%	13%
(iv)	No "X-rays on" warning signal	9%	5%
(v)	Dental staff holding film or tube head	20%	7%
(vi)	Dose per exposure		
	≥ 6.6 mSv*	57%	54%
	≥ 5.0 mSv	76%	70%

* Experimental work (1) has indicated that, depending on the kilovoltage of the equipment, a good radiograph should be obtained with exposures of between 1.3 mSv and 6.6 mSv as measured at the cone tip.

Further information derived from the survey included other important factors such as film processing techniques, timer accuracy and reproducibility, location and maintenance of equipment and the level of instruction and supervision given to dental staff.

CONCLUSIONS

Comparison of Group A with Group B tends to indicate an improvement in standards. This may be partially due to the general educative campaign; it may result from discussion within the profession prompted by the HSE Survey programme. Another possibility may be two "natural phenomena", these being the ageing of equipment and the ageing of dentists. In the former, old equipment is replaced by equipment which should meet present technical standards. As dentists retire they are replaced by dentists who have recently completed their formal education, it is the latter who are, generally speaking, more appreciative of the hazards and of correct radiographic procedure.

It is anticipated that the standard in the practices of Groups A and B will further improve when individual recommendations, based on survey results, have been implemented.

The postal method of survey, reinforced, where necessary, by visits from inspectors, is a cost effective method of promoting radiation safety for dental staff and is a useful aid to the enforcement of statutory requirements. Early results are encouraging; it is intended to resurvey each dental practice every 6-7 years and it is clear that the enforcing authorities must maintain contact with the suppliers of dental radiographic equipment and with the dental schools.

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A METHODOLOGY FOR THE EVALUATION OF COLLECTIVE DOSES ARISING FROM RADIOACTIVE DISCHARGES TO THE ATMOSPHERE

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INTRODUCTION

The usual approach to protecting the public from exposure to radioactivity in the environment has been based on identifying the critical pathways to man for particular radionuclides. By controlling the radiation exposure of the critical organs of individuals in the population who were most highly exposed the remainder of the population could be assumed to be adequately protected. The system of dose limitation proposed by the International Commission on Radiological Protection (ICRP) in Publication 26 (1) is based on the consideration of the total risk associated with radiation exposure and it has become necessary to evaluate the radiation exposure of individuals by all pathways from all radionuclides to which they are exposed. At the same time the total health detriment from a radiological practice has become recognised as an important quantity by which the radiological harm to exposed populations can be judged. The ICRP also recommends (1) that once a radiological practice has been shown to be justified, any resulting radiation exposure should be kept as low as reasonably achievable. The dual emphasis on total dose to an individual and on optimisation with the latter implying the need to evaluate total health detriment means that the exposure of the whole population must be considered, through all pathways and for all radionuclides for as long as they persist in mans environment. To achieve this objective when considering releases to the atmosphere it is necessary to have an understanding of the way in which radioactive effluents are dispersed in the environment, of the transfer processes which take place from the environment to man and of the spatial distribution of the population and of agricultural practices around the discharge point.

It is difficult to define a single radiological quantity which adequately describes the total number of health effects which will appear in an irradiated population. The collective effective dose equivalent commitment as defined in ICRP Publication 26 (1) has been used elsewhere as a first approximation to such a quantity although the limitations of this approach have been recognised (2).

In this paper a methodology is described for evaluating collective dose from the major routes of exposure following a release to atmosphere. The routes considered are inhalation both of the material in the initial cloud, and that resuspended from the ground, external irradiation from the radioactive decay of the material both in the cloud and deposited on the ground, and ingestion of radionuclides transferred through the foodchain.

ENVIRONMENTAL MODELLING

An atmospheric dispersion model (3) is used to derive the spatial distribution of the activity around the release point taking into account the height of the release and the meteorological parameters specific to that site. The radiation exposure of an individual from inhalation is evaluated from the air concentration at any point using an appropriate breathing rate and a knowledge of the dose distribution to human organs and tissues per unit of activity inhaled.

The individual exposure from external irradiation by both electrons and photons from the material in the cloud is also evaluated from the air concentration. A semi-infinite cloud model is used to evaluate the absorbed dose in air from electrons; since the sensitive basal layer of the skin is at an average depth of 70μm, a correction factor is necessary to obtain the relevant dose equivalent. The absorbed dose in air from photons is evaluated by means of a finite cloud model which takes account of the distribution in space of the material in the cloud. The absorbed dose in air is converted to dose equivalent in body tissues using factors derived by Poston and Snyder (4).

The amount of activity deposited on the ground by both wet and dry deposition mechanisms is evaluated and the radiation exposure from external irradiation from the deposited material is calculated taking into account the migration of the activity down into the soil (2). The air concentration from resuspension processes and hence the dose from inhalation of resuspended material is evaluated using a time dependent resuspension factor (2).

The radioactivity appearing in foodstuffs is estimated by using a time dependent foodchain model (5) which relates the level of contamination in the foodstuffs produced at a specific location to the amount of material deposited there. The dynamic nature of the model enables time dependent processes, which may be important for long lived radionuclides, to be represented : these include the build-up of activity in soil and animal tissues and the migration of activity through the soil. Other important mechanisms involved in the transfer of activity to foodstuffs derived from plants and animals are represented in the model; they are interception and retention of deposited activity on plant surfaces, resuspension, translocation and root uptake into plants, transfer to animals by consumption of contaminated pasture, inadvertent consumption of soil, inhalation of the initial cloud and resuspended activity and the metabolism of radionuclides in the animals following intake.

POPULATION AND AGRICULTURAL DATA

The models described above allow the radiation exposure of individuals to be calculated from a variety of routes and the activity appearing in various foodstuffs at all points around the discharge site to be evaluated. To complete the estimation of collective dose it is necessary to know the distribution of the population and of agricultural practices around the site of interest. In assessing the collective dose from the ingestion pathway it is assumed that all activity reaching foodstuffs is consumed by man; it is therefore

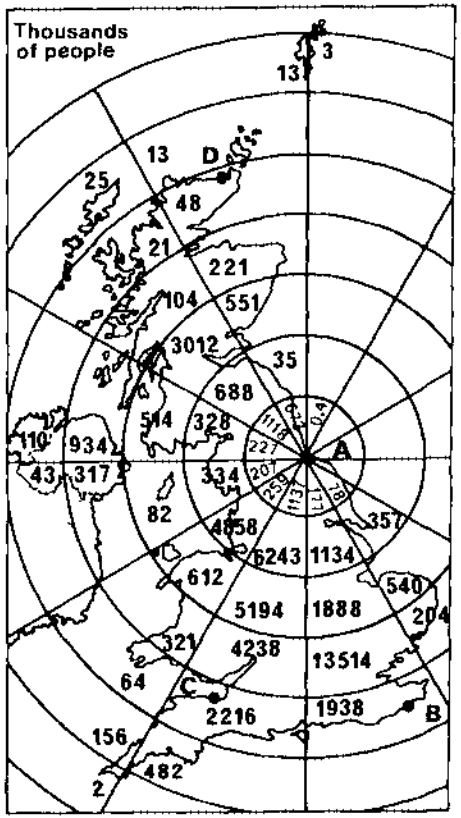

Fig 1 Location of sites and the population distribution around site A

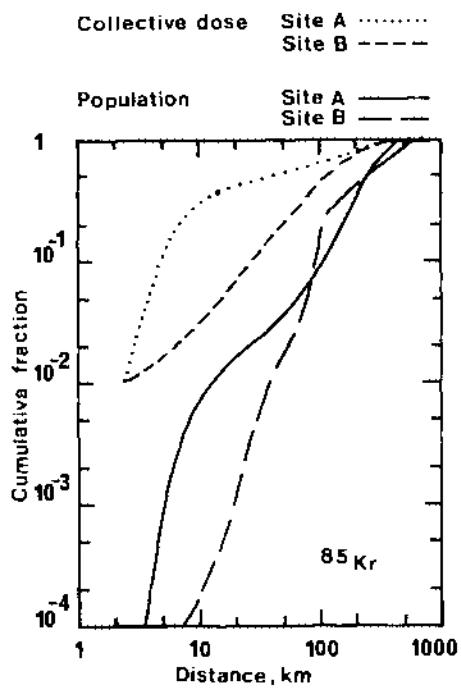

Fig 2 Cumulative population and collective dose from skin β irradiation v distance

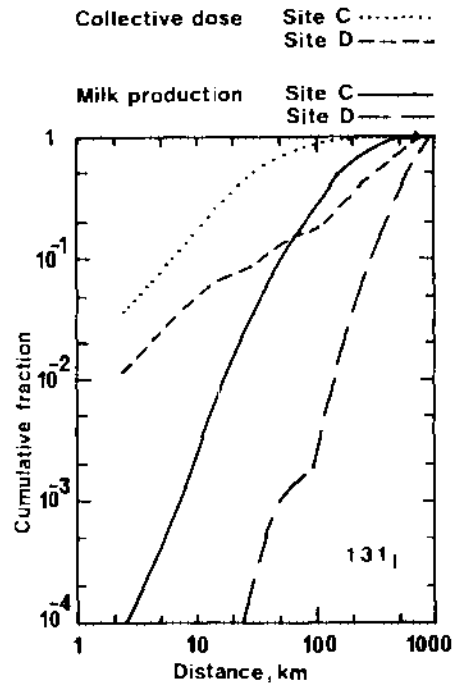

Fig 3 Cumulative milk production and collective dose from ingestion v distance

unnecessary to make assumptions concerning individual dietary intakes.

Population data in the form of a listing of the numbers of people living in each 1km square of the National Grid of Great Britain, as recorded at the 1971 census, together with similar data for Northern Ireland, are used to determine the number of people at a given distance in a given direction from the site, as shown in figure 1.

Similarly, grids of agricultural production have been generated based on national agricultural statistics. 5km grids exist for the most important agricultural products as identified in the foodchain model ie grain, green vegetables, root crops, beef, cows liver, mutton/lamb, sheeps liver and cows milk.

EXAMPLES OF CALCULATIONS

The contribution by each exposure pathway to collective dose from a small segment of area is calculated by combining estimates of the individual dose with the population of the segment or the dose per unit mass of foodstuffs with production in the segment. The total collective dose for each exposure pathway is the sum of the contributions from all such segments around the point of discharge. This process is repeated for each nuclide and pathway to evaluate the total collective dose from a discharge.

An example of the results of collective dose calculations is shown in figure 2. The variation of the cumulative populations and collective doses from β irradiation with distance from the point of discharge are contrasted for the discharge of krypton-85 from two sites A and B whose population characteristics are quite different. The ratio of the total collective dose from unit discharge at site A to that at Site B is 1.7. The rate of accumulation of collective dose with distance is very different for the two sites. The majority of the collective dose at site A comes from quite close to the site and consequently the individual doses contributing to the collective dose are generally higher than is the case for site B.

A similar example is shown in figure 3 where the variation with distance of cumulative milk production and collective dose from the ingestion of contaminated milk and milk products from the release of I-131 is given. Here the difference in the total collective doses from the two sites C and D which are located in markedly different agricultural areas is even more pronounced, the ratio being 20.

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COLLECTIVE DOSES DUE TO RADIOACTIVE EMISSIONS FROM NUCLEAR PLANTS

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In publication No.26 (1) ICRP recommends the use of cost-benefit analysis for the optimization of the retention of radioactive materials in nuclear facilities. This Requires the evaluation of the collective effective dose equivalent commitment. While the collective doses can be evaluated generally for airborne effluents the collective dose for the water pathway is highly dependent on the river under consideration.

COLLECTIVE DOSES DUE TO AIRBORNE EFFLUENTS

Due to different numerical methods it is useful to separate the evaluation of collective doses into first pass exposure and global distribution. The annual average air-concentration for first pass exposure within a 5o km range can be evaluated using the Gaussian Plume model. For greater distances a homogenous distribution over a constant mixing layer is assumed (2). Comparing the calculation of collective doses with an homogenous 360 deg. distribution and much more complex metheorological statistics it becomes obvious that there is no significant difference. The emission height also is of negligible influence on the collective doses. The contribution to the collective doses by the direct vicinity of the plant - where the emission height is of influence - is small, as shown in tab. 1.

Parameter analysis have shown that there is little difference in the equivalent collective doses for all sites in the F.R.G.

The global distribution can be evaluated with multicompartment models (3,4,5), see fig. 2 for ^{14}C e.g..

For the calculation of the ingestion dose caused by ^{3}H, ^{14}C and ^{129}I via first pass exposure, it is sensible to use the specific activity model. The average air concentration of ^{127}I for the F.R.G, as well as for other countries is about 1o ng/m^3 (7). An upper limit of collective doses due to iodine, except for ^{129}I and aerosols, can be evaluated with the models and data given in (8), taking into accound mean consumption rates of food. Caused by the high number of people and the very small change of air concentration in far distances, the contribution to the collective doses via first pass exposure in these areas is much more important than that in the vicinity of the plant. For this reason it is not necessary to regard the local statistics of harvest, which would make

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evaluations much more complicated.

^{3}H first pass exposure is a special problem. Being emitted as tritiated water, ^{3}H contributes to the radiation exposure via contamination of food and drinking water. The concentration of ^{3}H can be evaluated easily with a rainfall of approx. 400 mm/yr which is the contribution of the oceans to the total rainfall of approx. 700 mm/yr. When calculating with the total rainfall, evaporation has to be taken into account (9).

The collective effective dose commitment can be evaluated using the weighting-factors given in ICRP 26 (1). In this paper a total collective effective dose equivalent commitment S_T with additional consideration of the detriment after the 2nd generation (1) and the external β-radiation (1b) is defined.

Tab. 2 shows the total collective effective dose equivalent commitment S_T related to 1 Ci ^{3}H, ^{14}C, ^{85}Kr, ^{131}I, ^{129}I and aerosols for first pass exposure. For the aerosols a typical nuclide composition of the effluent of a reprocessing plant was used (11). The approx. distribution of ^{3}H collective doses for different countries is shown in tab. 3.

In addition, tab. 2 contains the total collective effective dose equivalent commitments due to globally distributed ^{3}H, ^{14}C, ^{85}Kr and ^{129}I as a function of integration time.

From the viewpoint of a cost-benefit analysis, the integration time occurs to be very problematic. As seen in fig. 2, the greatest contribution to S_T from ^{14}C and ^{129}I results from times longer than 500 years. For the cost-benefit analysis, the integration time should be 10^4 to 10^5 years, taking into account a world population of 10^{10} people. Using economical methods, the cost of detriment should be discounted. Tab. 4 shows the discounted collective dose.

For the fist pass exposure different sites were analysed. Stack releases at sites like Windscale which are geographically extremely different from the north-east of Germany result in a decrease of collective doses of approx. 30 %.

COLLECTIVE DOSES VIA THE WATER PATHWAY

The collective dose due to liquid releases into rivers can be evaluated for actual sites only. The utilization of the river is of great importance here. Assuming a river with a mixing volume of 1000 m^3/s, which is used by 10^6 persons as drinking water, the total collective effective dose equivalent commitment due to ^{3}H would be about 0.001 man·rem/Ci. If the consumption of fish would be 100 t/yr, S_T due to the most important radionuclides would be approx.: ^{60}Co - 0.001 man·rem/Ci, ^{90}Sr-0.1 man·rem/Ci, ^{134}Cs - 0.05 man·rem/Ci, ^{137}Cs - 0.02 man·rem/Ci.

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TABLE 1 Percentage of the first pass ^{85}Kr collective dose to the population in different areas relative to the first pass ^{85}Kr collective dose to the total world population

Area	Population	Percentage of ^{85}Kr collective dose [%]
Radius 0 - 50 km F.R. of Germany	8.1 E5	4.5
Radius 0 - 100 km F.R. of Germany	3.2 E6	9
Radius 50 - 100 km F.R. of Germany	2.4 E6	4.5
F.R. of Germany	5.2 E7	40
World	4. E9	100

TABLE 2 Total collective effective dose equivalent commitment S_T by an emission of 1 Ci/a q stock in north east of the Federal Republic of Germany

a Typical nuclide composition of a reprocessing plant

Nuclide	Total collective effective dose equivalent commitment [man rem/Ci]					
	First pass exposure	Global distribution dependence at integration time				
		70a	500a	10.000a	10^6 a	∞
H 3	0.013	5.7E-4	5.7E-4	5.7E-4	5.7E-4	5.7E-4
C 14	9	23	56	370	520	520
Kr 85	5.E-5	1.4E-3	1.4E-3	1.4E-3	1.4E-3	1.4E-3
I 131	5	0	0	0	0	0
I 129	1000	4.8	7.8	72	6.8E3	16.E4
Aerosols (reprocessing plant)	100 a	0	0	0	0	0

TABLE 3 Approximate contribution to the collective dose of the different countries over the first pass exposure for an emittent near Hannover (1 Ci/a first pass: S_T = 0.013 man rem/Ci)

Country	Contribution	Country	Contribution
Austria	1	Luxemburg	0.1
Belgium	4	Netherlands	5
Czechoslovakia	1	Norway	0.1
Denmark	1	Poland	15
France	6	Sweden	2
German Democratic Republic	15	Union of Soviet Socialist Republics	1
Germany, Federal Republic of	40	United Kingdom	1

Tab.4 Change of the global collective effective dose equivalent commitment with discounting the cost of detriment by infinite integration time

Nuclide	$S_{T,i}$			
	i=0%	i=1%	i=2%	i=3%
H 3	5.7E-4	4.6E-4	3.7E-4	3.0E-4
C 14	520	22	13	9
Kr 85	1.4E-3	1.1E-3	8.5E-4	6.8E-4
I 129	1.6E5	2	1.3	1

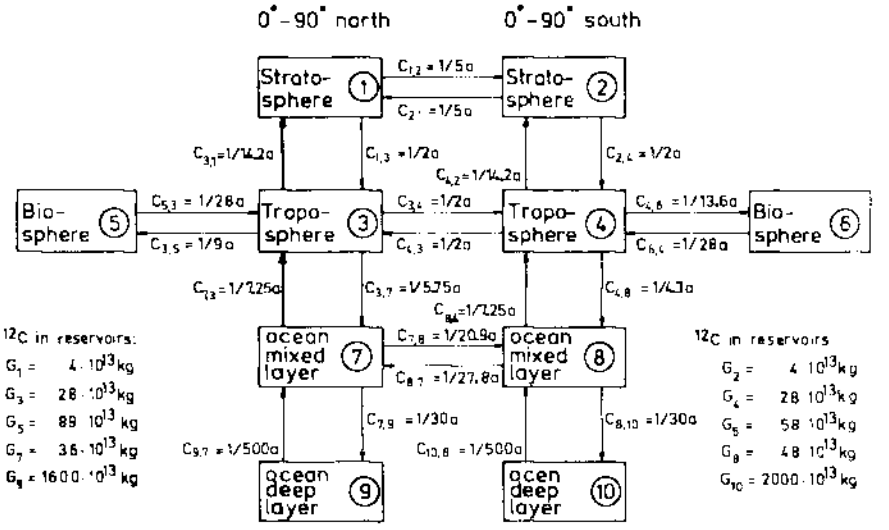

Fig. 1: Compartment model for ^{14}C

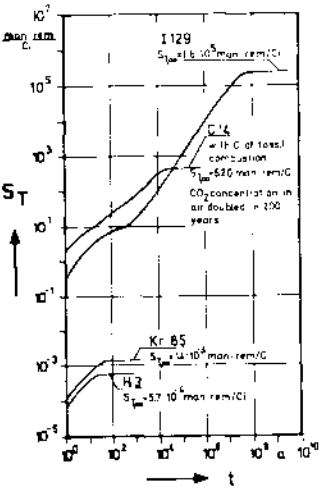

Fig. 2: Total collective effective dose equivalent commitment S_T from globally distributed radionuclides in dependence of integration time/world population 1E10)

INFLUENCE OF THE MOST IMPORTANT DATA ON THE CALCULATION OF THE MAXIMUM RADIATION EXPOSURE IN THE VICINITY OF NUCLEAR FACILITIES

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The Radiation Protection Ordinance in the F.R.G. demands that the radiation exposure resulting from the operation of a nuclear facility is kept as low as possible, taking into account the state of science and technology. In (1) the methods for the calculation of the maximum radiation exposure are determined.

RELEVANT RADIONUCLIDES

When analysing the radiation exposure due to radionuclides released from nuclear facilities in more detail, one can see that only a few nuclides deliver relevant contributions to the radiation exposure. Fig. 1 and 2 show the contributions of the most important nuclides in the vicinity of a pressurized water reactor via the air and water pathway (2).

In the case of the air pathway especially the noble gases and C 14, as well as I 131 for the thyroid, are decisive for the dose. The dose via the water pathway is determined by H 3, Cs 134 and Cs 137.

The dose in the vicinity of a reprocessing plant is governed by a few nuclides as well (2). Due to the long cooling time between the unloading of the reactor and the reprocessing of the fuel elements the relevant nuclides like H 3, Kr 85, Sr 90, Ru 106, I 129 and some transuranium isotopes are all long lived.

INFLUENCE OF THE MOST IMPORTANT DATA

The calculation of a radiation exposure has to consider the distribution of radionuclides in the atmosphere and the hydrosphere, the transfer into the foodstuffs and the transfer into the various organs after inhalation and ingestion.

Basing on this contamination of the environment, the maximum doses can only be calculated exactly for external submersion and, when using the specific activity model, for H 3 and C 14 (3).

The inhalation dose can also be calculated sufficiently exact, if the input parameters are known. Here the influence of the aerosol size distibution is only important for extreme aerosol sizes (very large or very small diameters). If the dose conversion factors are weighted over

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the size spectrum of the natural aerosols 1 µm can be deducted as a median diameter (4). Fig. 3 shows the inhalation doses for various particle diameters of a Sr 9o aerosol standardized on the dose for 1 µm.

The calculation of the ingestion dose is highly sensitive to the deposition behaviour of the airborne radioactive material, as well as the transfer and concentration factors in the food chains. When considering the size spectrum of the aerosols in the calculation of the deposition velocity for fallout and washout, it shows, that the present calculation procedure results in an overestimation of the deposition velocity by a factor of 5.

The transfer data for the most important nuclides Sr 9o, Cs 137 and I 131 have been determined after atomic bomb tests. As the values laid down in (1) are still being discussed, the importance for the dose calculation was examined.

Fig. 4 and 5 show the influence of the plant-, milk- and meat transfer factors on the ingestion dose. The dose calculation was simplified corresponding to the formulae stated in fig. 4 and 5, where $\bar{\chi}$ is the long time dispersion factor, N is the precipitation, x is the distance, $\bar{u}$ is the mean wind velocity and v is the food pathway (plants milk, meat, fish). Mostly for Sr 9o an increase of the transfer ground-plant has a considerable influence on the dose. This influence, however, decreases when the loss of radionuclides into deeper soil layers and due to harvest is considered (5,6).

For the water pathway the accumulation factors in fish and most of all the high consumption rate (39 kg/yr) are very important (fig. 6 and 7).

The variation of ground concentration due to loss of radionuclides via washing out into lower soil layers and harvest is shown in fig. 8 and 9.

It can, however, be assumed that the maximum possible doses for those nuclides which are of greatest importance for the dose, can be calculated sufficiently conservative.

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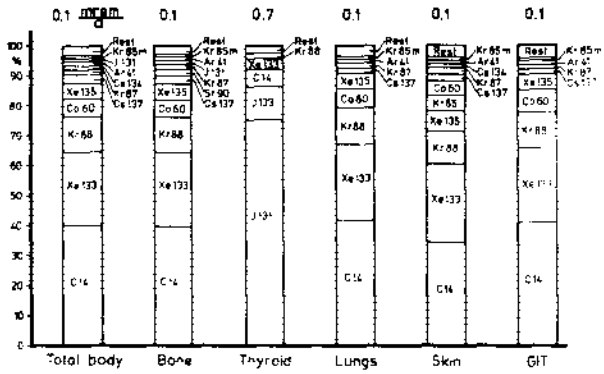

Fig. 1 Contribution of relevant radionuclides to the radiation exposure of an adult via the air pathway at the point of maximum exposure
(Pressurized Water Reactor 1000 MW_e, H=100m)

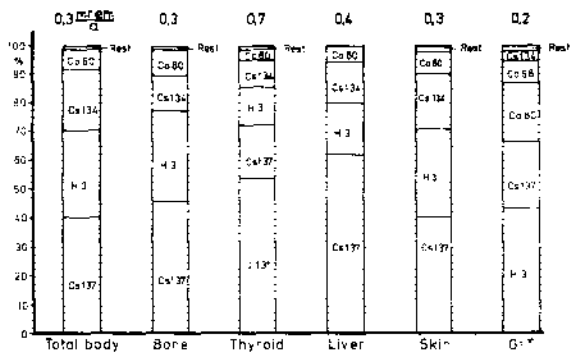

Fig. 2 Contribution of relevant radionuclides to the radiation exposure of an adult via the water pathway.
(Pressurized Water Reactor, Q = 100 m³/s)

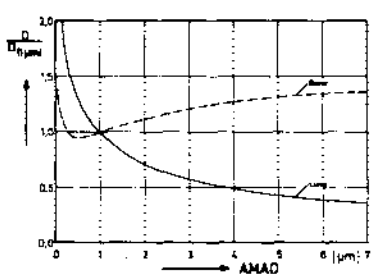

Fig. 3: Doses to lung and bone due to inhalation of Sr90 aerosols with various particle diameters, normalized on the dose for 1µm AMAD

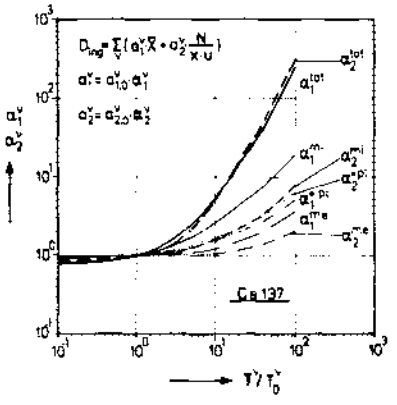

Fig 4: Change of α_1^ν and α_2^ν due to variation of transfer data.
* with loss of radionuclides

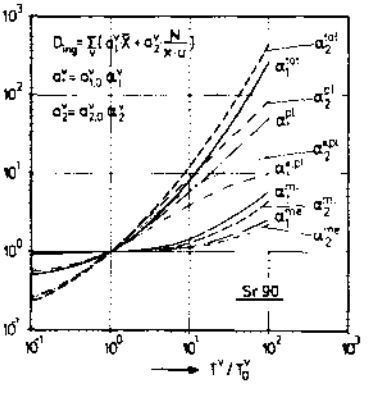

Fig 5: Change of α_1^ν and α_2^ν due to variation of transfer data.
* with loss of radionuclides

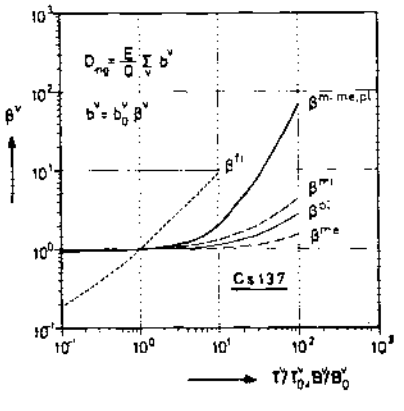

Fig 6: Change of β^ν due to variation of transfer data and concentration factor in fish.

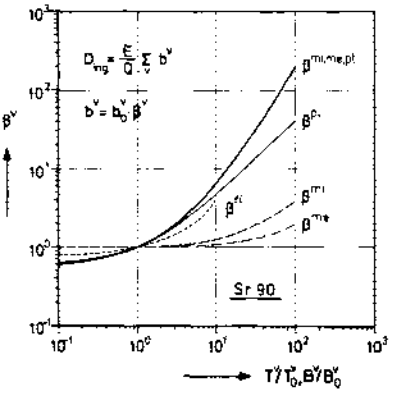

Fig 7: Change of β^ν due to variation of transfer data and concentration factor in fish.

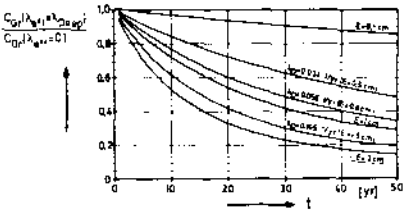

Fig 8 Relation of the ground concentration with and without loss of radionuclides due to transfer into lower soil layers E=Sinking velocity

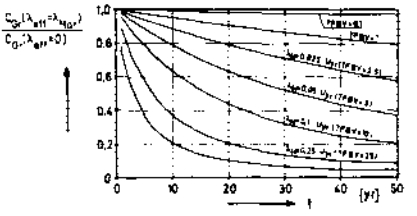

Fig 9 Relation of the ground concentration with and without loss of radionuclides due to harvest fFBV=Transfer factor soil-vegetation

IMPACT SANITAIRE DU PROGRAMME NUCLEAIRE FRANCAIS EN 1990.

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INTRODUCTION - RESUME.

Le développement prévu de l'industrie électronucléaire française dans les années à venir, soulève le problème de l'évaluation des conséquences radiologiques pour l'environnement et pour l'homme dans le cadre du fonctionnement normal de chaque installation.

Nous avons défini à l'horizon 1990 un "scénario" correspondant à l'état d'avancement prévu du programme français, en nous limitant à la filière à eau pressurisée (PWR).

Pour chaque installation du cycle du combustible (mines, usine de conversion, réacteurs etc ...), nous avons évalué à partir des données françaises disponibles, cinq types d'indicateurs du risque radiologique lié au fonctionnement normal pendant 1 an :

- les activités des rejets liquides et gazeux,
- l'équivalent de dose effectif individuel en bordure de site,
- l'équivalent de dose effectif collectif pour les populations dans un rayon de 25 km,
- l'équivalent de dose effectif individuel et collectif pour les travailleurs,
- les effets sanitaires théoriques radio-induits.

Enfin, dans le cas des radionucléides de très longue période rejetés par l'usine de retraitement du combustible irradié, nous avons calculé l'engagement d'équivalent de dose collectif étendu à la population nationale et mondiale.

LE SCENARIO A L'HORIZON 1990.

Le scénario "nucléaire" que nous avons défini tient compte de la perspective d'évolution de la demande et de l'offre d'électricité envisagée actuellement pour la France en 1990. Il est caractérisé par une production annuelle totale d'énergie électrique de 330 TWh, assurée par le fonctionnement de 30 tranches de 900 MW(e) et 28 tranches de 1 300 MW(e) de la filière PWR, pendant 5 200 heures par an en moyenne. Les quantités de matières mises en jeu par ce "scénario" au niveau de chaque étape du cycle du combustible nucléaire sont données dans le tableau n° 1. A ces quantités, il convient d'ajouter les produits de fission résultant du retraitement du combustible irradié, (environ 35 tonnes) ainsi que la production de plutonium (9t/an).

Tableau n° 1 : Quantités annuelles de matière.

Etape du cycle	Flux annuels relatifs à 63,4 GW(e) (T/an) ★
Usine de concentration	7 230 t U naturel
Usine de conversion	8 620 t U naturel
Usine d'enrichissement	8 620 t U contenu dans l'UF_6
Usine de fabrication	1 460 t U enrichi contenu dans l'UF_6
Réacteurs	1 460 t U
Usine de retraitement	1 460 t U contenu dans l'UO_2

★ 63,4 GW(e) = 330 TWh : 5 200 h.

IMPACT SANITAIRE SUR LES TRAVAILLEURS.

Les conséquences sanitaires pour les travailleurs du cycle du combustible nucléaire associé au programme que nous avons retenu, ont été évaluées à partir des données suivantes :

- nombre de travailleurs exposés,
- équivalent de dose effectif individuel moyen par voie d'exposition,
- coefficient dose-effets de la CIPR 26.

Le tableau n° 2 fourni les résultats pour l'ensemble du programme considéré [4].

Tableau n° 2 : Impact sanitaire sur les travailleurs.

Instal.	Effectif (statutaires)	Dose indiv. (mSv/an)	Dose collective (homme sievert)	Effets sanitaires
MINES	7 450	14	104,3	1,7
TRAIT.	2 020	5	10	0,16
CONVERS.	880 ★	5	4,3	0,07
ENRICH.	640	5	3,2	0,05
FABRIC.	1 350	5	6,7	0,01
REACT.	9 120	25	228 + 253 ★	7,9
RETRAIT.	4 450 ★	10	46	0,8
TRANSP.	790	4	3,1	0,05

★ Statutaires + Entreprises extérieures.

Il est à noter que l'exposition des mineurs due au radon n'est pas prise en compte dans le calcul de la dose individuelle moyenne : ce choix est lié à l'incertitude importante concernant la relation entre le "Working Level Mounth" et le "Sievert" (de 0,03 à 1 Sv/WLM) d'où la sous-estimation du risque des mineurs [1].

L'estimation de l'effectif pose le problème du personnel appartenant aux "entreprises extérieures" intervenant dans l'exploitation des installations pendant leur fonctionnement normal. Ce personnel n'est connu et n'a été pris en compte que pour deux installations (conversion et retraitement). Dans le cas précis des réacteurs, l'estimation du risque est basée sur l'hypothèse, conforme à l'expérience actuelle, d'une répartition de la dose collective entre les deux catégories de travailleurs : statutaires (40 %) et non statutaires (60 %) [3].

IMPACT SANITAIRE SUR LE PUBLIC.

Compte tenu de l'hypothèse de fonctionnement normal de toutes les installations du cycle et de leur répartition géographique (75 % en bord de rivière et 25 % en bord de mer, pour les réacteurs par exemple) nous avons estimé les effets sanitaires pour le public en général, dans un rayon de 25 km autour de chaque site. Le tableau n° 3 fournit les résultats pour l'ensemble du cycle du combustible [4].

Tableau n° 3 : Impacts sanitaires sur le public.

Installations sites		Population (< 25 km) (1)	Dose indiv. critique mSv/an (2)	Dose indiv. moyenne mSv/an	Effets sanitaires
Mines Traitement du minerai	5	293 900	(10^{-2}WL)(5)	(10^{-4} WL) $1{,}4.10^{-1}$ 3.10^{-2}	$7\ .10^{-1}$ $1{,}4.10^{-2}$
Conversion	2	358 500	-	-	-
Enrichiss.	1	202 000	$2{,}6.10^{-2}$	$5{,}2.10^{-5}$	$1{,}7.10^{-4}$
Fabrication du combust.	1	92 400	$8\ .10^{-3}$	$1{,}6\ .10^{-4}$	$2{,}4.10^{-3}$
Réacteur (6)	15	1 986 500	$1{,}4.10^{-2}$ (3) $5{,}2.10^{-2}$ (4)	$9\ .10^{-4}$	$1{,}4.10^{-2}$
Retraitement (7)	1	25 900	1.10^{-1} $7{,}2.10^{-2}$	$1{,}1.10^{-2}$	$4{,}7.10^{-3}$
Transport	-	-	-	-	0,2
TOTAL	27	2 958 600	-	-	0,44

(1) Calculée pour la population française en 1978
(2) Exposition de l'individu théorique le plus exposé pour chaque site
(3) Equivalent de dose effectif de l'adulte le plus exposé
(4) Equivalent de dose effectif de l'enfant le plus exposé
(5) WL : working level (unité d'exposition dans le cas du radon)
(6) Valeurs pour 3 types de sites, respectivement, bord de rivière, 4 x 900 MW(e) ; bord de rivière 4 x 1300 MW(e), bord de mer 4 x 1300 MW(e)
(7) Emetteurs α non compris

Dans le cas de l'usine de retraitement nous avons évalué, pour les radionucléides, de longue période (I^{129}, Kr^{85}, C^{14}, H^3) l'engagement de dose effectif collectif pour 500 ans dû à un an de fonctionnement normal, pour la population française et mondiale.

Compte tenu du comportement de ces radioéléments dans l'environnement ainsi que de leur propriété physique, nous avons obtenu les valeurs suivantes [2] :

Tableau 4 : Impact sanitaire des radionucléides de longue période.

Equivalent d'engagement de dose collectif (homme-Sievert)		Effets sanitaires radioinduits théoriques (★)	
National	Mondial	National	Mondial
197,6	286,6	3,3	4,7

★ Somatiques + Génétiques.

CONCLUSIONS.

A la lumière des résultats présentés ici nous pouvons souligner, entre autres, les points suivants :

1. La différence importante entre le risque collectif des travailleurs et le risque collectif du public.
2. La spécificité du problème de l'évaluation de la dose collective des mineurs.
3. Le pourcentage de la dose collective des travailleurs des centrales nucléaires exprimés par rapport à la dose travailleurs totale (~ 70 %).
4. Enfin le rôle particulier de l'usine de retraitement dans le risque collectif du public en général.

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DIAGNOSTIC X-RAY EQUIPMENT EVALUATION IN BRAZIL

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As in many countries all over the world, also in Brazil there was a marked increase of medical X-ray installations and the number of examinations "per capita" during the last decades. Up till now there are a very little statistics regarding frequency of X-ray examinations the main factors affecting radiation exposure and the conditions of the equipment available.

After 1977, the IRD had the possibility to perform a survey on the main part of diagnostic X-ray installations all over the country. Till the moment we have checked 790 X-ray machines.

This program included statistics as well as tests of the technical conditions. These tests included: radiation quality (i.e. kilovoltage and filtration), tube out put, beam collimation and protection devices.

About 34% of the X-ray equipment tested in this work are used for dental radiography and 66% are used for other routine examinations includind fluoroscopy and those performed by mobile units.

Table I presents a survey of the number of physicians (radiologists), radiographers, examinations per month, films used and lost, in 10 big Hospitals, where 120 X-ray machines are installed. When these measurements have been performed, around 10% of the machines were not functioning because of different reasons, mainly due to technical failures and lack of a correct place to be installed and/or of trainned people to run them.

TABLE 1. Survey of 120 diagnostic X-ray machines.

Number of Hospitals	10
Number of X-ray Machines	120
Physicians (Radiologists)	80
Radiographers	260
Examinations per Month	51100
Films Used per Month	157300
Films Lost per Month	4000

RADIATION QUALITY

Regarding radiation quality in practice means testing the coincidence of the quilovoltage indicated at the control panel with the quilovoltage applied to the tube and the inherent filtration. As it was proved impossible to use a voltage divider in such a field test we choose the method published by Ardran and Croods (1) for determination of the quilovoltage. It is based on film dosimetry using a special film-screen combination in specially prepared cassette. One half of the cassette contains a pair of high resolution screens and the other half a pair of high speed screens.

The cassette is covered with a sheet of lead in which two rows of holes are cut, one over each pair of screens. The row on top of the high speed screens is covered with a copper step wedge. When a film is exposed in this cassette it will show a row of spots with identical optical density resulting from the high definition screen an another with decreasing optical densities, according to the different copper thicknesses covering the high screen causing the exposure of this part of the film. From the spot in this row which shows the same optical density as the spots in the other row one can get the applied quilovoltage by means of a calibration curve established for this device.

TOTAL FILTRATION

The total filtration was determined by means of HVL measurement at a certain kV value determined in the described way. The total filtration corresponding to measured HVL was taked from NCRP Report 33. (2)

TUBE OUTPUT

The purpose of measuring the tube output under defined conditions was to get an idea about the conditions of the tube, i.e., the age or, if necessary, the power of an unknown gerator type.

The dosimeters used were a calibrated Baldwin Farmer with a 30 cc chamber and a Pitman with a 35 cc chamber.

EXPOSURE TIME

As most installations did not include an automatic

exposure control the correct functioning of the timing was of special importance. This was checked with an electronic device, developed by our Institute, covering a range of 1 ms up to 900s.

COLIMATION OF THE USEFULL BEAM

When the tube housing was equiped with a ligth beam collimator the coincidence of the light field with the radiation field was checked by exposing a film.

When only cones were available the field size in the plane of the patients surface, i.e., 20 cm above the table top was calculated. In Table 2 we can see the type of collimators used in the machines of this national survey.

TABLE 2. Type of collimator

Cone	36 %
Light Beam Diaphragm	51 %
Without Collimator	13 %

SPECIFIC TESTS

Besides this basic program applicable to all installations a few more tests were done in special cases like image amplifiers with TV chains and tomographic equiment. These tests consist of measuring the dose rate to the skin of the entrance side and to the entrance of the image amplifier using a phantom.

With tomographic equipment the simetry of the tomographic movement and the coincidence of the indicated height of the plane with the height of the plane with the actual height were tested according to ICRU Publication 89. (3)

RESULTS

From all the measurements realized ln each X-ray equipment, we got for: total filtration, kivoltage, exposure time and field size the figures shown in Table 3.

TABLE 3. Survey of 522 Diagnostic X-ray Machines

Kind of test	Correct (%)	Not correct (%)
Total Filtration	38	62
Kilovoltage	58	42
Exposure Time	63	37
Field Size	46	54

In regard to Radiation Protection, 40% of the rooms were in wrong conditions related to the staff and the public. We emphasize that almost all the X-ray Departments

visited do not use any personal dosimeter.

In the field of Odontological X-rays, we found that 30% of the rooms were incorrect in relation to radiation protection.

In conclusion, from 268 Odontological X-rays machines checked, only 9% were working in correct conditions, in regard to the technical parameters and the radiation protection.

Finally, Table 4 presents the figures for the technical parameters checked in the 268 machines.

TABLE 4. Survey of 26% Odontological X-ray machines

Kind of test	Correct (%)	Not correct (%)
Total Filtration	79	21
Kilovoltage	54	46
Exposure Time	52	48
Field Size	70	30

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SURVEILLANCE OF X-RAY MACHINES IN ISRAEL

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Ionising radiation is widely used in Israel for medical, industrial and research purposes. Early in 1964 it was recognized by the Israeli Health Authorites that the use of X-ray machines for diagnostic purposes is the main source of artificial irradiation of the population. Table 1 shows the distribution of radiation machines (mostly diagnostic) that are currently under surveillance of the Israeli Ministry of Health.

TABLE 1. RADIATION MACHINES USED FOR MEDICAL PURPOSES IN ISRAEL

Radiography	350
Radiography + Fluoroscopy	270
Radiography Portable	200
Fluoroscopy Portable	60
Mammography	9
CT	5
Dental	3,400
Panoramic	40
Ortovoltage	20
Co^{60}	10
Linear Accelerator	4
Veterinary	25

In order to assure that X-ray machines are used according to the recommended safety standards for both patients and operators, a nationwide surveillance program of diagnostic X-ray machines has been carried out by the Research Institute for Environmental Health, which is operated by the Ministry of Health and the Tel-Aviv University. According to the Israeli law, all the X-ray machines owners must apply to the Ministry of Health in order to obtain an operating license for the equipment. At present diagnostic X-ray machines are surveyed at least once every two years while dental machines are surveyed once every three to five years. The surveillance procedure includes the following items:

a) Registration of the diagnostic X-ray machines.
b) Check of the structural shielding (scattered radiation).
c) Check of the reproducibility and linearity of the X-ray machine.
d) HVL measurements.

e) Check of the X-ray-light field alignement.
f) Check of inherent filtration.
g) Instruction of machine operators in appropriate radiation control procedures.
h) Issue of recommendations regarding personal dosimetry.

All the tests are carried out following BRH's Routine Compliance Testing for Diagnostic X-ray Machines. Furthermore the Institute advices machine owners about the design of structural shielding.

In the last years there has been an increased concern about doses to the patient following X-ray examinations. In order to compare X-ray techniques used in different hospitals in Israel, the NEXT (Nationwide Evaluation of X-ray Trends) program, developed by the BRH was utilized. On the basis of the findings of this project, necessary correction steps were taken in order to reduce the radiation dose to patients. It is planned to extend the NEXT program in the future, in order to compare the dark room conditions among different hospitals. It is believed that the lack of standardization in the dark room conditions is responsible for overdoses to patients. Furthermore, a comparison of X-ray image qualities using appropriate patterns will be carried out.

Following an initiative of the Institute, a special Committee was nominated by the Minister of Health, in order to evaluate the health implications of the irradiation of pregnant women. The Committee recommended that doses below 5 Rem, in any stage of the gestation period do not justify performance of an abortion; while doses over 10 Rem, in the third to twelfth week might justify performance of an abortion. In any event the personnel of the Institute carry out the estimation of the dose to the foetus. This is done either by calculations, using NCRP Report 54 recommendations, or by in-phantom measurements.

Recently, the new recommendations of ICRP 26 were implemented in Israel. At present, doses beyond 150 mR/month are reported to the Institute by the Soreq Personal Dosimetry Service, and the reasons to this overexposure are investigated.

PATIENT DOSE EVALUATIONS FROM MEDICAL X-RAY EXPOSURE IN ITALY: AN ANALYSIS OF NEXT DATA

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INTRODUCTION

The average annual whole body dose to individuals is currently about 182 mrem; the contribution due to medical usage is about 73 mrem per year (1).

From a study of CNEN (National Committee for Nuclear Energy) and ISS (Istituto Superiore di Sanità), it was found that the radiographic films used in Italy during 1975 amounted to more than 90 millions, with an annual increasing rate of 8%. The estimated contribution to the genetically significant dose, due to radiographic examinations, was about 40 mrem (2).

Such non negligible values of genetically significant dose are due to the same factors as for other countries: maximum voltage and half value layer are two or three times the least value; current by time product for the same examination ranges up to two orders of magnitude. But as one of the most determining factors, usually beam dimensions are considerably wider than film dimensions.

Therefore, it can be drawn that the ICRP Publication 26 suggestions, as far as patient radiation protection is concerned, are not taken into due account, in Italy as elsewhere.

NEXT PROGRAM DESCRIPTION

NEXT is an acronimous for Nationwide Evaluation of X-ray Trends (3). It is a program aimed at collecting all the available information on the different techniques used when performing 12 selected radiological examinations.

During a NEXT survey, the facility operator is asked to set up the X-ray apparatus as though he were going to perform the most frequent examination for a patient with standard dimensions. The inspector records information about the type of facility and apparatus, workload, operator training; he also records the indicated voltage, current intensity, time, film size and then he measures the source-film distance, the exposure at a fixed distance, the exposure with three increasing aluminum thicknesses and the beam size. The ionization chamber used for the exposure measurements is calibrated by the central authority; a special apparatus is supplied to ensure the reproducibility of the exposure

measurements.

From the registered parameters and the reported measurements, it is then possible to calculate the half value layer, the skin entrance exposure and the gonad dose for all the inspected facilities.

Radiological techniques that cause too high gonad doses are easily picked out, by comparing the recorded, measured and computed parameters for the techniques used in a facility with those used in all the inspected facilities.

Each facility may then revise its wrong techniques; each local administrations may choose the most adequate program to improve the patient radiation protection.

NEXT PROGRAM IN ITALY: FIRST RESULTS

NEXT program was fitted to italian conditions and, after a pilot study, it was presented in May 1977 to the competent Regional Administrations.

As central authorities, CNEN and ISS reserved to themselves the data processing and instrument calibration. All the NEXT surveys should be performed by regional operators.

Regional Administrations have been operating in Italy only since 1975 and economical, social, organization problems have been braking their working.

Therefore, it must be considered as a great success that the Umbria Regional Administration immediately begun to collect the data for the NEXT program. After a few months, the Emilia Regional Administration made the NEXT program start in the territory of one of its provinces.

The program is now going to be completed in Emilia; three other Regional Administrations are very likely to include the NEXT program in their health programs.

Table 1 shows the most relevant results for the recorded parameters and for some selected projections; in particular it should be stressed that the current by time product ranges from 2 mA.s up to 200 mA.s for the chest P/A projection.

For the same projection, the voltage varies between 40 kV and 90 kV; the half value layer ranges between 0.9 Al mm and 4.0 Al mm and the beam area to film area ratio ranges from 0.6 to 8.2.

The other projections are less frequent than chest P/A projection and, therefore, less data are related to them; although, the involved parameters are spread over wide intervals.

Table 2 shows the minimum, maximum and weighted mean values for the computed parameters, such as skin entrance exposure, ovarian and testicular doses. The parameters range in remarkably wide intervals; though the weighted mean values are closer to least

values, nevertheless it should be stressed that the patient in radiographic examinations keeps receiving non negligible doses in a great number of cases.

CONCLUSIONS

The NEXT program has been carried out up to now only in two regions, slightly representative of the health conditions in Italy; a southern Regional Administration at least would be necessary.

Nevertheless, the results shown here are enough to indicate that often patient doses are unjustified and non optimized techniques are chosen. NEXT results show as well the fields where local Administrations should operate to reduce patient dose and to improve radiological techniques.

TABLE 1 - Minimum and maximum values of some registered parameters for a few selected radiological projections

Projection	Voltage kV		mA.s		SSD(°) cm		B/F(°°)		HVL Al mm	
	min	max	min	max	min	max	min	max	min	max
Chest P/A	40	90	2	200	90	200	0.6	8.2	0.9	4.0
Abdomen A/P	60	100	12	160	74	120	1.0	2.3	1.6	4.5
Cervical Spine A/P	52	70	25	200	80	136	1.2	2.8	1.4	2.2
Lumbar Sacral Spine A/P	55	100	20	200	75	114	0.9	3.5	1.5	5.0
Dental Bitewing P	50	60	2	11	9	13	1.7	2.3	1.1	1.6

(°) Source - Skin Distance

(°°) Beam area to Film area ratio

TABLE 2 - Minimum, maximum and weighted mean values of the computed parameters for a few selected projections

Projection	Skin Entrance Exposure mR			Ovarian Dose mrem			Testicular Dose mrem		
	min	max	mean	min	max	mean	min	max	mean
Chest P/A	8	375	56	0.5	26	3	0.5	15	0.5
Abdomen A/P	391	3328	1222	55	644	252	1	57	13
Pyelogram A/P	285	1808	784	34	418	153	1	20	6
Lumbar Sacral Spine A/P	187	3173	1311	21	538	286	0.5	46	15

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NEUTRON PRODUCTION AND LEAKAGE FROM MEDICAL ELECTRON ACCELERATORS *

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NEUTRON PRODUCTION

For medical accelerators operating above about 10 MeV, there is an unavoidable production of photoneutrons which add to the head leakage. The yield of photoneutrons produced by electron beams incident on thick targets has been calculated (1,2) and found to increase rapidly with primary electron energy up to approximately 25 MeV, and more slowly thereafter (Fig. 1). Dividing the neutron fluence for W (or Pb) by the photon absorbed dose (calculated by the Monte Carlo code EGS (3)) for the same conditions gives us the ratio plotted in Fig. 2. This ratio, representing the maximum achievable with practical materials, becomes nearly constant above about 25 MeV incident electron energy where its value is about 2.1×10^5 neutrons cm^{-2} rad^{-1}. Measurements that fall significantly below the curve of Fig. 2 are likely due to photon absorption in components that are not high-Z materials, or measurements made with the moveable jaws open. Points that fall significantly above probably mean there is substantial loss of electron beam elsewhere than the intended target.

Neutron spectra in the giant resonance can be adequately described by a Maxwellian distribution which involves a single parameter, the nuclear temperature T (in MeV) for the particular nucleus. The spectrum peaks at $\hat{E}_n$=T and has an average energy of $\bar{E}_n$=2T. The neutrons are emitted almost isotropically and values of T generally lie in the range 0.5 to 1.0 MeV for high-Z materials.

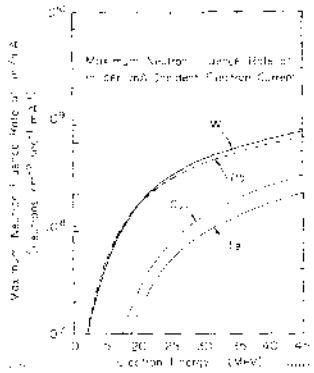

Figure 1. Maximum neutron fluence rate at 1 m.

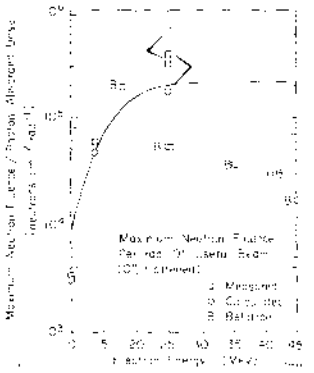

Figure 2. Neutron fluence per treatment rad (0°, flattened).

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DEGRADATION OF NEUTRON ENERGY IN THE TREATMENT HEAD

The typical medical accelerator has massive shielding around the target to provide photon shielding and produce a collimated beam of x-rays. The photon shielding is of some heavy metal such as W or Pb, with some Fe and Cu present. The significant neutron energy loss mechanisms in the heavy elements are inelastic scattering and (n,2n) reactions. Because these cross sections together amount to 1 - 2 b in W and Pb, the typical neutron undergoes several collisions within the shielding. In addition, a large amount of elastic scattering takes place (4 - 5 b). This increases the effective path length in the shielding, thereby offering greater opportunity for inelastic reactions. Although the energy degradation is significant, the attenuation of neutron fluence is small because the capture cross sections of W and Pb are small down to thermal energies. Fig. 3 shows the integral photoneutron spectra for 15-MeV electrons on W and for ^{252}Cf fission neutrons; these are quite similar. On the same figure, we show the neutron spectrum from 15-MeV electrons on W after the neutrons have penetrated 10 cm of W. Also shown is the further degradation due to reflection in the concrete room in which medical accelerators are usually placed. Concrete-scattered neutrons add to the neutron fluence but their contribution to the patient's dose is much smaller, because of the lower energy spectrum. The average energy of the room-energy component is about ¼ of that of the direct component.

NEUTRON SOURCES IN A MEDICAL ACCELERATOR

Most medical linacs provide fields up to about 35×35 cm^2 at 100 cm from the target. The main collimator covers all forward directions beyond the extremes, with a half-angle of about 14 degrees. If the target, flattener, jaws and main collimator are of the same material, it would be W or Pb which have nearly equal neutron yields (1). Only photons above about 8 MeV are effective in producing neutrons.

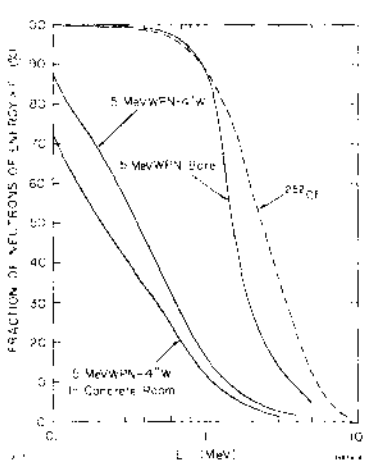

Figure 3. Neutron integral spectra, modified by shielding.

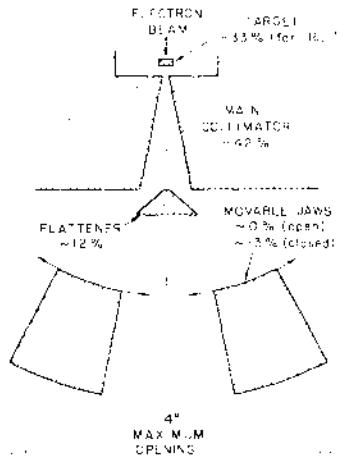

Figure 4. Neutron-producing components (not to scale).

These are mostly forward-directed and are absorbed in the main collimator or pass through and strike the flattener or the jaws. Using the Monte Carlo program EGS (3), we have calculated the angular distribution of photons of energy above 8 MeV (4). Using this angular distribution and the neutron yield as a function of material thickness (1) we have calculated the neutron yield from treatment head components. This is done for the choices of materials summarized in Table I.

TABLE 1. Summary of Neutron Source Calculations (25 MeV Electrons).

Material and Percentage of Neutrons			Fraction of Infinite W Yield	Neutrons per rad
Target	Flattener	Main Collimator		
W 38%	W 14%	W 48%	86.5%	2.2×10^{10}
W 43%	Fe 2%	W 55%	64%	1.6×10^{10}
Cu 9.2%	W 30%	W 60.5%	84%	2.1×10^{10}
Cu 12.5%	Fe 5.7%	W 82%	64%	1.6×10^{10}

The yields for an all-W design are also shown as percentages of the maximum possible in Fig. 4. Note that while the total neutron yield is not very different for the choices of materials given in Table 1, the fraction from different components varies. Similar calculations and measurements of total fast-neutron source strengths (5) for several accelerator models (Table 2) indicate that the overall head leakage can be calculated with an accuracy no worse than about ± 50%.

TABLE 2. Calculated and Measured Neutron Yield per Photon rad.

Accelerator	Energy (MeV)	Relative Neutron Yield (n/rad)	
		Calculated	Measured
ATC 25 MeV Betatron	25	6.8×10^{9}	6.9×10^{9}
Siemens 42 MeV Betatron	42	3.8×10^{9}	3.7×10^{9}
Varian Clinac 35 (Old)	25	4.3×10^{10}	8.1×10^{10}
Varian Clinac 35 (New)	25	2.2×10^{10}	6.2×10^{10}
Varian Clinac 18	10	3.9×10^{8}	4.2×10^{8}
Siemens Mevatron XX	15	5.8×10^{9}	7.6×10^{9}

LEAKAGE NEUTRON DEPTH-DOSE CURVES

Because the leakage neutrons are considerable degraded in energy, one would expect them to be attenuated quite rapidly in tissue. We have calculated this depth-dose distribution using the computer code MORSE (6), in which the source spectrum used was that of either a 14-MeV or 25-MeV (incident electron energy) photoneutron spectrum

surrounded by 10 cm of W. The phantom was a 1-m × 30-cm (diam) H_2O cylinder, centered at 1 m from the point source and oriented perpendicularly to the beam direction. Fig. 5 shows the rapid falloff of absorbed dose of these two spectra.

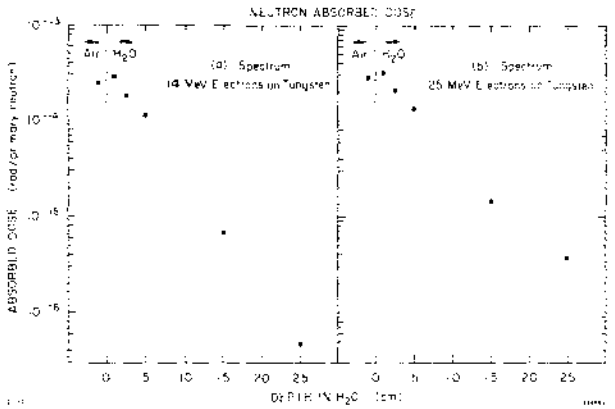

Figure 5. Depth-dose distributions in H_2O for neutron spectra from therapy targets, modified by 10 cm of W.

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RETENTION OF MOLYBDENUM-99 IN ADULT MAN

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The ^{99}Mo has been used, in water-solubie forms such as sodium and ammonium molybdate, in radionuclide investigations (5;6) and it is present as an impurity in eluates from molybdenum-technetium generators (1), extensively used in nuclear medicine.

The metabolism of molybdenum, which plays a essential role in flaving-dependent metalloenzyme systems, has been studied and transport, storage and excretion of this element are known (2;7): there is a rapid uptake of orally ingested Mo by all tissues with preferential accumulation in the liver, kidney and bone. Intravenously administered molybdenum disappears from circulatory system very rapidly (3). The available data, however, are not expressible in convenient mathematical form to represent retention with acceptable accuracy for radiological protection.

In the present note retention data obtained by measuring the levels of molybdenum-99 in excreta of 10 patients injected intravenously are reported and the whole body retention function derived by multicompartmental analysis of these data is proposed.

METHODS AND MEASUREMENTS

Subjects were patients of the Department of Medicine of the Sassari University, who have been investigate using sodium pertechnetate-^{99m}Tc. Table I lists the age, sex, waigth and pathology of these patients and administered doses.

Measurements of urinary and fecal excretion of ^{99}Mo were carried out with the method of the gamma-spectrometry, using NaI(Tl) crystals and multichannel analyzer, during the first 2 weeks after injection. Attempts to measure whole body radioactivity did not give good risults because of the low level of the ^{99}Mo. For the same reason, also the results of measurements carried out on samples of plasma and whole blood are poorly meaningful. Rosoff and Spencer (3), who carried out studies on the fate of molybdenum in man injecting tracer doses of 50-100 μCi, refer that 1 h

after injection the concetration of Mo in plasma and in whole blood ranges from 2.5 to 5% of the initial dose.

TABLE 1. Patients and administered doses

Patient	Age	Sex	Weigth	Diagnosis	Mo-99 Dose (μCi)
1	39	F	73	Carcinoma of the breast with metastases	0.2
2	47	M	57	Multiple myeloma	0.4
3	54	F	58	Carcinoma of the breast with metastases	1.2
4	60	M	60	Carcinoma of the lung with metastases	0.8
5	70	M	78	Metastatic epydermoid carcinoma	1.7
6	50	F	63	Carcinoma of the breast with metastases	1.0
7	49	M	68	Carcinoma of the kidney with metastases	0.1
8	66	F	60	Perivasculitis	0.5
9	54	F	55	Carcinoma of the breast with metastases	1.3
10	77	F	62	Carcinoma of the breast with metastases	2.0

RESULTS

Estimates of the retained molybdenum during the first 2 weeks, calculated from the difference between the initial dose and cumulative urinary and fecal excretion, are given in Table 2 and are reported in graph of Fig. 1. Fecal excretion was very low: the ratio of urinary to fecal excretion was between 25 and 30 for every patient. In Table 2 is also indicated the number of patients to whom the data refer.

The fractional retention, R, for ^{99}Mo can be represented accurately by a four-component exponential function of time:

$$R=\sum_{i=1}^{n} A_i \exp(-\lambda_i t) \qquad (1)$$

The parameters of equation (1) are reported in Table 3. Fig. 1 shows the fitting of the experimental data by this function.

TABLE 2. Retention of injected ^{99}Mo

Days after injection	Number of measurements	Whole body retention
1	10	89.8
2	10	86.4
3	8	83.4
4	8	81.3
5	7	78.9
6	7	76.4
7	7	74.6
8	5	73.4
9	5	72.1
10	3	70.3
11	2	69.4
12	2	68.3
13	1	67.3

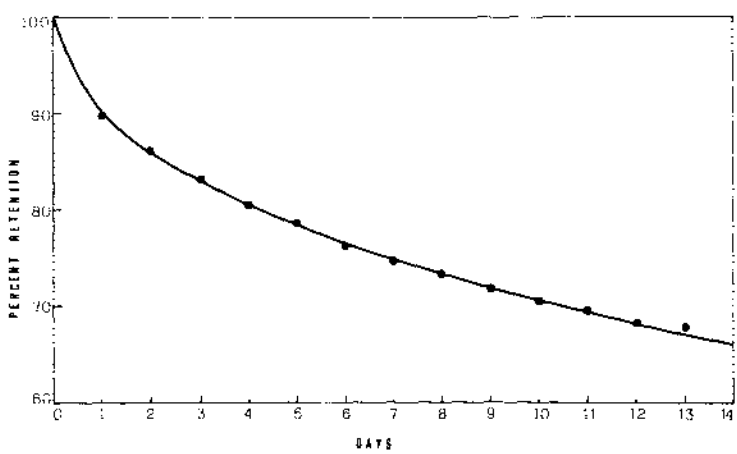

Figure 1. Whole body retention of ^{99}Mo in the first 13 days after injection. Data are fitted by function (1) with parameters of Table 3

TABLE 3. Results of four-component exponential analysis of retention data for 99-Mo between days 1 and 13. (See also Fig. 1)

Component no	Percentage of injected dose	Biological half-life (days)
1	9.5	0.58
2	17.8	5.33
3	8.2	25.67
4	64.5	86.62

Retention data of Table 2 and Fig. 1 are in good agreement with those of Rosoff and Spencer (3). They found that cumulative urinary excretion of Mo in 10 days was 29% and 24% of the injected in 2 subjects. The cumulative fecal excretion in the same period was less than 1% for the first subject and 6.8% for the second one. The higher value was due to the passage of 4% of the initial dose in only one stool specimen on day 4.

Internal exposure for 5, 15 and ∞ days after injection of 1 μCi of Mo-99, calculated by the proposed retention function, is of 2.52, 3.28 and 3.34 μCi.days. The corresponding dose, calculated from the data of Snyder et al. (4), is 0.85, 1.10 and 1.12 millirads.

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ORGAN DOSES IN DIAGNOSTIC RADIOLOGY

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Extensive use has been made of a dosimetry technique for calculating organ doses in diagnostic radiology. The computations are made with a Monte Carlo radiation transport procedure and various mathematical anthropomorphic phantoms. This method simulates and records the energy deposition of x-ray photons in the phantoms by following the radiation interaction histories of a large number of incident photons using known physical descriptions of the interaction processes and recording energy depositions at the sites of interaction. The physical processes treated are limited to the photoelectric effect and Compton scattering, since the initial photon energies in the diagnostic range are less than 150 keV. A variety of dosimetry data of general applicability for estimating organ doses from diagnostic x rays have been developed.

TISSUE-AIR-RATIOS FOR A REFERENCE ADULT PATIENT

Tissue-air-ratios, expressed as the average absorbed dose (rad) to the organ per unit exposure (R, free-in-air) at the reference plane of the organ, have been tabulated for a reference adult phantom for collimated, normally-incident 4-cm x 4-cm monoenergetic photon beams in the range 20 to 100 keV (1). These tabulations are for beams incident upon the phantom on the front, rear and lateral surfaces.

The tissue-air-ratios can be converted to organ doses for a specific x-ray projection using pertinent information on x-ray spectra, projection geometries, and field sizes and locations. The tissue-air-ratios for component beams included in the x-ray field and for energies included in the x-ray spectrum are combined to simulate the desired conditions. A computer program in FORTRAN IV which performs these calculations on an IBM 370/168 system for several organs (lungs, active bone marrow, ovaries, testes, thyroid, uterus (embryo), and total body) is available (2).

ORGAN DOSES - REFERENCE ADULT PATIENT

Organ doses for a number of radiographic views and projections have been computed for a reference adult patient (3). These doses are normalized to a convenient numerical entrance exposure of one roentgen. Two parameters, entrance exposure at skin entrance (free-in-air) and beam quality must be measured or estimated by the user to convert these values to the conditions at a particular clinical facility. Utilizing typical technique factors (beam quality, entrance exposure and number of films of various views) observed in a 1970 nationwide study of diagnostic x-ray exposure conditions, organ doses were computed for common radiographic examinations. Table 1 presents some of these results.

TABLE 1. Organ doses for some common radiographic examinations

	Organ Dose (mrad)				
Examination	Thyroid	Active Bone Marrow	Breasts	Testes	Ovaries
Chest	7	4	14	< 0.01	0.06
Thoracic Spine	75	43	276	< 0.01	0.6
Lumbar Spine	0.3	126	not computed	7	405
Upper GI	7	117	53	0.4	45
Barium Enema	0.2	298	not computed	58	787
Pelvis	< 0.01	27	not computed	57	148
Full Spine	271	35	234	10	100

The conditions simulated assume good collimation (field size equal to film size) and proper alignment of the x-ray field with appropriate anatomical landmarks. Each examination type has a different combination of exposed organs and absorbed dose to these organs. The data clearly demonstrate that no single organ dose can serve as an indicator of total radiation impact for all x-ray examinations.

MAMMOGRAPHY

In many discussions of absorbed dose in the breast from mammography, the typical dose to the breast is derived from an assumed depth-dose of approximately 20 percent at the midline (3-cm depth). After correcting for backscatter and the exposure-to-absorbed dose conversion factor thia is equal to about 250 mrad per 1 roentgen skin exposure (free-in-air). This assumption is an oversimplification, for there is a wide range of doses that result from the present day practice of mammography. Table 2 presents midbreast doses as a function of HVL per one roentgen exposure for mammography, interpolated from recent data of Hammerstein, et al (4).

TABLE 2. Mammography breast doses for 1 R entrance skin exposure (craniocaudad view)

	Midbreast Dose[a] (mrad/R)					
HVL, mm Al	0.2	0.4	0.6	0.8	1.0	1.5
Tungsten target, Aluminum filter	--	110	170	230	285	430
Molybdenum target, Molybdenum filter	25	85	145	--	--	--

a Absorbed dose in a small mass of mammary gland embedded at 3-cm depth in a 6-cm medium of 50 percent adipose and 50 percent glandular tisaue.

Current work will compute absorbed dose in varying breast compositions using the Monte Carlo technique and a number of geometrically described breast phantoms. The dosimetric quantities that can be computed include the midbreast absorbed dose, the average absorbed dose throughout the breast, and the absorbed dose to only the glandular tissue within the breast. A wide range of mammography conditions are being studied.

ORGAN DOSES - REFERENCE PEDIATRIC PATIENTS

Organ doses for a number of radiographic views and projections have also been computed for three reference pediatric patients, a newborn, one-year old and five-year old (5). It was too costly to generate tissue-air-ratios for the three pediatric phantoms; therefore, a more direct approach was used. In addition, the variation in technique factors and the variety of x-ray projections are more limited in pediatric radiology (6). For each projection, the computation started with the specified characteristics of the x-ray projection including source-ro-image receptor distance (SID), field size and location, and x-ray spectrum matched to the desired beam quality with respect to kVp and half-value-layer. The output of the computation is the organ dose (mrad) to the various organs for a one roentgen entrance exposure (free-in-air). Table 3 is a sample set of data for a pediatric projection.

TABLE 3. Pediatric organ doses for 1 R entrance exposure, AP abdomen

		Organ Dose (mrad/R)					
HVL, mm Al		2.0		2.5		3.0	
Collimation		a	b	a	b	a	b
Testes	Newborn	86	910	144	1,000	152	1,120
	1-year	105	1,070	105	1,070	105	1,070
	5-year	125	1,070	125	1,070	125	1,070
Active	Newborn	91	159	127	211	137	225
Bone	1-year	69	99	100	140	112	151
Marrow	5-year	55	69	83	101	90	112
Lungs	Newborn	49	439	66	497	67	498
	1-year	35	227	48	255	55	290
	5-year	39	102	47	123	54	135

a Field collimated to body part: newborn 13 X 13, 1-year old 18 X 21, 5-year old 23 X 30 (in cm).
b Field collimated to film size: newborn 20 X 25, 1-year old 25 X 30, 5-year old 28 X 36 (in cm).

SOMATIC DOSE INDEX

For somatic effects there are several organs at risk and each type of diagnostic examination results in a different spatial relationship between the significant organs and the x-ray beam. Each type of examination also employs different technique factors to yield the desired diagnostic information. Consequently, every examination results in a unique non-uniform distribution of absorbed dose among the organs, and some method of accounting for this non-uniform distribution is necessary to assess the overall impact from a given exposure.

A somatic dose index (I_D) has been formulated which represents the uniform dose to the organs at risk that has the same somatic detriment as the non-uniform doses absorbed by the individual organs (7).

$$I_D = \frac{\sum_{i=1}^{n} s_i \alpha_i \bar{D}_i}{\sum_{i=1}^{n} s_i \alpha_i}$$

where: s_i is the relative severity of the somatic effect induced in organ i,

α_i is the risk coefficient for the effect in organ i (cases/10^6 person-yr-rad),

$\bar{D}_i$ is the average absorbed dose in organ i.

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INTERNAL RADIATION DOSIMETRY OF F-18-5-FLUOROURACIL

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Although 5-Fluorouracil (5-FU) is widely used in the oncological clinic for palliative therapy of advanced colorectal and breast tumors, its efficacy in such patients is only about 30% (1,2). We had observed that a differential distribution of F-18 occurred in mice bearing the L-1210 Lymphocytic Leukemia tumor following administration of F-18 labeled 5-FU, when the responsive variant was compared to the non-responsive tumor (3). We had suggested that the differential distribution, kinetics and/or metabolism of this drug in patients might be of use in predicting individual response to chemotherapy with 5-FU, and subsequently perhaps of determining the optimal dosage and dosage regimen required.

The labeling of 5-FU with the positron emitting radionuclide F-18 (whose $t_{\frac{1}{2}}$ is 110 minutes) can readily be accomplished in high yields and in a manner suitable for humans (4).

In the following study the projected internal radiation dose to different human organs per millicurie of injected F-18-5-FU is calculated from rat distribution studies and human urinary excretion data. These calculations assume a similar distribution of labeled drug in humans as in rats, inasmuch as our preliminary human distribution studies appear to validate the use of the rat model for human dosimetry calculations (5).

METHODS

Animal Distribution. Organ distribution of F-18, when administered as F-18-5-FU, was studied in six groups of 12 rats each bearing various types of solid tumors and in two groups of healthy rats, up to 2 hours after the drug had been administered. F-18-5-FU was injected at a dose of 10 μCi per animal, and each series of 12 rats was then sub-divided into three groups and sacrificed 30, 60 and 120 minutes after the injection. Blood was withdrawn by heart puncture, and 11 organs and tissues (liver, muscle, kidney, blood, bone, lungs, pancreas, spleen, heart, genitals, thyroid and adrenals) were sampled and counted in a gamma-counter. Details of these F-18 distributions were reported in a former publication (6). From those results initial organ uptakes as well as biological and effective half-lives were calculated.

Urinary Excretion. Urine samples were not usually collected when studying the body distribution of F-18-5-FU in rats. However, this route of excretion eliminates a major fraction of the injected radioactivity. Urine was collected from patients and dosimetry calcula-

tions were made based on the total cumulated activity in the urine of a patient with normal renal function. In this way, the worst possible dose from bladder content to other organs was calculated.

Radiation Dosimetry. These dosimetry calculations make the following assumptions: 1) That 5-FU clears the plasma rapidly (7), and hence rapid tissue uptake will follow. Some of this activity, following 5-FU metabolism, may leave the tissue, but some of the metabolites formed may be reabsorbed by them. Therefore, the assumption of an instantaneous uptake of the total administered dose will give an upper limit, a "worst possible case;" 2) That the extrapolated initial fraction values are correct for practical purposes, thus assuming a possible overestimation rather than underestimation of the dose; 3) That there is a homogeneous distribution of the label within each organ studied; 4) That the remainder of the activity, not found in the total activity recovered from the blood and the 11 organs sampled (48.06% recovered 30 min post injection, 41.6% after one hour, and 28.82% after two hours), was either excreted from the body or deposited in other organs in minute amounts; and, above all, 5) That the distribution of the drug in rats is similar enough to that in humans to allow such a comparison. The internal radiation doses were calculated according to the MIRD technique (8).

Human Distribution. The distribution of F-18 was studied in six patients possessing different types of neoplastic disease (e.g., colorectal, breast, liver). Each patient was administered intravenously a dose of 5 to 6 mCi of F-18-5-FU containing 30-70 mg of 5-FU. The distribution of the radioactive material was studied using an Ohio Nuclear Rectilinear Scanner equipped with a high energy 511 keV collimator. The patients were scanned immediately after injection and every two hours thereafter, up to 12 hours post injection. The total urine excreted was collected and counted, whenever the patient's condition allowed adequate collection.

RESULTS

The initial organ uptake of F-18 in rats at injection times, as well as the biological and effective half-lives of the label in the above organ was determined, and used to calculate the cumulative activity (A). Utilizing the tables of absorbed doses to the main organs per unit of cumulated F-18 activity (8) as they refer to the standard man, we calculated the radiation doses to the organs counted, according to the following equation:

$$D_T(\text{mRad}) = \Sigma_J S_{NT} \cdot A_J$$

where D_T is the dose to the target organ due to the presence of unit cumulated activity (1 millicurie-hour) in the jth source organ, and A_J is the cumulated activity (in millicuries) from the jth source organ. The sum of multiplications of the cumulative activity (in mCi-h/mCi administered) by the absorbed dose of the various organs (in Rad/μCi-h) will yield the dose to the target organ (mRad per each mCi F-18 administered) as tabulated in Table 1.

The urinary excretion of one patient with normal renal function gave a fractional excretion dose of 15.75% (0-1 hr), 18.9% (1-2 hr), 14.55% (2-4 hr), 17.17% (4-8 hr) and 7.05% (8-13 hr), from which the cumulated activity was calculated. Based on the total cumulated ac-

tivity of .413 mCi-h/mCi administered, the additional radiation dose to relevant organs due to the activity accumulated in the bladder is given in Table 2.

Table 1. Activities Cumulated in 12 Target Organs for ^{18}F and the Radiation Doses Absorbed in Each Case

Source Organ	Cumulated activity (mCi-h/mCi administered)	Dose/Target Organ (mRad/mCi administered)
Liver	2.3×10^{-1}	113.5
Muscle	1.8×10^{-1}	10.4
Kidney	7.9×10^{-2}	182.6
Blood	5.9×10^{-2}	11.0
Bone	8.9×10^{-2}	26.2
Lungs	6.3×10^{-3}	12.2
Pancreas	2.1×10^{-3}	29.1
Spleen	3.0×10^{-3}	21.6
Heart	2.2×10^{-3}	10.3
Genitals	2.8×10^{-3}	52.6
Thyroid	8.2×10^{-4}	28.0
Adrenals	4.2×10^{-4}	37.8
Total Body	-	14.7

Table 2. Radiation Dose to Several Abdominal Organs (as generated from bladder contents) Following Administration of F-18-5-FU

Target Organ (T)	$S_{T \leftarrow \text{Bladder contents}}$ Rad/mCi-h	Dose to target organ from bladder content mRad/mCi administered
Liver	2.1×10^{-3}	.87
Kidney	2.8×10^{-3}	1.16
Pancreas	1.9×10^{-3}	.78
Spleen	1.7×10^{-3}	.70
Ovary	4.5×10^{-2}	18.60
Uterus	1.1×10^{-1}	45.60
Testes	3.7×10^{-2}	15.20
Adrenals	1.7×10^{-3}	.70
Bladder Wall	1.8×10^{0}	729

DISCUSSION

The distribution of F-18-5-FU in humans appears to be similar to the drug distribution in rats, with the liver, the kidney, and the urinary bladder being the critical organs. While the uptake of radioactivity by the gallbladder as seen in patients may be important in determining the most effective dose of the drug (9), its importance in calculating the radiation dose appears to be negligible. Therefore, we believe that the premise that the drug distribution in the rat is similar to that of the human appears to be a valid assumption.

Although the dosimetry calculations are based on the biodistribution of the label in tumor-bearing rats, it was noted that the differences in tissue uptake and clearance of F-18 in the organs when comparing the tumor-bearing and control rats, were small, if any.

The average tumor uptake, seemingly independent of the tumor type used, was about 0.8% per gram at 2 hrs post injection, and the few small differences between retention of F-18 in organs of both groups studied were of no significant impact insofar as the present dosimetric calculations are concerned. For this reason we chose the distribution in rats rather than in mice for our dosimetry calculations, as the uptake in the rat tumors was considerably less than in mouse tumors (about 7% per gram) and, thus, more radioactive material is available for organ uptake. Thus, again, the dose calculated here is the "worst possible case."

Because 5-FU is rapidly metabolized in the liver followed by fast renal excretion (10), the highest doses are expected to be in the bladder wall, the liver and the kidney. This, in fact, appears to be the case. Our calculationss reveal that the kidney and liver receive a dose of 182.6 and 113.5 mRad/mCi administered, respectively.

Assuming that 20 mCi will probably be the maximum dose of F-18-5-FU to be administered, the absorbed dose to the bladder wall, kidney, liver, testes and ovaries is about 14.6, .37, .23, .14 and .14 Rads respectively. These values represent the upper limit and while they are high, they are still not unusual for nuclear medicine procedures. As these studies are intended to be performed in patients with a fully diagnosed tumor, and these results of the F-18 study are to be used to decide on the appropriateness of a therapeutic regimen, the risk/benefit considerations are clearly acceptable. The radiation absorbed by the body will thus not be a limiting factor in the use of F-18-5-FU for the prediction of chemotherapy response and optimization of the dosage regimen of 5-fluorouracil.

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THE CURRENT CONTRIBUTION OF DIAGNOSTIC RADIOLOGY TO THE POPULATION DOSE IN GREAT BRITAIN

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Twenty years have passed since the last national assessment of the radiological hazards to patients arising from medical practices in Great Britain (1). Only two local surveys of the contribution from diagnostic radiology have been carried out in the meantime, limited to the Sheffield region in 1964 (2) and the Yorkshire region in 1976 (3). Many other countries have conducted their own national surveys in recent years and some have managed to repeat them after a reasonable interval, so that trends in the pattern of use of medical radiology can be observed (4).

In view of the considerable changes in both the extent and techniques of medical radiology that have taken place in Great Britain over the past 20 years, the National Radiological Protection Board felt that a fresh look at the hazards to patients was long overdue. A reappraisal of the genetically significant dose (GSD) to the population of Great Britain from diagnostic radiology was recognised as one of a number of worthwhile objectives for the review. The absence of any direct benefit to the descendants of patients undergoing X-ray examinations argues for a close watch to be kept on the magnitude of the genetic risk, and diagnostic radiology was given priority because of the relatively large proportion of the fertile population who experience it in a year. Whilst an assessment of the somatic risks and the contributions from radiotherapy and nuclear medicine are currently underway, this paper concentrates on the survey that has recently been completed to establish a current value for the GSD to the population of Great Britain from diagnostic X-ray examinations.

SURVEY METHODS

Calculation of the GSD from diagnostic radiology requires knowledge of the number of people examined annually in Great Britain broken down by examination type, age and sex, the child expectancy of the population as a function of age and sex and the average gonadal dose delivered by each examination type as a function of age and sex.

Information on the frequency of X-ray examinations was

obtained by sending questionnaires to a sample of about 100 hospitals spread throughout England, Scotland and Wales. The questionnaires asked for details of every X-ray examination carried out in a specified week in June 1977. The selection of the sample was based on the division of all the hospitals in the country into 8 strata depending on the workload of their X-ray departments. The proportion of the total sample to be allocated to each stratum was derived using the Neyman method of allocation (5) which minimizes the errors when extrapolating to the whole population. Individual hospitals were selected with probability proportional to their X-ray workloads. By this means the sample of hospitals that responded to the questionnaires was sufficient to achieve a standard error of 5% in the estimated total of examinations for the whole country. Annual totals were estimated by multiplying the weekly figures by 365/7 and then correcting for seasonal variations in workload based on film consumption data for Great Britain in 1977.

The child expectancy of the population was obtained from the official statistics provided by the Office of Population Censuses and Surveys.

Gonadal doses were measured directly on patients undergoing 13 selected diagnostic examinations at 21 hospitals. Thermoluminescent dosemeters were used that consisted of lithium borate powder contained in adhesive polythene sachets. They had been developed at NRPB specifically for use in medical dosimetry (6). For male patients they were attached to the inside of the thigh close to the scrotum and were assumed to receive the same dose as the testes. For female patients they were positioned so as to measure the entrance skin dose level with the ovaries. Skin doses were converted to ovarian doses using conversion factors obtained by exposing an anthropomorphic phantom to a range of typical diagnostic X-ray fields and measuring the doses at the ovary sites and on the skin. The 13 examination types selected for inclusion in the dose measurement survey were those that in previous surveys accounted for about 95% of the GSD. Doses for all other examination types were obtained from the current literature. The 21 hospitals visited came from all regions of the country and included a representative sample of X-ray department size and speciality. At the end of the survey the gonadal doses received by 4565 patients had been measured.

RESULTS

The frequency survey indicated that 21.3 million X-ray examinations were carried out in National Health Service (NHS) hospitals in Great Britain in 1977. This represents an increase of 64% over the estimate of 13 million from the 1957 national survey (1). The number of examinations per thousand of the population has risen by only 48%, since the population has increased during the period. This corresponds to an increase of 2.0% per annum in close agreement with estimates for other industrialised countries (4). In addition it was estimated that 1.5 million chest X-rays and 1.0

million other examinations (excluding dental) took place in institutions outside the NHS. This brings the total number of examinations up to 23.8 million, corresponding to 440 examinations per thousand of the population, which as the following table shows, is rather lower than in other industrialised countries.

TABLE 1. Frequency of X-ray examinations per thousand head of population for industrialised countries

Country	Year	Examinations per thousand
West Germany	1974	1658
Switzerland	1971	1350
Netherlands	1972	1186
Japan	1974	810
USA	1970	669
Sweden	1974-1976	650
Great Britain (this survey)	1977	440

Comparative data are from reference (4)

The relative frequency of particular examination types demonstrated some predictable trends. For example, chest X-rays have fallen from 48% of all examinations down to one-third, presumably reflecting the progress which has been made in controlling tuberculosis. The number of obstetric examinations per thousand live births has fallen from 114 in 1957 to 42 in 1977, no doubt as a consequence of the increased concern for possible foetal damage and the extensive use of ultrasound as an alternative to X-rays. There has also been a drop in the number of cerebral angiograms as a result of the advent of computerised tomography.

The gonadal dose measurement survey indicated that for some types of examination there has been an increase and for others there has been a reduction in the gonadal exposures delivered per examination. For example, the introduction of double contrast techniques in barium examinations of the gastro-intestinal tract has led to a doubling in the number of films taken per examination which in the case of barium enemas has more than doubled the gonadal dose. On the other hand simpler examinations involving only a few radiographs appear generally to be accomplished with lower gonadal doses than 20 years ago which is probably due to the trend towards faster films and screens. However, if the ratios of current to 1957 gonadal doses for each examination type and sex are weighted according to their relative contribution to the GSD, then the average of the ratios turns out to be about 1.07. Therefore there has been no overall reduction in the gonadal doses delivered by those types of examination of importance to the GSD.

Similarly there has been no reduction in the very large range of gonadal doses delivered throughout the country for the same examination type. Individual gonadal doses for the same examination ranged over 3 or 4 orders of magnitude throughout the country. Mean gonadal doses obtained at different hospitals for the same examina-

tion were found to differ by up to a factor of 10.

Our provisional estimate for the current value of the GSD to the population of Great Britain from all diagnostic examinations conducted both within the NHS and elsewhere is 17 x 10^{-5}Gy (17 mrad). This represents an increase of only 20% over the value of 14.1 x 10^{-5}Gy (14.1 mrad) found in 1957. Considering the errors involved in both surveys this is probably not a significant difference. Table 3 shows how the current value for Great Britain compares with recent estimates of the GSD in other countries.

TABLE 3. The GSD from diagnostic radiology estimated for various countries in the period 1970-1977 (10^{-5}Gy = 1 mrad)

Country	Year	GSD (10^{-5}Gy)
Sweden	1974-76	46
West Germany	1974	41
Italy	1974	30
Romania	1970	29
Netherlands	1972	28
USA	1970	20
Great Britain	1977	17
Japan	1974	17
Taiwan	1972	3.5
India	1967-72	1.1

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DETECTION OF LOW-LEVEL ENVIRONMENTAL EXPOSURE RATES DUE TO NOBLE GAS RELEASES FROM THE MUEHLEBERG NUCLEAR POWER PLANT.

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To verify the adherence to the regulations on limits for the release of radioactive waste gases from nuclear power plants (NPP's), monitoring networks with solid state dosimeters, usually TLD, are prescribed in practically every country. Because of variations in the natural background radiation and the low value of emissions from modern NPP's, the determination of the net dose above background in most cases is very difficult. At best, only the upper limits for the additional doses can be indicated. For that reason, for several years now, high pressure ionization chambers (Reuter & Stokes RS-EI), which are capable of detecting dose rate increases in the range of natural background radiation as small as a few tenths of a µR/h, have been used for monitoring NPP emissions in Switzerland. The dose rate is continuously recorded on magnetic tape (digital record every 30 sec.) and on strip charts, and is continuously integrated by a counter. The correlation of these measurements with the emission and wind measurements makes it possible to calculate the dose amounts attributable to the exhaust plume.

This paper covers such measurements made at the Mühleberg NPP near Bern Switzerland (BWR, 320 MWe). For several years, the continuous releases of radioactive noble gases from the stack have been for the most part less than 100 µCi/s ^{133}Xe-equivalent. In addition, several times per year as a result of transient operations, short-time higher releases (spikes) occur that are 30 to 50 times higher than the continuous releases. The Mühleberg NPP is located in the gently curved Aare Valley with 2 distinct prevailing wind directions. The nearest house, where the calculational model indicates the maximum doses would be expected, lies about 500 m west of the plant on the hill "Ufem Horn". This place was selected as the site for installation of an Ionization Chamber.

Because of the hilly terrain such as that at Mühleberg, the dispersion models have only limited validity. Consequently, by evaluation of the dose rate peaks, and correlating them with the stack release measurements, a determination was made of the short-time dispersion factors for Gamma-Submersion. Figure 1 (16 Jan.1978) and Figure 2 (22 April 1976) show the stack and dose rate records for such higher emissions. For 24 such spikes that occurred during the last two years, the integrals of the dose rate increases at the measuring point and the emission spikes from the stack (^{133}Xe-equivalent) were calculated, and hence, by using the dose factor for ^{133}Xe

(0.011 mrem/Ci per s/m^2), the short-time dispersion factor was calculated. The correlation line so obtained (correlation coefficient 0.76) is shown in Figure 3. The mean value of the short-time dispersion factor was found to be $(5^{+}3)\cdot10^{-3}s/m^2$. From weather statistics, one can determine a relationship of long-time to short-time dispersion factors to be approximately 0.1. The calculated value of the dispersion factor is thus in agreement with that of G. Schriber (1) for the long-time dispersion factor determined for the same place by other methods.

The determination of the net dose contribution from the continuous stack releases is very difficult because of the meteorologically dependent variations of the natural background radiation at a given place, and because of the very small releases. Thus, only the higher limits of this dose contribution can be determined. The dose rate records from magnetic tape are evaluated using a desk calculator to determine hourly average values. In addition, consideration is given to whether the wind at the stack is blowing in the direction toward the measuring point, or in another direction. In this way, one obtains the average hourly values of the dose rate when the wind is toward the measuring point and when the wind is in other directions. The difference between these two curves corresponds then to the net contribution from the plume. Figure 4a shows an example of such an evaluation for the period from 16 March 1977 to 26 April 1977. The averaged value of the dose rate is given for the wind in the direction toward the measuring point (dotted line) and for other wind directions (solid line). Figure 4b shows the difference between the two curves. For this example, the calculated average net dose rate is 0.05 ± 0.07 µR/h or ≤ 0.12 µR/h, which corresponds to the higher limit of the additional yearly dose contribution from the NPP, and amounts to less than 1.1 mR/y above background radiation.

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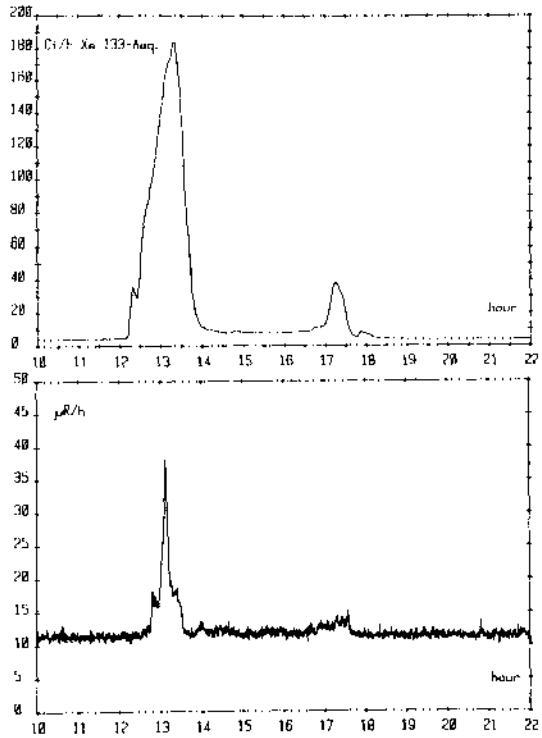

Fig 1

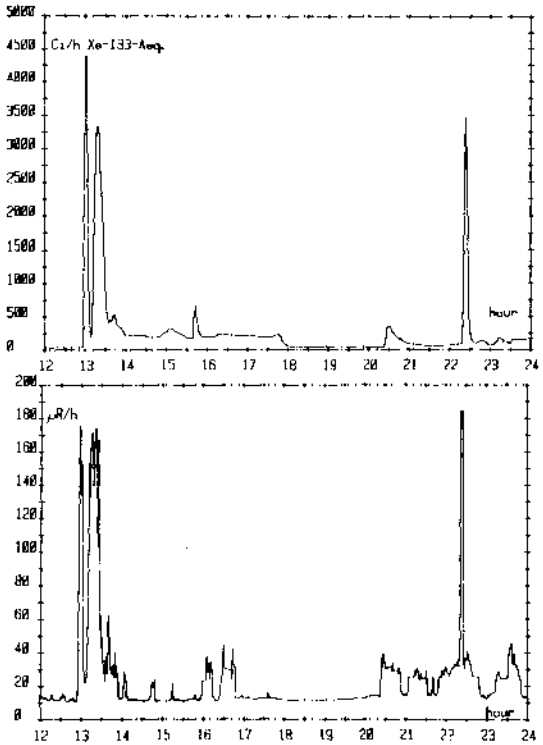

Fig 2

Time correlated Records of Gaseous Effluents Spikes at the Stack (Ci/h, $^{133}Xe_{eq}$ equivalent) and Dose Rate Spikes (μR/h) 500 m West from the NPP on the 16 Jan 1979 (Fig 1) and on 22 April 1976 (Fig 2)

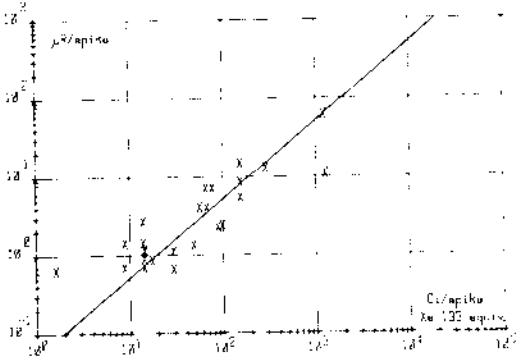

Fig 3 Correlation between Dose Spikes (µR/Spike), measured 500 m West from the NPP, and Stack Release Spikes (Ci/Spike, ^{133}Xe-equivalent)

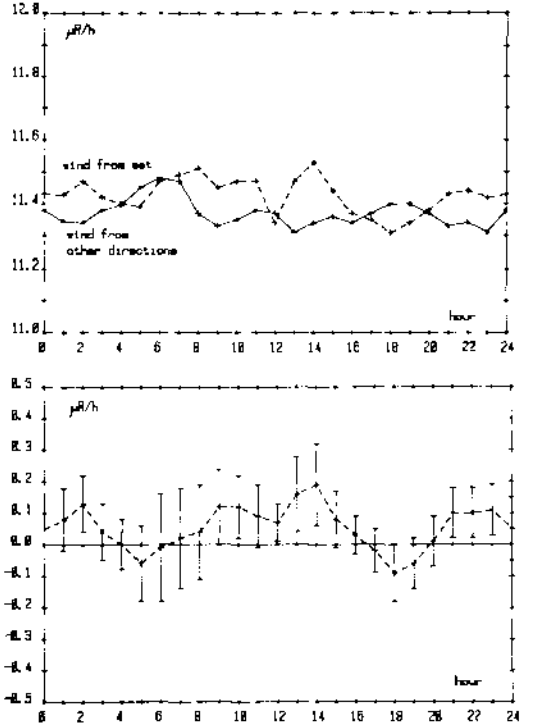

Fig 4a Mean hourly values of the Dose Rate (µR/h) at a Point 500 m West from the NPP with an East Wind (---), and with other Wind Directions (———) calculated for the Time Span: 16 March to 22 April 1977

Fig 4b Calculated Net Dose Rate for the same Time Span

RETENTION OF ^{14}C IN NUCLEAR POWER PLANTS AND REPROCESSING PLANTS

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INTRODUCTION

Up to about 1972 it was unknown that the emissions of ^{14}C from nuclear power plants and reprocessing plants represent significant contributions to radiation exposure. Calculations, however, have shown that ^{14}C may give the highest contributions to the individual dose (1),(2). Following measurements confirmed the emission rate (3),(4). The collective dose commitment, however, is of greatest importance for ^{14}C. Because of the long half life of 5730 a it is very high, too (5).

PRODUCTION RATES OF ^{14}C

In reactors, ^{14}C is mainly produced due to neutron reactions from $^{13}C(n,)^{14}C$, $^{14}N6n,p)$ and $^{17}O(n,)$. The ^{14}C produced in the fuel is not emitted until dissolution, while ^{14}C produced in the coolant enters the air-circulating plant via leakages and is emitted from there or is emitted via the off-gas-system. Tab. 1 shows the production rates in fuel and coolant of all common reactor types. Details of the calculation are represented in (6).

^{14}C RETENTION FACILITY

^{14}C is emitted as gaseous compound, mainly as $^{14}CO_2$, but also as ^{14}CO and $^{14}C_nH_m$ together with a considerably higher part of ^{12}C compounds. If significant amounts of other radio carbon compounds exist beside $^{14}CO_2$, it is sensible, to oxidize these to $^{14}CO_2$ before the retention. Then the "direct" or the "double" alcali process can be used to retain ^{14}C.

Both processes have two considerable advantages in comparison with other processes:

1. The absorption of the CO_2 is supported by a chemical reaction, whereby high decontamination factors are reached.
2. ^{14}C becomes available as calciumcarbonate, a solid stable compound that can be concreted.

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Up to now, mainly the direct alcali process has been examined, because it seemes technically more simple to realize and more economical (7),(8), although the chemical reactions of the double alcali process and its control action concerning changes in the mass flows are more favorable. This is important, because the ^{14}C release takes place instationarily from nearly all nuclear facilities; both the volume flow and the concentration of the carbon compounds often vary.

Therefore, a ^{14}C retention facility with the double alcali process has been developed at the professorial chair for reactor technology at Aachen Technical University. Concerning the process scheme, the retention facility has been constructed as simple as the direct alcali process, but it has all the advantages of the double alcali process. Fig. 1 shows the flow sheet.

From a reservoir, sodium hydroxid is pumped through a packed column where inert gas containing CO_2 flows in inverse current. When the alkaline solution is converted to 60 %, the total alcaline solution inventory is pumped into the agitator for regeneration with $Ca(OH)_2$. At the bottom of the agitator $CaCO_3$ is bottlinged as a viscous slurry with a high percentage of liquid. The high percentage of liquid, approx. 50 % wt, is used for concreting. Fig. 2 shows the facility, the agitator has not yet been finished. The facility is used to obtain operation experiences with the new process and to optimize the process parameters. The most important construction data are:

max. gas mass flow : 40 kg/h
max. liquid flow : 500 kg/h
operation pressure : 1.5 ... 3 bar.

The actual operation experiences are well. A ^{14}C retention facility with this process scheme could be installed in LWR's and reprocessing plants for LWR fuel elements.

COST-BENEFIT ANALYSIS OF THE ^{14}C RETENTION FACILITY

If the cost-benefit analysis recommended in ICRP publication 26 is used, see Fig. 3, it is obvious that a ^{14}C retention facility with oxidation-process should be installed in reprocessing plants for LWR fuel elements (5). An installation in LWRs doesn't seem sensible to us because of the low ^{14}C emission rates. For reprocessing plants for HTR fuel elements with burn-leach-head-end the proposed method cannot be used because the waste volumes become too large. In this case the head-end should be changed in order to solve the ^{14}C problems. In big reprocessing plants for nitridic fuel elements of the fast breeder such a high amount of ^{14}C is set free (10^6 Ci/a) that in spite of high decontamination factors so much ^{14}C would be released that this fuel cycle should not be continued any further.

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Tab. 1: Production of ^{14}C in different reactor types in Ci/GW(e)a

Specification			^{14}C-Production in Ci/GW(e)a						
			BWR	PWR	HWR	MAGNOX	AGR	HTR	FBR
Outer surface of pressure vessel		^{14}N	$5 \cdot 10^{-4}$	0,005	0,005	$5 \cdot 10^{-4}$	0,001	0,005	[illegible]
Coolant		^{13}C	$7 \cdot 10^{-7}$	$8 \cdot 10^{-7}$	$3 \cdot 10^{-5}$	0,06	0,06	$4 \cdot 10^{-7}$	$5 \cdot 10^{-7}$
		^{14}N	0,6	0,8	25	7,3	7,1	0,02	0,02
		^{17}O	5,1	6,2	175	1,1	1	$5 \cdot 10^{-5}$	$2 \cdot 10^{-5}$
Fuel element	Fuel	Fission	0,6	0,6	0,6	0,6	0,5	0,5	0,5
		^{13}C	$2 \cdot 10^{-4}$	$2 \cdot 10^{-4}$	$7 \cdot 10^{-4}$	$8 \cdot 10^{-4}$	$2 \cdot 10^{-4}$	$9 \cdot 10^{-5}$	$\cdot 10^{-5}$
		^{14}N	7,6	7,8	26	130	13	3,1	2
		^{17}O	4,4	4,5	13	0,01	3,3	1,6	3
	Canning	^{13}C	$3 \cdot 10^{-4}$	$5 \cdot 10^{-4}$	$7 \cdot 10^{-4}$	$4 \cdot 10^{-4}$	$6 \cdot 10^{-4}$	—	$1 \cdot 10^{-5}$
		^{14}N	17	20	34	35	32	—	8
		^{17}O	0,015	0,02	0,03	0,003	0,003	—	$2 \cdot 10^{-4}$
Graphite moderator		^{13}C	—	—	—	110	35	32	—
		^{14}N	—	—	—	180	59	54	—
		^{17}O	—	—	—	0,02	$7 \cdot 10^{-4}$	$7 \cdot 10^{-4}$	—
Total production rate			34	40	274	500	85	91	15

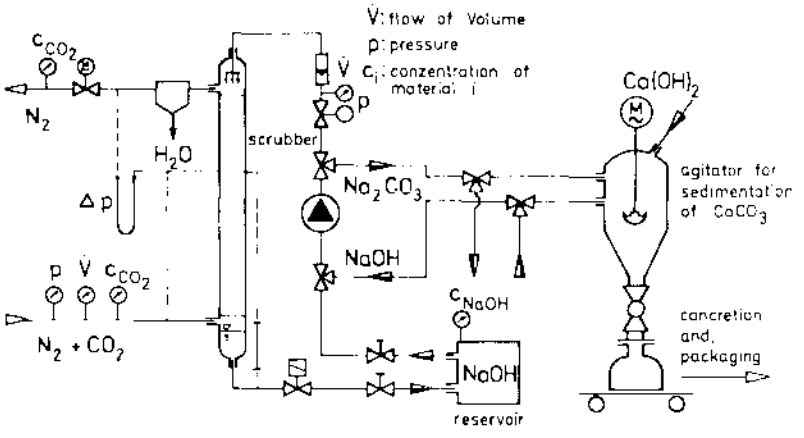

Fig.1: Flow sheet of the double alkali process

Fig.2: Experimental facility for the retention of ^{14}C

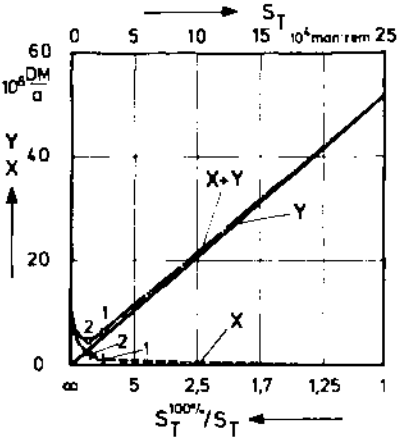

Fig.3: Cost-benefit-analysis of ^{14}C-retention in processing plants with a capacity of 1500 t/a
1: $^{14}CO_2$ only
2: $^{14}CO_2$ + ^{14}CO + $^{14}C_nH_m$

STATISTICAL CORRELATION OF ENVIRONMENTAL TRITIUM VALUES AT TROMBAY

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In the immediate neighborhood of a nuclear reactor releasing tritium into the environment, tritium monitoring is carried out as part of environmental surveillance in many nuclear centres in the world. The estimated doses from tritium releases have been reported by many workers (1-3). In almost all the cases the data is represented in a conventional way of diurnal, monthly or seasonal variations of discrete values.

As part of the present work tritium releases from a 40 MW(th) D_2O moderated reactor (Cirus) at Trombay and the environmental tritium concentrations (ambient air and vegetation samples) at different ground stations have been measured over a period of five years and the statistical correlation among the different sets of values were assessed. Sampling stations were chosen in the proximity of the reactor primarily with a view to obtain positive measurable values to yield statistically significant data base, for a meaningful interpretation. The three ground stations chosen were situated at 1500' NNE, 2500' NNE and 3300' SSW. Airsamples were collected using the 'coldstrip' method and vegetation samples (leaves of mango-Mangifera Indica,L.), processed by vacuum freeze drying technique. Tritium measurements were carried out with a Liquid Scintillation Spectrometer (Packard-Model 3255) with a detection limit of 0.8 $\pm$ 0.13 pCi/ml. for aqueous phase.

EXPERIMENTAL RESULTS AND DISCUSSION

The variation in the values of daily releases from Cirus reactor was quite low (of the order of a few tenths of a pCi/ml). The ambient air concentrations at the Ground Stations were usually of the order of a few nCi/m^3 with considerable variations around the mean. Similarly the aqueous samples extracted from the vegetation samples were of the order of a few pCi/ml and showed large variations around the mean. Normally these values are represented as discrete diurnal variations or in histograms indicating monthly or seasonal variations. Most of the earlier workers' attempt didnot extend beyond the determination of mean values of such data for different applications such as dilution rate factor determination etc.

The yearly cumulative frequency distribution of different sets of measured values were drawn on the basis of the method suggested by Seigmund Brandt (4). In all the three sets of distributions representing release values, ambient air concentrations and the vegetation samples similar patterns, having positively skewed normal distribution, were observed. These were caused by the occassional larger release concentrations of tritium which get reflected in the ambient air and

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the water compartment of the vegetation samples. Thus the median values are effectively shifted towards higher values.

The median values corresponding to the 50% fractiles, giving central values insize in each set of observations, as seen in a fractional frequency distribution pattern, differed widely from the mean values. (See Table I). The most probable values for a typical year are also given paranthesis in Table I for the sake of comparison.

TABLE I Average Tritium Concentration Values

Year	Tritium Concentration Values					
	at release point		in ambient air at Station A*		in vegetation samples at Station A*	
	Median	Mean	Median	Mean	Median	Mean
	pCi/cc	pCi/cc	nCi/m^3	nCi/m^3	pCi/ml	pCi/ml
1975	1.23	0.76	3.20	2.80	-	-
1976	0.90	0.60	6.10	3.90	334	330
1977	0.74	0.39	3.96	2.80	339	390
	(0.74)			(1.00)		(350)

* 1500' NNE

The cumulative probability curves for the different sets of values obtained at the different sampling stations also indicated very clearly that all the three distributions corresponding to the release concentrations of tritium, ambient air concentrations and the vegetation sample values showed positively skewed normal distribution patterns.

Inorder to test statistical correlation among the tritium concentration values at the release point and at the ground stations, the sample correlation coefficient, r, was calculated using the expression

$$r = \Sigma x_i y_i \Big/ \sqrt{\Sigma x_i^2 \cdot \Sigma y_i^2} \qquad \ldots\ldots\ldots(1)$$

where x_i and y_i were the observed values for the bivariates taken up for analysis. In case of different sets of observations at different stations, interrelationship among the different bivariates can be obtained as a correlation matrix $\| r_{ij} \|$. The sample correlation coefficients calculated for a 6x6 matrix was corrected for skewness using the expression

$$z = \frac{1}{2} \left\{ \ln (1+r) - \ln (1-r) \right\} \qquad \ldots\ldots\ldots(2)$$

The population correlation coefficient, ρ , as calculated on the basis of the expression

$$\rho = \Sigma x_i y_i - n \cdot \bar{x} \bar{y} \Big/ n \cdot \sigma_x \cdot \sigma_y \qquad \ldots\ldots\ldots(3)$$

were found to be nonzero, and indicated good correlation among the sets of values. Table II gives a set of typical values for three bivariates viz. tritium concentration values at the release point, ambient air concentrations and the condensates (aqueous) and the vegetation sample values. It can be seen that there is excellent correlation among the sets of values (better than 95%) both for sample correlation as well as for population correlation.

TABLE II Typical set of Correlation Coefficients Obtained

Sample Variables		Degrees of freedom for	Values of	Corrected values	% of significance	Degrees of freedom for	% of significance
x_i	y_i	r	r	z		ρ	
X2	X4	30	0.678	0.829	99	80	99
X2 (Aq)	X4	30	0.371	0.388	95	80	99
X1	X2	30	0.538	0.604	99	80	99

X1 : Tritium Conc. at Release Point (pCi/cc)
X2 and X2 (Aq) : Ambient Air concentrations at Station A in gaseous (vapour) and liquid phase. (nCi/m^3 and pCi/ml respectively)
X4 : Vegetation Sample Values at Station A (pCi/ml)

In order to test the interrelationships among the vegetation samples at different ground stations to those of air concentrations, the correlation matrix (5x5), corrected, were found to be well within 99% statistical significance. Thus the correlation among the ambient air concentrations and the water compartments of the vegetation was found to be statistically significant based on a set of environmental samples collected over a long period of time. It is emphasised that the above relationships were among the distributions themselves rather than among the discrete sample values.

The scatter diagram drawn for a set of bivariates has shown that for a fixed value of one variable there were a whole lot of population of values for the other variable indicating the influence of the environmental parameters on the sets of data. The propensity of the sample correlation coefficient, r, is normally reflected by the points which lie in a band in the scatter diagram, often shaped like an ellipse with its major axis sloping towards the appropriate direction depending on the polarity of 'r'. In the present case, where $r \neq \pm 1$, a pair of stright lines indicating the scatter of values around a mean value was obtained.

Thus in all cases it was clearly established that the statistical significance among the environmental tritium values, when taken up as a whole for a long period of time, was found to be excellent.

Utlising the values obtained from the above sets of values, the mean, median and the most probable values were obtained. The dilution rate factor, K ($m^3 . sec^{-1}$), was calculated for the Trombay environs using all the above values to study whether there are variations in the dilution rate factors due to the skewed nature of distribution of tritium in the environment. Table III summarises the values thus

obtained for three years. It is significant that the values obtained on the basis of the present study shows a lower value than the earlier results reported (5).

TABLE III Dilution Rate Factors (K)

Year	Based on Mean Values	Based on Median Values	Earlier Published Data(5)
1975	$0.13-0.38 \times 10^4$	0.28×10^4	$1.0 - 2.8 \times 10^4$
1976	0.10×10^4	0.23×10^4	
1977	0.10×10^4	0.29×10^4	
		(0.19×10^4)	

Value in paranthesis is based on most probable value.

CONCLUSION

The distribution patterns of environmental tritium in and around a reactor site on the basis of a set of data generated over a period of five years have shown that all the distributions are gaussian in nature with identifiable skewness, caused by occasional larger release. The effect of the mean, median and the most probable values on dilution rate factors were found to be significant. The correlation among the sets of data showing tritium concentrations in release air, the ambient air and the vegetation samples at different stations were found to be statistically good, and very nearly perfect.

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AN AIR MONITORING PROGRAMME IN THE ENVIRONMENT OF A MAJOR NUCLEAR ESTABLISHMENT: OPERATION AND RESULTS

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During the Windscale Inquiry in 1977 (1), the Board was asked by the Inquiry Inspector to measure the levels of airborne alpha activity in Ravenglass, a village on the coast approximately 10 km south of the British Nuclear Fuels Ltd., Windscale works. (Windscale is a fuel reprocessing plant). During a four week period in September 1977, the mean level of 239 + 240Pu was 5.2 μBq m^{-3} and that of ^{241}Am was 4 μBq m^{-3}. These levels were clearly elevated above average weapons fallout values (0.8 and 0.1 μBq m^{-3} respectively) but they were well below the maximum permissible concentrations in air for members of the public prevailing at that time (2200 and 7400 μBq m^{-3}) (2). In presenting its results at the Inquiry however the Board commented that the sampling was not sufficiently prolonged to enable annual average doses to be assessed with confidence. Furthermore, the levels might be higher at other locations.

The Board decided therefore to mount a programme of measurements in West Cumbria with the principal objective of assessing the exposure of the local communities to airborne radioactivity over a sufficiently long time to be confident that the results were indeed representative; a period of one year was deemed sufficient. It was also hoped to obtain some information about the immediate source of the airborne activity as well as the effect of environmental conditions, such as the weather, on the levels.

The programme of sampling ran from the summer of 1978 to the summer of 1979.

SELECTION OF SITES

A fully comprehensive sampling programme would have utilised a dense array of samplers around the Windscale works. However, because of financial limitations, the problems of analysing large numbers of air filters, and the difficulty of finding suitable sites, a smaller programme with five sites was mounted.

Sites were required which were reasonably secure and which had supplies of mains electricity. The air samplers were to be located remote from buildings, firstly to ensure representative environmental sampling and secondly because of the noise levels from the samplers. Five sites which met these requirements were located in the areas of St. Bees, 10 km to the north; Egremont, 7 km north-east of the works; Holmrook, 6 km to the south-east; Seascale, 3 km to the south and Eskmeals, 13 km south of the works. The sites were chosen to coincide with local centres of population, (St. Bees, Egremont, Seascale), as well as covering inland (Egremont, Holmrook) and coastal (St. Bees, Seascale, Eskmeals) locations. The Eskmeals site was a very open one approximately 100 m from the beach.

EQUIPMENT

The air monitoring equipment in Cumbria consisted of 6 high

volume air samplers, one of which was controlled by a switch sensitive to wind direction, and a cascade impactor. A high volume sampler was installed at each site; the wind direction sensitive sampler and cascade impactor were installed for 9 months at Eskmeals and then 3 months at Seascale.

Each high volume air sampler consisted of an upward facing filter holder at a height of 1.5 m from ground, coupled to a high velocity centrifugal fan (Secomak 575/1) and a rotary inferential gas flow meter (PCC). Polystyrene fibre filters (Microsorban 98) size 260 mm x 210 mm were used. Each sampler was housed in a wooden shelter approximately 1.3 m x 1.3 m x 2 m high with a sloping inlet on each side. The inlets measured 200 mm x 200 mm and were covered by a metal screen (6 mm mesh with 1.5 mm ribs) to minimise vandalism. The air from each sampler was exhausted at roof level, a deflector plate directing the air flow up and away from the inlets. The air flow rate using the above configuration was greater than 1 m^3/minute. The shelters were designed to provide good security and weather protection, while allowing radily respirable particles (i.e. particles up to about 10 μm AMAD) to be sampled adequately. However, larger particles are also important since the effective dose equivalent per unit intake for a $^{239+240}Pu$ class W aerosol remains fairly constant up to approximately 100 μm AMAD (3). The apparatus described above will sample some particles of this size but with a reduced and unknown efficiency. It is believed however that the aerosol in the shelter is likely to be less depleted than it would be inside an ordinary house where windows and doors are normally closed. Consequently an overestimate is likely to be made of the concentration to which most people are exposed because of the high indoor occupancy factors that prevail. The concentration of larger particles out of doors is however likely to be under-estimated. A subsidiary experiment is being carried out to determine the overall degree of depletion of the aerosol caused by the shelters.

As noted above, one of the samplers at Eskmeals was controlled by a switch sensitive to wind direction. A wind vane on a 4 m high mast activated a switch with a time constant of 120 sec when the wind was coming from the sea within an angle which was adjusted to $\pm 67\frac{1}{2}^{0}$.

The cascade impactor (BGI-30 with model 1A sampler) was kept in its normal aluminium shelter with the top removed. This was housed in one of the wooden shelters described above with the first stage at a height of 1.5 m. The impactor is a single slit device with four impactor stages and a final filter. Glass-fibre filter papers, moistened with olive oil, were used for the impactor stages with Microsorban as the final filter. The flow rate was almost 1 m^3/minute and was maintained constant by a pressure transducer in a feed back loop. The impactor has an effective cut off diameter of about 10 μm for the first stage and about 1 μm for the final stage.

During the one year operating period filters were changed weekly. The centrifugal fans used with the high volume samplers proved to be very reliable. The cascade impactor motor was not however so suitable for continuous running. Several replacement motors had to be used and brushes had to be changed frequently. Trouble was also experienced with the air flow rate meters which were repaired frequently.

The sites were not very accessible to the public and vandalism was not a problem. At one of the sites, nevertheless, the noise levels were unacceptable to local residents. The sampling programme ran for a year with no major problems.

ANALYSIS OF FILTERS

Each Microsorban filter was pressed into a shape 80 mm dia x 3 mm thick prior to γ counting with a lead shielded 105 cc Ge (Li) detector or a 75 cc Ge (Li) detector with an anticoincidence shield. Both detectors were housed in a steel room with 150 mm thick walls. Each Microsorban filter and each glass-fibre cascade impactor filter was counted for 1000 minutes and the activities of the following γ emitting nuclides was determined:
^{54}Mn, ^{95}Zr, ^{103}Ru, ^{106}Ru, ^{125}Sb, ^{131}I, ^{137}Cs, ^{141}Ce and ^{144}Ce

After γ counting, the ordinary high volume sampler filters were ashed individually. Cascade impactor filters covering a 2 week sampling period were combined before ashing. The ashed samples were totally dissolved in hydrofluoric acid and the plutonium and americium isotopes separated out using ion exchange resins. The samples were electroplated onto stainless steel discs and the $^{239+240}Pu$, ^{238}Pu and ^{241}Am activities were determined by α counting using surface barrier detectors under vacuum. Yields were monitored using suitable tracers.

RESULTS AND DISCUSSION

It is possible to present only a selection of early results in this paper: a full report is being prepared.

TABLE 1. Gross activity concentrations in West Cumbria, June to December 1978, μBq m^{-3}

	Nuclide	Seascale	Holmrook	Egremont	St.Bees	Eskmeals	Fall-out
Actinides	$^{239+240}Pu$	7.1	1.8	1.5	3.8	6.1	0.1
	^{238}Pu	1.5	0.4	0.4	0.7	1.1	0.02
	^{241}Am	3.3	0.7	0.6	2.1	3.7	0.05
γ-emitters	^{144}Ce	210	330	360	360	230	190
	^{106}Ru	610	230	280	280	320	160
	^{137}Cs	990	500	420	260	230	70
Distance from Windscale works km		3	6	7	10	13	-

The numbers in Table 1 are the means of the weekly values. The concentrations varied widely from week to week, however. For example, at Seascale the $^{239+240}Pu$ values ranged from 0.7 to 20 μBq m^{-3} and the ^{137}Cs values ranged from 30 to 4,100 μBq m^{-3}.

Considering first the actinides, the results for the 3 coastal sites (Seascale, St. Bees, Egremont) are similar to those found at Ravenglass a year earlier. All results are clearly elevated above

average fallout values. The derived air concentration (DAC) for members of the public is assumed to be 2500 μBq m^{-3}, i.e. one tenth of the value in ICRP-30, for each of the actinides in lung class W. (4). The actinide concentrations are clearly well within appropriate limits even when considered jointly.

Results for the γ emitters are also elevated above average fallout values, ^{144}Ce marginally so, ^{106}Ru more noticeably and ^{137}Cs rather markedly. However, the most restrictive DAC for both ^{144}Ce and ^{106}Ru is 5 x 10^{6} μBq m^{-3} for lung class Y and the DAC for all compounds of ^{137}Cs about 5.8 x 10^{7} μBq m^{-3}. The levels in Cumbria are four to five orders of magnitude below these DACs.

From Table 1, it can also be seen that the coastal sites (Seascale, Eskmeals, St. Bees) have the highest actinide results. The concentrations of γ emitting radionuclides however fall off with distance from the Windscale works, clearly so where there is a strong signal-to-noise ratio as in the case of ^{137}Cs.

Complete analysis of meteorological and other data is proceeding but partial analysis suggests that at coastal sites the concentration of actinides is dependent on wind persistence and to a lesser extent on wind speed. Analysis of the results from the directional and continuous samplers at Eskmeals shows that actinides concentrations are higher when the wind is off the sea, but that the same is not true for the γ emitting nuclides.

Because of certain metrological constraints, the results from the cascade impactor at Eskmeals are not so accurate as the others. However, some indication of the particle sizes can be given. Approximate AMADs in μm are as follows $^{239+240}$Pu 15, ^{238}Pu 15, ^{241}Am 10, ^{144}Ce 1, ^{106}Ru 2, ^{137}Cs 3. Fallout of course has a particle size less than 1 μm. The higher AMADs for the actinides probably points to different immediate sources of actinides and γ-emitters. The results are consistent with the hypothesis that the major source of actinides is the sea; but the situation for γ-emitting nuclides is more complex.

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SIX-YEAR EXPERIENCES IN THE ENVIRONMENTAL RADIOACTIVITY MONITORING ON TAIWAN

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An islandwide network for environmental radioactivity monitoring established in 1974 is described with reference to (1) radioactive fallout measurement from nuclear testings, (2) environmental monitoring for nuclear power plants and radioactive waste disposal site, (3) specially designated site by the government, and (4) instalments involving radiation. Both naturally occurring and artificial radionuclides are equally emphasized. All significant results are reported and discussed.

Observation of fallout from nuclear testings mainly at Lop Nor and an islandwide network (Fig.1) for radioactivity monitoring established in the Taiwan province of the Republic of China in 1974 to provide radiation baseline data for the governmental agencies were described with reference to types of environmental samples taken and types of radioactivity monitored. The Taiwan Radiation Monitoring Stating (TRMS) of the Atomic Energy Council of Executive Yuan has a primary responsibility to establish the radiation baseline data. The TRMS obtains environmental sampled from its own islandwide networks and involves the analysis for samples of air, water, soil, vegetation, food, and direct gamma-ray using thermoluminescent dosimeters (TLD).

1. Direct gamma-ray

The TLD's are $CaSO_4$:Dy in powder from. They are installed around the nuclear power plants No. 1 and 2. The calibration procedures can be seen elsewhere (1,2). The results are presented in Table 1.

2. Airborne radioactivity

The airborne radio activities are classified as gross beta activity in surface air by gummed paper (Fig. 2), for Kaohsiung City alone (Fig. 3),

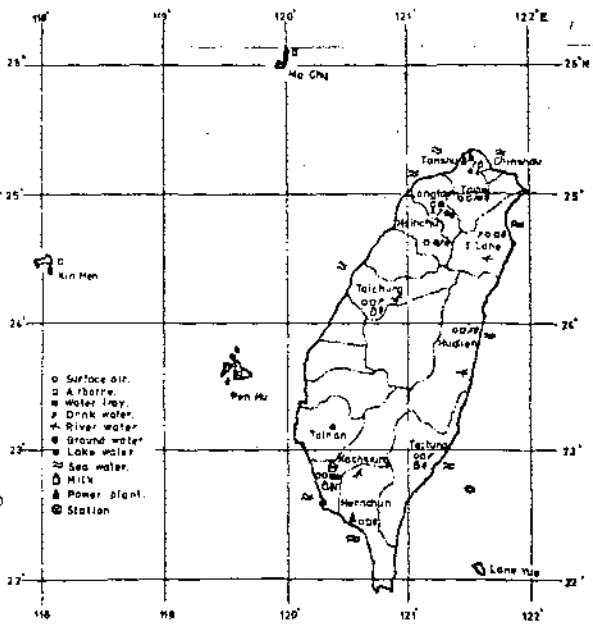

Figure 1. The map of sampling Salion.

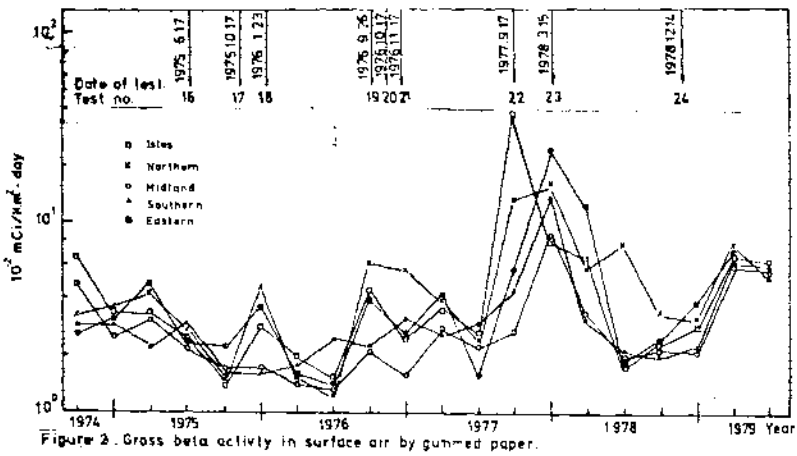

Figure 2. Gross beta activity in surface air by gummed paper.

and gamma-ray spectrum analysis (Fig. 4). Tritium in air is also collected by ethylene glycol with an efficiency close to 95% (3).

TABLE 1. Results of background gamma monitoring from 1977 to 1979

uR/hr

Exposure Period	Exposure Area					
	Towns and Cities		No.1 Power Plant		No.2 Power Plant	
	Range	Mean	Range	Mean	Range	Mean
1977	4.5-10	8.2	7.3-16.5	10.8		
1978	4.6-10.4	7.5	5.8-10.2 *5.7-10.4	7.8 7.7	4.3-10.4	7.1
1979	3.0-12.1	7.9	*6.7-10.8	8.4	3.2-10.3	6.5

*Commercial operation in October 1, 1978

3. Water sample

Various kinds of water samples are generally analyzed for gross beta activity, and sometimes for ^{90}Sr and ^{137}Cs as shown in Table 2. The $^{141/144}Ce$ concentration in water samples after nuclear testings are shown in Table 3. The activation products from nuclear power plant No. 1 such as ^{Mn}, ^{59}Fe, ^{57}Co, ^{60}Co, ^{64}Cu, ^{65}Zn have not been detected so far. Tritium in drinking water and rainwater varied from 0.88 to 3.2 pCi/ml with a lower limit of detection about 300 pCi/liter(4).

TABLE 2. Concentration of $^{141/144}Ce$ in Water Samples

pCi/liter

Samples	Dec 1976		Mar 1977	
Drink Water	96.5	94.5	52.8	50.2
Ground Water	63.8	54.8	57.5	55.4
River Water	122.6	61.8	B.K.	
Lake Water	77.1	52.3	B.K.	
Sea Water	112.4	44.9	B.K.	

B.K.:Background

4. Soil

Both surface (0~5 cm) and deep (10~25cm)soil samples are analyzed for gross beta activity (9~36 pCi/g), and for ^{137}Cs (1,218 pCi/kg) and ^{90}Sr (37.7~570.8 pCi/kg).

5. Vegetation

The gross beta activity of vegetation mainly comes from ^{40}K. The fission products such as ^{95}Nb-^{95}Zr, ^{144}Ce, ^{103}Ru,etc. were also detected after nuclear testings at Lop Nor.

6. Food

Ten kinds of major food such as rice, pork, fish, egg, powdered milk, potatoes, chicken, duck, flour, and vegetables in Taiwan were analyzed for ^{90}Sr and ^{137}Cs(5,6). The results are listed in Table 4. The tea leaves from nuclear power plant No. 1 were also analyzed for ^{90}Sr and ^{137}Cs.

All data reported in this paper are in good agreement with previous investigations. The estimation of the population dose based on present data is insignificant.

Several new experimental techniques have been developed. For example, the TLD's for environmental monitoring which have been proved to be quite efficient after participating the international intercomparison program, the ion-exchange method for water sample treatment, the chemical analysis of soil samples, wet-ashing method for food samples, and the chemical procedures for activation product analysis in the effluents from the nuclear power plant. Relatively high activity from nuclear testing at Lop Nor during 1974-1979 is indicated with corresponding dates and sequence number of tests in the figures concerned.

TABLE 3. Radioactivity of Various Water Samples during 1974-1979

pCi/liter

Sample	Gross beta			^{90}Sr			^{137}Cs		
	Min.	Max	Mean	Min	Max	Mean	Min	Max	Mean
Drink Water	0.9	2.4	1.7	0.06	0.95	0.4	0.14	0.34	0.3
Ground Water	0.5	3.6	1.6	0.13	1.27	0.6	0.13	0.38	0.3
River Water	1.1	7.8	1.6	0.08	0.48	0.5	0.12	0.44	0.3
Lake Water	1.0	2.5	1.7	0.13	1.88	0.6	0.14	0.6	0.4
Sea Water	1.0	5.9	2.0	0.18	0.91	0.5	0.53	0.6	0.6

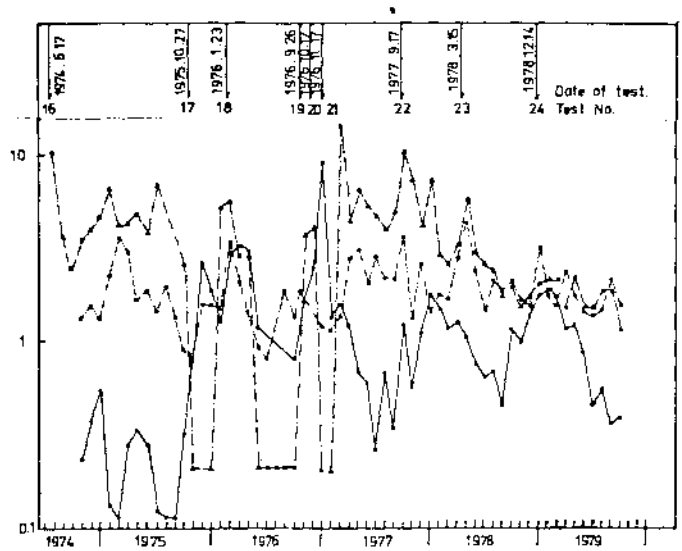

Figure 3. Monthly average beta activity of daily airborne particulates samples in Kaohsiung.

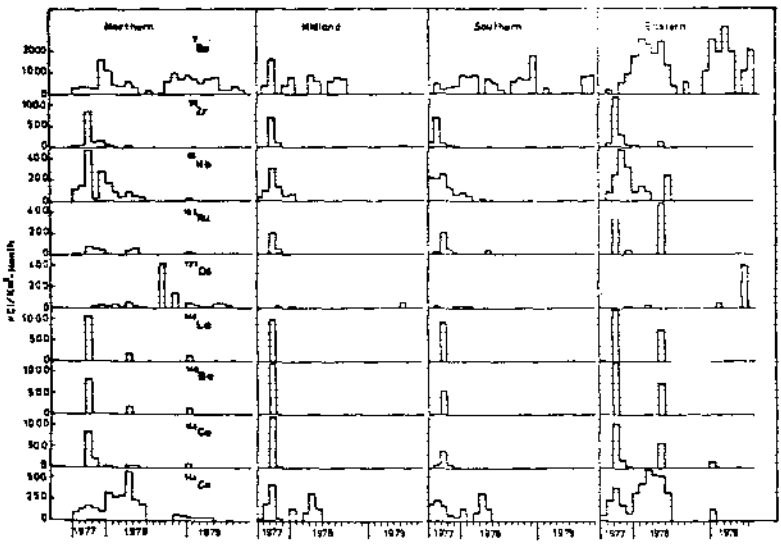

Figure 4. Concentration of fission products in airborne particles during 1977 to 1979.

TABLE 4. Concentration of Strontium 90 and Cesium 137 in ten important foodstuffs, and estimated total daily intake in the average diet.

Food	Personal consumption kg/year	%		^{90}Sr		^{137}Cs	
		Ca	K	pCi/kg	yearly intake %	pCi/kg	yearly intake %
Rice	132.7	0.04	0.3	3.9	43.4	29.1	44.5
Vegetable	98.0	0.01	0.34	3.5	15.6	3.3	3.7
Pork	2.2	0.11	0.18	6.6	3.7	16.3	4.2
Fish	21.9	0.59	0.36	6.5	12.0	6.6	1.7
Eggs	9.5	0.05	0.30	2.7	2.1	2.1	0.2
Milk	9.4	1.29	0.90	19.5	15.4	65.1	42.1
Potato	8.9	0.09	0.15	2.6	1.9	11.8	1.2
Chicken	7.7	0.18	0.26	2.4	1.5	4.3	0.4
Wheat	5.4	0.02	0.26	5.4	2.2	22.4	1.3
Duck	3.6	0.15	0.35	6.9	2.1	15.8	0.6
yearly intake	300 kg	374 g	652 g	1191.7 pCi		8672.3 pCi	
daily intake	820 g	1.02 g	1.79 g	3.48 pCi/g.Ca		23.8 pCi/d	

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THE PRE-OPERATIONAL MONITORING - HOW USEFUL ARE RECOMMENDATIONS OF INTERNATIONAL ORGANIZATIONS AND VARIOUS NATIONAL PROGRAMS

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NATIONAL LEGISLATION

Environmental monitoring around Nuclear Power Plant Krško was the first pre-operational monitoring to be performed in our country. The existing national legislation in the year 1974 included a basic law on radiation protection and accompanying regulations on radiation protection of workers, on registration of use, of production, transport and trade, as well as storage and release of radioactive materials, on monitoring the fall-out, food and animal feed, on education and health control of workers.

Six centers for radiation protection in six republics were responsible for monitoring and inspection. Some responsabilities were given to radiation protection laboratories in the existing three nuclear science institutes.

During the seventies the basic law with accompanying regulations was updated with some general requirements concerning siting, building and licensing nuclear installations, including some restrictions related to releases and storage of radioactive materials.

Until now no regulatory body for nuclear legislation with full time professionals was formed.

THE PRE-OPERATIONAL MONITORING PROGRAM DESIGN

The pre-operational environmental monitoring programs, in the period from 1974 to 1979 were based on recommendations of international organizations (1,2), the various national programs (3,4) and our own experience.

The general guidelines for the pre-operational radioactivity investigations necessary for estimating the dose from planned releases and for the establishment of limits and conditions of radioactive releases from an installation to the environment were given in Section IV, paragraph 405 of the reference (2). Yet in the above publication no suggestion was given on which samples are to be measured by specific and/or nonspecific methods, on the frequency of sampling, on the number of sampling sites etc. In paragraph 410 of the same publication it was stressed that the pre-operational investigations should provide quantitative data for the derivation of working limits and action levels for routine and emergency environmental monitoring.

National programs of pre-operational monitoring usually give detailed guidelines regarding the method of measurements for different samples and sampling frequencies. Yet all these programs are madefor specific needs for each site.

The first pre-operational monitoring program for Nuclear Power Plant Krško was defined by the investor and the laboratories, which were doing the measurements, and approved by the sanitary inspectorate of the republic. The results were periodically reveiwed and the program was improving until it was felt that the objectives were reached.

In the following, our first program will be described and the improvements which were made will be outlined.

RESULTS

Our first pre-operational program design included sampling and monitoring of surface and underground water, sediments and biota of Sava river, soil, agricultural products and natural radiation background. After the first review of results more emphasis was given to isotopic determination of water samples, biota and agricultural samples. Quality control was included in the program. After some time further modifications were introduced with the aim to identify the site specific critical exposure pathways. Modifications included the change and addition of sampling sites for underground water, river biota and agricultural produce as well as changes in frequency of sampling. Program of annual intercomparison measurements between the laboratories involved in the program and regular review of results was introduced.

DISCUSSION

Having in mind the main objectives of the pre-operational monitoring: to provide data on radioactivity levels in various environmental media and provide quantitative data for derivation of working levels and derived limits we have considered that quality assurance and the usefullness of information which the results provided should be the main criteria for a successful program. Examining the results of our five year long pre-operational monitoring program, we found that about one third of our results were unsatisfactory.

From our experience we learned that nonspecific measurements, unless they are made on the same samples as specific isotope determination, are of far less use than specific measurements. The results of specific measurements provide information which can be used for identification of critical exposure pathways, for determination of transfer parameters between compartments in an environmental model, and for establishing radioactivity levels in environmental samples in the pre-operational period.We realized also,

that critical examination of the relevance of program design as well as the examination of the quality of results is essential.

CONCLUSION

Lack of adequate national legislation and legislature body, as well as lack of authorized, experienced and competent radiation protection specialists resulted in our vaguely defined and poorly performed pre-operational monitoring in the year 1974-77. All improvements of the program were made by a concentrated effort of few competent radiation protection specialists from nuclear science institute laboratories.

We greatly emphasize the need for early education and training of radiation protection workers, as we cannot envisage a nuclear regulatory body, a superwising authority and a successful pre-operational and/or operational monitoring program without experienced specialists in radiation protection.

We feel, it may be useful that in their recommendations related to pre-operational monitoring the international bodies put more emphasis on site specific exposure pathways investigations, on the usefulness of specific activity determination of environmental samples as well as on regular critical examination and evaluation of results.

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MIGRATION OF RADIONUCLIDES FOLLOWING SHALLOW LAND BURIAL

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INTRODUCTION

This paper summarizes a study of radionuclide migration conducted at a facility used from 1944 to 1949 for the shallow land burial of radioactive waste. The waste material was produced during operations with two nuclear reactors and related nuclear research at this location. The facility is located in a wooded, sparsely populated area approximately 25 km southwest of Chicago (U.S.A.). It is situated in glacial drift, which is about 45 m thick at the burial plot. Underlying the drift is a generally level Silurian dolomite bedrock approximately 60 m thick. The thickness of the drift decreases as the surface slopes downhill in a northerly direction until the dolomite reaches the surface and forms the bed of a river, 700 m to the north.

Two aquifers exist in the area - one in the drift, in which the groundwater level varies with precipitation and season and averages about 9 m under the plot; and one in the underlying dolomite, a generally porous structure. Precipitation reaches the river either as surface runoff water or by infiltration into the glacial drift and dolomite.

From 1944 to 1948, material was buried in 2 m deep trenches and covered with earth. From mid-1948 to mid-1949, material was buried in steel bins. These were removed in 1949 and use of the burial plot was discontinued. In 1956, the facility was decommissioned by building a concrete wall on all four sides down to 3 m below the surface and covering it with a 30-cm thick concrete layer. The concrete was covered with two feet of soil and the surface seeded with grass. The area of the burial site was approximately 45 m by 40 m.

The reactors and laboratory research buildings were located 600 m south and 10 m uphill of the burial plot. In 1955, the reactors were dismantled; the heavy water, graphite, and fuel removed; the reactor tank and containment shell filled with miscellaneous reactor items and concrete; and buried in a 12 m deep excavation next to the reactor building. The top of the reactor shell is 7 m below the surface. The excavation was filled with building rubble, then with earth, and seeded with grass.

This study was begun in 1974, following the detection of tritiated water in two hand-operated picnic wells north of the facility, between the burial plot and the river.

METHODS

To establish the origin of the tritium, evaluate the water

migration rate and pathway, and determine if other radionuclides had migrated, the following work was done: 1) soil borings were drilled through and around the facility and near the buried reactor, and the soil samples were analyzed radiochemically; 2) test wells were drilled into the dolomite bedrock; 3) water permeability of the drift and groundwater velocity were determined; and 4) radiochemical analyses of surface water, surface soil, vegetation, and all existing wells in the area were conducted.

RESULTS

The tritium (as tritiated water) content of the two wells is shown in Figure 1. The seasonal fluctuations are apparent. In the well closest to the burial plot (Well A) the concentrations ranged from about 14 nCi/l in the winter to about 0.1 nCi/l in the summer, except in 1977 when the summer concentrations remained at 8-9 nCi/l. Since that anomalous period the usual seasonal variations have resumed. Well B also showed elevated tritium concentrations that varied with time, from about 0.1 nCi/l in the autumn to 6 nCi/l in the spring, about 4-6 months later than the peak concentration in Well A.

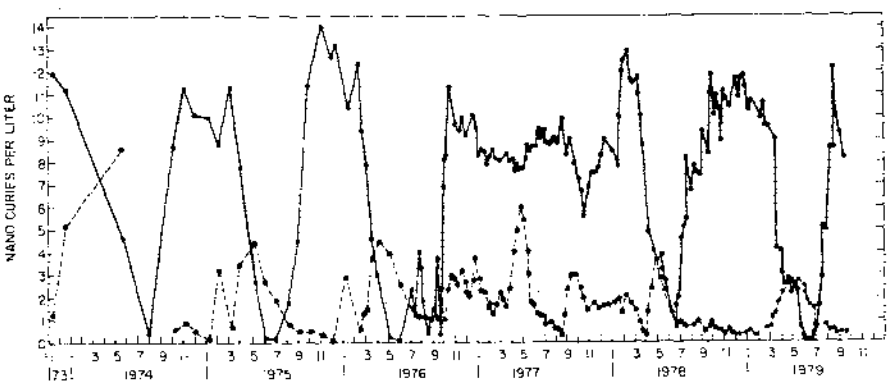

Fig. 1. Tritium concentrations in two wells near burial site.
— Well A, 350 m from site: --- Well B, 650 m from site.

The tritium content of the surface water and subsurface soil showed that the tritium in the wells originated in the burial plot and not in the area that contained the buried reactors. The burial plot is drained by an intermittant stream that flows in a northerly direction toward the river. Tritium in this stream during the spring, when water is present, has varied from < 0.2 nCi/l upstream of the plot to 30-100 nCi/l adjacent to the plot, then decreased with distance downstream as a result of dilution to about 5 nCi/l.

Between the burial plot and the reactor site, soil borings down to 25 m contained less than 0.1 pCi of tritium (as water)/g. Soil

borings near the buried reactor down to 20 m contained from < 0.03 pCi/g to 15 pCi/g. In the vicinity of and beneath the burial plot the subsurface tritium content was considerably higher, 0.2 pCi/g to 5×10^4 pCi/g. The tritium content of the soil borings showed the direction of water flow to be as expected, from the facility in a northerly and downhill direction. The deepest sample obtained from the glacial drift was 2-3 m above the dolomite bedrock and contained about 300 pCi/g.

The soil borings were also examined for Pu, U, ^{90}Sr, and gamma-ray emitters. Beneath the burial plot concentrations were normal, except for Pu in the first 2 m directly below the buried material. The maximum concentration here was 0.07 pCi/g, about twice the fallout concentration in this area. Higher concentrations of several radionuclides (U, Pu, ^{90}Sr, ^{137}Cs, and ^{152}Eu) were found at the depth of the buried material. None of the soil borings outside of the facility contained abnormal concentrations of these elements. The solid-element nuclides evidently migrated very little after burial.

Surface soil from the vicinity of the burial plot was also collected and analyzed. A small area (30 m x 60 m) in a swale 50 m northwest of the plot contained elevated concentrations of normal U (10-60 pCi/g) and Pu (0.07 to 0.7 pCi/g). The usual concentrations are about 2 pCiU/g and 0.04 pCiPu/g. The U and Pu in this area is believed to be the result of spillage during burial and removal operations and not from leaching of the buried material, since the surface and subsurface soil between the plot and the contaminated area contained only normal concentrations.

All other water wells in the same glacial drift area were examined, and none contained abnormal concentrations of the radionuclides found in the burial area.

DISCUSSION

The results of all the surface and subsurface measurements indicate that tritium is migrating out of the burial site as tritiated water, but that no other radionuclides have left the plot.

The tritium concentrations decreased with increasing distance from the plot. Tritium has been found in the subsoil as deep as samples have been collected, so it must be assumed that the ground beneath and immediately around the plot contains tritium down to the dolomite aquifer. Integration of the concentrations as a function of depth and distance yields a total tritium content for the glacial drift of the order of 3000 Ci. The dolomite aquifer north of the plot must also be assumed to contain tritium from the plot, based on the well water results. However, in the river that drains the plot, tritium was not detected above normal levels due to dilution.

The time of travel of water from the burial plot to the nearest well is estimated to be 54 months. This estimate is based on permeability measurements, from which a groundwater velocity in the till of 8.2 cm/day was obtained and on matching the peak concentrations in the wells, which gave a flow rate in the dolomite of 238 cm/day. Future dolomite tritium levels may be estimated from the concentrations beneath the plot, where a maximum in the tritium content occurred 20 m below the surface. If the tritium was placed in the burial area about 1947, and has moved downward as a pulse for the past 30 years,

the average rate would be about 0.67 m/yr. This would imply the peak concentration would reach the dolomite in about 35 years. By this time, 86% of the tritium would have disappeared by radioactive decay.

The cyclical nature of the tritium content in the two wells implies that tritiated water is carried from the burial site by the spring rains when they recharge the groundwater supply. During this period, the groundwater rises and a portion of the tritium in or beneath the plot is mobilized, travels underground and in the surface stream, and eventually enters the aquifer. At other times, the groundwater level drops and little water moves out of the plot.

The principal pathways for radiation exposure are consumption of water from the picnic wells and inhalation of resuspended material from the surface soil in the swale northwest of the plot. The dose due to drinking well water can be evaluated from the ICRP model for ingestion of tritiated water. Depending on the quality factor used for the tritium beta particle, the biological half-life for body water, and the daily water consumption, the model estimates the dose to an individual who consumes 1 liter of water daily at the average concentration in Well A (7 nCi/l) to be 0.4 to 0.8 mrem/yr. The U.S.E.P.A. limit for drinking water is 4 mrem/yr.

The potential inhalation dose can be estimated as follows. Resuspension factors for actinides in vegetation-covered soil measured in this area are (1 to 5) x 10^{-10}/m. These factors, applied to the average surface soil concentrations in the swale area, give air concentrations of about 0.4 fCi^{240}Pu/m^3 and 0.008 fCi^{239}Pu/m^3. Air breathed continuously at these concentrations would result in lung doses of less than 0.2 mrem/yr. The U.S.E.P.A. limit for airborne transuranium elements in the environment is 1 mrad/yr, or 10 mrem/yr if a quality factor of 10 is assumed. The potential doses given above are quite conservative since individuals do not live in the immediate area. Exposure times to airborne material and consumption of well water by individuals are small and infrequent.

The two most probable sources for the tritium in the plot are neutron-irradiated heavy water from the CP-3 reactor and lithium which had been irradiated with neutrons for tritium production. From measurements of the relative amounts of tritium, deuterium, and protium in water draining from the facility, it is apparent that the source was neutron-irradiated lithium.

Since water is the vehicle by which buried material is moved, reduction or elimination of water movement through the burial plot would reduce the rate of radionuclide migration. Some possibilities are waterproofing the concrete cap and installing drain tile around the concrete sides to divert water past the area. Although this would have little effect on the tritiated water deep in the soil, it would reduce the probability of future migration of the other radionuclides in the plot. Also, deeper aquifers are being examined to determine if water containing less tritium is available in the immediate vicinity.

The results of this study should be applicable to the evaluation of other shallow land burial sites.

PRINCIPLES AND RESULTS OF ENVIRONMENTAL SURVEILLANCE OF THE AUSTRIAN RESEARCH CENTER AT SEIBERSDORF WITHIN THE LAST TWENTY YEARS

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1. INTRODUCTION

The Research Center at Seibersdorf uses its 12 MW reactor for isotopes production, fuel testing and activation analysis and operates also several active laboratories, an accelerator, a waste managing department with an incineration plant and an intermediate store for low and medium active wastes.
On the area of the center also the Safeguards Analytical Laboratory of the IAEA was errected.
Our monitoring system has principally the following aims:

- to control the radioactivity which is emitted into air and waste water
- to ensure that governmental regulations are fulfilled
- to detect any changes of the environment by longterm emissions.

The monitoring of the center is performed by emission and immission measurements supported occasionally by research projects.

2. MONITORING DEVICES

2.1 Installations for controlling the emissions

The ASTRA reactor is the main emittent of shortlived radionuclides:

TABLE 1: Shortlived radionuclides emitted by the ASTRA reactor

Radionuclide	Activity per year
Ar 41	200 ± 20 Ci
H 3	1 Ci
Fission gas	0,2 Ci
Aerosols, Rb 88, Cs 138	91 Ci
I 131	5.10^{-5} Ci

For monitoring the breathing air in the reactor hall it was therefore a counting unit in the off gas tunnel installed which has a detection limit of 3.10^{-8}Ci/m³. Another unit with the same capacity is posted in the 25 m off gas tunnel.

For the determination of the aerosols, 100 m³ air are by-passed over a micropore filter. Activities up to 7.10^{-14} Ci/m³ can be counted with a methane flow counter.
I-131 is absorbed on charcoal and measured with a Ge(Li)-detector giving a detection limit of 3.10^{-14} Ci/m³. The off gas tunnels of the hot cells and active laboratories are monitored by GM-counters with fixed alarm levels of 1000c/min or 4000 c/min respectively.
All air streams leasing outside of SAL are controlled very strictly for their alpha-, gamma- and beta-activity.

2.2 Permanent monitoring of the air

By a pumping station in the middle of the center daily 100 m³ air are sucked through a glasfibre filter and the alpha-, gamma- and beta-activity measured after 10 min, 3 hrs and 120 hrs after collection.
If the 10 min value is higher than the normal scatter - the values will change with the meteorological conditions about a factor of 3 - this filter will be measured with a gammaspectrometer.
The 120 hrs-values from the year 1978 differ e.g. form 0,034 to 0,152 pCi/m³.
A second aerosol monitor is located on the roof of the institute for radiation protection. The collection and measurement are taken continously with a filter band. The evaluation is made with plastic scintillators to alpha- and alpha+beta-activity.
In the same place also the meteorological datas are collected.

2.3 Controlling of water activity

In our center the following water systems exist:

- raining water:
  it is drained into ground, but from the first water flowing down the pipe, a 2 litre sample is taken.

- wasting waters:
  coming from baisins, showers, kitchen and toilets; they are conducted directly to the biological purification plant.

- radioactive waters:
  all waste waters coming from the active laboratories are led by a special piping directly to the decontamination plant.
  High radioactive fluids are not allowed being poured in the sinks, but have to be brought in protected vessels to the plant.

- possible radioactive waters:
  they are gathered in tanks, the activity is measured and either dispatched or led to the decontamination plant.

All cleaned sewage waters coming from the biological purification- or decontamination plant, are collected in a reservoir and pumped to a near river.
For monitoring, daily samples are gathered from pumping reservoir, the cooling pond and the reservoirs of the decontamination plant before releasing.
Weekly water from three wells in the center and a series of wells around the area, a sample of a near ditch and one of the drinking water is taken.
Also samples of the roof waters are measured every week if it is raining. The averaged activity is about 6 - 15 pCi/l. After a longer dryness the activity increases to about 100 pCi/l.
Further wells and surface waters in the environment are collected in half a monthly, monthly and tree months cycle The values differ from 0 to 99 pCi/l.

2.4 Controlling of the soil samples

Furthermore soil samples and vegetables are collected from six different points. The soil samples are taken from an area 10 x 10 cm and the depth of 5 to 10 cm. The grass is taken from 1 m^2. The values in 1978 were:

- Pu 239: grass <3 to 21 fCi/g ash.
  soil <3 to 12 fCi/g ash
- beta-activity: grass 72 to 284 pCi/g ash.

The samples are taken quarterly.
From five places from the river Leitha - above the mouth of the waste water pipe, 10 m, 100 m and 1000 m below and in the next village - various samples are taken: sand, mud, seeweads and small craw fishes. After preparation the activity is determined. The sensitivity of the measurement is good enough that ^{90}Sr and ^{239}Pu can be determined in the size of the fallout concentrations.
Half a year samples are collected from diverse parts of the purification plant and every year sand, fishes and biological material.

2.5 Installations for controlling the immissions

The measurement of the immissions are made in principle for preventive events, to realize the limits set up by the authorities and to verify the values which are calculated from the measurements of the emissions.
The environmental surveillance to the gamma dose is made by TLD dosimeters. We use simultaneously card and bulb dosimeters which are housed in an Al-cylindre for protection against weather (1).

Dosimeters are placed in two about concentric circles around the center to superintend the environment independent from different emission parameters, the inner circle

in 30° and the outer in 90° steps. The change of the dosimeters and the evaluation is made monthly.

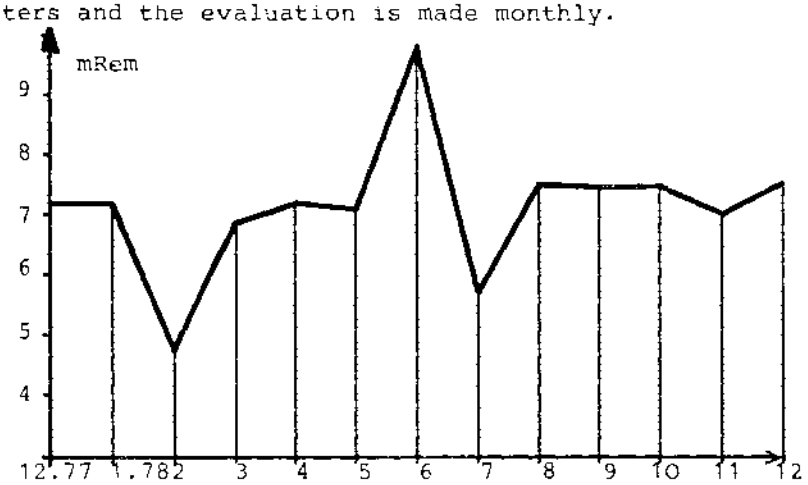

Fig.1 Fence-dose in the main-wind direction TLD

Additionally, the fence dose is measured with two integrating gamma radiation doserate monitors. One is situated in the main wind direction and the second in one of the surrounding villages. The detectors are sensitive GM-tubes equipped with a registering unit.

3. DISCUSSION

Since the center lies in an agricultural area, at first the main stress was led on the monitoring of water and also of air. The described system was extended by rings of TL-dosimeters and doserate meters and aerosol monitors. The described system was sufficient up to now in monitoring alpha-active laboratories and waste management. For comparison, in research work the Pu-fallout was determined all over Austria (2). For verification of the valid calculations for the spreeding of activity in the air, a filter pump unit to collect 5000 m^3/h on a charcoal filter is in construction.

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SIX-YEAR EXPERIENCES IN THE OPERATION OF A LOW LEVEL LIQUID WASTE TREATMENT PLANT

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After the review of the techniques and experiences of other countries and considering the specific situation in Taiwan, we at the Institute of Nuclear Energy Research (INER) constructed a low-level liquid waste treatment plant in 1973 (Fig.1). Thereafter, the fuel fabrication plant, the isotope production plant and other research laboratories have been set up one by one. The amount of liquid waste has been increased from 600 tons to 2500 tons every month. The activity concentration ranges from $10^{-5}\mu Ci/ml$ to $10^{-3}\mu Ci/ml$. The waste volume and the radioactivity as well have increased gradually with the growing of the INER. The treatment plant has also been improved every year to fulfil the requirement of the INER.

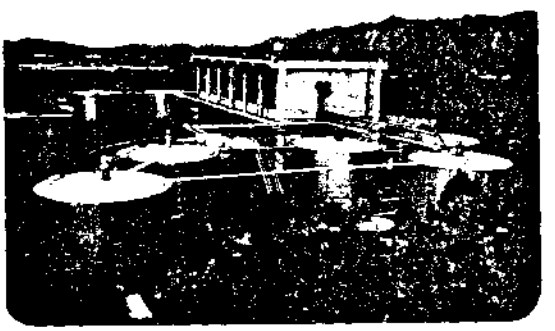

Figure 1

CONTINUOUS PROCESS

The flow digram of the low level radioactive liquid waste treatment is shown in Fig.2[1].

Receptor for liquid waste

After sampling with gross β counting and the adjustment of P^H at 6-9, the radioactive liquid waste from the operation and maintenance of the Taiwan Research Reactor and other plants was purged to the liquid waste treatment plant. The waste is classified into two catagories. The first catagory is the liquid waste whose gross β activity is below 4×10^{-5} $\mu Ci/ml$ and is purged to the tail end of the storage tank in the treatment plant. It is then discharged to the ground disposal area. The second catagory is that above $4 \times 10^{-5}\mu Ci/ml$ and is purged to the head end of the storage tank for treatment.

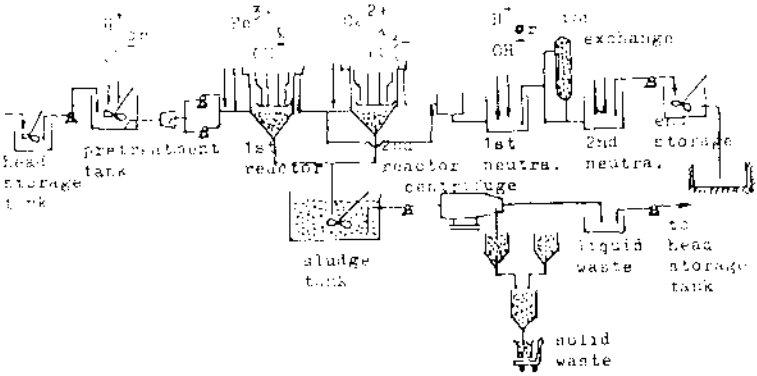

Figure 2

Liquid Waste treatment

Pump the liquid waste from the storage tank to the pretreatment tank for sampling, p^H measuremtns, radioactivity counting, and nuclide analysis. We then choose the proper chemicals and determine the quantity. After passing through the first reaction tank and the filtration tank, if the radioactivity is decreased to the discharge level, then the liquid waste will flow directly to the first and the second neutralization tanks for proper p^H adjustment, and finally it flows to the tail end of the storage tank. After sampling and counting, the liquid waste is discharged to the ground disposal area. If the liquid waste does not reach the discharge level, the clear liquid will be pumped to a second precipitation tank for precipitation with calcium phosphate. After precipitation, if the upper clear solution is below the discharge level, it will be pumped to the neutralization tank, and then to the tail end of the storage tank and finally to the ground disposal area. Otherwise it will be pumped to an ion exchange tank. If it still does not reach to the discharge level, it will be pumped to the head end of the storage for recycle.

Treatment of precipitate

Most precipitates in the two precipitation tanks are $Fe(OH)_3$, $Ca_3(PO_4)_2$, and $Ni_2Fe(CN)_6$. They will be centrifuged to a concentrate sludge which has 10-20% solid content. The sludge will be stirred with cement at the weight ratio of 2 to 1, then it will be transferred to a 53 gallon barrel for dryness and solidification. After being covered and sealed,the barrel is stored for further treatment.

Discharge of Waste liquid

Although the waste liquid in the process is monitored frequently, effluent samples should be taken and counted before discharged to the qround disposal area for the prevention of violating the regultions. The trace amount of nuclides will be absorbed or adsorped in the soil after discharged. There are 20 wells whose depths vary from 25 to 30 m around the disposal area for monitoring the contamination of under-

ground water.

OPERATION EXPERIENCES AND IMPROVEMENT OF EQUIPMENT

The advantages and disadvantages of continuous and batch processes

The batch process uses the pretreatment tank for the addition of chemicals. The quality control (QC) group takes sample for analysis. Taking the optimum condition based on QC, the p^H of liquid waste is adjusted and the chemicals are added. The liquid waste is stirred sufficiently and settled for 6 h or overnight to precipitate the radionuclides. From the upper clear liquid, sampling is performed for counting and analysis and to determine whether it may be discharged or not.

The advantages of batch process are very short operation time and large capacity. The disadvantage is that it requires more operation tanks which are not necessary for continuous process. The continuous operation for 24 h, on the other hand, has lower capacity and requires more man power.

Improvement of ion exchange capability for vermiculite

Vermiculite, with good water permeability and high selectivity of cesium ion, is one the cheapest ion exchangers available. It is appropriate for treatment of low level radioactive liquid waste.

From the practice of the plant operation, it showes that the high concentation of ion will cause the efficiency of the vermiculite exchange capability to decrease. If the waste liquid pretreated with vermiculite before chemical treatment, then the efficiency of decontamination will be increased and the exchange capacity of vermiculite will be improved.

As the experimental data show, vermiculite has poor selectivity for ^{90}Sr-^{90}Y, and its efficiency for ion exchanges is low. The more efficient method for treatment of ^{90}Sr-^{90}Y is the precipitation with calcium phosphate.

The improvement of peat absorption for liquid waste treatment

Since the combustion capability of peat, it is not suitable to be used as a fuel. In the past years, peat was widely used in soil improvement and gardening. Recently its application shifts to the utilzation of its specific characters.

The p^H value of the solution has great influences over the absorption capability of peat. This is due to that the humic acid in peat has lower exchange or chelation reaction under high H^+ concentration. The peat has more absorption as weak alkali. Usually, the waste liquid after chemical treatment is in this range.

A 10 cm layer of peat on the ground disposal area will decrease the contamination, and make the liquid waste treatment more appropriate. After saturation, the peat should be removed for solid waste treatment. The disadvantage is the small volume reduction of peat after incineration.

Improvement of sludge treatment

The sludge, the coprecipitate from liquid waste treatment plant with solid content about 3-6 W%, is centrifuged after dryness. The volume is decreased and then it is treated with cement. The solidified products are ready for final treatment. The centrifuge is a continuous horizontal type. The solid is separated from liquid by a conveyor screw. The character of the centrifuge is its ability to separate different properties of suspensions by varying the radius of rotation and the difference RPM between bowl and conveyor.

The effluent from the separation of sludge still has too high solid content. It cannot be transferred to the liquid waste storage tank. Replace the liquid storage tank with two 50 m^3 tanks as feeding and effluent recieving tanks for decanter, then the operation can meet with the requirement[2].

Environmental monitoring

The final liquid waste after treatment is discharged to the ground disposal area. The adsorption capability of soil keeps the ground water from contamination. To guard the underground water resources and prevent them from contamination, there are 20 deep wells around the disposal area for environmental monitoring. Routine sampling and analysis for several years has shown that the radioactivity is about the background. They give no evidence of contamination.

CONCLUSION

The construction and operation of this low level waste treatment plant is the first experience in Republic of China. Some improvements have been made as mentioned before.

The total volume of liquid waste treated in 6 yr was 84725 m^3. The decontamination factor was from 5 to 50. The gross beta activity discharged after treatment is 2299 mCi, less than 400 mCi per year on average which is less than the predicted discharge activity of 1 Ci per year. From the experience of operation for 6 yr, we consider batch process to be more practical than continuous process especially the capacity of treatment is concerned. The monitoring of well samples shows that the liquid waste discharged to the ground disposal area does not contaminate the environment.

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THE DEVELOPMENT OF CRITERIA FOR LIMITING THE NON-STOCHASTIC EFFECTS OF NON-UNIFORM SKIN EXPOSURE

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Several recent papers (1, 2, 3) have outlined some of the limitations of the latest recommendation of the International Commission on Radiological Protection (ICRP) in the case of skin exposure. ICRP publication 26 (4) for example gives little or no consideration to the importance of

(1) the incidence of radiation induced skin cancer
(2) alpha and low energy beta irradiation of the basal layer, particularly for thin skinned personnel
(3) partial-body skin exposure such as that to the fingers
(4) very non-uniform localised exposure such as that from small sources and radioactive particulates.

Items 3 and 4, which we will refer to as cases of non-uniform skin exposure, will be the subject of this paper.

NON UNIFORM SKIN EXPOSURE - THE PROBLEM REVIEWED

When personnel are exposed to external and/or internal sources of radiation the resultant exposure of the body is rarely uniform. In order to limit the potential detrimental effects of a non-uniform skin exposure it is necessary to consider the probability of producing stochastic effects (cancer) and early or late non-stochastic effects (ulceration, fibrosis, cosmetic changes, etc.).

For chronic exposures of large areas of skin stochastic effects may be limiting, particularly if skin cancer incidence rather than mortality is considered (1). The whole body skin dose limit may need to be reduced below the present 0.5 Sv a^{-1} if skin cancer incidence is considered to be of concern for medical or industrial-relations reasons.

In practice the skin of the whole of the body is rarely, if ever, exposed near to the dose limit. As the exposed skin area is decreased the stochastic risk can be expected to be reduced on the basis of there being a proportional decrease in the number of irradiated cells (5). At some particular value of skin area the stochastic dose limit may be so high that non-stochastic effects become limiting. The area at which this occurs depends upon an assessment of the relative detriment of skin cancer incidence versus cosmetic changes. The present ICRP non-stochastic dose limit of 0.5 Sv a^{-1} is that which corresponds to unacceptable chronic skin changes, mainly of a cosmetic nature. Little relevant data exist to provide a basis for this limit other than the work of Sulzberger (6). Recent followup studies of occupationally exposed groups

indicate that the limit may be too high (7). More clinical, epidemiological and experimental evidence is clearly needed before a rational system of skin dose limitation can be implemented which applies to the range of skin areas irradiated in practice.

For very small area exposures, perhaps less than 1 cm^2, such as those from small sources or radioactive particulates, the likelihood of chronic exposure of the same body site is reduced and the acute response to high local doses becomes of significant practical importance. The ICRP offer little guidance for this situation (4, paragraph 183).

An evaluation of the mean dose to an area of 1 cm^2, which was previously recommended by the ICRP (8), is no longer considered appropriate for comparison with the dose limit. In the absence of suitable guidance the averaging procedure will however probably continue to be used and is likely to represent a conservative approach for most situations. A sounder basis for operational procedures and a knowledge of the magnitude of pessimism embodied in averaging procedures is clearly necessary.

There is at present so little clinical or experimental data for non-uniform exposures that the ICRP (4) advice to predict local skin reactions on the basis of the absorbed dose distribution cannot be complied with. What data is available to aid a prognosis?

RÉSUMÉ OF NON UNIFORM SKIN EXPOSURE STUDIES.

Only three (9, 10, 11) systematic studies appear to have been attempted with the specific aim of providing guidance to predict the likelihood of non-stochastic effects in non-uniformly irradiated skin.

Kreb's (9), as part of a theoretical study of the response of mammalian skin to irradiation with particles of irradiated nuclear fuel, produced guidelines on the effects of radioactive particulates on skin. In a preliminary study to verify the usefulness of these criteria Forbes (10) evaluated the effects of irradiated uranium microspheres on the skin of mice and pigs. The point depth dose (i.e. the dose in tissue at a depth of 100 μm directly below the particle) and the 'Kreb's dose' (the dose at the edge of a circular field of 4 mm radius, at a depth of 100 μm below the skin surface) were both found to correlate well with the area of damage in pig skin (Fig. 1). However, both ulceration and desquamation in pig skin occurred at lower Kreb's doses than the 15 Gy predicted largely on the basis of clinical data using X and γ radiation. The minimum Kreb's dose required to produce a small but recognisable ulceration in pig was below 4 Gy. Extrapolation of the data to derive threshold doses is difficult since no sub-threshold data was generated by Forbes.

It is interesting to note that Dean, Langham and Holland (11) reported the response of human skin to acute irradiation with fuel particles of the same type used by Forbes. Exposure of the forearm to peak basal layer doses of ≥ 500 Gy produced transient point erythema and dry desquamation respectively.

Mahlum (12) has irradiated the skin of Hanford miniature swine, to doses of 10, 25, 62.5, 150 and 400 Gy (surface dose) of β radiation from circular ^{204}Tl plaques with areas of 0.01, 0.5, 1.0 and 5.0 cm^{-2}. The aim of the study was to investigate the dependence of the non-stochastic response on dose and area irradiated.

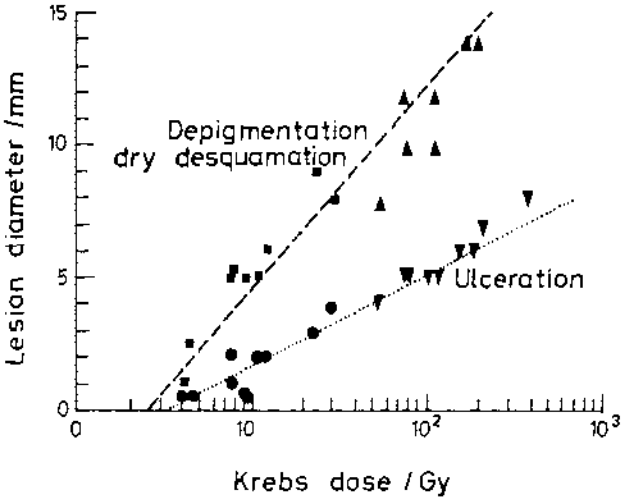

Figure 1. The response of pig skin following irradiation with radioactive particles. Small particles (dia ∿ 100 μm) are represented by squares and circles; large particles (dia ∿ 250 μm) by triangles. Data from Forbes (10). Peak doses = 200 (600) x Kreb's dose for large (small) particles.

Due to the limited number of radiation doses used in this study it is not possible to provide precise threshold doses, but from an examination of colour slide records it was possible to provide values for adult Hanford miniature pigs (Table 1).

Area of $^{204}Tl/cm^2$	Surface Skin Dose/Gy	
	erythema	ulceration
10^{-2}	> 400	> 400
0.5	62.5 - 150	150 - 400
1	62.5	150
5	62.5	62.5 - 150

Table 1. Threshold doses for Erythema and Ulceration in Pig Skin using small ^{204}Tl beta sources (13).

Moritz and Henriques (14) irradiated the lateral and dorsal surfaces of young Chester White pigs (10-15 kg) with circular (1 cm^2 β-emitting plaques. Little correlation was seen between surface dose and biological response for the different β-energies but good correlation was seen when the dose was calculated at a depth of 9 mg cm^{-2}, in the vicinity of the basal layer. The most important result of this study is the clear illustration of the importance of β-energy in any consideration of the biological response.

SUMMARY

The studies of Forbes, Mahlum and Moritz and Henriques, though all necessarily limited, show the important role which animal experiments can play in the development of protection criteria for acute non-uniform skin exposures. There is a continuing requirement for studies of the dependence of threshold doses on source area and radiation quality. More than one animal species should be studied and extrapolation to man should be on the basis of histology and cell kinetic studies, taking into account the biological and morphological differences between animals. A more general model than that proposed by Krebs could then be evolved for application in the more complex practical situations encountered by radiation workers. We are involved in a collaborative radiobiology programme of this kind with the Churchill Hospital Research Institute, Oxford and St. Bartholomew's Hospital Medical College, London which involves the beta irradiation of pigs and mice. The rationale of the programme has been described elsewhere in some detail (15, 16) and the results will be published in due course. Preliminary data supports the view that there is a reduced acute biological response as either the area of the irradiated skin or the depth of penetration of the radiation is reduced. This work will indicate the level of pessimism entailed in dose averaging procedures.

Clinical and epidemiology studies should continue to be encouraged in order to underwrite, and further develop, criteria for limiting large area skin exposures.

ACKNOWLEDGEMENT

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INVESTIGATION OF THE NATURE OF A CONTAMINATION CAUSED BY TRITIUM TARGETS USED FOR NEUTRON PRODUCTION

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1. INTRODUCTION

The purpose of this paper is to show that the risk of handling targets used for generating fast neutrons in a Van de Graaff accelerator is caused by metal-tritide particles which may be emitted from these targets. It is obvious that this fact, also described by Fehér and Biró (1) who arrive at a slightly different conclusion regarding the dosimetry, constitutes another type of risk than most other cases of handling tritium, where incorporation in the gaseous phase must be considered.

The targets consist of a thin layer of titanium with occluded tritium, on a backing of Cu, Al or Ag. Our investigations started in consequence of a widespread contamination in a Van de Graaff accelerator building, after 40 targets containing an average of 7×10^{10} Bq (varying from 4×10^{10} to 56×10^{10} Bq) of tritium per target had been stored, prepared, transported and applied over a period of 7 years. The total size of the area was 3000 m^2. The degree of contamination over this area varied and there were some 1000 hot spots on floors, work benches and walls. The high level of this contamination was only realized after windowless gas flow counters had been introduced in the area. Faecal and urine samples of persons concerned were taken. Samples of the contamination were collected from the surfaces and auto-radiographed. When microscopic examination of the samples showed solid particles a few were further examined microscopically and by X-ray spectroscope. The presence of titanium and tritium in the particles was established. The method to arrive at the radiation dose after incorporation of the particles is discussed. The procedure of removing the contamination and the steps taken to prevent further contamination is not mentioned here.

2. DETECTION METHODS AND RESULTS OF MEASUREMENTS

The contamination check of the area was carried out using windowless gas flow counters. The total activity found was 3.7×10^7 Bq, distributed unevenly over the contaminated surfaces, with maxima of up to 5.8×10^5 Bq measured over surfaces of 1 dm^2. The derived working limit for surface contamination by low energy beta emitters is 37 Bq/cm^2 for low activity areas (2). For more detailed testing some radioactive material was collected from the surface on adhesive tape by covering a contaminated spot with the tape, applying light pressure, then pulling the tape off. These samples were auto-radiographed (3) either with Ilford nuclear research plates (K-5) or with liquid emulsion (L-4). After an exposure time of up to 8 hours the developed films showed black spots, underneath which solid particles were located after examination under an optical microscope.

In order to test our assumption that the particles had been

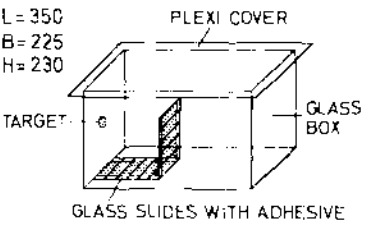

Fig. 1

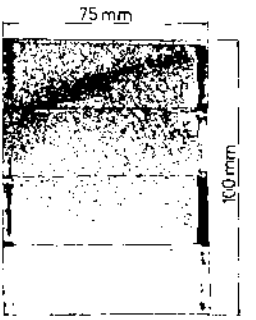

Fig. 2a HORIZONTAL

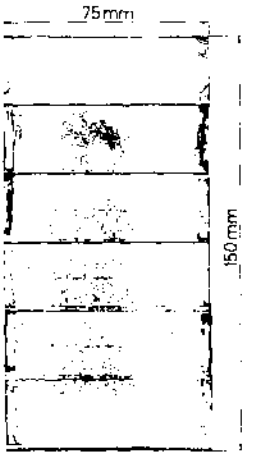

Fig. 2b VERTICAL

expelled from the targets a special laboratory experiment was developed.

An unused target was attached to the wall of the box as shown in Fig. 1. Microscopic slides covered with an adhesive were put in front of the target, in a horizontal and in a vertical position. After a period of 100 hrs the slides were removed and covered either with a liquid photographic emulsion or with nuclear research plates. The results of autoradiographs taken with nuclear research plates are shown in Fig. 2a and b.

The maximum physical sizes of 300 particles, measured microscopically and plotted on logarithmic paper follow a log normal size distribution, with a median diameter by count (CMD) of 10 μm and a standard deviation σ_g of 2.2 μm (Fig. 3). The smallest size detectable by this method is 1 μm; no particles with sizes below 3 μm were found. The corresponding aerodynamic diameter (CMAD) is found from the relation CMAD = CMD $\sqrt{\text{density}}$. Using a density of titanium of 4.5 g.cm^{-3} we find CMAD = 21 μm; the Mass Median Aerodynamic Diameter, MMAD, is calculated from the relation

$\log \text{CMAD} = \log \text{MMAD} - 6.91 (\log \sigma_g)^2$.

We find MMAD = 140 μm.

The presence of tritium in the particles was shown with the aid of a specially built, very small windowless gas flow counter with an opening of 2 mm diam. This was placed over a single particle by fixing the detector to the microscope, adjusting the co-ordinates of the window to that of a particular, isolated large particle. The β-spectrum thus measured corresponds to the spectrum of a tritium calibration source.

The presence of titanium was shown by measuring the characteristic line spectrum of the particles by X-ray spectroscopy as shown in Fig. 4; the Ag, Cl and S lines are due to elements in the

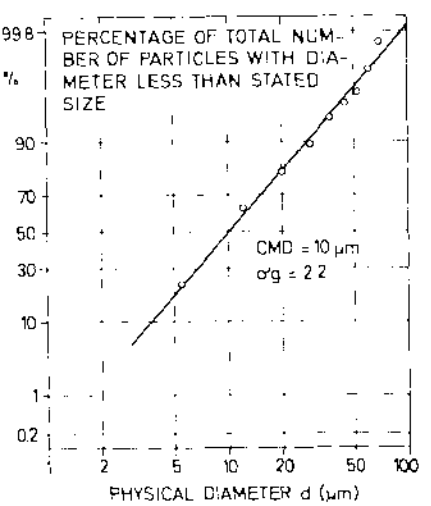

Fig. 3

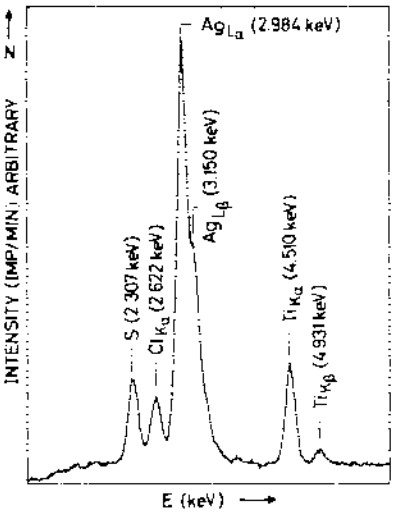

Fig.4

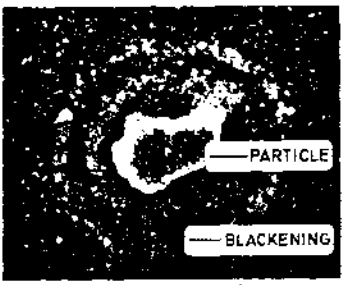

Fig.5a

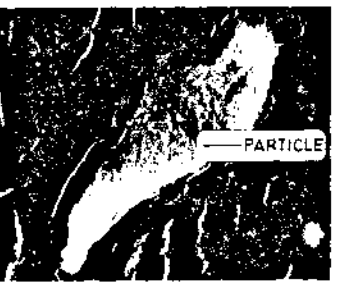

Fig. 5b

photographic emulsion. Fig.5a and 5b are scanning electron microscope pictures of a titanium particle with and without covering by emulsion.

The results of the urine sample analysis did not point to the presence of tritium in the urine. The faecal sample contents from 10 persons analysed during the period 1972-1979 ranged from 1.9×10^3 Bq - 0.4 Bq/24 hr sample. The tritium was unevenly distributed within the samples, which fact was found as each sample -for practical reasons- was divided into 4 parts before the analysis was carried out. The analyses were made by Dr.G.Koch of the S.C.K. laboratories at Mol. The fresh sample was dried under vacuum at room temperature in a disposable nickel crucible, the distillate being collected in a trap cooled with liquid nitrogen. The dry residu was burnt in a combustion train in an oxygen stream and the combustion water was collected in a trap cooled with dry ice. The radioactivity of the distillate and of the combustion water was measured in a liquid scintillation spectrometer. The detection limit for an aliquot of 10 cm^3 water is 1.9×10^{-2} Bq/cm^3.

An estimate at which the particles spontaneously detach from a target was obtained by applying the method described before (Fig.1) to a number of targets. An estimate of rate varied from 20 to 8000 particles per 24 hr.

3. DISCUSSION

Several results of our investigation point to the fact that the contamination consisted mainly of particles. The fact that

tritium was found in the faeces and not in the urine is the first indication. Tritium in gaseous form being inhaled or otherwise incorporated would have been distributed in the body fluids and excreted in the urine. The inhomogeneous distribution of the tritium in the faeces forms another indication. The results of the auto-radiographic analyses of the samples collected in the work as well as in the laboratory experiment confirm the "particle theory", together with the observation of particles by electron microscope. The X-ray spectroscopic analysis confirmed that the particles consist of titanium; the β-spectrum identified the tritium.

The distribution of inhaled tritium-titanium particles in the body organs can be estimated by applying the particle size data to the Dosimetric Model for the Respiratory System (4). According to the large size of the particles the initial deposition will exclusively take place in the naso-pharynx. Those deposited in the pharynx will subsequently be removed with the mucus into the mouth and then swallowed. The nose-deposited particles will either be removed by nose blowing or be swallowed.

The radiation dose to the naso-pharynx will be small because of the short passing time and the low energy of the β-radiation. The same applies to the gastro-intestinal (GI) tract through which the tritium will pass after having been swallowed. The surface cells of this tract are very insensitive for radiation (5). As metal tritides are not taken up into the body from the GI-tract (6) the material will be excreted after having passed this tract.

In order to determine the radiation dose quantitatively the relation between particle size and radioactivity must be better known than yet established. Work on this is going on. It can however tentatively be concluded that the dose must be considerably smaller than in cases where tritium is incorporated in comparitive quantities as a gas and that the dose to the lung in our case will be neglible.

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LES PROBLEMES DE RADIOPROTECTION RENCONTRES DANS UN LABORATOIRE DE MARQUAGE DE MOLECULES AU CARBONE-11.

H. VIALETTES - A. MOREAU.

LE CARBONE-11 EN MEDECINE NUCLEAIRE.

L'intérêt que suscite le carbone-11 par le marquage des produits radiopharmaceutiques est dû :

- à sa courte période (20,3 mn) qui permet de diminuer de façon très sensible la dose engagée pour le patient ;
- à sa grande radioactivité spécifique, qui permet de n'injecter que des masses très faibles et, par suite, sans effet physiologique, de produits toxiques ;
- à son mode de désintégration par émission β^+. Par annihilation, deux photons de 511 keV sont émis à 180° et des images tomographiques de la répartition de la molécule peuvent être données à l'aide d'une caméra à positons ;
- au fait qu'étant isotope du carbone naturel, son introduction dans une molécule organique ne modifie pas ses propriétés chimiques ou physiologiques.

A contrario, ces qualités mêmes, et, en particulier, la courte période, se révèlent être un inconvénient important sur le préparateur qui doit :

- faire une préparation pour chaque examen,
- partir de radioactivités élevées pour aboutir à une activité injectable suffisante au bout du temps de préparation. En effet, il est nécessaire, au cours de ce temps de préparation, de s'assurer de la parfaite pureté radiochimique, de la stérilité et de l'apyrogénécité du produit qui sera injecté au malade, de sorte qu'en moyenne, ce temps correspond à 2 ou 3 périodes ; ceci oblige à manipuler au départ des activités de l'ordre du curie.

LES RISQUES D'IRRADIATIONS LIES AUX OPERATIONS.

Le carbone-11 est produit par réaction $^{14}N(p, \alpha)\ ^{11}C$, à l'aide d'un cyclotron, il est présenté sous forme de $^{11}CO_2$.

L'incorporation du carbone-11 dans une molécule organique s'effectue par un marquage intermédiaire d'une des trois molécules suivantes appelées précurseurs : formaldéhyde, iodure de méthyle et acide cyanhydrique. Dès l'étape du marquage du précurseur, la source radioactive qui était produite sous forme gazeuse se retrouve sous forme liquide. L'activité est alors de l'ordre d'une fraction de curie recueillie dans un volume de plusieurs centaines de µl contenu dans des tubes en pyrex de forme cylindroconique de 1 à 2,5 ml. Les opérations sont effectuées à l'intérieur d'une hotte blindée mais nécessitent des interventions manuelles en certaines des étapes. Il est nécessaire de saisir à la main les tubes de réaction afin d'effectuer l'addition de réactifs, le prélèvement de fractions adéquates à l'évaporation de solvant. Le temps total de préhension du tube est inférieur à 1 mn. Une irradiation est également entraînée par les opérations de rinçages de l'appareillage, de prélèvements et d'addi-

tion de réactifs pendant lesquelles les doigts des opérateurs sont placés à 5 cm environ de la source pendant un temps de l'ordre de 1 mn. Enfin, des opérations de chromatographie nécessitent, une injection à l'aide d'une seringue. Les doigts sont alors au contact de la seringue pendant quelques dizaines de secondes.

LA DOSIMETRIE DES MANIPULATIONS.

Il est évident a priori que le film dosimètre traditionnellement porté au poignet n'est pas représentatif de l'irradiation subie au cours de ces opérations, qui intervient surtout au bout des doigts. Pour préciser cette irradiation, nous utilisons des dosimètres thermoluminescents constitués de plaquettes de téflon chargé de fluorure de lithium dont les caractéristiques sommaires sont les suivantes :
- épaisseur : 0,4 mm, longueur : 8 mm, largeur : 4,5 mm.
- proportions en masse : 40 % fluorure de lithium et 60 % téflon.
- enveloppe en polyéthylène de 7 mg/cm^2.
Ils sont étalonnés avec le rayonnement du cobalt-60 ; leur réponse relative à la dose absorbée dans les tissus mous, en fonction de l'énergie des photons de 10 keV à 1,3 MeV, est satisfaisante puisqu'ils ne présentent qu'une hypersensibilité de l'ordre de 15 % à 15 keV. En revanche, leur réponse pour le rayonnement β est inconnue. Afin de juger la validité des mesures de la dose absorbée au niveau des doigts, des mesures comparatives ont été effectuées à l'aide de dosimètres thermoluminescents d'une part, de stylodosimètres d'autre part. Les stylodosimètres utilisés sont du type SEQ.6 et SEQ.7 dont les parois ont des épaisseurs respectives de 300 mg/cm^2 et 7 mg/cm^2. Placés à une distance de 10 cm de la source, nous estimons qu'ils permettent d'effectuer une mesure correcte de la dose absorbée due aux photons et aux β sous les épaisseurs indiquées. Par ailleurs, la comparaison entre les réponses de stylodosimètres des deux types permet de déterminer les contributions respectives du rayonnement β, dégradé par le passage à travers la paroi des récipients et des photons d'annihilation. La comparaison entre la réponse des stylodosimètres SEQ.7 et des dosimètres thermoluminescents permet, d'autre part, d'effectuer l'étalonnage de ces derniers par le rayonnement mixte β et γ en cause.

Les résultats expérimentaux sont les suivants :
. pour les rayonnements en cause, la mesure de la dose absorbée, sous 7 mg/cm^2 à l'aide des dosimètres thermoluminescents, est correcte à 20 % près ;
. la contribution du rayonnement β à la dose absorbée sous 7 mg/cm^2 n'est que de l'ordre de 30 % de celle due aux photons. Ceci s'explique par l'atténuation de ce rayonnement β dans la paroi du récipient;
. le débit de dose sous 7 mg/cm^2 au contact des récipients est de l'ordre de 3 rad/h pour une activité de 1 mCi de carbone-11. Il est clair que des valeurs extrêmement différentes seraient obtenues, pour une même activité de carbone-11, en cas d'utilisation de récipients de taille et d'épaisseur de parois différentes.

IRRADIATION DU PERSONNEL.

Le débit de dose au contact des tubes de réaction est donc de l'ordre de 1000 rad/h au début des opérations. Une dosimétrie systé-

matique des extrêmités est effectuée à l'aide des dosimètres thermoluminescents décrits plus haut. Ceux-ci sont collés sur les parties des doigts entrant en contact avec les récipients contenant les solutions radioactives. Les mesures de routine se limitent à l'extrémité de l'index droit. L'activité mise en jeu pour les opérations de marquage est de l'ordre de 200 à 500 mCi au début des opérations et de 20 mCi à leur achèvement ; la dose absorbée est en moyenne de 1,5 homme-rem par manipulation à l'extrêmité des doigts. Les opérations se répétant avec une fréquence de 15 par mois, l'irradiation subie par les agents concernés aurait atteint la limite maximale annuelle de 60 rem en une dizaine de mois environ.

Cette situation a rendu nécessaire la mise au point d'une méthode de préparation limitant le plus possible les interventions manuelles. L'automatisation des opérations de routine, qui sont au nombre de 8 par semaine environ, limite l'irradiation du personnel à l'étape de sortie de la molécule marquée purifiée dans la seringue utilisée pour l'injection. Globalement, elle a permis de réduire la dose collective de moitié alors que l'activité manipulée était doublée. On peut donc considérer que les opérations de marquage des molécules utilisées pour le diagnostic sont maintenant exécutées dans des conditions de radioprotection satisfaisantes. La vigilance demeure grande toutefois pour les opérations de mise au point qui, elles ne peuvent pas être automatisées.

IRRADIATIONS DES EXTREMITES ET RECOMMANDATIONS DE LA CIPR.

Une comparaison a été effectuée entre les résultats fournis par le dosimètre "bout de doigt" et ceux obtenus à partir d'un dosimètre thermoluminescent au fluorure de lithium pur fritté placé dans le chaton d'une bague portée à la 2ème phalange de l'index droit, le chaton étant tourné vers l'intérieur de la main. Ce travail a montré que, pour le type de manipulation considéré, le rapport entre ces résultats était raisonnablement constant et égal à 2,5 ± 0,5. Etant donné les gros avantages pratiques présentés par le 2ème type de dosimètre par rapport au premier, il a été décidé de l'adopter pour le contrôle de routine. La valeur de la dose absorbée aux extrêmités est calculée en multipliant la valeur mesurée par le dosimètre bague par le coefficient 2,5. Nous retenons ainsi, pour la comptabilité individuelle des doses, la dose absorbée maximale au point de contact des doigts avec les sources radioactives. La surface de la peau concernée est de l'ordre de 1 cm^2 à l'extrêmité du pouce, de l'index et du majeur de chacune des mains.

Cette attitude a été guidée à l'époque par le souci de respecter les recommandations publiées par la C.I.P.R. en 1965. La publication 9, en effet, précise dans son paragraphe 28 que : "... en cas d'irradiation externe de la peau, en particulier quand la distance à la source est très faible ou quand la surface irradiée est très petite, il ne serait pas judicieux d'effectuer la moyenne de la dose reçue sur la peau entière. Il est recommandé d'effectuer une moyenne sur une surface de un centimètre carré dans la région recevant l'irradiation la plus importante".

Ces recommandations sont parfaitement claires et la procédure dosimétrique qui vient d'être décrite les suit rigoureusement. Là où le bât blesse, c'est qu'en 1977, la CIPR a publié de nouvelles recommandations, faisant l'objet de la très célèbre publication 26 et présentées comme une nouvelle philosophie de la radioprotection, dans lesquelles ce problème de l'irradiation externe partielle de la peau n'est pas abordé de façon aussi claire et précise.

Il faut insister sur le fait que ce problème des irradiations des extrémités n'est pas un problème académique mais un cas bien réel correspondant à la préoccupation de nombreux praticiens de la radioprotection. Une autre communication de notre Service (communication n° 1129) décrira les mesures effectuées avec d'autres radionucléides placés dans différents conteneurs dont les résultats vont dans le même sens que ceux publiés ici. On sait par ailleurs que les agents manipulant manuellement des échantillons d'uranium subissent des irradiations importantes aux extrémités ; enfin, l'utilisation de produits radiopharmaceutiques dans les laboratoires médicaux et les milieux hospitaliers tend à se répandre rapidement, entraînant un accroissement important du nombre d'opérateurs.

Il est donc fondamental pour la radioprotection de ces opérateurs qu'il soit répondu de façon aussi claire et précise que possible aux questions suivantes :

- qu'entend-on par "extrémité" ?
- à quel niveau doit être effectuée la mesure ?
- quelle est la surface de tissu à considérer pour la détermination de la dose ?
- comment cette dose doit-elle être comparée à la limite d'équivalent de dose ? Dans l'attente de la traduction dans les réglementations nationales des recommandations de la publication 26, doit-on retenir comme valeur limite l'équivalent de dose maximal admissible au niveau des extrémités ou au niveau de la peau étant donné que l'irradiation intéresse à peu près toujours la même surface de tissu ?

NUCLEAR MEDICINE

From somewhat primitive beginnings some 30 years ago (using open bench procedures and a spirit lamp to sterilise the air!) we have progressed to air-conditioned radiopharmaceutical clean rooms housing laminar flow cabinets, at both branches of our hospital (3). During early periods of low work-load our chief concern was contamination control. However, in the late 1960's our work-load increased sharply; in 1969 a total activity of 630 GBq (17 Ci) in a wide range of radiopharmaceuticals was administered to some 3000 patients. The accumulation of external dose by a small group of staff was then evident but still relatively small. The work-load has, however, continued to increase with the 1979 total of activity administered exceeding 3 TBq (80 Ci) chiefly Tc-99m in various forms, to some 10,000 patients for a wide range of diagnostic examinations. External dose accumulations by staff also continued to increase and by the mid-1970's the annual collective dose-equivalent for our nuclear medicine group (~20 staff - medical, radiography and nursing) had risen to some 40mSv.

In considering the problems of external dose, we note particularly the gain resulting from the use of more sensitive equipment (in reducing patient measurement time) and from attention to techniques of administration.

We have made detailed investigations into the control of dose to the hands (4), (5), (6).

RESEARCH

We have given much attention to procedures involving I-125, and have noted the problems reported elsewhere (7). In one pathology laboratory here some 20 members of staff are involved in labelling techniques, using up to 74 MBq (2 mCi) in each procedure and over 11 GBq (300 mCi) per year. A Class B laboratory with high grade fume cupboard is used for this work. Staff are instructed to self-monitor, including the thyroid, by contamination meter, and counter-check measurements are made periodically by Physics Staff using a thin crystal NaI detector. Under these conditions during two years we have only detected I-125 in about one third of these workers, at levels generally of 370 - 740 Bq (10 - 20 nCi).

One traumatic experience with tritium deserves mention. In spite of our centralised control on radionuclide acquisition, a member of our staff acquired, by an unusual method of requisition, an aqueous solution of an organic compound supposedly labelled at high specific activity with 92.5 GBq (2.5 Ci) T and supposedly free of labile tritium. In spite of our Instruction Manual, our local rules, our B-laboratory facility and our safety officer, an open bench evaporation of the solution was conducted. Later analysis showed that virtually none of the tritium had ever been associated with the compound. As a result of the evaporation procedure, the laboratory and its equipment were substantially contaminated and body burdens reached the ICRP investigation level in several staff. The decontamination procedures were lengthy and the interruption of important work programmes was severe.

SUMMARY

By giving steady attention to the design of facilities and the arrangement of procedures, and with an active personnel monitoring policy, relatively large scale radiation commitments within medical and research organisations can proceed with individual whole-body doses to staff being held well below 15mSv/annum. Such control generally relies heavily on the experience of a small group of staff within the organisation, who now face increased problematical administrative commitments from recent legislation.

In spite of detailed attention to control of radiation work, traumatic radiation incidents may still occur.

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EVALUATION DES DOSES INDIVIDUELLES ET COLLECTIVES RESULTANT DE REJETS EN RIVIERE

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Une évaluation des doses individuelles et collectives résultant de rejets en rivière à partir d'une installation située sur le Rhône, à 120 km de son embouchure en Méditerranée, a été effectuée. Les résultats sont présentés pour Co-60, Sr-90, Ru-106 et Cs-137 pour lesquels on suppose un rejet de 1 Bq s^{-1} pendant un an. Les voies de transfert étudiées sont :
- l'ingestion d'eau de boisson,
- l'ingestion de poissons,
- l'ingestion de produits alimentaires contaminés par irrigation à partir de l'eau du fleuve,
- l'irradiation externe à partir des sédiments situés sur le lit du fleuve et sur ses berges.

I - DISPERSION PHYSIQUE DANS LE FLEUVE

La dispersion dans le fleuve a été estimée à l'aide du modèle semi-empirique de SCHAEFFER (1). Les expressions mathématiques correspondant au modèle sont présentées par ailleurs (2) pour diverses situations. Nous nous contenterons ici de donner les principales hyptohèses et de définir les paramètres utilisés.

1-1. Concentration dans l'eau

Le modèle suppose la dilution instantanée de l'effluent dans tout le débit q du fleuve et une décroissance exponentielle de la concentration de l'eau en fonction de la distance du point de rejet. La constante de décroissance k est donnée par $k = k' + \lambda/w$ où k' représente l'effet des mécanismes d'appauvrissements physiques, chimiques, ou physico-chimiques, λ la constante de décroissance radioactive et w la vitesse de l'eau du fleuve. La concentration de l'eau se compose de l'activité en phase soluble et de l'activité retenue sur les matières en suspension présentes dans l'eau. Ces deux quantités sont reliées entre elles par le coefficient de distribution K_d en eau douce de l'élément considéré et dépendant de la masse de matières en suspension par unité de volume d'eau.

1-1. Concentration dans les sédiments

Les sédiments non consolidés se comportent comme un fluide et "s'écoulent" sous l'influence de la gravité et de la force tractrice qu'exerce l'eau sur la couche super-

ficielle. On considère une vitesse moyenne v de déplacement d'une couche d'une trentaine de centimètres d'épaisseur sur la largeur L du fleuve.

1-3. Détermination des valeurs des paramètres

Le modèle présenté ci-dessus a été appliqué au cas du Rhône pour une installation située à 120 km de son embouchure en Méditerranée. Comme les caractéristiques physiques du fleuve varient le long de ce trajet de 120 km, il a été procédé à un découpage en trois tronçons sur lesquels on suppose que les paramètres utilisés ont des valeurs constantes. Les valeurs sont présentées à la partie a) du tableau 1, tandis que la partie b) donne les valeurs retenues de paramètres qui sont caractéristiques du radionucléide considéré mais que l'on admet être indépendants du tronçon de fleuve étudié.

Les valeurs adoptées résultent de mesures directes pour tous les paramètres à l'exception de deux d'entre eux, qui sont la vitesse de déplacement des sédiments et le coefficient de fixation k'. Les valeurs de ces deux paramètres ont été obtenues par la recherche du meilleur accord entre les concentrations calculées et les concentrations mesurées pour quatre points du fleuve, situés à environ 10, 25, 55 et 65 km en aval du point de rejet, et pour six années (de 1974 à 1979 inclus). Le tableau 2 présente, dans le cas de Cs-137, les rapports des concentrations calculées à l'aide des valeurs des paramètres indiquées au tableau 1 aux concentrations mesurées. Un bon accord général peut être constaté bien que ces résultats aient été obtenus à l'aide d'approximations grossières. En effet, le modèle calcule des valeurs moyennes sur l'année et il n'est pas du tout certain que les mesures, effectuées trimestriellement ou avec une périodicité plus longue, soient représentatives de ces valeurs moyennes en raison des variations importantes du débit du fleuve. Par ailleurs, les points auxquels les concentrations sont calculées ne correspondent pas toujours de manière précise aux points où les prélèvements ont été faits.

2 - EVALUATION DES DOSES INDIVIDUELLES ET COLLECTIVES

Le modèle de dispersion physique décrit ci-dessus sert de point de départ à l'évaluation des doses correspondant aux voies de transfert retenues. La méthodologie utilisée pour étudier ces voies de transfert, est exposée en détail par ailleurs (2), a été appliquée à des rejets dans le fleuve de Co-60, Sr-90, Ru-106 et Cs-137 au taux unitaire de 1 Bq s^{-1} pendant un an.

Les tableaux 3 et 4 montrent les résultats des évaluations des engagements d'équivalent de dose effectif individuel et collectif. La comparaison des résultats relatifs aux diverses voies de transfert montre que, avec les hypothèses

utilisées, l'ingestion de légumes verts conduit aux doses collectives les plus élevées tandis que c'est l'irradiation externe qui est en général la voie de transfert la plus importante sur le plan des doses individuelles.

Ces résultats ne peuvent pas être généralisés à tous les fleuves ni à l'ensemble des radionucléides car ils dépendent de manière étroite de nombreux paramètres parmi lesquels figurent l'utilisation des eaux et des sédiments du fleuve ainsi que le mode de vie des populations exposées.

TABLEAU 1 - Valeurs retenues des paramètres utilisés dans le modèle de dispersion

a) paramètres dépendant des sections du fleuve Section	(0-35 km)	(35-80 km)	(80-120 km)
Débit du fleuve q ($m^3\ s^{-1}$)	1540	1700	1700
Vitesse de l'eau w ($m\ s^{-1}$)	0,78	0,74	0,70
Matières en suspension M ($t\ m^{-3}$)	4.10^{-5}	$4,5.10^{-5}$	5.10^{-5}
Vitesse des sédiments v ($m\ s^{-1}$)	$4,9.10^{-4}$	$3,9.10^{-4}$	$2,8.10^{-4}$

b) paramètres indépendants des sections du fleuve	Co-60	Sr-90	Ru-106	Cs-137
Constante radioactive λ (s^{-1})	$4,15.10^{-9}$	$7,78,10^{-10}$	$2,17.10^{-8}$	$7,29.10^{-10}$
Coefficient de distribution Kd ($t\ m^{-3}$)	30 000	2 400	37 000	27 000
Coefficient de fixation k' (m^{-1})	1.10^{-3}	2.10^{-5}	1.10^{-4}	1.10^{-5}

Tableau 2 - Rapports des concentrations calculées aux concentrations mesurées de Cs-137 dans les sédiments pour plusieurs années et plusieurs points en aval du point de rejet

Année	Distance en aval du point de rejet (km)			
	5	35	55	65
1974	-	9	1,0	15
1975	1,3	2,2	1,3	3,1
1976	0,4	0,7	1,1	1,3
1977	0,3	0,9	1,6	6,1
1978	0,5	1,5	0,7	2,2
1979	0,3	0,3	0,4	2,6

Tableau 3 - Engagements d'équivalent de dose effectif collectif (homme Sv) correspondant à un rejet de 1 Bq s^{-1} pendant un an

Radionucléide	Engagement d'équivalent de dose effectif collectif (homme Sv)			
	Eau de boisson	Poisson	Irrigation de légumes verts	Irradiation externe
Co-60	1.10^{-7}	6.10^{-8}	4.10^{-7}	3.10^{-7}
Sr-90	4.10^{-6}	2.10^{-7}	3.10^{-4}	0
Ru-106	3.10^{-7}	4.10^{-9}	1.10^{-6}	8.10^{-10}
Cs-137	8.10^{-7}	1.10^{-6}	8.10^{-6}	1.10^{-7}

Tableau 4 - Engagements d'équivalent de dose effectif au groupe critique (Sv) correspondant à un rejet de 1 Bq s^{-1} pendant un an

Radionucléide	Engagement d'équivalent de dose effectif (Sv)			
	Eau de boisson	Poisson	Irrigation de légumes verts	Irradiation externe
Co-60	2.10^{-13}	8.10^{-13}	5.10^{-14}	2.10^{-10}
Sr-90	8.10^{-12}	2.10^{-12}	3.10^{-11}	0
Ru-106	5.10^{-13}	6.10^{-14}	1.10^{-13}	9.10^{-13}
Cs-137	2.10^{-12}	2.10^{-11}	8.10^{-13}	1.10^{-10}

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MODELISATION DES ECHANGES DE PRODUITS ALIMENTAIRES EN VUE DE L'EVALUATION DES DOSES COLLECTIVES

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RESUME

A partir d'informations statistiques sur les productions, transformations et échanges, des modèles sont construits en vue d'établir les relations entre la contamination initiale des produits et celle des aliments consommés, et d'évaluer les doses collectives et leur répartition dans diverses hypothèses de contamination du milieu ambiant.

INTRODUCTION

La prévision des contaminations potentielles des produits livrés à la consommation humaine, en cas de rejets normaux ou accidentels dans le milieu ambiant, est l'un des éléments d'évaluation des doses collectives et individuelles. Connaître le cheminement et la destination des produits afin d'établir les schémas de répartition des doses résultant de contaminations éventuelles plus ou moins intenses et étendues n'est pas chose aisée, ainsi qu'on l'a précédemment exposé à propos des cas du lait de consommation [1] et des céréales [2] [3] . Aux problèmes de méthodologie générale s'ajoutent ceux particuliers à chaque catégorie de produits. Les résultats très brièvement présentés ici concernent les produits de la mer et les produits laitiers.

PRODUITS DE LA MER [4] :

On a rassemblé les informations concernant les zones de pêche, les quantités pêchées en chacune d'elles par les pêcheurs français, la destination des prises, les agents intervenant dans la production et la distribution des produits de la mer. Une étude des transports et des échanges extérieurs a complété la précédente.

Un modèle d'échanges a été établi sur ces bases, tant pour les produits consommés en frais que pour les conserves et surgelés. Il permet de connaître l'origine, selon les zones de pêche, des quantités (pêchées ou importées) disponibles pour la consommation dans les différentes régions françaises. Il permet également d'estimer les consommations (totales et moyennes individuelles) par régions (exemple tableau 1).

TABLEAU 1. Produits de la mer - Estimation des consommations dans une région donnée, en 1976 : a/ par région, en tonnes, b/ par personne, en kg

Régions	Poisson frais salé, fumé		Crustacés, coquillages		Conserves de poisson		Surgelés, congelés	
	(a)	(b)	(a)	(b)	(a)	(b)	(a)	(b)
1.NORD	17925	4,54	10343	2,62	10007	2,53	3964	1
2.PICARDIE	8845	5,22	3459	2,04	4459	2,63	1768	1,04
3.REGION PAR.	58070	5,83	49324	4,95	32981	3,31	13070	1,31
4.CENTRE	11320	5,22	4426	2,04	5707	2,63	2263	1,04
5.HAUTE NOR.	8412	5,22	3289	2,04	4241	2,63	1682	1,04
6.BAS.NORM.	6873	5,22	2688	2,04	3465	2,63	1374	1,04
7.BRETAGNE	17854	6,82	20472	7,82	7225	2,76	2862	1 09
8.LOIRE	19043	6,82	21829	7,82	7708	2,76	3053	1 09
9.POITOU	10503	6,82	12043	7,82	4251	2,76	1684	1,09
10.LIMOUSIN	3605	4,82	2019	2,70	1787	2,39	709	0,09
11.AQUITAINE	12390	4,82	6950	2,70	6154	2,39	2438	0,9
12.MIDI PYR.	11012	4,82	6169	2,70	5461	2,39	2167	0,9
13.CHAMPAGNE	7040	5,22	2759	2,04	3554	2,63	1407	1,04
14.LORRAINE	7630	3,25	3027	1,29	6492	2,77	2572	1,1
15.ALSACE	4992	3,25	1978	1,29	4246	2,77	1681	1,1
16.FR.COMTE	3478	3,25	1381	1,29	2965	2,77	1173	1,1
17.BOURGOGNE	8287	5,22	3240	2,04	4178	2,63	1657	1,04
18.AUVERGNE	4408	3,28	1829	1,36	3633	2,70	1441	1,1
19.RHONE	15856	3,28	6584	1,36	13063	2,70	5184	1,1
20.LANGUEDOC	10932	6,06	5274	2,92	5596	3,32	2377	1,3
21.COTE D'AZ.	22475	6,06	10829	2,92	12325	3,32	4882	1,3
TOTAL	270950		179912		149498		59448	
MOYENNE		5,11		3,39		2,84		1,42

Ce modèle d'échanges susceptible d'être généralisé au niveau européen avec les données adéquates permet d'établir la relation entre les niveaux de contamination des zones de pêche et les niveaux de contamination au stade de la consommation en différentes régions.

On peut en déduire les activités ingérées. La figure 1 donne, par exemple, l'activité ingérée par tête en trois régions (Bretagne, Région Parisienne, Pyrénées) en fonction de répartitions hypothétiques dans les zones de pêche d'une contamination de l'eau de mer par le césium 137. Les zones concernées sont, d'une part, l'ensemble Mer du Nord, Manche Est et Ouest et Canal de Bristol, d'autre part, la Mer d'Irlande, les niveaux dans les autres zones, inspirés de résultats d'observations, demeurant faibles et inchangés. Les différences de consommation dans les diverses régions ne suffisent pas à expliquer les différences qui sont constatées entre les activités ingérées. Le choix des lieux de pêche et les caractéristiques de la distribution interviennent aussi de manière sensible.

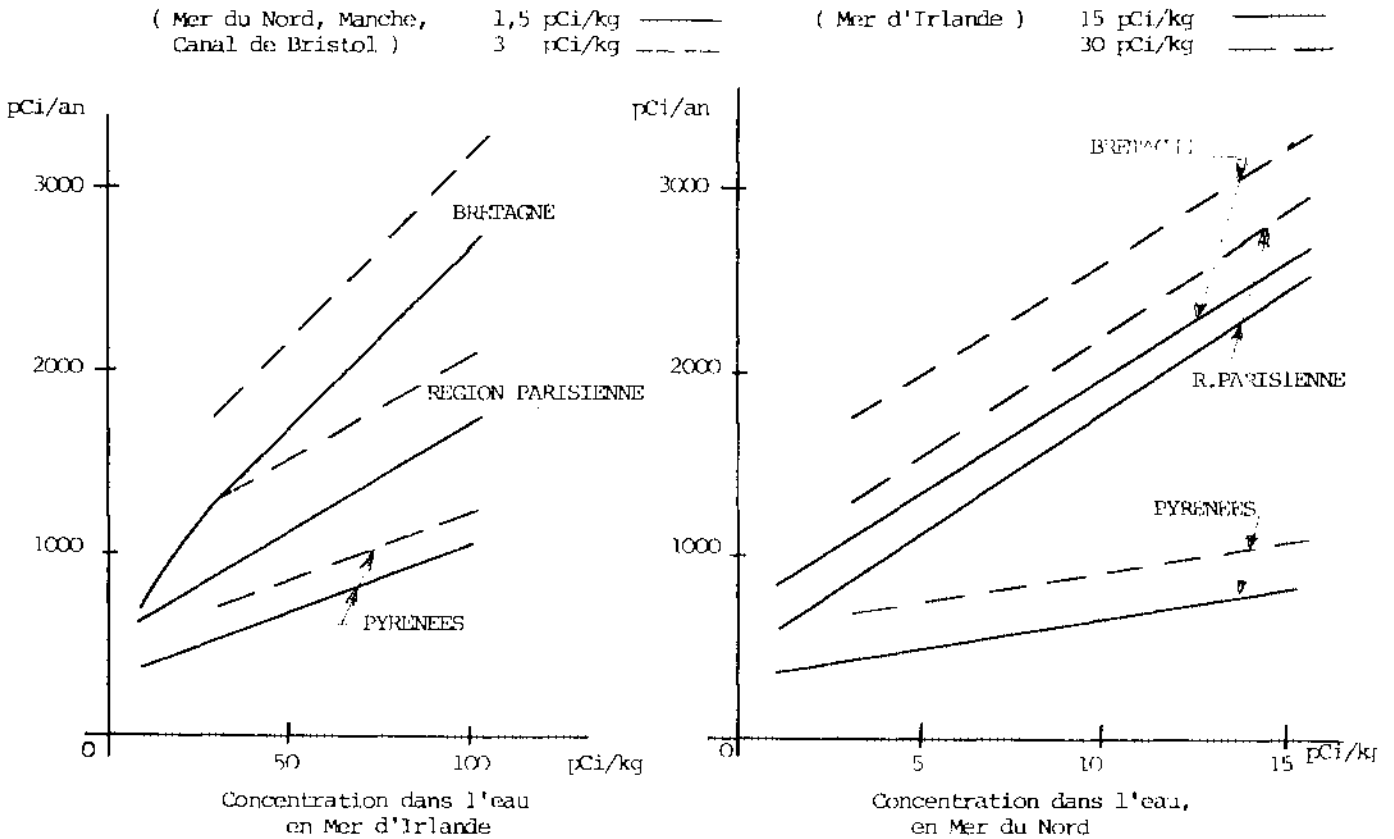

Fig.1: Activité ingérée par tête en fonction de la contamination des zones de pêche (^{137}Cs)

PRODUITS LAITIERS [5] :

On a recueilli à l'échelle européenne et pour plusieurs années consécutives les données concernant la production, la collecte et la structure de distribution. Des regroupements de produits ont été imposés tant par le nombre et la variété des produits que par le but de l'étude : l'accent a été mis sur l'utilisation du lait de chaque région ou pays en tant que produit frais, produit transformé, ou produit non destiné à l'alimentation humaine. Les disparités observées entre pays sont importantes. Ainsi, pour le Marché Commun, et pour les années 1973-1976, la proportion de lait entier collecté, par rapport à la production, a été de 80% en moyenne (73 à 95,6% selon les pays), et son utilisation en produits frais a été de 25,7% en moyenne (11,7 à 65,7% selon les pays), les utilisations correspondantes en beurre et fromage ont été respectivement de 44,9% (12,6 à 63) et 22,2% (5,8 à 51,2). Le lait écrémé obtenu est surtout utilisé pour les conserves : 57,5% en moyenne (26 à 78,5). De semblables disparités sont observées au niveau de la consommation des différents produits (celle du lait varie en 1976 de 71 kg/tête à 206 kg/tête parmi les neuf pays de la Communauté Européenne). Une analyse détaillée a permis d'estimer les échanges entre les régions françaises et allemandes et les pays de la Communauté Européenne et de connaître l'origine et les quantités de produits frais consommés en chaque région. Les tableaux d'échange mettent généralement en évidence la prédominance d'utilisation de la production régionale de cette catégorie de produits. Par contre, les échanges sont importants pour les produits transformés (fromage et poudre) et ils concernent essentiellement les régions fortement urbanisées.

L'effet des échanges sur les activités ingérées totales par an et par tête n'est donc pas négligeable. Il est supérieur à 10% pour la moitié des régions d'Europe dans l'hypothèse d'une contamination à la production variant relativement peu (de l'ordre d'un facteur 5 entre les extrêmes). Mais dans l'hypothèse de contamination très élevée d'une région, le schéma de répartition des activités ingérées par tête est fortement influencé par la structure des échanges.

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ON THE CHARACTERIZATION OF RISK

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This paper analyses the concept of Risk, proposes a general definition of Risk in mathematical terms,and derives some practicable ways to describe the risk adequately for use in risk assessments.

THE DEFINITION OF RISK

There is nothing new in the concept of Risk, though its quantitative assessment in connection with the use of new technologies is quite recent. Correspondingly, qualitative definitions abound and can be found in any dictionary. However, there is no accepted technical definition of Risk in the literature on the risk assessment of technologies (to be distinguished from its use in insurance). In fact, one can recognize in the literature a search for a technically adequate definition of the term. By "adequate" it is meant that the definition would be of a sufficiently general nature, and that it would be exact in the mathematical sense, so that quantitative practical description of the risk could be derived for any case of interest.

Qualitative definitions

Let us first examine two qualitative definitions of Risk in order to extract the conceptual elements of the term. A dictionary (9) defines *Risk as exposure to the chance of injury or loss*. A modern text on Risk, with a strong philosophical bias (7), gives the definition *Risk is the potential for realization of unwanted, negative consequences of an event*. The second definition differs from the first mainly by explicitly limiting the consideration to the results of an event, the limitation being necessary for risk analysis of specific systems or accidents.

From the definition it is seen that risk has three conceptual elements: *An event*, *Harmful consequences*, and *Probability*. The harmful consequence can be an immediate result of the event, but it can follow a development of the event, which can be described as a chain of events (Fig. 1). For example, a pipe-break in a water-cooled nuclear reactor can be followed by loss of coolant, damage to fuel elements, contamination of the coolant, escape of radioactivity, irradiation of people and contamination of buildings, and finally by impairment of the health of a few of these people, loss of property, and probably other damage as well. The event that initiated the accident can be unique, it could be an earthquake in our example, but the choice of the event whose risk we consider is arbitrary to a large extent. We may choose the initiating event, but also any of the

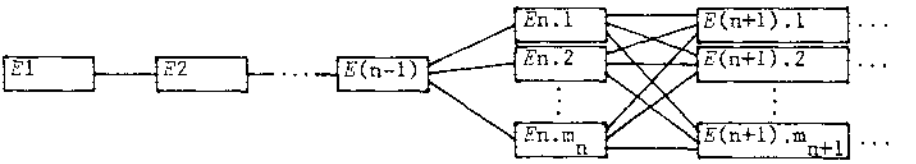

Figure 1. The development of event *E*1, a schematic representation.

following or even preceding events, like the loss of coolant or escape of radioactivity in our example, or the existence of a nuclear power plant. Similar arbitrariness exists in the choice of the conequence, which, in the example, could be the final results, but also the irradiation and contamination, the escape of radioactivity, or even the pipe-break. the last examples of a consequence shows that there is no inherent distinction between an event and a consequence. An event can be regarded as *the* event in one analysis, and as a consequence in another.

The probability of a consequence is combined of the conditional probability of the consequence given that the event has happened, and the probability of the event. However, limiting the definition of Risk to that of a specific event, it would be convenient to take the conditional probability as *the* probability, bearing in mind that it is conditional and depends on the event.

Technical definitions

In common usage, *Risk* is thought of as *the probability of an undesirable occurence* (4), and this definition is taken for granted in many discussions on the risks of technologies (e.g. 6). This definition is adequate for one-consequence events, but fails in the case of several-consequence events. Another common definition is that *Risk is the product of the frequency* (or probability per unit time) *of an event and the magnitude of its harm* (10). Like the former, this definition strictly applies only to one-consequence events, but it also involves the idea of proportionality (i.e. that changing the probability is equivalent to changing the magnitude by the same factor). The risk is given as the expectation of the consequence, and it can easily be generalized to events that can have many consequences of different magnitudes but of the same kind, by defining *Risk* as *the expectation of the consequence*. This can be expressed in many ways, like *the probability per unit time of the occurence of a unit cost burden* (8).

It is evidently easy to farther generalize the definition of Risk to a linear (weighted) combination of the expectations of consequences of different kinds, but there remain two major objections even to this definition. The first is an almost universal feeling that the idea of proportionality, on which any linear expectation definition is based, is unacceptable (e.g. 5,7,10). For very frequent and relatively minor events, which occur several times every year, the expectation of the consequence, preferably accompanied by a

measure of its variability like the standard deviation, is quite satisfactory. This can apply to small industrial accidents, side-effects of medical treatments and the like, from societal (but not the personal) point of view. In connection to the nuclear industry it applies to small leaks and routine releases of radioactivity. However, risk assessment of modern technologies is often concerned with major accidents, and there , 1000 deaths with a probability of 1:1000 per year would not seem equivalent, to most people, to one death every year.

The second objection is that there are no accepted ways to compare harms, injuries or damages of different kinds, like deaths and illnesses, or ruin of property and ruin of archeological sites. The last objection would hold against any attempt to characterize the risk by one number, even a non-linear function of the consequences and probabilities. It can be hoped that one day such a function would be agreed upon (5), but attempts to establish one resulted in differing and contradicting results (3,7).

For these reasons Okrent (3) defined *Risk* as *probability and consequence*, meaning that the whole probability distribution function is needed to characterize the risk. Indeed, notwithstanding any formal definition, modern risk assessments presented their risk estimates as probability, or frequency distribution functions of the magnitude of the consequence (2,10), as was originaly suggested by Farmer (1). This definition is still unsatisfactory. A few variables (e.g. deaths and illnesses), however useful, would not characterize the risk completely. Two types of risk emerge to be relevant, individual or personal risk and societal or national risk (2), and the relationship between them is not clear. An even more general definition is needed.

It is proposed here to define *the Risk of an event* as *the probability space on the space of possible harmful consequences* (to which the no-harm event has been added). The terms of this definition is taken from mathematical probability theory, appart from the term *harmful consequence* which replaces the term *event* in probability. The definition, though abstract and not directly usable for the description of risk, puts at our disposal the conceptual and manipulational mechanisms of probability theory.

THE REPRESENTATION OF RISK

One outcome of the definition as a probability space is the possibility to define random variables in it, that is numerial functions on the space of consequences. The joint probability distribution, or density function (in the generalized function sense) induced by the space on a set of any harm-measuring functions, shall be called *a representation* of the Risk. The above-mentioned risk assessments (2,10) presented estimates of some representations of the risk they assessed.

In choosing a representation of the risk of an event, there are two levels of freedom or arbitrariness. The first is in choosing the space, that is in deciding what events should be regarded as the conequences. The second is in choosing the representation or representations, that is deciding on relevant and practical variables that would characterize the consequences. In the example of nuclear

reactor accidents, one can choose the consequence space of radiation-doses and contamination, or the space of health effects and property loss. In each space one can choose first the receptor of the risk to be considered, and than the parameters that describe the consequence from the viewpoint of that receptor. Two such viewpoints are the individual and the societal, which lead to individual and societal representations, but there can be many other intermediate, viewpoints. The risk to a local community or to an industrial company, for example, would lead to corresponding communal and company representations. However, since any individual is also a citizen of his country, a member of his community and so on, a representation from one viewpoint would not suffice for proper evaluation of the risk.

Calculating the probability distribution of a set of parameters, it should be remembered that in any one possible outcome of the event, many different consequences happen together. In the example of a nuclear reactor accident, a certain number of people would have immediate radiation illnesses, others would develop late cancers, et cetera (in health-effects space from societal viewpoint); or a certain person would receive different radiation-doses to each of his organs (in radiation-dose space from the individual viewpoint). For this reason the joint probability distribution is needed for representation of the risk. For proper evaluation of the risk in risk assessments, this statistical interdependence of the parameters of a representation should be described at least by the correlation between the parameters.

The choice of meaningful representations and their description should be one of the main concerns of risk assessments.

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STOCHASTIC AND NON-STOCHASTIC EFFECTS . A CONCEPTUAL ANALYSIS .

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Stochastic and non-stochastic effects are defined on the basis of descriptive laws relating dose to the appearance of effects .

Stochastic effects appear randomly in exposed populations and the severity of effects does not depend on dose exposure . The incidence of effects increases with dose .

On the other hand , non-stochastic effects are necessary and all individuals are affected provided the dose exposure reaches a certain threshold . The severity of effects depends on dose . The higher the dose the more intense the effect . Carcinogenesis, increased mutation rate , chromosomal aberrations and congenital anomalities represent stochastic effects . Aplastic anaemia , myelofibrosis , skin lesions, processes involving the destruction and secondary fibrosis of specific tissues or organs represent non-stochastic effects .

The theory suggests that the effects of ionising radiation can be classified in two separate and mutually exclusive classes , stochastic and non-stochastic effects . In addition it suggests that there are two types of events and two types of dose-effect relationship which correspond prima facie to two different processes . These two distinct processes are described by different natural laws as shown by the presence or absence of a threshold , necessary vs probable effects, quantitative vs quantal outcome . The terminology underlines a fundamental conceptual difference between these two processes . In the first case the law which describes them supposes a natural necessity and in the second case a probability law . The theory suggests that stochastic and non-stochastic effects represent a partition of the class of effects into two mutually exclusive subclasses .

1. On the necessity of effects . *Are there such things as necessary effects ?*

Philosophy of science shows there are no necessary empirical relations in nature . If a and b are two events (e.g. a,exposure to radiation or to a toxic agent and b,its effect)either a differs from b , in which case a can happen in the absence of b , or the two events are a single and unique event in which case b necessarily follows a . In other words if two events are bound by a law of necessity the two events are identical .

Let us suppose that event a , swallowing a lethal dose of potassium cyanide , occurs at time t_1 , and that event b be the death of the person . As long as there is a time interval (t_2-t_1) between the two events , it is possible that the second event will not occur because of the possibility of intercurrent events or interventions (the subject may vomit or else the repletion of his G I tract may slow down absorption..) . One can object that this example is misleading . Event a should be defined as the swallowing of cyanide and its passage through the wall of the G I tract into the blood . But this will not do either , since the subject might be receiving some

sort of antidote . The only way to establish a necessary connection between a and b would be to include"cerebral death" in the definition of a ; a would be a lethal dose of cyanide which causes death so that the dose-effect relationship becomes necessary . However a is now no more distinct from b and the necessity which binds outcome to exposure is definitional and no more empirical or contingent . This discussion illustrates a well-known principle of logic : necessity is of semantic nature . Natural laws describe regularities of nature and never express necessary connections but only contingent ones .

2. On the complementarity of stochastic and non-stochastic effects . *Are non-stochastic effects the residue left after substracting stochastic effects from the class of effects ?*

If the terminology is correct , the answer is yes . But is it correct ?

Actually stochastic and non-stochastic effects are not two different empirical or biological processes but rather two ways of looking at reality which means they mainly differ in the conceptual approach .

In the case of stochastic effects the exposed population (rats, cells, persons ..) is defined by its exposure and the observed outcomes are analysed according to dose . A *dose-response* or *dose-per cent curve* is obtained which is a qualitative relation between dose and percentage of response . In the case of non-stochastic effects the exposed population instead of being defined by exposure is defined by outcome and one obtains a *dose-effect relationship* . The outcome in the second case is quantitative while in the first case , percent of response is quantal or qualitative . Radiodermatitis can be described as a quantitative effect of ionising radiation and in this case it represents a non-stochastic effect . If on the other hand one considers the dose-response of a population of skin cells it is possible to describe the radiodermatitis as a stochastic effect with the recruitment of an increasing number of cells with increasing radiation dose .

High thresholds are needed to produce so-called non-stochastic effects since a certain percent of response is needed to obtain a phenomenological , i.e. clinical , skin erythema, a cataract or a tissue fibrosis .

Conclusion

1. It is argued that stochastic and non-stochastic effects are not complementary i. e. that they do not represent a partition of the class of radiation effects .

2. Stochastic and non-stochastic effects represent different ways of describing outcomes and the "non-stochastic" or necessary character of some effects is of semantic nature .

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RISK ESTIMATES OF STOCHASTIC EFFECTS DUE TO EXPOSURE TO RADIATION - A STOCHASTIC HARM INDEX

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I. The ICRP system of dose limitations for radiation workers is based on the following requirement: The probability for the induction of fatal malignancies due to exposure to radiation should be limited to a rate not exceeding the occupational fatality rates in industries generally recognised as safe, i.e., to a rate less than 10^{-4} cases per year, per person . Exposures to small "critical" groups of the population are limited by the use of a similar approach with a limiting risk rate lower than the above by an order of magnitude or more.

This risk rate is generally comparable in Western societies with the risk of immediate death from any of the following causes: accidental drowning, poisoning, death in an air-crash or electrocution and is lower than the societal risk of being murdered. Exposures of the population at large should be evaluated by risk benefit analysis, attributing a proper monetary value to a man-rem ($1000 in the U.S.) or - indirectly - to a statistical life.

It is important to emphasise that these upper limits of risk levels for late effects (assumed to be socially acceptable) are established by comparison and adoption of actual immediate death rates experienced and passively accepted by Western societies from comparable causes.

Truly, the validity of the linking working assumption between the exposures and their late effects, the linear dose-effect relationship for low level low rate exposures is more and more questioned. According to the generally evolving opinion the model overestimates the effects for the considered range of exposures, but this matter is beyond our consideration here.

The significant difference between immediate and late fatality is noted in ICRP Publication 26, but is not explored there in detail. Unfortunately in many presentations and evaluations no differentiation is made between the two.

II. As every living creature is bound to die, the only open question in this respect is the timing, i.e. the age at which the death occurs - in other words, the life span or life expectation. The effects of exposure to low level radiation on the survival probability and life expectation have been investigated, using the LDER assumption, the risk rate per man-rem adopted by the ICRP and the competing risk technique.

The 1977 vital statistics of Jewish males in Israel have been used as baseline, mainly the data on normalized survival probability and life expectation as functions of age. On these data, assumed effects of exposure have been superposed and the net differences calculated. The late effects of exposure have been assumed to be 10^{-4} deaths/man-rem in a flat distribution between 2.5 to 27.5 years

after the exposure, i.e. 4×10^{-6} deaths/man-rem,yr during the period in which a given dose burden is considered as effective. It was further assumed that the effects of an effective dose burden are lost with death, including those from competing causes. In the mathematical treatment the accumulated effective dose burden carried over from one to the next period has been reduced in proportion to the survival probability at the given age. For the purpose of calculations single exposures of 10 rem at different ages and annual exposures of 5 rem/yr and 0.5 rem/yr from age 20 to 65 have been taken as examples.

Fig. 1 shows the base data, i.e., the mortality rate and the additional life expectancy for unexposed persons as a function of age, for Jewish males in Israel, in 1977.

Fig. 2 shows the number of excess deaths resulting from single exposures of 10 rem of 10^4 persons as function of age and the age at exposure under the quoted assumptions.

Fig. 3 shows the loss of life expectancy due to a single exposure (in days per man-rem) as a function of age at exposure.

The following conclusions were reached:

(a) The detrimental effects of exposure are proportional to the assumed risk rate and the exposure;

(b) The number of excess fatalities over unexposed population is calculated to be less at any time than the integral dose (man-rems) times the risk factor. The maximal excess occurs 18-25 yrs after the exposure and its value decreases sharply with the age at exposure (from $\sim 0.95 \times 10^{-4}$ excess dead/man-rem for exposure at 18 to $\sim 0.4 \times 10^{-4}$ at 60).

(c) The loss of life expectancy per unit exposure varies even more strongly with the age at exposure from approx. 1.2 days/man-rem at 18 to about 0.16 days/man-rem at 60.

(d) The initial life expectancy of the Israeli male (72 yrs.) is higher than of males in many other countries. The detriments of exposure for populations with lower life expectancy are smaller than the quoted values.

(e) Calculations for exposures of constant rates (18-65) within the MPD showed that the maximum number of excess fatalities occurs at the age of 65 and its value is approximately 0.6×10^{-4} /man-rem, while the averaged loss of life expectancy due to exposure is about 0.55 days/man-rem.

The presently available details of the vital statistics in Israel are not sufficiently accurate to verify such small differences. Many probably important factors, as will be noted in the following, are missing and are not available for correlations. These factors are relevant also to epidemiological studies of the late effects of other potentially harmful agents.

III. It was found that mortality among the employees of the IAEC was significantly lower (by approx. $40 \div 50\%$) than the statistically expected value according to their age distribution and the corresponding death rates in Israel. Possibly several factors contribute to this, including:

(a) The high level of occupational safety maintained within IAEC.

(b) The exceptional medical care given to the employees.

(c) Employee selection procedures including preemployment health examination.

(d) Some cases of early retirement due to failing health.

(e) The somewhat different cultural, social and ethnic structure of the group from the average of the Israeli society.

(f) The higher than average income of the group resulting in generally above average living standards and conditions.

The exploration of such factors could provide valuable insights.

Calculation shows that even a marginal reduction of the natural death rates among radiation workers is sufficient to balance the assumed harmful stochastic effects of low level exposures. Fig. 4 shows the required reduction of death rate to balance such effects due to continuous exposure at a rate of 1 rem/year. For a typical age distribution of workers less than 1 lifesaving per 10^4 workers per year is the breakeven value. According to the available crude information probably more than this is achieved.

If we attribute a modest 10 percent of the previously mentioned reduction of the natural death rates among the IAEC employees to the instant availability of medical aid and to the existence of particular medical care which includes periodic medical and laboratory examinations (general and bioassays), early detection and treatment of health irregularities and diseases, we find that this service compensates several-fold for any possible harm from exposure to ionizing radiation.

Similar statement can be made with regard to the effects of maintaining high-level occupational safety, although here the gain is probably smaller.

A corresponding phenomenon can be revealed from vital statistics data for the past three decades in Israel. There was a significant improvement in the medical care, partly due to the developments in the medical science and profession and partly due to organizational improvements. These improvements have been accompanied by a corresponding decrease in the death rates and increase of life expectancy. In parallel the relative and absolute incidence of malignant neoplasms increased, mainly on the account of more easily curable diseases.

IV. In this context it is worthwhile to note that as malignant cancers are the cause of ~20% of the deaths, every life saved from death due to other causes will generate in the future ~0.2 additional cancer deaths. If studied detached from other data of vital statistics, every improvement in the general health care will virtually worsen the cancer statistics. If the excess cancer cases are attributed to radiation effects, then, using the ICRP assumptions, every life saved could imply the stochastic effects of ~2000 man-rems. If the establishment of a nuclear installation is accompanied by improved material well-being of the neighbouring population, leading among other changes to better medical care and the resulting changes in causes of death, this could falsely imply the existence of stochastic radiation effects from very low level exposures or exaggerate their magnitude.

CONCLUSIONS

1. The realistic late effects of exposure to radiation are generally less than calculated by multiplying the collective dose by the risk factor. The effects are strongly age-dependent, decreasing sharply with the age at a single exposure and with the starting age of continuous exposure. A good estimate of the assumed harm is an age-

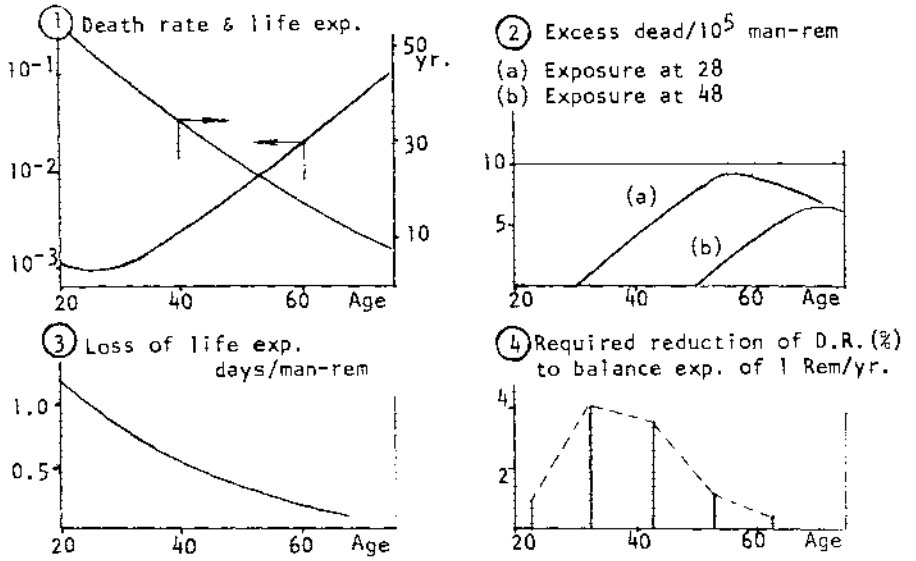
dependent function of expected life shortening per unit exposure. The values of this function may range from 1.25 to 0.15 days/man-rem.

II. A possibility to account for this age dependent effect in radiation protection could be the use of a stochastic harm index, defined as the product of the effective dose equivalent index and a weighing factor derived from the above function. This weighing factor can be either an age dependent normalizing factor or the estimated lost life expectancy in terms of days/rem. In the second case the new index has the dimensions of time. Such a magnitude is easy to comprehend, to integrate over years or populations and easy to use in comparative studies of different risks.

III. The assumed harm due to exposure to radiation can be more than balanced by improvements in medical care, industrial and home safety, increase in the standard of living, etc. The assessment of the health and safety of exposed populations (for example, radiation workers) should be based on total balances and not on the segregated exploration and use of single factors.

IV. Cancer-statistics detached from the general vital statistics is misleading when used for the establishment of risk factors for exposure on epidemiological basis. The accuracy and the information recorded in vital statistics should be improved to allow its proper use in the epidemiological evaluations of stochastic effects of radiation or other agents.

V. Medical follow-up of exposed populations is intrinsically accompanied by improved medical care and affects the results in the direction of a decreased mortality but an increased cancer incidence. These effects should be accounted for.



MEDICAL IRRADIATION AND THE USE OF THE "EFFECTIVE DOSE EQUIVALENT" CONCEPT

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The Effective Dose Equivalent Concept defined by the ICRP (5,6) considers the total risk from occupational irradiation including both hereditary and somatic effects. In the field of medical irradiation there is a tradition to report the genetically significant dose equivalent as an index of harm to the population to this type of irradiation although it only includes the geretical effects.

This paper will deal with the definition of a similar concept for somatic effects which added to the GSD will better approach an index of total harm to the population from medical irradiation.

By applying the linear-dose effect relation one can transform the weighting factors for total risks recommended by ICRP to weighting factors for somatic effects. Another weighting factor M_S has to be used which takes into account the dependence of the latence period for occurence of malignant disease and the variation of the incidence with time. On this basis a somatically significant dose equivalent can be defined as "the dose equivalent which if received by every member of the population, would be expected to produce the same total somatic injury to the population as does the actual dose equivalent received by the various individuals". In the first approximation it can be written as

$$SSD = \sum_{g,j,k} \left(\frac{N(g,j,k)}{N_{tot}}\right) \sum_{i} M_S(g,i,j,k) \cdot \omega_S(g,i,j,k) \cdot \bar{H}(g,i,j,k)$$

where g = (FM) or (M) denote the sex

$\frac{N(g,j,k)}{N_{tot}}$ = the relative frequency of the age class 'k' subjected to class 'j' exposure
(N_{tot} = the total number of individuals in the population)

$\omega_S(g,i,j,k)$ = the relative weighting factor for somatic effects in tissue or organ 'i' of individuals in age class 'k' subjected to class 'j' exposure.

$\bar{H}(g,i,j,k)$ = average dose equivalent to tissue or organ 'i' in individuals of age class 'k' subjected to class 'j' irradiation.

$M_S(g,i,j,k)$ = malignancy significant factor for a malignant disease of organ or tissue 'i' in individuals of age class 'k' subjected to class 'j' radiation.

In the second approximation the malignancy significant factors are taken to one i.e. $M_S \simeq 1$ which is an overestimation, because M_S by

definition is < 1. But still it is required a lot of detailed information about age distributions which are very seldom available in practice. Therefore one must assume that there is no age dependence in the incidence and in the frequency distributions. Then one arrives to the following expression which is very similar to the 'somatic dose index' suggested by Laws and Rosenstein (8).

$$SSD = \sum_{g,i,j} \left(\frac{N(g,j)}{N_{tot}}\right) \cdot \omega_S(g,i,j) \cdot H(g,i,j)$$

The aproximation of no sex-dependence makes the final expression for the somatically significant dose equivalent

$$SSD = \sum_{i,j} \left(\frac{N(j)}{N_{tot}}\right) \sum_{i} \omega_S(i,j) \cdot \bar{H}(i,j) = \sum_{i} \left(\frac{N(j)}{N_{tot}}\right) \cdot H_{S,E}(j)$$

where $H_{S,E}(j) = \sum_{i} \omega_S(i,j) \cdot \bar{H}(i,j)$ is the somatically effective dose equivalent for the type of irradiation 'j'.

The incidence for fatal somatic effects and the relative weighting factors for somatic effects in different organs or tissues 'i' are given in table 1 (5,6,12). Morbid somatic effects will not be considered in the present paper.

TABLE 1. Incidence for fatal somatic effects and the relative weighting factors for somatically effective dose equivalent (5,6,12).

Organ or tissue	Incidence percent per Sv	Relative somatic effective weighting factors ω_S
Breast	0.25	0.19
Red marrow	0.20	0.16
Lung	0.20	0.16
Thyroid	0.05	0.04
Bone surfaces	0.05	0.04
Remainder	0.50	0.40
Skin	0.01	0.01
Total	1.26	1.00

DIAGNOSTIC X-RAY EXAMINATIONS

Estimation of organ doses from diagnostic X-ray procedures can be made either by direct measurements of absorbed dose or by calculations. The Monte Carlo method has been used extensively for calculation of absorbed dose received by each organ of interest during diagnostic X-ray examinations.

TABLE 2. Calculation scheme for the somatically effective dose equivalent $H_{S,E}(j) = \sum_i \omega(i) \cdot D(i,j)$

	Average absorbed dose to various organs and tissues from different X-ray examinations D(i,j) mGy							
	Breast	Red Marrow	Lung	Thyroid	Bone surfaces	Remainder	Skin	$H_{S,E}$ mSv
NERVOUS SYSTEM								
Head (skull)	0.04	0.54	0.04	3.7	1.00	10.0	11.76	1.87
Head-angiography	-	5.87	-	-	-	-	-	-
THORAX PLUS NECKORGANS								
Heart and Lung	1.95	0.73	2.9	0.97	2.6	11.1	10.5	6.62
Angiocardiography	-	1.42	-	-	-	-	-	-
Whole chest	0.27	0.13	0.42	0.10	0.26	0.66	0.78	0.48
Chest-mass survey	-	0.72	-	-	-	-	-	-
Tomography (Lung)	-	1.93	-	-	3.18	20.1	17.0	-
DIGESTIVE ORGANS								
Barium swallow (stomach)	3.4	4.6	8.6	1.01	6.7	30.2	26.8	17.8
Barium meal (intestine)	0.81	5.2	1.0	0.06	5.3	38.6	35.6	20.4
Barium enema (colon)	0,17	5.9	0.25	0.05	9.2	18.9	17.8	10.8
Stomach-mass survey	-	0.60	-	-	-	-	-	-
Gallbladder	0.37	1.39	1.29	0.02	1.48	15.5	10.36	7.8
UROGENITAL ORGANS								
Abdomen-Kidney Urether and Bladder	0.29	1.26	0.14	0.02	2.9	9.3	7.5	4.9
Urography	3.7	3.4	0.67	0.21	4.7	38.6	16.0	18.5

TABLE 2. Continued

	Breast	Red marrow	Lung	Thyroid	Bone surfaces	Remainder	Skin	$H_{S,T}$ mSv
Cystography (Bladder)	-	3.0	-	-				
Abdomen-angiography	-	3.8	-	-	-	-	-	-
Abdomen-obstretic	-	2.3	-	-	-	-	-	-
Salpinography	-	1.81	-	-	-	-	-	-
Placentography	-	-	-	-	-	-	-	-
Other	-	0.99	0.60	0.01	-	-	-	-
SKELETON, EXTREMITIES								
Dental (Full mouth 14 films)	0.1	0.05	0.01	0.5	0.025	*)	0.3	1.17
Cervical spine	0.05	0.33	0.12	4.1	1.00	3.4	8.8	2.5
Thoracic spine (dorsal)	4.0	1.95	5.6	5.0	3.4	9.7	9.1	6.3
Lumbar spine	1.58	2.3	0.95	0.08	4.1	11.5	16.8	7.3
Lumbo Sacral joint	0.03	2.9	0.23	0.01	2.7	7.1	6.0	4.1
Hip and upper femur	0.03	0.80	0.04	0.01	2.7	7.1	6.0	3.7
Leg and foot	0.01	0.11	0.01	0.20	0.18	0.18	1.25	0.24
Clavicle and Shoulder	0.64	0.35	0.99	0.50	-	-	-	-
Arm and Hand	-	-	-	0.01	-	-	-	-
OTHER								
Mammography	-	-	1.43	-	-	-	-	-
Lymphoangiography	-	-	2.03	-	-	-	-	-

*)Salivary glands 6 mGyx0.01
Brain 0.2 mGyx0.01
Remaincer 0.005 mGyx0.38

Estimation of the somatic effective dose equivalent needs data on the absorbed dose to breast, red bone marrow, lung, thyroid, bone surfaces, skin and up five more highly exposed organs or tissues. The absorbed dose to these organs from various types of X-ray examinations are given in table 2. These data are extracted from measured and calculated data published by various authors (1,8,3).

The somatically effective dose equivalent $H_{S,E}$ for the various types of X-ray examinations which has been derived from the absorbed dose values in table 2 by applying the weighting factors given in table 1 are given in the right column of table 2. The highest somatically effective dose equivalent are received by X-ray examinations of the small intestine (barium meal), descending urography (IVP) and the stomach and oesophagus (barium swallow) which all give values in the order of 18-20 mSv (1.8-2 rem).

DIAGNOSTIC USE OF RADIOPHARMACEUTICALS

The absorbed dose to various organs and tissues can be derived from the well known MIRD formalism but this must be extended to involve also the target dose resulting from the radioactive material distributed within the remaining parts of the body. Generally, the absorbed dose D(i) in any target organ (i) is composed of three parts

$$D(i) = D(i \leftarrow i) + \sum_h D(i \leftarrow h) + D(i \leftarrow RB)$$

where $D(i \leftarrow i)$ is the absorbed dose due to self-irradiation from the target organ (i), $\sum_h D(i \leftarrow h)$ is the absorbed dose due to irradiation from specified source organs (h), $D(i \leftarrow RB)$ is the absorbed dose due to irradiation from radioactivity in the rest of the body (RB).

In medical applications a unit bolus injection of activity A_o is often used which means that the cumulated activity in an organ can be written as

$$\tilde{A}(o,\infty) = \tau_k \cdot A_o$$

where τ_k is the mean residence time of the radioactivity in region k. The somatically effective dose equivalent may thus be given as

$$H_{S,E} = A_o \cdot \sum_i w_S(i) \{ \tau(i) \cdot S(i \leftarrow i) + \sum_h \tau(h) \cdot S(i \leftarrow h) + \tau_{RB} \cdot S(i \leftarrow RB) \}$$

where the mean absorbed dose per unit cumulated activity S for the remainder of the body (i.e. total body excluding the source volumes) may be represented by the following expression (11).

$$S(i \leftarrow RB) = S(i \leftarrow WB) \frac{m_{WB}}{m_{RB}} - \sum_h S(i \leftarrow h) \frac{m_h}{m_{RB}}$$

where $S(i \leftarrow RB)$, $S(i \leftarrow WB)$ and $S(i \leftarrow h)$ are the mean absorbed dose per unit cumulated activity for the target region, considering the remainder of the body, the total body and the h:th source volume respectively.

Taking into account the relationship between the residence time for the rest of the body, total body and the target organs ($\tau_{RB} = \tau_{WB} - \sum_h \tau_h$) the somatic effective dose equivalent may be given

TABLE 3. Somatic effective dose equivalent (μSv/MBq) and absorbed dose to 'critical' organs per unit of administered activity for some radiopharmaceuticals (1 μSv/MBq =3.7 mrem/mCi ;1 mrem/mCi = 0.27 μSv/MBq)

Compound	Function or organ examined (Average administered activity MBq) (1 mCi = 37 MBq)	Absorbed dose μGy/MBq 'Critical' organs	Average whole body	$H_{S,E}$ μSv/MBq
Cr-51 EDTA	GFR (3,2)	kidney:80-120 -	3	9-16
Tc-99m colloids	liver (300)	liver: 92, spleen:57	5.1	12
Tc-99m DTPA	kidney (150)	kidney:24	1.7	3.5
Tc-99m MAA, HAM	lung (70)	lung:57, liver:10	3.7	15
Tc-99m pertechnetate	brain; thyroid (400, 60)	thyroid:35	38	8
Tc-99m-phosphates	skeleton (350)	skeleton:15	4,7	5
I-123 iodine	hypothyroid (20)	thyroid:2 400	11	100
	euthyroid "	thyroid:5 400	9	220
	hyperthyroid "	thyroid:13 000	8	520
I-125-iodide	hypothyroid (1.4)	thyroid:250 000	140	10 000
	euthyroid "	thyroid:470 000	260	19 000
	hyperthyroid "	thyroid:800 000	320	32 000
I-131-iodide	hypothyroid (0.4)	thyroid:230 000	80	18 000
	euthyroid "	thyroid:570 000	110	41 000
	hyperthyroid "	thyroid:1 000 000	180	80 000
Au-198 colloid	liver (40)	liver:11 000	380	1 230

by the following expression

$$H_{S,E} = A_o \sum_i w(i) \{\tau(i)\cdot S(i\leftarrow i)+\sum_h \tau(h)\cdot S(i\leftarrow h)+(\tau_{WB}-\sum_h \tau(h))\cdot$$

$$(S(i\leftarrow WB)\frac{m_{WB}}{m_{RB}} - \sum_h S(i\leftarrow h)\frac{m_h}{m_{RB}})\}$$

The somatically effective dose equivalent per unit of administered activities for some of the most frequent examinations are given in table 3 together with the absorbed dose to some critical organs.

The calculation of these values are mainly based on the data summarized by Kaul and Roedler (7,10,9)

RADIATION THERAPY

There are surprisingly enough no complete information on various organ doses available in the literature for an estimation of somatically effective dose equivalent in radiation therapy procedures. The high absorbed dose used in radiation therapy makes it further difficult to estimate which weighting factors should be used in the calculations. Therefore it seems not at the present time to be meaningful to make any estimation of somatically effective dose equivalent in radiation therapy.

Attempts have, however, been made to calculate leukemia significant dose equivalent based on absorbed dose to the marrow and a leukemia significant factor (2).

In table 4 is given the leukemia significant dose equivalent LSD for beam therapy (μSv per person and year) in Japan.(2,4).

SOMATICALLY SIGNIFICANT DOSE EQUIVALENT

By applying the relative frequency of various types of X-ray examinations and examinations with radiopharmaceuticals to the somatically effective dose equivalent one can easily arrive to an approximative value of the somatically significant dose equivalent.

TABLE 4. The leukemia significant dose equivalent LSD (μSv per person and year) from beam radiation therapy in Japan 1971 and 1978 (2,4)

Irradiation Source	LSD (μSv per person and year)			
	Male		Female	
	1971	1978	1971	1978
X-ray: HVL 2 mm Al	5.9	-	18	1.5
X-ray: HVL 0.5-2 mm Cu	2.6	-	12	0.1
Co-60 gamma rays	39	31	180	9.5
Accelerator: photons 4-30 MV	13.6	37	85	63
Accelerator: electrons 8-35 MeV	4.8	0.4	7.4	2
Total	66	68	302	236

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IS THE DOSE EQUIVALENT INDEX A QUANTITY TO BE MEASURED?

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PRIMARY RADIATION PROTECTION QUANTITIES

EFFECTIVE DOSE EQUIVALENT

With its Publication 26 /1/, ICRP limited the risk of stochastic radiation effects in man by setting a limit to the annual effective dose equivalent

$$H_e = \sum_j w_j H_j \qquad (1)$$

which is a weighted (w_j) sum of average dose equivalents H_j in particular tissues. Of course, H_e cannot be determined by routine radiation protection measurements, the chief obstacle being the intricate geometrical shape and the intrinsic structure of the human body as well as the necessity to determine dose equivalent distributions in many tissues. Yet it should be emphasized that H_e is unequivocally defined by eq. (1) in any radiation environment irrespective of its variation with time or of the movement of the exposed person.

SECONDARY OR OPERATIONAL RADIATION PROTECTION QUANTITIES

DOSE EQUIVALENT INDEX

Hence, for practical measurements operational quantities must be introduced which approximate H_e. If such an operational quantity is to be applicable to whole body irradiation by all kinds of ionizing radiations it must be defined in an anthropomorphous phantom, simulating the trunk of the human body, as e.g. in a neutron field secondary radiations originating in the irradiated person contribute substantially to H_e. A simple phantom sufficiently anthropomorphous for radiation protection in many situations of external irradiation is the 30 cm diameter sphere composed of soft tissue as introduced by ICRU /2/. The definition of the maximum dose equivalent in this sphere as a possibly operational quantity, which is termed dose equivalent index H_I, was then obvious.

Eventually, ICRP /1/ took up this idea and stated that with external exposure to penetrating radiation the limitation of the deep dose equivalent index $H_{I,d}$ i.e. the maximum dose equivalent in the inner 28 cm diameter core of the sphere /3/, would afford at least as good a level of protection as the limitation of H_e. For simplicity's sake, only the unrestricted dose equivalent index H_I as introduced above will be considered in the following.

EFFECT OF IRRADIATION HISTORY

Whereas the introduction of the dose equivalent index certainly simplifies calculations in a given stationary radiation field, ambiguity arises when the dose equivalent to a moving person is to be estimated, as the 30 cm sphere has no inherent coordinate system and hence cannot be oriented in space. Likewise, when the radiation field varies with time with the spere remaining fixed, the dose equivalent index H_I is generally not the time integral of the dose equivalent

index rate $\dot{H}_I$,

$$H_I \leq \int_0^T \dot{H}_I(t)dt \qquad (2)$$

as the location of the dose equivalent rate maximum in the sphere may vary with time:
- H_I is non-additive with respect to its components in time.

EFFECT OF IRRADIATION GEOMETRY

This peculiarity of the dose equivalent index also shows up in the superposition of various angular components. Let $d\vec{\Phi}(d\Phi/d\Omega)$ be the differential fluence of incident particles with directions whithin the solid angle element $d\vec{\Omega}$. Then the corresponding differential maximum dose equivalent will be $d\hat{H} = d\vec{\Omega}(d\hat{H}/d\Phi)/(d\Phi/d\Omega)$. As the maxima produced by various angular components may occur at different locations within the sphere, the relation

$$H_I \leq \int_{4\pi} \frac{d\hat{H}}{d\Phi} \frac{d\Phi}{d\Omega} d\vec{\Omega} \qquad (3)$$

is valid for the resulting maximum H_I:
- H_I is non-additive with respect to its components in solid angle.
Relation (3) allows the maximum uncertainty in H_I coming from this kind on "non-additivity" to be estimated. For radiation of fluence Φ incident unidirectionally to the sphere, this is also the maximum fluence at the surface of the sphere, and numerically equal to dN/dF, the number of particles per surface element passing the surface of the sphere at that point where the particles hit this surface element dF perpendicularly. For radiation of the same fluence Φ incident isotropically to the sphere, the maximum fluence at the surface is only $\Phi/4$ at any point of the surface of the sphere, if the sphere fully shields the incident radiation[1]. This means that also the number of particles passing the surface per surface element dF is (1/4)dN/dF. In those cases where the sphere only partly shields the incident radiation, the anisotropy factor

$$k_\Omega = H_I / \int \frac{d\hat{H}}{d\Phi} \frac{d\Phi}{d\Omega} d\vec{\Omega} \qquad (4)$$

is between 1/4 and 1 (cf. /4/).

EFFECT OF RADIATION ENERGY

Similarly, the location of the dose equivalent maximum in the sphere depends on the energy or energy distribution in the incident radiation. Again, the dose equivalent index resulting from various energy components in the incident radiation will generally not be the sum of the dose equivalent maxima corresponding to these components:

$$H_I \leq \int \frac{d\hat{H}}{d\Phi} \frac{d\Phi}{dE} dE \qquad (5)$$

---

[1] This is illustrated by recalling that in the unidirectional case, the total number of particles incident to the sphere of radius r is $N = \Phi\pi r^2$. In the isotropic case this number N is to be distributed equally over the whole surface $4\pi r^2$ which leads to a "shielded" incident particle fluence $\Phi\pi r^2/(4\pi r^2) = \Phi/4$.

- H_I is non-additive with respect to its components in energy. According to Harvey /4/, the factor

$$k_E = H_I / \int \frac{dH}{d\Phi} \frac{d\Phi}{dE} dE \qquad (6)$$

ranges between 0.7 and 1.

The maximum uncertainty in H_I coming from the non-additivity of angular and energy components is then characterized by the extreme values

$\check{k} = \check{k}_\Omega \cdot \check{k}_E = 0.175$ and $\hat{k} = 1$, the ratio of which is

$\hat{k}/\check{k} = 5.7 \qquad (7)$

INSTRUMENT PERFORMANCE AND CALIBRATION

These considerations lead to two essential requirements for an instrument intended to measure H_I:
- The instrument must exhibit the full scattering and absorption properties of the 30 cm dia. sphere;
- The dose equivalent distribution must be explored throughout the entire sphere in order to locate the resulting maximum.

Notwithstanding the fulfilment of these requirements there is no simple and unique way to derive H_I from dose equivalent index rate measurements. The uncertainty mentioned above will subsist. In practice, however simple instruments with a single fixed detector will be used. Whereas it should be possible in principle to construct an instrument to indicate H_I for a range of incident radiation energies in a fixed irradiation geometry, e.g. unidirectional incidence, the calibration performed under these conditions is not transferable to other irradiation geometries. The maximum deviation is given by expression (7). Yet, a measurement which may be uncertain to a factor of 5, loses its sense.

OTHER OPERATIONAL QUANTITIES

Among possible candidates for operational quantities, which do not show the shortcomings of H_I are,
- Dose equivalent ceiling $H_c = \int d\hat{H}$ (cf. /4/) which is the sum of dose equivalent maxima in the sphere for all radiation components;
- Average dose equivalent $\overline{H}$ in the sphere;
- The dose equivalent H_d in a specified depth at a particular location in the sphere.

That operational quantity H_O for which the ratio H_e/H_O is least energy dependent should be most suitable. As H_e is additive with respect to all its components, the same should apply to any quantity H_O. The three mentioned above fulfil this condition. As results of calculations and experiments to answer these problems are still scarce, fig. 1 can give only a few examples for photon radiation based on ref. /5/. It is to be noted that none of the scalar quantities mentioned above can give full information on H_e, since H_e depends among other things on the orientation of the exposed person in the radiation field and on its variation over the occupied space.

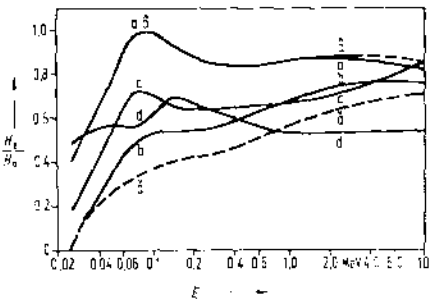

Figure 1. Ratios H_e/H_O of effective dose equivalent H_e to various operational quantities H_O (cf. /5/).

Curve a: $H_O \equiv H_{I,d}$ deep dose equivalent index, irradiation geometry (IG): parallel beam (PAR) rotating perpendicularly to vertical axis of the human body (ROT)

Curve ă: " minimum value of $H_e/H_{I,d}$ } IG: from PAR

Curve â: " maximum value of $H_e/H_{I,d}$ } to ROT

Curve b: $H_O \equiv H_{I,d}^{PAR}$ Reading of an isotropic detector calibrated by means of a parallel beam to indicate $H_{I,d}$; IG: ROT

Curve c: $H_O \equiv H_{I}$ (unrestricted) dose equivalent index; IG: ROT

Curve d: $H_O \equiv 2\overline{H}$ Average dose equivalent in the 30 dia.sphere; IG:ROT

The curves for the minimum (ă) and maximum (â) values of the ratio $H_e/H_{I,d}$ indicate its range for various exposure conditions from parallel beam incidence perpendicular to the vertical axis of the body to a parallel beam rotating around this axis. The solid curves give various ratios for the rotational beam. Obviously, the ratio $(H_e/\overline{H})/2$ (curve d) shows the least variation with incident radiation energy. It should be pointed out that the absolute value of the ratios is not of great consequence, as a conservative estimate of H_e can always be achieved by applying an appropriate scale factor or by introducing secondary limits to the respective operational quantity.

Future investigations should include the fully isotropic case, extend the range of incident energies and in particular consider kinds of radiations other than photons. Quantities of vector or matrix character should be explored in order to estimate H_e to a closer approximation.

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A THEORETICAL APPROACH FOR THE MEASUREMENT OF THE EFFECTIVE DOSE EQUIVALENT FOR EXTERNAL RADIATIONS

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The quantity 'effective dose equivalent', H, was introduced by the ICRP [1] to express the maximum admissible dose limits for radiation induced stochastic effects. In this paper, the relation between the response of different types of detectors and H is given explicitly, and construction requirements are deduced. Criteria for 'operational quantities' facilitating the measurements are given. It is found that the 'dose equivalent index' [1,2] is not consistent with these criteria.

THE THEORETICAL APPROACH

An irradiation situation is considered in which an upright standing antropomorphous phantom is irradiated from a spacial distribution of radiation sources located in the horizontal plane, far away from the position of the phantom. The radiation fluence ϕ at the position of the phantom in receptor-free condition [2] can then be expressed in terms of the spectral angular fluence by

$$\phi = \int_0^\infty dE \int_{2\pi} d\alpha \cdot \phi_{E,\alpha}(\alpha, E) \qquad (1)$$

where the angle α indicates the direction and E the energy of the radiation. The spacial orientation of the phantom (its 'nose') is indicated by the angle α^H. Then the effective dose equivalent can be written.

$$H(\alpha^H) = \int_0^\infty dE \int_{2\pi} d\alpha \cdot M^H(\alpha - \alpha^H, E) \cdot \phi_{E,\alpha}(\alpha, E) \qquad (2)$$

where the function M^H can in principle be calculated by Monte Carlo calculations and may therefore be considered as a known quantity.

For practical measurements, the angular and the energy range are decomposed in intervals $[\alpha_j, \alpha_j + \Delta\alpha_j]$ and $[E_k, E_k + \Delta E_k]$. With the definition

$$M^H_{j,k}(\alpha^H) = M^H(\alpha_j - \alpha^H, E_k) \qquad (3)$$

one can write approximately

$$H(\alpha^H) \approx \sum_{j=1}^{j_m} \sum_{k=1}^{k_m} \int_{\Delta E_k} dE \int_{\Delta\alpha_j} d\alpha \cdot M^H_{j,k}(\alpha^H) \cdot g_{j,k}(\alpha, E) \cdot \phi_{E,\alpha}(\alpha, E) \qquad (4)$$

where the functions g are derived by interpolating $M^H(\alpha-\alpha^H,E)/M^H_{j,k}$ within the given interval. Defining the quantity

$$\Phi^H_{j,k} = \int_{\Delta E_k} dE \int_{\Delta\alpha_j} d\alpha \cdot g_{j,k}(\alpha,E) \cdot \Phi_{E,\alpha}(\alpha,E) \qquad (5)$$

one can write

$$H(\alpha^H) = \sum_{j=1}^{j_m} \sum_{k=1}^{k_m} M^H_{j,k}(\alpha^H) \cdot \Phi^H_{j,k} \qquad (6)$$

where the matrix M^H is called 'phantom response matrix'.

Now the phantom be replaced by a 'directionally dependent spectrometer', the most general type of detector considered in this paper. Its response $R_{J,K}$ is assumed to be distributed in two sets of discrete variables $J = 1,2\ldots, J_m$ and $K = 1,2,\ldots, K_m$. The variable J is related to the measurement of the radiation direction j, and may, for instance, be substantiated by turning the instrument to J_m different directions. The variable K is related to the measurement of the energy k of the radiation, and may, for instance, be substantiated by a pulse height spectrum. Even for fixed values of j and k, the measured values $R_{J,K}$ are allowed to be distributed. Simplifications introduced below will allow to consider less complicated detectors.

Defining corresponding quantities as in eqs. (3) and (5), the response of the detector can be written

$$R_{J,K} = \sum_{j=1}^{j_m} \sum_{k=1}^{k_m} M^R_{J,j,K,k} \cdot \Phi^R_{j,k} \qquad (7)$$

The detector response matrix $M^R{}_{J,j,K,k}$ can be measured using 'monoenergetic' radiation with energy k and incident direction j. Inherent symmetries of usually employed detectors will in practice reduce the number of different matrix elements.

The inversion of eq. (7) is given by

$$\Phi^R_{j,k} = \sum_{J'=1}^{J'_m} \sum_{K'=1}^{K'_m} P^R_{j,J',k,K'} \cdot R_{J',K'} \qquad (8)$$

Insertion of eq. (8) in eq. (7) gives

$$\sum_{j=1}^{j_m} \sum_{k=1}^{k_m} M^R_{J,j,K,k} \cdot P^R_{j,J',k,K'} = \delta_{J,J'} \cdot \delta_{K,K'} \qquad (9)$$

It is assumed that this system of linear equations can be used to express the projection matrix P by the measured response matrix M^R. A necessary condition is $J_m \cdot K_m \geq j_m \cdot k_m$ with $J_m = J_m'$ and $K_m = K_m'$.

It is further assumed that $M^H(\alpha-\alpha^H,E)$ and $M^R(\alpha-\alpha^R,E)$ have similar interpolation properties within the intervals. Then

$$\Phi^H_{j,k} = \Phi^R_{j,k} = \Phi_{j,k} \tag{10}$$

The effective dose equivalent can be expressed by eqs. (6), (8) and (10):

$$H(\alpha^H) = \sum_{J=1}^{J_m} \sum_{K=1}^{K_m} M_{J,K}(\alpha^H) \cdot R_{J,K} \tag{11a}$$

where

$$M_{J,K}(\alpha^H) = \sum_{j=1}^{j_m} \sum_{k=1}^{k_m} M^H_{j,k}(\alpha^H) \cdot P^R_{j,J,k,K} \tag{11b}$$

As mentioned above, both matrices, M^H and P, can be considered to be known.

From eqs. (11a), (11b) the construction requirements can be deduced for four different types of detectors:

1. Requirement for the directionally independent dosimeter: M is independent of J and K, i.e.

$$H(\alpha^H) = M(\alpha^H) \cdot \sum_{J=1}^{J_m} \sum_{K=1}^{K_m} R_{J,K} = M(\alpha^H) \cdot R \tag{12}$$

The dosimeter only needs to indicate one single value R which is linearly proportional to H.

2. Requirement for the directionally dependent dosimeter: M is independent of K, i.e.

$$H(\alpha^H) = \sum_{J=1}^{J_m} M_J(\alpha^H) \cdot \sum_{K=1}^{K_m} R_{J,K} = \sum_{J=1}^{J_m} M_J(\alpha^H) \cdot R_J \tag{13}$$

This dosimeter only needs to indicate the values R_J.

3. Requirement for the directionally independent spectrometer: M is independent of J, i.e.

$$H(\alpha^H) = \sum_{K=1}^{K_m} M_K(\alpha^H) \cdot \sum_{J=1}^{J_m} R_{J,K} = \sum_{K=1}^{K_m} M_K(\alpha^H) \cdot R_K \tag{14}$$

This spectrometer only needs to indicate the values R_K.

4. The directional spectrometer, the most general case, has been discussed above. This instrument needs to indicate the complete information $R_{J,K}$ (see eq. (11a)).

OPERATIONAL QUANTITIES

In practice it may be advisable to simplify the construction requirements based on eqs. (11a), (11b).

In such cases a simplified phantom response matrix $\bar{M}^H$ can be determined such that

$$\bar{H}(\alpha^H) = \int_0^\infty dE \int_{2\pi} d\alpha \cdot \bar{M}^H(\alpha - \alpha^H, E) \cdot \phi_{E,\alpha}(\alpha, E) \qquad (15)$$

is an approximation to $H(\alpha^H)$. The quantity $\bar{H}$ with the corresponding response matrix $\bar{M}^H$ may facilitate the problem of measuring H, and it is defined as 'operational quantity'. Either one or both of the following simplifications can be introduced in order to construct an operational quantity:
(i) purely mathematical simplifications, for instance fitting the phantom response matrix to a simplifying analytical expression.
(ii) the phantom itself can be simplified and the corresponding response matrix be used.

DISCUSSION

It is concluded that Monte Carlo calculations for anthropomorphous phantoms are required as a prerequisite to construct the response matrix of the phantom. If, in addition, the response matrix of the detector is measured, then the effective dose equivalent can explicitly be expressed by the measured detector response. The extension of this concept to a three-dimensional distribution of distant sources does not pose any basic problems.

Regarding operational quantities, in the past, these often have been defined as the maximum of the spacial distribution of the 'dose equivalent index' [1,2], for which, however, a response matrix $\bar{M}^H$ (see eq. (15)) cannot be constructed because the corresponding superposition properties are lacking. Within the framework of the approach given here, such a quantity appears to be unsuitable.

ACKNOWLEDGEMENT – We like to thank Prof. Dr. S. Wagner for valuable discussions.

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BIOLOGICAL EFFECTIVENESS OF FAST NEUTRONS

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The main difficulty in deriving human risk estimates for the purpose of radiation protection lies in the lack of information on the radiation action at the extremely low doses and dose-rates. In practice, this difficulty is overcome by extrapolating estimates of risk obtained at whatever dose is available down to the smallest dose levels, by making use of the hypothesis of non-threshold linearity between dose and effect. In the absence of direct data, risk estimates for high-LET radiations are usually derived from low-LET data through conversion factors, such as the Quality Factor, Q. Quality factors are extremely important in these procedures because they allow scaling of information relating to different radiation qualities according to one family of variables, that of dose-equivalent.

The above general procedure entails two main assumptions. Firstly, the assumption of linearity, to which a great deal of theoretical and experimental radiobiological work is and has been devoted in an effort to obtain supportive evidence. Secondly, the assumption of a quality factor independent of dose and dose-rate or, alternatively, the choice of quality factors applicable at the dose levels and for the effects of interest, which remains a controversial matter.

The present paper is concerned with the influence that different shapes of dose-effect relationships might have on the assessment of risk estimates and the adoption of quality factors.

GENERALITIES

Before doing so, it is essential to recall notions that are quite clear in principle, but often confuse such discussions. Firstly, the levels of dose and dose-rates at which the following considerations apply are orders of magnitude below the range where radiobiological effects are detectable. At these very low levels of dose and effect the actual shape of the function linking the two variables may bear little relationship to the experimental data observed at higher doses and, therefore, the analysis must of necessity be speculative and inferential. Secondly, there is a difference between the concepts of Relative Biological Efficiency (RBE) and Quality Factor, Q, the first being an experimentally determined relative quantity, applying to a given effect and exposure condition, the latter a conventionally defined factor, which is intended to allow for the influence of any given radiation quality on biological effect /5/, simply for the requirements of radiation protection. The factor Q is formulated as a continuous function of the collision stopping power, L_{∞}, alone /6/, an approximation which is considered to be acceptable at the level of accuracy required for radiation protection /4/.

DOSE-EFFECT CURVES

Among all possible forms of dose-effect relationships for high-LET and low-LET radiations compared for determining values of RBE, three examples will be discussed hereafter. The first is that of two curves having the same form and

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differing only for a dose-modifying factor. In this case the RBE would be independent of the level of effect and risk estimates derived at high levels of dose would be applicable down to the lowest exposures. This situation, which includes the case of linearity generally used in radiation protection, is however difficult to be verified experimentally.

Radiobiological experience shows, on the contrary, that for many biological end-points the reference low-LET radiation has a more than linear dose-effect dependence of the form $I(D)=aD+bD^2$, while for the same end-point the dose-effect relationship is more nearly linear in the same range of effect for high-LET radiation, such as neutrons. The linear + quadratic relationships for low-LET radiation has been examined by several authors for many radiation induced carcinogenic, mutational and cytogenetic end-points /1, 11/. For many of the end-points considered the linear term dominates the response up to a dose of approximately 100 rad. Consequently, assuming a linear non-threshold relationship could overestimate the risk at low doses and dose-rates of low-LET radiation by about a factor of two. At the same time the value of RBE corresponding to a given level of dose could still be used as an estimate of the effectiveness of the high-LET radiation at lower doses, although the RBE would become larger for low dose exposures. In fact, the RBE of the high-LET radiation, would be given by the expression

$$RBE = \frac{2a_H}{a_L + (a_L^2 + 4a_H b_L D_H)^{0.5}}$$

where the indices H and L refer to high- and low-LET radiation, respectively. Thus, RBE would increase with decreasing dose up to a maximum equal to a_H/a_L, which is reached when $D_H a_H/a_L \ll a_L/4b_L$, that is $D_H RBE_{max} \ll a_L/4b_L$. For a_L/b_L =100 rad and for RBE_{max}=10 (which is very probably an underestimate) the constant RBE would be reached for $D_H \ll 2.5$ rad and, therefore, RBE would in practice be increasing for absorbed doses down to very low levels. Actually, neutron RBE values relative to X- or gamma-rays exceeding 100 have been measured /9/. It should be pointed out that, under the present assumptions, the increase of RBE at very low doses would not be caused by an increased effectiveness of neutrons, but by a decreased effectiveness of the low-LET radiation. Problems may arise when quality factors are based on RBE's which are measured at much higher doses than required for human risk estimates, as it is usually the case, and Q might thus be taken to be lower than it should.

A third ideal case would be one where the dose-effect relationship for the low-LET radiation is linear through the origin, and that for the high-LET radiation less than linear. Assuming, for instance, relationships of the type $I(D)=a_H D^{0.5}$ for the high-LET radiation and $I(D)=a_L D$ for the low-LET radiation, the RBE would be of the form

$$RBE = \frac{a_H}{a_L D_H^{0.5}}$$

Essentially, the dependence of RBE on high-LET radiation dose would still apply, as in the previous example, although no constant value would be reached at very low doses. Under these conditions risk estimates made at any dose level would apply to any low dose of the low-LET radiation, but using a constant Q would underestimate the risk of very low doses of the high-LET radiation.

EXPERIMENTAL EVIDENCE

Here follow a few examples of biological effects where, irrespective of

the level at which the effect is expressed (cellular, tissue, whole body), there is a suggestion of an increasing RBE with decreasing dose. The first example is on protoplasts of Nycotiana plumbaginifolia /8/, a biological system suitable for studies at low doses for its high radiosensitivity and plating efficiency and for the possibility of obtaining large amounts of synchronized cells. As shown in Fig. 1 the surviving fraction of cells decreases bi-phasically with increasing absorbed dose of 16 MeV neutrons, which could be described by the superimposition of two exponentials. At corresponding levels of effect the curve for the low-LET radiation may be taken as a simple exponential within the experimental errors of the data points. As a result, values of RBE increasing with decreasing neutron dose are found in the low-dose region and a value of RBE larger than 70 would apply to a neutron dose of about 1 rad. The shapes of the response curves suggest that the increasing RBE is due to increased neutron effectiveness at low doses.

The case of mouse testis weight loss at 28 days postirradiation (see Fig. 2) /2/ is similar in that at low doses the curves would be interpreted as if a consistent fraction of cells would show an enhanced response to high-LET radiation giving rise to high values of RBE below about 5 rad.

Another example is provided by cataractogenesis in the mouse (Fig. 3). Recent experiments /3/ have shown that in the dose region of up to few Gy the dose-effect curve for 250 kVp X-rays is well described by a straight line through the origin. Values of RBE increasing with decreasing neutron dose are found, approaching 50 for about 1 rad of 1 MeV neutrons. Here again, the data suggest that the increase of RBE results from an increased neutron effectiveness at the low doses.

Neutron RBE values exceeding 100 have been derived at low doses for the induction of mammary tumors in Sprague-Dawley rats /10, 7/. The very important feature of these results for radiation protection is the sublinear dose dependence at low neutron doses, with a dose exponent of about 0.5.

Similar results have been found from a comprehensive analysis of data on life shortening in the mouse following acute irradiations. In the dose region up to few Gy a linear non-threshold dose dependence is suggested by the data for X- and gamma-ray exposures. For neutrons a sublinear dose dependence describes the available results, with an optimum value of 0.5 for the dose exponent.

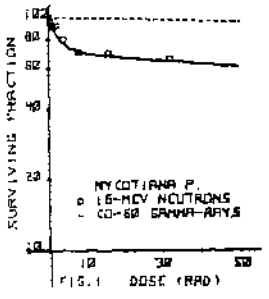

FIG.1

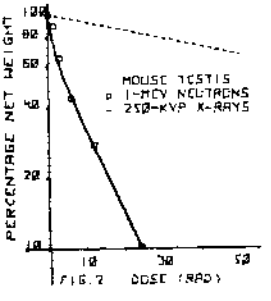

FIG.2

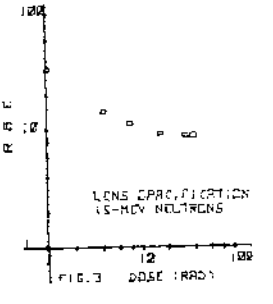

FIG.3

CONCLUSIONS

Although more-than-linear dose dependences generally apply to the induction of biological effects by X- or gamma-rays, in several cases terms higher than first order seem to play a negligible role in the region of importance

for radiation protection. However, in the same cases neutron RBE values show a definite tendency to increase with decreasing neutron doses, which is compatible with the assumption of an increased neutron effectiveness at low doses.

The above considerations are purely empirical and apply to effects observed mostly on highly radiosensitive biological systems. They are by no means to be generalized until sufficiently convincing explanations might be provided to account for the increased neutron effectiveness at low doses or dose-rates. Neverteless, these observations applying to a wide range of biological effects make the matter sufficiently interesting for further investigation on suitable materials and meaningful end-points. It is possible that following the above and further evidence the values of the Quality Factor, Q, would have to be reconsidered.

Finally, microdosimetric results suggest that quantities with a direct influence on radiation effectiveness, such as $\bar{y}_D^+$, have a large dependence on radiation quality, ranging up to a factor of two for neutrons between 1 and 15 MeV. Therefore, adoption of a unique value of Q for all fast neutrons might not be the best choice for radiation protection, unless it were proven that in the very low dose region the neutron effectiveness becomes sufficiently independent of neutron energy /2/.

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HEALTH PHYSICS EVALUATION OF AN ACUTE OVEREXPOSURE TO A RADIOGRAPHY SOURCE

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Industrial radiography in South Africa has grown since 1948 to the present use of, inter alia, 130 iridium-192 sources (up to 200 Ci) by 29 firms. Although it remains the main cause of overexposure (1) in the use of radioisotopes, only two persons have suffered clinically observable injury before this incident in 1977. The one person who was here acutely overexposed, was followed-up by physical dosimetry, chromosome studies and medical surveillance.

1. THE INCIDENT

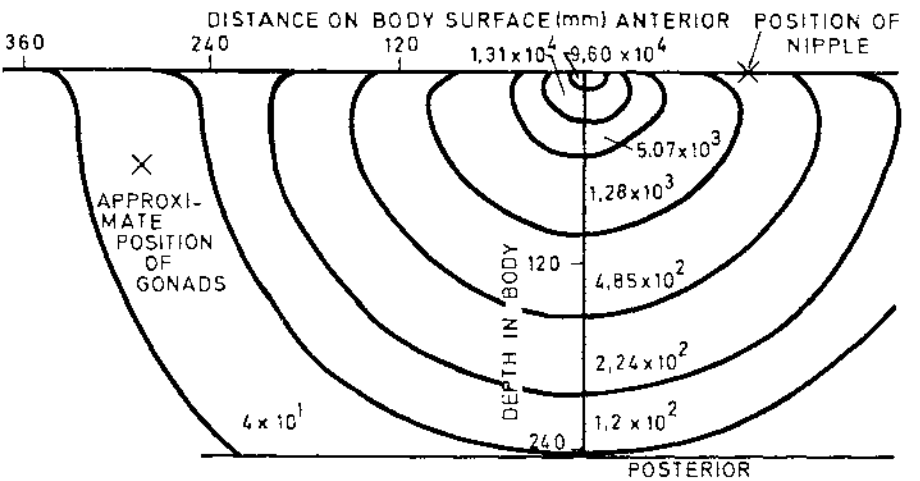
During radiography at a construction site a 6,7 Ci ^{192}Ir-source fell out of its container on Saturday morning, 8 January 1977. This was not noticed by the radiographer because of a faulty monitor, and he subsequently left the construction site. About 3 h later a construction supervisor, A, picked up the bright metallic object which he assumed to be a component of a mobile crane, and placed it in the left breast pocket of his shirt. Subsequently, he travelled home in a small bus with 6 other occupants who alighted at various points along the route, with A reaching his home after 40 min when he sat down to watch a television program. About 40 min later he became nauseous and vomited, and removed his shirt which was placed in a cupboard, and went to bed. The next morning he removed some money and the source from the shirt and placed it in the drawer of his bedside table. The family spent the rest of the day away from their house, retiring to bed at 20h00 with A immediately next to the bedside table and his 6-year old son sleeping between him and his wife. On Monday he left for work at 06h00 while his wife and son remained at home.

The loss of the source was discovered at 11h30 on the Monday when a search was instituted by the firm with the aid of radiation monitors. As the search proved fruitless, a replica of the source capsule was shown to the workers on site with subsequent identification and recovery of the source from A's bedside table at 15h45. In the meantime, the loss of the source was reported to the Atomic Energy Board, the South African regulatory body, at 13h45.

2. HEALTH PHYSICS

Due to the magnitude of the accident, A was admitted to hospital for observation and his family as well as his colleagues were immediately placed under medical supervision. Subsequent dosimetric calculations indicated that in addition to his family, only three colleagues (X, Y and Z) were to remain under medical observation, but that at no stage was any member of the public exposed to a level which warranted surveillance.

Although statements were obtained from A, his superiors, colleagues and the industrial radiographer, it was extremely difficult to reconstruct the accident in detail. It could, however, be derived with a fair amount of accuracy that the source remained in his pocket for 2 h 40 min, although its position in relation to his body (it was a loose-hanging shirt) could not be determined accurately. For dosimetric purposes it was considered that 33 % of the time was spent in the sitting position with the source approx. 60 mm from his body, whereas the source capsule was considered to be in contact with his skin in the standing and prone positions. Isodose curves were calculated using the shielding program PELSHIE (2), assuming a point source irradiating the body, considered as a slab of water - as presented in the following figure.



ISODOSE CURVES IN RAD (PATIENT A : 1,83 m 85 kg)

Drastic assumptions had to be made for the calculation of the whole-body dose, estimated as 133 rad.

Although the eventual biological effects were underestimated during the initial clinical examinations, the physical dosimetry led to A being kept under close medical observation as well as the decision to request chromosome dosimetry.

3. CHROMOSOME DOSIMETRY

On 31 January 10 ml blood samples from 6 people were placed in sterile heparinised specimen tubes and despatched by air to the National Radiological Protection Board in England. Here lymphocyte cultures were set up following a routine technique, incubated for 48 h and metaphase preparations examined for the presence of unstable chromosome aberrations (3).

The aberrations found and the resulting dose estimated together with their 95 % confidence limits based on scoring statistics, are

presented in the following table.

	No. cells	Dicentrics	Centric rings	Acentrics	Dose (rads)	95 % Confidence limits Lower	Upper
A	1 000	86	2	60	116	98	133
Wife	500	2	-	10	17	2	40
Child	500	1	-	3	10	2	34
X	500	-	-	1	-	-	26
Y	500	-	-	3	-	-	26
Z	500	-	-	5	-	-	26

The estimates of dose were available on 4 February, 5 days after the blood samples were taken, and were made by reference to an *in vitro* calibration curve produced with cobalt-60 gamma rays (4).

The cytogenetic method provided estimates of whole-body dose in agreement with the uncertain calculations, but did not permit quantitative estimates to be made of the absorbed dose to specific parts of the body. However, in the case of A, the distribution of the aberrations was not Poisson, which provided a firm indication that his exposure was not uniform.

4. MEDICAL OBSERVATIONS

The dosimetric evidence was confirmed by subsequent medical examinations and only A showed any clinical evidence of radiation injury.

The patient developed a slight nausea and loss of appetite within hours after exposure, that lasted only for about 24 h. ECG studies done at 1 week, 6 weeks and 18 months after exposure did not reveal any abnormalities. His blood pressure remained constant. Examinations of his urine done at weekly intervals were normal. The patient's wife gave birth to a full-term normal baby about a year after the incident.

Mentally the patient became depressed, being unable to do his normal work as a result of his injured left hand. No shortness of breath or infection occurred during this follow-up period. Chest X-rays done at monthly intervals, were within normal limits and no fibrotic changes were observed.

4.1 Laboratory Studies

Full blood counts as well as the sedimentation rate were determined, first at two-day intervals and then at weekly intervals. The haemoglobin, white-cell count, as well as the differential white-cell count remained within normal limits. The only abnormality was a change in the sedimentation rate which was normal on day three and rose to 50 (NV 0-9 Wintrobe) on day nine. Twenty-four days after exposure it was back to normal.

Liver and kidney function remained within normal limits.

4.2 Local Reactions

The thumb and index finger of the right hand started with an

erythema reaction on day 18, that developed into a wet dermatitis on day 20, but eventually healed completely within the next 10 days. Clinical estimation of the radiation dose was difficult but judging by the reactions (5) about 1 000 rad was received.

The thumb and index finger as well as the middle finger of the left hand started with an erythema on day four, which developed into a wet dermatitis as well as blister formation after ten days. The hand was very painful and on day 20 the full skin thickness was shed. The healing of the fingers was very slow and incomplete and after three months were covered with atrophic skin that tended to break down repeatedly after minor trauma. This required amputation of the index and middle finger after 24 months. The estimated dose equivalent is over 5 000 rad.

Two days after exposure erythema developed over about 180 cm^2 of the anterior chest wall. This progressed to wet desquamation on day six, followed by necrosis in the centre. Healing started from the periphery and two months after exposure there was a 60 cm^2 (8 cm x 15 cm triangle) necrotic area surrounded by slightly atrophic depigmented skin, which was very painful. The necrotic area remained unchanged and excision and pedicle skin graft was required 18 months after the incident. By means of the Strandqvist method (6) the central necrotic part was estimated to have received between 5 000 and 10 000 rad. The adjoining area received between 1 000 and 2 250 rad. This is in remarkable agreement with the physical calculations, especially as the source was not stationary.

5. CONCLUSION

This accident has pointed out the inherent weakness of pneumatic radiographic equipment and, consequently, the further use of such equipment in South Africa has been prohibited. A system of log-sheets has also been introduced whereby the compulsory monitoring of a source inside its container must be recorded before storage.

The radiography firm as well as two of the employees have been criminally prosecuted and convicted.

The direct physical evaluation of radiation exposure proved its value in keeping the severely overexposed subject under medical surveillance.

6. ACKNOWLEDGEMENT

The expert assistance of Miss J I Thompson with the physical dose calculations is gratefully acknowledged.

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CONTAMINATION OF PERSONS OCCUPATIONALLY EXPOSED TO NATURAL RADIOACTIVITY IN A COAL FIRED POWER PLANT

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INTRODUCTION

On the premise that a study of a work population in a coal fired power plant with a higher than average radioactivity and sulphur content can be of value, this paper is presented to stimulate investigations of practical means of assessing general exposure levels of employees in basically nonradioactive industries.

Air samples were taken, surface contamination surveys of the work areas were made, and urine specimens and blood specimens were analyzed for radioactive contamination and mutagenic analysis.

METHODS AND EQUIPMENT

The personnel in the original study group were sixty employees assigned to operations and maintenance activities in the coal fired power plant. The operation was a continuous one employing three shifts of operational personnel. During the study period the men worked rotating eight-hour shifts, with a duty cycle of seven days and a rest cycle from 24 to 48 hours. The environmental exposure involved uranium, dust particles, ^{226}Ra, ^{210}Pb, ^{210}Po, mixed with fly ash bottom ash and slag, with additional contamination from ^{222}Rn in ambient air.(1,2) After coal combustion the concentration of these radionuclides was increased tenfold(3).

The possibility of chemical synergism was not taken into consideration. The major route into the body had to be by inhalation, so ^{210}Pb in urine was expected to be present. ^{210}Pb was sampled in 24 h urine. It was radiochemically separated (4). ^{210}Pb concentration in urine of a control group was taken as "blank level".

The chromosome analysis was performed in 48 lymphocytes cultures. Two hundred cells were analyzed for each subject (5). A control group was investigated at the same time with the test group. The test group was chosen on the ground of good health and none previous mining experience.

This paper is based on work performed under IAEA contract No 2346/RB

RESULTS

The results are presented in the following tables:

TABLE 1. CONTAMINATION OF WORKPLACES

WORKPLACE	µg U/m^2/24 h
1. Automatic control operator	360.0
2. Conveyer belt operator	23.0
3. Steam turbine machinist	7.1
4. Turbine operator	7.1
5. Mechanical engineer	7.1
6. Water controller	7.1
The ground 10 km^2 distant from the coal fired power plant	0.07

Around the place of the automatic control operator the radiation at the time of sampling and measurement was 0.240 mR/h. The highest radioactivity ever registered at the ash hopper was 0.5 mR/h. On the average the level of radioactivity was lower - 0.05 0.05 mR/h. The surveys have shown that the contamination at the workplaces corresponded with the frequency of chromosome aberrations in persons working in these same places as indicated in tb 1. in numerical order. The same is valid for automatic control operator with ^{210}Pb in urine. 14,47 pCi ^{210}Pb/l urine and the turbine operator 7,15 pCi/l. All the workers had a higher amount of ^{210}Pb in urine than the control group.

DISCUSSION

It is evident from the tables that the contamination with low levels of radioactivity in the coal fired power plant can be detected and tnat the the threshold is very low. This may be due to synergism with chemical contaminants. The contamination of workplaces correlates with urine and chromosome results.

TABLE 2. FREQUENCY OF CHROMOSOME ABBERATIONS IN PERSONS OCCUPATIONALLY EXPOSED TO NATURAL RADIOACTIVITY AND IN CONTROL SUBJECTS

occup.exp. persons	Age years	Years of exposure	Structural aberrations					Numerical aberrations		Total %
			Dicentric	Ring	Invers.	Break	Gap	Polyploid.	Hypodiploid	
Automatic control	39	3	3	1	1	7	3	-	-	7.5
Conveyer belt operator	35	7	0	0	1	6	5	-	-	6
Steam turb. machinist	52	24	2	1	0	6	3	-	-	6
Turbine operator	39	20	3	0	1	8	6	2	-	10
Mechanical engineer	26	3	1	0	0	5	5	1	2	7
Water controller	33	2	2	1	0	8	4	1	-	8
No Control										
1.	48		-	-	-	2	1	-	-	1.5
2.	35		-	-	-	3	4	-	1	4
3.	35		-	-	-	2	4	-	-	3
4.	39		-	-	-	4	4	-	1	4.5
5.	24		-	-	-	2	5	-	-	3.5
6.	28		-	-	-	3	2	-	-	2.5

TABLE 3. FREQUENCY OF ELEVATED ^{210}Pb LEVELS IN THE URINE OF PERSONS OCCUPATIONALLY EXPOSED TO NATURAL RADIOACTIVITY

occupationally exposed persons		Age years	Years of exposure in CFP plant	pCi ^{210}Pb/l urine
Automatic control operator		38	3	14.47
Conveyer belt operator		35	3	1.89
Steam turbine machinist		52	24	5.80
Turbine operator		39	20	7.15
Mechanical engineer (shift supervisor)		26	3	4.31
Water controller		33	2	2.29
No Control:	1.	48	-	1.06
	2.	35	-	1.24
	3.	35	-	1.10
	4.	39	-	1.07
	5.	24	-	0.70
	6.	28	-	0.86

The mean concentration of all control samples was used as blank level = 1.05 pCi ^{210}Pb/24 h urine, and it was deducted from all test samples.

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PERSONNEL DOSIMETRY SYSTEM BASED ON TLD AT PNC TOKAI WORKS

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INTRODUCTION

Power Reactor and Nuclear Fuel Development Corp. Tokai Works develops thermoluminescent dosimetry and applies it for routine personnel monitoring on a large scale in Japan. More than 3000 personnels per 3 months who work at nuclear fuel reprocessing plant, plutonium fuel fabrication plant and uranium enrichment facilities have been monitored with the PNC TLD badge and finger ring for past six years.

In order to measure the neutron dose which will be recieved by the personnel handling the plutonium of the order of Kg at the plutonium facilities and also the β absorbed dose recieved by handling high radioactive materials at the reprocessing plant, we developed the PNC TLD badge for whole body exposure and two types of finger ring for partial exposure, which consists of some TL elements. It is now possible to evaluate gamma, beta and neutron doses on routine basis by using the PNC TLD badge.

PNC TLD BADGE AND FINGER RING DOSIMETER

The external view of the PNC TLD badge and two types of finger ring is shown in Fig. 1 and the composition of badge in Fig. 2. The external size is 46 × 78 × 11 mm and a material of the badge case is ABS plastic.

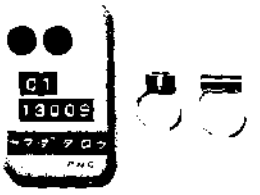

Fig.1 PNC TLD badge and finger rings

The TLD badge is composed of two TL elements ($CaSO_4$:Tm) for γ-ray, two thermal neutron sensitive TL elements (^6LiF + $CaSO_4$:Tm), one thermal neutron insensitive TL element (^7LiF + $CaSO_4$:Tm), six β-ray TL elements and one In foil.

All TL elements are products of Matsushita Electric Industrial Co. LTD.

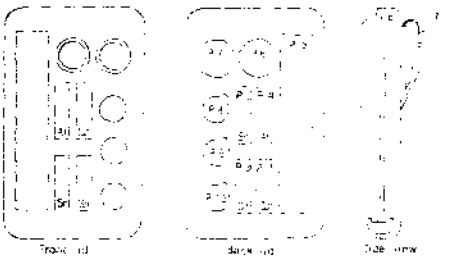

Fig. 2 Composition of PNC TLD badge

METHOD OF DOSE EVALUATION

γ-ray dose

γ-ray dosimeter (UD-200S) with energy compensation shield around the TL element ($CaSO_4$:Tm) composes the PNC TLD badge. Though the minimum detectable

amount of this dosimeter is several mrem, 10 mrem per 3 months is employed as a recording level for the routine individual monitoring.

When γ dose is calculated, background dose of each dosimeter is automatically substructed by computer system.

This dosimeter has the good energy characteristics, the low fading effect, the high sensitivity and the ease of handling for the routine individual monitoring.

β-ray dose

The TL elements (UD-100M8) consists of thin Al film base of 30μm and thermoluminescence material ($CaSO_4$:Tm) in thickness of 60um. Fig. 3 shows a composition of β-ray dosimeter and two sets of this dosimeter composes the FNC TLD badge.

The method of dose evaluation is the following.

Incident β and γ-rays are absorbed by TLD-1 and TLD-2 and in the TLD-3, only γ-ray is absorbed.

Consequently, the amount of thermoluminescence emitted from each TL element can be given, in case of the mixed β and γ radiation field, by the following equations;

$$L(1) = b_1(E)D_\beta + g_1(E)Dg + C \quad \ldots (1)$$

$$L(2) = b_2(E)D_\beta + g_2(E)Dg + C \quad \ldots (2)$$

$$L(3) = g_3(E)D_g + C \quad \ldots (3)$$

where

L(i) = amount of luminescence emitted from TLD(i)
bi(E)= sensitivity of TLD(i) for β-ray with Emax(MeV)
gi(E)= sensitivity of TLD(i) for γ-ray with E(MeV)
D_β = β-ray dose
D_γ = γ-ray dose
C = amount of noise luminescence

Accordingly, obtaining bi(E) and gi(E) previously, the γ and β ray doses can be calculated by the following equations

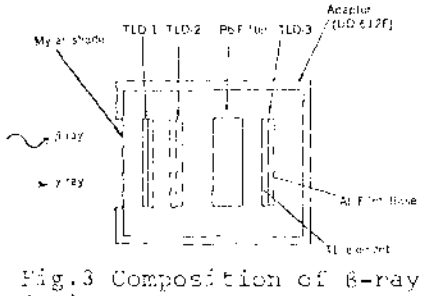

Fig.3 Composition of β-ray dosimeter

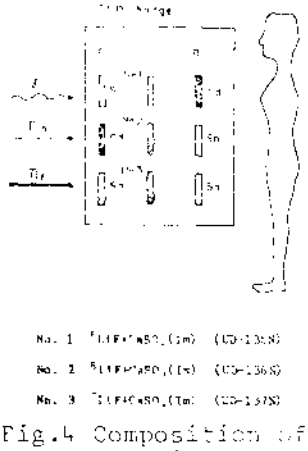

Fig.4 Composition of neutron dosimeter

$$D_\gamma = (L(3) - C)/g_3(E) \quad \text{(mrem)} \ldots (4)$$

$$D_\beta = \frac{(L(1)-C)-[g_1(E)/g_3(E)](L(3)-C)}{b_1(E)} \quad \text{(mrad)} \ldots (5)$$

Neutron dose

Neutron dosimeter incorporated in the PNC TLD badge consists of three TL elements and Cd Sn filters. Fig. 4 shows a composition of this dosimeter.

In the mixed radiation field of fast neutron, thermal neutron and γ-ray, the amount of thermoluminescent response of TL elements is interpreted by the following equations;

$$L(1) = n\sigma(E)\phi_{th} + G_1 \quad \ldots (6)$$

$$L(2) = n\sigma(E)(\phi'_{th} + \phi'_f) + G_2 \quad \ldots (7)$$

$$L(3) = G_3 \quad \ldots (8)$$

where

L(i) = amount of luminescence emitted from TLD(i)
n = proportionality constant
σ(E) = the $^6Li(n,\alpha)^3H$ cross section
ϕ_{th} = the incident thermal neutron flux
ϕ'_{th} = the backscattered thermal neutron flux
ϕ'_f = the tissue moderated fast neutron flux, that is albedo-neutron flux
G(i) = the luminescence caused by γ-ray to TLD(i)

If the effect of Cd and Sn filter to γ-ray is equal, G_1, G_2 and G_3 are equal, and so the thermal neutron dose (D_{th}) and the fast neutron dose(D_f) are calcurated by the following equations;

Table 1. The example of dose evaluation in the mixed exposure of β & γ rays

Source		Dose					Maximum Energy of β-ray (MeV)	Error* (%)
		Calculated			Measured			
β-ray	γ-ray	β(mrad)	γ(mrem)	β/γ	β(mrad)	γ(mrem)		
$^{90}Sr-^{90}Y$	^{137}Cs	153	37	5.9	206	200	2.3	26
$^{90}Sr-^{90}Y$	Xray (100 keV)	346	32	10	1350	90	2.3	25
^{204}Tl	^{226}Ra	446	47	9.5	410	90	1.2	4
$^{90}Sr-^{90}Y$	-	154	0	-	320	0	2.3	10
^{204}Tl	-	61	0	-	50	20	1.0	18
^{147}Pm	-	1930	0	-	1930	0	0.23	0
-	^{60}Co	0	190	-	0	210	-	-
$^{90}Sr-^{90}Y$	^{60}Co	324	310	1.0	480	340	1.1	48
^{147}Pm	^{60}Co	1000	330	3.1	1140	360	0.21	14
$^{90}Sr-^{90}Y$	^{60}Co	51	62	0.8	80	60	1.4	38
$^{90}Sr-^{90}Y$	^{60}Co	169	62	2.7	200	60	2.5	21
^{204}Tl	^{60}Co	122	62	2.0	60	80	2.3	51

* Error = $\frac{(\text{Calculated}-\text{Measured})}{\text{Calculated}} \times 100(\%)$

$$D_{th} = \frac{L(1) - L(3)}{K_1} = \frac{n\sigma(E)\phi th}{K_1} \quad (rem) \quad \ldots (9)$$

$$D_F = \frac{[L(2)-L(3)]-f[L(1)-L(3)]}{K_2} = \frac{n\sigma(E)\phi' f}{K_2} \quad (rem) \quad \ldots (10)$$

$$f = \frac{\phi' th}{\phi th} = \frac{L(2) - L(3)}{L(1) - L(3)} \quad \ldots (11)$$

Where, k_1 and k_2 are calibration constants to convert the luminescence into the dose equivalent, and f is the fraction effected by incident thermal neutron backscattered to the TLD(2), that is albedo-rate of thermal neutron.

Thus, it became possible to evaluate separately fast neutron, thermal neutron and γ-ray doses in the mixed radiation field.

This neutron dosimeter was calibrated by PuO_2, Cf and AmBe neutron sources and paraffin phantom.

RESULTS

Exposing the β-ray dosimeter to the mixed field where β-ray and γ-ray doses are known by calculation, the dose evaluation was practically done with the method described above. The results are given in Table 1.

As seen from Table 1, the larger the β/γ ratio, the evaluation precision for β-ray dose improve more. In the routin monitoring, we think that minimum detectable amount of β-ray dose is about 100 mrad.

The sensitivity ratio (b_1/b_2) to β-ray between TLD-1 and TLD-2 in β-ray dosimeter is shown in Fig. 5, using the various β-ray sources. By this figure, we are able to obtain the information on β-ray maximum energy and radionuclide.

Table 2 shows comparison between personnel monitoring data and radiation dose rate by survey meter in plutonium facilities. The ratio is almost the same, showing that the neutron evaluation method is adequate. We think that minimum detectable amount of neutron dose is 10mrem for thermal neutron and 20mrem for fast neutron in the routine individual monitoring.

Table 2. Comparison between personnel monitoring data and radiation dose rate

Sections or Groups	Personnel monitoring data by TLD Badge			Radiation dose rate by survey meter		
	n	Doses	D/d			D/d
1		120 / 80	1.5	1.8 / 1.3		1.8
2		120 / 80	1.5	3.0 / 2.9		1.7
3		120 / 80	1.5	2.2 / 1.5		1.4
4		70 / 40	1.8	2.5 / 5.0		0.5
5		60 / 30	2.0	2.5 / 1.5		1.7
6		90 / 70	1.3	4.0 / 1.6		2.5
7		90 / 50	1.8	3.0 / 3.0		1.0
8		80 / 40	2.0	4.5 / 2.0		2.3
9		40 / 30	1.3	6.0 / 2.0		2.0
10		70 / 40	1.8	3.5 / 1.6		2.2
Ave.			1.7			1.6

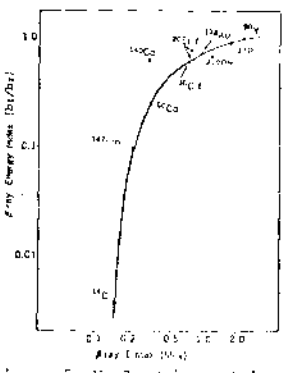

Fig. 5 Relationship between b_1/b_2 & Egmax

MEAN AND INDIVIDUAL RADIATION EXPOSURES OF THE STAFF OF THE KARLSRUHE NUCLEAR RESEARCH CENTER, 1969 - 1978

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INTRODUCTION

At the Karlsruhe Nuclear Research Center, radiation protection monitoring of the staff by far exceeds the stipulations of the Radiation Protection Ordinance of the Federal Republic of Germany. In order to obtain an overall view of the radiation exposure of the staff due to external radiation, for instance, not only occupationally exposed workers, but all members of the staff are monitored with solid state dosimeters (glass dosimeters).

AVERAGE ANNUAL DOSE

The aggregate dose including natural background, measured between 1969 and 1978, amounts to 50 man-Sv. Calculating the average of the annual radiation dose of 0.82 mSv received by persons not exposed occupationally as natural external background, a radiation dose of approximately 25 man-Sv is obtained in ten years, which is due to handling radioactive substances and ionizing radiation at the Karlsruhe Nuclear Research Center. This is an average annual occupational dose to radiation workers of 1.6 mSv.

Occupational external exposure follows a logarithmic normal distribution (Fig. 1). The line determined by the least squares fit indicates an average radiation dose due to occupational exposure of 1.59 mSv/a, which is in good agreement with the 1.6 mSv/a value calculated from the total dose.

DOSE DISTRIBUTION OVER DIFFERENT WORKING AREAS

A detailed insight into the radiation exposure of the staff is obtained by an analysis of the annual dose distribution according to individual areas of work. Table 1 very clearly shows that the members of services responsible for the treatment and final conditioning of radioactive waste - the waste treatment plant of the Karlsruhe Nuclear Research Center handles not only the radioactive waste of the Research Center proper but also that of two prototype nuclear power plants (D_2O, 50 MWe; LMFBR, 20 MWe) and a reprocessing plant (40 t/a) - are exposed to a higher radiation burden. The staff working in these services, although they make up but 5% of the total staff, received 42% of the whole occupational radiation dose over the years 1969 to 1978. It is not very surprising that the Health Physics staff rank second on this list - 2.2% of the staff, 8% of the dose - since this group of persons always work in the very front line. Further details are given in a recent paper (1).

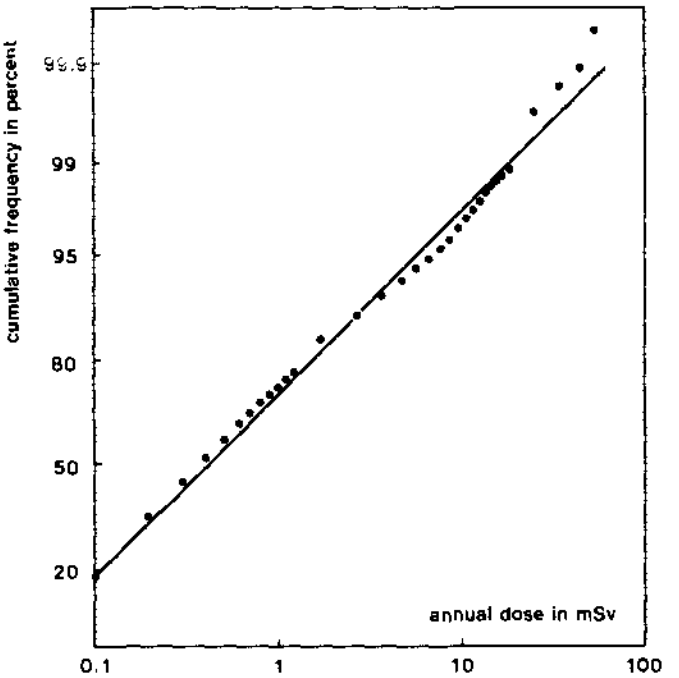

Fig. 1: Log-probability plot of annual doses (without background) Karlsruhe Nuclear Research Centre, 1969-1978

Table 1: Percent of dose and staff for different areas, Karlsruhe Nuclear Research Center, 1969-1978

Working area	Occ. dose man-Sv 1969-1978	Percent of dose	Percent of staff	Dose / Staff
Waste Handling	10.8	42	5.0	8.4
Health Physics	2.0	8	2.2	3.6
Cyclotron	1.4	5	1.8	2.8
Reactor	5.7	22	9.1	2.4
Chemistry	4.2	16	9.1	1.8
Supply Service	1.4	5	8.9	0.6
Physics	0.5	2	16.4	0.1
Biology	0	0	2.1	0
Other	0	0	45.4	0

ANALYSIS OF DOSES > 15 mSv/a

Analysis of all data for the decade from 1969 to 1978 indicates a number of 29,578 persons monitored by personnel dosimeters at the Karlsruhe Nuclear Research Center. 1903 cases had doses higher than 5 mSv/a, and only 349 cases had doses above 15 mSv/a. This means that, under the Radiation Protection Ordinance, only 6.4 % of the total staff of the Karlsruhe Nuclear Research Center would have to be classified as occupationally exposed persons, 5.2% of them falling under category B - annual dose, 5 to 15 mSv - and 1.2% falling under categroy A - annual dose, > 15 mSv.

The 349 cases of doses higher than 15 mSv/a in the period 1969 to 1978 are distributed over 124 individuals (see Fig. 2). On the average, these individuals were employed at the Karlsruhe Nuclear Research Center for 7.7 years. The mean annual occupational dose to these individuals amounts to 12.7 mSv. The highest average annual dose to anyone of these persons was 33 mSv (monitoring period: 5 years). In 19 cases involving 12 persons, annual doses were higher than 50 mSv, with a maximum of 68 mSv. Most of these cases occurred in 1972/73 during intervention and repair work at the facility for handling liquid radioactive wastes.

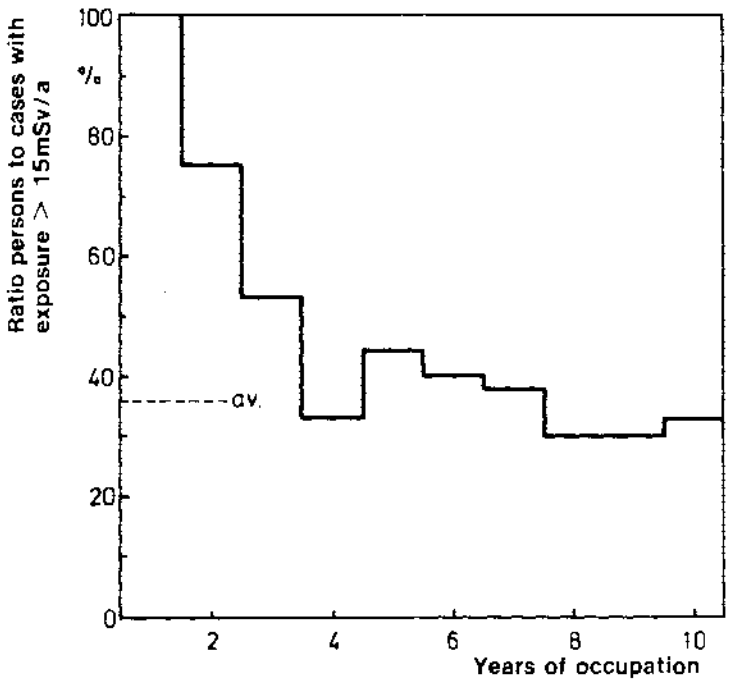

Fig. 2: Ratio persons to cases with exposure > 15mSv/a as a function of years of occupation, Karlsruhe Nuclear Research Center, 1969-1978

DATA OF BODY COUNTER AND LUNG COUNTER MEASUREMENTS

In addition to extensive personnel dosimetry all persons handling unsealed radioactive meaterials are monitored on a routine basis at six months intervals as the minimum either in the whole body counter or lung counter or by excretion measurements for incorporations. In the decade between 1969 and 1978, a total of 24,012 persons of the Karlsruhe Nuclear Research Center were monitored for incorporation in the body counter and the lung counter. Including multiple recordings in successive measurements, incorporations were found in 1628 persons. This is 6.8% of all persons measured. This figure includes the staff of two experimental nuclear power stations and the reprocessing plant. These facilities are located on the premises of the Karlsruhe Nuclear Research Center, but operated by independent companies. The 1628 measurements with positive findings yielded a total of 2147 incorporations (incorporations of one or more nuclides). Cs-137, Co-60, Ru-106/Rh-106, Co-58, I-131 and Mn-54 make up 90% and are the most frequent radionuclides in body counter measurements. In 96% of all cases the activity incorporated was below 1% of the MPBB values specified by ICRP (ICRP Publication 2). Only in five cases values above 100% MPBB were recorded. Incorporation monitoring by excretion analysis is performed by the Medical Department. Results have been published in a recent paper (2).

The radioactivity incorporated is not routinely converted into the body dose and added to the personnel dose determined by personnel dosimeters, since this is not generally required under the Radiation Protection Ordinance of the Federal Republic of Germany. However, it is evident from the data given that the dose statistics described in the preceding chapter is not substantially altered by an inclusion of the values of these measurements.

CONCLUSION

Our experience in radiation protection and monitoring at the Karlsruhe Nuclear Research Center and the data presented can be summarized as follows:

- ICRP recommendations on dose equivalent limits for workers are met in general.

- The average annual occupational exposure of radiation workers is remarkably low, 1.6 mSv.

- The number of persons with annual exposures > 15 mSv is about 1%. Only a few of these individuals exceeded the 15 mSv/a value repeatedly over the decade 1969 - 1978.

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PERSONNEL HAZARDS FROM MEDICAL ELECTRON ACCELERATOR PHOTONEUTRONS*

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INTRODUCTION

For medical accelerators, neutron penetration through the room entry door is the major personnel hazard. Most therapy accelerator rooms are designed with at least a rudimentary maze to avoid the use of massive doors. Often, however, the maze may be similar to those shown in Fig. 1. In Fig. 1, scale outline drawings of some medical electron accelerator rooms are shown where the authors have made neutron measurements outside the doors which were of different thicknesses and compositions. The results are tabulated in Table I. It should be noted that there can be significant dose equivalents (H) at the door when a maze is inadequate, and that all three components - fast neutron, thermal neutron, and neutron capture γ-rays - can be equally important. Also, these capture γ-rays are very penetrating; (TVL ≈ 5-7 cm of lead).

SIMPLE METHODS OF CALCULATING MAZE EFFECTIVENESS

For a good review of neutron penetration of mazes, the authors suggest Chapter 4 by Selph of Ref. 1. Most of the extensive work on mazes is not directly applicable to medical electron accelerators, however, for various reasons. Monte Carlo or albedo computer calculations have been shown to correctly calculate neutron maze penetration.

We have explored several simpler methods of predicting neutron penetration of a maze which do not rely upon computer codes or difficult calculations. Method 1 is an albedo method based upon the work of French and Wells (2), and is described as follows: On a room drawing, the portion of the walls, floor and ceiling that could be directly irradiated by neutrons from the accelerator, and then scatter the neutron directly to the door, are outlined, and their areas determined. An effective center, P, is chosen for each. The incident and reflected angles are measured from these points. Next, the dose albedo α_d (2), is used;

$$\alpha_d = \alpha(E_0) \cos^{2/3}\theta_0 \cos\theta \qquad \text{Eq. 1}$$

where θ_0 and θ are the incident and reflected angles, respectively, measured from the normal to the wall. For the range of neutron spectra from medical accelerators, a single value for $\alpha(E_0)$ of 0.11 can be used for concrete. Next, H is assumed to propagate according to the inverse square law for the distances, Ra and Rb, from the accelerator source to P and from P to the door, respectively. H at the door then is the sum of the individual contributions from each of the n illuminated areas; that is,

$$H = \sum_1^n \frac{H_0}{Ra_n^2} \cdot \frac{A_n \alpha_{d_n}}{Rb_n^2} \qquad \text{Eq. 2}$$

where H_0 is the dose equivalent at 1 meter from the source, A_n is the area of the nth shaded wall, and α_d is the dose albedo from Eq. 1 above.

Method 2 is one due to Kersey (3) which appears to be an empirical solution based on his measurements of several rooms. The details are given in Ref. 3; essentially he calculates neutron H using inverse squares at a mid-point in the maze which can 'see' the source, and then applies a maze attenuation based on the center line length of the maze and a value of 5 meters of maze length (irrespective of bends) to reduce H by a factor of 10. That is;

$$H = \frac{H_0}{Ra^2} e^{-Rb/2.17} \qquad \text{Eq. 3}$$

* Work supported by the Department of Energy under contract number DE-AC03-76SF00515.

where Ra and Rb are given in meters.

Method 3 is based on the "cookbook" approach of McCall, et al (4) and more exact calculations which show that most of the neutrons at a door have scattered from the wall directly opposite it. In essence, the number of neutrons entering a maze is calculated by the cookbook method. The area of the maze entrance is determined by the shaded wall (A' in Fig. 2), and a current albedo, a_c, from Fig. 7.9 of Ref. 4, is used. These reflected neutrons then are propagated down the maze according to inverse square, and the resulting current illuminates the cross sectional area at the end of the maze (A" in Fig. 2). The resulting fluence is then converted to dose equivalent. That is;

$$H = \frac{\Phi_o A'}{Ra'^2} \times \frac{a_c A'' C}{Rb^2} \qquad \text{Eq. 4}$$

where Φ_o is the fluence at a meter from the source, C is the fluence-to-dose equivalent conversion factor and Ra' is the distance from source to maze entrance. From Ref. 4, the average energy of the scattered neutrons at the door is about 100 keV, which implies a value for C of 2.4×10^8 n/cm^2-rem.

Though there are many assumptions in the above three methods that are difficult to defend from a physics standpoint, they do give reasonable answers as shown in Table II where the results are compared with measurements. For overall accuracy, methods 1 and 3 would seem to be the best choices.

IMPROVING EXISTING MAZES

The simplest solution for improving an existing maze is to add shielding to a door. Neutrons at the door will be attenuated by polyethylene with a dose equivalent TVL of about 4 cm. The outer portion should be borated to capture thermal neutrons. However, this will have little effect on capture γ-rays which contributed about 1/3rd of the total dose outside the doors of the rooms in Fig. 1.

Another solution is to improve a maze. From measurements made at the door of Fig. 2, we have found that all components of H were proportional to the area of the maze entrance, i.e., C-D. Once an accelerator is installed, a maze entrance can be reduced in size by hand-stacked shielding to that necessary for bringing in patients.

A third solution is to add a second hydrogenous door in the maze. An example is shown in Fig. 2 where a 5 cm polyethylene door was added as an internal maze. This arrangement gave reductions in H to 0.12 for thermal plus fast neutrons and 0.36 for capture γ-rays for a total reduction in H to 19%.

Adding a door such that it extends the maze wall, but is illuminated by the source is only about 50% as effective as adding the door across the maze where it is shadowed from the source by the maze wall, such as the doors shown in Fig. 2.

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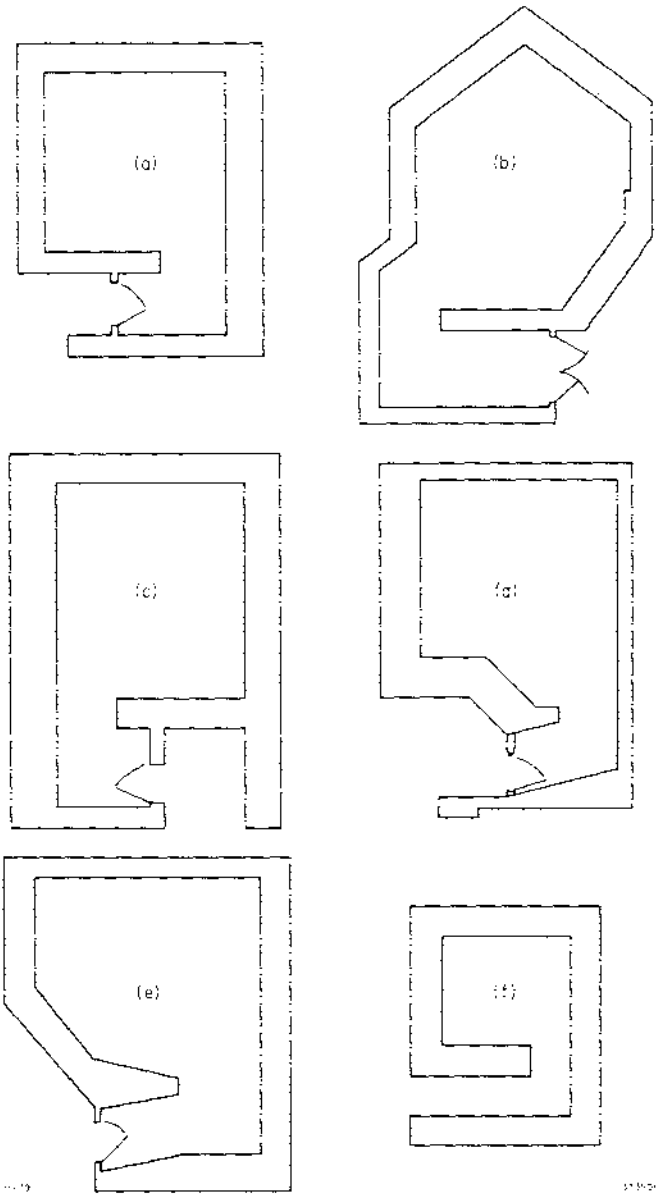

Fig. 1. Various rooms and mazes where measurements have been made.

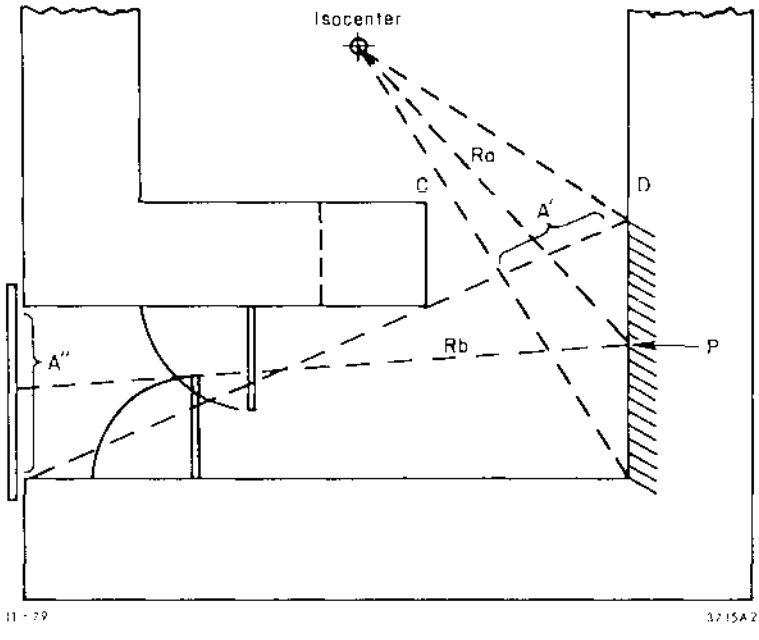

Fig. 2. Maze geometry showing various components described in text.

Table I. Measurements outside doors shown in Fig. 1.

Room	Dose Equivalent/Year (Work Load = 10^5 rads/week) Rem			
	Thermal Neutrons	Fast Neutrons	Neutron Capture Y-rays	Total
1A	4.7	3.7	5.7	14.1
1B	1.0	2.5	Not Measured	3.5 + Y-ray
1C	1.8	2.3	1.4	5.7
1D	2.3	0.7	4.6	7.6
1E	1.4	1.4	1.5	4.3

Table II. Comparison of calculational methods and measurements.

Room	Calculated Neutron Dose Equivalent / Measured Neutron Dose Equivalent		
	Method 1	Method 2	Method 3
1A	0.68	2.0	0.97
1B	1.4	5.5	1.4
1C	0.94	1.6	0.97
1E	1.7	4.5	1.8
1F (Cf)	1.0	2.0	0.74
1F (Cf/W)	0.71	1.4	0.73

INVESTIGATION LEVELS OF RADIOISOTOPES IN THE BODY AND IN URINE. CONSEQUENCES OF THE RECENT RECOMMENDATIONS ON THE ANNUAL LIMITS OF INTAKE

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The recommendations of Committee 2 of the International Commission on Radiological Protection (ICRP) concerning annual limits of intake (ALI) for workers (1) have recently been published. These limits differ in many cases from the maximum permissible annual intake (MPAI) recommended previously by the same committee (2,3). The new recommendations directly influence the derived health physics parameters, such as the acceptable total body burden and concentrations of radioisotopes in the urine.

Radioactivity in the body can be monitored routinely either by whole body counting or indirectly by urine analysis. Thus the monitoring laboratories have to know the relation between the activity in the urine or the body and the committed dose for calculating the latter from their measurements.

The activity of a radioelement in the body at any time t after intake of a unit of activity is given by its retention R(t):

$$R(t) = F_1 \exp(-\lambda t) \sum_i A_i \sum_j B_{ij} \exp(-\lambda_{ij} t)$$

where F_1 is the coefficient expressing the fraction of the intake transferred to the transfer compartment; A_i are coefficients expressing the fractions transferred from the transfer compartment to the i-th organ; B_{ij} are the coefficients of the linear combination of exponentials with decay constants λ_{ij} representing the retention in the i-th organ (1,4) and λ is the physical decay constant.

The amount of activity U(t) excreted in the urine at any time t after the intake of a unit of activity, is given by the first derivative of the biological retention function, multiplied by F_u the fraction of the excretion that is excreted through the urine (3,5):

$$U(t) = F_1 F_u \exp(-\lambda t) \sum_i A_i \sum_j \lambda_{ij} B_{ij} \exp(-\lambda_{ij} t)$$

When the decay constants λ are expressed in $days^{-1}$ then U(t) is the daily excretion. The average daily urine volume is 1.4 liters (5); thus division by 1.4 yields the concentration of the radioelement per liter.

The investigation level at any time t after intake was defined as the concentration of activity in the urine arising from an intake of 1/20 of an ALI (3). An analogous definition is used here for the total body investigation level. A computer code was written which

receives as input the various coefficients $F_l, F_u, A_i, B_{ij}, \lambda_{ij}, \lambda$ and the ALI (1,4) and calculates the investigation levels. Tables 1 and 2 list the investigation levels in the body and the urine of a few commonly used radioisotopes, as a function of time after ingestion. Different tables should be used for the case of inhalation.

TABLE 1. Total body investigation levels as a function of time after ingestion.

Isotope	Organ	Chemical form	Investigation level (μCi) days after ingestion 3	7	30	60
^{22}Na	T.B.*		14	12	4.0	0.97
^{42}K	T.B.		0.51	$2.2 \cdot 10^{-3}$	-	-
^{51}Cr	T.B.	Trivalent	9.1	6.5	2.0	0.73
	T.B.	Hexavalent	91	65	20	7.3
^{57}Co	T.B.	Inorganic	8.4	6.5	3.3	2.5
	T.B.	Organic	26	20	10	7.9
^{60}Co	T.B.	Inorganic	0.52	0.41	0.22	0.18
	T.B.	Organic	1.2	0.92	0.50	0.40
^{59}Fe	T.B.		3.9	3.6	2.5	1.6
^{65}Zn	T.B.		8.5	8.1	6.5	5.3
^{67}Ga	T.B.		0.17	0.056	$0.19 \cdot 10^{-3}$	-
^{75}Se	T.B.	Elemental	5.4	4.2	2.1	1.1
	T.B.	Inorganic	30	23	11	6.2
^{99}Mo	T.B.	Sulfide	1.1	0.39	$0.96 \cdot 10^{-3}$	-
	T.B.	Other	27	9.4	0.023	$9.0 \cdot 10^{-6}$
^{99m}Tc	T.B.		0.66	$3.5 \cdot 10^{-6}$	-	-
^{125}I	Thyroid		0.39	0.36	0.25	0.16
^{131}I	Thyroid		0.31	0.21	0.026	$1.7 \cdot 10^{-3}$
^{137}Cs	T.B.		5.0	4.7	4.0	3.3
^{144}Ce	L.L.I.*		$3.2 \cdot 10^{-3}$	$3.2 \cdot 10^{-3}$	$3.0 \cdot 10^{-3}$	$2.8 \cdot 10^{-3}$
^{226}Ra	B.S.*		$4.8 \cdot 10^{-3}$	$2.9 \cdot 10^{-3}$	$1.5 \cdot 10^{-3}$	$1.2 \cdot 10^{-3}$
^{232}Th	B.S.		$7.3 \cdot 10^{-6}$	$7.3 \cdot 10^{-6}$	$7.2 \cdot 10^{-6}$	$7.2 \cdot 10^{-6}$
^{238}U	B.S.	Hexavalent	$13 \cdot 10^{-3}$	$9.6 \cdot 10^{-3}$	$3.4 \cdot 10^{-3}$	$1.6 \cdot 10^{-3}$
	T.B.	Tetravalent	$7.2 \cdot 10^{-3}$	$5.4 \cdot 10^{-3}$	$1.9 \cdot 10^{-3}$	$0.9 \cdot 10^{-3}$
^{239}Pu	B.S.		$24 \cdot 10^{-6}$	$24 \cdot 10^{-6}$	$24 \cdot 10^{-6}$	$24 \cdot 10^{-6}$
^{241}Am	B.S.		$30 \cdot 10^{-6}$	$30 \cdot 10^{-6}$	$30 \cdot 10^{-6}$	$30 \cdot 10^{-6}$

* T.B. = total body; B.S. = bone surface; L.L.I. = lower large intestine

The following assumptions are inherent in the calculations:

a) The activity build-up time in the organs is assumed to be negligible compared to the decay time. Since the exponential approximation is in any case too crude to use for calculations for the first day no attempt was made to insert the build-up effect in the calculations. Therefore, this calculation should not be used for the first day.

b) The urinary excretion fraction F_u is taken as one constant for all organs and at any time. It will be possible to insert better approximations into the computer code when more biological information is available.

The computer code and Tables 1 and 2 give the levels in the urine and the body arising from an intake that corresponds to a particular committed dose. In the future we shall use the same tables and routines to reverse the procedure and calculate the committed dose from a measured activity.

TABLE 2. Investigation level in urine as a function of time after ingestion.

Isotope	Organ	Chemical form	Investigation level (μCi)			
			days after ingestion			
			3	7	30	60
^{3}H	T.B.	Water	81	62	12	1.6
^{22}Na	T.B.		0.46	0.39	0.13	0.032
^{32}P	T.B.		0.49	0.18	0.017	$1.3\cdot10^{-3}$
^{35}S	L.L.I.	Elemental	0.37	0.068	0.025	$7.2\cdot10^{-3}$
	T.B.	Other	9.3	1.7	0.63	0.18
^{36}Cl	T.B.		3.3	2.5	0.50	0.063
^{42}K	T.B.		$7.1\cdot10^{-3}$	$31\cdot10^{-6}$	-	-
^{45}Ca	T.B.		0.32	0.12	0.024	$8.1\cdot10^{-3}$
^{51}Cr	T.B.	Trivalent	0.21	0.11	$8.2\cdot10^{-3}$	$1.7\cdot10^{-3}$
	T.B.	Hexavalent	2.1	1.1	0.082	0.017
^{57}Co	T.B.	Inorganic	0.37	0.17	0.019	$6.0\cdot10^{-3}$
	T.B.	Organic	1.1	0.52	0.058	0.019
^{60}Co	T.B.	Inorganic	0.023	0.01	$1.2\cdot10^{-3}$	$0.42\cdot10^{-3}$
	T.B.	Organic	0.052	0.024	$2.8\cdot10^{-3}$	$0.96\cdot10^{-3}$
^{65}Zn	T.B.		0.014	0.012	$6.0\cdot10^{-3}$	$2.9\cdot10^{-3}$
^{67}Ga	T.B.		$2.2\cdot10^{-3}$	$0.6\cdot10^{-3}$	$0.8\cdot10^{-6}$	-
^{75}Se	T.B.	Elemental	0.12	0.039	$9.3\cdot10^{-3}$	$3.2\cdot10^{-3}$
	T.B.	Inorganic	0.67	0.21	0.05	0.017
^{85}Sr	T.B.	Titanate	0.021	$5.6\cdot10^{-3}$	$0.32\cdot10^{-3}$	$0.11\cdot10^{-3}$
	T.B.	Other	0.42	0.11	$6.4\cdot10^{-3}$	$2.2\cdot10^{-3}$
^{90}Sr	T.B.	Titanate	$2.7\cdot10^{-3}$	$0.76\cdot10^{-3}$	$55\cdot10^{-6}$	$27\cdot10^{-6}$
	B.S.	Other	$4.8\cdot10^{-3}$	$1.3\cdot10^{-3}$	$0.1\cdot10^{-3}$	$47\cdot10^{-6}$
^{99}Mo	T.B.	Sulfide	0.010	$2.1\cdot10^{-3}$	$5.0\cdot10^{-6}$	-
	T.B.	Other	0.24	0.051	$0.12\cdot10^{-3}$	-
^{99m}Tc	T.B.		0.074	-	-	-
^{125}I	Thyroid		$3.2\cdot10^{-3}$	$0.8\cdot10^{-3}$	$0.54\cdot10^{-3}$	$0.34\cdot10^{-3}$
^{131}I	Thyroid		$2.6\cdot10^{-3}$	$0.46\cdot10^{-3}$	$57\cdot10^{-6}$	$3.7\cdot10^{-6}$
^{137}Cs	T.B.		0.055	0.026	0.014	0.012
^{204}Tl	T.B.		0.87	0.58	0.059	$3.0\cdot10^{-3}$
^{226}Ra	B.S.		$32\cdot10^{-6}$	$8.6\cdot10^{-6}$	$0.6\cdot10^{-6}$	$0.3\cdot10^{-6}$
^{238}U	B.S.	Hexavalent	$0.18\cdot10^{-3}$	$86\cdot10^{-6}$	$16\cdot10^{-6}$	$4.3\cdot10^{-6}$
	T.B.	Tetravalent	$0.1\cdot10^{-3}$	$48\cdot10^{-6}$	$8.9\cdot10^{-6}$	$2.4\cdot10^{-6}$

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HEALTH PHYSICS DOCUMENTATION

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When dealing with radioactive material the health physicist gets innumerable papers and documents within the field of researching, prosecuting, organizing and justifying radiological protection.

One group of documentation comprises the description and publication of scientific experiences. It is used to give directions and obligations to health physics in international recommendations, national ordinances and standards.

On the other hand a lot of measurements are registered in order to prove the functioning of radiological protection itself.

Decisions when comparing scientific documentation with measured values still produce more papers.

Book-keeping and storage of radioactive and fissile materials cause more reports on stock, balance and modifications.

In consequence the German Radiological Protection Ordinance requires from the licensee the following written documentation:

- license to handle radioactive material
- appointment of responsible persons
- radiological protection directions
- accident prevention
- instruction of exposed persons
- measurement results: personal dose
  air
  water
  environment
- medical supervision
- radiation passport
- functional tests of radiation monitoring instruments
- inventory of radioactive material

Some figures of the Karlsruhe Nuclear Research Center as an example:

		Fixed data
occupied persons		4 000
occupational-exposed persons		2 500
health physics experts		150
registrars		10
institutes		40
atomic energy act licenses		300
fissile material balance areas		10
		Annual amounts
scientific publications in the field of health physics		70
measurements:		
survey of persons	- dosimeter exploitations	40 000
	- incorporation controls	5 000
working areas	- wiping tests	500 000
	- contamination controls	175 000
environment	- waste water	16 000
	- other activity measurements	45 000
book-keeping reports		2 500
Sum data per year		10^6

On the whole radiological protection has to cope with

10^6 data per year = 5 000 data per working day
= 500 data per hour
= 50 data per hour and expert
= 1 datum per minute and expert

The result of all data concerning occupational-exposed persons is

only 7 exceedings of personal dose limits
or 200 pecularities (= > 1.5 rem/a) within 10 years.

All data have to be recorded according to law.

Personal dose	30 years
radiation passport	during the whole professional life
material referred measurements	5 years

The necessity of keeping these values should be examined and all registration should be clearly arranged.

As a good motivation for health physics documentation we assume

- distribution of health physics experiences
- check of an positive result of a planned operation
- responsibility for the public.

As a bad motivation for documentation we assume

- demonstration of work
- justification of cost
- uncertainty of measurement results
- political reasons

More than 95 % of all data are useless and will never be used again according to the following decisions:

- Is the measurement correctly made?
- Is a dose limit exceeded?
- Did radiological protection function?

Health physics need more confidence in expert decisions.

By contrentation of

longer periods of time
collective dose
risk areas

individual data can be omitted.

Modern administration technics are a real help, i.e.

- office machinery
- computer documentation
- microfilms.

In the Karlsruhe Nuclear Research Center data of personal dosimetry are already dealt with by computer documentation, for material referred values this will be provided in the near future.

Summary:

- Health physics are practical work
- We cannot do without an accompanying documentation
- The scope of nowadays' used documentations could be reduced by critical selection
- documentation is an instrument not a justification of the health physicist.

ANALYSIS OF MEDICAL OCCUPATIONAL EXPOSURE TO IONIZING RADIATION ON TAIWAN DURING PAST TWO DECADES

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Analysis of the data was obtained from the very inception of the centralized laboratory for personnel desimetry service operated by the National Tsing Hua University on Taiwan of Republic of China from 1960 to 1979 for the yearly occupational exposure to ionizing radiation. During the 20 yr monitoring period, analysis was performed with reference to (1) medical occupational exposure, (2) maximum and average yearly dose-equivalent, (3) range of dose-equivalent, (4) percentage of maxmium permissible dose-equivalent, (5) number of workers including sex and age, (6) detailed quarterly analysis for the years 1977 ~ 1979, (7) types of radiation sources, (8) estimation of genetically significant dose-equivalent, and (9) estimation of marrow and leukemia significant dose-equivalents.

ESTIMATION OF GENETICALLY SIGNIFICANT DOSE-EQUIVALENT

The genetically significant dose-equivalent (GSD) can be calculated with the following formula:

$$D = \frac{\sum_j \sum_k \left(N_{jk}^{(F)} W_{jk}^{(F)} d_{jk}^{(F)} + N_{jk}^{(M)} W_{jk}^{(M)} d_{jk}^{(M)}\right)}{\sum_k \left(N_k^{(F)} W_k^{(F)} + N_k^{(M)} W_k^{(M)}\right)} \qquad (1)$$

where

- D = annual genetically significant dose.
- N_{jk} = number of individuals of age-class k, subjected to class j exposure, i.e., either radiographic or fluorographic X exposure or other sources.
- N_k = total number of individuals of age-class k.
- W_{jk} = future number of children expected by an exposed individual of age-class k subsequent to a class j exposure.
- W_k = future number of children expected by an average individual of age-class k.
- d_{jk} = gonad dose per class j exposure of an individual of age-class k.
- (F) = female.
- (M) = male.

ESTIMATION OF MARROW DOSE AND LEUKEMIA SIGNIFICANT DOSE-EQUIVALENT

The leukemia significant dose-equivalent (LSD) estimated here was calculated according to a weighting factor. This factor takes into account the shape of the time-incidence curve of radiation-induced leukemia, and the survival statistics for the various age groups in the population. The data for this factor were obtained from the leukemia incidence among the Hiroshima A-bomb survivors located within 3000 m from the hypocenter at the time of the bomb (1).

The population mean marrow dose equivalent and the leukemia dose-equivalent are calculated by the following formulas;

$$D_p = \frac{\sum_j \sum_k N_{jk} d_{jk}}{\sum_k N_k} , \qquad D_\ell = \frac{\sum_j \sum_k N_{jk} d_{jk} L_{jk}}{\sum_k N_k} \tag{2}$$

where

D_p = population mean marrow dose-equivalent (MMD).
D_ℓ = leukemia signigicant dose-equivalent.
L_{jk} = significant factor of leukemia incidence for an average individual of age-class k, subjected to a class j exposure.
d_{jk} = mean bone-marrow dose-equivalent per class j exposure of an individual of age-class k.

Jones (2) has made measurements with a phantom of the dose received by critical body organs relative to the exposure at the conventional position of a personal dosimeter. His experimental results were used to calculate the GSD and LSD.

The average expected children census data released by our National Health Administration are that for age 18∿45 yr, the average expected children are 2.30 for both sexes, and for age above 45 yr, the average expected children are 0.015 for both sexes.

RESULTS AND DISCUSSION

The distribution of Tsing Hua film badge users in each county or city is shown in Fig.1 where the numeric numbers indicate the number of hospitals or clinics being monitored. The results of film badge monitoring expressed in terms of the ranges of dose-equivalents in μSv's *vs.* number of personnel monitored during the past two decades (1960∿1979) are shown in Fig.2; the majority is in the undetectable range whether they are in diagnostic radiology (DR), radiotherapy (RT), or nuclear medicine (NM). Taking into account the average expected children for an average individual of age-class k subjected to a class j exposure and the critical body organ dose-equivalent relative to the exposure of film badge, we obtain the GSD by Eq.(1) and the results of calculation are shown in Fig.3. The LSD is shown in Fig.4 by Eq.(2) where the MMD is not shown but it differs from LSD by a weighting factor. The average annual dose-equivalents of all personnel being monitored are shown in Fig.5 and the maximum dose-equivalents for an individual ever detected are shown in Fig.6. Some of the results are not shown in the figures such as sex of workers, detailed quarterly analysis for the years 1977∿1979, and the percentage of maximum permissible dose-equivalent due to space limit and next in importance. The types of radiation source are mainly 602 diagnostic X-rays (75∿250 kVp), 13 ^{60}Co units, a few LINAC's, and ^{131}I and ^{99m}Tc as the main radioisotopes.

The dose-equivalents recorded by some of our film badges at the early period such as in the years of 1964 and 1968 in diagnostic radiology are questionable due to the uncertainty in dose evaluation at very high dose level. It happened that a few minor incidents occurred during those years. A good indication over the last decade was that the incidents of over-exposure were reduced though the number of radi-

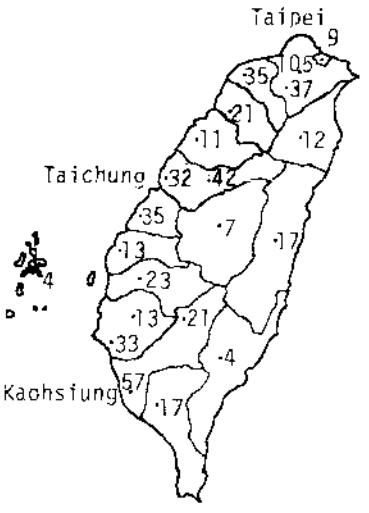

Figure 1

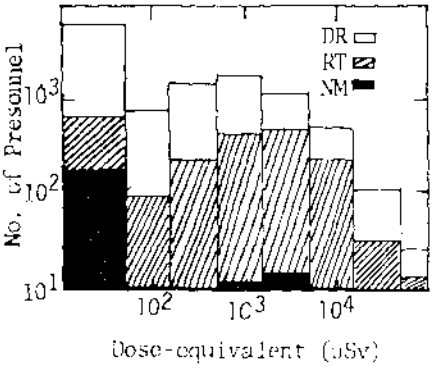

Figure 2

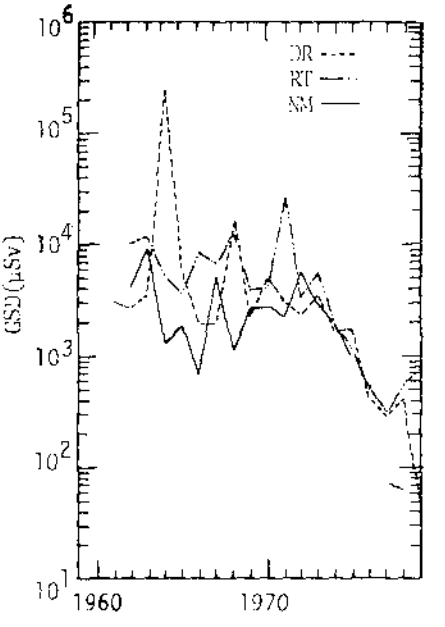

Figure 3

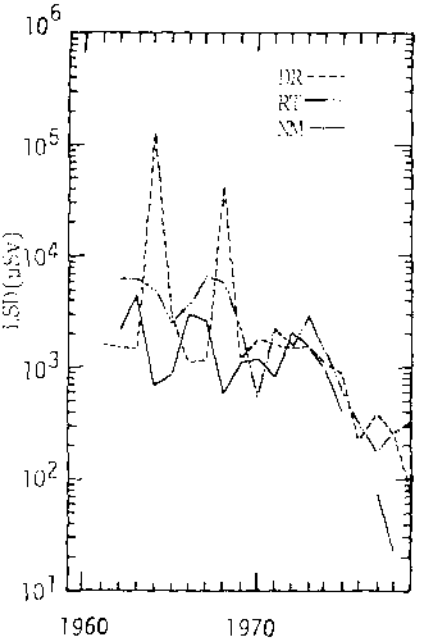

Figure 4

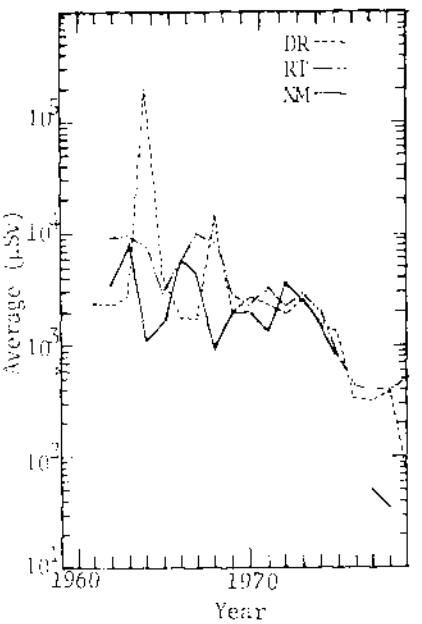

Figure 5

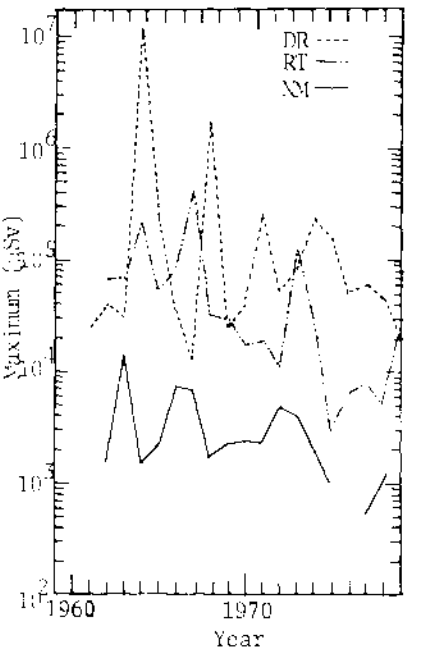

Figure 6

ation workers increased rapidly. Another good indication over the last 8 yr was that the average year dose-equivalent received by all medical radiation workers was decreasing and well below 1/10 of the maximum permissible dose equivalent.

In contrast to other countries experience, females were less than 10% of the total number of radiation workers monitored. The majority of radiation workers is in the ages 18~45. It is noted that significant high exposure were found among diagnostic radiology. Radiotherapeutic personnel were second.

The annual GSD and LSD were decreased though the number of radiation workers monitored increased every year. This may be attributable to the implementation of a nationwide training program on radiation protection for all radiation workers in the Republic of China. Each session on the training course is formulated for one week and several courses are held by the joint efforts of the Atomic Energy Council and the National Health Administration every year. In addition, the promulgation of medical radiation control regulation in February 1973 and enacted thereafter may also be an important factor to reduce the exposure.

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RISK RATIOS FOR USE IN ESTABLISHING DOSE LIMITS FOR OCCUPATIONAL EXPOSURE TO RADIATION

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1. Introduction

One function of the Licensing Branch of the South African Atomic Energy Board is to establish authorised limits for occupational exposure to radiation incurred at nuclear installations. The approach adopted to establish these limits has been described previously by Winkler and Simpson (1). Briefly, this involves an investigation of fatality rates experienced in South African industry in order to determine a representative average industrial fatality risk. From these investigations and considerations of future trends an average level of risk has been selected on which to base limits for occupational exposure. This level of risk combined with an appropriate risk coefficient enables one to determine an average annual dose limit.

Figure 1. Average Fatal Accident Rate 1965 - 1970

Industrial Class	Average Annual Fatality
Agriculture and Forestry	3,28 E-04
Fishing	2,33 E-03
Mining	8,35 E-04
Building and Construction	7,68 E-04
Food, Drinks, Tobacco	2,27 E-04
Textile	2,87 E-05
Wood	3,90 E-04
Printing, Paper	7,30 E-04
Chemicals	2,49 E-04
Leather	1,75 E-05
Glass, Bricks, Tiles	3,42 E-04
Iron, Steel	2,23 E-04
Diamonds, Asbestos	2,05 E-04
Trade, Commerce	1,73 E-04
Banking, Finance, Insurance	6,78 E-05
Transport	1,00 E-03
Local Authorities	4,85 E-04
Personal Services	5,70 E-05
Entertainment, Sport	1,52 E-04
Professional Services	4,25 E-05
Medical Services	4,75 E-05
Educational Services	1,34 E-04
Charitable, Religious Trade Organisations	1,15 E-04
TOTAL	3,03 E-04

It was recognised that studies of industrial fatality rates, as illustrated in Figure 1, do not provide information on possibly elevated levels of risk experienced by an individual or subgroup within a given occupation. It was decided therefore to conduct an investigation of industrial fatality rates in an attempt to identify the extent to which the risk of fatality for a given occupation e.g. electricians, varies throughout industry. From this investigation a ratio of maximum to average occupational fatality risk representative of industry in South Africa was derived.

Having established an average annual dose limit the risk ratio can be used to establish a maximum individual annual dose limit. The average annual dose limit would be applicable to the occupationally exposed population of a particular installation whilst the maximum limit would be applicable to individual employees on an annual basis. This paper describes the results of initial investigations conducted thus far.

2. Sources of Data

A report on accident statistics is issued annually by the Workmen's Compensation Commission of South Africa. Although this reports fatality rates by industry rather than by occupations, such information is stored in the Commission's data bank (2). The second source of data utilized was the Manpower Survey No 11 (1975)(3) issued by the South African Department of Labour.

A cursory examination of the data revealed only six occupations in which fatal accidents occured in six or more classes of industry in any one year. Figure 2 lists the particular occupations and the number of industries in which fatal accidents occurred.

Figure 2. Occupations Selected for Investigation

Occupation	Number of Industries Having Fatal Accidents
Watchman	7
Foreman/Supervisor	7
Driver	13
Labourer	19
Electrician	6
Fitter	8

Figure 3. Annual Deaths of Electricians by Industry

Industry	Average Number of Deaths Annually
Mining	1,33
Building and Construction	2,67
Glass, Brick Tiles	0,33
Local Authorities	4,0
Professional Services	0,33
South African Railways	2,0
Total	10,67

Although the two sources of data use different classification systems it was possible to do a cross correlation so

as to ensure the proper matching of the two sets of data.

Data on fatal accident rates taken from the Workmen's Compensation Data Bank were available in sufficient detail for the years 1971, 1972 and 1973. The data for these three years was averaged and taken as representative of annual occupational fatality rates. A typical example is shown in Figure 3 which lists the average number of annual occupational fatalities experienced by electricians in the various classes of industry.

The data of industrial populations derived from the Department of Labour manpower survey was tabulated for the particular occupations in the industries concerned. Figure 4 shows the populations of electricians by industries. It is recognized that the population data was obtained for 1975 whilst the fatality rates were obtained for the years 1971, 1972 and 1973. However, as the objective of the study is to derive a ratio of occupational risk throughout industry, using these two sets of data is acceptable since the ratio of persons employed in various occupational groups throughout the various industrial classes is not expected to vary appreciably over such a short time.

Figure 4. Industrial Population of Electricians

Industry	Population
Mining	4 253
Building and Construction	4 098
Glass Brick Tiles	588
Local Authorities	3 942
Professional Services	208
South African Railways	4 459
All Industries	32 000

Figure 5. Annual Fatality Rates for Electricians

Industry	Average Annual Fatality Rate
Mining	3,13 E-04
Building and construction	6,52 E-04
Glass Brick Tiles	5,61 E-04
Local Authorities	1,01 E-03
Professional Services	1,59 E-03
South African Railways	4,49 E-04
All Industries	3,33 E-04

3. Occupational Fatality Rates

Fatality rates for the various occupational groups e.g. electricians, fitters etc., were derived for the industrial classes using the three year average fatality figures and the total industrial population for the particular occupational group. Figure 5 presents an example of the data, detailing the average annual fatality rate for the occupational class electrician in the various industries chosen for investigation. The industrial class with the highest fatility rate for a particular occupation was identified as indicated by an asterisk.

4. Results

The maximum and average fatality rates experienced by the various occupation groups are tabulated together with the populations involved and the ratios of maximum to average fatality rates for each occupation in Figure 6.

Figure 6. Fatality Risk Rates and Maximum to Average Ratios

Occupation	Total Industrial Population	Average Annual mortality risk	Maximum Annual Mortality risk	Maximum/Average risk ratio
Watchman	29 000	$6,21 \times 10^{-4}$	$7,83 \times 10^{-3}$	12,6
Foreman/Supervisor	64 000	$1,41 \times 10^{-4}$	$1,03 \times 10^{-3}$	7,3
Driver	105 000	$1,11 \times 10^{-3}$	$2,72 \times 10^{-3}$	2,45
Labourer	1 507 000	$7,25 \times 10^{-4}$	$5,32 \times 10^{-3}$	7,34
Electrician	32 000	$3,33 \times 10^{-4}$	$1,59 \times 10^{-3}$	4,77
Fitter	40 000	$2,66 \times 10^{-4}$	$7,33 \times 10^{-3}$	27,6

5. Discussion

Having derived ratios of maximum to average fatality rates for various occupations within different classes of industry, a method is needed to combine the values in order to establish what ratio could be considered representative of industry as a whole.

An examination of the values obtained displays quite a variation from 2,45 for drivers to 27,6 for fitters. Examination of the data concerning the maximum risk groups, shows the maximum risk rate for fitters was derived from a group of only 45 persons in which only one fatality occurred in the three year period. The validity of this figure as being representative of a maximum risk rate is therefore questionable. The second highest risk for fitters $3,71 \times 10^{-3}$ deaths per person per year derived from a group of some 898 persons is probably more representative and results in a ratio of 13,9.

Combining the ratios obtained by simply taking the mean results in a value of 10,34. The mean of the risk ratios derived using the reduced value for the occupational group of fitters results in a value 8,06. If one combines the ratios by deriving a weighted mean to take account of the working population, the resulting values are 7,55 and 7,15.

6. Conclusion

The study described appears to indicate that a ratio of 7 is representative of the maximum to average risk of fatal accidents whithin industry in South Africa. A postulated interpretation of the result is that since persons in a particular occupation move about in industry, apparently accepting the variations in risk of incurring a fatal accident the occurence of similar ratios within one industry would presumably also be acceptable.

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COMPARAISON DES COUTS MARGINAUX DE PROTECTION DANS LES CENTRALES THERMIQUES CLASSIQUES ET NUCLEAIRES

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L'objet de cette communication est d'évaluer les risques sanitaires (pour le public) et les coûts de protection relatifs aux centrales thermiques classiques et nucléaires, puis de les comparer.

L'EVALUATION DES RISQUES SANITAIRES ET DES COUTS

Dans le but de déterminer les coûts marginaux de protection, nous allons évaluer les risques et les coûts dans des situations concrètes, autant que possible.

A - Le cas des centrales thermiques classiques

La création d'une zone dite de " protection spéciale " contre la pollution atmosphérique dans la périphérie de Paris a entraîné depuis l'année 1979 l'obligation de recourir à du fuel résiduel à basse teneur en soufre (BTS) dans les centrales thermiques entre le 15 Novembre et le 15 Mars [1] .

Pour les trois centrales concernées, Vitry-sur-Seine, Gennevilliers et Saint-Ouen, la quantité annuelle de combustible BTS (2% S) consommée est de 220 000 tonnes environ; le restant de l'année, 440 000 tonnes de fuel ordinaire sont brûlées(3,5% S).

Le surcoût annuel lié à l'emploi du fuel BTS est de 9,4 10^6 F (1978) [1] .

L'impact en terme de réduction de la mortalité annuelle liée à la pollution atmosphérique a été quantifié à l'aide de deux modèles de relation exposition-risque. Cet impact a été calculé sur l'agglomération parisienne, dont l'effectif est d'environ 7.10^6 habitants.

Il faut souligner qu'il n'existe pas actuellement de relations exposition-risque communément admises dans le domaine de la pollution atmosphérique par les oxydes de soufre et les poussières. De nombreux auteurs ont suggéré des relations dont aucune n'échappe en définitive à la controverse. Dans cette situation, il nous a paru utile d'effectuer les calculs à l'aide de deux modèles élaborés aux Etats-Unis ,conduisant à des résultats sensiblement différents, afin d'illustrer le caractère incertain de telles évaluations.

a/ Le modèle de Liu et Yu [2]

L'équation de base est la suivante :

$$M = N \times 10^{-4} \times \exp\left(1{,}25 - \frac{25{,}52}{C_{fond}+c}\right) - \exp\left(1{,}25 - \frac{25{,}52}{C_{fond}}\right)$$

avec C_{fond} : concentration annuelle SO_2 de fond
c : concentration annuelle SO_2 ajoutée par la centrale
M : mortalité annuelle supplémentaire attribuable à la concentration ajoutée c
N : effectif soumis à la concentration c.

Dans le cas présent, la concentration annuelle ajoutée par les centrales a été calculée en nous fondant sur une étude de diffusion décrite dans l'étude du CEPN [2] . On trouve que la concentration annuelle moyenne apportée par les centrales est d'environ 1 $\mu g/m^3$ sur la région parisienne. La concentration de fond est d'environ 110 $\mu g/m^3$ d'après [1] . La mortalité annuelle associée aux trois centrales ne brûlant que du fuel à 3,5% S est, pour un effectif de 7.10^7 habitants : 4 morts/an. Le passage au combustible BTS (2% S) en hiver permet d'éviter 0,6 mort/an.

Un tel modèle implique que la mortalité est associée à la pollution atmosphérique de fond, exprimée en moyenne annuelle.

b/ Le modèle de Buehring et al. [2]

On peut supposer que la mortalité est imputable essentiellement à des pointes de pollution et qu'il existe un seuil en-deçà duquel la mortalité est nulle. Dans le modèle à seuil considéré ici, l'équation de base est $F(i) = - 0{,}0404 + 76\ 10^{-6}\ S_{24}(i)$ avec

$S_{24}(i)$: concentration en SO_2 sur 24 h pour le jour $i \geq 530\ \mu g/m^3$.
$F(i)$: fraction de la mortalité totale attendue au jour i qui est associée à l'existence de $S_{24}(i)$.

En nous fondant ici encore sur l'étude [2] , nous trouvons :
sans fuel BTS : 10 morts/an
avec fuel BTS : 5 morts/an.

On constate que le modèle de Buehring conduit à des valeurs sensiblement supérieures.

c/ Coût marginal dans les centrales thermiques classiques

Compte tenu des résultats précédents, le coût de l'effet évité est situé entre 2.10^6 F et 16.10^6 F.

B - Le cas des centrales nucléaires

L'étude du CEPN [3] porte notamment sur un réacteur PWR de 1300 MWe situé en bordure de fleuve.

Le réacteur entraînerait, en l'absence de toute option de traitement des effluents, une mortalité annuelle de 0,18 mort/an. Notons que cette valeur ne reflète pas le risque sanitaire total lié au réacteur dans la mesure où l'étude [3] ne calcule les effets que dans un rayon de 100 km autour du réacteur. En réalité, quatre options de traitement sont appliquées : TEG (stockage des effluents gazeux), DR (traitement des drains + effluents chimiques), BF (pré-

geage des iodes), LV (stockage effluents des laveries).

TABLEAU 1. Performances et coûts du traitement des effluents radioactifs

Option	Coût annualisé	Nombre d'effets annuels évités par l'option
TEG	158	0,16
DR	740	0,01
BR	48	≠ 0
LV	130	≠ 0

On constate qu'aux deux options BR et LV, correspondent des coûts de l'effet évité pratiquement infinis. Ceci tient au fait que ces options ont été introduites essentiellement dans un but de minimisation des équivalents de dose reçus par les individus du groupe critique. Par contre, les deux autres options TEG et DR ont une influence sur la réduction du risque collectif, lequel passe de 0,18 mort/an avant traitement, à 0,02 après TEG et à 0,01 après DR. On peut dès lors considérer que le coût du dernier effort de protection collective est celui de DR. Le coût marginal est donc de l'ordre de $\frac{0,740\ 10^6}{0,01} = 10.10^6$ F/effet évité en arrondissant.

COMPARAISON ET CONCLUSION

Notons que ni la nature, ni la gravité des affections liées aux deux types de nuisances ne sont directement comparables et par conséquent l'indicateur de risque gagnerait certainement à être exprimé, par exemple, en années de vie perdues plutôt qu'en mortalité annuelle supplémentaire.

Il est cependant possible de procéder à une première comparaison des résultats obtenus plus haut.

TABLEAU 2. Comparaison des risques et des coûts

	Centrales thermiques au fuel production 2,8 TWh/an	Centrale PWR 1300 MWe production 10,4 TWh/an
Risque résiduel avant protection	4 - 10 morts/an	0,2 mort/an
Risque résiduel après protection	3,4- 5 morts/an	0,005(?) mort/an
Coût marginal de protection	2 - 16 10^6F/mort évité	70 10^6 F/mort évité
Coût annualisé de protection	9,4.10^6 F/an	1,1.10^6 F/an

Compte tenu des incertitudes associées aux évaluations des coûts et des risques, le tableau précédent suggère essentiellement que le risque sanitaire et le coût annualisé de protection par KWh produit sont plus élevés dans les centrales au fuel.

La comparaison entraîne des conclusions moins nettes pour les coûts marginaux de protection, à cause des incertitudes. Si l'on fixait le budget total de protection (fuel + nucléaire) au niveau actuel, le principe d'égalisation des coûts marginaux de protection conduirait à un transfert de ressources de protection du nucléaire vers le fuel. Si l'on devait évaluer le budget total de protection, la situation serait indéterminée : on pourrait aussi bien accroître le budget total afin que le coût marginal de protection dans le secteur fuel rejoigne celui du nucléaire, ou bien diminuer le budget total de sorte que le coût marginal dans le nucléaire soit réduit au niveau existant dans le secteur fuel.

L'application d'un autre principe, celui de l'égalisation des risques sanitaires résiduels par unité d'énergie produite, conduirait (pour des raisons en fait contingentes) à des conclusions identiques aux précédentes.

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EVALUATION QUANTITATIVE ET COMPARATIVE DES RISQUES LIES AUX CENTRALES NUCLEAIRES

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Malgré l'ampleur des études sur l'évaluation du risque nucléaire et sa comparaison avec d'autres risques, le monde scientifique et non scientifique continue de s'intéresser à cette question, sans doute à cause de l'extension envisagée du programme de production d'énergie d'origine nucléaire. Il est certain que ces comparaisons ont servi plus ou moins implicitement de point de repère pour les experts chargés de proposer les normes d'exposition. Mais en pratique, les industries nucléaires exposent l'homme à des doses bien inférieures aux limites autorisées et il est utile de situer le risque calculé correspondant parmi les autres risques auxquels l'homme se trouve chaque jour exposé.

BASES DE L'EVALUATION DU RISQUE

Rappelons que les organismes nationaux et internationaux autorisés, tels que la CIPR, l'UNSCEAR et le Comité BEIR, publient périodiquement des études, destinées soit à recommander des règles de radioprotection, soit à faire le point des connaissances et à évaluer les risques. Le caractère aléatoire des affections radioinduites retenues, cancers ou anomalies génétiques, est reconnu par tous et un certain consensus s'est dégagé sur le choix de l'hypothèse de la relation linéaire entre dose et effet. La plupart des experts reconnaissent cependant que cette base de calcul est majorante aux faibles doses, la surestimation du dommage pouvant être de 2 à 4 selon certains, voire de 10 pour d'autres [1, 2].

L'estimation moyenne du risque par mSv est actuellement de 10 cas de cancers par million de personnes (sur la durée de vie). Lorsque la dose est répétitive (cas de l'industrie nucléaire) l'estimation est faite pour la dose cumulée, en ne tenant pas compte du mode de distribution de la dose dans le temps, ce qui amène en général un coefficient de sécurité supplémentaire dans l'évaluation.

Pour un travailleur en centrale nucléaire, par exemple, qui serait exposé de 18 à 65 ans à 5 mSv par an, et qui aurait ainsi cumulé en fin de vie professionnelle 235 mSv, la probabilité qu'il aurait de faire un cancer du fait de l'irradiation serait de $2,35 \times 10^{-3}$, à comparer à la probabilité spontanée, qui est actuellement supérieure à $2,2 \times 10^{-1}$.

Ceci signifie qu'une exposition professionnelle répétitive d'un travailleur à la dose estimée comme la dose moyenne, amènerait théoriquement à une augmentation de la probabilité qu'il aurait de mourir

de cancer, dont la limite supérieure serait de 1%. Il s'agit bien là, répétons le, d'un calcul majorant, par suite des hypothèses de base retenues.

CAS DES TRAVAILLEURS

Les risques auxquels les travailleurs sont exposés peuvent être de deux natures : soit des risques d'accident, se soldant par un handicap à long terme ou par la mort, soit des risques de maladies spécifiques de la profession, pouvant ou non conduire à la mort, à plus ou moins bref délai. La comparaison peut être établie selon plusieurs critères, parmi lesquels le critère du risque de mort annuel rapporté à un million de travailleurs et celui du nombre d'années ou de jours de vie perdus.

Si l'on considère les risques de mort par accident des différentes professions, on constate que la moyenne à EDF est légèrement inférieure à celle de l'ensemble des industries et que les accidents de cause purement électrique ne constituent qu'environ le quart de la totalité (tab. 1). Dans les centrales thermiques et nucléaires, la mort par accident constitue un risque encore inférieur à la moyenne de toutes les industries. Parmi celles-ci, les moins exposantes sont celles du textile et du vêtement, les plus dangereuses étant celles des travaux publics, des mines, des docks, de la pêche en haute mer.

Pour l'ensemble des accidents graves, l'âge moyen de leur survenue est aux alentours de 30 ans ; aussi, que l'accident soit mortel ou non, les années de vie perdues ou altérées sont-elles nombreuses.

Il peut en être tout autrement pour les maladies professionnelles, dont la plupart demandent de nombreuses années pour se manifester. Les risques spécifiques d'origine nucléaire appartiennent à cette catégorie. Aussi pensons nous qu'il faille comparer ces risques surtout à ceux des autres maladies professionnelles.

E. Pochin a donné des valeurs de risques de mort inhérents à diverses professions, d'après des statistiques britanniques ou américaines[3]. Le risque théorique nucléaire demeure inférieur à celui constaté dans d'autres professions (tab. 2). Le risque de la silicose des mineurs n'y est pas porté, or la prévalence de cette affection est estimée à l'heure actuelle, en France, de l'ordre de 80 000 par million de mineurs.

J. Reissland a récemment proposé de baser la comparaison sur le nombre de jours de vie perdus[4]. Le risque encouru par le travailleur du nucléaire, exposé depuis l'âge de 20 ans et durant 45 ans à la moyenne de dose de 5 mSv par an, serait ainsi 20 fois inférieur à celui du mineur.

CAS DE LA POPULATION

Quant à l'individu de la population, soumis en moyenne à 10^{-3} mSv par an du fait de l'énergie nucléaire, il recevrait en 70 ans 0,07 mSv supplémentaire, ce qui se traduirait par un risque de cancer de 7×10^{-7} qui représenterait, environ, le trois millionième du risque spontané. Pour une personne du groupe "critique", supposée exposée en continu

à une dose maximale de 0,05 mSv par an, elle aurait ainsi à la fin de sa vie un risque surajouté de mourir de cancer de $3,5 \times 10^{-5}$, soit environ le deux dixmillième du risque spontané. Approximativement, on pourrait alors prévoir environ 1 mort annuel par cancer dû au nucléaire dans la population française et environ 1 cas tous les trois ans, parmi les travailleurs, si ceux ci étaient 10 000. Indépendamment du caractère spéculatif

TABLEAU 1. Comparaison des fréquences de morts par accidents professionnels (par million de travailleurs et par an).

Moyenne toutes industries	≈ 250 *	
EDF (toutes causes)	190	
" (origine électr. seule)	40	
Vêtement	16	(1968-72)
Papeterie et imprimerie	30	"
Textiles, cuirs et peaux	44	"
Metaux	115	"
Chimie	155	"
Agriculture	360	"
Carrières	370	"
Bâtiment et travaux publics	490	"
Mines	≈ 500	"
Transport, manutention	540	"
Aviation civile	≈ 1 000	(1974)
Docks	1 020	
Chalutiers	1 636	

* accidents de trajet compris

TABLEAU 2. Exemples de risques de cancers professionnels (morts par million de travailleurs et par an en France et à l'Etranger).

Professions	Risque	localis. des cancers
Chaussure	130	fosses nasales
Bois (machines)	700	fosses nasales
Amiante (filage, tissage, isolation)	≈ 3 000	poumon, plèvre
Cadmium	14 000	prostate
Nickel (raffinage)	6 600 - 15 500	sinus et poumons
Nucléaire (an 2000)	50 (théorique)	cancers et leucémies

de ces calculs, de tels résultats n'ont de sens que dans la mesure où ils sont comparés à tous les autres effets cancérogènes et mutagènes de l'environnement. Une tentative de comparaison est faite par les recherches fondamentales sur la rad-équivalence des nuisances chimi-

ques, qui doivent être completées par les enquêtes épidémiologiques. Celles-ci auraient permis de relever, par exemple, à Londres, de 1948 à 1962, plus de 7 000 morts imputables à des accidents aigus de pollution soufrée et particulaire. Par contre, le nombre de décès qui serait dû à la pollution chimique et particulaire de fond ne peut être évalué. Par ailleurs, la pollution médicamenteuse provoque des morts en France au nombre d'environ 5 par million de personnes et par an.

TABLEAU 3. Comparaison de quelques causes de mortalité en France avec les risques théoriques.

Causes de mort	Risque de mort annuel (10^6 pers)	Risque indiv. relatif	Risque calculé par mSv
Mortalité (toutes)	10 500	1	
Maladies (toutes)	9 500	0,9	
Cancers totaux	2 300	0,22	10^{-5}
Bronch - pulm.	400	$3,8 \cdot 10^{-2}$	$2 \cdot 10^{-6}$
Leucémies	80	$7,6 \cdot 10^{-3}$	$2 \cdot 10^{-6}$
Tabagisme	1 300	$1,2 \cdot 10^{-1}$	
Alcoolisme	400	$3,8 \cdot 10^{-2}$	
Accidents totaux	950	$9 \cdot 10^{-2}$	
" de la route	240	$2,3 \cdot 10^{-2}$	
Noyades	60	$5,7 \cdot 10^{-3}$	
Incendies	40	$3,8 \cdot 10^{-3}$	
Acc. médicaments	5	$4,7 \cdot 10^{-4}$	
Electrocution	4	$3,8 \cdot 10^{-4}$	
Asphyxies (gaz)	≈ 1	$\approx 9 \cdot 10^{-5}$	
Foudre	0,5	$4,8 \cdot 10^{-5}$	
Chute d'avion	$\approx$ 0,02	$\approx 1,9 \cdot 10^{-6}$	

La comparaison à des risques d'activités volontaires serait encore plus frappante, comme à celui du tabac (1 300 par million), à celui des accidents en vacances (environ 180 par million de personnes) et en partie à celui des accidents de la route (240 par million). Le risque inhérent à la production d'électricité d'origine nucléaire en fonctionnement normal reste hypothétique et de toute façon négligeable par rapport à tous les autres risques auxquels l'homme se trouve exposé dans le vie courante.

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RADIATION AND THE PERCEPTION OF RISK IN THE U.S.A.

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A complete approach to radiation protection must take into account perceived risk--the subjective judgment of the risk involved in the technology which may produce an exposure to radiation. The necessity for including consideration of perceived risk is briefly outlined below.

Calculated risks, sometimes called objective risk estimates, are almost always associated with uncertainties and therefore usually require subjective judgments about probabilities, consequences, and their distributions. Gaps in knowledge about the technology and its impacts and interrelationships within the total environmental system are often large, making evaluation of radiological consequences a difficult and speculative art. Lack of specific experience with a technology makes assignment of probabilities especially difficult and speculative. Multiplying these uncertainties makes their product even more suspect.

The point being made is not the obvious one, namely that risk assessment is a challenging enterprize, but rather that the bases of risk calculations are often founded on expert perceptions of technological hazards. Thus, when dealing with discrepancies between "perceived" versus "calculated" risk, it is vital to recall that more correctly we are talking about "lay" versus "expert" perceptions of risk. To be sure, the basis on which each group forms perceptions may be quite different, and that some approaches may be more defensible than others, but the two sets of perceptions probably obey the same psychological laws since we are all human beings.

Understanding of the processes which control perceptions of risk is largely lacking and research efforts are still only in a formative state. The notion of "acceptable risk"--that is, the establishment of absolute levels of risk acceptable to the public--may well be a will-o'-the-wisp. We say this based on clear evidence that comparable risks (in terms of expected value) are rarely equally acceptable. Risk is measured not only by its own value but also by the value of what the risk is taken for, or in other words, the benefit. Thus, people make an integrated judgment to answer the question: "Is this risk worth taking?"

With respect to nuclear power, people can and have made this judgment. In the United States, approval and disapproval levels have been surprisingly constant over the years. This can best be illustrated by opinion poll data taken from Harris and Cambridge Reports (1, 2). These two major and long term studies have over the years 1975-79 shown a majority or plurality of Americans supporting nuclear power plants (Fig. 1). However, the accident of March 28, 1979, at the Three Mile Island Nuclear Generating Station (TMI-2) appears to

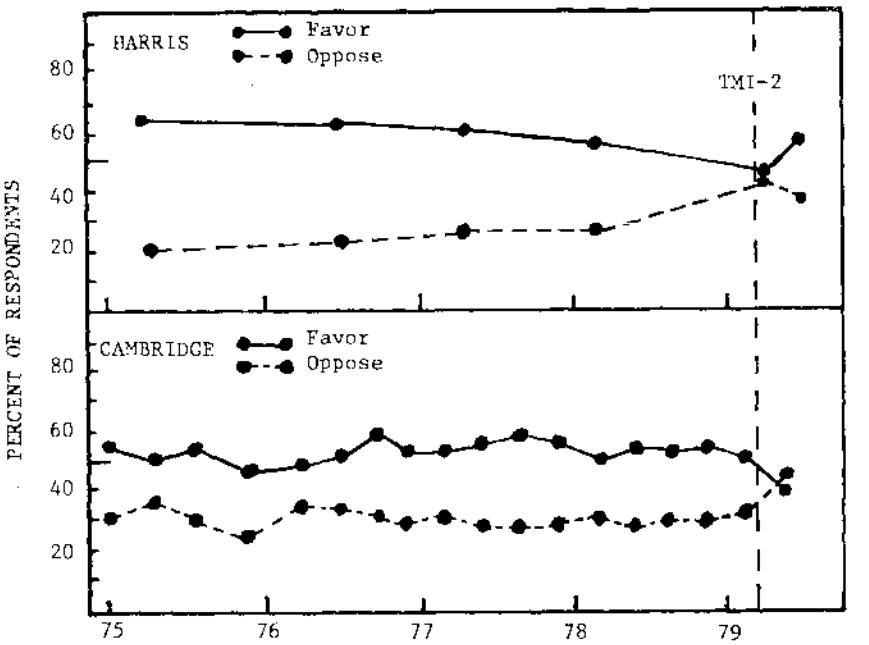

Figure 1. Percent of respondents favoring or opposing continued nuclear power plants in two surveys over time (1975-1979).

have had a strong effect on public opinion: polls taken shortly after the accident showed for the first time nearly equal opposition and support for nuclear power. Clearly, the extensive publicity given to the event coupled with the appearance of poor institutional response had a pronounced effect on the judgment of risk worth taking. However, a curious effect is seen in the post-TMI-2 data of Harris: a rebound effect. There appears to be movement toward previous levels of approval (and disapproval) as a more realistic assessment of TMI-2 (and its risk) is made. The rebound is not seen in the Cambridge data but may yet appear.

This difference in the two studies may be attributable to slight technical differences between the two studies. The Cambridge data generally show a lower level of approval and a higher level of disapproval than that of Harris. Cambridge asks, "Do you favor or oppose the construction of more nuclear power plants?" Harris asks, "In general, do you favor or oppose the building of more nuclear plants in the United States?" The slight wording differences in these essentially identical questions may be highly significant. The first question could well be taken as specific to the community, while the second is clearly non-specific and in general. Moreover, as noted, it is possible that the rebound effect will be seen in the Cambridge results at a later date. If so, the lag may reflect a different threshold evoked by the different wording of the question.

There are also curious differences in the data between male and female respondents. In general, women in the United States have been much less supportive of nuclear power. This difference may have been exaggerated by the TMI-2 accident. Figure 2 shows the level of

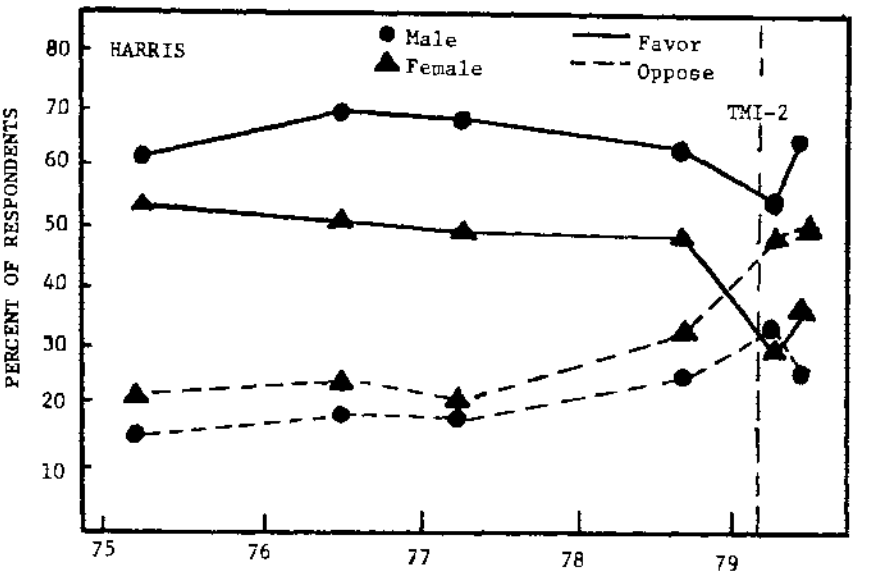

Figure 2. Male vs. female opposition or support for continued nuclear power over time (1975-1979).

nuclear power support and opposition for men and women as obtained by Harris. For women, the TMI-2 incident has led to majority opposition to nuclear power and the rebound has not occurred with sufficient vigor to return support to previous levels. Whether this effect is permanent is yet to be seen. The point is that there are likely important differences in the perception of risks worth taking, not only between sexes, but across other demographic groups as well. This may reflect different emphasis placed by various groups on widely held human values such as economic security or a world at peace rather than a differing ability to accurately perceive risk.

The usual approach to risk assessment, as taken by technical persons including health physicists, is to look at risk associated with a specific technology (nuclear power generation, for example) along with associated benefit and make a judgment on the basis of the "evidence" or available data that the risk is or is not worth taking. This approach is logically defensible as a self-contained problem. However, members of the public may approach the problem differently. They may ask about calculated risk, but go further yet. They likely place the proposition in the context of their own lives, asking these kinds of questions: Is the new benefit proposed (or probable benefit) desirable to (or, perhaps, advantageous to) me? Is the new risk additional to other risks I'm already bearing? The cumulative risk and benefit may be the important risk calculus. The marginal or incremental value of risk and benefit is probably more relevant to the protection of the public than an absolute level.

The foregoing discussion of nuclear power and public perception of risk leads logically to the question: What are the implications for radiation protection? There are, of course, serious and

significant implications and practical lessons for the health physicist. The first of these might be termed the small increment problem, a situation in which any one small risk (or increment of dose) may be acceptable in isolation but not in a cumulative sense or when considered in conjunction with other risks. Thus, the first incremental dose may be of no concern, while the nth increment may be totally unacceptable. Thus it is the cumulative dose or risk that must be considered, and this may or may not be perceived as time related.

The second implication might be termed the last technology in problem, in which a new and "unproven" technology (such as nuclear power) may have to bear a greater burden for the increment problem than "old" familiar technologies. Applying historically acceptable risk levels or evaluations may simply be not acceptable for this new technology. To some extent, this may be underlain by a fear of the unknown, or by a lack of experience on the part of the general public, or by a desire to avoid recently perceived errors in risk evaluation made with the "old" technology. Whatever the reason, the net effect is to require less risk for comparable benefit.

An important lesson lies in the area of standards setting: the radiation protection standards setting process needs to be sensitive to demographic differences in valuing risk and benefit. Broad representation of these groups in the standards setting process would be ideal, but, failing that, awareness of the differences is essential. Thus, national standards can logically and realistically differ among countries and from international recommendations; indeed, if the small increment problem is fully considered, there might even be regional and local standards which differ from national or international recommendations. An interesting example of the demographic differences in risk perception is the generally higher level of acceptance of nuclear power plants by the communities in which they have been located.

Finally, there needs to be a differential application of standards to those technologies whose benefits are trivial versus those whose benefits are great. Clearly, this is a scientific and political judgment, cutting both broadly and deeply, and including many considerations beyond the scope of the typical standards setting body. If, however, compelling scientific evidence could be presented that a safe (i.e., nonzero) or threshold level exists, then this differential approach could be avoided.

In summary, the radiation protection standards setting process should ideally consider not only hard scientific data but also perceived risks and benefits of the technology under consideration. In addition, the demographic character of the affected group needs to be factored in as well. Public opinion data can provide useful and otherwise unobtainable information to the standards setting body.

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Physical Characteristics of the Japanese in Relation to Reference Man

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Introduction

Quantitative description of physical and other characteristics of the human body is one of the basic data for estimating dose equivalent and calculating annual limit on intake of radionuclides in line with the ICRP Recommendations(1,2). ICRP Reference Man is practically based on the data reported for populations the habitat of which is Western European and North American areas although it is stated that "it is neither feasible nor necessary to specify Reference Man as representative of a well-defined population group"(3). In order to be more realistic and quantitative in the dose equivalent estimation and ALI calculation in Japan populations of which are different from those of European and North American countries in physical dimensions and other aspects, standard or reference values for the Japanese, i.e. mass and dimensions of body and organs, and the daily intake, distribution, and metabolism of elements have been being studied particularly on the basis of more recently obtained data(4). These values that has been obtained are compared with those of authorized ICRP Reference Man(3) with the intention of establishing "reference Japanese man" and contributing to improvement of models of man used in radiation protection. A calculation of dose equivalent commitment and annual limit on intake of radioiodine using the obtained data for the Japanese adult is referred. The present approach may be of importance regarding the demographic contribution of Asian populations to the world population.

Methods

The weight and size of the body and organs were measured in autopsy for subjects who died of sudden death, mostly from traffic accidents. Autopsy was carried out 12-24 hr after death at the Tokyo Medical Examiners Office, Otsuka, Bunkyo-ku, Tokyo. From protocols of 10,598 cases recorded during the period 1971-1976, 2,880 cases were selected, using as a criterion those individuals who had been physiologically normal. Data for subjects having pathological changes in any single organ which tend to cause a change in the normal weight of organs were rejected in the statistical study carried out at the National Institute of Radiological Sciences. The obtained results, therefore, are considered to represent the normal Japanese in the strict sense.

Separate from this, bone and other autopsy tissue samples were collected from a few districts including Tokyo during the period 1961-1976. Chemical analysis are being carried out using techniques

of trace analysis developed for the present purpose, including vacuum drying and plasma-dry ashing of autopsy tissues, atomic absorption and emission spectroscopy and also partly employing rooms specifically designed for use in trace analysis, and, in some part, being assisted with simultaneous multielement analysis techniques.

A low dose of 74 nCi of I-131 were orally administered as sodium iodide to two voluntary, normal adult male subjects and the uptake and retention of ingested radioiodine in the thyroid gland was measured using the N. I. R. S. 8 in.-dia. NaI(Tl) Human Counter in a collimation mode.

Results and Discussion

The total body weight and length as presently studied was in good agreement with the published data for the Japanese of different ages reported by the Ministry of Health and Welfare and the Ministry of Education for essentially the same period. The data(5,6) is plotted as a function of age and compared with the data used for ICRP Reference Man in Fig. 1. Average total body weight of the Japanese adult male and female was approximately 59 and 51 kg, respectively being comapared with approx. 75 and 58 kg for the European and American counterparts, respectively. Average length of the total body was approx. 165 and 155 cm for the male and female, respectively which are compared with approx. 175 and 164 cm for the Caucasian counterparts, respectively.

The mean weight of twelve organs for males and of eleven organs for females were obtained as a function of postnatal age as well as the relative weight of organs with respect to total body weight. Mean organ weight of the normal Japanese adult is shown in Table 1 along with the literature data for the Europeans and Americans being referred to in the ICRP Publication(3) for comparison. Remarkably lower weight of the spleen was noticed in the Japanese male, and, however, comparable values were seen in children between the data. Apparently higher values were found for the thymus and pancreas in the Japanese adult as compared to those referred to by ICRP(3). The

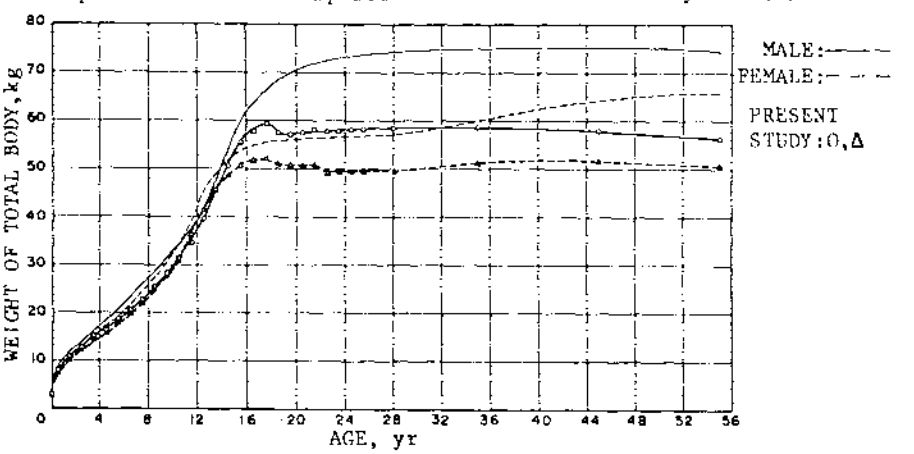

Fig. 1. Total body weight of Japanese as compared to that of Europeans and Americans.

weight of the spleen in the Japanese adult male was 127 g which is compared with the literature value of 192 g(3). The mean weight of the thymus and pancreas was 31.7 and 135 g in the Japanese adult male which are compared with those referred to by ICRP, 19.7 and 96.1 g for the European and American counterpart, respectively. Comparative weights were observed for other organs, e.g. adrenals, hypophysis or pituitary gland, testes, thyroid gland, heart, kidneys, brain and lungs. The mean liver weight in the Japanese was smaller than that referred to in Reference Man(3). These may be explained by a possible correlation between the weight of some organs and the total body weight. Similar results were seen for the Japanese adult female as also shown in Table 1. Relative weight of a given organ to the total body weight found in the present study was apparently larger than that given in Reference Man for most of organs studied except spleen. This should be stressed because it suggests that estimating weights of organs of a certain population using relative weights reported for different populations is not necessarily valid.

Table 1. Weight of organs and total body weight proposed for reference Japanese adult as compared with ICRP Reference Man values.

Organ	Japanese adult				ICRP Reference adult			
	Male		Female		Male		Female	
	(g)	(%)	(g)	(%)	(g)	(%)	(g)	(%)
Adrenal glands	13	(0.026)	13	(0.028)	14	(0.020)	14	(0.024)
Brain	1400	(2.6)	1300	(2.8)	1400	(2.0)	1200	(2.1)
Heart	350	(0.66)	280	(0.61)	330	(0.47)	240	(0.41)
Kidneys	330	(0.59)	280	(0.60)	310	(0.44)	275	(0.47)
Liver	1600	(2.9)	1400	(2.9)	1800	(2.6)	1400	(2.4)
Lungs	1100	(2.1)	900	(1.9)	1000	(1.4)	800	(1.4)
Pancreas	135	(0.24)	100	(0.24)	100	(0.14)	85	(0.15)
Pituitary gland	0.6	(0.0010)	0.6	(0.0013)	0.6	(0.00086)	0.7	(0.0012)
Spleen	130	(0.23)	120	(0.26)	180	(0.26)	150	(0.26)
Testes	33	(0.058)	—		35	(0.050)	—	
Thymus	30	(0.054)	25	(0.053)	20	(0.029)	20	(0.034)
Thyroid gland	19	(0.034)	17	(0.035)	20	(0.029)	17	(0.029)
Total body weight	60000		51000		700000		58000	

The present data for most of organs studied exceeds that reported for the normal Japanese in 1952(7). Improvement in the general nutritional condition may have had possible influences. Differences between the reported data for pathological cases and normal cases are also pointed out.

A comparison of *per capita* daily consumption of principal nutrients and categorized foodstuffs between Japan, and European countries and the United States. Calcium as well as fat is factor of two less consumed in Japan than in the latter countries, even on the unit body weight basis. Fourty-eight per cent of the total fat comes from vegetables and other plants. Consumption of dairy products, meat and milk is factor of two to four lower in Japan and, however, fish meat is consumed at a rate approx. four times larger than in European and American countries. Consumption of many kinds of sea algae and their products is regarded also characteristic to the Japanese population.

For the elemental composition of the human body and daily intake, data is being accumulated and, in the present paper, strontium is referred in relation to calcium. Strontium in bone has been extensively studied using samples of various ages including gestational stage and the mean strontium to celcium concentration ratio in the adult(20-49 yr) bone was found 0.51 mg Sr per g of Ca. Skeletal mass was estimated tentatively using the ICRP assumption, 0.14 of the total body weight until a valid data for the Japanese will be available. Apparently larger skeletal strontium content in the Japanese adult male than the ICRP estimate for European and American counterpart. This may be due to the dietal Sr/Ca ratio which was found factor of two higher in Japan and this may be related to the characteristically low consumption of milk and dairy products.

Table 2. Skeletal content and daily intake of Ca and Sr in adult.

	Present work	ICRP(3)
Skeletal content:		
Calcium	840 g	1000 g
Strontium	430 mg	320 mg
Daily intake:		
Calcium	0.54 g	1.1 g
Strontium	2.3 mg	1.9 mg

Regarding the remarkably high level of natural iodine intake in the Japanese up to between 500 and 1000 μg per day per person which has been reported, an isotope dilution effect is expected. Fraction of I-131 incorporated in the thyroid gland, K_2(8) was found 0.11 in one subject taking normal meals containing a kind of kelp and other marine algae and their products throughout the experiment. In the other subject for whom the intake of these kinds of foodstuffs had been restricted as far as possible for two weeks until the oral administration of I-131, K_2 was found 0.33. Biological half-life was estimated to be 29.9 and 40.4 days for the former and the latter subject, respectively. It is suggested that the thyroidal uptake rate can be estimated as 0.15 to 0.20 and that the biological half-life is probably estimated as 35 days in the Japanese adult. The data is considerably smaller than that adopted by ICRP(8).

Establishment of reference Japanese man data will be useful for one to calculate more realistic dose-equivalent commitment and also annual limit on intake for radioactive isotopes. A result of such calculation employing a unique transformation method developed to apply the MIRD absorbed dose fraction data to an individual of an arbitrary physique and the weight of organs and other data obtained in the present study(9) shows a lower annual limit on intake for I-131 in the Japanese adult as compared with that calculated for the MIRD phantom.

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COMPETING RISK THEORY AND RADIATION RISK ASSESSMENT

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The paper presents a summary of the research in competing risk theory during the last decade and applies some new statistical procedures to estimate cumulative distribution functions (c.d.f.), force of mortality, and latent period for radiation-induced malignancies. It is demonstrated that correction for competing risks influences the shape of dose-response curves, estimates of the latent period, and of the risk from ionizing radiations. We show the equivalence of the following concepts: force of mortality, hazard rate, and age or time specific incidence. This equivalence makes it possible to use procedures from reliability analysis and demography for radiation risk assessment. Two methods used by reliability analysts — hazard plotting and total time on test plots — are discussed in some detail and are applied to characterize the hazard rate in radiation carcinogenesis. C.d.f.'s with increasing, decreasing, or constant hazard rate have different shapes and are shown to yield different dose-response curves for continuous irradiation. We point out that the absolute risk is a sound estimator only if the force of mortality is constant for the exposed and the control group. Dose-response relationships that use the absolute risk as a measure for the effect turn out to be special cases of dose-response relationships that measure the effect with cumulative incidence. We explain how life tables — a popular demographic tool — should be used to calculate the risk to a population from a risk estimate obtained from another exposed population.

INTRODUCTION

The basic idea of competing risk theory was first outlined by Daniel Bernoulli in his "Memoir" published in 1760 (1). In this publication he attempted to answer the question: Is mandatory vaccination against smallpox beneficial, and what are the quantitative effects of vaccinations on the survival experience of a population (1)? Each individual was considered as exposed to two risks: death from smallpox and death from other causes. The analogous "modern" question is: Is exposure to a certain dose of ionizing radiation detrimental, and what are the quantitative effects of a radiation exposure on the survival experience of a population? Each indidivual is again subject to two competing risks: death from radiation-induced cancer and death from other causes not related to the radiation exposure. During the last few decades competing risk theory developed rapidly because of the need to answer very similar questions in reliability analysis. There a device can fail due to different failure modes, and it is desired, for example, to estimate the probability that the device will fail due to one particular failure mode not later than a certain time t. The life of a person or a device is therefore a non-negative random variable T [$P(T < 0) = 0$]. In the

field of radiation risk assessment, we are interested in the estimation and comparison of $F_T(t,D)$, the cumulative distribution function of T for a control population radiation dose (D = 0) and an exposed population (D > 0). T is the life length from exposure to death or diagnosis of a radiation-induced malignancy. In the following section we will introduce some concepts related to F_T and discuss their estimation and connection with dose-response curves.

CUMULATIVE HAZARD, DISTRIBUTION FUNCTIONS, AND DOSE-RESPONSE CURVES

If $F_T(t,D)$ or, for short, F is the c.d.f. for the life or in our context synonymously the tumor appearance time (latent period) after exposure then

$\bar{F} = 1 - F$ = survival function
$f = dF/dt = F'$ = probability density function
$h = f/\bar{F}$ = hazard function or age (time) specific tumor rate
$H = \int_0^t dt'h(t')$ = cumulative hazard

The arguments t,D have been suppressed for brevity. One easily sees that

$$F = 1 - \exp(-H) \tag{1}$$

Intuitively $h(t,D)\Delta t$ can be interpreted as the conditional probability that an individual will die from a tumor in the interval $(t,t+\Delta t)$ having survived to age t. For this reason h is sometimes also called the force of mortality. Every c.d.f. of tumor appearance times has a corresponding force of mortality h(t). If $F = 1-\exp(-\lambda t)$, the force of mortality is λ = constant. This is a unique characterization of the exponential c.d.f. These probabilistic concepts and the different relationships among them are well known and very simple. The difficult aspect is the estimation of F, $\bar{F}$, or H from observed quantities. To understand why this is so, consider the following situation: Groups of N_i individuals are exposed to doses D_i $(i = 0,1,2\ldots; D_0 = 0)$. The exposure is instantaneous and takes place at t = 0. As t increases, we observe for each dose group D_i a sequence of N_i indicators like

0,0,1,0,1,1,0,1,1,0,1,1,1,0,0,0,1,0,0

A "1" means in our context that the individual died from cancer possibly induced by radiation. A "0" means that the individual died from another cause, was lost from the study, or is still alive. Each 0 or 1 has associated with it the time when the event occurred. All the times where a 0 occurred can be considered as observations of a random time variable L. "L" stands for loss to indicate that the 0 — events represent lost information for the estimation of $F_T(t)$. The problem of estimating F if losses (0 — events) occur was solved by Kaplan and Meier (2) under the assumption that T and L are statistically independent. An estimator for H is also available in the literature (3,4).

A dose-response curve at a fixed time t_0 for instantaneously exposed individuals (e.g., atomic bomb survivors) can be defined as a

plot of $F_T(t_o,D_i)$ versus D_i on linear graph paper (5). This replaces the plotting of the ratio, number of tumors, n_i/number of individuals exposed, N_i. The magnitude of n_i/N_i is clearly influenced by the number of 0 — events (e.g., accidents) and does not measure solely the effect of the radiation exposure. The same dose D_i can therefore produce different effects — measured by n_i/N_i — depending on how many other deaths occur. Since $F_T(t_o,D_i)$ differs from this ratio, the shape of $F_T(t_o,D_i)$ versus D_i will differ in most cases from n_i/N_i versus D_i [see (5) for more details]. Most of the time the ratio n_i/N_i is used to measure the effect of D_i in exposed animals. In human epidemiological studies the fraction, n_i/total number of person-years, is used instead, and sometimes the difference $\Delta\lambda$ or ratio ρ of these fractions for an exposed and an unexposed group are plotted versus D_i [see e.g. (6)]. The fraction $\alpha = \Delta\lambda/D_i$ is the absolute risk, and the ratio ρ is the relative risk (6). Both are incorrect measures of the radiation risk because of hidden and unwarranted assumptions. To see the mistake, consider Equation 1. For $H \ll 1$, one obtains

$$F \approx H \tag{1'}$$

For the exponential distribution — cf. the earlier remark — $H = \lambda t$, where λ, the force of mortality, is constant. If F is exponential, λ can be estimated by n_i/total number of person-years (7), but the assumption of exponentiality (i.e., λ = constant) is incorrect for both the unexposed and the exposed group for time periods longer than two to three years [see e.g. (8)]. For spontaneous and radiation-induced cancer $h = h(t)$. This fact can be demonstrated by graphical techniques-hazard plotting (3,4) and total time on test (TTT) plots (9). If Equation 1' holds and the force of mortality is time dependent, then only $\Delta H = H(t_o,D_i) - H(t_o,D_o)$ and $H(t_o,D_i)/H(t_o,D_o)$ are meaningful measures of the radiation effect at t_o. The net radiation risk $R_N(t_o,D_i)$ at time t_o due to an instantaneous dose D_i should be defined as the difference of the cumulative incidences:

$$R_N(t_o,D_i) = F_T(t_o,D_i) - F_T(t_o,D_o) \tag{2}$$

This definition assumes that cancer is the only risk acting. If Equation 1' holds, $h_T(t,D_i) = \lambda_i$ = constant and $h_T(t,D_o) = \lambda_o$ = constant, then Equation 2 becomes

$$\begin{aligned} R_N(t_o,D_i) &= H_T(t_o,D_i) - H_T(t_o,D_o) \\ &= (\lambda_i-\lambda_o)t_o = \Delta\lambda t_o \end{aligned} \tag{3}$$

If in addition $\Delta\lambda = \alpha D_i$, then

$$R_N(t_o,D_i) = \alpha D_i t_o \tag{4}$$

The derivation of Equation 4 shows that the absolute risk concept and linear dose-response curves based on this concept are valid only under very special circumstances.

For continuous radiation exposure, the dose $D(t)$ is an increasing function of time. If $F_T(t)$ has been estimated, $G_D(d)$, the c.d.f. for

the random variable D, can be obtained by a simple transformation (10). For $D = \delta T$ — the simplest case with constant dose rate δ — one obtains $G_D(d) = \Pr\{\text{tumor for } D \leq d\} = F_T(d/\delta)$. If $F_T \approx H_T$, then G_D increases like H_T. A plot of G_D versus d is a possible definition of the dose-response curve for continuous irradiation. The shape of G_D depends on the behavior of $H_T(t)$ and will be different for increasing (decreasing) hazard rate $h_T(t)$. A serious problem with this definition stems from the fact that one does not know up to what time after first exposure dose effectively induces cancer. The dose delivered after this time is "wasted." A deeper understanding of radiation carcinogenesis is necessary to solve the "wasted" radiation problem.

The general relationship $\bar{F} = \exp(-H)$ can be used to calculate the potential crude radiation risk from D_1 at t_0 to a so far unexposed population, if ΔH has been estimated from data on an exposed population. The survival function $S(t)$ for the actual life of the population can be found in a life table. The crude radiation risk can then be defined as

$$R_C(t_0, D_1) = S(t_0)\ [1-\exp(-\Delta H)] \qquad (5)$$

The crude radiation risk in Equation 5 is the additional risk from radiation exposure in the presence of all other competing risks.

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A SYSTEMATIC APPROACH TO PERSONNEL NEUTRON MONITORING

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Introduction

A good personnel dosimetry program requires an integrated approach to personnel dose assessment. No single measurement technique or information source can be relied upon solely to provide accurate dose measurement. This is particularly true for personnel neutron monitoring, because the problem of accurate measurement is so difficult and the personnel dosimeters currently available have severe limitations, preventing a wide range of applications for any one dosimeter type[1]. Neutron monitoring requires detection and measurement of neutron doses at 1/10th the level for the accompanying gamma rays. The range of neutron energies generally spans at least nine decades (thermal to 10 MeV), and, in some accelerator facilities at least another decade of energy may be involved. The usual dosimetric problem of angular dependence and body orientation effects add to the difficulty of proper dosimeter interpretation.

It is clear that the information provided by the response of the dosimeter must be used in conjunction with other sources of information to provide the most accurate interpretation of the neutron environment. Other information sources that may provide information necessary for accurate dosimeter interpretation include gamma dosimeter response, instrumental measurements of the gamma and neutron dose rates in the environment, worker stay times, etc. Most dosimeters have a very poor energy response - that is the neutron response does not adequately mimic the dose equivalent conversion curve across the wide range of neutron energies encountered in personnel monitoring. Therefore, a piece of information that is important for accurate neutron dosimetry is the spectral quality of the worker's environment. This information not only improves the accuracy of the dosimeter interpretation, through more accurate assessment of the calibration data, but also serves as a basis for acceptance or rejection of new dosimeters based on their ability to measure the important portion of the dose equivalent spectrum through a radiation facility.

Current Neutron Dosimeters

Before discussing monitoring techniques that can be used in support of a dosimetry program, it is important to review the characteristics of currently available dosimeters. We will place emphasis on the detecting element rather than the system as a whole, because the detector characteristics are the primary limitation of the dosimetry system.

Photographic neutron detectors - NTA film - have been used for operational dosimetry longer than any other dosimeter[2,3]. Briefly, the neutron interacts with a proton in the emulsion of a

small piece of film, causing the proton to move some distance through that emulsion. When the film is developed, the track of the proton is revealed as a thin trail of silver grains in the film. Dosimetry is done by optical measurement of the number of tracks per unit area using a high magnification optical microscope. In practical use, NTA film has a threshold at about 0.5 to 1 MeV, which is equivalent to a 3 or 4 grain track in the developed emulsion. NTA film is also capable of detecting low energy neutrons from the (n,p) reaction with nitrogen in the emulsion. However, practical experience indicates a poor sensitivity for such low energy neutrons. The energy response of NTA film compared with the ICRP dose equivalent conversion curve for neutrons is shown in Fig. 1a. One of the most serious criticisms of NTA film has been its rapid fading property[4]. Although some investigators have had success by packaging the film in hermetically sealed wrapping[5], fading, sensitivity to low energy gammas that fog the film, and the tedious counting involved, are all negative characteristics of NTA.

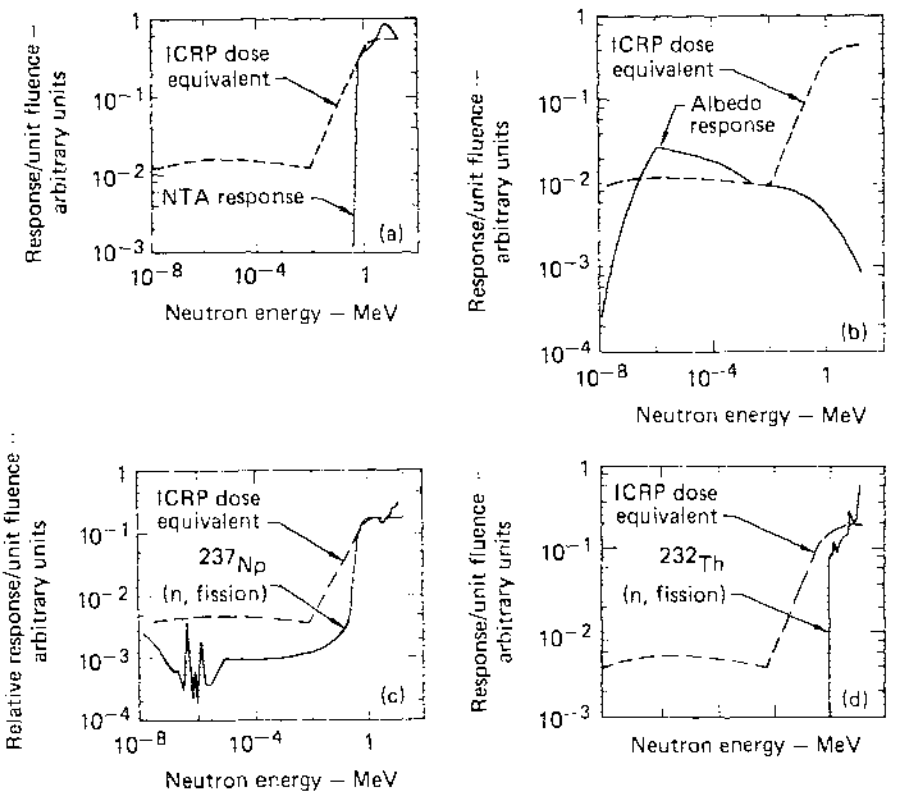

Figure 1. Comparison of the Relative energy responses of personnel neutron dosimeters with the ICRP Dose Equivalent Conversion.

Albedo detectors, which depend on interaction with the human body to thermalize fast neutrons, have had the most rapid increase in use of any of the dosimetry systems used in recent years[6,7,8]. The detecting element is a TLD crystal, having normal or enriched levels of neutron sensitive ^{6}Li. The two distinct advantages of albedo dosimetry are high sensitivity (particularly for low energy neutrons) and the wide availability of automated TLD readout systems. By far, the most significant disadvantage of albedo detectors is a very poor dose equivalent energy response simulation (Fig. 1b). Of all of the dosimeters in use, the albedo system is the most sensitive to spectral variations in the working environment and requires the most supplemental monitoring information. However, for highly moderated neutron environments such as those in nuclear power reactors, albedo dosimeters may be the only real choice.

Two additional dosimeters, which rely on dielectric track etch techniques have been adopted on a much more limited scale than either the NTA or albedo systems. Fission track detectors require the combination of a fissionable radiator and a track registration material. The radiator material (^{237}Np, ^{232}Th, ^{235}U or ^{238}U) is chosen because these nuclides have n,fission cross sections that duplicate all or part of the dose equivalent conversion curve. Fission foil - track etch systems have been used at various laboratories in the United Kingdom, Switzerland, United States, and other countries for about ten years [9,10,11,12]. Of these systems, ^{237}Np dosimeters most faithfully reproduce the dose equivalent conversion curve (Fig. 1c). The biggest drawback of fission foil dosimeters is that the wearer must carry small but significant amounts of radioactive material in the dosimeter. Therefore, they are often issued to personnel only on a limited and controlled basis. Of the nuclides used for this purpose, neptunium has the highest external gamma dose rate from a dosimeter having enough radiator mass to provide sufficient neutron sensitivity. In addition to the disadvantage of having to use radioactive material, the sensitivity of many fission track systems is marginal for routine personnel dosimetry and would probably become unacceptable if higher, more stringent neutron quality factors are adopted. Unlike the NTA type of track detector, the fission track detectors suffer little from the problem of fading[3].

Some laboratories have adopted track detector systems which do not require fissionable irradiators[13,14,15,16]. These detectors rely on direct interaction of the neutrons with light nuclei in the plastic (C, N and O). The charged nuclei recoil, leaving damage tracks that can be revealed by various etching methods. The sensitivity of these systems for fast neutron spectra such as that from a ^{252}Cf fission source is of the order of 50 to 500 mrem, depending on the etching techniques, plastic and the definition of sensitivity used. One of the major problems with direct recoil plastics for routine dosimetry is the relatively high energy thresholds associated with the reactions. It is, however, possible to enhance the low energy response of these detectors using the n,alpha reactions from non-fissionable ^{6}Li or ^{10}B radiators. Unfortunately, little experience, save that of CERN[16], has been obtained with this technique. The manual counting required for

dosimeter evaluation is also a limitation. Certainly automated optical systems could be used for this purpose, but they are generally too expensive for small-scale dosimetry programs.

We should point out that the discussion to this point has considered a dosimetry system based on only one detector or one detecting element. In fact, however, it may be necessary, particularly in facilities that have a wide range of neutron spectra, to use a multi-element system. Such systems have been used[16,17,18] and generally involve the combination of an albedo detector with a threshold detector. The responses of the detectors can be combined to synthesize a better simulation of the dose equivalent conversion curve. Moreover, combination systems are, in effect, simple spectrometers. They add to the complexity of the dosimetry, but the improved accuracy and information available may well justify the added effort.

Dosimetry Developments

Perhaps the most promising new detector now being widely investigated is CR-39 plastic. CR-39[19,20,21] is a carbonate, and has physical properties similar to that of glass. The processing of CR-39 for electrochemically etching requires 7 to 10 hours of a combined pre-etch and electrochemical etch. Unlike polycarbonate, the threshold for neutron detection with CR-39 is about 100 keV (Fig. 2), and is capable of detecting less than 20 mrem of fast neutrons. High sensitivity can also be obtained by conventional etching only, however the optical counting of the much smaller tracks is more tedious than when electrochemically etching is used. Although it does not provide as good a replicate of the dose equivalent curve as one would like, it represents a significant potential improvement over other track etch base systems. A personnel dosimetry service with CR-39 is now commercially available.

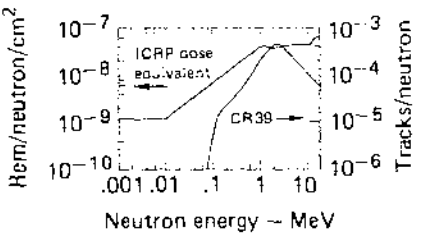

Figure 2. Comparison of the energy response of electrochemically etched CR-39 (after pre-etching (22)) with the ICRP dose equivalent conversion.

A number of other neutron detectors have been and are being investigated for potential personnel dosimetry applications, but each appears to have one or more deficiencies that, for the present, prevent use in operational dosimetry programs. Such detection methods include a super heated drop detector investigated at Yale University[23], lyo-luminescence dosimetry that has been investigated by a number of laboratories[24,25], the development of TLD's having hydrogenous material built into their crystals[26], TSEE

(now pursued with much less interest than at one time), and others. One of the more promising TLD based techniques makes use of LET dependent differential glow-peak formation[27]. This system continues to be investigated and may be commercially available in some form in the near future.

Neutron Surveys

Neutron dosimeters are interpreted on the basis of calibrations that normalize detector response to dose equivalent. Traditionally, we use unmoderated fast neutron sources - ^{252}Cf, $^{241}AmBe$, $^{238}PuBe$, etc. - with known neutron emission rates to provide calibrations. We now realize there are usually large differences between the calibration and exposure spectra, making those calibrations inaccurate and uncertain. In recent years, the need for in-field measurements to provide a correction factor to these calibrations has become evident.

The concept of in-field calibrations is that measurements are made with instrumentation having an energy response like the personnel dosimeters. These measurements are then normalized to dose equivalent measurements made at the same locations in the working environment. The normalizations provide the basis for correcting neutron source calibrations. In-field calibrations are necessary only because of spectral differences between the calibration source and working environment. As long as the working area spectra do not change, the measurements need be done only once.

The most traditional and straight forward approach is the survey of a facility with a remmeter (Anderson Braun, spherical moderator, etc.), followed by long-term (hours to days) exposures of personnel dosimeters placed on phantoms at the same locations. This technique is technically sound, but takes a long time and may be severely limited if the dose rates are low. Moreover, the necessary information is not available until the dosimeters are processed, often resulting in the need for resurveying.

A more rapid technique has been developed for facilities using albedo dosimeters[28]. A BF_3 detector placed in a 3-inch diameter polyethylene sphere has been shown to have an energy response very similar in shape to that of the albedo dosimeter. The use of phantoms in an initial survey is replaced by a one- or two-minute measurement with the 3-inch moderator. If we use a 9-inch remmeter, a single location can be surveyed in less than 5 minutes, even at millirem dose levels. Additional information on the thermal neutron contribution can be obtained by taking counts with a bare BF_3 probe.

We use the sphere response ratio to identify locations that represent the range of spectral variation in any facility. The survey can then be completed by exposure of a few phantom mounted dosimeters, with the confidence that the proper locations have been selected. We can also sample many more positions in a short time, with a minimum of phantom deployment.

Neutron Spectrometry

The measurement of neutron spectra has, for the most part, been regarded as a laboratory concept, useful in high energy physics, but without practical application in health physics. However, moderated multisphere detector measurements with computer unfolding can now be used to augment conventional survey techniques. The instrumentation required is much less sophisticated than the detectors and electronics used for experimental physics applications, and well within the capability and budget of a health physics program. As an extension of simpler survey methods, multisphere measurements yield spectral and dose equivalent information over the full range of occupational neutron energies. This information can then be used to estimate the error in response of portable remmeters, as well as allowing the health physicist to predict the spectrum weighted response of any personnel monitor in use or considered for use at the facility. The spectra give the health physicist the most clear view of the facility neutron environment.

Health physicists have used Bonner spheres for 20 years[29], but recent computer unfolding codes[30,31] and response calculations[32] contribute to making simple, accurate spectral measurements. New generations of commercially available portable pulse height analyzers make in-field use of multisphere much more mobile. At LLL, we use a portable analyzer, a ^{6}LiF scintillation detector and 3, 5, 8, 10 and 12 inch spheres of polyethylene to make the necessary measurements. The detector is used, in turn, in each of the spheres together with bare and cadmium covered measurements. The detector pulse height responses from the $^{6}Li(n,alpha)$ reaction in each of the seven detector-moderator configurations are used as input for the unfolding code. The spectra determined have poor resolution qualities, but high resolution is not essential for neutron health protection.

In practice, we make spectrum measurements at key facility locations identified during the detector-remmeter survey described above. We choose these locations to represent the range of spectral variation implied from the range of 3/9 inch ratios. Usually only about three or four locations are measured with this system. Although we use a computer to fully unfold the spectra, the responses of the 3, 8 and 12 inch detector measurements can be used as input for simple matrix inversion programs available on programmable hand calculators, to obtain dose equivalent estimates that are within 15% of the fully unfolded calculations. This gives us an immediate comparison with the remmeter results before leaving the area. Obvious errors in the data can be detected, allowing remeasurement without having to return days later.

We have used the 3/9 inch sphere ratio technique, with multisphere spectrometry to survey our own facilities (a high flux 14 MeV neutron generator, transuranic isotope storage vaults and glove box facilities, a 3 MW pool type reactor and our own neutron calibration facility), as well as a number of power reactors through the United States. The survey technique has significantly improved dosimeter calibrations. The spectral information has been used (even at dose equivalent levels as low as 0.1 mrem per hour) to

predict the poor performance of certain threshold detectors in heavily moderated environments, as well as determining the effective over-response of moderated remmeters used in the facilities.

Summary

NTA film and albedo detectors represent the major portion of personnel dosimeters now used for occupational neutron monitoring. However, recent attention to the spectral response of these systems has demonstrated the need for detectors that have a better match to the fields being monitored. Recent developments in direct recoil track etch dosimeters present some intriguing alternatives, and careful use of ^{237}Np fission fragment detectors offers the advantage of a good dose equivalent spectral match. Work continues on a number of other new detector mechanisms, but problems with sensitivity, energy response, gamma interference, etc. continue to prevent development of most mechanisms into viable personnel dosimeters. Current dosimeter limitations make a systematic approach to personnel neutron monitoring particularly important. Techniques have been developed and tested, using available portable survey instruments, that significantly improve the quality of dosimeter interpretation. Even simple spectrometry can be done with modest effort, significantly improving the health physicists ability to provide accurate neutron monitoring.

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THE PRESENT STATE OF NUCLEAR ACCIDENT DOSIMETRY

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1. INTRODUCTION

During the last two decades various types of nuclear accident dosemeters have been developed and tested. Important contributions to this development have been made through a series of intercomparisons organized by the Oak Ridge National Laboratory in the USA (1), and a co-ordinated research programme, including four international intercomparisons, established by the IAEA (2). The IAEA has issued a compendium of neutron spectra in criticality accident dosimetry (3) and is now preparing a comprehensive technical manual on nuclear accident dosimetry (2).

By "nuclear accident dosimetry" is usually meant criticality accident dosimetry, since it is for this purpose that special accident dosemeters have been developed. For other types of radiation accidents, involving, for example, contamination or sealed radioactive sources, the dosimetry methods used routinely for low-level personnel monitoring are usually adequate, provided that dosemeters are worn.

The primary purpose of nuclear accident dosimetry is to provide, in case of an accident, a rapid assessment of radiation doses for the guidance of the medical services in the appropriate treatment of more heavily exposed persons and for reassuring personnel who have been only lightly exposed. A rapid dose assessment is also desirable for public relations reasons. Furthermore, reliable estimates of dose are important for record purposes and of scientific value in studies of the effects of acute radiation exposure in man.

In this paper the present state of nuclear accident dosimetry is discussed on the basis of a critical review of nuclear accident dosemeters prepared by the author (4).

2. NUCLEAR ACCIDENT DOSIMETRY METHODS

Nuclear accident dosimetry systems are generally based on a combination of personnel dosemeters and area dosemeters and, in addition, the induced activity in the body of the exposed person will usually be measured after an accident involving a neutron exposure. Installed neutron dosemeters can contain more components than it is practicable to use in a personnel dosemeter, so area dosemeters can provide more detailed information on the neutron spectrum and consequently a more accurate neutron dose estimation. However, area dosemeters may be subject to recovery difficulties following an accident, and information from personnel dosemeters is generally needed for the transfer of the results obtained with

This paper has been prepared in connection with a study sponsored by the Commission of the European Communities (4).

area dosemeters to the positions of the exposed persons. Simple personnel dosemeters can provide a fairly rapid dose assessment after an accident, but the use of a simple dosemeter means that greater emphasis is to be placed on the methods of interpretation.

As in low-level personnel dosimetry, a basic problem in criticality accident dosimetry is that generally no single neutron detector is available which adequately covers the whole energy range of interest. Therefore, the determination of the neutron dose received by a person involved in a criticality accident generally requires a knowledge of the neutron spectrum incident on the body. Several nuclear accident dosimetry systems have been designed to provide some spectral information by including at least two neutron detectors with different energy responses.

2.1. Neutron dosemeters

In existing nuclear accident dosimetry systems the measurement of neutron doses is usually based on activation detectors or to a smaller extent on fission fragment track detectors. The main advantages of activation detectors are that they are normally insensitive to beta- and gamma-radiation, and most activation detector foils are inexpensive. Generally, a combination of fast-neutron (threshold), intermediate-energy neutron (resonance) and thermal-neutron detectors is used, and the neutron dose is derived from the induced activities using assumptions on the neutron spectrum.

Like activation detectors, fission fragment track detectors are normally insensitive to beta- and gamma-radiation. A further advantage is that their information content following an accident is not subject to radioactive decay. On the other hand, fission fragment track detectors are more expensive, and the radiotoxicity of the fissile material is a disadvantage which in some countries excludes their use in personnel dosemeters.

Silicon diodes may be utilized for fast-neutron detection in the energy range from about 0.2 to 15 MeV. Their main advantages are a relatively flat energy response, easy and fast evaluation and negligible response to other types of radiation. Their main disadvantage is a limited overall accuracy.

Albedo dosemeters are being increasingly used for routine personnel monitoring, but so far their use in nuclear accident dosimetry systems have been very limited owing to their wide variation in response as a function of neutron energy.

2.2. Gamma dosemeters

The measurement of doses from gamma-rays following an accident will generally be made with the same types of dosemeters as are used for routine personnel monitoring, i.e. film, thermoluminescence and radiophotoluminescence dosemeters. In some systems gamma-ray detectors are incorporated into the nuclear accident dosemeter badge, whereas in others the gamma dose is determined using a routine personnel dosemeter.

2.3. Special nuclear accident dosimetry methods

A chemical nuclear accident dosemeter has been developed by Dvornik and co-workers (5). The dosemeter measures the total (neutron + gamma) absorbed dose.

Analysis of chromosome damage produced in human blood lymphocytes following over-exposure is a useful biological dosimetry technique. In a mixed gamma and neutron radiation field, the gamma and neutron dose components can be estimated from the chromosome aberration yields if the ratio of the gamma to neutron doses is known(6).

If a person who is not wearing a dosemeter is accidentally exposed, the dose may be estimated from chromosome aberration analysis, from the induced activity in the body (e.g. Na-24 in blood or P-32 in hair) or from reactions in items carried by the exposed individual. Chromosome-aberration analysis and the measurement of body sodium activity are well-established methods, but in mixed radiation fields additional information is needed in order to derive gamma and neutron doses. More unusual methods (measurement of induced activity in coins etc.) may provide some information, but difficult interpretation problems are to be expected if methods that are not in routine use or have not been designed and tested for nuclear accident dosimetry must be relied on.

2.4. New possibilities

A recent promising development in fast-neutron dosimetry, which may find application in nuclear accident dosimetry, is the registration of fast-neutron-induced recoil-particle tracks in polycarbonate foils or in other plastics using electrochemical etching.

Further developments in albedo neutron dosimetry could result in an increased application of albedo dosemeters in nuclear accident dosimetry.

Self-irradiation of thermoluminescence phosphors containing elements which form radioactive isotopes after neutron capture could also be utilized in nuclear accident dosimetry, and the use of lyoluminescence dosemeters (7) is a further possibility. However, lyoluminescent materials are still too insensitive to be used for routine personnel monitoring.

3. DISCUSSION AND CONCLUSIONS

The experiences gained at national and international intercomparisons (1,2), and from experimental studies (8) show that even simple personnel criticality dosemeters perform satisfactorily in neutron spectra that are not too degraded from an uncollided fission spectrum. For spectra rich in intermediate-energy neutrons, on the other hand, the performance of simple dosemeters is generally not satisfactory when routine methods of interpretation are used.

Installed neutron dosemeters containing several activation and/or track detectors can provide an estimate of the general shape of the neutron spectrum, so usually they can give a reasonable determination of the neutron dose at the position of the dosemeter. However, in well-moderated spectra, the commonly used activation

detectors (In,S,Mg etc.) may be insufficient. The performance may be improved by adding a neptunium track detector to the activation detector set. However, even for this combination the neutron dose may be underestimated by a factor of about 2 in particularly difficult cases (8). Threshold detector systems that include a set of fission-fragment track detectors contained in a B-10 shield can usually give a satisfactory determination of the neutron dose. However, such threshold detector systems are expensive so they are used to only a very limited extent.

Accidental gamma doses can usually be measured with an uncertainty of less than 25%, using the same personnel dosemeters (film, TLD,RPL) as are used in routine personnel monitoring. However, the uncertainty may be increased in a mixed radiation field with a heavily moderated neutron spectrum if appreciable corrections have to be applied to the reading of the gamma dosemeter for the response to thermal neutrons and to the radiations from activated foils in the vicinity of the dosemeter.

The present state of nuclear accident dosimetry is satisfactory in the sense that several nuclear accident dosimetry systems are now available that perform within the criteria proposed by a panel of experts convened by the IAEA in 1969 (9). However, the proper interpretation of the responses of existing nuclear accident dosemeters may require a high degree of expertise, so, in the authors opinion, there is a need for a further development that should aim at simplification. It would be desirable if accidental neutron doses, as is usually the case for the gamma doses, could be measured with the same dosemeters as are used in routine personnel monitoring. Whether or not this simplification will eventually be achieved is dependent on further research and development in personnel neutron dosimetry.

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PHOSPHATE GLASS DOSIMETRY: A POTENTIAL ALTERNATIVE IN PERSONNEL MONITORING

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INTRODUCTION

Radiophotoluminescent phosphate glass was the first solid state dosimeter in personnel monitoring produced in millions of species already in the middle of the fifties. Later on, however, it decreased in importance as compared to TLD due to unbalanced efforts of researchers and manufacturers on these subjects, particulary in USA. This trend may be explained by somehow emotional reasons as well as by the dosimetric imperfections of the early glass generation with respect to background luminescence (predose), non-reproducibility and energy dependence of reading. The new generation of low-Z glass dosimeters that meanwhile has been developed in Japan remained rather disregarded in spite of remarkable improvements of the dosimetric properties such as sensitivity, batch uniformity and precision.

Mainly in Germany this type of glass dosimeter found application in personnel monitoring and, to a limited extend, replaces or backs up the so far official film dosimeter. From review, the predose and the energy-dependent detector material may be considered the main reasons that prevented a major break-through of glass dosimetry. The missing development of appropriate automatic readers is only a consequence of this situation.

Contrary to glass dosimetry there are at least 12 automatic TLD reader versions commercially available. But inspite of promising earlier prognosis also this dosimetry method does not easily penetrate into personnel monitoring since a number of institutions concerned rejected or postponed to replace the film by TLD. To a certain extend, this may be explained by the high financial investment for a system at a state of technology, which is characterized by considerable malfunctioning and not yet elaborated standard of dosimetric reliability.

Under these aspects and on the basis of long-term experiences with both TLD and glass dosimetry we feel encouraged to draw the attention again to glass dosimetry as a so far neglected but potential alternative in personnel monitoring. This holds the more since essential physical and technological advance has recently been achieved.

RECENT TECHNOLOGICAL ADVANCES

For better understanding of the essential advances of glass dosimetry the following properties are compared with LiF TLD-systems. New glass composition: A new glass type FD-7 [1] or DOS8 [2] offers a lower energy dependence with an oversensitivity of a factor 3.6 related to ^{60}Co which hitherto was a factor 7 for the still used FD1

(Toshiba) or DOS2 (Schott) glass. Both glass types show a high batch uniformity of <±0.5% compared to the TLD response which may vary up to ±30% calling for individual calibration. Individual predose values of 20 mR to 250 mR depending on the reader type can be determined reproducibly. The zero dose reading of TLD detectors, on the other hand, may scatter by one order of magnitude due to batch quality and radiation/annealing history of the single detector.

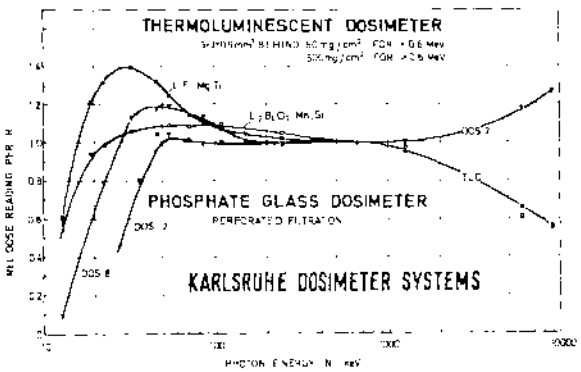
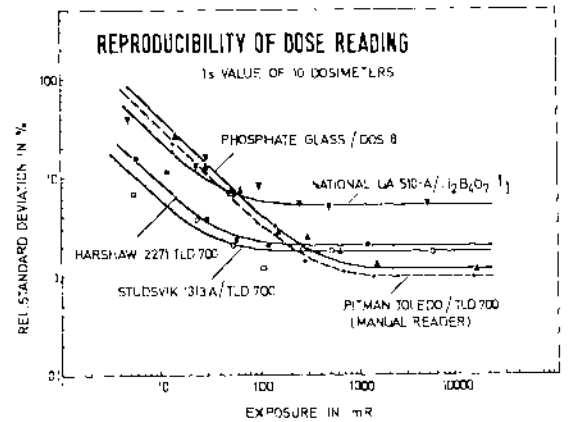


Fig.1 Energy dependence of the automatic DOS 8 [4] and the spherical DOS 2 [3] glass dosimeter system compared to TLD systems

In RPL dosimetry perforated filter are used to improve energy dependence. This is shown in Fig. 1 for the routinely used DOS 2 in the spherical capsule [3,11] and for a new automatic RPL system on DOS 8 basis [4] resulting in energy independence within ±20% above 25 keV. The RPL characteristic is similar to LiF for low energies and differs mainly for high energy photons above some MeV. Sophisticated combinations of glass and filter may be chosen such that the reading is proportional to specific depth doses [5] or fits new quantities such as dose equivalent or dose equivalent index [6].



1) according to prototype reader, 1978

Fig.2 Reproducibility of the automatic glass dosimeter systems [4] and TLD systems of an interlaboratory test [12]

Automatic readout: At Karlsruhe a glass dosimeter system with automatic readout device has been developed which uses a glass plate of 14x15x1.5 mm^3 and a perforated energy compensation filter resulting in an energy independence within ±20% above 25 keV (Fig.1) [4]. In addi-

tion, dose fractions in the energy ranges $E < 30$ keV, 30 keV $< E <$ 100 keV and $E > 100$ keV can be assessed by applying movable screens during readout. The reproducibility of the automatic glass dosimeter system (Fig.2) is characterized by a relative standard deviation of $s = 1\%$ above 1 R and due to predose influences $s = 15\%$ at 40 mR. According to these figures the automatic glass system is comparable to automatic TLD systems [12]. Present readout frequency is around 250 h^{-1} which figure can still be increased.

Multi-directional scanning: An experimental setup for automatic scanning along the three axis of a glass cube of 8x8x8 mm^3 has been realized for a microprocesser assisted estimation of radiation quality and direction of radiation incidence [7]. In addition to dose assessment this technique offers the advantage to identify unidirectional exposures above 1 R and to discriminate between penetrating and nonpenetrating radiation by means of the directly measured depth dose profile in glass.

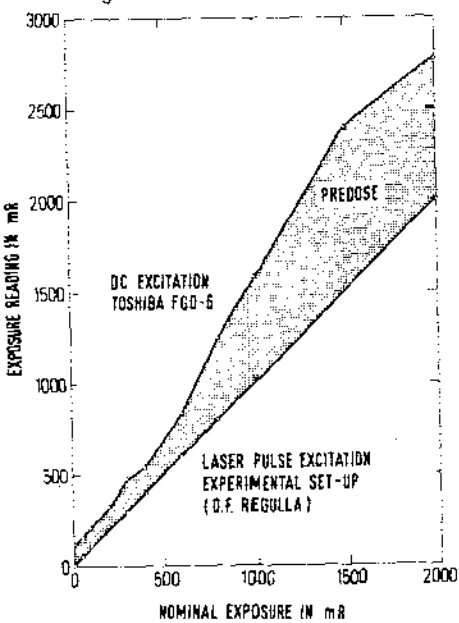


Fig.3 Conventional and laser pulse readout of heavity surface contaminated glass detectors vs. exposure

Laser pulse excitation: The conventional readout technique of glass dosimeters provides a refined cleaning of the glass with detergents before readout which procedure probably does not fit to automation. Recently, the fluorescence phenomenon of predose was found to follow an intrinsic mechanism different from radiation induced fluorescence (RPL) [8] and was correlated to surface contamination. It decays rapidly after UV pulse excitation with time constant of $\tau < 100$ ns, i.e. 10 times faster than RPL. This effect was successfully elaborated for a general predose suppression using laser pulse excitation (FWHV 10 ns). Fig. 3 compares results of conventional DC with laser pulse excitation for a heavily surface contamined glass detector without any cleaning procedure applied. The new readout technique is good for exposure assessment as low as 10 mR or even down to 0.1 mR

when using the time constant τ itself as a measure for dose [9]. Present improvements at Neuherberg concern the application of a compact nitrogen laser, electronic noise suppression and computer assisted signal processing and data interpretation [10].

CONCLUSION

Large-scale application in personnel monitoring apparently by-passed phosphate glass dosimetry in the past. The reason for this evolution seems rather random, retrospectively. Definitly, glass dosimetry combines the capabilities for repeated evaluation, long-term documentation and analysis of radiation field parameters appreciated from film dosimetry, with the advantages of solid state dosimetry, i.e. the accuracy of dose measurement, the long-term stability of the stored RPL signal and the ability to perform intermediate evaluations during long-term exposure. Unlike other solid state dosimetry methods, glass dosimetry lends itself to automation because of the simple and straightforward reader calibration and read-out procedure. Reading time can be a fraction of a second. Based on known fluorescence centers that are not distroyed by readout and an extremely good batch and inter-batch uniformity, detector readout is by far independent of the reader. This enables, with the same dosimeter, repeated or even frequent readout during the monitoring period, e.g. for exit controls in power reactors, and subsequent control reading at a central dosimetry service if necessary.

These capabilities may truely simplify and improve the organization of operational radiation protection. Another aspect is the permanent availability of information on a person's accumulated dose without any further data processing. Last not least, the energy dependence may be adjusted by change of glass composition and filters such that it fits the energy function of the quantity of interest, e.g. exposure or absorbed dose, organ dose or deep dose equivalent index.

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TRACK STRUCTURE CALCULATION OF THE THERMOLUMINESCENT YIELDS OF HEAVY CHARGED PARTICLES*

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Extensive effort is currently being invested in the investigation of the use of LiF thermoluminescent dosimeters (TLD) in exotic radiation fields, such as mixed neutron-gamma and high LET particulate radiation fields. For this reason it is important to investigate the relative TL response, η, to various radiation fields (η is defined as the TL signal/imparted energy by the radiation field in question/TL signal/imparted energy by Co-60 γ rays, both at low absorbed doses of the irradiated mass). The question of the universality of the TL-LET behaviour is especially pertinent to the use of LiF-TLD's in mixed n-γ or charged particle radiation fields. For example, large discrepancies exist among published fast neutron sensitivities and high energy electron sensitivities even in the cases when the problem connected with the determination of the imparted energy are claimed to be solved. Unfortunately, the possibility of non-universality of the TL-LET response has been generally overlooked, perhaps, because it has been proposed (Tanaka et al., (1) and many others) that, indeed, the relative TL-LET response curve is universal, i.e., the stopping power, S, of the directly ionizing radiation is the dominant factor that influences the relative TL response and that there is a unique curve $\eta = \eta(S)$ common to many TLD materials. Our purpose is to show that the situation is far more complicated than the simplistic picture proposed by Tanaka et al., and others.

In the first instance many groups have reported data that illustrate that the η of a particular type of radiation depends on the type of dosimeter (e.g., LiF, $Li_2B_4O_7$, BeO, CaF_2, quartz, etc...). Furthermore, we have carried out extensive studies (2,3) irradiating various batches of LiF and $Li_2B_4O_7$ (Harshaw TLD bulb dosimeters) with 13.54 meV neutrons and 3.8 MeV alphas, which showed that purchase of a particular type of TLD even from the same supplier and with the same nominal dopant concentrations (Mg, Ti in LiF, Mn in $Li_2B_4O_7$) does not guarantee identical or even similar TL-LET response characteristics (Table 1).

Another question in the application of the dependence of η on LET of a particular dosimeter is whether the LET of the ionizing radiation is the only parameter of the radiation field that influences η, or whether other parameters of the radiation field (e.g., the velocity of the directly ionizing particles) have to be taken into account. We have therefore studied the TL induced in LiF-TLD's by fission fragments using a 5 μm mylar degraded flux from a Cf-252

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source in vacuum (4). Approximately 1/3 of the imparted energy arises from fission fragments with average specific $\overline{LET}$ in LiF-TLD of approximately $3 \cdot 10^4$ MeV $g^{-1}cm^2$. Most of the remainder of the imparted energy arises from α particles emitted from Cf-252.

Table 1. Relative Response of LiF and $Li_2B_4O_7$ to neutrons and alphas

13.54 meV Neutrons			3.8 MeV α particles		
Type	Batch	$\bar{\eta}$	Type	Batch	$\bar{\eta}$
LiF	1-TLD-100	0.340±0.007	LiF	1-TLD-100	0,170±0.024
LiF	2-TLD-600	0.420±0.007	LiF	2-TLD-600	0.210±0.007
$Li_2B_4O_7$	1-TLD-800	0.775±0.007	LiF	3-TLD-700	0.290±0.013
$Li_2B_4O_7$	2-TLD-800	1.105±0.017			

All the models predicting the dependence of η on $\overline{LET}$ and the experimental data on η versus LET indicate, to the best of our knowledge, a decrease of η with increase of LET in the region of high $\overline{LET}$. For example, the one and two trap model (5) used by Jähnert predicts relative TL efficiencies (fission fragments to 5.4 MeV alpha particles) in LiF-TLD's of 0.05 and 0.23 respectively. We have measured a value of $\eta_{ff}/\eta_{\alpha} = 1.26 \pm 0.28$ in serious contradiction to these predictions (the statistical error comprises only a small part of the indicated total error of η_{ff}/η_{α}.). This result together with other published experimental data hint at the multiplicity of the TL-LET function, i.e., the type of the directly ionizing particle must be taken into account in the study of η versus LET and that at least in the high $\overline{LET}$ region η increases with particle mass for a particular $\overline{LET}$.

THEORY

The reduced relative response of LiF-TLD to high LET radiations is widely explained to arise from the recombination of the liberated charge carriers by the primary heavy charged particle, or from saturation of the activator sites (which depends on the concentration of locally available activators) or from both effects. The influence of these effects on η must be treated in a three dimensional model which takes into account the delta rays, which escape from the heavily irradiated column around the path of the primary particle and produce TL signal with high efficiency. The radial distribution of absorbed dose imparted by the ejected electrons around the path of the heavy charged particle determines the track structure and is believed to be a relevant parameter to describe the radiation end effect. In the theory of track structure (6) the dose-response function of a system to gamma rays is coupled with the spatial distribution of dose from secondary electrons to yield the response of the detector to the heavy ion. Information about the ability to produce TL signal of the heavily irradiated volumes around the track can be arrived at using electron dose-TL response.

Such calculations require the knowledge of the dose-TL response dependence of the particular dosimeter under study. We define a dose response function, f(D), to be the ratio of the TL response (TL signal/imparted energy) at a particular absorbed dose from a radiation

field to the TL response at low dose from the same radiation field. Since f(D) is quality dependent and the mean energy expended per ion pair formed is energy dependent for low energy electrons, the dose-TL response must be generated by electrons of initial spectrum (as far as this is possible) similar to that of the initial electrons ejected by the heavy charged particles, HCP, and not by γ rays as is usually applied in the theory of track structure (6,7). Since the maximum energy of ejected free electrons by a 4 MeV α particle is about 2 keV, it is therefore improper to generate and use a f(D) function using electrons with initial energies two orders of magnitude greater.

If the heavy irradiation in the wake of the densely ionizing HCP is the only effect that governs the dependence of η on the $\overline{LET}$ of the radiation and there exists cylindrical symmetry on the radial distribution of the absorbed dose, $D(r,\ell,W)$, imparted by the ejected particles around the track and along the path of the HCP (with average path length R in the TLD) the η of the radiation will be given by

$$\frac{\eta_\alpha}{\eta_e} = \frac{\int_0^R \int_0^\infty f(D)D(r,\ell,W)2\pi r dr d\ell}{\int_0^R \int_0^\infty D(r,\ell,W)2\pi r dr d\ell} \qquad (1)$$

The radial distribution of absorbed dose in nanometric scale for various HCP (protons, α particles, I-127 ions etc...) $D(r,W)$ have been calculated by W. Baum and collaborators (8) based on their measurements of the ionization current produced by a monoenergetic beam of particles within a small movable ionization chamber of transparent mesh located in a large cylinder filled with tissue equivalent gas at variable low pressure.

The radial distance in LiF-TLD from the data in tissue can be simulated using the relation

$$r_{LiF} = r_t\left[\frac{dE}{dx}^{e,t} \Big/ \frac{dE}{dx}^{e,LiF}\right] \qquad (2)$$

where $(dE/dx)^{e,t}/(dE/dx)^{e,LiF}$ is the average ratio of the stopping power of the ejected electrons in the two media, and the radial distribution of absorbed energy in LiF, $D_{LiF}(r_{LiF},W)$ by

$$D_{LiF}(r_{LiF},W) = \left[\frac{dE}{d\xi}^{HCP,LiF} \Big/ \frac{dE}{d\xi}^{HCP,t}\right] \cdot \left[\frac{\rho_{LiF}}{\rho_{tis}}^2 \cdot \frac{dE}{d\xi}^{e,LiF} \Big/ \frac{dE}{d\xi}^{e,t}\right] \cdot D_t(r_t,W)$$

where $(dE/d\xi)^{HCP,LiF}/(dE/d\xi)^{HCP,t}$ is the ratio of the mass stopping power of the heavy charged particle at energy W in LiF and tissue and ρ_{LiF}, ρ_t are the densities of the two materials.

RESULTS AND DISCUSSION

Preliminary results were obtained via H-3 irradiation of LiF (TLD-100). The mean energy of the emitted beta particles after appropriate backscattering corrections was calculated to be 6 keV. Beta particles were preferred over ultra-soft X-rays and lower energy

electrons in order to assure that similar volumes of the TLD's were irradiated by the electrons and the HCP employed.

The calculations for 3.8 MeV alpha particles stopping in the TLD lead to values of η about two to three times greater than those we measured with the same TLD's (even allowing for a factor of two error in the dose determination, η changes by only 20%). In addition we have calculated the ratio of the TL response of 33 MeV I-127 ions to the TL response of 3 MeV α particles to be approximately 0.25, while we have found experimentally that the ratio of the TL response of degraded Cf-252 fission fragments (with mean energy of the light and heavy fragments 49 and 31 MeV respectively (4)) to the TL response of 5.4 MeV α particles (both stopping in the TLD) is five times greater. Unfortunately radial dose distribution data for I-127 ions at lower energies are not available, so that the calculation could not be carried out over the entire range of the fragments. These results were obtained with dosimeters annealed in air and there exists the possibility of surface contamination altering the TL response to HCP's and to beta particles. We are therefore repeating these experiments using more sophisticated techniques to eliminate such a possibility (10).

Tochilin et al., (9) measured the dose response function of BeO to X-rays with effective energy = 9 keV and found for 10.4 MeV/amu C-12 and Ne-20 ions $\eta = 2.06 \pm 0.14$ and 2.25 ± 0.38 respectively. Application of our method using data for 9 MeV/amu O-16 ions leads to $\eta = 1.5$.

These results indicate that the main (if not the only) effect that governs the values of η is the local concentration of the imparted energy around the track of the HCP and point to the possibility of calculating the TL response of any dosimeter to any ionizing radiation field (e.g., epithermal and fast neutrons) by measuring only the dose response function to electrons. Further investigations we are carrying out in our laboratory will explore the extent and accuracy of the applicability of Track Structure Theory to thermoluminescence.

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EVALUATION DE LA DOSE COLLECTIVE A L'ECHELLE EUROPEENNE DUE A DES REJETS ATMOSPHERIQUES

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Le programme de calcul DOSCOL a pour objet la détermination de la dose collective délivrée à l'ensemble de la population d'Europe de l'Ouest, par irradiation externe et par inhalation, due à un rejet continu et à débit constant d'un gaz ou d'un aérosol radioactif, en un point quelconque du territoire européen.

Ce calcul nécessite la mise en oeuvre du programme TALD (1), (2), qui calcule les concentrations atmosphériques intégrées annuelles, et les données démographiques relatives à l'ensemble de l'Europe de l'Ouest, disponibles sur une bande magnétique élaborée antérieurement (3). Sur cette bande, l'Europe de l'Ouest est divisée en carrés d'environ 10 km de côté. Pour l'application au code DOSCOL, et afin de ne pas multiplier les calculs, il a été décidé, dans un premier temps, de regrouper ces populations sur une grille dont la maille est d'environ 10^4 km^2. Cette population est supposée concentrée au centre des mailles.

Les équivalents de dose effectifs collectifs, pour une concentration unitaire, pour l'inhalation et l'irradiation externe, sont tirés de (3). Ils sont rappelés dans le tableau 1.

1. UTILISATION DU PROGRAMME TALD

1.1. - Présentation

Le calcul comporte deux étapes successives :

- *l'établissement des trajectoires* suivies par le polluant, à partir de données météorologiques réelles ;

- *le calcul des concentrations* dans le nuage associé à ces trajectoires. Le modèle de diffusion adopté est un modèle "panache".

Le rejet est décomposé en rejets élémentaires bi-quotidiens. Une trajectoire est tracée pour chacun de ceux-ci; elle est supposée représentative de l'ensemble de celles qui pourraient être tracées durant les douze heures de rejet.

Le tracé des trajectoires est effectué à partir des données météorologiques. Pour les années 1975 et 1976, la vitesse et la direction du vent moyen entre les surfaces isobares 850 mb et 1000 mb, à 0 h et à 12 h chaque jour, en 400 points de la grille de prévision météorologique ont été

fournies par la Météorologie Nationale. Cette grille est approximativement limitée par les méridiens 20° W et 20° E, et par les parallèles 30° N et 65° N.

Il faut connaître d'autre part la direction et la vitesse du vent au point d'émission durant les années 1975 et 1976. Ces vents sont répartis en huit classes en fonction de leur direction.

Les trajectoires sont tracées par segment, en supposant les conditions météorologiques constantes durant 6 h. Durant les 3 premières heures, on utilise le vecteur vitesse du vent au point d'émission, et ensuite le vecteur vitesse moyen entre 850 et 1000 mb fourni par la Météorologie Nationale.

Le calcul des concentrations repose sur un certain nombre d'hypothèses dont les plus importantes sont celles qui permettent l'emploi d'un modèle de panache gaussien (vent constant en vitesse et direction; diffusion longitudinale négligeable devant le transport par le vent).

On suppose que l'épaisseur de la couche de mélange est de 1000 m sur l'Europe Occidentale et que le rejet est effectué au sol.

Les écarts-types σ_y et σ_z de la distribution de la concentration dans les directions perpendiculaires à l'axe panache sont calculés à l'aide de relations de la forme :

$$\sigma = (At)^k$$

où t est le temps de transfert, A et k sont des paramètres dépendants du temps de transfert.

L'appauvrissement du panache dû au dépôt sec et à la décroissance radioactive est pris en compte. Par contre le dépôt par temps de pluie ne l'est pas.

Le programme DOSCOL a été appliqué à des rejets hypothétiques de 1 Ci par an de ^{85}Kr, de ^{131}I et de ^{239}Pu, à partir du Centre d'Etudes Nucléaires de Saclay, pour les années 1975 et 1976, années pour lesquelles nous possédons les données météorologiques nécessaires à l'exploitation du programme TALD.

1.2. - Résultats

1.2.1. - Influence du dépôt -

Alors que le rapport des facteurs de dose irradiation externe du $^{85}Kr/^{239}Pu$ est de 28,1, le rapport des doses collectives est de l'ordre de 100. Il s'ensuit que la prise en compte du dépôt sec ($v_d = 5.10^{-3}$ m.s^{-1}) diminue, *pour le site considéré*, les doses collectives, d'un facteur voisin de 3.

1.2.2. - Influence de la décroissance radioactive

Cette influence apparaît lorsque l'on compare les résultats pour l'iode et le plutonium. Le rapport des facteurs de dose irradiation externe I/Pu est de 4890. Le rapport des doses est de 4260. Pour l'inhalation, le rapport des facteurs de dose est $9,2.10^{-5}$; il est de 8.10^{-5} pour les doses collectives. Ceci signifie que, *pour le site considéré*, la prise en compte de la décroissance radioactive de l'iode ($T_{1/2} = 8$ j) ne modifie pas sensiblement les doses.

2. UTILISATION D'UN MODELE A TRAJECTOIRES RECTILIGNES

Le modèle à trajectoires rectilignes utilisé est décrit dans (3). C'est un modèle de panache gaussien qui prend en compte les phénomènes d'appauvrissement de décroissance radioactive et les phénomènes de réflexions sur le sol et sur le sommet de la couche de mélange. Les conditions de diffusion sont celles du schéma de DOURY (3), et définies par les conditions de stabilité et la vitesse du vent. On tient compte de roses des vents différentes par temps sec et par temps de pluie. Les probabilités d'occurence des différentes conditions de diffusion ont été calculées à partir des données observées sur le site et fournies par le Centre d'Etudes Nucléaires de Saclay.

Les données relatives à la population sont celles présentées dans (3), mais réparties en 104 secteurs de couronne (8 secteurs et 13 rayons de couronne), à partir de la grille de 10 km de côté.

3. RESULTATS - COMPARAISON DES DEUX MODELES (Tableau 2)

a) Examen global des années 1975 et 1976 (indépendamment de la direction de l'émission) - Les doses collectives obtenues avec DOSCOL sont 2 à 4 fois plus faibles que celles obtenues avec le modèle à trajectoires rectilignes. Par ailleurs, la variation d'une année à l'autre est au maximum de 20 %, les écarts les plus grands étant obtenus avec le modèle à trajectoires rectilignes. Il faut toutefois remarquer que, bien que ces différences ne soient pas très significatives, les valeurs de 1976 sont inférieures à celles de 1975 avec le modèle à trajectoires rectilignes, alors qu'elles sont supérieures avec le modèle DOSCOL.

b) Etude de l'année 1976, par section d'émission - Le maximum, pour le modèle à trajectoires rectilignes, est obtenu quand le vent vient du Sud (influence de l'agglomération parisienne) et du S-W (vents dominants à Saclay). Avec le modèle DOSCOL, ce maximum est atteint lorsque le vent vient du N et du N-E. Cette anomalie est due au fait que d'une part dans la grille population utilisée dans le code DOSCOL, la population parisienne est regroupée en un point situé au Sud du point d'émission, et d'autre part que les trajectoires sont rarement rectilignes aux échelles de distance considérées.

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RADIOELEMENT	^{85}Kr	^{239}Pu	^{131}I
Irradiation externe	$3,74.10^{-4}$	$1,33.10^{5}$	$6,5.10^{-2}$
Inhalation	0	$8,48.10^{4}$	7,82

Tableau 1 - Facteurs de dose utilisés, exprimés en rems par Ci.s.m^{-3}

		N	NE	E	SE	S	SW	W	NW	TOTAL 1976	TOTAL 1975
^{85}Kr Irrad.ext.	1	$1,36.10^{-5}$	$1,54.10^{-5}$	$3,09.10^{-6}$	$1,3.10^{-6}$	$1,0.10^{-6}$	$3,82.10^{-6}$	$5,94.10^{-6}$	$5,64.10^{-6}$	$4,97.10^{-5}$	$4,1.10^{-5}$
	2	$3,07.10^{-6}$	$3,63.10^{-6}$	$4,27.10^{-6}$	$8,24.10^{-6}$	$2,87.10^{-5}$	$1,90.10^{-5}$	$1,23.10^{-5}$	$2,62.10^{-5}$	$8,16.10^{-5}$	$8,4.10^{-5}$
^{131}I Irrad.ext.	1	$6,96.10^{-4}$	$7,07.10^{-4}$	$1,74.10^{-4}$	$4,30.10^{-5}$	$2,68.10^{-5}$	$1,04.10^{-4}$	$2,33.10^{-4}$	$2,4.10^{-4}$	$2,22.10^{-3}$	$1,8.10^{-3}$
	2	$2,41.10^{-4}$	$2,80.10^{-4}$	$2,85.10^{-4}$	$6,81.10^{-4}$	$2,91.10^{-3}$	$2,20.10^{-3}$	$6,33.10^{-4}$	$2,65.10^{-4}$	$7,42.10^{-3}$	$7,8.10^{-3}$
^{131}I Inhal.	1	$8,37.10^{-2}$	$8,5.10^{-2}$	$2,1.10^{-2}$	$5,17.10^{-3}$	$3,22.10^{-3}$	$1,25.10^{-2}$	$2,8.10^{-2}$	$2,9.10^{-2}$	$2,68.10^{-1}$	$2,2.10^{-1}$
	2	$2,01.10^{-2}$	$2,26.10^{-2}$	$1,88.10^{-2}$	$4,26.10^{-2}$	$2,29.10^{-2}$	$1,74.10^{-2}$	$5,40.10^{-2}$	$2,23.10^{-2}$	$5,83.10^{-1}$	$6,2.10^{-1}$
^{239}Pu Irrad.ext.	1	$1,6.10^{-7}$	$1,66.10^{-7}$	$3,97.10^{-8}$	$1,06.10^{-8}$	$6,54.10^{-9}$	$2,6.10^{-8}$	$5,5.10^{-8}$	$5,7.10^{-8}$	$5,21.10^{-7}$	$4,3.10^{-7}$
	2	$4,98.10^{-8}$	$5,75.10^{-8}$	$5,70.10^{-8}$	$1,23.10^{-7}$	$5,94.10^{-7}$	$4,46.10^{-7}$	$1,32.10^{-7}$	$5,43.10^{-8}$	$1,51.10^{-6}$	$1,6.10^{-6}$
^{239}Pu Inhal.	1	1020	1060	253	67,3	41,2	166	351	363	3320	2740
	2	226	254	212	477	2536	1920	612	248	6488	6860

Tableau 2 - Distribution des équivalents de dose effectifs collectifs, par secteur d'émission pour l'année 1976, et globalement pour les années 1975 et 1976

1 - Par le modèle DOSCOL

2 - Par le modèle à trajectoire rectiligne,

exprimés en homme-rem pour un rejet de 1 Ci/an.

THE STATUS OF RADIOACTIVE WASTE MANAGEMENT: NEEDS FOR REASSESSMENTS

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The purpose of this paper is to examine several aspects of the radioactive waste problem in the hope that so doing may emphasize the need for a fresh examination of not only the examples to be given, but the entire multifacited subject.

Wastes from Biomedical Clinics and Laboratories

It has been estimated that U.S. biomedical laboratories shipped a total of 2487 curies of radioactive waste to the burial grounds in 1978 (1). Relatively short-lived nuclides such as P-32, I-131, I-125, and S-35 contributed appreciably to the total. Because these nuclides have half lives measured in days, the reported annual shipment must be corrected for decay. Assuming that the wastes are generated at a uniform rate throughout the year, and allowing for decay, the total accumulation at year end would have totaled about 1300 curies. Seventy-two percent of the radioactivity would be due to two nuclides, tritium (720 Ci) and carbon-14 (221 Ci).

These quantities of tritium and carbon-14 are insignificant as a potential source of public exposure. Both nuclides are produced in nature by the interaction of cosmic rays with the atmosphere and both have been produced in even greater amounts in the testing of nuclear weapons. The worldwide steady state inventory of natural C^{14} is estimated to be 230×10^6 Ci, which delivers a dose of about 0.7 mrems per year to the world's population (2). It is estimated that by 1972 a total of 5.8×10^6 Ci were injected into the atmosphere as a result of weapons testing. This one source was equivalent to more than 2000 years of natural C-14 production.

If all C-14 used in biomedical laboratories in the United States were incinerated to $^{14}CO_2$, the steady state environmental inventory would eventually increase the dose from C-14 by less than 0.007 millirem per year.

Tritium accounts for about 55% of the waste radioactivity produced in 1978 by the biomedical facilities, a total of about 720 curies. This nuclide is also produced naturally by cosmic ray interactions, and the worldwide steady state inventory is estimated to be 34×10^6 Ci (2). The dose from naturally-occurring tritium is estimated to average 0.001 mrem per year. Tritium released without restriction by biomedical clinics and laboratories would gradually diffuse into the environmental hydrogen pool and its contribution to the human dose would be proportional to the amount present. The increase in dose would be about 0.002% of the tritium background dose of 0.001 mrem/yr.

Tritium, like C-14, was also produced copiously in tests of thermonuclear weapons, which resulted in injection of an estimated 4500 x 10^6 Ci into the environment. This was more than 100 times the steady state inventory from natural sources. The nuclide is also produced by nuclear reactors, which discharge between 0.1 and 1 Ci per megawatt electric (MWe) per year.

The figures given are of course applicable only to the wastes generated in the U.S. However, if the amounts were to be multiplied tenfold, which would be more than sufficient to account for wastes generated by all countries, the global impact of the incinerated emissions would still be negligible.

There remains the problem of evaluating the potential for exposure to individuals who live close to the incinerator and may therefore be exposed directly to the incinerator plume.

For purposes of estimating the potential magnitude of such exposure, it is assumed that a single large facility discharges 1% of the total quantity of radioactive wastes generated by the medical facilities and universities in the U.S. The wastes are incinerated, and the gaseous products are discharged at a uniform rate for 200 days per year, 8 hours per day. The exhaust gases are discharged from the incinerator stack at a rate $\sim 1\ m^3\ s^{-1}$. For planning purposes, it is specified that members of the general population should not be exposed to more than 10% of the limits recommended by the International Commission on Radiological Protection and the National Council on Radiation Protection and Measurements.

The concentration of tritium and C^{14} would, under these conditions, be 62 and 3.2 times the target level of 0.1 MPCa at the point of discharge from the stack. Even under the most adverse meteorological conditions, the effluent would be diluted within a few meters to concentrations well below the target level.

From the above, it appears that the biomedical and clinical laboratories could be permitted to dispose of most of their radioactive waste with no regulatory requirements other than those applicable to the wastes because of their chemical or physical characteristics. Instead, procedures established for management of these wastes are of themselves a waste; they waste time, money and resources. The elaborate record keeping, the careful packaging, the shipment for long-distances and the burial practices themselves are an unnecessary ritual.

Some Perspectives on High Level Wastes

The problems associated with the management of high level wastes are of course far more complicated than disposal of low level wastes from biomedical facilities. Apart from the fact that the subject is more complicated technically, a rational approach

to high level waste management has become entangled in a morass of political and quasi-scientific considerations to such an extent that the future of nuclear development in several countries is in jeopardy because of what I believe is a widespread misperception that a feasible way of managing high level radioactive wastes has not been demonstrated. We should not at this point become involved in the question of whether nuclear power development should be encouraged or discouraged. That question is beyond the scope of this paper and, in any case, should be settled independently of the question of how to manage radioactive wastes generated by nuclear power.

One of the most pressing questions is whether high level wastes can be isolated from the biosphere for a sufficient period of time by emplacement in geological repositories. In the debate over this issue, we have lost sight of the fact that mineral deposits in a wide variety of chemical forms remain isolated in nature for hundreds of millions of years under many environmental conditions. Perhaps we have been negligent until now in not having studied the many opportunities provided by nature. The mobilization rate of a deposit, and its rate of entry into the biosphere, should be quantifiable in terms of the properties of the deposit and hydrological and geochemical parameters that are capable of description. We should develop models that describe the physical and biological transport of trace elements from deposits found in nature. Studies of this kind have only recently begun and are few in number.

The natural reactor at the Oklo mine in the West African country of Gabon is one excellent example of what we can learn from nature (4). A study of the Morro do Ferro in Brazil is of more recent origin, having begun early in 1979 (5,6). The Morro do Ferro is a hill in the State of Minas Gerais, near the surface of which is an estimated 12,000 metric ton deposit of thorium that is believed to be as old as 80 million years. Because the deposit is in an advanced state of weathering, and because of the close chemical similarities between thorium and plutonium, studies of the rate at which the thorium is being mobilized from the deposit should provide useful information about the behavior of plutonium in a geological repository that has been breached.

Any conclusion that high level wastes can be isolated from the biosphere requires agreement as to the length of time for which isolation will be required. Pigford and his associates (7) have argued that after only 1,000 years, the potential risk is no greater than that from the ores from which the uranium was originally obtained. This approach is subject to the criticism that the relative hazard of the various nuclides depends on their chemical, physical and biological properties so that, curie for curie, the hazard indices may not be equivalent. Nevertheless, this analysis does serve to drive home the message that although some of the nuclides in high level radioactive wastes may have long half-lives,

they are present in relatively small quantities after a relatively short period of time. Storage times of a few thousand years are well within the range of human experience. There is a need to achieve a consensus as to the length of time during which the wastes must be isolated from the biosphere. It would help to resolve the contemporary controversy if there could be agreement that we are in fact concerned only with the need to isolate the wastes for about 1,000 years.

What Should Be the Role of the Marine Environment in Radioactive Waste Management

I will now turn to the oceans as an examule of a neglected environment that deserves a role in any program of radioactive waste management. The oceans cover 70% of the earth's surface and are the recipients of vast quantities of organic and inorganic debris carried into them by the rivers of the world.

More than two decades ago (8,9), the National Academy of Sciences began to examine the problems that would be encountered if the ocean were to be used for disposal of radioactive wastes. These and other studies estimated that huge quantities of radionuclides could be placed in the ocean deeps without hazard. However, a great prejudice has developed against using the ocean for waste disposal of any kind (non-radioactive as well as radioactive) and the U.S., along with many other countries, has stopped ocean disposal, even for low level wastes.

Society must be careful that the oceans as an ecosystem are not damaged by indiscriminate dumping of wastes. We must be careful that we don't allow accumulation of chemicals such as PCBs and DDT that degrade slowly and are known to be toxic to aquatic biota. But if we can find a waste form that can be deposited in the oceans subject to some common sense restrictions that will avoid ecological or cosmetic injury to the ocean environment, then why should we not take advantage of the opportunity.

Testing nuclear weapons has resulted in widespread dissemination of a broad spectrum of radionuclides in the oceans, particularly the Pacific. The total explosive yield of these tests is estimated to be about 366 megatons (MT) of TNT equivalent (10). Of this total, an estimated 72 MT was exploded under ground and can be neglected for the purposes of these discussions. The yields of all explosions conducted above ground (or under the oceans) thus total 294 MT.

Sources quoted by Miskel (11) estimated that 41% of the total yields were due to fusion and the remainder to fission. Using these ratios, it can be estimated that the fusion component of atmospheric explosions through 1978 totaled 122 MT and the fission component totaled 172 MT. As noted earlier, the bombs produced an estimated 4.5×10^9 Ci of H^3 and 5.8×10^6 Ci of C^{14}.

The fission and activation products produced by the explosions have been disseminated throughout the world, have entered the biosphere, and have been the subject of intense study by radioecologists from many countries. The data have been summarized elegantly in the periodic reports from the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR). However, reference to these reports shows that the dose estimates to human populations has been estimated from studies of the terrestrial food chains. This, despite the fact that the oceans cover 70% of the earth's surface and that many of the larger tests were conducted on Pacific atolls where fallout in the vicinity of the tests contributed additional radioactive debris to the Pacific. The dose commitments from marine food chains have not been taken into consideration presumably because they do not add significantly to the dose commitments estimated from terrestrial foods.

More than 95% of the total explosive yields of the tests took place between 1954 and 1962, at which time a limited test ban agreement was consummated which prevented atmospheric testing among the major nuclear powers. Deposition of the principal nuclides (H-3, C-14, Sr-90, Cs-137, and Pu-239) has been well documented and most of the debris has by now deposited on the earth's surface.

Despite the fact that the oceans have been the recipient of enormous quantities of radioactivity, marine sources of food have not contributed significantly to the dose received from fallout in those countries of the world for which data are available. The UNSCEAR emphasis has been on the terrestrial food chains because most food is derived from land sources. For example, in San Francisco, where representative diets have been monitored for strontium-90 for many years, fish and shellfish account for no more than about 0.2% of an annual strontium-90 intake that has ranged between about 1,000-3,000 pCi/yr. Similar data have been reported from diet studies in New York.

The lifetime dose commitment to endosteal cells of persons in the temperate zones from all strontium-90 produced in nuclear explosions up to the end of 1975 is estimated to be 116 mrad (13). Based on the San Francisco and New York data, the contribution of marine foods to this dose commitment would be about 0.1%, or 0.116 mrad. Note that this is the dose commitment, i.e., the dose that will accrue to an individual over his lifetime. The dose would of course be higher in populations which consume more seafood than people in New York or San Francisco.

Measurements of Pacific albacor during the period 1965-71 showed an average Cs^{137} concentration of 74 pCi Kg^{-1} wet (14). If a person consumes 1 Kg of albacor per week, the dose commitment will be about 1 mrem for each year of fish consumption. This would amount to total dose of about 30 mrem for a lifetime of fish consumption--a dose that is approximately 0.4% of the lifetime dose

received from nature. The dose from consuming albacor is probably representative of the dose to individuals who subsist on a high proportion of Pacific fish. It is the dose that has resulted from deposition (without precautions) of an estimated 27 million curies of Cs^{137}.

The British have set a unique example by the rational manner in which they have utilized the marine environment for disposal of radioactive wastes from windscale. They used critical pathway analysis to identify the limiting nuclides and the ecological pathways by which the nuclides can reach humans. Their preliminary studies began nearly 30 years ago, and a 10,000 Ci experimental release of wastes took place in 1952 over a period of about six months. Based on studies of the ecological behavior of the radioactivity releases during this and subsequent experiments, the quantities released were gradually increased, so that during the period 1955-65 the releases ranged from 3742 Ci to 7659 curies per month. The critical nuclides in these releases was shown to be Ru^{106}, which has a half life of 1 year (15).

Summary

Policies that dictate the procedures for management of radioactive wastes are being influenced by superstitions and prejudices that have no place in modern society. Even innocuous low level wastes are subject to absurd regulations that should be re-examined. For example, the recent problem encountered by biomedical facilities in the U.S. because of the closing of low level burial grounds could have been avoided in the first place since most of the wastes could be disposed of safely by onsite incineration or other methods applicable to the nonradioactive wastes from the laboratories.

Many forms of wastes can be safely emplaced in the marine environment where an inadvertant experiment resulting from the fallout of massive quantities of radioactive debris has provided us with information about the ecological behavior of the individual nuclides.

Several aspects of the high level waste management problem requires reexamination. Perhaps most important is the question of how long it is necessary to isolate the wastes from the biosphere. Plans for waste management would be greatly simplified if, as some believe, an isolation period of 1,000 years will be sufficient. If longer periods are required, we should look to nature for guidance. There are many mineral deposits that have remained in place for tens of millions of years under a variety of environmental conditions. Knowledge of the factors that influence the mobilization rates from those deposits should greatly assist construction of models to predict the behavior of long-lived nuclides in a geological repository.

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Projections of Organ Doses from Diagnostic Radiology in Radiation Protection and Control

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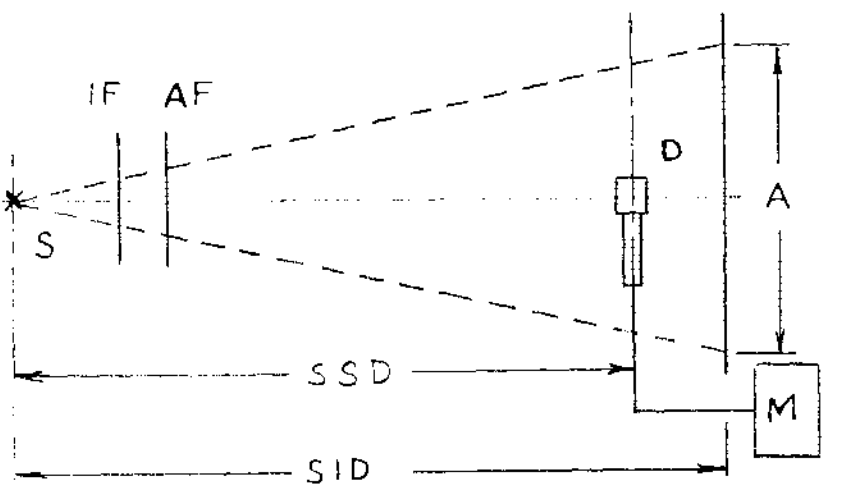
"Quality assured" means that we can expect a good image from a given x-ray machine; it does not necessarily mean that the dose from a given procedure is minimized. Since diagnostic x-rays contribute the largest fraction of manmade readiation exposure to the public (1,2), all practical measures should be taken to minimize organ doses while achieving the best possible film image. This implies integration of quality assurance and technique with organ dose estimation. Procedures for diagnostic quality assurance are described in numerous publications (3,4,5). However, the estimation of organ doses may be more or less laborious, depending on which of several methods (which may give different estimates) is used (6,7,8,9,10,11). For integration into a quality assurance program, we found the data in Rosenstein's Handbook on Selected Organ Doses (12) to be useful. These organ dose estimates are based on a combination of the handbook data with measured operating paremeters of the x-ray machine. The work presented here is based on measurements under typical operating conditions of two quality-assured "identical" x-ray machines, and demonstrates the incorporation of the estimated dose to the ovaries from these operating conditions into "minimum-dose" quality assurance.

Materials and Methods: The setup used is shown in the diagram below. All exposure measurements were made



with an MDH Industries model 1015C x-ray monitor (D-detector, M-digital readout in the diagram). The detector had an air-wall chamber whose energy response was relatively uniform over the range of energies used in diagnostic radiology, and the chamber diameter and length were comparable, thus minimizing directional dependance (8). Measurements were made on 2"identical" Picker single-phase full wave rectified x-ray machines with 0.5 mm Al inherent filtration and 2.0 mm Al added filtration, which resulted in x-rays whose HVL was 2.3 mm Al when the machines were operated at 85 kVp. For a given projection and view,; SID, field size, mAs, and kVp, the entrance exposure, E_x, was measured in mR. For kVp values other than 85, the HVL of the beam was determined from data in NCRP 33(14). The data in Rosenstein's handbook were used to estimate organ doses for several different diagnostic techniques.

Results: The entrance exposures from the 2 "identical" quality assured units were found to differ considerably. In a series of 9 measurements at kVp's of 40 to 120, one machine was always higher than the other; the mean ratio of the x-ray outputs was 1.72 ± 0.22!

As an example of dose estimation from a given diagnostic procedure, consider the dose to the ovaries from an AP projection of the lumbar spine on 14" x 17" film, when the measured entrance exposure was 224 mR, and the machine paremeters were: 85 kVp, 50 mAs, SID = 40", and HVL = 2.3 mm Al. From Table 9 in Rosenstein's handbook, which deals with the lumbar spine, we find the dose to the ovaries from a normalized skin exposure of 1000 mR, under the given operating paremeters, to be 168 mrads. The estimated ovary dose, therefore, is

$$\frac{224}{1{,}000} \times 168 = 38 \text{ mrads.}$$

To account for field sizes and SID values different from those in the handbook, the estimated dose, ED, is given by

$$ED = \frac{E_x}{1{,}000} \times D_{oh} \times \frac{(SID)_h^2}{(SID)^2} \times \frac{A}{A_h}$$

where E_x = measured entrance exposure, mR

$(SID)_h$ = SID value in handbook

SID = measured value

A_h = field size in handbook

A = measured field size

D_{oh} = normalized handbook value of organ dose.

When the HVL value differs from those listed in the handbook, as in the case illustrated here, then the value for D_{oh} is obtained by interpolation between the handbook values. The second x-ray machine, which was operated under conditions identical to the first one, produced a skin entrance exposure of only 152 mR, which leads to an estimated ovarian dose of 26 millirads.

Discussion: The results of this study confirm the findings of other investigators that there may be large variations in exposure rates from apparently "identical" machines. Thus, while quality assurance leads to good films, the actual patient dose may be questionable. Since the dose to any organ of concern, such as the ovaries in the illustrative example, is a function of the operating parameters, those parameters should be chosen not only to produce a good image on the film, but also to minimize the radiation dose to critical organs or tissues that are incidentally irradiated during radiographic examinations. Incorporating projected organ doses into a quality assurance program should contribute significantly towards attainment of this objective.

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COMPUTER AIDED DESIGN OF FAST NEUTRON THERAPY UNITS

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During the last decade several radiotherapy centers all over the world used fast neutrons with considerable success to treat certain types of cancer. Randomized clinical studies have shown that at least some groups of cancer respond better to fast neutrons than to megavoltage X-ray therapy. Correspondingly, there exists a strong demand to develop and to construct more and better fast neutron therapy machines to make this therapeutic modality available to more patients.

In an effort to help meet this continuously increasing demand and at the same time to reduce design costs significantly, a novel approach has been used at KMS Fusion, Inc. laboratories: A computer code, TBEAM, has been developed that can determine the physical characteristics of the neutron beam generated in a machine of given geometric configuration and materials' composition.

Neutron flux, energy flux, spectral composition of the incident beam, collimation and shielding efficiency, activity induced by the interaction of fast neutrons with various components of the equipment are among the design parameters determined by TBEAM. These and several other parameters indispensible to the evaluation of patient dose, scattered dose, occupational exposure of personnel are furnished by TBEAM, without the cost and effort involved in the actual construction of a candidate design. By the same token, TBEAM is eminently applicable to parametric studies as well as to comparison of the effects of various construction materials and geometric configurations used in candidate designs, and thus to optimization of design.

Required physical characteristics of the beam are to be determined by criteria set by the clinician. To be therapeutically useful, the fast neutron beam should have sufficient penetration to treat deepseated tumors, the biological shielding of the facility should protect personnel and patients. Collimation should be adjustable so as to restrict the beam to the size of the area to be treated, i.e., the penumbra around the edges of the beam resulting from stray radiation between the treated and shielded zones should be high. Beam intensity should be sufficient to permit relatively short treatment times. Induced radioactivity in various components should be shortlived to avoid both long waiting times between treatments and potential occupational hazards to the operating personnel. Neutron beams with narrow spectral width are conducive to faster, more accurate dosimetry.

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It appears that due to its unique space, time and energy characteristics, a neutron source generated by laser fusion could be used in a therapeutic facility with considerable advantages. Such a source, being monoenergetic at 14.1 MeV, has approximately the same penetration as Co^{60} gamma rays; being a point source collimation is simplified; and since the neutrons are delivered in very short bursts, the dose can be spread over several pulses spaced into time so that the integrated dose, in a given treatment session, can be accurately controlled.

A typical treatment schedule of 1560 rads, administered in 12 sessions requires 130 rads per session. Assuming a source-skin distance of 120 cm, and emission of 4×10^{14} neutrons at 14.1 MeV in each pulse, the average dose absorbed by a superficial tissue layer facing the incident beam is approximately 18.35 rads per pulse. Two pulses per minute would provide a convenient 3 - 4 minute irradiation time per session.

Laser fusion experiments are being performed at several laboratories on a routine basis. There is considerable experience at hand that is applicable to the development of a therapeutic facility that could satisfy all design criteria dictated by clinical considerations. Conceptual design of a therapeutic facility of this kind is under consideration at KMS Fusion, Inc.

The basic mechanical design of such a fast neutron therapy unit would consist of a spherical shell-shaped shielding structure at the center of which is located the point source of fusion neutrons. The therapeutically useful beam exits through a conically shaped collimating aperture. The opening angle of the collimator could be varied, according to instructions of the therapist by insertion or removal of appropriately shaped collimator liners. Should it prove necessary, remote control of collimator placement can be used to protect the operator from excess exposure to radioactivity that might have been generated due to previous exposure of structural components to the neutron field.

No attempt is made here to describe details concerning the optical path of the laser beam nor the fuel (pellet) injection mechanism, although these problems are being considered. The patient to be treated could be positioned as comfortably as possible on a bed-like treatment table and could be rotated isocentrically around the source if the clinical situation thus requires.

TBEAM has been developed as part of the effort toward a viable conceptual design. The input file of TBEAM contains, among others, the following design data describing the facility: relative configuration of source, shield, collimator and irradiated area, materials' composition of the shield collimator assembly and of related components, source strength, source geometry and possibly other parameters of the facility. Materials' composition data are entered as appropriate number densities. "One Hundred Group Neutron Reaction Cross Section Data Generated by SUPERTOG from ENDF/B" (published by ORNL-RSIC. DLC-24) is used as cross section input file.

TBEAM uses the method of statistical sampling (Monte Carlo) to solve the space, time and energy dependent neutron transport equation associated with the configuration and materials' composition of the design specified by the design engineer. The code traces the indi-

vidual source neutrons as they propagate throughout the shield collimator assembly and determines the energy and the position of each neutron at the instance of incidence. Those results, in turn, serve as input to a set of subroutines which compute spatial flux distribution, activation and spectral distribution, as requested by the user. The TBEAM library also contains a graphical package that puts out diagrams of spectral composition, incident flux vs position, energy flux vs position, and also a multicolored diagram of neutron traces, labeling points of incidence of uncollided and of scattered neutrons on the various regions with differently colored symbols.

Comparison of TBEAM generated narrow beam attenuation results show good agreement with published measured data. The TBEAM design code, due to its built-in flexibility, can accommodate energy deposition in homogeneous and heterogeneous media including such heterogeneous phantoms as, e.g., the standard reference man, along with the isodose curves required in treatment planning.

To make TBEAM applicable to accelerator or cyclotron driven fast neutron machines, an option has been included to deal not only with point shaped, but also with plate shaped sources; not only with monoenergetic sources, but also with sources of any given spectral composition. In a sample problem, the results of which are presented in Table 1, TBEAM has been used to compare physical characteristics of fast neutron beams emitted from two facilities having identical design except for the source geometry which was a point in Case A and a circular plate in Case B.

Results of the sample problem and of various other design studies performed with TBEAM, favor the point neutron source generated in laser. Fusion. They suggest that, with all other design characteristics being identical, the point source compared to a plate source generates a beam having a harder incident spectrum on the field of irradiation (conducive to better penetration); better collimation (i.e., penumbra/umbra contrast); better shielding (i.e., less scattered and leakage energy flux per unit energy flux on target and a softer spectrum of the scattered and leakage flux).

Table 1. COMPARISON OF PHYSICAL CHARACTERISTICS OF FAST NEUTRON BEAM A (POINT SOURCE) VS BEAM B (PLATE SOURCE)

	CASE A Point Source	CASE B Plate Source
Relative Energy Flux Incident on Field of Irradiation	192 :	100
Fraction of 14.1 MeV Neutrons in Total Number Flux Incident on Field of Irradiation	85:100	72:100
Ratio of Energy Flux Incident on First Penumbra vs That on Field of Irradiation	1.7:100	10:100
Ratio of Energy Flux Incident on Second Penumbra vs That on Field of Irradiation	0.17:100	4:100
Ratio of Energy Flux Incident on the Total Body Area vs That Incident on Field of Irradiation	0.07:100	83:100

The above example is meant to illustrate the capacity of TBEAM to evaluate and compare a wide variety of candidate designs at a fraction of the cost that would be involved in actual construction and experimental testing.

MEASUREMENTS OF THE EFFECT OF "THYROID BLOCKING" IN PATIENTS INVESTIGATED WITH ^{125}I-FIBRINOGEN

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The diagnosis of venous thromboses with around 4 MBq of ^{125}I-labelled fibrinogen is today a routine nuclear medicine investigation. At the degradation of the ^{125}I-fibrinogen, $^{125}I^-$ is released and is available for thyroid uptake. The biological halflife of fibrinogen is around 4 days. To reduce the uptake of $^{125}I^-$ in the thyroid stable iodine is given.

At Swedish hospitals the patients normally receive 100-300 mg KI per os daily for 1-2 weeks. The aim of the present work is to study the effect of such a blocking regime on the ^{125}I uptake in the thyroid.

MATERIAL AND METHODS

The measurements were carried out on 25 female patients over 50 years of age, who were given ^{125}I-fibrinogen for thrombosis detection in connection with gynecological surgery. The patients were given 300 mg KI daily mostly in 10-12 days after the injection of ^{125}I-fibrinogen. The first tablet of KI was given 1 h before injection.

The measurements of the ^{125}I content in the thyroid were made by means of a 124 mm (diam) x 1.5 mm NaI(Tl)-detector placed in the Malmö low background iron room. The detector was fitted with a brass collimator (100 mm length, 100 mm diameter) which was centered to the neck between jugulum and larynx with a neck-to-collimator distance of 10-20 mm. All patients were measured 9-12 days after the injection. At this time 80-90% of the ^{125}I was released from the fibrinogen.

The activity remaining in the plasma and tissue was found to give a background countrate over the neck which was 4-5 times higher than the countrate caused by the activity in the thyroid itself. This makes it imperative that the background correction is accurate.

An individual determination of this background was made for 14 patients by means of additional measurements over the arm and over the heart during the first two days. At this time the circulating activity almost completely makes up the countrate over the neck and therefore the relation between the countrate from circulating activity over the arm, the heart and the neck can be determined. The background on day 9-10 was then calculated by this relation from measurements over the arm and the heart on the actual day. The value of the background became somewhat different depending upon whether it was based on the arm or the heart measurements. The mean difference was 6%.

The reason is the difference in proportions of blood and tissue within the field of view of the detector. In the results the mean value of the two measurements was used.

For the group af patients who were measured only on day 9-12 the subtraction of the background was made by means of a mean value of the relation calculated from the 14 patients. This value was: (neck countrate/arm countrate) = 3.5±0.4. The uptake was measured a

second time on 13 patients, 38-59 days after the injection of ^{125}I-fibrinogen. At that time the background activity was negligible.

The data of countrates were transformed into activity in the thyroid by means of calibration with an IAEA standard neck phantom. Because of the great attenuation coefficient in tissue for the ^{125}I-photons, this calibration is not very accurate for individual patients. An accurate way to determine the activity of ^{125}I is to use the summation peak in the pulse height distribution (1). This could be done on the 10 patients who were measured day 38-59 when the background was negligible.

RESULTS AND DISCUSSION

For the 14 patients for whom an individual background subtraction was carried out the uptake of ^{125}I in the thyroid on day 9-12 is given in figure 1 as a fraction of the total injected ^{125}I activity.

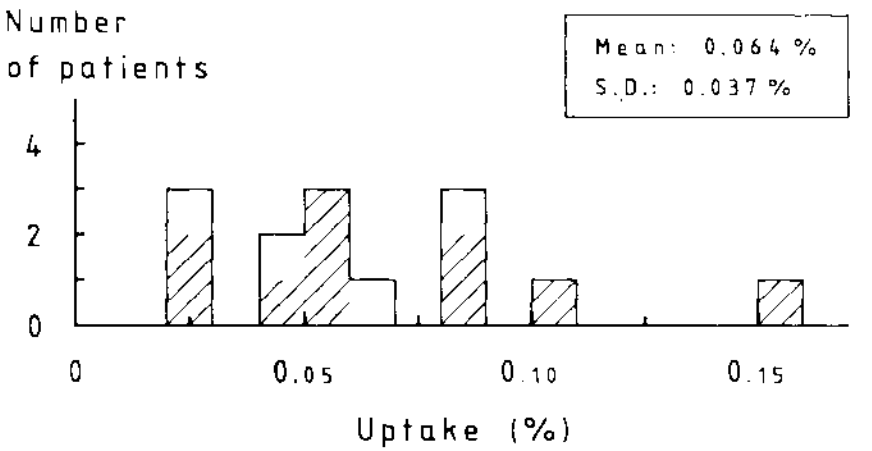

Figure 1. The content of $^{125}I^-$ in the thyroid on day 9-12 as a fraction of injected ^{125}I activity. The shadowed areas indicate uptake values calculated with the sum coincident method. The uncertainty due to counting statistics was ±5% (S.D).

The mean value was found to be 0.06% with a standard error of 0.01%. If the 11 patients for whom no individual background subtraction was made is included in the result the thyroid uptake was 0.07%. Thus the blocking regime used reduces the thyroid uptake by a factor of around 200.

It is of special interest to follow the ^{125}I content in the thyroid after the cessation of the blocking the ^{125}I activity in the thyroid of the 13 patients measured between day 38 and 59 after the ^{125}I-fibrinogen injection is given in figure 2. The mean value of the uptake was 0.5% with a standard error of 0.1%. This is around a

factor of 10 higher than at the end of the blocking. This means that after the withdrawal of the KI about 3% of the available $^{125}I^-$ has been taken up by the thyroid.

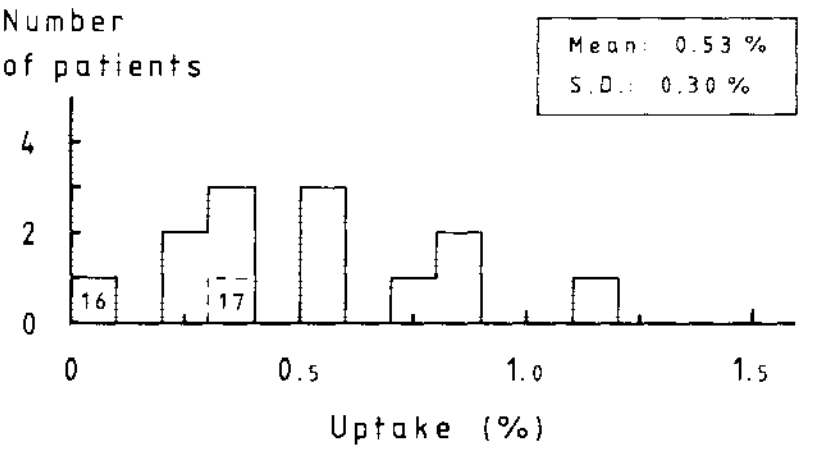

Figure 2. The content of $^{125}I^-$ in the thyroid on day 38-59 as a fraction of injected ^{125}I activity. The cessation of blocking was made on day 10-12 for 12 patients and on day 16 and 17 for two patients.

Two of the patients with low uptake values have got 300 mg KI daily for 16 and 17 days respectively instead of 10-12 days after injection.

SUMMARY AND CONCLUSION

A thyroid blocking using 300 mg KI daily for 10-12 days after injection of ^{125}I-fibrinogen gives a very efficient reduction of the thyroid uptake.

After cessation of the blocking the uptake increases. Studies of the effect of longer blocking periods with smaller daily amounts of KI are in progress.

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REDUCING OCCUPATIONAL RADIATION EXPOSURES AT LWRs

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Abstract:

The paper reviews briefly the occupational radiation doses received by nuclear power plants personnel, during a period of several years of operation. Comparisons are made between the data for BWRs and PWRs in order to identify the more "critical" reactor type, from a radiological point of view. Attention is devoted to GCRs too.

Furthermore the areas which contribute most to personnel doses are considered and briefly reviewed. The main actions to be carried out in order to reduce occupational radiation exposures at LWRs are discussed.

1. INTRODUCTION

The problem of occupational radiation exposures in nuclear power plants has been receiving increased attention in the latest years. Light-water reactors, in particular, are affected by high radiation levels in areas with high occupancy factors, what results in high occupational doses, as compared with doses absorbed by workers in other reactors, e.g. GCRs.

This fact highlights the importance of controlling, limiting and possibly reducing the radiation detriment to personnel during the operation and the maintenance of LWRs.

In order to reduce occupational doses we must have a realistic picture of the existing situation; in this paper we shall examine such a picture of the occupational radiation exposures at LWRs, with also the aim of identifying the reactor type that is more "critical" from a radiological point of view.

Attention is briefly devoted to the occupational doses in gas-cooled reactors, as compared with LWRs.

In addition we shall try to identify the factors which affect the doses at LWRs, on which it is poesible to act in order to reduce them.

2. ANNUAL COLLECTIVE DOSES

The dose data examined refer to 29 BWRs and to 40 PWRs of the western world, ranging from 150 to 1150 MW(e). Such data were obtained from about 100 reports and from informations directly collected by the Authors in 20 plants.

The reported values refer to the mean collective doses; variations in the doses, observed in different plants, may reflect either basic differences in the plant design or particular operational problems. Anyway if the differences in occupational exposures among similar LWRs could be traced to differences in plant design, rather than to accidental causes, it would help in designing better plants in the future.

The examination of the average annual radiation detriment allows to make a first evaluation of the radiological risk connected with the operation and maintenance of the plant; the time trends of the average collective dose equivalents are of particular interest in assessing the influence of plant age on the occupational hazard.

2.1. Light-water-cooled reactors

The general trend, during plant life, of the mean values of man-rem/year, man-rem/year-MW(e) and man-rem/MWy per reactor unit is examined. The two latest variables were considered in order to evaluate the radiological cost of the produced energy and of the installed power. The results are reported in fig. 1 and in table I for all the 69 LWRs.

Moreover, the 56 LWRs with electric power greater than 400 MW(e), which belong to newer generation, are examined (tab. I), in order to make an intercomparison between the radiological hazard in them and in all the LWRs.

2.1.1. Man-rem/year

The annual collective dose per reactor unit has a mean value, calculated over all the years of operation of the plants, equal to 370 man-rem/year in BWRs and to 315 man-rem/year in PWRs.

The behaviour of the average dose as a function of the plant age shows that, after an initial upward period of about four years, it seems to reach a levelling off value of about 600 man-rem/year in BWRs and of about 450 man-rem/year in PWRs (fig. 1). These data refer to about 300 reactor-years.

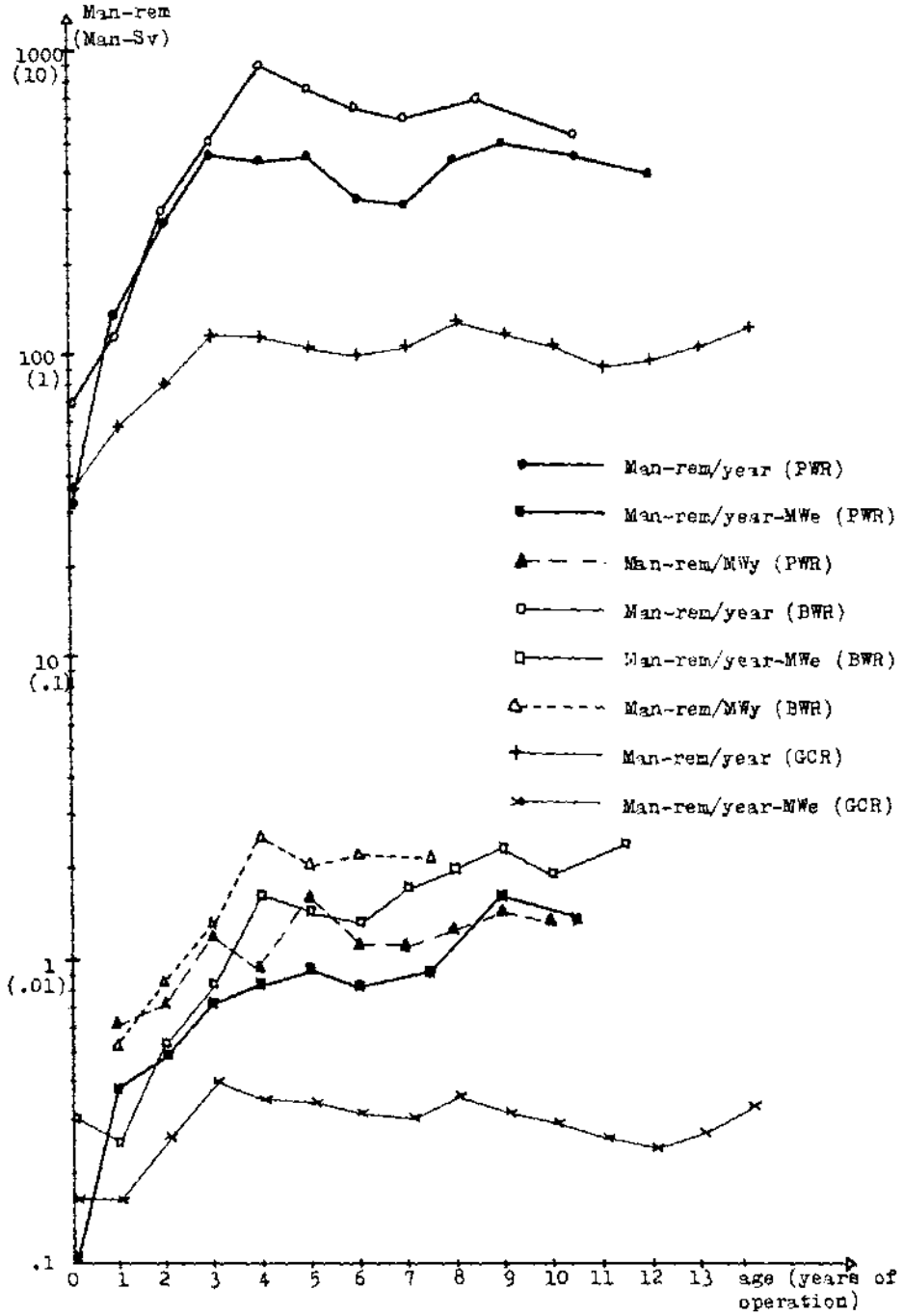
Man-rem
(Man-Sv)
1000
(10)
100
(1)
10
(.1)
1
(.01)
.1
0
1
2
3
4
5
6
7
8
9
10
11
12
13
age (years of operation)
Man-rem/year (PWR)
Man-rem/year-MWe (PWR)
Man-rem/MWy (PWR)
Man-rem/year (BWR)
Man-rem/year-MWe (BWR)
Man-rem/MWy (BWR)
Man-rem/year (GCR)
Man-rem/year-MWe (GCR)

FIG. 1

2.1.2. Man-rem/year-MW(e)

In order to consider on the same basis the nuclear power plants with a different electric power, we examined the annual mean collective dose normalized at the installed electric power.

The average values, for all the years of operation, are about 0.93 man-rem/year-MW(e) at BWRs and 0.75 man-rem/year-MW(e) at PWRs.

The time trend shows that, after an initial increase in the first 4 years of operation, the mean annual collective dose per electric power unit increases slightly from 1.5 to 2.0 man-rem/year-MW(e) in BWRs and from 0.8 to 1.3 man-rem/year-MW(e) in PWRs, during the following six years (fig. 1).

2.1.3. Man-rem/MWy

The behaviour in relation to age of this variable (fig. 1) shows that, after the remarkable increase during the first 4 + 5 years of operation, there is a levelling off at a value of $\sim$ 2.4 man-rem/MWy in BWRs and of $\sim$ 1.3 man-rem/MWy in PWRs.

Table I describes the situation of occupational doses at BWRs and PWRs. It is possible to see that the LWRs with electric power greater than 400 MW(e), which are also of new type, present a more favourable situation from a radiological point of view.

For what concerns the difference between occupational doses at BWRs and PWRs, at present time PWRs are responsible for lower doses: the difference between all BWRs and PWRs ranges from 16 % to 87 %, while for BWRs and PWRs of the new generation the difference ranges from 41 % to 108%, as far as the annual collective doses are concerned.

2.2 Gas-cooled reactors

In order to make a comparison between occupational radiation exposures in LWRs and in GCRs we considered the average annual collective doses also in these reactors.

Figure 1 shows the remarkable difference between the doses at LWRs and at GCRs: the mean values in GCRs, after about three years of operation, are about 100 man-rem/year per reactor unit and about 0.3 man-rem/year-MW(e). These values are calculated over 18 plants ranging from 150 to 590 MW(e).

TAB. I : Comparison between the mean values of annual collective doses in BWRs and PWRs

	All LWRs			LWRs > 400 MWe (new generation)		
	BWR	PWR	Δ *	BWR	PWR	Δ *
- Man-rem/year (for all the years of operation)	370	320	16 %	510	340	50 %
- Man-rem/year (after 4 ÷ 5 years of operation)	600	450	30 %	1000	480	108 %
- Man-rem/MWy (after 4 ÷ 5 years of operation)	2.4	1.3	85 %	2.2	1.1	100 %
- Man-rem/year-Mwe (for all the years of operation)	0.93	0.75	24 %	0.72	0.51	41 %
- Man-rem/year-MWe (after 4 ÷ 5 years of operation)	1.5÷2	0.8-1.3	87÷54%	1.5	0.8	87 %
- Rem/year per individual (for all the years of operation)	0.74	0.70	6 %	0.71	0.65	9 %

* The percentage difference Δ is referred to the dose value in PWRs :

$$\Delta = [\text{Dose(BWR)} - \text{Dose(PWR)}]/\text{Dose(PWR)}$$

3. ANNUAL INDIVIDUAL DOSES

Achievement of low collective doses is a desirable objective but it is not sufficient, as it could be achieved by having available a number of well-trained and skilled workers who could accumulate high individual doses.

On the contrary, the control of individual doses alone could be achieved by having available a large number of workers so as to share the dose among them; but in this way the total dose to personnel might be increased.

Only an adequate balance between the two requirements, individual doses reduction and collective dose reduction, can reduce the detriment to workers due to the operation and maintenance of the nuclear plant; for this reason, in addition to the collective doses, the individual doses also must be controlled and reduced. So our survey was extended to the examination of average annual individual doses at BWRs and PWRs: in this case also the situation is more favourable to the pressurized reactors. The mean annual individual dose, for all the years of operation, is about 0.74 rem/year at BWRs and 0.70 rem/year at PWRs, as reported in table I.

4. REDUCING OCCUPATIONAL EXPOSURES AT LWRs

As the ALARA criterion is difficult to be applied in the practice, mainly owing to the lack of a methodology for assessing the economic impact of the man-rem, another approach might be attempted; this process includes a review of the occupational doses with the aim of ensuring that the design and operating methods are such that the exposures are reduced.

A comprehensive program to reduce occupational radiation exposure at the new plants should be made of several measures that should act on various areas (fig. 2). The measures to be taken during the design stage should regard layout, ventilation, structural materials, monitoring, radioprotection program, and so on, while some of the action areas for consideration are:

1) reduction in maintenance and inspection time;
2) reduction in radiation fields;
3) reduction in failure rate of components;
4) contamination control.

The first area would include improved accessibility, which can be obtained by acting on the layout, and careful maintenance planning.

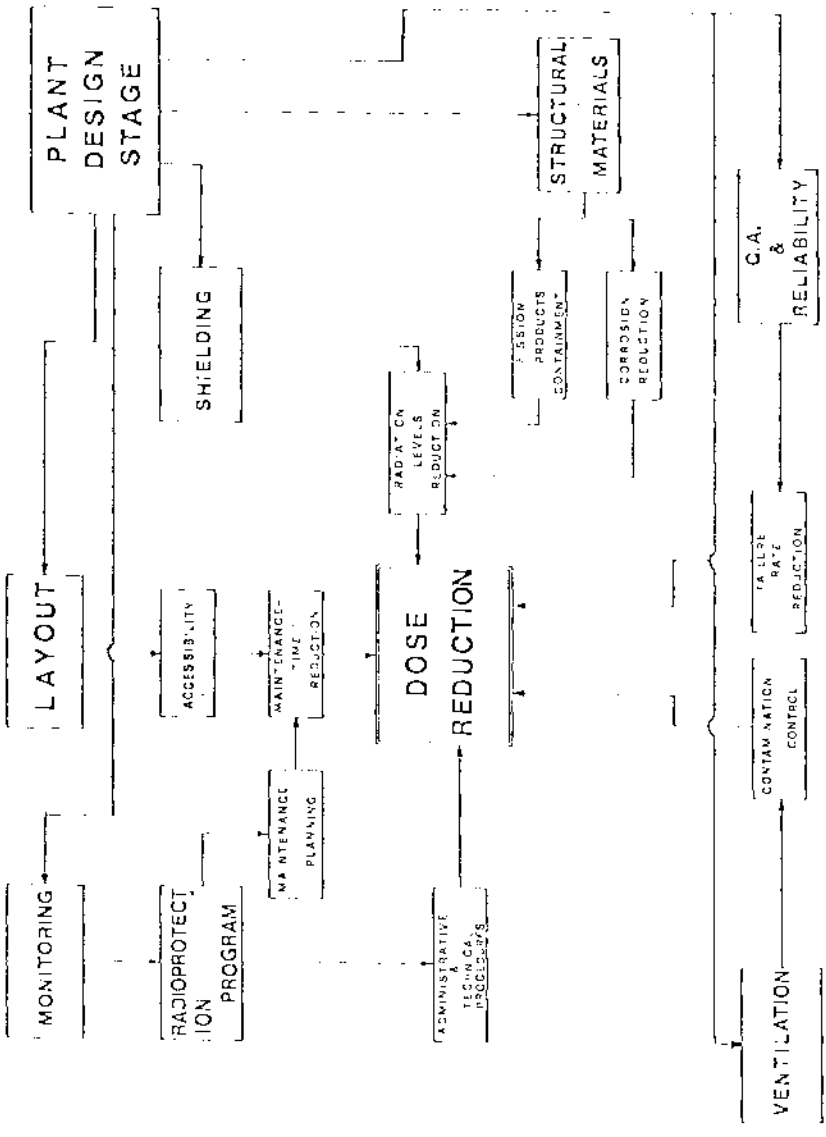

FIGURE 2 : SOME WAYS TO REDUCE THE OCCUPATIONAL DOSES

The layout of the plant, that is the arrangement of the buildings, of the working areas in them and of the components and equipment in the rooms, is a very important factor as the availability of space, improved accessibility and easiness of maintenance can reduce the maintenance time, optimize the radiation fields configuration and so reduce the collective and individual doses. In particular, potentially high radioactive components, that may require frequent maintenance throughout plant life, should be given a high priority in the plant design stage as to allocation in areas with low radiation and contamination levels and separation from other components.

The maintenance planning, that must be an important part of the radioprotection program, should include preventive maintenance programs which could lead to better use of time during shut-down and to dose reduction.

The second goal can be achieved by the use of adequate shielding, both permanent and temporary, and by materials selection in order to contain fission products and to reduce the formation of crud; also the cobalt content of materials must be limited.

The third area would include materials selection for durability and improved reliability especially of equipment with high radiological risk.

Finally the contamination control can be achieved by an adequate and flexible ventilation system, as well as "air-lock" doors.

At the plants already in operation the reduction of the doses can be achieved by an adequate maintenance planning and by suitable administrative and technical procedures established on the basis of the informations given by the monitoring system.

CONCLUSIONS

The occupational radiation exposures received by nuclear power plants personnel afford a method by which the level risk of workers can be evaluated and the "critical" reactor type can be identified; in addition the dose analysis is a useful tool by which the effectiveness of the measures used in a radiation protection scheme may be judged.

It is the Authors' belief that the implementation of the above-considered design measures, evaluated together through a balanced program, is a practicable approach that can lead to considerable savings in terms of occupational radiation exposure.

WHOLE BODY ELECTRON THERAPY USING THE PHILIPS SL75/10 LINEAR ACCELERATOR.

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Whole body electron irradiation is frequently used in the treatment of mycosis fungoides, a disease which may involve large areas of the body surface. The major technological problems encountered in whole body electron therapy (WBET) are as follows:

1. Delivery of a radiation dose to a layer 10-15 mm deep uniformly over all of the patient's body.
2. Minimizing contamination of the beam by Bremsstrahlung X-rays generated in electron interactions with matter located between the electron source and the patient.
3. Achievement of a practical treatment session time, taking into account limitations on the maximum dose rate available and on the maximum dimensions of the treatment beam.

In the most common method of WBET, patients are irradiated by beams of 3-4 MeV electrons while standing at a distance of 3-7 meters from the electron source to produce large treatment fields (1,2). An electron scattering layer is needed to achieve satisfactory dose uniformity. Since the beam intensity decreases because of the inverse square law while, at the same time, the X-ray/electron dose ratio is enhanced by preferential attenuation of electrons in the scattering layer and in air, this method requires an accelerator with a high electron output together with a low intrinsic level of X-ray contamination.

At a distance of 3 meters, the maximum intensity of the standard 4 MeV electron beam from our Philips SL75/10 linac is only about 10 rad/min when a 3 mm Perspex scattering plate is present while the X-ray contamination increases from a tolerable value below 2% at 1 meter to about 8%. These conditions are not acceptable, hence a long distance method is not suitable for our machine.

An alternative procedure is to treat the patient at a shorter distance with an electron beam that scans the surface of his body. For example at Manchester a scanning system has been constructed for the SL75/10 linac in which a moving platform replaces the usual treatment couch (3). This approach was rejected because it was felt that the adopted method should introduce minimum changes in the routine of an already busy treatment machine.

In this paper we describe the method used at our center for WBET which was developed under the constraint that no special equipment or modifications to the linac should be required.

MATERIALS AND METHODS

Treatment was performed with an electron beam of nominal energy 4 Mev from the Philips SL75/10 linac. No electron applicator was

attached and no scattering layers were present except for the beam exit window, the monitor ionization chamber and the air space between the linac and the patient. During irradiation the patient reclined on the regular treatment couch at a source-to-skin distance of 150 cm. The collimator was fully open.

Measurements of the depth dose properties of the electron beam were made with the aid of thin polystyrene sheets and a parallel plate ionization chamber (S.H.M. build-up chamber) connected to an electrometer (Keithley Model 600B). The ionization chamber was calibrated against a Cobalt-60 source using the method recommended by the H.P.A. (4). Dose distributions in a Rando Phantom were determined by means of thermoluminescent detectors (Harshaw TLD-100) and X-ray film (Kodak Type XV-2). Dosage to patients was monitored by taping TLD detectors to the skin.

RESULTS

Properties of the radiation beam

To achieve the clinically desired penetration, the energy of the electron beam was adjusted so that the 80% depth dose occurred at a depth of 13 mm. Fig. 1 shows the behavior of the central axis depth dose of the beam used for whole body treatment. At a depth of 5 cm the ratio of the X-ray dose to the peak electron dose is 1.7%. Fig. 2 shows profiles of the electron beam and its X-ray component. Both beams are approximately gaussian in shape with a F.W.H.M. of 50 cm.

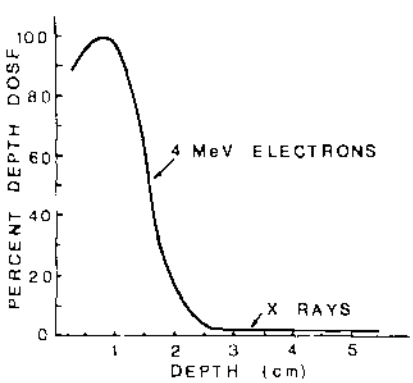

Fig. 1: Central axis depth dose of the treatment beam.

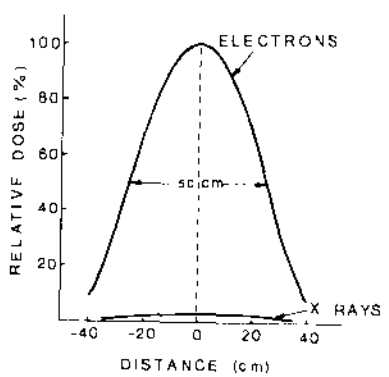

Fig. 2: Profile of the treatment beam.

Treatment technique

To cover the entire surface of the patient's body it is necessary to apply multiple fields. Because of the gaussian nature of the beam the matching of adjacent fields is not critical. Fig. 3 shows how a uniform dose is produced over a large cross section when 2 beams are joined at the 50% value of the profiles. An error of 1 cm in the separation distance causes an overdosage or underdosage of only 5%.

The patient lies in prone and supine positions during treatment. In each case a pair of fields is needed to include the width of the body and 5 pairs are used to produce uniformity over a patient up to 200 cm tall.

If the beam axes are perpendicular to the plane of the treatment couch, at the lateral edges of the body the radiation will intercept the body cross section at tangential incidence. This leads to regions of underdosage at the body edges (3). To overcome this problem, the gantry was rotated through a small angle to shift the points of tangential incidence away from the midplane. The same angle (20^{o}) was used for the anterior and posterior fields. The beams incident on a typical body cross section are shown in Fig. 4.

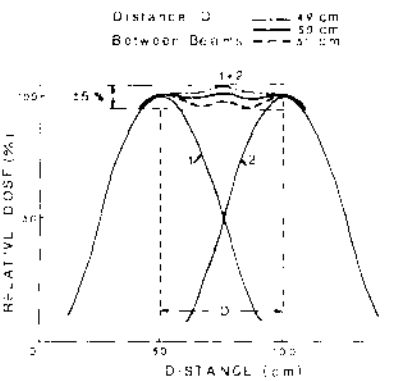

Fig. 3: Resultant profile when 2 beams are joined.

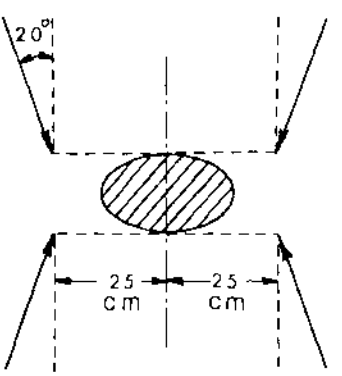

Fig. 4: Beams incident on a body cross section.

TLD and film measurements made with the Rando Phantom indicated that variations in the peak electron dose are within ±15% over the entire body except for the groin, axillae and soles of the feet which are shielded by other parts of the body. The depth of the peak dose depends on the angle of incidence. Fig. 5 shows a film representation of the dose distribution in the phantom when exposed

to the set of WBET beams.

Treatment is given in 4 fractions per week. One quarter of the body surface is irradiated to 400 rads at each session. The dose rate at the patient is 100 rad/min. and the time for each treatment session is about 30 minutes. The groin, axillae and soles of the feet are treated separately by small electron fields to compensate for under dosage during the whole body irradiation. The eyes, fingernails and toenails are protected by lead shields during treatment.

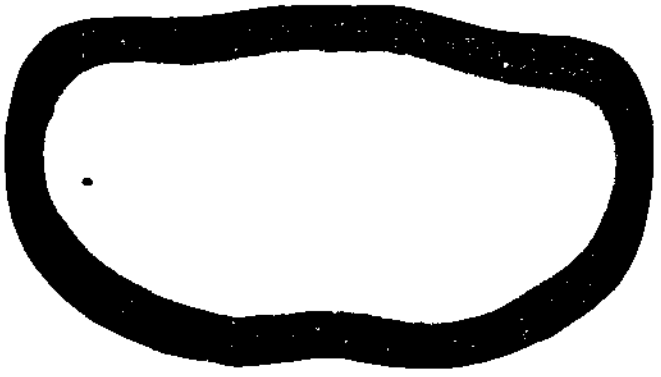

Fig. 5: WBET dose distribution in the Rando Phantom as recorded on X-ray film.

CONCLUSION

Our technique provides a practical solution to the clinical requirements for WBET with respect to uniformity of electron dose and low X-ray contamination. Its implementation doesn't require special equipment or modifications to the linac.

ACKNOWLEDGEMENT

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RADIOECOLOGICAL MODELS FOR ESTIMATING SHORT AND LONG-TERM EFFECTS OF RELEASES IN THE NUCLEAR FUEL CYCLE

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INTRODUCTION

The increasing use of nuclear energy will mainly through the production and reprocessing of nuclear fuel and the long-term disposal of the radioactive waste introduce increasing amounts of radionuclides into the biosphere. For the estimation of the radiological consequences when these radionuclides,sooner or later,reach man it is necessary to know not only how the nuclides are distributed and retained within the body,but also how the nuclides are transported in the biosphere before they directly,or through food-chains,reach man. The only way to study the transport from one environmental pool to another is the direct measurement of the activity concentration for the radionuclide of interest. Such measurements have now been carried out for a considerable time. The source of the radionuclides studied has up to now mainly been fall-out from nuclear weapons tests. Because of this,it has been possible to study only a limited number of radionuclides and still a great deal of experimental work concerning the behaviour of long-lived fission products and transuranium elements,other then plutonium,remains. The results can be used to design a more complex system for transport in the biosphere or in the human body. Such systems,normally referred to as compartmental systems can be used to predict individual and collective dose equivalent arising from the use of some nuclear energy practice.

However,due to the lack of reliable data for many radionuclides regarding their behaviour in the biosphere the use of mathematical models is at present in most cases limited to yield upper limits of the parameters studied rather than quantitative estimation of these parameters. For some applications,especially regarding the disposal of nuclear waste, calculations extending millions of years ahead are used. For static systems, the reliability of the transport factors used will in such cases be of vital importance. The need for efficient mathematical methods of solution is also evident.

A GENERAL MATHEMATICAL MODEL

For a compartment system consisting of n compartments the net transport rate of the substance y into the compartment i can be expressed using the general expression

$$\frac{dy_i(t)}{dt} = \sum_{j=1}^{n} y_j k_{ji}(t) - \sum_{j=1}^{n} y_i (k_{ij}(t) + \lambda) + \frac{ds_i(t)}{dt} \qquad (1)$$

where k is the transfer coefficient expressing the fraction of substance y that per unit time is transferred from compartment i to compartment j and λ is the physical decay constant. The last term corresponds to the presence of a source of the substance in compartment i. A somewhat simpler way of expressing equation (1) is a notation of the form

$$\dot{\vec{Y}}(t) = \vec{Y}(t) \cdot K(t) + \dot{\vec{S}}(t) \qquad \{2\}$$

where K is the transfer coefficient matrix and $\dot{\vec{S}}$ is the source term vector. The method of solving equation {2} will strongly depend on the time variation characteristics of the matrix K and the vector $\dot{\vec{S}}$. Some, or all, of the transfer coefficients and elements in the source vector can independently be chosen to continously vary with time following, for instance, some given function or polynomial approximation of given points. Furthermore, these terms can behave in a stochastic manner. The generation of the specific values during the calculations is preferrably done with Monte Carlo technique. The number of transfer coefficients with stochastic behaviour must, however, be limited due to the very long computer time to obtain good statistics.

In both of these cases with a dynamic coefficient matrix and source vector, the change in numerical value of the derivatives may be so great that the step in time when performing the integration of the equation system must be made very short. Iterative methods are therefore recommended, as the Kutta-Merson algorithm or the Adams-Moulton predictor-corrector scheme.

In the case of a static coefficient matrix and source term it is possible to use another algorithm to solve the equation {2}. (1) The solution to this set of equations will be of the form

$$\vec{Y}(t) = \vec{Y}(0) \cdot e^{K \cdot t} = \vec{Y}(0) \cdot B \qquad \{3\}$$

The quantity B is also a matrix, which can be evaluated from the matrix K using serial expansion. The main problem when performing this, is that a large number of terms must be included if some eigenvalues of the matrix K·t are large. This can be avoided if the resulting serial expansion matrix is multiplied with a geometrical series which is chosen in a way that only a limited number of terms in the serial expansion will be used. The inverse quotient of this series is used to convert the resulting matrix by a binomial expansion resulting in the matrix B. Using the solution {3} instead of an iterative method will make the calculations run up to 10 000 times faster on a computer depending on the characteristics of the coefficient matrix. Figure 1 represents a simple system which is a good test on the precision in the algorithm used, especially for compartments 1 and 2. Table 1 gives the amount of substance in compartment 2 after a time of 2 time-units after a pulsed injection in compartment 1 for some different mathematical methods. It is a well known fact that the Kutta-Merson and Adams-Moulton methods both are very slow giving, however, excellent precision.

Figure 1.

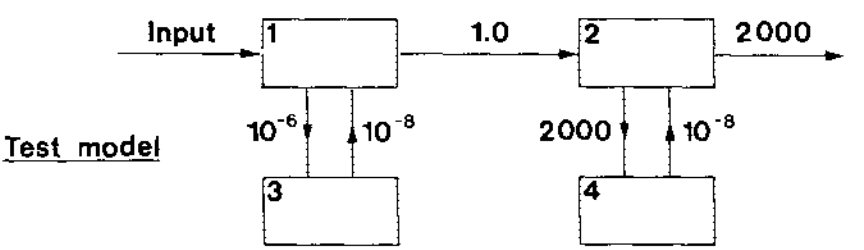

Table 1

Method of solution	Step-length	Amount of substance	Rel. error (%)	CPU time (s)
Analytical	-	6.8'536·10^{-5}	-	-
Kutta-Merson	h	6.8'595·10^{-5}	-	360
Adams-Moulton	h	6.8'506·10^{-5}	-	313
Matrix exponentiation	h	6.76877·10^{-5}	-0.8	1.1
Matrix exponentiation	h/20	6.8'486·10^{-5}	-0.003	16
Matrix exponentiation	10 h	4.96676·10^{-5}	-14.6	2.6

The matrix exponentiation mentioned above will, for the same step-length give a result within 1% of the exact value with a substantial decrease in computer time. It can also be seen that this algorithm must be used with caution. An increase in step-length immediately reduces the precision to an unacceptable level. For the same precision as the two iterative methods, the computing time increases but is still much shorter.

EXAMPLES OF APPLICATION

A field where mathematical models are frequently applied is on the spreading of radionuclides from the disposal of high-level radioactive waste. We have studied the two ways of disposal proposed by the Swedish Nuclear Industry (2,3).

The first alternative concerns the deposition of waste from the reprocessing of 10000 tons of UO_2 fuel. The second deals with the direct deposition of the burned-out fuel. The deposition takes place in primary rock at a depth of 500 m. The main difference between the two alternatives is the encapsulation material which for the latter mainly is copper. In this case the capsules are assumed to withstand corrosion for a certain time (100 000 years),after which the radionuclides during 500 000 years leak out into the biosphere. On their way to the first recipient in direct contact with the biosphere they are being delayed in the primary rock.

We have applied our technique on a 20-compartment global model to study the effects of earlier breakdown of the capsule and a faster leaking rate, (4). Furthermore, in order to yield the "upper limits" we have taken no notice of a possible delay in the primary rock. A combination of these factors are of particular interest for the daughters in decay chains. It is almost impossible without computers to calculate their input activity rate as a function of time due to the cooperating factors of capsule breakdown and leaking rate on the amount which at that time is built up in the capsule. An example illustrating this is the 4n+1 series. Np-237→Pa-233→U-233→Th-229. In table 2, the relative activity inflow rate of Th-229 into the primary recipient normalized to a capsule breakdown at 100 000 years after deposition and a leakage time of 500 000 years is given.

The combined effects of breakdown time and leaking rate can clearly be seen in the bottom row, which has a quite irregular pattern due to the balance between leaking rate and buildup during leaking time. These figures are also proportional to the individual absorbed doses

Table 2

Th-229	Leaking time (years)			
Capsule breakdown (year)	500 000	50 000	5 000	500
100 000	1.0(225000)	7.29(100000)	72.9(100000)	729(100000)
10 000	0.77(210000)	1.16(35000)	3.48(10000)	34.8(10000)
1 000	0.75(201000)	0.80(31000)	0.27(4000)	0.46(1000)

The figures in parenthesis indicate the time (in years) when maximum inflow occurs.

from thorium in the vicinity of the waste site. For the collective doses thorium is of particular interest due to its high sedimentation capability. The collective doses arising from thorium isotopes are not proportional to the amount that is directly injected into the shallow and deep ocean water compartments, nor to the small amount that is resuspended from the sediments, but only to the amount that is produced from the decay of its mother nuclide in these compartments.

SUMMARY AND CONCLUSION

Mathematical models are, if used with caution, an important tool for evaluating effects of releases of radionuclides. For some elements, very little is known regarding their behaviour in the biosphere, and complementary experimental work must be carried out. The non-static and stochastic nature of the biosphere must be taken into account regarding the transport of substances, thus having demands on the algorithms to be increasingly efficient.

Appendix: Brief description of algorithm used.

$$e^{K \cdot t} \approx \left(1 + K \cdot t + \frac{(K \cdot t)^2}{2!} + \frac{(K \cdot t)^3}{3!} + \ldots\ldots\right) = \sum_{n=0}^{\infty} \frac{(K \cdot t)^n}{n!}$$

$$\sum_{n=0}^{\infty} \frac{(K \cdot t)^n}{n!} \approx \left(\sum_{n=0}^{m} \frac{(K \cdot t)^n}{n!} \cdot \frac{1}{\beta^n}\right)^{\beta}, \text{ where } \beta^m \geq \frac{(K \cdot t)^m \cdot 10^m}{m!}$$

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RISK ASSESSMENT PERSPECTIVES IN RADIATION PROTECTION

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Introduction

Risk assessment involves two separate activities; risk determination and risk evaluation. The first activity involves identifying and estimating the risk from some action; the second activity involves a socio-political decision about the action and residual levels of risk after risk reduction techniques have been applied. Since risk determination is a well-developed activity in the radiation protection field, the major focus of this paper is on risk evaluation.

The International Commission on Radiation Protection has published a system of dose limitations which to some extent addresses the risk evaluation process. The first step in the system is optimization where collective dose is reduced by the addition of controls, using the marginal cost-effectiveness of the cost controls for reduction of detriment as a criterion. A value is assigned to the avoidance of a man-Sievert to establish a cut-off value. The establishment of this value is a socio-political decision. The second step is justification where a balance of benefits and costs are made. This is, indeed, a socio-political decision, and will be reviewed in terms of risk evaluation. Application of dose limits, the final step, is an equity balancing step to assure that no individual receives an undue proportion of risk.

Optimization and the establishment of dose limits are not purely socio-political decisions, but represent managerial decisions for ease of administrating radiation protection policies. The choice of fixed values for the optimization criterion (value of a man-Sievert) and for dose limits are decisions to ease the process of regulation; otherwise, each case would be balanced on its own merits. The impact of such managerial judgments involves the balancing of costs to the public (including risks) against the ease and cost of regulation.

Justification is another matter. It is a balancing of all factors after optimization is complete.

GENERAL FORM OF JUSTIFICATION

Although a justification decision is not purely numerical, a mathematical structure of the justification process can be quite useful for understanding the process and application. The structure presented here is thus a study mechanism not an end in itself.

Justification is determined in general form by examining the difference between the new (or changed) practice (n) and a reference condition (r) in the form of a difference equation.

$$(B_n - B_r) = (V_n - V_r) - (P_n - P_r) - (X_n - X_r) - \alpha_j (S_n - S_r)$$

EQUATION 1

where B = Benefit (NET)
V = Worth of practice
P = Production cost of introducing the practice
X = Cost of radiation control
S = Detriment from increased exposure
α_j = Value of a man-Sievert for justification

$$\Delta B = (\Delta V - \Delta P) - \Delta X - \alpha_j \Delta S_j$$

EQUATION 2

The parenthesis around $\Delta V - \Delta P$ is to emphasize the difference of these parameters from those associated with radiation protection and exposure parameters. Thus justification consists of two parts, the first involving the worth and costs of the practice, and the second involving the implementation of radiation protection practices and a judgment of the social costs of the remaining detriment. This separate identification of the value of alpha for justification is to indicate that it may be a different value from that used in optimization. Usually optimization preceeds justification and is based upon assuring that:

$$\frac{\Delta X}{\Delta S_o} = -\alpha_o$$

EQUATION 3

where the subscript $_o$ refers to optimization for both the value of alpha and the change in detriment.

Optimization is well covered elsewhere, and will not be covered in detail here, but it is important to note the differences for alpha and detriment for justification and optimization. The total detriment, ΔS, consists of two parts: 1) the detriment removed by optimization at cost ΔX, and 2) the remaining detriment not removed, but attributable to the practice. It is the latter detriment which is of concern for justification since ΔX in monetary units will have been spent to remove ΔS_o. Thus:

$$\Delta S = \Delta S_o + \Delta S_j$$

EQUATION 4

The remaining change in detriment from the change in practice, ΔS_j, is the parameter of concern for justification.

The value of alpha for justification, α_j, may differ from that for optimization, α_o, but α_j cannot be less than the α_o. This arises since the value of alpha for optimization, α_o, may be limited by the availability of resources. When all radiation protection and other hazard reduction needs are considered and the value of prevention of a stochastic fatality or illness is considered "across-the-board", only a limited amount of resources may be available for the total. When such limits apply, a selection of a lower value for alpha in justification means that radiation protection will have a disproportionately lower regard in justification decisions. This in turn would reflect

an uneven allocation of resources.

Alpha may be higher for justification since here one uses a value inferring what society ought to do (not just what it can do for limited resources in optimization), and in many cases the radiation impact costs may not even be specified in monetary terms since the judgments involved may be at a higher level of abstraction.

Radiation Risks Versus Other Societal Risks

The justification equation must take into account other risks in addition to radiation risks. For this reason, the justification format must be expanded:

$$AB = (AV-AP) - \sum_{i=1}^{n} (\Delta x_i + \alpha_{ij} \Delta S_{ij})$$

EQUATION 5

where subscript 1 might be for radiation detriment and other subscripts might be for other types of risks such as exposure to toxic chemicals.

This immediately implies that radiation detriment can be related to other detriments in society such that choices of α_{ij} represent judgment as to the relative impact of these. There are many different kinds of detriment and risks (the potential from harm) involved with many factors affecting the valuation of these risks, i.e., choice of value of α_{ij}. Some of the factors, which are treated in greater detail in my book AN ANATOMY OF RISK, are:

Voluntary vs. involuntary imposition of the threat
Immediacy vs. latency between cause and affect
Controllability of the threat
Individual risks vs. population risks
Familiarity with a risk and its consequences
Certain vs. uncertain consequences
Etc.

Risks can be put into perspective by comparing them with benchmarks, i.e., risks of a similar nature to those being considered. They are used to provide some insight as to the magnitude of risks, but do not imply acceptability.

For radiation two cases are considered here to illustrate other α values: 1) the relative potency of radiation vs. carcinogenic chemicals, and 2) the problem of addressing rare events.

Relative Potency

Relative potency is a means of comparing two or more carcinogens to determine their ability to cause cancer in certain organs based upon similar concentrations of material. It does not represent the relative risk of the materials, since an exposure pathway must exist for risk to occur.

One approach for relative potency is the parts per million (or per billion) to cause a lifetime risk of one in one million. Some values for ingestion of toxic chemicals expressed in this form are shown below.

TABLE 1
SOME EXAMPLES OF RELATIVE POTENCY
Values are in Parts per Billion (Ingestion)

COMPOUND	DOSE (FDA)(1)	DOSE (Wilson)(2)
DDT	0.4	
Dimethylnitrosamine	0.05	0.002
Ethylene Thiourea	2.0	
NTA	260.0	
Vinyl Chloride	6.7	
Saccharin		600,000
Aflatoxin		0.0002
Benzo-a-Pyrene	700	
Acrylonitrile		3

Radiation risks to individuals are such that an exposure of .01 Sv/yr. (1.0 Rem/yr.) over a 70 year period results in a risk of $2x10^{-4}$ fatalities/year (See Appendix). A level of 0.7μ Sv/yr. (70 μ rem/yr.) represents a one in a million lifetime risk on this basis.

Maximum permissible doses (MPC's) are based upon the daily concentration in air or water for 50 years of occupational exposure for 168 hour week. In the absence of ICRP #30 where new allowable limits for intake (ALI) will be published, other values from 10 CFR 20 can suffice. Three isotopes have been selected for the ingestion pathway assuming the material is in soluable form. These are Sr^{90}, Pu^{239}, and I^{129}, selected for their range of specific activity. MPC's must be converted into μg/ml to get parts per billion by weight through the specific activity factor and reduced to get an equivalent risk of 0.7μ Sv/yr from approximately 0.5 Sv/yr, a factor of about $1.4x10^{-5}$.

Nuclite	MPC (10CFR20) Water Soluable	Specific Activity	PPB for 1 in a million lifetime risk
Sr^{90}	$3x10^{-7}$ μC_i/ml	$1.44x10^{2}$ C_i/gm	$3x10^{-5}$
Pu^{239}	$5x10^{-6}$ μC_i/ml	$6 \quad x10^{-2} C_i$/gm	1
I^{129}	$6x10^{-8}$ μC_i/ml	$1.8\ x10^{-4} C_i$/gm	5

Ingestion of Sr^{90} appears to be nearly a hundred times more potent than aflatoxin while ingestion of Pu^{239} and I^{129} seem comparable with ingestion of vinyl chloride or acrylonitrile. Materials with high specific activity are, of course, more potent on a weight basis than a curie basis.

Rare Events

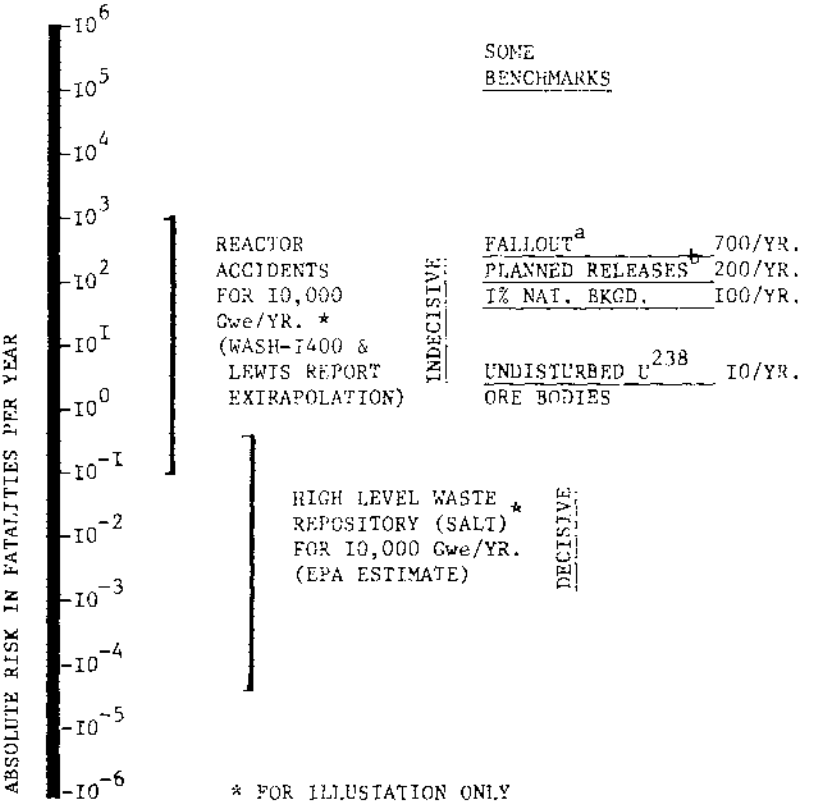
Nuclear and waste disposal accidents are also risks that may be considered in the justification equation. However, there are limitations to estimating the likelihood of rare events. Statistical methods provide measures of the relative likelihood of events for

those of relatively high frequency, which can be validated. There are also models for estimating the relative likelihood of rare events which cannot be validated. However, in no case do they predict when, where, or of what magnitude an event will occur. In the case of rare events, this is exacerbated by the uncertainty in the specification of the models themselves and their inherent lack of validation.

These methods can only be effective on an absolute risk basis if the very wide bands of uncertainty common to such estimates do not encompass the range of acceptable levels of risk. If the bands of uncertainty of probability estimates encompass the range of acceptable risk levels, the decision cannot meaningfully be based on probability estimates. Figure 1 illustrates this problem using nuclear accidents and high level radioactive waste disposal as examples.

FIGURE I

ABSOLUTE RISKS, UNCERTAINTY, AND BENCHMARKS



* FOR ILLUSTATION ONLY

a. WORLD WIDE RISK FROM EXISTING WEAPONS FALLOUT-Ce^{137}, C^{-14}, Pu

b. 10,000 Gwe/YR. BASED UPON 40 CFR 190

The scale on the left is a measure of absolute risk in terms of the probability of the number of fatalities that might occur in a year. Some benchmarks are shown on the right, including world wide fall-out from nuclear weapons already committed, planned releases from the nuclear fuel cycle for 10,000 Gwe-years of operation (the maximum production possible from available uranium resources without breeding), one percent of natural radiation background, and radon and radiation from undisturbed uranium ore bodies. These benchmarks are only to provide perspective; they do not, by themselves, imply acceptability.

The range of risk estimates for a high level waste repository for all high level wastes (10,000 Gwe-years) lies well below the benchmarks. Thus, a decision on high level waste is resolvable by probablistic methods. The range of risk estimates for all nuclear reactor accidents (10,000 Gwe-years of operation) is shown based upon WASH-1400, WASH-1400 COMMENTS, The Lewis Committee report, and extrapolation. The exact range may be argued, but it probably envelopes all of the benchmarks, making any decision based upon probablistic analysis alone indecisive. In this case, although it may be possible to refine the estimates to some extent, it may be impossible to reduce the residual uncertainties to a level for which meaningful decisions can ever be made by this approach.

Thus, it seems reasonable to expect that a probablistic analysis will be useful for resolving the high level radioactive waste question but not for nuclear accidents. The nuclear accident question cannot be addressed using absolute risk models. Relative risk models may, however, be useful here since these models provide insight into the causes of risk and where resources for reducing the risks may be best spent. For indecisive cases the amount of money to be spent on risk reduction and the acceptability of the risk question remain socio-political decisions that cannot be resolved on a technical basis.

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APPENDIX

In this report individual exposure to risk from radiation is based upon a "Cohort Analysis of Increased Risks to Death" (CAIRD) (1) where the increased risk of death from exposure to radiation of 100,000 people whose lifetime risk is profiled has been calculated using both relative and absolute risk models from the 1972 BEIR Report. In this model an exposure over an average lifetime of 70.75 years of a cohort or 100,000 for a constant exposure of .01 Sv (1.0 rem) is

4,400 early fatalities on a relative risk lifetime plot
670 early fatalities on an absolute risk lifetime plot

This respresents about 6×10^{-4} fatalities/year to an individual for a relative risk model and about 9×10^{-5} fatalities/year to an individual for an absolute risk model for a 10 man-Sv exposure per year. A mid-range value of 2×10^{-4} fatalities/year was selected as a

reasonable reference value.

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SOME NON-SCIENTIFIC INFLUENCES ON RADIATION PROTECTION STANDARDS AND PRACTICE

Lauriston S. Taylor*

I. INTRODUCTION

In the practical application of the principles of the achievement of protection against harmful radiation effects, our greatest obstacles today do not include a lack of knowledge about the biomedical effects of ionizing radiation. Today, we know about all we need to know to adequately protect ourselves from ionizing radiation.

Let me repeat that. Today, we know about all we need to know to adequately protect ourselves against ionizing radiation. Therefore, I find myself charged to ask: What is the problem and why is there one? I suspect that most of us here today know in a general sort of way where the problem lies and that basically it is not a scientific one. Rather, it is a philosophical problem with all the ramifications implied in the term. Or perhaps it may be a political problem, that is, one requiring prudence and sagacity in devising and pursuing measures adapted to promote the public welfare, or perhaps the problem may not be as much protecting ourselves against radiation as protecting us against ourselves. In any case, I sometimes think that today we are - in many areas - tormenting ourselves through our obsession with health (W.P. 8-28-79).

The control and management of any toxic agent, including radiation, requires a critical knowledge of the properties, characteristics, and biomedical effects of the agent. Furthermore, if control is to be absolute in the scientific sense, there must be either an establishable threshold below which there is no effect, or total elimination of the agent.

It is obvious that as far as we know today, neither of the above points can be met for ionizing radiation. Therefore, we must resort to other means, either political or philosophical or both, to arrive at some acceptable solution to the radiation control problem.

In developing my theme I shall mention, at least briefly, several non-scientific factors which may influence protection practices directly or indirectly and thus, in turn, influence the setting of our numerical protection standards. By and large, it is usually the needs of practice that dictate the setting of standards and the standards themselves must always represent some degree of compromise between a politic use of radiation and its elimination.

Little said today can be new or innovative but will be designed mainly as reminders to radiation protectionists of the things that they should be aware of and prepared to discuss in public forums. Some items need to be promoted, others rationalized, and still others, defended. Throughout, in relations with the public, we must avoid being patronizing and avoid any appearance of self-promotion.

Let us turn now to a brief discussion about the state of our current knowledge of the biomedical effects of ionizing radiation.

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II. BIOLOGICAL EFFECTS OF RADIATION

Collectively there exists a vast array of facts and general knowledge about ionizing radiation effects on animal and man. It cannot be disputed that the depth and extent of this knowledge are unmatched by any of the myriads of other toxic agents known to man. It is because of this knowledge, portions of which have become known to the public, that the public has come to expect sharp, clear, definitive, and undisputed answers to any questions involving radiation. This is an understandable, if somewhat irrational, position. However, it leads to the difficulty that when there may be some indication of a lack of knowledge by, or disagreement among, scientists, the public feels that somehow they have been let down or led astray by the scientific community. A good example of this is the current so-called "controversy" within the protection community centering around the effects of radiation delivered in low doses at low dose rates. Were it not for a few congressional committees, more interested in headlines than facts, aided and abetted by a willing press and a few publicity-seeking "scientists", it is likely that the question would drone on in the normal scientific meetings and committees at a proper pace, commensurate with its importance. It's not that it is unimportant, but its priority should be low compared with so many other insults that man faces.

Let us quickly review some of the facts related to ionizing radiation and the injuries that may be caused by it.

Ionizing radiation, delivered in sufficiently large amounts, can cause determinable effects or injuries to any biological system. However, for any particular effect observed, radiation would not necessarily have to have been the causative agent.

Radiation effects are generally proportional to dose when delivered acutely in moderate amounts, say 100 rads upwards, to the regions observed. Precise proportionality is difficult to establish.

There may be long latent periods between the time of exposure and the appearance of any effects that might reasonably be attributed to that exposure. Large doses above 500 rads can show effects within minutes or hours. Low doses below 50 rads may not show any effects for periods up to several tens of years, if ever. In general, the lower the dose and the rate at which it is delivered, the longer will be the period of latency before the effect manifests itself. We have here a generally inverse relationship between dose and latent period. The problem becomes especially critical in the low-dose region, say below 25 or 50 rads, delivered acutely, for which the latent period may be three or even four or more decades. During that long a period any individual would be subjected to hundreds of other insults, any number of which might produce the same effect as the radiation.

Man has always lived in a radiation environment which, except for a very small increment due to weapons testing, has been exponentially decreasing. His exposure today is less than half of the level existing during the Biblical period.

There is uncertainty about the existence of threshold effects for ionizing radiation. There are very few threshold effects, although there are clearly some.

For the purpose of numerical protection standards, it is assumed that unless the contrary is clearly identified, any radiation may cause an effect, if not an injury. The development of a clear-cut position on this question runs into complications. Here we encounter a further difficulty. If one is concerned about the degree of hazard in the region where effects cannot be found or identified, to what extent should an attempt be made to further "reduce the hazard" to some fraction of what could not be found in the first place? The question is "how large is half of something that cannot be measured?"

Dose effects are not cumulative; there is some process of repair or recovery or replacement of cells, both of a genetic and somatic nature.

Today we know enough about dose-effect relationships to state unequivocally that at least for low-LET radiations the relationships cannot be strictly linear over the whole dose range and that even for high doses they are probably not linear.

The difficulty is that since we do not know the precise relationship - and perhaps it doesn't make much difference anyway - it is assumed, as a matter of cautious procedure, that the dose-effect relationships are linear throughout the entire dose range. This assumption is constantly being subjected to hard scrutiny because, taken too literally, it may lead to unnecessary and unjustifiable restrictions on the use of ionizing radiations.

From the mere fact that radiation may cause some identifiable effect, it does not follow that the effects are necessarily detrimental.

For purposes of protection against ionizing radiation, we have to deal with effects; detrimental effects; risks; quantities observable or unobservable; and so on. We have now ventured outside of the scientific arena.

III. NON-SCIENTIFIC ASPECTS OF RADIATION PROTECTION

Over the past several decades, there has been a gradually developing consciousness of the inadequacy of scientific data or reasoning that alone will lead to the establishment of unequivocal numerical radiation protection standards. In the late 1940's it was clear to the NCRP, and probably to other bodies, that non-scientific factors would be involved in permissible dose standards. In 1957 I argued (cite):

"Radiation protection is not only a matter for science. It is a problem of philosophy, morality, and the utmost wisdom." At later times I have added "economics, politics, and public involvement" but actually they are all segments of an overall philosophical approach. I shall select a few of these for special comment.

PHILOSOPHY. Absence of a threshold leads immediately to the difficulty that there is no line of demarcation between the regions where scientific evidence does or does not exist; where evidence is not found, it is simply assumed to exist - a judgmental decision. It is obvious that, in reaching such a decision, a very non-scientific matter would play an important role. I refer to the pure emotions, not only of some scientists themselves but also of the lay persons who understand only bits and pieces of the problem and who realize

that they have to depend upon the scientist.

"What are the judgment elements entering into a standards setting process?" Basically, the arguments would center around the degree of risk that those who were setting the standards would be willing to inflict on others or, as a part of the public, to accept for themselves. How do you evaluate and describe quantitatively a situation, or more likely a combination of situations, each having its own set of values and its own descriptive units, and none having any unique relationship to ionizing radiation. For example, what is an effect and what is an injury and when is one not the other?

Comparisons of effects and injuries have been attempted in a variety of ways, particularly in the last decade or so, and it would seem that the only comparison unit which has come to our imagination has been monetary, such as the dollar or the mark. The arguments for choosing monetary value seem at times to be frivolous, but for obvious reasons nothing better seems to have turned up. The problem becomes even more involved if one tries to evaluate, let us say, pain or mental anguish which can be two obvious "effects" that might be caused by radiation, but still short of death.

Let us consider that risk, however we decide to describe it, is roughly proportional to radiation dose. Why are people willing to accept any risk at all? This argument applies to practically everything we do in life, with radiation being perhaps one of the smallest risks that we normally have to contend with. For all practical purposes, it is only in the use of medical procedures involving radiation that the risk, if any, is compensated by some benefit to the person at risk. It is also the area where one is most likely to find the situation that the risk of not carrying out some action (for example, an x-ray examination) is more hazardous than any conceivable risk to an individual from radiation.

Since risk questions do not really have discrete and scientific solutions, we are compelled to accept a philosophical approach. What is needed, on top of our scientific knowledge, which I contend is adequate at this point, is a large supply of basic wisdom and understanding. Question: Who has it? To whom do you look for it? How far can it alone suffice to complete the problem and develop a rational action policy?

The past supply of wisdom has come mostly from the scientists themselves, who consciously or unconsciously, recognizing the limits of their scientific knowledge, have made strong and important judgment actions regarding the amounts of radiation considered to be acceptable for radiation workers or the public or the patient. This has not been a bad thing because, after all, the scientists involved cover a wide range of disciplines, ways of living, nationality, ethnic background, and everything else that makes for an effective melting pot.

That this has been effective is evidenced by what I consider to be the fantastically fine radiation safety records that they have accomplished. No one has been identifiably injured by radiation while working within the first numerical standards set by the NCRP and the ICRP in 1934. The theories about people being injured have still not led to the demonstration of injury and, if considered as facts by some, must only be looked upon as figments of the imagination.

I do not argue for leaving the philosophical decision process in the hands of the scientist where, by default if nothing else, it has largely rested for the past 80 years. Nor do I argue for removing the process entirely from his hands; a combined scientific and non-scientific approach is indicated. A difficulty here is the current public attitude that if a person has worked in a field (e.g., radiation) he must be suspected of some kind of conflict of interest if he becomes involved in any related decision-making process. Actually, because of their basic training and their having to be imbued with a basic sense of objectivity, a good argument can be made that scientists, as such, are about as devoid of special interests as any group that may be found.

Aside from our experienced scientists, trained in radiation protection, where do we look further for our supply of wisdom? Personally, I feel strongly that we must turn to the much larger group of citizens generally, most of whom have to be regarded as well-meaning and sincere, but rarely well-informed about the radiation problems that they have to deal with.

To return to the basic philosophical question of setting standards for protection we can, with some over-simplification, reduce the problem to two choices. One choice is to more or less follow the present course of theorizing that we are dealing only with a single, linear, no-threshold, dose-effect relationship. However, in doing this we must take more specific steps in the future to keep in front of the public that (1) this is only a theory; (2) it is used only because we don't know the precise relationship; and (3) it is probably conservative, for most practical purposes.

The second choice would be to follow the practices used for many decades by the toxicologists. For permissible concentrations of some toxic substance they would set a level somewhat below that at which any effect could be found. A judicious blending of the two philosophies (and that is all they are) may well provide us with the most sensible solutions to the protection problem.

Before leaving the basic philosophical questions, there is one more item that must be considered and one which has personally worried me since the day that it was first introduced. That question relates to the system by which we have different kinds and classes of permissible exposures, or dose limits, for different classes of people. On technical grounds, I would not argue either as to the pragmatic need or to the acceptability of such a procedure. However, on philosophical grounds, we have problems. Technically, the differential is acceptable.

A logical question may be asked - why should our workers be subjected to higher radiation levels than the general population? The answer is along the lines given above, but it is rarely understood that way. From a philosophical point of view, a strong argument could be made for setting the same standards for radiation workers as well as for the public. On the other hand, for sound pragmatic reasons, and because thus far there is no evidence of injury even to radiation workers, this would certainly introduce a tangible and unacceptable economic cost for a gain that cannot be quantitated.

POLITICS. As already noted, to be politic means "to be prudent and sagacious in devising and pursuing measures adopted to promote public welfare.' (Webster). In the sense of radiation protection and in many other matters as well, political consideration really means the pragmatic combination of all of the elements bearing on a particular situation.

In the case of radiation protection, we might almost group all of the elements which I have listed above under politics and we might add some such as legal considerations, economic considerations, social considerations, etc. In this sense, the scientist generally, and the radiation protectionist particularly, must devote more thought and attention to constructive and objective politics, including direct approaches to people through the Congress, the press, and the telecommunications media. He must develop constructive and especially objective discussions and explanations for what is happening in the field of radiation.

From about 1946 to 1977, practically all federal matters in the United States relating to ionizing radiation were handled through the Joint Committee on Atomic Energy. The Joint Committee, with a stable membership from both the House and the Senate, was dedicated to developing facts and an understanding of atomic energy, rather than looking for newspaper headlines and votes. Among its many studies and reports, 21 major efforts dealt with radiation protection alone.

In its place there are some two dozen congressional committees, lacking in stability and without an overview power. Rarely does the chairman or staff of these committees have any knowledge in deoth of the broad subject of ionizing radiation. But equally distressing is their failure to keep each other informed as to their operations and intentions.

In spite of technical shortcomings in the political arena, both federal and state legislatures exert strong influences on the development of numerical radiation protection standards. Because of the likely influence on governmental committees by vocal but prejudiced witnesses or witnesses having some personal case to plead, we are today faced with the possibility of unreasonably restrictive limitations being placed on legitimate uses of ionizing radiation (cite: Mancuso, Bross).

THE MEDIA. One of the first political needs we must always recognize in dealing with groups of people is education. It is almost a cliché. We must be able to bring to bear and bring into perspective a wide variety of elements, many of them not seemingly interrelated and many expressed in terms of different quantities and different values. The prime agent of education (outside of formal schools) in these times is in the radio-television, newspapers, comic books, books generally, and books written by scientists. The order above is probably in that of declining readership. Of these, the "news media" clearly dominate, and here lies one of our most critical problems and the most fruitful area in which the radiation protectionist must assist in the education of the public. First, however, we have to persuade the media (and I use the term rather broadly now) that they have a national obligation to assist the country in educating its public about radiation matters.

Attacks on the news media for one reason or another are common as

is their own defense under the First Amendment. The First Amendment to our Constitution in the United States is an essential bulwark of freedom and is not paralleled by any other countries as far as I am aware. However, in my opinion, the First Amendment also carries with it an obligation on the part of the press to completely and properly report the news.

In the case of ionizing radiation which, of course, is the area with which I am most familiar, there are constant and continuous violations of this principle. The press will report and accent the news items and details which it thinks the public wants to hear about and what will help sell their papers...the latter, apparently, being their prime objective. The media must make money to stay alive and viable. They must sell their products and avoid wasting time on non-paying items. Thus, consciously or unconsciously a selective process begins. They say that they are supplying what the public wants. But as far as the public is concerned, they take that because it is the only thing they can get. However, as inept as our press may be in the United States in some respects, it is at least open and uncontrolled by the government - uncontrolled even to the point where it will "steal" and publish such government secrets as can be found.

The fact remains that we need greater responsibility on the part of the news media in the objective presentation of uneditorialized news. As far as radiation is concerned, our people must somehow persuade the press that it is irresponsible to subordinate radiation facts to stimulate sales. This will be a slow and painful process, but any gain is worth the effort.

MORALITY. What can we say in dealing with the problem of protecting people from possible adverse effects of ionizing radiation? It must be clear from the discussions above that a slightly higher radiation exposure permitted to a group of people under a given set of conditions may cause more injuries or even deaths to that group, and we are faced with deep moral considerations.

For example, the theoretical risk for radiation workers receiving the full MPD (5 rem/year) is ten times the risk for individuals in the public who receive the full Dose Limit (0.5 rem/year). Such an unqualified statement by itself is just part of a simple and immoral numbers game of which we have all too many in the broad safety field. A person dying from occupational radiation exposure is no different from one dying from non-occupational exposure.

An equally mischievous use of the numbers game is that of calculating the numbers of people who will die as a result of having been subject to routine diagnostic x-ray procedures. An example of such calculations is those made before a Congressional Committee in 1967 which were based on the literal application of the linear, non-threshold, dose-effect relationships, treated as a fact rather than a theory. By this procedure he calculates 30,000 deaths per year resulting from x-ray diagnosis. Of course, there has been no statistical or other verification of this calculation. Unfortunately, the technique has been picked up by others (cite: Gofman and Tamplin, Sternglass). These are deeply immoral uses of our scientific heritage.

Morality cannot be dictated by law or subjected to rule or

control. Morality is almost invariably an individual matter and the best and most sincere and thoughtful of people probably have widely variant moral viewpoints on a given question. On the other hand, we cannot sweep our moral obligations under the rug, nor can we settle them or bring them into agreement by vote or edict or law. Dealing with the moral aspects of radiation protection problems demands a kind of leadership and guidance and overall understanding that is not easily found.

LAWS AND REGULATIONS. There is no question but that the legislative and regulatory action in relation to radiation protection presents one of our most formidable two-edged swords. On the one hand, it aids in providing a needed degree of uniformity in radiation protection procedures. It provides a base upon which action for redress can be taken as might be needed under a variety of circumstances. On the other hand, the very power behind the legal system tends to stifle initiative and innovation in many areas, invites litigation and other legal actions, and greatly increases the cost of radiation, not only in industry but perhaps more importantly in medicine.

As far as radiation matters are concerned, we cannot live without a substantial legal system to protect both industry and the public, but there are times when we wonder if the atomic energy and radiation oriented industries can survive much longer within the complex of laws and regulations that have been spun, especially over the last ten or fifteen years.

The growth of bureaucratic involvement in radiation matters during the past two decades in the U.S. Government almost boggles the imagination. The programs may be divided into four categories. (1) Those involving radiation as a tool; (2) Research and development; (3) Regulation of the safety and uses of radiation and radioactive materials; and (4) Military applications. Without attempting to be complete as to numbers or details, let us make a listing of the more or less current government units embracing major interests in matters of ionizing radiation.

There are at least fourteen agencies of which six have regulatory responsibilities. Six have research and development responsibilities and three have advisory roles. In the legislative branch of the government, there may be some twenty-four House or Senate committees playing some role in radiation matters (the exact identification of these is not easy).

With as many agencies and congressional committees, each vying for its piece of the budgetary pie or prestige with its constituents, is it any wonder that there is competition, overlap, waste, and confusion in the radiation regulatory field? The wonder is that there is not more. An excellent example of recent vintage was the struggle for leadership in the development of radiation protection standards, primarily between the EPA and NRC, but with DOE and BRH anxiously watching from the wings. This led to the so-called "Libassi Study" of all radiation matters in the government. A remarkably fine report was produced, including organizational recommendations made to the President.

The Kemeney Report to the President on the Three Mile Island incident directs attention to "... a preoccupation with regulations" by the NRC. It goes on "... we are convinced that regulations alone

cannot assure safety. Indeed, once regulations become voluminous and complex as those regulations now in place, they can serve as a negative factor in nuclear safety." And later it states "... the nature of some of the regulations, ... may in some instances have served as a deterrent for utilities or their suppliers to take the initiative in proposing measures for improved safety."

The principle of keeping radiation levels as low as practicable was intended to stimulate protection initiative and innovation. The implications arising from the attempted use of the "least practicable" concept in regulations are of concern. The attempt to specify by regulation what is "least practicable" appears to be an unfortunate melding of concepts which vitiates the merits on each side. It undoes the assured uniformity of regulations because deviation on a case-by-case basis would appear essential, and yet removes from those subject to the regulation the responsibility for ascertaining what need be done to meet the "least practicable" criteria. Application of the ALAP principle should be primarily a political-management action presumably designed to promote the public welfare. But is it? Surely not, if it impedes independent initiative and innovation.

Another point of interest here has to do with Workman's Compensation in the nuclear or radiation industry. Let me hasten to add that I personally believe in Workman's Compensation as a legitimate charge against industry. In the U.S. we are having a gradual increase in the number of "radiation injury" compensation cases settled in favor of the worker, usually by compensation boards, but also by the various courts. Most of these cases center around individuals who have had very low doses in the course of their radiation work, but who have developed a malignancy. Of the cases that have come to my attention so far, most have incurred a lifetime exposure of not more than 5 or 10 rads acquired over a period of several years.

A malignancy may, as we know very well, be caused by radiation, although on the basis of our knowledge of dose-effect relationship, the likelihood would be extremely remote at the levels mentioned. On occasion, I have said sometimes seriously and sometimes facetiously that it would be less expensive and perhaps more humane if it were decided that any time a person who had worked at any time during his life with radiation and subsequently developed a cancer which might be ascribed to radiation, at any level, he be given free treatment, together with what would be a normal compensation for the family. With all of our national health plans in being or in prospect, this would scarcely add a drop in the bucket to overall health care in this country.

ECONOMICS. The possible influence of economics on the standards for radiation protection must be so obvious that it scarcely needs mentioning. There is constant pressure to lower protection standards by some radiation protectionists as well as "consumer advocates' and generally concerned members of the public. Too often their arguments are based mainly on the theoretical estimates of effects that have never been observed and in turn on calculations of the theoretical deaths or cancers due to specific sources of radiation exposure. Any degree of protection can be achieved - at a cost. The problem is to evaluate the risk, the cost of reducing it, and the gains to be achieved. The process is frequently referred to as balancing the

risk vs. the benefit. The principle is so simple as to be disarming. The difficulty lies in the quantitative elusiveness of both risks and benefits.

In a medical installation it sounds like a simple action to reduce the leakage radiation from a therapy tube by a factor of 2, but before the chain of events is completed a new building structure may be called for.

In the case of power reactors the economics would be much more difficult to evaluate. Where the different radiation levels from a particular reactor are known, they are likely already to be too low to evaluate except in terms of assumed risk, by such theorizing as has already been discussed. So this is a case of reducing by some factor something that you did not know in the first place. If someone were today to decide on a reasonable de minimis level for radiation exposure, it would probably be found that most of our radiation installations are already well below it.

Risk comparisons can be made between effects at high doses where we do have information and those at low doses where we have no information. The numerical values of these risks are possibly basically without meaning but are at least internally consistent and so evaluation of the costs of increasing or decreasing protection in a given installation can be made in terms of the arbitrary risk numbers.

A different kind of economic problem arises in connection with regulatory operations. In the process of defending, say, a budget request, a federal agency will work up an elaborate cost-benefit tree designed to show that by a certain addition to their facilities or program, they can save so much radiation exposure per person averaged over the public. This, in turn, will reduce the risk to the public and hence reduce the number of radiation effects. But the bottom line is to show that these imaginary effects will reduce hospitalization, will reduce Medicare, so that there is an overall saving to the taxpayer. I've seen at least one example where an agency showed a saving for a given program and yet someone on the outside used precisely the same input information to show that it would make a much more costly program overall.

IV. POSSIBLE CORRECTIVE ACTIONS NEEDED

EDUCATION. In the development of an overall understanding and acceptance of radiation as one of man's most valuable tools, we need two things: (1) better communication within and between scientific and technical groups on the one hand, and the general public on the other; and (2) much broader education of and dissemination of information to the public. Perhaps a third item should be added. These communication and educational projects should be carried out basically by non-governmental organizations, aided and assisted, however, by some limited government support.

As far as the public is concerned, there has been a mysticism about radiation and for reasons, some valid and some not, the public has come to recognize that it has on occasion been told untruths or part truths about some radiation matters, primarily by government agencies. I believe a good case can be made for some of the misinformation that has been supplied to the public. But in the

matter of communication, the radiation protectionist profession must play a stronger role, together with coordinated and concerted effort by other national and international organizations having long familiarity and responsibilities with problems of radiation protection and measurements.

It is my belief that much of the blame for the public's fears and apprehensions with respect to radiation matters are due to our media - newspapers, magazines, radio, and television. No particular one is better or worse than the other. The difficulty here is that of the general public. But where must people go for their information? Primarily, to the press. Yet, in my opinion, the press is failing in its responsibility under its Constitutional freedoms. Let us make one simple clear statement before continuing. The media are in existence primarily to make a profit under our free enterprise system. At the same time, the media enjoy protection under at least our First Amendment in the gathering and dissemination of information, not to mention their editorial treatment, which is not news but opinion. We must not take any action which could conceivably destroy this basic freedom.

It should be required that in reporting a news event, the media report it all as it occurs, not only the segment that is in line with their publication policy. Mistakes in observation will be made; these are excusable. Mistakes, however, of selecting only certain aspects of a news item and suppressing others is a general, but totally unacceptable practice. The press rationalizes the situation by saying that they supply the public with what the public wants. As I have noted before, this is a very specious argument and it is self-defeating. The alternatives would be government controlled and supported media which totally reflect the policies of the government. That would be intolerable!

There is yet another criticism that must be directed to the media, namely, their constant use of a small number of individuals who are clearly out of step with the radiation protection community. In the U.S. alone there are some 3500 health physicists and 1800 radiological physicists. The National Council on Radiation Protection and Measurements has, over the years, utilized over 550 scientists covering every professional field having any conceivable bearing on radiation protection standards. Yet the media will, for some newly breaking news story, seek out some of a half dozen individuals who are willing to make willfully deceptive statements regarding radiation. Collectively, they account for more news lines than the hundreds of reliable professionals accepted by their peers. (I refer to them as the U.S. Six). If the media want to improve their professional image, they must studiously avoid the sensationalism produced by the U.S. Six, but which they presently believe sells their wares.

The U.S. Six has a strange mixture of talents. One or two still have some degree of professional reputation left and they will carefully hedge their statements as self-protection among their colleagues. At the same time, however, their statements are interlaced with enough of the usual fear catchwords which are often the only part captured by the reporter inexperienced in the nuances and matters of radiation protection.

SCARE BOOKS AND ARTICLES. Of a collection of "popular' books published over the last decade or so dealing with radiation matters there is not a single one which is not riddled with half-truths, untruths, and evidence of basic lack of knowledge of the subject. All carry a high level of sensationalism and an eye-catching title or subtitles. These books are usually written by individuals who have no basic background knowledge of nuclear energy or radiation.

Unfortunately, many people sincerely concerned about many of our present-day problems in the nuclear field read these books and believe that they read the facts. This happens simply because they do not know enough about the subject to recognize much or any of the guileful and misleading statements. The books that I have been speaking about have largely been written with a profit motive but there are occasional others, obviously written to support their opposition to nuclear power, government research support, and so on. In spite of being written by one-time scientists, their books carry all of the sensationalism of the media.

Before leaving this part of my discussion, I invite attention to yet another insidious practice designed to keep the public alarmed about radiation matters. This is the constant linkage made between the atomic bomb and any discussion about radiation, including medical and industrial applications. For example, in a television documentary presentation on ionizing radiation or a news story about some small accident in a nuclear installation, or a large or small accident in a nuclear power plant, practically the first thing that is presented to the reader or the viewer is a story about a bomb, a picture of a bomb exploding, reference to radiation through the term "fall-out", and so on ad nauseam. These are the catchwords.

Understandable, but equally preposterous, is why an article about a nuclear reactor accident, such as the TMI case, should always be preceded by some reference to the bomb or an explosion and fallout, the standard warfare terms, when the public has been informed by hundreds of good and reliable sources that a power reactor simply cannot explode like a bomb.

CREDIBILITY OF SCIENTISTS. In some respects, the scientist stands apart from most other individuals who can be placed in some definable pattern plan. In the first place, the average scientist starting in research is not very likely to have financial gain as his prime objective. He does have to earn enough to live in reasonable comfort, have some freedom from financial worries, and to have a family life. For the true researcher, the man at the bench, the highest real reward is in terms of a professional reputation, acceptance among his colleagues, his reputed objectivity, and his reputed intellectual honesty. If he fails in any one of these elements, he is destroyed.

The implication is implausible that nuclear or other researchers are susceptible to ready suborning by the people or the organizations who support them and make their work possible. Great play has been given to the disagreement between some scientists either as individuals or as part of some scientific groups. This is pictured by the media as chaotic, self-serving, cover-up, or such. Actually, it is the normal, proper, and healthy intercourse of scientists.

I plead that we cease the seemingly endless procession of studies, congressional committees, and hearings on the problem of "low level ionizing radiation", just to choose one of the problems that plague us today. About this, we know what we know and we know what we do not know; there is reasonable and rational agreement as to the degree of disagreement. So where does this leave us? Either we forget the whole "problem" or we theorize or postulate a dose-effect relationship.

However, this is what has led us to our present dilemma because these technical concepts have been grasped by the press, by the congress, by some government agencies, and hence by the public as established facts, rather than as the scientific ruminations, which they are.

Somehow, we as radiation protectionists must develop an unassailable counterforce against such misguided actions as outlined above. This counterforce should act with such strength and integrity and persistence as to compel public attention and respect.

5th INTERNATIONAL CONGRESS OF THE INTERNATIONAL RADIATION PROTECTION ASSOCIATION

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RADIATION PROTECTION FOR INDUSTRIAL RADIOGRAPHY IN THE AEROSPACE INDUSTRY

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"Industrial Radiography" has been a dirty word to Health Physicists all over the world. This is true because the Industrial Radiographers are involved in more radiation accidents, receive more overexposures, and are cited for more serious items of non-compliance by governmental agencies than any other group in the nuclear field.

The Boeing Company employs eighty Radiographers, operates ninety radiation sources, and takes ten-thousand radiographic exposures per month. Our Radiographers average less than 100 milliRem Whole Body exposure per year. They have never been cited for a governmental non-compliance, nor have had any radiation accidents. X-ray units, Accelerators, and Radioactive Materials are used in our Radiographic Inspection Program on aircraft, missiles, and seacraft.

The major components of our Radiation Protection Program are: Hazard Analysis, Facility and Equipment Design, Program Administration, and Evaluation of the Protection System.

HAZARD ANALYSIS

A Hazard Analysis starts when a Quality Control Organization within the Company wants to use a source of radiation to perform radiography. Depending upon the thickness and density of material to be radiographed, the Operating Group will determine the type of radiation source required.

If a Radioisotope such as Cobalt 60, Iridium 192, Ytterbium 169, etc., is to be used, the strength of the source in Curies is determined. With this information, the External hazard is analyzed from the standpoint of energy and amount. The Internal hazard must be assessed from the method of encapsulation and the type of radioactive material.

When an X-Ray Unit is involved, we are given the tyoe of tube they are going to use, such as a 360 degree emission or side emission tube, and the maximum kiloVolts (KV) and milliAmps (MA) capabilities of the unit. Similar information is also provided on Particle Accelerator operations.

Once the characteristics of the source are determined, we must agree upon where and how the source is to be used. If the source is going to be used in a shielded room near an unpopulated area, the shielding and protection requirements will be minimal. A stationary source in a populated area will require more shielding and a complete protection program. When a source is used in a shielded area, the main beam wall use factors, distance to people, and occupation factors are necessary information to determine the shielding requirements.

However, if the source is to be a portable unit and used mainly for field radiographic inspections, we then need to know

where and how often these unshielded sources will be used. The type of radiographs taken will dictate the maximum KV, MA, and "ON" time of a machine, or required "exposure" times for sources of radioactive materials. These parameters are considered when evaluating the size of a hazard or restricted areas.

The interaction of all employees and a source is analyzed at great length. This analysis covers both the Radiographer using the source, and the Employees working in and around the source location. The Radiographer and his Helpers are classified as "Occupationally Exposed Workers". The other Employees are classified as "Non-Occupationally Exposed Individuals", and can only receive one-tenth of the radiation exposure of an Occupationally Exposed Worker.

Operations conducted by the Radiographer using the source are reviewed to accurately establish: (1) The amount of time the Radiographer and other employees are exposed; (2) The intensity of the exposures; (3) The areas of their bodies that are exposed. The possible exposure dose will scope the Radiation Protection Program.

Having assessed the hazard associated with normal radiographic operations, we then consider the consequences if the equipment fails, or the Radiographer errs. If a human error or failure of the equipment can cause high External or Internal exposure, then every possible protective device is used to eliminate the hazard.

FACILITY AND EQUIPMENT DESIGN

When the analysis is complete and the degree of hazard is known for the operation, necessary facilities and equipment can be designed to contain the hazard. At Boeing, we try to design out all the hazards associated with radiographic inspections.

Our industrial radiographic operations are classified in three general categories: (1) Shielded Room Operations; (2) Portable Self-Contained Protection Systems for use outside of a shielded room; and (3) Open Field Radiographic Inspections.

In a Shielded Room design, we place enough shielding in the walls, floors, and ceilings to keep the radiation exposure outside the shielded area to less than 10 mR per week. We shield the room according to the maximum KV and MA, work load, use, and occupational factors associated with the area. We reduce the cost of shielding by limiting the walls at which the Primary Beam may be directed.

All entrances to the radiation area within the room are provided with Interlocks that shut down the x-ray production when opened. The Interlocks are all of a fail-safe nature, and must be re-activated by both closing the door and re-setting manually. The interior of the exposure room is posted with a sign, "Do Not Occupy Area When Door Is Closed". A Flashing Warning Light is placed outside of each entrance to the room, and also inside the room. The Warning Light on the outside illuminates a sign which says, "Radiation On". The Warning Light inside (preferably a Rotating Beacon) is activated twenty seconds prior to the production of radiation, and remains activated during irradiation.

An Audible Warning Device is also used inside each shielded area. This audible signal is activated twenty seconds prior to the production of radiation. "Caution——High Radiation Area" Signs are

placed at the entrance to each shielded room to warn personnel of the hazard within the room. Emergency Power Cutoff Switches (Scram Buttons) are placed in each exposure room, and are easily identifiable because of the large red square painted on the wall behind it. A Sign is located above the switch which says, "If Alarm Sounds, Push Switch and Evacuate Room Immediately".

Also placed in each room is a Continuous Radiation Monitoring Device, which gives off an audible signal when radiation is present. These are placed in the room as a secondary protection system, to ensure that the Radiographer knows if the x-ray beam is on, or a radioactive source is exposed within the enclosure. These continuous warning devices do not replace the Portable Survey Instruments which are used by the Radiographer upon entrance to the room.

Before any of the shielded rooms are put into service, a complete radiation survey is performed of all occupied areas adjacent to the walls, ceilings, and floors of the room. Each safety device is checked out to make sure that it is operational and fulfills its requirements. If they all check out, the room and radiation source are certified by the Radiation Health Protection Organization as safe to operate. This certification will be discussed later, in the "ADMINISTRATIVE" section.

In-place shielding devices are also designed for special applications. This may be a small lead box used in radiographing small parts, or a shielding fixture which allows the radiographing of an in-place hydraulic tube on an aircraft. The radiation exposure levels do not exceed 2.0 mRem/hr at 30 cm from the shields. Where possible, these shields are fitted with safety equipment such as Interlocks, Flashing Lights, etc. Signs, instructing Non-Occupationally Exposed Individuals to stay approximately one meter from these devices, are posted near the equipment. The Interlocks ensure that the in-place shielding device is closed or fits tightly to the working surface, to eliminate leakage. The Flashing Light is activated when the radiation source is turned on or exposed. Both X-ray units Radioactive Material Sources are utilized in the In-Place Shielding Devices. By use of this type of equipment, radiographs can be taken on an aircraft in the assembly line, without having to remove any of the nearby workers; a great cost savings for the Company.

For Field Radiographic Inspections, posted barriers such as ropes, fences, and barricades are used to designate the hazard area. Distance from the source of radiation is used as our primary protection method. However, portable lead shields are also used during field radiographies. Shields are generally 1.2 meters by 2.4 meters Sheets of 9.5 millimeters and 12.7 millimeters Lead, to accommodate the high energy sources. They both may be used for Primary and Secondary radiation, depending upon the energy of the source. Use of shields cuts down the size of the hazard area, and allows more employees to continue working nearby the radiographic operation.

In addition to the portable shields, we also use Lead Cones on the Gamma and X-Ray Beams, to limit the field of radiation. The Cones are fitted to the x-ray tube or on the exposure tube of the radioactive material device. Both Audible and Visual Warning Alarms are used at the radiographic site. Flashing Warning Lights are activated twenty seconds before radiation is present, and all the

time during the exposures. The Audible Alarm is activated by the radiation source. The barriers, signs, and warning devices are put in place by a Radiation Monitor from the Radiation Health Protection Organization. Often, the Radiographer aids in this process; the two work as a team. During actual radiographic exposures, both members of the team patrol the hazard area to ensure continuous evacuation of Non-Occupational personnel.

PROGRAM ADMINISTRATION

After the facilities and equipment have been designed and obtained, the Administrative functions are called upon to provide the rest of the Radiation Protection Program. The general policy of The Boeing Company is to make each Operational Supervisor responsible for the health and safety of his employees. The Radiation Health Protection Organization serves them in an advisory capacity. If the Radiation Health Protection Organization did operate as an enforcement group, they would have to be present during all industrial radiographic inspections, even the ones conducted in a shielded room. This is impractical, and very costly! Also, the employee who works in a hazardous area must be willing to accept some of the responsibility for his own protection. We have also determined that the workers and the supervision of Operating Groups tend to accept advice much more willingly than to comply with commands or ultimatums.

There is one exception to this policy. If an operation is conducted in a manner that could cause a serious radiation hazard, the Radiation Health Protection Organization has the power to stop the operation. Because of the high risk involved in a field radiographic inspection, a Radiation Monitor from the Radiation Health Protection Organization is always present to aid in the evaluation of the hazard.

Our command media plays a very important part in the administrative portion of the Radiation Protection Program. The Radiation Health Protection Organization reviews or writes all Administrative, Operating, and Emergency Procedures which involve the use of radiation for industrial radiography.

The Administrative Procedures establish the Radiation Protection Program, and give general rules and requirements.

The Operating Procedures are written for specific industrial radiographic operations, and tell the Radiographer in a step-by-step method how he should perform his work. Operating Procedures also assign protection responsibilities to the Radiographic Group as well as the Radiation Health Protection Organization.

Emergency Procedures are written to cover every type of industrial radiography operation, and document the possible hazards with the necessary controls. Responsibilities are assigned to the Operating Group as well as other Emergency Organizations, such as Fire, Plant Security, Maintenance, Occupational Medicine, and the Radiation Health Protection Organization.

The backbone of the whole Radiation Protection Program is the education and training provided for the Radiographers, Radiation Health Protection personnel, Management, and Emergency Groups.

At Boeing, every Radiographer and Radiation Health Protection

employee is required to take a 24-hour Radiation Protection Course prior to working in a radiation area. These Courses are taught by the Radiation Health Protection Organization, and help to cement relations between the two groups. Subjects such as Basic Radiation Physics, Biological Effects of Radiation, Methods of Controlling Radiation Hazards, Operating Procedures, Emergency Procedures, Legal Requirements, and Recent Radiation Accidents are covered in the Radiation Protection Course. A written examination is given at the end of the Course, and the students are required to pass with at least a 75 percent score. Refresher Courses are required for all Radiographers and their Helpers. On-the-job training is also an on-going part of the training program.

An Equipment Certification Program plays an important part in the Boeing success of reducing its employee exposures. Each source of radiation is certified to operate only under specific conditions. These certifications are written by the Radiation Health Protection Organization, and the contents approved by the Operating Groups.

The Certification Form contains the conditions under which an X-ray, Gamma source, or Accelerator may operate. The write-up includes the name of the piece of equipment, the identification number, the location where the equipment may be used or stored, the organization who owns the equipment, and personnel who are approved to operate the equipment. It specifies the maximum operating power range in kiloVolts (KV) and milliAmps (MA), or Curies for Radioactive Material, and the maximum time the equipment may be operated in a given week. It tells the Operator at which walls he can point the radiation source, what power levels he can use, and the minimum distance the source must be from the wall whenever it is activated. These conditions are spelled out for the four walls, the ceiling, the floor, and any door within the enclosure.

A list of operating limitations is also included in the Certification. Such things as "This equipment can only be operated by a trained and authorized Radiographer", "All Safety Devices must be in working order", "A Safety Device Checklist shall be completed prior to each day's operation", "Any operation not complying with the limitations on the certification must be monitored by the Radiation Health Protection Organization", "Radiation Health Protection personnel must be contacted prior to any field radiographic inspection using this radiation device", and "A copy of all Administrative and Operating Procedures must be posted along with this certification" are some of the limitations placed on operations.

All operations are certified annually. Quarterly inspections ensure a continued safe operation. The certification form is reviewed by each Operator. When the Radiographers have read and understand the limitations placed on the operation of a particular radiation source, they sign a Form confirming they have reviewed and understand the document.

EVALUATION OF THE PROTECTION SYSTEM

After the Administrative Program has been set up and the Industrial Radiography Protection Program is underway, the Radiation Protection Program is evaluated through Physical Examinations, Personnel Dosimetry, and Radiation Area Surveys.

Normally, Physical Examinations are given to employees prior to working in a restricted area, on an annual basis, and when the employee terminates his work with radiation. If real or potential overexposures are suspected, special physical examinations are administered.

Personnel Dosimetry is worn by all personnel working in radiation areas. These include Film Badges, Pocket Dosimeters, and Thermoluminescent Dosimeters (TLD's) for special monitoring programs. The Radiographers are required to wear the Film Badge and Pocket Dosimeter at all times. TLD's are used in special programs to determine if certain areas of the body are receiving more radiation than others. TLD's are placed on the head, hands, legs, and other body surfaces to determine localized doses.

If there is any possibility of internal deposition of radioactive material, a Bio-Assay is done on each Radiographer involved. The Film Badge Program is on a monthly basis, with Pocket Dosimeters being issued and read on a daily basis. Film Badges are purchased from, and processed by, a disinterested third party. This ensures that The Boeing Company will not under-read a film badge in case of an overexposure. Each film badge that reads over 100 mRem in a period of a month is investigated, and a letter of explanation is required from the person receiving the exposure.

Personnel working under field conditions are issued Personnel Audible Warning Devices, which alarm when in a radiation field. Of course, each field radiography (as explained earlier) is monitored by a Representative of the Radiation Health Protection Organization.

Periodic (quarterly) Radiation Surveys are also conducted on shielded installations. These surveys include monitoring of the shielding, safety devices, and radiation source. Reports of all surveys are maintained in our records for review by governmental agencies. The results of the Boeing program have been excellent. Only one individual in twenty years of operation was exposed to 1 Rem in a year. The average exposures for one year are below 100 mRem.

Boeing's Industrial Radiographic Program has never been cited for items of non-compliance by any governmental agency. This includes the United States Nuclear Regulatory Commission and the Washington State Radiation Control Agency. The potential for high acute or even low chronic exposure to radiation has essentially been eliminated at The Boeing Company, and the term "Industrial Radiography" is not looked upon as a dirty word by any of the Health Physicists or Radiation Health Protection people associated with this Program.

SIMULATION AND COMPUTATION IN HEALTH PHYSICS TRAINING

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The Health Physicist is an applied Scientist and his professional skills must extend over many subjects. Probably the most difficult skills to acquire are those of numerical analysis, particularly the application of this analysis to time dependent phenomena. In the ten years of experience of graduate level courses in Radiological Protection, the Royal Naval College has made extensive application of digital and analogue computers. Computer programmes have been developed for training in radiation shield design, optimisation methods, the study of the environmental impact of a reactor accident and the prediction of detector responses in various radiation fields. Computers play an important supporting role in the study of the design process and in the provision of information input to reactor accident exercises. Time dependent phenomena are not always easily assimilated so analogue computers are used on reduced or accelerated time scales; in some cases the calculation can be stepped as required by the student. Whilst these alterations in time scale are useful for initial understanding it is essential to relate them to real time, and thus the College also uses a research reactor for the final phases of training. This paper will be illustrated by a number of applications of computers in Health Physics training and these will be restricted to the digital computer applications.

Computers have a more important role to play in education than the simple direct application in the solving of set course work problems. They can be considered (1) in two principle modes "behavioural control" and "discovery learning". In the first the teacher controls the learning environment by selecting and arranging the computer material to illicit the required response from the learner. This technique derives from the original concept of programmed learning but has met with limited success. In the discovery learning mode, control is exercised by the learner who through computer based procedures can build up a base of knowledge and extend his knowledge beyond the basic given information. These procedures are associated with computer modelling and simulation but their effectiveness depends very strongly upon the quality of supporting teaching. A typical application will present a computer based model of a system or process and offer the student interactive access to the principle parameters with the opportunity to compute and display the effects of variation. As an extension of this the student may be involved in the process of building and validating the model before using it for the study of a system response or to achieve an objective in the design of a system.

An example of this is the RNC programme SPS (Shield Parametric study) which is one of the first programmes developed in this field. The programme models a pipe containing radioactive material and

simulates this as a line source which may be located anywhere in a plane passing normally through the slab of shielding material and through the dose point. The programme uses a linear attenuation coefficient and a linear build up factor and the source is simulated by point isotropic elements which can be varied in physical size. The student inserts initialising parameters or may use a set of test data and can then select any parameter which may be varied by a selected positive or negative factor. At each increment the dose rate is calculated and printed out and subsequently a graph may be plotted for each parameter over the selected range of change. Students may use the programme to synthesise more complicated sources.

An elementary shielding programme such as SPS will overlook important changes in response function due to radiation quality. Programmes which model the response of gamma detectors to photons emitted from internally distributed radioactive material (2) have been adapted. One programme simulates photon transport using a Monte Carlo technique to determine the energy loss spectrum in cylindrical geometry. Data sets are made available for cross sections applicable to sodium iodide, caesium iodide and germanium detectors. Application of this created interest in the dynamics of internal radiation dose and an ICRP lung model (3) was used in a student project applying the digital computer for ease of program writing. Preliminary work using the analogue computer permitted non-linear rates of activity input to be represented. The models developed in these projects have then been combined in a simulation of noble gas releases using a simplified hemispherical model for a building shield (4). In this programme the external gamma exposure can be calculated from radiation sources both outside and inside the building. This model has been tested against Argon 41 sources created by the research reactor.

Role playing is a valuable vehicle for education of the individual and the team and introduces the real time world through the planning of health physics control or the management of accidents. On the whole it is not possible to play the game totally in real time, nor is it possible to assign complete roles to individuals or to allow them to interact freely. Best results are obtained when students work as syndicates of 4 or 5 to which information and questions are addressed with solutions being offered by the syndicate as a group. At the present state of development the computer is used to provide realistic information on dose rates and its principal value is that of a labour saving aid to the Directing Staff of the Exercise. Further programmes are under development to provide feedback of data on collective dose under working conditions determined by the syndicate, having regard to the ALARA principle. It is here that interactive graphics with the ability for 3-dimensional simulation of working areas has much to offer.

To be effective, all these applications of the computer require as much realism as possible. For example, although the playing out of an accident need not be in real time, it is essential that the radiation dose conditions postulated are seen to be as realistic as possible. To this end, simulation of the accident

sequence, and its subsequent development can be carried out using available codes such as TRICY or WEERIE (5, 6). It is now recognised however that less comprehensive codes are adequate, and have the advantage that they can be made interactive and hence can be used not only for exercise planning but also by students during, or independant of, the exercise. Such a code is under construction at RMC and in its present state of development it uses a point source of fission products, allows for building attenuation when calculating the gamma shine doses, and uses a standard gaussian model for plume dispersion with building entrainment and dry deposition. From the existing model it has been possible to compute the time integrated dose as a function of distance down wind and to indicate its sensitivity to release height, and Pasquill weather conditions with the major aim of convincing the students that a reactor accident will not necessarily follow a predetermined course. Future models used in exercise simulations will insert a degree of randomness in weather conditions, release mechanisms and in simulated protective response.

Digital computers are ideal for the application of optimisation methods and a number of these are briefly described in Reference 7. The aim of the training process is to ensure that the students become familiar with the fundamental design criteria; the student should gain an overall appreciation of the design process and above all understand the inter-relationship of design methods.

The dose reduction philosophy of ICRP 26 has demanded increased attention to dose reduction techniques. The RMC programme developed to demonstrate dose reduction simulates a containment structure represented by a 3-dimensional grid within which various system components may be represented by a combination of point, line and plane sources. Simple line of sight shielding methods are used to estimate the exposure at any location within the grid and it is possible to extract the relative contribution of to the dose from each system component. It is therefore possible to assign a collective dose to a given operational procedure taking place within the containment, for example, a maintenance operation. Reduction of this collective dose may then be pursued through changes in operational procedures to take account of dose gradients for example (7) or by introducing protective facilities such as additional shielding or reduction in specific sources.

The work described in this paper is a response to the demand for optimisation and cost benefit analysis in radiation protection. These computer based methods can be implemented using a variety of computer systems not only main frame computers but also dedicated mini-computers and it is possible that microprocessor based terminals will have an important application. The development of systems will increase the availability of computer based support for the operational health physicist and thus the computer must be an essential feature of his education.

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RETENTION OF RADIOACTIVE SUBSTANCES, AEROSOLS AND POISON GAS BY SANDFILTER

L. Balcarczyk, V. Hess, H. Sorantin

INTRODUCTION

In the last wars sheltering rooms have proved their effectiveness if they are properly equipped.
One of the most important components is the filtering system. It consists generally of a non-burnable prefilter filled with gravel, sand or other loose material which acts as heat and shock protector followed by an activated charcoal filter to absorb volatile poisons.
We tried to develop a filter system with only one non-inflamable medium with sufficient capability for the retention of harmful gases and aerosols.

EXPERIMENT

Testing of filtering materials

Various kinds of inorganic materials as gravel, sand, dolomite, slags etc. were tested. For the use in filters, sands of basalt or dunite proofed to be the best, but others can be taken if they fulfill the following conditions:
-large inner surface (>3 m²/ℓ)
-airflow resistance (3,5 mbar)
-particle diameter 0,2 - 0,025 cm.

Influence of humidity on sandfilters

It is a well known fact that in the presence of humidity the capacity decreases very quickly.
In a series of experiments air, saturated with water, was sucked through a sandfilter. The humidity in the bed was measured by hygrometers which had been posted in different depths.
No significant change during 60 hours could be observed on the hygrometer in the sheltering room, because - as could be noticed from the other hygrometers - an equilibrium between the humidity of the air and the water content of the sand was formed.

Behaviour of sandfilters against hot air

In case of accidents higher temperatures may occur in the surroundings of the shelter. To simulate this conditions an airstream was heated to 120°C and conducted through the sandfilter, in which several thermometers had been inserted. After six hours no remarkable temperature rise was observed till 75 cm distance from the surface of the filter. Later the temperature increased in the following 18 hours to 60°C and remained after 48 hours at 70°C.

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Behaviour of filtersands against aerosols

For checking on laboratory scale a definite volume of air was led over an emanating sample of ^{228}Th. After a decay time of 10 min for Thoron, the air stream was devided. The first part was directly sucked through a micropore filter while the second had to pass a sandfilter before reaching also a micropore filter. The activity gathered on the two micropore filters showed that the aerosols were absorbed completely by the sand.
In a similar experiment paraffin aerosols with a medium diameter of 0,3 µm were produced with a special generator, mixed with air and a steady stream of 1,5 l/min was sent through the sand.
The measuring apparatus consisted of a plastic tube of 6 cm diameter, which was filled to 1 m height with the sand. On one side of the tube, in distances of 10 cm, holes were drilled and connected with an exhausting pipe and a case holding a micropore filter. By the two way valve the airstream loaded with the aerosols was sucked through the sand or the aerosol could be extracted from different height out of the sandfilter.
While in 28 cm depht 1,4 mg aerosol could be sucked off in 10 min, a layer of 28 cm was sufficient to with hold the aerosols. Totally, the filter column was loaded with 9 g paraffin during 100 hours.
For comparison, a similar column was filled with activated charcoal of 1,5 mm particle diameter till a height of 38 cm. The break through was observed after 25 hours, in which 2,25 g paraffin had been precipitated. The sand showed therefore under the mentioned conditions a four times greater capacity.

Behaviour of filtersands against methyliodide

After reactor incidents ^{131}I is one of the most volatile fission products. The iodine can be present in form of element, acids or organic compounds. Form the latter methyliodide can only be retained by charcoal filter impregnated with alcali iodides or amines and at higher temperatures, only molecular sieves containing Ag or Cu are successful.
We tested the sandfilters with ^{131}I labelled methyliodide and could only register a very poor retention about 0,1 % of the starting activity.
Better results about 1 % were obtained when the loaded airstream was mixed with ozon coming from an electrical generator.

Retention of tear and poison gases

Since chemical warfare is not prohibited the presence of poison gas and aerosols must be taken into account.
In the first series of tests we used bromoacetone, which effects the eyes even in a concentration of 1 mg/m³ very

severely.
To simulate conditions in the sheltering room an airstream of 1,5 l/min was charged with the teargas and the carried amount determined by adsorption on charcoal. In the following test the stream was conducted through the sandfilter and the break through registered by coloration of filterpaper impregnated with 2,4 dinitrophenylhydrazine. An ordinary sandfilter with a volume of 1 m^3 can therefore retain 1,67 g of bromoacetone.
From the poison gases phosgen was chosen, because it is one of the most toxic. The experiments were conducted in similar way in gloveboxes. The dried airstream was mixed with phosgen coming from a steel bottle. On the normal sandfilter 166 g phosgen were retained. For comparison, specification for charcoal filters ask for a retention of 155 g.

SUMMARY

Selected filtersand showed a climatizing effect for humidity and hot air. By their greater outer surface and moisture content they display a very good retention of aerosols and are in this case even effectiver than activated charcoal filters.
Methyliodide passes through sand and untreated charcoal filters and can only be withheld by sands to a small extent after ozonisation.
Our sandfilters showed because of their moisture content very good retentions of bromoacetone and were even superior to charcoal in the presence of phosgen.

A STUDY OF EFFLUENT CONTROL TECHNOLOGIES EMPLOYED BY RADIOPHARMACEUTICAL USERS AND SUPPLIERS*

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INTRODUCTION

The medical use of radionuclides has resulted in the evolution of a large radiopharmaceutical industry. This industry is made up of suppliers that produce or package radionuclides and users primarily hospitals and physicians. The quantities of radiopharmaceuticals produced for in-vivo diagnostic and therapy procedures has been estimated to be growing at the rate of 16% per year,[1] based on 1978 sales figures. Nuclear medicine facilities are experiencing an average annual growth rate of 5% per year.

The principle radionuclides produced and used for nuclear medicine are ^{131}I, ^{133}Xe and ^{99m}Tc. Of particular concern is that amount of these radionuclides which might become airborne and escape into the environment during the process of manufacture or during aliquotting or administration by hospital personnel. Therefore, the radiopharmaceutical industry facilities have been reviewed to identify those parameters and mechanisms that could lead to the airborne release of radioactive isotopes and assess the control technology employed.

HOSPITAL USAGE

The radionuclides used in nuclear medicine procedures are grouped according to use as follows:

Internal Therapy - ^{131}I, ^{198}Au, ^{32}P
In Vivo Studies - ^{99m}Tc, $^{123,125,131}I$, ^{133}Xe, ^{51}Cr, ^{18}F, ^{67}Ga, ^{196}Au, ^{59}Fe, ^{111}In, ^{81m}Kr, ^{75}Se, ^{89}Sr, ^{201}Tl
In Vitro Studies - $^{125,131}I$
RIA - ^{125}I, ^{3}H, ^{57}Co

These radionuclides, except for krypton and xenon, are handled in a solid or liquid form and thus are not likely to be released as an airborne effluent. In hospitals, the iodine usage rate is 800-1350Ci/y. Preparation and transfer techniques for iodine tends to retain it in the liquid phase. The emission of airborne radioiodine is minimal during normal operating procedures except for protein-iodination by researchers. Therapeutic ^{131}I is readily volatilized and can become an airborne contaminant when used in selected therapeutic procedures.

^{133}Xe usage rate is estimated at 1600-3000Ci/y. ^{133}Xe is an inert gas that can be released as an airborne effluent. ^{133}Xe has a biological half-life of about 1-2 minutes which results in low

*This study was performed under USEPA Contract No. 68-01-5049.

patient dose and is usually administered in 15-25mCi doses for patient imaging. Following administration, the patient exhales the ^{133}Xe gas into a spirometer, either with or without charcoal treatment[2] or a Douglas bag with ultimate exhaust through a roof stack.

^{99m}Tc is currently used in large quantities at hospitals at an annual usage rate of 15,600-30,600Ci/y. It is used as prepared by the manufacturer or is eluted in liquid form from a radioisotope generator (^{99}Mo - ^{99m}Tc) and prepared as needed. ^{99m}Tc is included as a potential airborne effluent primarily because of the quantity used in medical procedures.

On the basis of these considerations and assessment of reports on measurements of effluent radioactivity from hospitals,[3,4] the potential airborne effluents from hospitals were identified as ^{131}I, ^{133}Xe, and ^{99m}Tc.

MONITORING OF FACILITIES FOR AIRBORNE EFFLUENTS

Effluent monitoring data for the isotopes of interest were obtained from seven suppliers and a 1976 State of New Jersey study.[5] That data show that the radioiodine dominates the airborne releases from these facilities.

Reports of hospital effluents were mainly concerned with liquid effluents. However, a recent paper[6] detailed the volatilization of radioiodine during preparation and administration of therapeutic liquid $Na^{131}I$. A steady-state release rate of 5 nCi/min from a 30 mCi solution was reported. In a literature search, no quantification of airborne effluent releases of radioiodine from hospitals was found.

CONTROL TECHNOLOGY

Like the commercial nuclear power industry, the radiopharmaceutial industry evolved under strict federal and state limits on releases of radioactivity to the environment. In general, the suppliers and users of radiopharmaceuticals rely on the guidance of 10 CFR 20, Appendix B,[7] to determine the maximum permissible concentrations (MPC) for radionuclides in air or water that are contained in effluent releases and the workplace environs.

Essentially all of the effluent control equipment that was found in use in radiopharmaceutical facilities: (1) was developed in the commercial nuclear power industry, (2) follows proven, reliable designs copied from nuclear power plant equipment, and, (3) consists of off-the-shelf components or complete systems readily available from commercial vendors.

Generally, the means to control airborne radioactive effluents fall into two classes according to function. The controls either: (1) dilute and direct the effluent to a specific point of release, or (2) hold up the effluent to reduce by decay the amount of radioactivity released.

The first class may be described generally as air flow or ventilation controls. These provide no hold-up or time for decay of routine releases except the delay due to the effluent's transit time through the ventilation system. These controls direct the radioactivity from its source within the building into the ventilation system

and then to the controlled release point. Examples include fume hoods, glove boxes, wall fans, and vent stacks. Each is intended to provide, within a radioactive materials facility, a zone of air pressure lower than in the adjacent nonradioactive areas. The air inflow from adjacent zones also dilutes the radioactivity in the low-pressure zone. Ventilation controls may be used alone but are often combined with the second class of controls (i.e., those that reduce the amount of radioactivity released).

The second class of controls may be described generally as radioactive effluent mitigators. They remove the radioactive materials from the effluent exhaust stream and hold them physically or chemically, reducing by decay the total radioactivity in the effluents. One example is activated carbon, which effectively adsorbs heavy vapors and gases, such as iodine and xenon. Other examples in this class of controls include silver-exchanged zeolites and silver-impregnated alumina and silica. Another example is the cryogenic trap for noble gases, which liquefies them and holds the liquefied gases for decay. For molybdenum and technetium, which may be found in particulate form in the effluent stream, a high efficiency particulate absolute (HEPA) filter can provide effective effluent control.

Suppliers Effluent Controls

From contacts with the radiopharmaceutical suppliers, it was found that the types of effluent controls used depend on the type and amount of each isotope handled in the facility.

Effluent control cost varies widely from installation to installation, depending on such factors as (1) volume of air to be treated, (2) degree of treatment needed, (3) ease of installation, (4) chemical quality of air treated, and (5) equipment reliability. Differences in routine maintenance and monitoring contribute most to differences in equipment performance from site to site.

Activated carbon and/or HEPA filters in the exhaust stream of the building and fume hood ventilation systems appear to be the rule among large radiopharmaceutical firms and common among small firms handling radioiodine or technetium. This equipment appears to function economically, reliably, and effectively and to be the choice for radioiodine control by radiopharmaceutical firms. It is estimated that the average cost of installing such systems to be approximately $8,000 per 1000 cfm, of maintaining such systems to be $1,000/y per 1000 cfm, and the dose reduction provided by such controls to be approximately factors of 10 to 100 for iodine and up to 3333 for technetium (the dose reduction provided depends on operating conditions). Xenon control at supplier facilities presently requires more elaborate equipment for which no good data could be found.

Users Effluent Controls

Because hospitals consume most of the radiopharmaceuticals produced, they were the focus of the study of radiopharmaceutical users' effluent controls. The most significant difference between suppliers' and users' effluent controls is the difference in scale. Small hospitals (less than 300 beds) appear to operate with no radioactive effluent controls because the principal isotope used (^{99m}Tc) is

handled in liquid solution and because the total activity handled per day is low (a few millicuries, typically). Both the medium-size (300 to 500 beds) and large (more than 500 beds) hospitals varied from using no controls to using extensive controls. Generally, the large hospitals appear to use controls like those of the radiopharmaceutical suppliers, because they handle large amounts of activity per day (tens or hundreds of millicuries) and they handle a wide variety of isotopes. Controls at the large hospitals range from fume hoods with HEPA and carbon filters and xenon traps to unfiltered fume hoods and no xenon traps. The medium-size hospitals tend to use xenon traps and unfiltered fume hoods, but may have no controls if they use nuclear medicine as infrequently as small hospitals, or meet NRC MPC requirements without controls. The carbon xenon trap offers a convenient, effective means to reduce xenon releases from hospitals and is the xenon effluent control preferred by the NRC.

SUMMARY OF EFFLUENT CONTROLS IN THE RADIOPHARMACEUTICAL INDUSTRY

From contacts with radiopharmaceutical suppliers, users, and effluent control equipment vendors, it was found that the control equipment is readily available, reliable, and effective in reducing radioactive releases from radiopharmaceutical facilities. The cost of controls appears to increase proportionately with the dose reduction provided by the controls. NRC requirements[8] and owner/operator perceptions of the controls' cost-benefit ratios determine what controls are used at a given facility. Based on the above it is believed that the effluent controls available and used today in the radiopharmaceutical industry adequately protect the environment and the public health and safety.

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A BIOETHICAL PERSPECTIVE ON RADIATION PROTECTION AND "SAFETY"

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By way of developing the central theme of this Congress, I propose to review three problems of major concern to policy makers whose task it is to protect public health by setting criteria and standards for "safe" radiation management. The first problem is to decide if current conceptual tools for assessing potential biohazards--namely, basic harms to valued living systems--are ethically adequate. The second problem is how to set safety standards on the basis of informed consent to scientific evidence presented by experts who disagree in interpreting that evidence. The third problem is how to resolve value-conflicts underlying expert disagreement, namely conflicting philosophies about radiation protection.

A fundamental bioethical principle must be firmly established if we are to analyze and organize scientific evidence concerning radiation exposure, and to separate genuine from counterfeit claims to credibility. Social justice and equity require an equitable management of sources of basic harms, that is, potential hazards which might have adverse health effects and unjustifiable social consequences. By "equitable management" I mean that policy makers should first be comprehensively informed about the broad spectrum of both natural and ordinary man-made hazards that may have health effects for large segments of the population; then make comparisons of the actual risks as well as costs per capita to reduce these effects; and only then make policies and set safety standards that will get the most public health protection for the many out of a finite amount of money. Potential hazard management is ethically equitable only if it is proportional in relation to actual basic harm that can be identified and reduced by expenditures of human effort, time, and money.

CONCEPTUAL TOOLS

Contrary to a popular misconception, "hazards" have neither a bare facticity nor an intrinsic morality predetermining how human beings should behave in relation to them. Hazards are not baldly "there" in nature or in human transactions with it. What people regard as hazardous in any given era reflects what they have come to know about their environment, and what they value as essential or desirable on a scale of real possibilities. In short, human beings structure hazards; they are, in that sense, human artifacts. A hazard is not by definition "toxicity of substance" or "violence of event" or "magnitude of consequences" that can be known, classified, and predicted. A hazard exists only when, and to the degree that, harmful exposure of and assimilation by the human body or other valued living systems becomes a genuine and not merely an imaginable possibility. That possibility exists only when there is an inability or failure to devise and maintain controlling actions or safeguards.

Because there are vast uncertainties about "how the world works," it serves no human purpose to bewail our "legacy of risks to future generations," and then make the fraudulent claim that the goal of hazard management should be to assure centuries of control over toxic elements or prediction of future adverse events. Prof. William Clark states that hazard management is "the adaptive design of hazard structure," and that the primary goal of hazard management is "to increase our ability to tolerate error and to take productive risks."(1) His statement stands in sharp contrast to a popular yet unexamined notion, expressed as well as anyone by Wolf Häfele, that "we are locked in a world of untested hypotheses (of unimplemented trials) because we dare not let experience prove us wrong. The costs of failure have grown too great."(2) Not only does this notion reflect the New Pessimism spawning defeatism and pseudoscientific dire predictions which now pervade our cultural climate; but it also constitutes in itself the ultimate hazard -- the failure to design and maintain structures of social resiliency. It is the social ideal of resiliency that has been a major driving force behind the emergence of highly complex and technologically advanced societies. The social ideal of resiliency accounts for the burgeoning art of risk-analysis.

The hazy connection between hazards and risks gives rise to another common misconception. If popular literature on the subject is any indication, "risk" is steadily acquiring the moral opprobrium reserved for other four-letter words. I do not intend to add to that moralizing. Suffice it to say that many have adopted the uncritical assumption that risk is a normative concept for certifying consequences to human beings that are harmful, dangerous, or "bad." These contrast sharply with consequences that are beneficial, pleasurable, or good--hence by implication risk-free. This assumption is altogether understandable because it reflects a basic value-conflict about the nature of risk-taking.(3) For some persons, risk-taking is by definition hazardous, harmful, and perhaps the result of a demonic compulsion suppressing nobler human pursuits. For others, the word risk stands for the opportunity to undertake what is challenging and venturesome, innovative and fulfilling to the human spirit in its endeavor to live "the good life." This value-conflict has developed because risk-taking is not inherently good or bad -- neither in a psychological sense nor in a moral sense. The fact that the concept of risk is negatively overloaded in popular usage has no analytical justification.

Because of a facile identification of risks with hazards, a false antithesis has been set up between risks and benefits--as if there were a way to have one without the other. The trouble with the phrase, "risk-benefit analysis" is twofold: it fails to express a proper symmetry, and it tends to obscure the primary motivating force of human activity, i.e. the foreseen and intended benefit which can be gained or lost. In concrete decisions, what is actually "at risk" is the possibility that intended benfit may not materialize, and instead harm may occur. When harm results, it is clearly unwanted and unintended. Risks and benefits are inseparable, not antithetical.

A major problem about the growing dispute over radiation protection and radioactive waste management is the inadequacy, not of risk analysis, but of harm-benefit analysis. Some refinement in the notion of benefit is essential. Okrent and Whipple suggest three qualitative distinctions in benefits, namely those goods essential to society

(e.g. food, water, energy) or basic goods; advantageous to society (e.g. most manufacturing); and of peripheral if any value to society (e.g. aerosal deodorants having substitutes at lower cost and likelihood of harm.) Each qualitative benefit has corresponding levels of harm. (4) Basic harms may result from being deprived of good essential to subsistence and material well-being. Justice and equity require a society to provide access to basic goods and avoid basic harms. As for second-level benefits, the total outcomes of any social policy toward such improvements will have an unclear mix of benefits and harms. Auto and airplane manufacture afford major economic benefits to employees, capital investors, travelers, and the general health of international economies. Yet each time someone drives a car or enables an airplane to take off, the benefits pursued may entail the possibility of unintentionally causing the death or serious impairment of a fellow human being. Any society must, at some point, deliberately decide how we ought to balance economic benefits and costs against possible harm or loss of life.

According to critics of such balancing, a human life is of infinite value, and its loss or impairment cannot be put in a class with other "negative consequences," much less be given a finite monetary value. To do so indicates the moral bankruptcy of our materialistic, consumerized, decadent society. Cost/risk/benefit quantifications, say its critics, manifest a loss of respect for the sacredness of human life. Those who defend this conceptual tool have often used simple observations, such as "There are necessary tradeoffs in any public policy decision," or "Everyone puts a finite, monetary value on one's life when buying life insurance, installing safety mechanisms in a home or auto, taking hazardous jobs because they pay higher wages." Although true, such analogies are not sufficient. The public must be confronted with the fact that any society has but a finite amount of money to spend on health protection and safety, and that the ethical problem is to get the most protection for the most people from this finite amount.

As a conceptual tool which attempts to enhance informed consent, cost/risk/benefit quantifications are simply one tool among many others whereby policy makers endeavor to allocate finite amounts of money in a just and equitable manner. They are not tools for putting some callous dollar value on human life or injury as a moral judgment of individual worth, much less of using economic losses to society as a measure of personal expendability. We are in fact maximizing the value we place on human life when we endeavor to allocate limited amounts of money in such a way as to reduce widespread hazards, thereby preventing as much loss of life and protection from injury as possible.

The fact that our tools for balancing economic costs against risk to human life are not morally or ethically objectionable does not amount to saying that they are easy and acceptable to the public. Far from it. The task of public education in this matter is monumental. Moreover, as my colleague in social ethics, George Pickering, observes: "We are going to have to do more than find some level of 'acceptable risk;' we are going to have to come to terms with the question of 'justifiable harm.' There are, after all, some kinds of harm which cannot be avoided; but there are other kinds of harm which any society should not allow and against which it should adopt protective or remedial measures to the best of its ability."(5) Which is which?

We must face up to the disconcerting task of developing a more enlightened concept of--and method of informed consent to--unavoidable hence justifiable harm, and not divert attention away from it by focusing exclusively on "acceptable risk" criteria. Our failure to take up this task lies at the root of the second problem noted above: the frustrating dilemma of a policy-maker who wishes to set safety standards on the basis of informed consent, yet when he turns to scientists upon whom he relies for "expert testimony," he finds they have basic disagreements about what data should count, how it should be interpreted, and what level of health protection is "safe enough to be safe."

EXPERTS, REGULATORS, AND STANDARDS FOR "SAFETY"

Aaron Wildavsky has recently observed, "Experts are used to disagreeing, but they are not so used to failing to understand why they disagree." (6) At the heart of the matter lies a misconception about safety, especially as it relates to risk acceptability.

A case in point is the unending controversy over whether or not there is a threshold for radiation below which no harmful effect occurs. For most toxic elements, a threshold concept has been accepted. It carries the implication that below a threshold dose any exposure is "absolutely safe." But over the twenty years of evolution in radiation protection philosophy, the ICRP and NCRP came to adopt a conservative assumption, namely that it would be more prudent to assume some harmful effect from any radiation dose, however small, than to assume a threshold dose and then discover data proving it to be false. This conservative assumption carries the implication that there is no absolutely safe radiation dose except zero, and every dose greater than zero entails a corresponding possibility of genetic or somatic harm. In the ensuing process of applying a linear no-threshold hypothesis to the development of standards, regulatory institutions and some of their expert advisers seem to have forgotten that their quest for radiation limits rests only on a hypothesis, a conservative assumption, and not on a scientifically established fact. According to G. Hoyt Whipple, "The data on the biological effects of radiation can be interpreted in terms of a threshold dose, but even the vast amount of radiobiological data cannot conclusively prove the existence, or absence, of a threshold."(7)

Given this state of affairs, the dilemma of a policy-maker could be mitigated if two factors were clarified: (1) the meaning of "safe" and (2) the ambiguity of a threshold concept.

A profound misconception of "safety" dominates the controversy over radiation protection. The working assumption has been that safety is an intrinsic, measurable, absolute property that a given system or product or activity can and should possess. Our society has institutionalized and appointed the regulator to measure approximations to that elusive property. The mandate of the regulator is to make ever more stringent regulations, presumably to come ever closer to that property by reducing risks. But the only risks he is expected to monitor and minimize are a small percentage of the total spectrum of risks tolerated by members of society as a whole. Intent on making a set of risks publicly "acceptable" as an index of "safety," the professional regulator must continue to propose risk-reduction without regard to economic costs or social impacts of ever-changing regulations.

Presumably he is "only giving the public what it wants," namely safety. This spiral is likely to continue unless or until the public comprehends the fact that safety is not an intrinsic property measured by approaching zero-risk. Safety is an evolving, relational value judgment derived from current personal or social priorities. Whereas risks can be measured, quantified, and predicted, safety cannot be measured, much less predetermined by the presence or absence of risks.

Judgments of safety are judgments about the justifiability or unjustifiability of harm. The process of reasoning for ethical safety-policy decisions should be dictated--not by risk avoidance, an impossible ideal--but by comprehensive risk/risk assessments and cost/risk/benefit ratios. When these comparisons make it clear that a point of diminishing returns on allocations of money, time, and effort has been reached by comparison with other potential hazards in a society, then the product or process under scrutiny is "safe enough." If indeed unintended and unwanted harm should occur, then such harm can be judged justifiable because unavoidable or negligible by comparison with other harms and essential benefits.

Greater clarity about the process of making safety judgments would help to clarify the ambiguity of a threshold concept. As noted above, regulatory standards have been predicated--not on a scientifically established fact--but only on a hypothesis, an assumption that there is no threshold below which harmful effects will not occur. As a result, reduction in radiation exposure levels has been required to become "as low as reasonably achievable," (ALARA). J. J. Cohen has this to say about ALARA: "Philosophically, this is based on the premise that, since we do not know the effects of low-level radiation exposures, the conservative standard will effectively minimize them. Supposedly any degree of reduction in radiation exposures will do some good. However, some evidence indicates that there might in fact be a net beneficial effect of radiation at low levels. Since we do not in fact have a complete understanding of low-level exposure phenomenologically, perhaps we should recognize the possibility of beneficial as well as harmful effects. If the net effects are in fact beneficial, then by insisting upon the application of ALARA--rather than being conservative--we may actually be causing harm."(8) Speaking from an ethical perspective, I must ask what scientific evidence justifies marginal reductions if "conservative" actually means "doing the least harm?" Since net beneficial effects of low-level exposures can be proven with other toxic materials (e.g. copper, selenium, fluoride), why not look at radiobiological data through the lens of that assumption?

The assumption of a zero-threshold for "safe" radiation exposures may be justifiable for some radiobiologists. But for the policy maker there can and must be a practical threshold below which the possibility of unintended and unwanted harm is ethically justifiable because it is either unavoidable or negligible by comparison with other potential hazards against which citizens ought to be protected.

The dissensus amongst health physicists and biostatisticians may mean that what we need is not more stringent regulations; but rather we need to devise innovative institutional methods for dealing fairly with complaints without undermining still further public confidence in experts, in safety-policy decisions, and in regulatory actions. Whatever these institutional innovations may be, they must somehow take

account of the possible origin of basic disagreements over safety judgments, namely conflicting philosophies.

PHILOSOPHICAL ORIGINS OF VALUE CONFLICTS

The mounting controversy over radiation protection and radioactive waste management has revealed basic value conflicts, compelling us to probe more deeply into the philosophical and ethical principles from which values derive their justification.

For some time now, representatives of environmental protection organizations, together with special interest groups purporting to protect the public's health and best interests, have espoused as a fundamental philosophical principle "non-degradation of the environment"--defining a "degraded environment" as any place that human actions have affected or changed.(9) Some representatives adopt the "pre-existing natural state" of any given environment as an appropriate standard for human transactions with nature because "it emphasizes the role of a trustee as one who maintains the non-renewable environment as it was originally, to pass on to the next trustee." This fundamental goal is a key consideration, "because if any degradation is allowed (in the name of 'allowable radiation exposure'), there is no clear bound at which degradation becomes, by anyone's standard, too much." On behalf of the public,these representatives are of the opinion that the ethical principles of equity and participation require criteria for radiation protection against energy technologies and waste disposal to be neutral to future generations, stating that "the least unfair way of managing intertemporal relationships is for each generation to try to leave the earth as it was when they arrived. As a goal, the only acceptable distribution of hazards and benefits is the neutral allocation, where no pattern of benefits and hazards is imposed."

Formulas such as these obscure two questionable assumptions: first that an untouched "natural environment" by definition manifests a superior, if not sacred order which human interventions or changes violate to some degree; and secondly, that a trustee of a so-called "natural environment" can do nothing more nor less than pass it along in its original pristine state; to do otherwise is to be guilty of a moral wrong.

The philosophy of non-degradation has a long history, as is clear to anyone who has read Book I of Georgius Agricola's De Re Metallica, published in 1556. This sixteenth century inventory of objections to disturbing a pre-existing natural state of the environment make it abundantly clear that the arguments advanced in the name of protection of public well-being are specious.(10)

The philosophy of non-degradation uncritically assumes the idea that a benign environment is rapidly being ruined by human beings. However the historical record attests that an untamed environment has repeatedly wrought massive human degradation through catastrophic effects of famines, plagues, floods, earthquakes, and so forth. The basic problem, therefore, is not a question of pursuing an ideal of "non-degradation" of the environment, but rather represents a highly complex challenge of both protecting life-sustaining and aesthetic qualities of the biosphere and developing technologies that provide basic human goods as a necessary condition for maintaining a preferable environmental quality. As a fundamental, meaningful principle

for securing that environmental protection, non-degradation is vacuous.

From a bioethical perspective, it is justifiable for policy makers to establish criteria and standards for health protection by reference to naturally occurring radiation sources from which man-made applications are derived. But it is not justifiable on the basis of a philosophy of non-degradation or trusteeship over some pre-existing natural state.

Those responsible for providing access to basic goods, methods of informed consent, and an equitable management of biohazards have an ethical obligation to derive value judgments of safety, acceptable risk and justifiable harm from a philosophy of congruence with a pattern of benefits and harms already established by naturally occurring radiation sources with which human beings have lived and evolved throughout recorded history. The philosophy of congruence and of logical consistency require a policy maker to form value judgments on the relative benefits of providing protection against radiation by first taking account of wide variations in personal exposures and population exposure from naturally occurring background sources. (These include external sources from cosmic rays, together with the radionuclides they produce and primordial radionuclides in the earth; and internal exposure from natural radionuclides inhaled or ingested via food and drinking water.) Large segments of the population in the U.S.A. receive natural external radiation doses varying from 40 to 105 mrem per year simply because of geographic location. Variations in natural exposure to thorium in monazite sands along the southeastern coast of India range from 130 to 2,800 mrem; while on the coast of Brazil, exposure ranges from 90 to 2,800 mrem with an average of 550 mrem per year. There is no scientifically established evidence that there are basic harms to those so exposed.

Human tolerance for, indeed dependence upon, such wide variations in natural radiation sources for several millenia demonstrate that increments from man-made applications of those natural sources can be kept well within the range of those variations without inflicting either unjustifiable harm or deprivation of basic goods to members of a society.

The philosophy of congruence and corresponding ethical principles set forth here are in contrast to what has been assumed by regulatory agencies when they have set excessively conservative standards in the past. However, with increasing knowledge of a pattern of benefits and harms from natural radiation, there is ethical justification for their gradual revision. It is a matter of fact that the largest increment from man-made radiation exposure comes from medical and dental health practices. These exposures are 10 to 100 times greater than other man-made sources which by contrast are stringently regulated.

From the perspective of bioethics, the inequitable management of biohazards in general--and of radiation protection in particular--ought to be reviewed and remedied. There is clearly a category of negligible risk and negligible harm which in practice ought to be ignored. This category coincides with the ethical principle of justifiable harm.

The perceptual problem of managing radioactive waste could benefit greatly from a philosophy of congruence applied to performance criteria for radioactive waste disposal. As they are proposed, these criteria require that ultimate waste disposal shall be conducted in such a way

that there is no net increase in risk of harm by comparison with the typical ore body of natural uranium which yields the energy from which the wastes are derived. In other words, the wastes would be disposed of in a way that returns them to the same (if not better) level of risk which natural uranium ore in the earth's crust poses. The waste form would be required to have the same stability as the original ore body; the medium containing the wastes would be required to retain the same integrity as the medium containing the ore; and the geological media surrounding and isolating the wastes would be required to retain the same integrity of isolation from the biosphere as that isolating the original ore bodies. If technologies exist to meet these requirements, then the public and professional critics cannot logically demand greater "safety."

CONCLUSION

In view of the above reflections, I suggest that the following principle might serve as guidance in the formulation of social policies for radiation health protection:

Any involuntary risks imposed by social policies for radiation protection must be congruent with, must not be in excess of, and may be reasonably less than, those involuntary risks imposed by the wide variations in naturally occurring toxic elements and harmful effects from our natural environment.

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ETHICAL NORMS IN THE USE OF RADIATION AND NUCLEAR POWER

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There is a very extensive literature and philosophy treating the ethics, morals and the principles underlying our behaviour as individuals and as members of an organized world. But when it comes to the point of applying ethical norms to practical (real or hypothetical) situations involving decisions by more than one or a few individuals, the principles become less clear and the guidelines less specific. Patterns of behaviour which appear simple and straightforward when applied to individuals' behaviour and life, become fuzzy and difficult to handle when upscaled to encompass many individuals and many lifespans.

Ethically satisfactory behaviour in work with radiation should be governed by the same basic moral requirements as are our other functions in society. These requirements may be condensed into three concepts: honesty, consistency, and generosity.

The first two are easier to define and explain acceptably to most people, though both of them carry their own intrinsic difficulties. Honesty is telling the whole truth and nothing but the truth, and admitting and stating ignorance when information is lacking. The ethical problem comes perhaps mainly in how far honesty should be pursued in terms of information: Should the cancer patient always be told of his prospects? Should the public be informed to the last confusing and disturbing detail?

Consistency means applying the same scales and standards to all comparable situations. This principle is easy to follow, if the scales and standards are accepted generally and across the board. Difficulties arise if differing scales and new standards are brought to bear on the same subjects.

The resolution of the latter type of conflict may lead us into the last category, that of generosity. This is the soft element of the structure. Certainly it is the most difficult one to define and delimit, and - simply due to its softness, and ad hoc non-rational character - also the least acceptable, in relations already so complex that the ground rules must be kept as simple as possible. But this softness is the factor which elevates ethics to a set of rules above the cook-book recipies for human behaviour. Also, it introduces an aspect of flexibility which enables the set of rules to cover new and unforseeable consequences of a given situation or action.

As examples for discussion are chosen three or four situations where ethics are to some extent involved in the way questions or problems are approached and might be solved. These are

a) the way we specialists present our information and our arguments to the non-specialists and the public in general, and
b) the way we handle possible radiation damage and fears of damage in individual cases today, and
c) the way we evaluate the effects inflicted, today or later, on future generations.

These are areas with real problems and difficulties, which may be resolved according to various models, and where ethical attitudes may be important for the choice of solution. Many of us have worked through these questions not only once but several times, and yet the same and definite answer does not come up every time.

First, the presentation of our - the specialists' - views and arguments to our employers - the public. Let us start with a look at two cost-benefit analysis presentations.

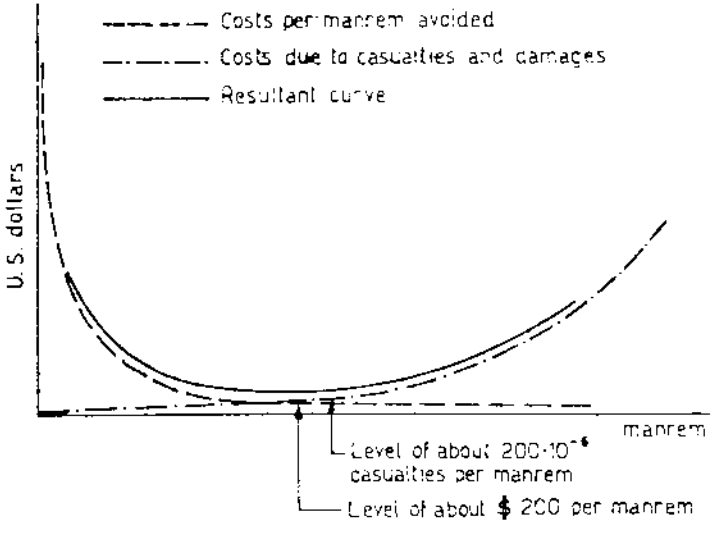
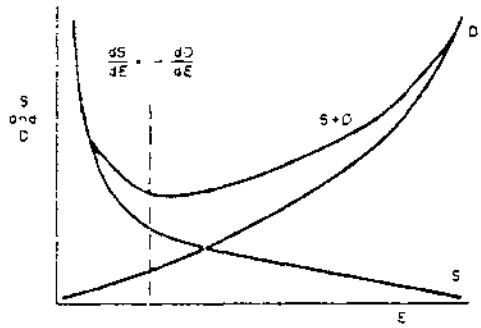


Fig. 1

Fig.1 is a curve I showed some years ago indicating cancer risk and cost per man-rem avoided, according to the numbers which were accepted at that time. The point I wanted to make is very clear from the way I chose my dimensions. My speaking point was that the trough of the resultant curve is flat, and that there is a fairly wide dose range where the curve gives little advice on where

to settle. My conclusion was then that experts with responsibility for protection would tend to advice a lower, more restrictive limit, whereas experts - equally expert - with responsibility tied to production will tend to settle at the higher limit of the flat area, leaving a range where a politically determined choice will have to be made.



Differential cost-benefit analysis.
E = a variable reflecting the exposure, possibly in man-rems.
S = total cost of achieving a value of E.
D = total cost of detriment associated with a value of E

Fig. 2

In some contrast then, the official ICRP version in Fig.2 which represents an idealized description of the elements entering into a cost-benefit analysis, resulting in a very well defined minimum. I have even heard this minimum described as the point where the tangents of the two curves have equal and opposite values.

There is nothing unethical about either of these presentations. In the first case, the form of the curve was chosen to demonstrate the complexity of the situation and to emphasize the responsibilities of the non-specialist authorities. In the second case, the presentation is constructed to demonstrate the elements and principles involved.

The question of ethics comes in when models of this kind are used without comments, and presented as if they were the full and absolute truth, with no doubts, uncertainties or alternatives.

Table 3. Estimated number of cases of serious genetic disease per 10^6 man rem delivered at low doses and dose-rates (see Oftedal and Searle, 1979)

Type of effect	nos. at Equilibrium	1st and 2nd Generation
Unbalanced translocations: Risk at birth	30	23 + 6 = 29
Trisomics + XO	30	30 + 0 = 30
Simple dominants + sex-linked	100	20 + 16 = 36
Dominants of incomplete penetrance etc.	160	16 + 14 = 30
Multifactorial disease not maintained by mutation	0	0
Recessives	.	–
Total	320	89 + 36 = 125

Fig. 3

Now, let us look at a genetic risk estimate developed for an ICRP Task Group. On the basis of discussion of the limited and unsatisfactory scientific facts before us, and on our evaluation of them, we arrived at a risk estimate formulated as shown in Fig.3. The significant features of this table are

1) that it was made for radiation protection purposes, which means that it was evolved under the instruction that an updated genetic risk estimate was needed, in connection with the then forthcoming ICRP Publ. 26. If the geneticists did not produce specific numbers, someone else - presumably less qualified - would have to do so. The ethics of this situation then comes in as a conflict between the moral standards of the scientist - which tells you not to overinterpret your data - and the demand for us to shoulder the responsibilities as functioning members of public health/technocracy sectors of the international society. In the actual situation, the obligation to act as advisors was deemed more important.
2) On the other hand, it became a necessity to make it quite clear that the risk estimate contains an element of non-scientific though honest and well informed guesswork. This comes out in the estimate of the risk for complexly inherited genetic ill health - dominants of incomplete penetrance etc, for which the basis for estimation is rather unsatisfactory - to be chosen as the sum of the chromosomal, simple dominant and sex-linked mutations - for which the basis for estimation is rather better.

The end figure comes out as a single number - which is what the commission needed - but there is obviously no claim to a mathematical exactness, nor any expression of the error attached to the figure. The number is quite close to the range of the BEIR and the UNSCEAR 1977 estimates.

The advantages are two-fold, as I see it: The most insecure number - the 160 due to incompletely penetrant dominant mutations etc - is linked to the more secure numbers, and any broad revision may be taken care of in the same way for all categories. Secondly, the number of calculating steps is reduced to a minimum.

The ethics of the exercise then really boils down to the very basic dogmas that we all learned in our homes, schools and undergraduate labs: Be honest, and don't take for your own things for which you have not worked. In that sense, this example is one of those that seems to be relatively simple to cope with.

The second situation pertains to a more or less legalistic problem, namely that of compensation for damage due to radiation, in particular cancer after relatively low doses. The problem arises because with stochastic events of this kind, cause may be unknown - and the case is usually termed "spontaneous", or it may be due to radiation exposure, but there is no way of distinguishing these two categories.

It is possible - by the use of Baye's theorem on conditional probability - to calculate what the relative probabilities are. The spontaneous frequencies of various cancers are quite well known in many countries, and the induction rates are fairly well established for many types. If the dose then is known, the conditional probability for one or the other being the cause may be calculated. However, at this point, we are in danger of moving into a numbers' game, which is bound to be unjust in some cases, and the outcome therefore not satisfactory. Regardless of whether one establishes probability criteria of 9:1, or 3:1 or 1:1 or 1:9, there is bound to be uncertainty and injustice, and also a sometimes disgraceful argument and dispute concerning the actual numbers used in the calculations.

An important aspect from an ethical point of view would be to shift the areas of dispute or of conflict of evaluation, into a frame of reference which is already well known and understood by all those involved. In other words, that the sense of security, or of assurance of justice, should depend not on technical subtleties (e g in dosimetry) or statistical refinements (e g in local cancer registration), but on straightforward and familiar categories like job description, and subjective risk evaluation.

Finally, the most difficult ethical questions of all, those of our attitudes to actions today which may mean trouble or ill health in lands far away and in future generations.

There is no doubt that the use of radiation and nuclear power puts a load on future generations. Partly so by introducing into present generations genetic damage which is not expressed in the immediately following generation but may crop up later, and partly by delayed exposure of future generations through the release of

long lived radioactivity from reactors, reprocessing plants, and waste repositories.

In both these areas - the induction and manifestation of recessive mutations in humans, and the future radiation loads from repositories - we are fully ignorant - within limits - which means that we ought to use as our design criteria the most pessimistic assumptions, as advocated in principle by modern risk philosophy under such circumstances. This is in contrast to circumstances of uncertainty (and not ignorance) where a weighted risk may be accepted.

The ethical issue then falls into two parts:

Firstly, is it ethically permissible at all under any circumstance to charge future generations with the load of manifestations of damaged genetic material, in untold generations and therefore unavoidably in large absolute numbers, in order to attain advantages - e g energy - for ourselves and our immediate descendants.

And secondly, is it ethically permissible to charge distantly later generations with the obligation to maintain a social organization capable of an appropriate control function for the physical containment of radiation sources.

Parallel situations may be presented, showing that the problems are not unique for radiation. In general, we fall back on the same basic elements as was discussed above:

Honesty demands that we explain the situation fully to everyone interested. And if everyone interested are not able to grasp and understand the problems, this is our - the experts' - responsibility.

Consistency demands that the risks we ask our distant descendants to accept is not greater than those we would accept - in the absence of any corresponding benefits. This makes a cost-benefit evaluation very difficult, and the use of our present day ICRP standards for setting norms for the distant future may appear doubtful, and certainly needs a separate discussion and justification.

Generosity might be taken to mean that we should show reticence in utilizing our resources, even within our self-imposed limits, so as to leave our distant descendants a maximum of options to handle situations and problems which may be totally unknown today and completely unforeseen by us.

THE IRRADIATION OF HUMAN VOLUNTEER SUBJECTS IN RESEARCH

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The uses of radiation in diagnostic radiology and nuclear medicine are justified by the benefits that are expected to accrue to the persons being examined. But in medical research it is sometimes necessary to obtain data from normal healthy individuals. These subjects do not themselves gain any physical benefit from such research.

CONSTRAINTS

Because of the possible detrimental effects of radiation, besides its ALARA concept, the ICRP recommends[1] that in the case of pure medical research:

(a) the irradiation of persons should be undertaken only by properly qualified and trained persons,
(b) the irradiation should have the consent of the institution,
(c) such consent should be based on advice of an appropriate expert committee and be subject to local and national regulations,
(d) the volunteer subjects should fully exercise their free will,
(e) the estimated irradiation risks should be explained to subjects,
(f) the magnitude of the detriment to the volunteers should be authorised for each research program.

The ICRP states that the higher the dose the more rigorous should be the selection of true volunteers and their capability of understanding the risk. In the case of children and other persons incapable of giving their true consent, experimental irradiation should be undertaken only if the expected dose-equivalent is of the order of one-tenth of the limit applicable to members of the general public[1].
In Australia the NHMRC has recommended the following principles[2]:

(g) annual doses should not exceed those for members of the public,
(h) the actual doses delivered should be the lowest practical,
(i) adequate precautions should be taken against overdosage,
(j) only true volunteers should participate,
(k) an expert committee of the institution should evaluate a comprehensive safety assessment for each proposed irradiation.

In 1977 the World Health Organisation made recommendations for the control of experimental radiation. The WHO Report[3], which appeared subsequent to the investigations discussed below, provides guidelines rather than prescriptions for making decisions in specific cases. It suggests four categories into which projects can be classified depending upon the amount of total body or weighted single-organ radiation dose to be received by a subject. Category 1 contains projects which produce an annual total body dose of less than 50 mrem; i.e. within variations of natural background and hence of negligible risk[3].

Two examples of the irradiation of human volunteer subjects in research are described below.

EXAMPLE 1

As part of a study of artificial hip joints, information is needed as to the extent of shortening of the femur, in vivo, when the normal load of the body weight is applied. This information can be deduced by superimposing two diagnostic radiographs of the femur, the first radiograph obtained when a subject bears his weight on one leg only, and the second when the subject bears no weight on that leg (i.e. it is suspended). The proposed x-ray plant settings are: 68 kV, 100 mA, 0.12 sec, 1.83 m FFD, 2 mm Al filtration.

For dosimetry purposes, ideally, measurements of absorbed doses could be made in a phantom, but an estimate can be made as follows[4]. From the literature typical measurements of exposure made at a distance of 1.68 m (corresponding to the anterior surface of the leg) under similar plant conditions to the above, yield an average exposure value of 5.2 μC/kg (20 mR) per film. Neglecting attenuation within the leg, the absorbed doses at 70 kV in bone (97 Gy.kg/C=2.5 rad/R) and in soft tissue (bone marrow and gonads; 35 Gy.kg/C=0.9 rad/R), yield dose equivalents of 500 μSv (50 mrem) in bone and 180 μSv (18 mrem) in bone marrow and gonad tissue. These values are considerably less than the respective annual maximum permissible doses[5] of 3,000 mrem and 500 mrem for volunteers by factors of 60 and 27 respectively. As the smaller number is the more restrictive, up to 27 films could be obtained from a single volunteer. The dose is minimised by requiring only two films per volunteer, totalling 7% of the annual maximum permissible dose-equivalent. In current ICRP terms[1], the Effective Dose Equivalent ($\Sigma w_T H_T$) for gonads + red bone marrow + bone surface is 30 mrem, placing this project in WHO category 1.

An expert committee was established to evaluate the safety of the proposed project. After consideration the committee gave its approval to the project, subject to the following conditions.

1. Only the least number of volunteers needed should participate.
2. All volunteers must be at least 21 years of age (now 18 years).
3. The investigation be restricted to male subjects and a gonad shield be used for all x-ray exposures.
4. Each volunteer must sign a Form of Consent after having read the NHMRC Statement on Human Experimentation[6] and a description of the proposed project. Subjects must have the opportunity of asking any questions and of reading the project safety assessment.

EXAMPLE 2

To improve dialysis treatment of renal failure, basic information is needed as to the rates of movement of various blood solutes between different compartments in the body[7]. The measurements can be performed by injecting into patients undergoing regular dialysis treatment, a known quantity of radioactively labelled solute, and subsequently analysing the activity contained in sequential blood samples.

Each subject will receive two intravenous injections of ^{14}C-labelled urea and two of ^{14}C-labelled creatinine. Each injection will contain 2.4 MBq (65 μCi) activity of ^{14}C-labelled solute dissolved in 15 ml of sterilised saline. At least two weeks will elapse between consecutive injections. Both solutes are expected to be distributed in a total volume of about 30 litres. The amount of labelled urea or creatinine remaining after two weeks would be about 3.7 kBq (0.1 μCi). The project envisages the participation of four stabilised chronic haemodialysis patients of either sex, aged between 20 and 40 years and having a life expectancy ranging from 5 to 10 years.

Adopting an average beta energy value of 0.052 Mev, and assuming that the absorber is infinitely large with respect to the range of radiation, then the initial absorbed dose-rate from a single injection is given by:

$$\frac{2.4 \times 10^{6}\,s^{-1} \times 0.052\ \text{Mev} \times 1.6 \times 10^{-13}\ \text{J.(Mev)}^{-1}}{30\ \text{kg}^{-1}}$$

$$= 0.67\ \text{nGy.s}^{-1}\ (0.24\ \text{mrad.h}^{-1})$$

Hence with unit quality factor, the initial dose-equivalent rate, D_o, is 0.24 mrem.h^{-1} (0.67 nSv.s^{-1}). The decrease in activity due to radioactive decay in a year is negligible. Thus using the above biological elimination data the effective decay constant λ is given by:

$$\frac{3.7\ \text{kBq}}{2.4\ \text{MBq}} = \exp(-\lambda t) = \exp(-\lambda \times 2 \times 168\ \text{h})$$

$$\text{Hence } \lambda = 0.0193\ \text{h}^{-1}$$

The annual dose-equivalent, D, is obtained by substituting these values into the equation for cumulative dose.

$$D = \frac{D_o}{\lambda}\left[1-\exp(-\lambda T)\right] = \frac{0.24\ \text{mrem h}^{-1}}{0.0193\ \text{h}^{-1}}\left[1-\exp(-0.0193\ \text{h}^{-1} \times 8736\ \text{h})\right]$$

$$= 12.4\ \text{mrem (124 μSv)}$$

The total annual dose-equivalent from four such doses is 50 mrem. This value is only 10% of the whole-body annual maximum permissible dose-equivalent.

As in Example 1 an expert committee gave its approval subject to previous conditions 1,2,4 and the following:

5. Each subject should be exposed to the minimum number of tests.
6. Supervision is necessary to prevent overdosage.

The Committee also considered the dependence of dose on the rate of elimination of the active solute; responsibilities resulting from multi-institutional participation; the free expression of consent and the lodging of signed consent forms.

Subsequently the Committee approved an extension to the project to include 2.4 MBq (65 μCi) of ^{14}C-sucrose and 5.5 MBq (150 μCi) of ^{3}H-vitamin B12. Using the above method, and allowing for double injection and different volumes of body fluid it was found that the ^{14}C-sucrose would deliver an annual dose-equivalent of 50 mrem (500 μSv). The ^{3}H-vitamin B12 has a different energy and different effective decay constant which will produce an annual dose-equivalent of 37 mrem (370 μSv)[4]. This project, which was in WHO Category 1, has since been completed[7]. In current ICRP terms[1], the annual dose-equivalent of 12.4 mrem is 10% of the dose-equivalent limit.

Because of rapid excretion, the committed dose-equivalent is also 12.4 mrem.

GENERALISED USE OF HUMAN SUBJECTS

In experimental procedures that involve the use of human volunteer subjects, there is a need to protect the rights and welfare of the subjects, and to safeguard the interests of the institution and the investigator. Generally, it is not easy to satisfy the separate ideal criteria of the three interested parties. Minimum requirements are listed in two authoritative documents[6,8].

At the University of New South Wales a set of rules governing experimental procedures of any type involving human subjects has been adopted[9]. The rules have been subject to lengthy consideration within the University. They provide for a committee to advise the Vice-Chancellor on all applications to undertake human subject research, and they make provision for certain exclusions.

Some difficulties which have been encountered include the wording of the consent form which, if to fully satisfy lawyers, would deter most potential subjects. To obtain "informed" consent, an investigator must give a full and frank disclosure of the proposed procedures and of all the risks entailed; this sometimes can defeat the purpose of the research. Insurance policies are usually effected to cover liability for possible claims resulting from negligence. In human subject research the claims could result from intentional acts. With greater community awareness of the need for social responsibility in science, the Committee will have challenging situations to consider.

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LONG-RANGE TRANSPORT OF RADIOISOTOPES IN THE ATMOSPHERE AND THE CALCULATION OF COLLECTIVE DOSE

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In calculating collective dose and in considering, under Article 37 of the Euratom treaty, possible transfrontier effects of unplanned releases, it may be necessary to model the dispersion of atmospheric releases of radionuclides over considerable distances - out to 1000 km or more from the source. Unfortunately it is difficult to extrapolate simple modelling techniques used close to the source and based on source meteorology because, in general, material will not move steadily on along straight line trajectories and spreading and dilution, and depletion by dry and wet deposition will vary along each trajectory in accordance with changing meteorological conditions. Under a EURATOM/CEA contract and with added support from the European Commission we have developed a model which takes account of these varying effects; this model, called MESOS, uses a data-base of real meteorological data over Europe and, although not sufficiently sophisticated to use in a predictive mode if and when an accident should occur, gives adequate derivation of probability distributions of different degrees of contamination for hypothetical accident studies. By summing the effects of consecutive short releases, maps of exposure from continuous routine releases are also deduced and combined with population density and food production matrices to yield collective dose.

THE MESOS MODEL AND DATABASE

The MESOS database (1) currently covers most of the year 1973 for the area of Europe shown in Figure 3; it supplies pressure fields and other meteorological data including cloud cover, precipitation intensity deduced from "present weather" codes, and temperature, at 3 hourly intervals from observations at synoptic stations and ships. A new database for 1976 covering a larger map area, including the Mediterranean region, is in preparation.

In the model, puffs released at 3 hourly intervals are tracked through the evolving pressure fields using quasi-geostrophic techniques. Along each trajectory a puff is treated as a vertical column expanding laterally, and evolving vertically according to the state of the boundary layer as deduced from local meteorological data

and the underlying surface of land or sea. Depletion by decay, dry deposition and wet deposition when it rains, are also included.

A continuous release over a 3 hour period between successive tracked puffs is treated as a series of puffs following intermediate trajectories and dispersion. Thus the area between calculated trajectories is exposed, representing the lateral spreading in the synoptic scale wind field. Integrated atmospheric concentrations are calculated for Kr85, Xe133, Xe135, Cs137 and I131, and dry and wet deposition for the last 2 nuclides. Exposure for longer term releases is calculated by summing the effects of consecutive 3 hour releases. More detailed descriptions are given in references (2) and (3).

SIMULATION OF WINDSCALE INCIDENT ON 10th OCTOBER 1957

The Windscale incident (3) provides one of the few instances in which monitoring (of I131 over Europe) allows comparison between measurements and predictions from the MESOS model (see Figure 1). Assuming a pattern of release from a 120 metre stack of about 30,000 Ci of I131 in total, with peaks late on October 10 before passage of a weak front, and after 9 hours on October 11 when water was poured on the pile, the model predicts that most of the first part of the release crossed over to the East of

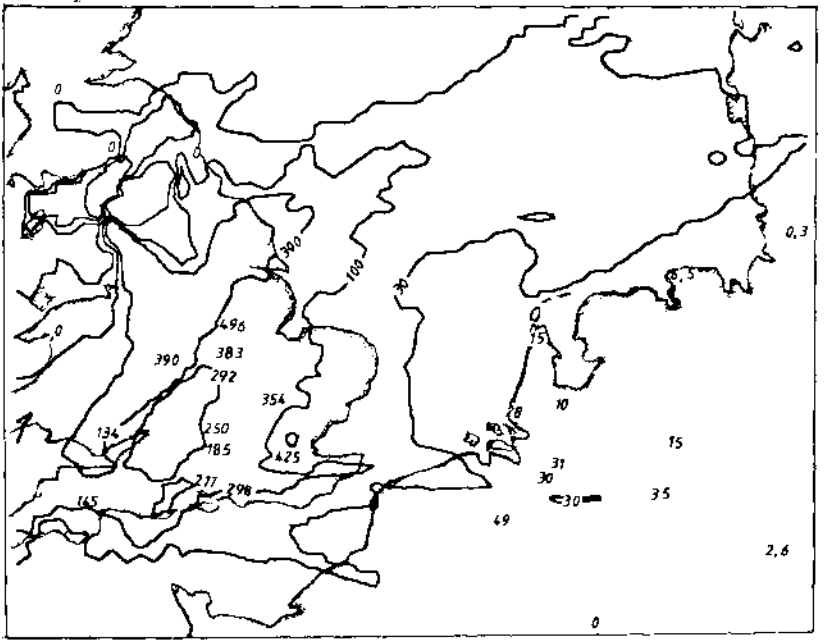

Figure 1. Measured time integrated air concentrations of I131 (p Ci, $d.m^{-3}$), resulting from the 1957 Windscale release, compared with predicted contours.

England above an inversion, penetrating to ground level with fumigation the next morning, whereas the second part gave rise to a plume down the W.Cumberland coast. Subsequent travel was influenced by an anticyclone moving East over N.France, and the release was eventually swept off East in moderate winds. The model predictions are sensitive to exact trajectories; for example a small increase in backing angle from the geostrophic direction would have implied reduced levels over N.Wales and higher values over S.England. Nevertheless the trajectories, predicted concentrations, and times of arrival of activity are in reasonable agreement with observation.

APPLICATIONS OF THE MESOS MODEL

The MESOS model is currently being applied to a few widely distributed hypothetical sites in European Community countries. Illustrative results are given here for MOL which is conveniently central in the study area. Figure 2 shows frequency distributions of different degrees of contamination by dry deposition from 24 hour

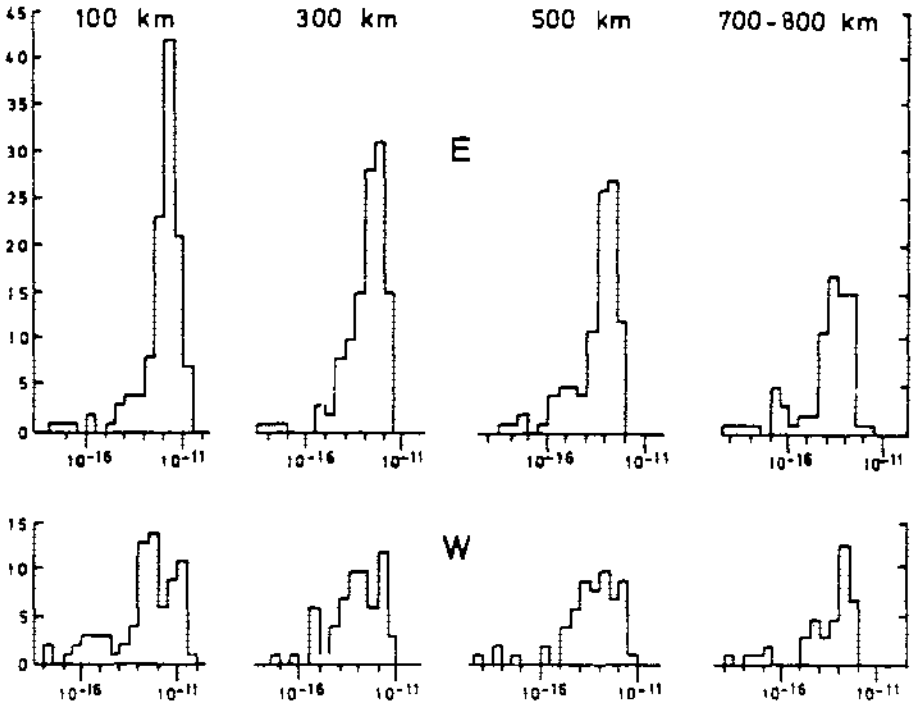

Figure 2. Frequency distributions of dry deposition for hypothetical releases of 1 Ci of I131 over 24 hours from MOL. Dry deposition (horizontal axis) is divided into half decade intervals in Ci m^{-2}.

releases of I131 at 4 distances East and West from MOL. For this nuclide 0.3 cm s^{-1} was assumed for the basic dry deposition velocity, and a washout coefficient $\Lambda=1.6.10^{-4}$ $J^{0.8}$ where J is the rainfall rate in mm/h.

Figure 3 shows contours of estimated annual wet deposition per Ci of I131 released. The distribution of dry deposition is similar to that of atmospheric concentrations, but wet deposition is greater to the N. and E. and lower to the S. where dry anticyclonic trajectories contribute more. Rainfall patterns are important; for example, orographic rain over Norway.

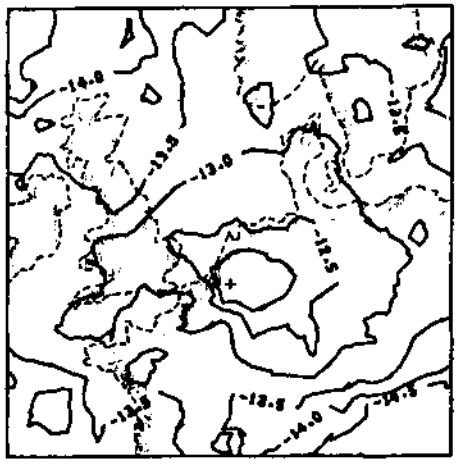

Figure 3. Contours of $\log_{10}$ (annual wet deposition in Ci m^{-2}) per Ci of I 131 released from MOL as a notional release site.

CONCLUSION

MESOS provides a more realistic model of atmospheric dispersal over long distances, and application to the 1957 Windscale incident produces encouraging comparison with measurements. For short-term releases MESOS is yielding useful statistics for hypothetical accident studies e.g. across a frontier. Data on population and food production supplied will enable us to estimate collective dose and its geographical distribution for routine continuous releases.

ACKNOWLEDGEMENTS

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COMPARISON OF THE ATMOSPHERIC DISPERSION CALCULATION WITH THE MEASUREMENTS OF γ-RADIATION LEVELS FROM AR-41 RELEASES

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The purpose of the work was to check the accuracy of a simple gaussian atmospheric diffusion model for a complex terrain condition. In the frame of the existing surveillance system, the annual gamma doses due to Ar-41 emission from a research reactor have been measured in the environment. The radiation level measurements were done mainly with sensitive G-M detectors.

The detectors were placed in the prevailing wind directions up to a distance of 4 km where villages (points W, D, V, R in Fig. 1) are located.

To supplement these measurements, TLD (CaF_2 (Dy)) detectors were used at two locations and for a short period, one high pressure ionisation chamber (RSS-111) was employed. The meteorological data (wind velocity, direction and atmospheric stability class) were recorded continuously during the whole year from December 1975 to November 1976. These data are shown in Table 1. The prevailing winds are from N-NE direction with mostly inversion condition, and from S-SW + NE directions with more neutral atmospheric stability. The winds from other directions (E-SSE) are very infrequent partly subject to the topographic situation, and hence are not listed in the Table. It can also be seen that during that year, the atmospheric stability classes were 45 % neutral (Pasquill D) and 39 % stable (Pasquill categories E and F).

The total emission of Ar-41 was 241'000 Ci at an average effective height of 100 m.

The computer code AIREM (1) was used to calculate the atmospheric dispersion. This code is based on a simple sector averaged gaussian diffusion equation for long-term average calculations. For the calculation of cloud γ-dose rates the finite extent model (Computer Code EGAD (1)) was used.

RESULTS

The measured annual doses at various locations are compared with the calculated values, which are shown in the following Table 2.

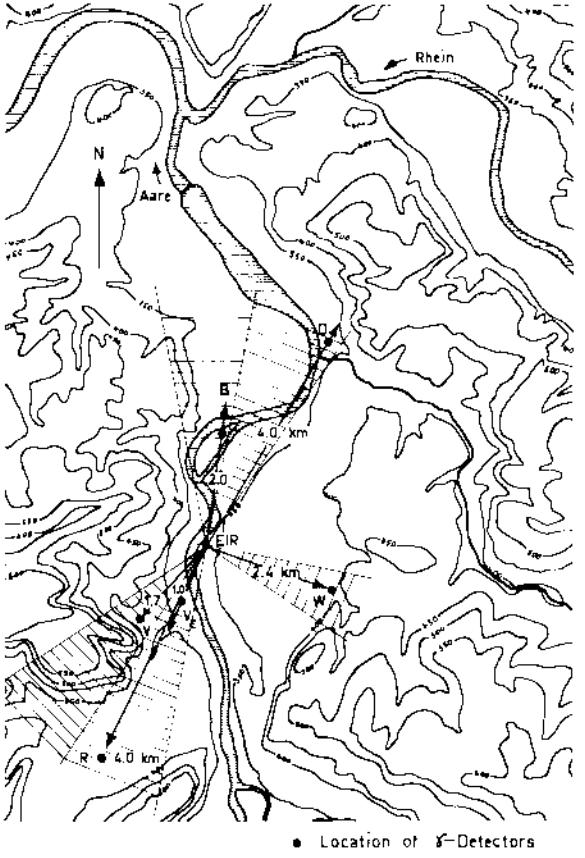

Fig. 1 Transport Directions and Sectors

Wind-sectors	Annual Frequ. %		Wind velocity (m/s) and frequency (%) for different stability categories				
			A	B	D	E	F
N	10.0	m/s	2.0	2.0	2.0	1.7	2.0
		%	0.5	0.7	3.2	5.0	0.6
NNE	18.7		2.0	2.0	2.0	1.7	2.0
			0.9	1.3	6.0	9.3	1.2
NE	18.9		2.5	2.5	2.5	2.0	2.0
			1.5	2.2	8.2	6.6	0.4
ENE	5.2		4.5	4.5	4.5	2.7	1.8
			0.6	0.8	2.4	1.3	0.1
S	8.4		3.0	3.5	3.0	1.8	-
			0.3	0.6	5.2	2.3	-
SSW	15.4		3.0	3.0	3.0	1.8	-
			0.5	1.8	8.2	4.9	-
SW	9.0		1.5	1.8	3.0	1.7	1.5
			0.5	0.7	4.3	3.3	0.2
WSW	5.8		3.7	3.8	4.0	2.0	1.5
			0.7	0.7	2.9	1.3	0.2
W	5.7		3.8	3.8	4.0	2.0	1.5
			0.2	0.7	2.9	1.3	0.1
WNW	1.3		3.8	3.7	4.0	2.0	1.5
			0.2	0.1	0.7	0.3	<0.1
NW	0.8		2.5	2.5	3.0	2.3	-
			0.1	0.1	0.3	0.3	-
NNW	0.8		2.5	2.5	3.0	2.3	-
			0.1	0.1	0.3	0.3	-

Table 1 Meteorological data

Table 2 Comparison of calculated to measured annual dose in the environment of EIR

Point of location	Detector	Downwind distance km	Measured annual dose (mrem)	Calculated annual dose (mrem)	Ratio of calculated to measured dose
Point V_E	TLD(CaF_2(Dy))	1	20	69	3.4
Point V	G.-M.	1.7	3.7	31	8.4
Point B	G.-M.	2	5.5	16	2.9
Point B	TLD(CaF_2(Dy))	2	9	16	1.8
Point W	G.-M.	2.5	0.22	2.1	9.6
Point D	G.-M.	4	0.78	8.2	10.5
Point R	Ionisation chamber RSS-111	4	0.32	0.84	2.6

The results show that at fairly flat locations (V_E, B, R) the ratios of calculated to measured doses vary between 1.8 to 3.4. However, it should be pointed out that there is a difference between the measured values at the location B where the dose was measured with two kinds of detectors (G.M. and TLD). At point R the measurments were made with a high pressure ionisation chamber over a period of only about 6 weeks.

Due to the topographic effects, at locations V, W and D, located in front of hills with elevations of 100 to 250 m, the results from the model show an overestimation of the experimental results. However this was expected by the authors because advective winds are not channelled in the valley due to the modest elevation of the ridges. The air masses will be lifted over the ridges, a problem which has not been sufficiently investigated. A deviation of wind direction due to Black Forest (north of the Rhine)and Jura Range (W-E) from straight line flow can also not be excluded.

There is no significant difference between the calculated values due to Pasquill's dispersion parameters and the dispersion parameters measured at Jülich, West Germany (2), as this difference is small compared to the disagreement between the model and the measurment.

It can be concluded that for long-term dispersion calculations there is a good agreement within a factor of 2 to 3 between the model and the measurements in the fairly flat terrain within the valley (N/NNE and S/SSW).

Comparable results were found by the comparison of shortterm diffusion calculations with the measured dose rates due to short-term emissions of Ar-41 (Refer to earlier paper (3)).

Acknowledgement: The authors thank Mr. W. Brunner (Federal Institute for Reactor Research) for his patience in processing the data used for this paper.

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REALISTIC ENVIRONMENTAL EXPOSURE CALCULATION FOR A MULTI-SOURCE FACILITY

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The annual radiation exposure caused by a nuclear facility usually is calculated from the measured release volumes of the individual nuclides and the so-called long-time diffusion factor, which results from the meteorological statistics of the site. This procedure is performed to make two assumptions: firstly, the release rates of the radioactive substances are quasi-continuous, and secondly, the superposed short-time diffusion factors of the changing weather conditions during the time of interest are comparable with the long-time diffusion factor. Because of their operational condition, these assumptions cannot be fullfilled by all nuclear installations.

Therefore we have designed a model and a computer code to calculate the radiation exposure under changing weather conditions and instantaneous release rates. This model is a multi-source model, i.e. the exposure due to the single installations will be superposed to the real location within the nuclear facility.

MODEL AND METHODS

The short-time diffusion factors are calculated by the hourly measured release volumes, the wind direction, the diffusion category and the mean wind velocity in emission height. The calculation method used herein is the Gaussian plume model, the parameters of which were measured by us for two emission heights (1). These short-time diffusion factors are superposed due to the real location of the sources to receive the environmental concentration distribution of each nuclide. Within this method quasi-continuous release rates can be described as well as instantaneous release volumes.

In order to calculate the radiation exposure of the total body and the other organs the following exposure pathways are considered: gamma-submersion, beta-submersion, soil radiation, inhalation and ingestion. In the case of the first four pathways the usual calculation methods are taken into account. The ingestion model is modified because of the instantaneous character of the diffusion calculation. The uptake from soil is derived from releases

throughout the whole year. In this term long-lived nuclides released in former years are also regarded. In our model the direct foliar deposition is derived only from really released volumes during the vegetation period. Based on a statistic time-dependance of the individual releases during the growing season of the different relevant foods, this term of our model includes the usual ingestion model in the case of uniform release rates (2).

EXAMPLE

As an example we present the annual radiation exposure of the Juelich Nuclear Research Centre in the year 1978. The Research Centre comprises several individual sources of emission. Depending on their operational condition, these individual installations release radioactive substances in one part in a quasi-contineous manner, and in another part as single emissions. The composition of the airborne radioactive substances differs very much. The main sources are listed in table 1.

Table 1. The airborne radioactive releases of the main installations of the Juelich Nuclear Research Centre in the year 1978

installation	nuclides	release in 1978 (Ci/a)	
reactor FRJ-1	AR 41	140^{+}	
	J 131	$1.1\ 10^{-4+}$	$2.5\ 10^{-5++}$
	HG 203	$1.8\ 10^{-2+}$	
reactor FRJ-2	AR 41	130^{+}	
	H 3	199^{+}	
	J 131	$1.6\ 10^{-4+}$	$5.6\ 10^{-6++}$
reactor AVR	H 3	399^{+}	
hot cells	KR 85	2.6^{+}	
	J 131	$3.5\ 10^{-3+}$	
fuel cells lab.	KR 85	1.6^{+}	
	J 131	$1.7\ 10^{-3+}$	
chemic lab.	J131	$4.2\ 10^{-3+}$	$3.1\ 10^{-4++}$

\+ release volumes during the whole year
++ release volumes only during the vegetation period from 1.4.1978 til 31.10.1978

During that year the release rates of the reactors FRJ-1 and FRJ-2 were in the main quasi-continueously, whereas the hot cells, the fuel cells lab. and the chemic lab. had irregular emission volumes. The main part of the tritium emission of the reactor AVR took place during an incident.

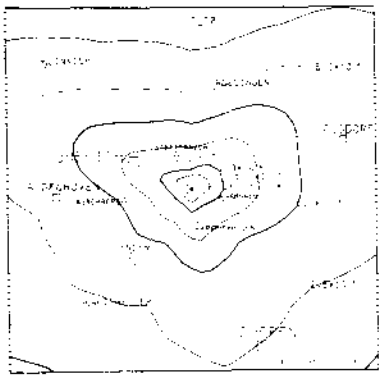

Figure 1

Annual dose of the total body from gamma-submersion

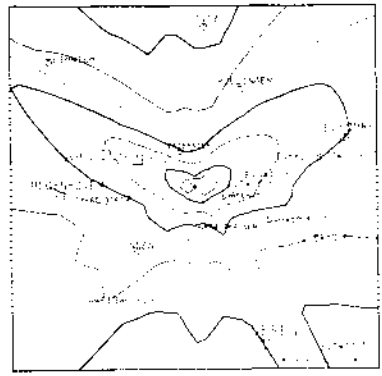

Figure 2

Annual dose of the total body as the sum of all relevant exposure pathways

Figure 1 shows the annual total body dose from gamma-submersion only, whereas figure 2 describes the annual total body dose as a sum of all relevant exposure pathways. The plotted isodoses of this last figure reflects the tritium incident of the reactor AVR and the predominant two weather conditions at that time. The curves are marked with the corresponding dose values in mrem per year. The asterisk describes the Nuclear Research Centre. The area of the map is 26 km by 26 km.

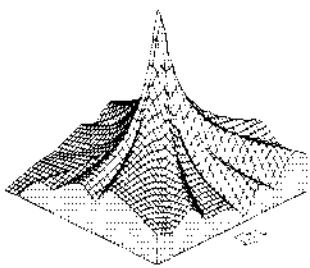

Figure 3

Relief-plot of the thyroid ingestion dose of the infant by J 131

Figure 3 shows the thyroid ingestion dose of infants by J131 only. The shape of the short-time diffusion factors are seen due to some few single emissions.

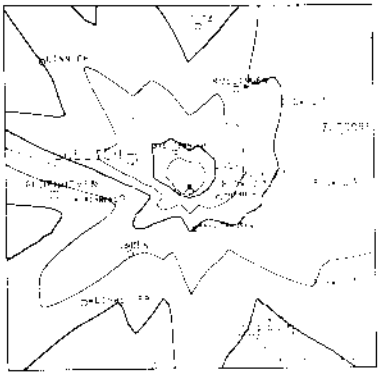

Figure 4

Inhalation dose of the thyroid of the infant by J 131

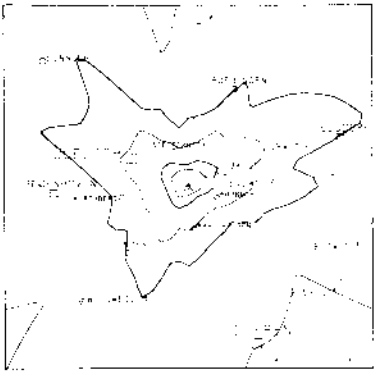

Figure 5

Ingestion dose of the thyroid of the infant by J 131

The figures 4 and 5 show the thyroid inhalation dose and the thyroid ingestion dose of infants by J 131. The inhalation dose in figure 4 includes also the main release volumes of J 131 during the winter period (see table 1). Because of the short life time of J 131 these release volumes had not to be taken into account for the ingestion dose calculation seen in figure 5. These two figures demonstrate very clearly the use of our model and that it is impossible to calculate the doses with the long-time diffusion factor.

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THE IMPORTANCE OF PLUME RISE IN RISK CALCULATIONS

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In certain circumstances, gaseous effluent emitted from a nuclear reactor may be sufficiently hot to allow the possibility of considerable plume rise. The Rasmussen report[1], for example, shows that, in the category of accident known as PWR1, the rate of release of heat can be 100 MW or so. In the context of the commercial fast reactor, it is plausible that large 'notional' releases such as those discussed in ref. (2), with up to 10% of the core vaporizing and escaping to the atmosphere, would be accompanied by the burning of several tonnes of sodium and the liberation of several hundred gigajoules of heat of combustion. Thus the consequences of such large releases cannot adequately be predicted in the absence of a convincing plume rise model.

RESUMÉ OF EXISTING MODEL

Refs. (3) and (4) contain a description of a systematic scheme for the calculation of plume rise, the elements of which are as follows: The basic equations are taken from the work of Gifford[5] and are modified to take account of the presence of the turbulent atmospheric boundary layer as described by Briggs[6]. Gifford's work gives predictions of the height of rise and radius of a continuous plume as a function of travel time or distance downwind, so long as the turbulence generated within the plume is so vigorous that the atmosphere outside may be regarded as being non-turbulent. During this stage, ground level concentrations under the rising plume are very small. Briggs discusses criteria for determining when plume rise effectively ceases so that the dispersion of the plume is governed thereafter by the action of atmospheric turbulence alone. These criteria include (a) the equating of turbulence energy dissipation rates within and outside the plume to determine when the turbulence within the plume ceases to be more 'vigorous' than that in the atmosphere; (b) the comparison of the expected rate of plume rise with typical updraught and downdraught velocities, in conditions in which the atmospheric turbulence is highly convective; (c) rise into a stable boundary layer until the temperature of the plume no longer exceeds that of the atmosphere, and (d) contact with an overhead inversion lid.

LIFT-OFF

A plume emitted from a reactor during an accident may first be mixed throughout the turbulent building wake. The question, "does such a plume lift cleanly off the ground, or is its buoyancy partially or totally destroyed?" has not yet been answered unambiguously. Briggs[7] has given a speculative prescription for determining whether or not such a plume will rise. Firstly, if the density difference between a plume and its surroundings is $\Delta\rho$ and its depth is h, then a typical buoyancy induced velocity is $V_b \propto \sqrt{g\Delta\rho h}$ where g is the acceleration due to gravity. Secondly, the friction velocity u_* is characteristic of the rate of spread of a passive plume. If $u_* >> V_b$, it is expected that atmospheric turbulence will be dominant, whereas

if V_b is much the larger, then buoyancy effects should be more important. Considerations such as these led Briggs to propose a non-dimensional parameter $L_p = (gh\Delta\rho)/\rho_a u_*^2$, where ρ_a is the density of the air. The critical value of L_p above which the plume is expected to lift off cleanly, is about 2.5 and Briggs estimates that there is at least a factor of 4 uncertainty in this value.

Figure 1 shows that rate of heat release, Q MW, required to ensure that lift-off takes place when the windspeed at a height of 10 metres is u ms^{-1}. Making the pessimistic assumption that the critical value of L_p is 10 and assuming typical values of 100m for w and 0.1m for z_o, then curve 3 shows that even for a rate of heat release of 10 MW, plume rise is not expected if $\bar{u}<4$ ms^{-1} - this observation illustrates the important point that very large rates of heat release are needed for current methods to predict lift-off. Of the various accident sequences defined by Rasmussen, for example, only three (PWR1a, PWR2 and BWR1) liberate heat at a rate in excess of 10 MW. In general, it is not sufficient merely to prove that a radioactive release is accompanied by substantial rates of heat release, Q: unless Q is very large indeed it cannot be assumed that there will be plume rise unless it has also been shown that the effluent does not mix throughout the reactor building wake. Preliminary indications from recent wind tunnel experiments are that the critical value of L_p is bigger than expected, perhaps as much as 30; the effect of this is shown on curve 1. The Safety and Reliability Directorate is sponsoring wind tunnel experiments designed to investigate the importance of the parameter L_p.

RAIN

A second phenomenon that can act so as to destroy or reduce the benefits of plume rise is rain, which can wash radioactive material out of the plume, deposit it on the ground and cause subsequent relatively high rates of irradiation by γ-rays.

The structure of a typical warm frontal rainstorm is described in ref. (8) and four levels of rainfall are identified. The largest is contained within a synoptic boundary and typically covers several tens of thousands of square kilometers and lasts for several hours to a few days. Next, there are large mesoscale areas (LMSA), roughly one to each 12,000 km^2 and occupying one third of the storm area. These persist for one to twelve hours and, if the rainfall rate in the synoptic area is R, that in the LMSA is 2R. Each LMSA contains five or so small mesoscale areas (SMSA) which last for about an hour, occupy about one ninth of the storm area and have an average rainfall rate 4R. Finally, each SMSA contains cells of area about 10 km^2, occupying about 2% of the storm area with typical rainfall rate 25R and a lifetime of a few minutes.

If a plume travels for a time t during a period of constant rainfall, and if the quantity of radioactive material airborne at time t = 0 is q, then the quantity remaining airborne at time t is $q\,e^{-At}$, where A is the washout coefficient. If the rainfall rate is R mm hr^{-1}, ref. (8) gives a phenomenological relationship between R and A, namely $A = CR$ s^{-1} where C is typically 10^{-4} s^{-1} $hr\ mm^{-1}$ for a warm frontal storm.

As an example, the total inhaled plus external dose to the bone marrow received as a result of the release of quantity of long-lived

^{137}Cs is shown on Fig. 2, for a range of assumptions about plume rise and rainfall. This is the dose that would be used in the calculations of early deaths[1] and is given by (½(7-day + 30 day inhaled dose) + external dose accumulated in 7 days, with an assumed shielding factor of 0.5). In all cases, the release takes place into a building wake 100m wide by 50m high and persisting for 5 building heights downwind. The dry deposition velocity is taken to be 0.003 MS^{-1}.

Curves (1,2) and (1,3) show the difference made by plume rise when there is no rain. Here, contributions from activity deposited on the ground are relatively unimportant. If there is rain, ^{137}Cs is brought down to the ground and, over a period of seven days, the external radiation from this deposited material makes the largest contribution to the dose and is the same whether plume rise takes place or not, see curves (1,4) and (2,4). Curves (1,5) and (2,5) show that 'hot spots' of deposited activity can be caused if the plume runs into heavy rain; again, the levels of activity are independent of whether plume rise takes place or not.

In conclusion, if there is no rain, plume rise can give a substantial reduction in the radiation doses likely to be received by people living in some areas downwind of the source. If there is rain, levels of deposited activity are independent of the plume rise so that areas of unacceptably contaminated land are not reduced by taking plume rise into account. If rapid evacuation is possible, however, the accumulated external radiation doses can be much reduced and, as in cases (1,2) and (1,3), some benefits will be seen from plume rise.

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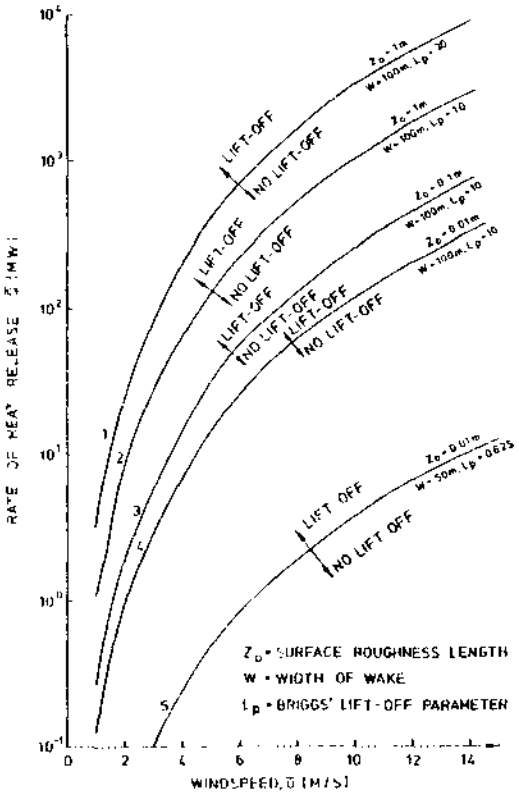

FIG 1 CRITICAL VALUES OF $\bar{Q}$ AND $\bar{u}$ FOR LIFT-OFF - CONTINUOUS RELEASE

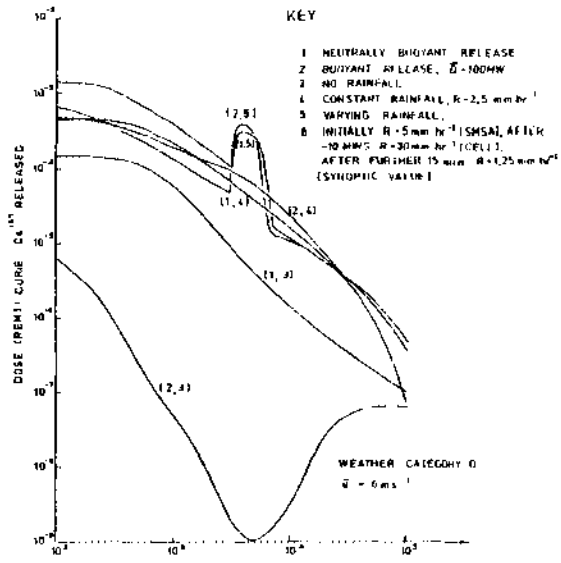

FIG 2 DOSE COMMITMENT AS A FUNCTION OF DOWNWIND DISTANCE

STATISTICAL STUDIES ON THE LIMITATION OF SHORT-TIME RELEASES FROM NUCLEAR FACILITIES

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The licenceable annual release volumes from a nuclear facility are calculated, on the assumption of a constant release rate, from the dose limit values, the radioecological transfer and dose factors, the release height, and the site-specific long-time diffusion factors which are dependent on the exposure pathway. In this paper, a statistical investigation relating to the necessity of limiting individual releases is being presented. In this connection, it is being checked whether a limitation of the hourly release volume will also be required in addition to the limitation of the daily release volume to 1/100 of the annual release volume as stipulated in the directive /1/ of the Federal Republic of Germany.

For the statistical investigation it is being assumed that the same radioecological and biological parameters can be used for releases featuring a constant release rate and for individual releases, so that only the different diffusion factors are to be taken into account. For calculating the radiation exposure, the following exposure pathways must be considered: gamma submersion, beta submersion, soil radiation, inhalation and ingestion. The exposure due to beta submersion and inhalation is proportional to dry deposition on the soil or on vegetation (fallout). For the determination of soil radiation and ingestion doses, the deposition due to precipitations (washout) is also required in addition to the fallout. It is therefore sufficient to compare the long-time diffusion factors for fallout, washout and gamma submersion with the corresponding quasi-longtime diffusion factors. As is illustrated in detail in /2/, the quasi-diffusion factors are ascertained by superposition of the short-time diffusion factors (hourly values), averaged over the 30° sector and featuring, in each specific case, release rates of 1/100 of the annual release volume for 100 statistically selected hours. This is being carried out using the meteorological statistics of Jülich for the year 1977. Each quasi-longtime diffusion factor is calculated 25 times with different data records of always 100 hours, so that a statistical statement is possible.

In the case of fallout and gamma submersion, calculations were carried out for the release heights of 50, 100 and 200 m. For the formulation described in /1/, it may be shown /2/ that the ratio of quasi-longtime to longtime washout factor does not depend on the release height and source distance.

The results summarized in Table 1 show that the quasi-longtime diffusion factors vary by the respective long-time diffusion factors. The arithmetic mean values coincide (except for washout) with the long-time diffusion factors. In the case of washout, the geometric mean value of the quasi-longtime washout factors corresponds better with the long-time washout factor. This may be explained by the larger variations naturally resulting from the fact that, on an average, it rains less than ten times during the 100 hours which enter into each individual calculation of the quasi-longtime washout factors. In the bottom line of the table on cumulative frequency distribution, the standard deviation of the quasi-longtime diffusion factors from their mean value corresponding to the associated long-time diffusion factor is shown. It may be observed that the standard deviations for all exposure pathways and release heights (except for washout) range between 34 and 45 per cent. This means that the quasi-longtime diffusion factor only exceeds the double long-time diffusion factors in less than 1 per cent of cases (3σ-error). In the case of washout, the standard deviation is four times as high. This would lead to the conclusion that, after all, the long-time washout factor is exceeded by the factor 5 in less than 1 per cent of cases. For, the high upward deviations result from the hours involving high precipitation intensities. For a high precipitation intensity, however, the proportional formulation of washout coefficient and precipitation intensity is overconservative according /4/ and /5/. A proportionality constant fixed for a mean precipitation intensity overestimates the washout coefficient for a high precipitation intensity approximately by a factor of 5, so that the maximum underestimation due to the long-time diffusion factor should not be rated more pessimistically for washout in this range than for any of the other exposure pathways.

However, this possible additional exposure due to short-time releases does not occur in reality because

- not all of the individual releases occur within one hour
- a considerable portion of total releases is made up by quasi-continuous releases, so that short-time releases are reduced
- the number of short-time releases never corresponds to the theoretically possible value of 100
- in the case of several emission sources at one site, as a rule, the releases via the individual stacks are restricted, so that the individual release corresponds to less than 1/100 of the total release of the site
- for long-lived nuclides, the accumulation over 50 years to be performed according to model /1/ will lead to a compensation of the individual maxima falling on different wind directions and source distances.

In summary, it may therefore be concluded that a limitation of short-time releases to 1/100 of the permissible annual release value under the conditions of /1/ is considered to be sufficient for a realistic assessment. If a more conservative approach is preferred, a limitation of individual releases to 1/200 of the annual value would come into consideration. An application of the long-time diffusion factors to individual releases thus delimited would lead to a minor underestimation of the environmental impact calculated according to /1/ in less than 1 per cent of cases.

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Table 1 Cumulative frequency distribution, mean value and standard deviation of the ratio of quasi-longtime to long-time diffusion factors at the maximum of the long-time diffusion factor

	cumulative frequency (%)						
exposure pathway	fallout			gamma submersion			washout
release height	50 m	100 m	200 m	50 m	100 m	200 m	
ratio smaller than							
0.2	0	0	0	0	0	0	12
0.4	8	12	12	0	0	0	28
0.6	24	20	16	0	12	28	40
0.8	40	48	44	16	40	44	48
1.0	60	80	80	68	64	56	60
1.2	76	88	88	72	64	64	60
1.4	92	96	96	80	80	80	68
1.6	100	100	96	84	84	88	72
1.8			96	96	88	96	72
2.0			96	96	96	100	76
2.2			100	96	100		76
2.4				96			76
2.5				100			76
3.0							92
4.0							92
5.0							96
6.0							96
7.0							100
mean value arithmetric	0.90	0.84	0.84	1.08	1.07	0.99	1.40
geometric	-	-	-	-	-	-	1.01
standard deviation	0.34	0.36	0.38	0.42	0.45	0.44	1.58

WASHOUT AND DRY DEPOSITION OF ATMOSPHERIC AEROSOLS

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INTRODUCTION

Essentially all air pollution is eventually cleansed from the atmosphere by the natural processes generally referred to as precipitation, scavenging and dry deposition. The purpose of this report is to present below-cloud rain scavenging rates or washout coefficients and the deposition velocities to rough surfaces on the earth for submicron aerosols.

The washout coefficient $\Lambda(d)$ for a monodisperse aerosol with a diameter d is the integrated product of the flux densities of the rain drops, their cross sections, and their collection efficiencies $E(d,d_{Tr})$:

$$\Lambda(d) = \frac{\pi}{4} \int_0^\infty \frac{\partial N_{Tr}}{\partial d_{Tr}} v_{Tr} E(d,d_{Tr}) d_{Tr}^2 \partial d_{Tr} \qquad (1)$$

The flux densities of the drops are obtained from the number distribution for raindrops $\partial N_{Tr}/\partial d_{Tr}$ and their terminal velocities v_{Tr}. The collection efficiencies of the raindrops were measured in dependence on particle (d) and droplet size (d_{Tr}) (1).

The major mechanisms of the dry particle transport from an air flow to a rough surface are eddy diffusion, sedimentation by gravity, Brownian diffusion and inertial forces. The deposition velocity v_g, is defined as the flux to a surface F devided by the aerosol concentration c: $v_g = F/c$. The influence of particle size, roughness of the surface and wind velocity on the deposition velocity were studied in a wind channel (2).

EXPERIMENTAL TECHNIQUES

The collection efficiencies of the raindrops were studied in a wind channel using spheres as simulated drops. The particles were labelled with radioactive Pb-212 and sucked into the wind channel, where the wind velocities agreed with the terminal fall velocities of the rain drops simulated by spheres. The wind velocity, the precipitated activity on the sphere and the activity concentration in air were measured. With this technique collection efficiencies were measured for spheres in the size range of 0.35 - 7 mm and for the monodisperse aerosol particles with diameters between 0.04 and 4µm (standard deviation 3 - 7 %).

For the deposition studies the monodisperse radioactive particles were led into a second wind channel with a cross section of 20 cm x 20 cm, where the floor of the working section (2 m long) was completely covered with the test surface. The used test surfaces consisted of plants (barley), filter paper and simulated grass, constructed by metal cylinders (2 mm diameter) planted in a plasticine layer. The fractions of particles filtered by the roughness elements and deposited on the substrate were measured separately. The height (H) and the density (n) of the roughness element were varied. From the measured wind profile the roughness length (z_o) and the friction velocity (u_*) were calculated.

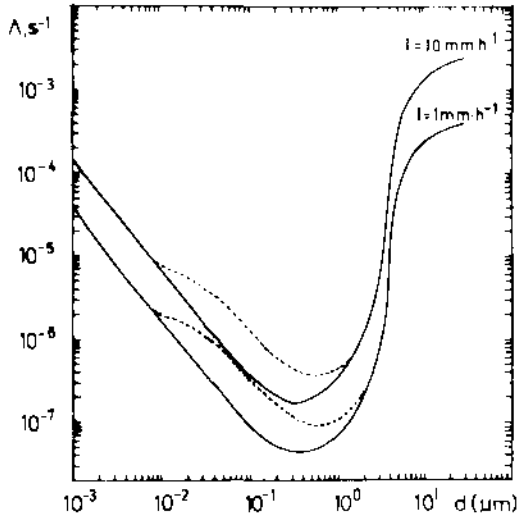

Fig. 1 Washout coefficients for neutral (——) and charged (- - -) raindrops as a function of the particle diameter for rainfall rates of 1 and 10 mm/h

RESULTS OF THE WASHOUT EXPERIMENTS

In a number of investigations the following parameters were studied:

1. The influence of drop shape, compared to the shape of the volume equivalent sphere on the collection efficiency was checked and found to be negligible (1).
2. The sticking probability of the aerosol (DES-droplets) up to 4 µm diameter on spheres (stainless steel) 0.37 - 7 µm diameter was measured to be unity (1).
3. The influence of the electric charge of the droplets for a normal rainfall and aerosols with Boltzmann charge equilibrium on the collection efficiency (electrostatic forces) and the washout coefficient, respectively, is negligible (Fig. 1, solid line). Only for typical charges of raindrops in the case of thunderstrom (3) there is a threefold increase of collection efficiency in relation

to collection without electric forces (Fig. 1, dotted line).
4. The collection efficiencies as a function of droplet and particle size were measured and washout coefficients (Fig. 1) for rainfall rates of 1 and 10 mm/h were calculated (equation (1)).

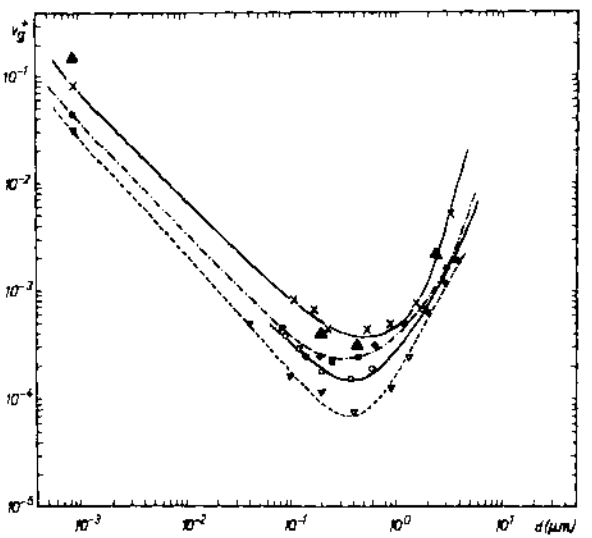

Fig. 2 Deposition velocities $v_g^+ = v_g/u_*$ of aerosols as a function of the aerosol diameter:
-x—x- Simulated grass n = 1,00 cm^{-2} H = 7 cm
-o—o- Simulated grass n = 0,25 cm^{-2} H = 7 cm
-•-•- Filter paper
-▽- -▽- Smooth metal surface
▲ ▲ Barcly n = 1,00 cm^{-2} H = 7 cm

RESULTS OF THE DRY DEPOSITION EXPERIMENTS

The experimental results of these investigation can be summarized (2):
1. The height (H) and the density (n) of the roughness elements have a great influence on the deposition velocities.
2. The particle fluxes to the roughness elements is much greater than to the substrate.
3. The measured v_g^+ values ($v_g^+ = v_g/u_*$) in dependence of particle size for relatively smooth surfaces (filter paper) are nearly the same as for grass or simulated grass surfaces between the surface structures and friction velocities (Fig. 2).

Table: The fractions of washout, rainout and dry deposition for different atmospheric aerosols

REMOVAL PROCESS	Activity size* distribution of Pb-210 $v_g = 3.9 \cdot 10^{-4}$ m/s $\Lambda = 5.4 \cdot 10^{-4}$ h^{-1}	Atmospheric** Aerosol (mass distribution (4)) $v_g = 7.6 \cdot 10^{-3}$ m/s $\Lambda = 7.6 \cdot 10^{-2}$ h^{-1}	Aerosol of*** a power plant plume; wind-velocity: 6 m/s $v_g = 3.9 \cdot 10^{-4}$ m/s $\Lambda = 5.4 \cdot 10^{-4}$ h^{-1} source height: 100 m distance from source: 2500 m
Dry FALLOUT $u_* = 40$ cm/s	5.3 %	87.4 %	99.3 %
WASHOUT rainfall rate: 1 mm/h rain: 600 mm/a	0.2 %	9 %	0.7 %
RAINOUT rainfall rate: 1 mm/h rain: 600 mm/a cloud depth: 2km cloud height: 2 km rainout coeff.: 1.1 h^{-1}	94.5 %	3.6 %	-

* Activity size distribution of the short lived radon daughters is assumed (4); The vertical Pb-210 concentration distribution is after MOORE et al. (5)

** The vertical concentration distribution is after WIGAND (6)

*** The activity size distribution of the short lived radon daughters is assumed (4)

A COMPARISON OF THE REMOVAL PROCESSES

The particle fluxes of different atmospheric aerosols to the earth surface under consideration our experimental results were calculated. The fractions of washout, rainout and dry deposition for the mass distribution of the natural aerosol, the Pb-210 aerosol and for a radioactive aerosol of a power plant plume are listed in the table.

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THE FUNCTIONAL DEPENDENCE OF THE TOTAL HAZARD FROM AN AIR POLLUTION INCIDENCE ON THE ENVIRONMENTAL PARAMETERS

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A general case of release to the atmosphere of a pollutant is considered. Various chemical and radioactive hazards may result from inhalation of the pollutant, deposition, resuspension, ingestion and external and internal radiation, if the pollutant is radioactive. According to the ICRP-26 recommendations, the total risk summed over all pathways and tissues should not exceed a certain limit.

The total dose received (D_t) is a monotonous function of the source strength. However, in general, it does not vary monotonously with some of the physical processes involved. For instance, an increase in the deposition velocity results in larger deposition along the cloud's trajectory, which reduces the amount of activity reaching downwind distance X and thereby the cloud and direct inhalation dose. On the other hand, though less activity reaches X, more activity is deposited there (for higher V_d), increasing the doses from external radiation from deposited material, from inhalation of resuspension and from ingestion. Similarly, it will be shown that, taking into account previous deposition, D_t at X does not always increase with decreasing wind speed or with decreasing source height.

In the process of hazards evaluation one usually tends to estimate the processes involved conservatively so as to maximize the computed doses. The worst cases (which give maximum D_t) are not always easily identified (1). The present work helps to identify them. In addition a model of the total dose is presented and its variations are studied as a function of wind speed-$\bar{u}$, deposition velocity-V_d and source height-h. The value of each parameter giving the highest total dose as a function of the model's parameters is determined.

MODEL

This preliminary study is based on the simplest and widely used assumptions of an instantaneous, elevated point source; deposition is considered using the Chamberlain model through Van Der Hoven's curves (2). The external dose is calculated with the semi-infinite homogeneous approximation.

THE TOTAL DOSE EQUATION

$$D_t = \left(\frac{c\bar{u}}{Q}\right)\frac{Q_o}{\bar{u}}\left(\frac{Q_x}{Q_o}\right)\left(F_c + F_s^D V_d + B_r S_B^D\left(1 + \frac{F_r}{\lambda} V_d\right)\right)$$

$\left(\frac{c\bar{u}}{Q}\right)$ - normalized concentration at distance X.

Q_o - source strength.

$$\left(\frac{Q_x}{Q_o}\right) = \exp\left\{-\sqrt{\frac{2}{\pi}}\frac{V_d}{\bar{u}}\int_o^x \frac{\exp(-h^2/2\sigma_z^2)}{\sigma_z}\,d\zeta\right\}$$

σ_z-clouds vertical standard deviation. $F_C \equiv 0.25\ E$, E-γ energy, $F_S^D \equiv \frac{F_S}{\lambda} + \sum \omega_i F_I^i$, F_S-ratio of the γ dose from deposition to the surface contamination. λ-decay constant, ω_i-ICRP-26's weights, F_I^i-ratio of the dose from ingestion + drinking to the surface contamination, B_r-breathing rate, $S_B^D \equiv \sum \omega_i S_B^i$, S_B^i = specific inhalation dose for to organ i, F_r-resuspension factor.

METHOD

The dose equation was derived twice with respect to $\bar{u}$, V_d and h. Extremum and inflection points are identified, which enables to study the behavior of D_t.

RESULTS

A. Variation of D_T as a function of V_d:
Fig. 1 gives the results for $X < X_o$

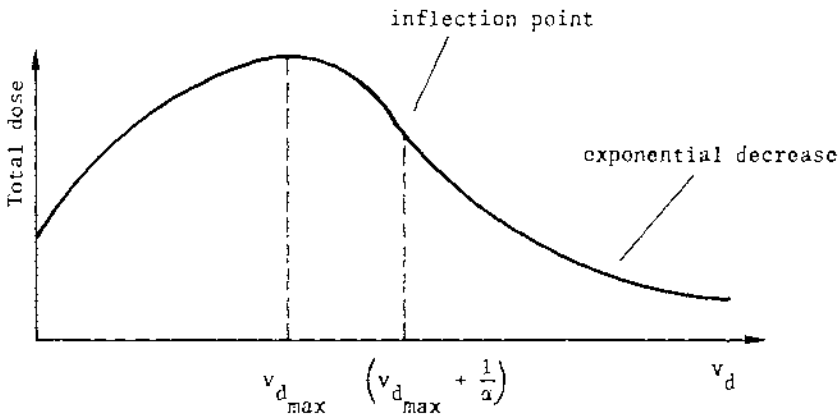

Fig. 1. Schematic representation of the variation of the total dose as function of the deposition velocity.

$$V_{d_{max}} = \frac{1}{\alpha} - \frac{1}{\alpha_o}, \quad \alpha \equiv \sqrt{\frac{2}{\pi}}\,\frac{1}{\bar{u}} \int_0^X \frac{\exp(-h^2/2\sigma_z^2)}{\sigma_z}\, d\zeta$$

$$\alpha_o \equiv \frac{B_r S_B^D \frac{F_r}{\lambda} + F_S^D}{F_C + B_r S_B^D} = \alpha\Big|_{X=X_o}.$$

For $X > X_o$ D_t decreases monotonously with increasing V_d.
Example: X = 9 km, stability D, $\bar{u}$ = 5 m/s, h = 10 m → α = 24 (from the Van Der Hoven curves). A very simple case: No γ energy → $F_C = F_S = 0$, no ingestion or drinking dose, only inhalation dose - $F_I^i = 0$. Taking $F_r = 10^{-4}m^{-1}$, $T_{\frac{1}{2}}$ - one week→α_o=87→$V_{d_{max}}$=3cm/s.

B. Variation of D_t with wind speed: Defining

$$\gamma \equiv \left(\frac{2}{\pi}\right)^{1/2} V_d \int_0^x \frac{\exp(-h^2/2\sigma_z^2)}{\sigma_z} d\zeta \quad , \quad \text{we receive Fig. 2:}$$

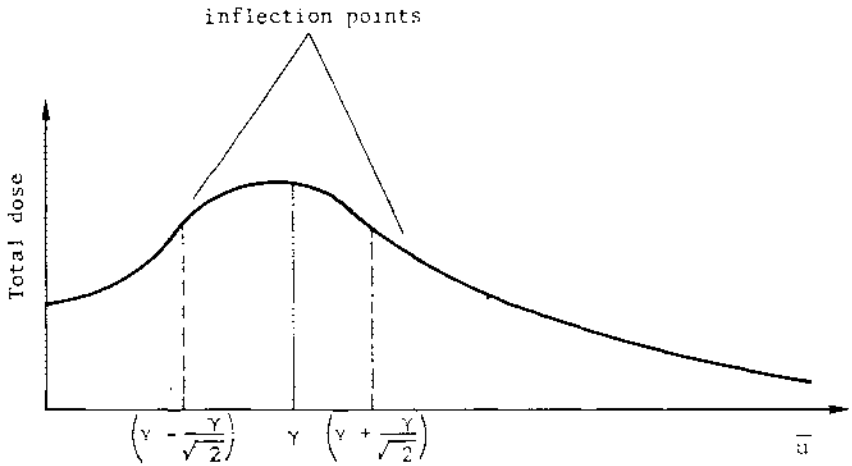

Fig. 2. Schematic representation of the variation of the total dose as a function of wind speed.

Example: The conditions are the same as above - D_t is maximal for $\bar{u} = \gamma = 3.6$ m/s, for $V_d = 0.03$ m/s.

C. Variation of D_t with h: A single maximum D_t exists when:

$$\frac{1}{\delta\sigma_z(X)} = \int_0^\pi \frac{\exp(-h^2/2\sigma_z^2)}{\sigma_z^3} d\zeta \qquad [1] \qquad ; \qquad \delta \equiv \sqrt{2}\,\frac{V_d}{\pi\,\bar{u}} \quad .$$

The numerical solution for stability D is given in graphical form in Fig. 3.

Example: same as above, for $V_d = 0.03$ m/s, $\bar{u} = 3.6$ m/s $h_{max} \approx 80$ m.

CONCLUSIONS

A. D_t does not vary monotonously as a function of V_d or $\bar{u}$ or h. A maximum total dose exists for certain values of these variables, which is a function of the stability, X, the other parameters of the model and the properties of the pollutant.

B. The worst cases (and corresponding $D_{t_{max}}$) can be determined and used in hazards evaluation.

C. Maximum D_t cannot exist simultaneously as a function of both $\bar{u}$ and V_d.

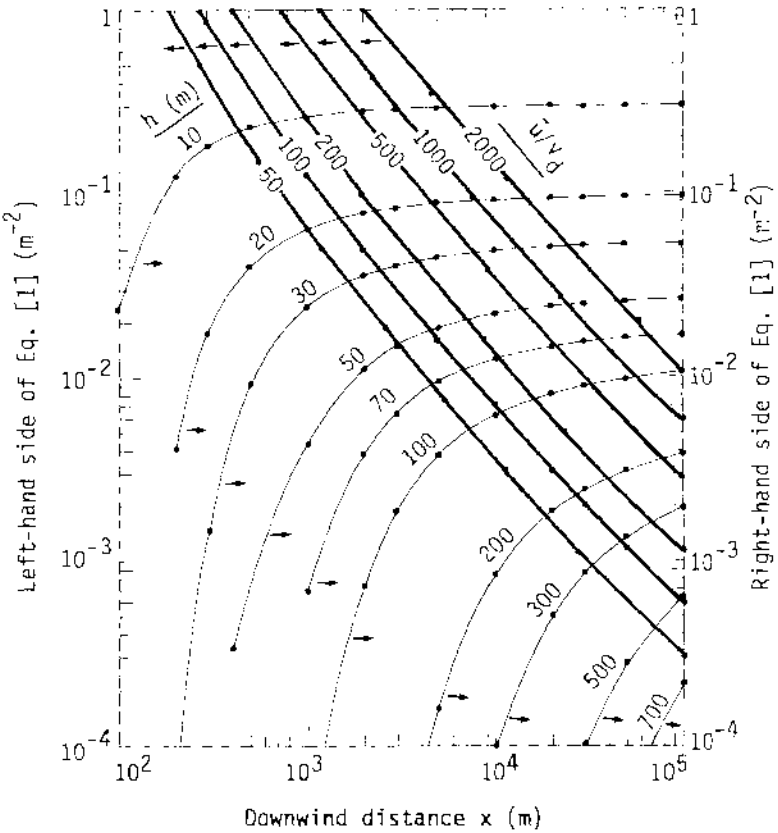

Fig. 3. Graphical solution of the equation for the height, which indicate the maximum D_t. The two sets of curves give the left and right hand sides of the equation for different $\bar{u}/V_d$ and h (respectively), as a function of X, for stability D.

D. In this preliminary study the simplest, widely used analytical model was used. Refinements may be incorporated when more information is obtained concerning the detailed analytical form of the dose equation.

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REDUCTION OF THE ENVIRONMENTAL CONCENTRATION OF AIR POLLUTANTS BY PROPER GEOMETRICAL ORIENTATION OF INDUSTRIAL LINE SOURCES

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Industrial line sources of air pollutants are frequently characterized by a relatively low height of release. Their contribution to the concentration of air pollutants in the environment may be significant and cause violation of the ambient air quality standards. Therefore, measures sometimes have to be taken to reduce the environmental concentration of air pollutants released by such line sources. Among the measures considered may be the reduction of the amount of pollutant at the source prior to release or the improvement of the atmospheric dispersion, either by increasing the height of release or, in the case of a line source, by changing the geometrical orientation of the individual sources.

A case study was undertaken of two line sources, one composed of 10 and the other of 20 individual sources. The height of release ranged from 15.7 to 39.6 m, with a uniform rate of release of a gaseous pollutant of 1 Ci/s for each source.

Average environmental concentrations of air pollutants were calculated for the original setting of the line sources and then compared with those obtained from other settings, in which the individual sources of the same physical height of release, were differently orientated geometrically. These concentrations were also compared with those obtained from a single source which released the total amount of pollutant (30 Ci/s) at a height of 70 m. The concentrations were calculated using a diffusion model in which the environmental conditions were represented by a three-dimensional matrix and were based on the Gaussian equation of the form (1):

$$\bar{\chi}(x_j,\theta_j) = \sum_{p=1}^{N_s} \sum_{r=1}^{N_w} \sum_{j=1}^{N_j} \frac{2.032 F_{pr}(\theta_j)\, Q_{ipr}(x_j)}{\sigma_p(x_j)\, u_r x_j} \exp\left(-\frac{h_{prj}^2}{2\sigma_p^2(x_j)}\right) \qquad (1)$$

where:
$\bar{\chi}(x_j,\theta_j)$ is the average ambient concentration, resulting from pollutant point sources j, at a downwind distance x, along a 22.5° arc θ, which includes both the source and receptor.

N_s, N_w, N_j are indices denoting, respectively, the number of atmospheric stability classes, wind velocity groups and point sources
$F_{pr}(\theta_j)$ is the fraction of the time during which the wind is in direction θ, for atmospheric stability class p and wind velocity group r
$Q_{ipr}(x_j)$ is the point source strength of a pollutant i, corrected for dry and wet deposition, occurring along a distance x
$\sigma_p(x_j)$ is the vertical dispersion coefficient at a distance x from source j, for atmospheric stability class p
u_r is the average velocity for wind group r
h_{pr} is the effective plume height for atmospheric stability class p and wind velocity group r.

A computer program, adapted for the Hebrew University CY74 computer, was written to solve equation (1).

Figure 1 shows the initial geometrical position of the two line sources and 16 ground level receptor points for which the average pollutant concentration was calculated.

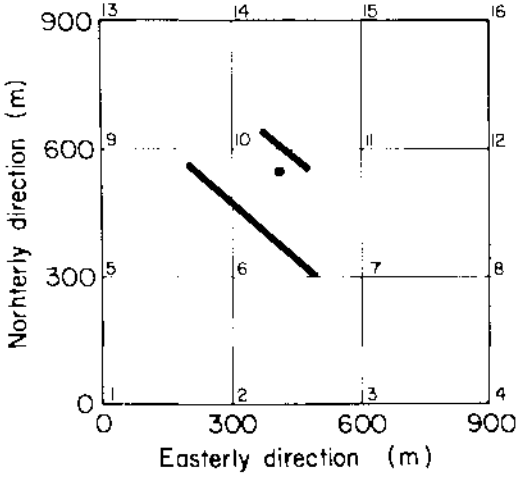

Figure 1. Initial geometrical position of the line sources and location of receptors 1-16 .

Line source
Rotation point

The 16 receptor points are distributed within a square 900 m x 900 m, at a distance of 300 m from each other. The computations of the average concentration of pollutant at the receptor points were performed for summer days, of characteristic joint atmospheric stability and wind velocity frequency (2).

The influence of the geometrical orientation of the line sources on the pollutant concentration at the 16 receptor points was investigated by veering both line sources from the original position 45° at a time around a given point A (Fig. 1). Figure 2 shows the average

pollutant concentration at the receptor point 3 as a function of the rotation angle of the line sources.

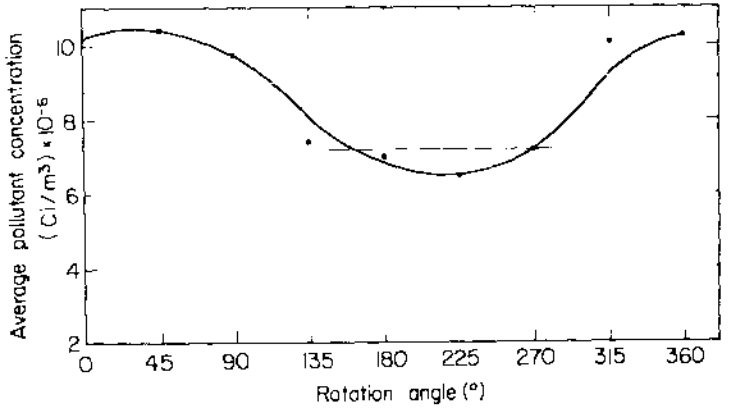

Figure 2. Average pollutant concentration at receptor point 3 as a function of the rotation angle of the line sources. The horizontal line shows the concentration of pollutant caused by a single source of 70 m height, the strength of which is equivalent to that of the line sources.

It is seen that at a given rotation angle, the average pollutant concentration is at a minimum at a certain receptor point. Similar results are obtained for all receptor points. The reduction of the pollutant concentration by a particular rotation of the line sources may attain values of up to about 50% as compared with the concentration obtained from the original position of the line sources.

Calculations were also made to compare the environmental pollutant concentration from given settings of the line sources, with the concentration caused by a single source of a height of 70 m, assuming that the strength of the single source is equivalent to that of the line sources. Figure 2 shows that at certain rotation angles of the line sources, the environmental concentration is lower even as compared with a single high source. In the case shown in Fig. 2 the minimum concentration obtained by rotating the line sources is about 10% lower than that caused by a 70 m high integrated source. However, for other receptor points and for other rotating angles, up to a 3-fold reduction of the average environmental concentration may be obtained by rotating the line sources, as compared with the concentration caused by a 70 m high integrated source.

The minimization of the average environmental concentration by rotating the line sources is different for each receptor point and calculations have to be performed to optimize the reduction of the pollutant concentration for all the receptor points in the area of interest, as related to each discrete point source of which the line sources are formed. The influence of a discrete point source on a given receptor point does not depend only on the reciprocal geo-

metrical configuration of the source and receptor. It also depends on the atmospheric conditions, such as wind parameters and atmospheric stability, of which the plume width and height are functions, and on the probability of the receptor point being within the sector of influence of the source. Because the probability function $F_{pr}(\theta_j)$ is a discrete statistical parameter, there is no analytical procedure to find the minimum average concentration for all receptor points as a function of the rotation angle of the line sources. Numerical calculations therefore have to be performed to optimize the configuration of the line sources in order to get the minimum average concentration for all the receptor points in the area of interest.

A given geometrical orientation of the line sources can thus be found which minimizes the pollutant concentration both as compared with the original setting of the sources and with a single higher source integrating all the given point sources. Considering the substantial increase in cost of augmenting the height of release of pollutants as a means of reducing the air pollutant concentration, determining the proper geometrical orientation of the line sources should be considered as an economical means of improving the environmental air quality.

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RESUSPENSION OF PLUTONIUM FROM CONTAMINATED SOILS

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INTRODUCTION

Resuspension factor (RF), defined as the ratio of concentration in air (Ci/m^3) to the concentration in top soil (Ci/m^2) is an important parameter useful for evaluating the inhalation hazards. Such factors were studied by various workers (1-4).

For desert conditions like Nevada Test Site areas, such factors for plutonium lie in the range of 10^{-5} to 10^{-7} m^{-1} at the beginning, decreasing subsequently to 10^{-9} to 10^{-10} m^{-1} due to weathering and downward movement of plutonium in soil (5-7). The study on resuspension factors from humid climatic soil conditions has not been reported so far. The present study was carried out to estimate resuspension factors for plutonium from such humid beach soils in the vicinity of low level liquid effluents discharge point from a Fuel Reprocessing Plant.

EXPERIMENTAL WORK

The work was carried out in two parts.

1. Field Measurements :

Air samples were collected on Whatmann GF/A glass fibre filter papers, using portable battery operated sampler, through a 25 mm sampling head at a suction rate of 12 litres/min. Sampling was done at two different heights of 1 and 2 meters and the sampling time was four hours. Plutonium concentrations and moisture contents of surface soil samples collected from the top 5 to 10 mm were determined. Studies were carried out both before monsoon and after monsoon.

2. Laboratory Measurements :

Series of laboratory experiments on the contaminated soils collected from the field were done, varying parameters like soil conditions, moisture content, wind speed etc. Experiments were carried out in a fumehood by uniformly spreading the soil in an aluminium tray (30cm x 30cm x 2cm). Air samples were collected at 1 meter above the tray using 25mm sampler at a flow rate of 45 litres/min. A total of 10 cubic meters of air was sampled every time. Knowing the airborne concentration and the concentration in the top soil (5mm), resuspension factors were calculated.

Different parameters varied during studies were as follows:

i) Soil conditions. Soil used in different studies were, dried and powdered soil, original wet soil with 20% moisture content and the wet soil with small plant sapling. The third condition simulates marshy area soils with some vegetation.

ii) Moisture contents of soil were varied from 10% to 30% in three different sets of experiments.

iii) Wind speed in Trombay area vary in the range of 3 to 19 Km/hr

(8). Three wind speeds simulated were 5.5, 8.0 and 11.5 Km/hr. This simulation was achieved by blowing air above the tray, uniformly above the entire surface.

iv) Respirable samples were also collected using cyclone sampling method (9) for two different humidities of soil and at two different wind speeds.

RESULTS AND DISCUSSION

Resuspension factors obtained during field studies carried out at different sites near and around effluent discharge areas, before and after monsoon are given in Table 1 and were found to be of the order of $10^{-7}m^{-1}$. With increasing moisture content in the soils, resuspension factors decreased. Factors at lower heights were relatively higher. Postmonsoon studies show slightly lower values compared to premonsoon ones.

TABLE 1. Resuspension Factors for Field Measurements.

Site No.	Collection Height (Meters)	Soil Moisture Content (%)	R.F. ($\times 10^{-7} m^{-1}$)
		(Premonsoon)	
1	2.0	8.81	3.79 ± 0.15
2	1.0	13.04	3.07 ± 0.14
	2.0	13.04	2.72 ± 0.11
3	1.0	26.59	3.19 ± 0.12
	2.0	26.59	1.15 ± 0.05
4	2.0	40.13	0.74 ± 0.03
		(Postmonsoon)	
1	2.0	10.13	2.61 ± 0.12
2	2.0	14.69	1.42 ± 0.09
3	2.0	28.24	0.96 ± 0.04
4	2.0	39.43	0.60 ± 0.03

Laboratory experiment results are given in Tables 2 to 5. Table 2 gives resuspension factor values for three different wind speeds for soils with different moisture contents. Increasing wind speeds increase resuspension factors only marginally, whereas resuspension factors decrease significantly with increasing soil moisture. Table 3 gives resuspension factor values for dry powdered soil at different wind speeds. In Table 4, are given values for soil covered with small plant saplings, studied at three different wind speeds. Resuspension factors obtained were comparatively lower than those obtained from the soil as such, indicating that plant growth in such soils help in reducing resuspension. Table 5 gives results for estimation of respirable fractions of these resuspended activities. Experiments carried out on dry soil as well as soil with 20% moisture content, at two different speeds show that for both the types of soils,

the respirable fractions were of the order of 65% indicating an activity median aerodynamic diameter of 2.6 μm (Ref.9) for airborne particles.

TABLE 2. Resuspension Factors (Lab. Measurements)

Soil Moisture Content	Wind Speed (Km/hr)		
	5.5	8.0	11.0
(%)	R.F(x $10^{-8}m^{-1}$)	R.F.(x $10^{-8}m^{-1}$)	R.F.(x $10^{-8}m^{-1}$)
10	2.66 ± 0.11	2.24 ± 0.11	2.97 ± 0.37
20	1.92 ± 0.10	1.65 ± 0.11	1.49 ± 0.08
30	1.71 ± 0.09	1.34 ± 0.07	1.36 ± 0.07

TABLE 3. Resuspension Factors For Dry Powdered Soils(Lab. Experiment)

Wind Speed(Km/hr)	Resuspension Factor (x $10^{-8}m^{-1}$)
5.5	1.45 ± 0.06
8.0	2.19 ± 0.09
11.0	2.49 ± 0.12

TABLE 4. Resuspension Factors For Soils Covered With Plant Saplings. Soil with 20% humidity (Lab. Experiment)

Wind Speed (Km/hr)	Resuspension Factor (x $10^{-9}m^{-1}$)
5.5	7.16 ± 0.41
8.0	8.46 ± 0.45
11.0	8.91 ± 0.49

TABLE 5. Respirable Resuspended Activities (Lab. Experiment)

Type of Soil	Wind Speed (Km/hr)	Respirable Fraction (%)
Dry soil	5.5	66.84 ± 3.11
Dry soil	8.0	68.68 ± 3.10
Soil with 20% moisture	5.5	64.67 ± 3.12
Soil with 20% moisture	8.0	60.28 ± 2.82

CONCLUSION

The resuspension factors obtained in the present study are useful in arriving at the standards for sea discharges.

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A REVISED METHOD TO CALCULATE THE CONCENTRATION TIME INTEGRAL OF ATMOSPHERIC POLLUTANTS

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1. ABSTRACT

It is possible to calculate the spreading of a plume in the atmosphere under instationary and inhomogeneous conditions by introducing the "particle-in-cell" method (PIC). This has a systematical advantage over the steady state Gaussian plume model usually used with averaged data. In particular the concentration time integral is calculated directly instead of being substituted by a locally integrated concentration at a constant time as is done in the plume model. Thus inaccuracies due to the computational techniques may be avoided for short-time emissions because both integrals do not lead to the same results as is shown herein. Additionally the PIC-method makes it possible to consider height-dependent input data and their variations in time.

2. CALCULATIONS OF THE CONCENTRATION TIME INTEGRAL

The near-ground concentration c of a plume spreading in the atmosphere is almost only described by the Gaussian plume model. It depends on a solution of the Fickian diffusion law by a Gaussian normal distribution. The effect I of the airborne material on live corresponds with the concentration time integral over the time T of influence at a fixed point:

$$I(x,y) = \int_T c(x,y,z=o,t)\cdot dt$$

An analytical solution of this time integral does not as yet exist if any time-dependent meteorological input data are to be used. However, it may be approximated if it is replaced by an integration that adds up all plume parts in concern locally at a constant time, i.e. $dx=u\cdot dt$. That is only possible if the meteorological data are constant all over the plume and if the cloud passes the exposure point unchanged.

If there are instationary or inhomogeneous effects to be considered as may be an advection velocity that varies locally and in time, a numerical solution is to be preferred e.g. the PIC-method. Its basis is to calculate the diffusion in the cloud and its transport step by step from time to time. The region around the cloud is divided into cells between which particles representing single parts of the whole concentration are moved according to the diffusion laws. The velocities of the particles are composed of the advection velocity and of the diffusion

velocities derived from the different concentrations in the cells. This movements are executed step by step in such a way that the input data may be fitted locally and temporally to the circumstances. The method has been described by Sklarew et al. [1]. In practice it has been shown by Voelz et Schultz [2] that many dispersion processes may be simulated in a good sense over longer periods. And this PIC-method allows to calculate either the realistic time integral or the substituted local integration.

3. COMPARISON OF THE INTEGRALS IN BOTH METHODS

If the PIC-method is used with local and temporal mean values of the input data, it normally gives the same concentration values compared to the Gaussian plume model under the same conditions. It gives comparable integral values, too, if these are calculated by the substituted local integration (1a).

This is also true if the integral values are evaluated using the PIC-method with either the substituted local integration for a height-dependent wind speed or the realistic time integral for a mean wind speed (1b, 2a).

Contrary, the realistic concentration time integral when used for a height-dependent wind speed differs from adequate calculations by the Gaussian model. Now the maximum value increases and lies nearer to the source. This is an effect that has indeed been observed during diffusion experiments at elevated sources (2b).

Fig. 1 shows the integral values downwind for a neutral diffusion type. They are calculated once for a height-dependent wind speed (b) and once for an adequate mean wind speed (a) with either the realistic concentration time integral at a fixed exposure point (2) or the substituted local integration at a constant time (1). The difference between the curves (2a) and (1a) is only due to the numerical computational techniques. On the other hand, it may be seen that the realistic concentration time integral used for a height-dependent wind speed (2b) differs from those integral values that are calculated with either a constant wind speed using the realistic time integral (2a) or using the Gaussian model.

Fig. 2 shows the velocities within a cloud from a 100 m high point source emission 3100 m downwind influenced by a height-dependent wind speed in a neutral atmosphere. There are shown on the right-hand side the diffusion velocities on those corner points situated in a vertical plane through the cloud center line and on the left-hand side the height-dependent advection velocities.

4. RESULTS

Both, the concentration time integral and the substituted local integration do not lead to the same results. The

reason may be explained as follows: The higher integral values near the source appear when using the realistic time integral. This happens as the cloud is not so much twisted by the height-dependent wind speed and because it passes more slowly the exposure point, that is to say, with a wind speed representative for the near-ground atmospheric layer. And besides that, during the passage some particles move at the tail-end from the upper parts downwards due to the decreasing concentration gradient. This downward diffusion stops at about 1200 to 1400 m far from the source. Then the cloud passes unchanged the exposure point only with the wind speed representative for the near-ground layer. Finally, the cloud becomes twisted at the tail-end by the height-dependent wind speed that forces the upper parts to cross more quickly. Therefore an upward diffusion starts due to the downward increasing concentration gradient. As a consequence, the realistic concentration time integral becomes lower more and more with the increasing downwind distance from the source in relation to the Gaussian model.

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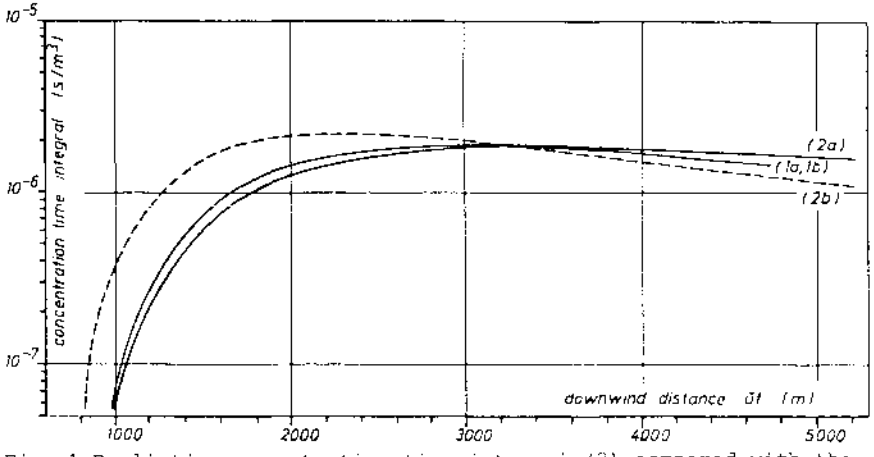

Fig. 1 Realistic concentration time integral (2) compared with the substituted locally integrated concentration (1) calculated using the PIC-method for a constant (a) and a height-dependent (b) wind speed for a neutral diffusion.

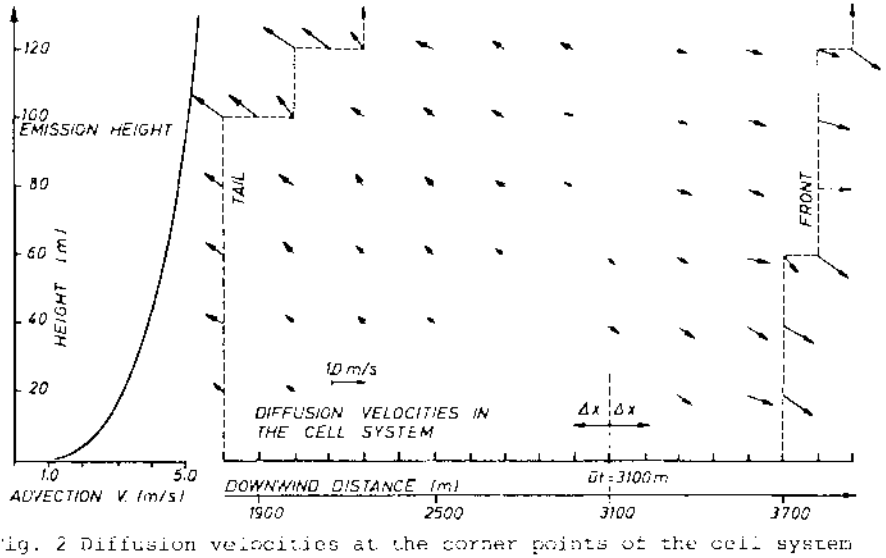

Fig. 2 Diffusion velocities at the corner points of the cell system in a vertical plane after the cloud of a 100 m high point source emission has travelled ūt=3100 m downwind controlled by the height-dependent wind speed u(z).

COMPARISON OF TWO- AND THREE-DIMENSIONAL ATMOSPHERIC DISPERSION MODELS

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In many models describing the atmospheric dispersion of waste air plumes (1,2), the diffusion in the direction of the wind is neglected, because this is only a minor effect compared to the convective transport by the wind. This simplification is certainly adequate for long downwind distances and continuous releases. However, the concentration of pollutants may be much underestimated for short distances and for elevated release. In this paper the results of two- and three-dimensional calculations are compared in order to check the validity of the simpler model.

COMPARISON METHOD

Because of the linearity of the differential transport equation, any release can be regarded as a succession of short puffs, which all follow the same dispersion laws. The total concentration at any point of the environment is then the superposition of the contribution of each puff. In order to appraise the effect of neglecting diffusion in the direction of the wind, it is therfore sufficient to follow the dispersion of a single puff. Thus we assume the following unit source strength:

$$Q(t) = \delta(t) \qquad (1)$$

where $\delta(t)$ denotes the Dirac delta function.

To avoid unnecessary complication of the problem, all the depletion processes, such as radioactive decay and dry or wet deposition of pollutants on the ground are neglected. These processes affect primarily the pollutant content of the plume but not the dispersion. Owing to this assumptions, the waste content of the plume will remain constant at unity.

The three-dimensional diffusion equation predicts a Gaussian distribution of the pollutants along the three axes x, y, z (downwind, crosswind and vertical respectively). Because of this variable separation, it is sufficient to consider only the concentration along the x-axis on the ground plane. If assuming, as customary, a total

relexion from the ground, the groundlevel concentration at downwind distance x and at time t after release is:

$$C(x,t) = \frac{2}{(2\pi)^{3/2}\sigma_h^2\sigma_v} \exp\left\{-\frac{1}{2}\left[\left(\frac{x-ut}{\sigma_h}\right)^2 + \left(\frac{h}{\sigma_v}\right)^2\right]\right\} \quad (2)$$

In equation (2), h is the release height and u is the mean wind velocity. The horizontal and vertical diffusion parameters $\sigma_h(\xi)$ and $\sigma_v(\xi)$ respectively are functions of the transport distance $\xi = ut$ and depend on the atmospheric stability conditions and the surface roughness.

When neglecting the diffusion in wind direction, the Gaussian distribution along the downwind axis will be degenerated in a Dirac delta function, i.e.:

$$\frac{1}{\sqrt{2\pi}\,\sigma_h} \exp\left\{-\frac{1}{2}\left(\frac{x-ut}{\sigma_h}\right)^2\right\} \longrightarrow \delta(x-ut) \quad (3)$$

Thus the concentration with the two- dimensional approximation will be

$$C'(x,t) = \frac{\delta(x-ut)}{\sigma_h\sigma_v} \exp\left\{-\frac{h^2}{2\sigma_v^2}\right\} \quad (4)$$

According to equation (4), the concentration C'(x, t) at any downwind distance x is always zero, except at time $t = x/u$, when it is infinite. Hence this is not a good basis for a comparison. However, for assessing the consequences of pollutant releases, one needs the time integrated concentration. We define the time integrated concentration as:

$$D(x, t) = \int_0^t C(x, t')\,dt' \quad (5)$$

In the case of the two-dimensional model, the integration of equation (5) can be carried out analytically, leading to:

$$D'(x, t) = \frac{1}{\pi u\,\sigma_h(x)\,\sigma_v(x)} \exp\left\{-\frac{h^2}{2\sigma_v^2}\right\} \quad (6)$$

for all times $t \geq x/u$. For $t < x/u$, D'(x, t) vanishes. In the case of the three-dimensional model, the integration has to be performed numerically. The effect of neglecting the diffusion in the direction of the wind can now be

estimated by comparing the time integrated concentrations D and D'.

RESULTS

In order to evaluate the discrepancy between the two results, we introduce a correction factor

$$f(x) = \frac{D(x,\ t = \infty)}{D'(x,\ t = \infty)} \qquad (7)$$

The integral for D, equation (5), can be rewritten as

$$D(x,\ t) = \frac{1}{u} \int_0^{\xi} C(x,\ \xi')\ d\xi' \qquad (8)$$

with $\xi = ut$ denoting the transport distance. One can see now that the correction factor f(x) does not depend on the wind velocity u but only on the release height h and the diffusion parameters σ_h and σ_v. Numerical integration of equation (8) and subsequent determination of the correction factor f(x) was performed for a release height h = 100 m and for the six Pasquill diffusion categories A to F (3). The Jülich diffusion parameter system was chosen, which describes the diffusion parameters by the power function:

$$\sigma(x) = p\ x^{q} \qquad (9)$$

The results of these computations are presented in figure 1, showing the correction factors f(x) for the six stability classes. From this figure one can see that for the unstable and neutral diffusion categories the simpler model gives a very good approximation beyond the distance of a few hundred meters. For the stable categories E and F, agreement is achieved after one or two kilometers. There is a very noticeable increase in value of the correction factors with decreasing downwind distance, denothing an underestimation by the two-dimensional model of several orders of magnitude. However this underestimation is not serious, because the factors apply to concentrations that are much below the maximum values. These maximum integrated concentrations are slightly overestimated by the simpler model as indicated in Figure 1.

From these results one can conclude that the simpler two-dimensional model is accurate in most cases of interest. Because of the much easier application of this model, its use can be advocated. However, care must be taken for short downwind distances, where neglecting diffusion in wind direction leads to a drastic underestimation of

the concentration. Finaly the simpler model is not appropriate to describe the short time behaviour of the concentration especially in emergency situations.

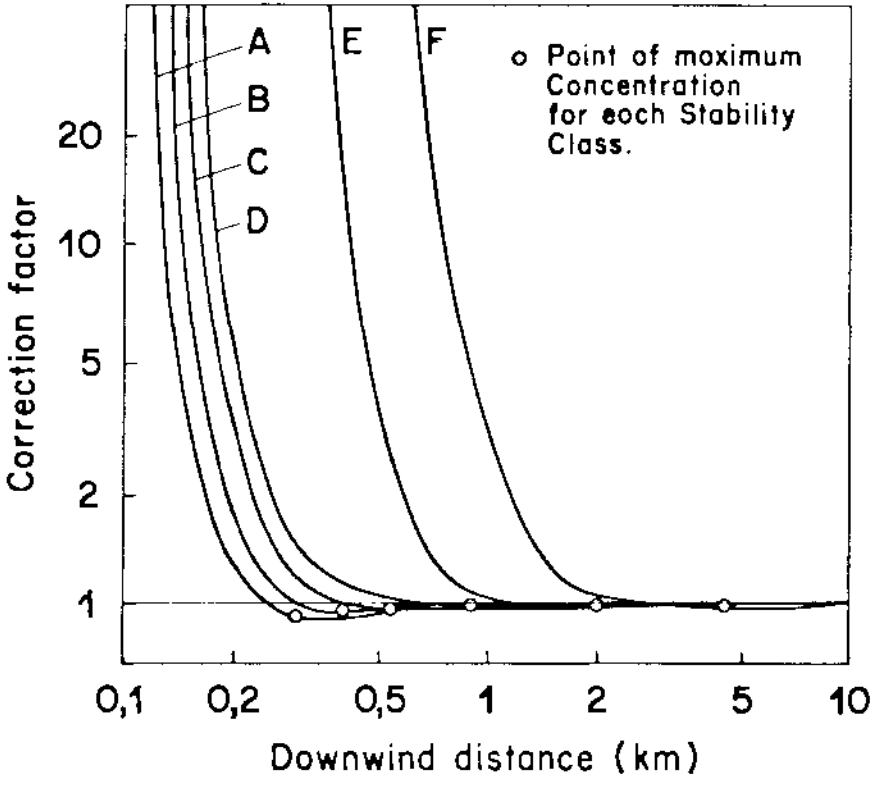

Figure 1: Correction factors F(x) as defined in equation (7) for the six diffusion categories (effective release height h = 100 m

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A CASE OF INTERNAL CONTAMINATION WITH PLUTONIUM OXIDE

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This paper describes a case of plutonium and americium internal contamination due to an accidental glove-box explosion occurred in 1974 at the Casaccia Plutonium Plant. The involved person showed a small contaminated wound (3x 0.5 cm) on his right cheek, a diffused contamination on the hair and a considerable activity in the nose which would indicate a possible incorporation by inhalation. On the basis of the information obtained at the Plutonium Plant the material contained in the exploded glove-box resulted to be a powder of PuO_2 calcinated at high temperature. In order to know just the isotopic and weight composition of the contaminating material the following radiometric and chemical measures were carried out on the nose-blow sample: gamma spectrometry (^{241}Am); gamma+X spectrometry with a thin NaI(Tl) crystal and Be window (^{241}Am and Pu); liquid scintillation (^{241}Pu and alpha emitters); alpha spectrometry (^{238}Pu + ^{241}Am and ^{239}Pu + ^{240}Pu); chemical separation of americium from plutonium. The following data were obtained: 97.56% in weight for alpha emitters (0.17% ^{238}Pu, 97.05% $^{239,240}Pu$, 0.34% ^{241}Am) and 2.44% ^{241}Pu; the activity distribution was 95.9% beta activity (^{241}Pu) and 4.1% alpha activity ($^{238,239,240}Pu$ and ^{241}Am); the distribution of alpha activity resulted to be 65% $^{239,240}Pu$, 25% ^{238}Pu and 10% ^{241}Am. The knowledge of the isotopic composition was necessary to correctly estimate the initial plutonium and americium activity in the wound, the lung burden calculated by W.B.C. and the dose committment to the different organs.

DIAGNOSTIC AND THERAPEUTIC ACTIONS

The following actions were taken to reduce the initial contamination and to get the maximum information on the residual contamination and on the dose committment. a) The wound was washed with DTPA and the activity was removed by a surgical toilet; b) the hair and the nose were decontaminated; c) some direct lung countings were performed; d) many urine and fecal samples were analyzed for Pu and ^{241}Am; e) some blood samples were collected for the determination of plutonium and for the detection of possible chromosomial aberrations; f) at the second day a DTPA treatment was started consisting on three dayly 0.5 g DTPA intravenous injections followed by 3 others on alternate days and on a 0.5 g DTPA aerosol inhalation during 2 consecutive days.

RESULTS

The following results were obtained. a) The activity in the wound was determined (1) by using a special NaI(Tl) probe suitable to detect the weak X emission of plutonium (17 KeV) and the X+gamma emission of ^{241}Am (17 KeV and 60 KeV); the localization of the superficial alpha activity was obtained by using a probe with a 7 mm^2 solid state alpha detector. The initial activity resulted to be ~ 30 nCi, and it was reduced to background levels by washing with DTPA and by carrying out a surgical toilet. b) The initial activity in the hair was ~83 nCi and it was reduced to negligible values by using a shampoo containing DTPA. The activity of the nose blow, collected just after the incident, was 7.5 nCi. A direct lung counting (2,3) of the subject, based on the detection of both the 17 KeV X-rays emitted by the plutonium isotopes and the 60 KeV gamma-rays of the ^{241}Am, was performed at various times after the incident. A 12.5 cm diameter x 0.1 mm thick NaI(Tl) phosphwich crystal positioned on the right lung or over the sternum was employed. The calibration factor applied to lung counting of the plutonium isotopes was obtained on the basis of both phantom and "in vivo" calibration (3,4) taking into account both the chest size of the subject and the isotopic composition of the contaminating mixture. The calibration factor for the ^{241}Am in vivo counting was based on phantom calibration only. The ^{238,239,240}Pu lung contents $\pm$ 2σ as a function of time elapsed from the incident were the following: 56 $\pm$ 20 nCi (5 h.); 25 $\pm$ 15 nCi (22 h.); 13 $\pm$ 10 nCi (5 d.); $<$10 nCi (19 d.). The corresponding ^{241}Am lung contents $\pm$ 2 σ resulted to be: 2.0 $\pm$ 0.5 nCi; 1.5 $\pm$ 0.5 nCi; 0.7 $\pm$ 0.3 nCi; 0.4 $\pm$ 0.3 nCi (40 d.); 0.15 nCi (70 d.). d) Taking into account the 55 urine analyses (37 of Pu and 18 of ^{241}Am) and the 34 feces analyses (26 of Pu and 8 of ^{241}Am) (5), the excretion curves shown in Fig. 1 and 2 have been obtained. e) No plutonium activity greater than the sensitivity limit (0.04 pCi) was detected in 10 ml of blood and no cromosomial aberration was found in 200 cells. f) No effect due to the DTPA treatment was shown in the urinary excretion curves.

DOSIMETRIC EVALUATION

Taking into account the data supplied by the lung counting and by the excretion curves, the following conclueions can be drown: a) the ratio ^{238,239,240}Pu/^{241}Am for fecal excretion is about 10, just as the ratio of the alpha activity present in the contaminating material: the similar metabolism observed for Pu and Am can be due to the fact

that both the elements were present as a very insoluble oxide; b) both the Pu and Am fecal excretion curves are very steep during the first few days (Peak activity/Plateau activity $\sim 10^5$) and this datum is in good agreement with the sharp decrease of the lung content in the same period: it appears threfore that the material granulometry was high (1 ÷ 10/um), mainly deposited in the upper part of the respiratory tract and thus fastly removed by the ordinary clearance mechanisms. c) The high ratio E_f/E_u ($\sim 10^4$ in the first few days) and the inefectiveness of DTPA confirm the biological non-trasportability of the contaminant. d) The plutonium activity excreted in the first few days with feces is $\sim$130 nCi which may correspond to an initial lung burden comprised between 13 and 65 nCi; this value is in good agreement with that found by the W.B.C. at the first day (56 ± 20 nCi). e) Taking into account the fecal curve after the first ten days, a lung half-time of about 100 days can be deduced in accordance with the values reported in the literature (6) for insoluble compounds. f) The fecal excretion after 100 days (0.5 pCi) would indicate a plutonium residual lung burden of 0.25 ÷ 1 nCi (6). g) Taking into account the urinary excretion after 100 days (0.2 pCi) a plutonium systemic burden of 3 nCi can be obtained (7). h) The committed lung dose, calculated on the basis of reference (6) and a biological half-life of 500 days for Pu and Am, resulted in the range of 60 ÷ 240 mrad with a corresponding maximum dose rate of 30 ÷ 120 mrad/y. i) The dose due to systemic contamination has been evaluated on the basis of reference (8) and considering the following percentage depositions and biological half-lives: Pu 42% in bone ($T_b = 5.5 \cdot 10^4$ d.), 56% in liver ($T_b = 5.5 \cdot 10^4$ d.); Am 25% in bone ($T_b = 7.3 \cdot 10^3$ d.), 35% in liver ($T_b = 3.5 \cdot 10^3$ d.) and 3% in kidneys ($T_b = 2.7 \cdot 10^4$ d.). For the contribution of lung contamination to systemic dose, the T_b in lung was considered 90 days. l) The calculated absorbed dose rate for bone was rather constant being in the range of 30 ÷ 40 mrad/y slowly increasing with time; for liver a rather constant dose rate of 200 mrad/y; for kidneys a rather constant dose rate of 3 mrad/y slowly decreasing with time. m) The committed dose equivalents, calculated on the basis of ICRP recent metabolic models (9,10) with Q = 20 for alpha particles, are: lung 1.2 ÷ 4.8 rem (12 ÷ 48 mSv); bone 40 rem (400 mSv); liver 100 rem (1 Sv); kydneys 3 rem (30 mSv). The effective total body committed dose equivalent is 7.5 ÷ 8 rem (75 ÷ 80 mSv). From a medical point of view, the operator was readmitted

to unlimited radiation work, but caution was taken not to involve him in high-risk contamination areas or operations.

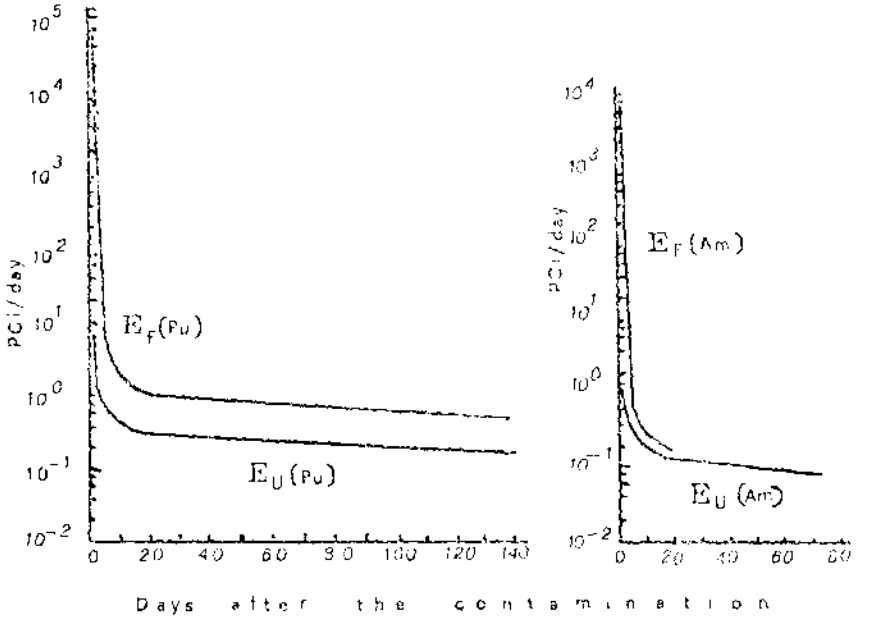

Fig.1 and 2. Urinary (E_u) and fecal (E_f) excretion of plutonium and americium.

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THE USE OF DTPA TO INHIBIT THE EXTRAPULMONARY DEPOSITION OF CURIUM-244 IN THE RAT FOLLOWING THE BRONCHIAL INTUBATION OF OXIDE SUSPENSIONS

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INTRODUCTION

Much experimental evidence has accumulated on the use of the calcium or zinc salts of diethylenetriaminepenta-acetic acid (Na_3Ca DTPA and Na_3Zn DTPA) for removing actinides from animals (1). However, such studies are based on experiments in which the actinides have been administered as the citrate or nitrate complexes. The present work looks at the efficacy of Na_3Ca DTPA and Na_3Zn DTPA for enhancing the excretion of curium after pulmonary intubation of curium-244 dioxide (CmO_2).

MATERIALS AND METHODS

High fired CmO_2 was supplied by the Radiochemical Centre (Amersham, Bucks, U.K) and was fractionated by ultrafiltration as described previously (2).

In the animal experiments Na_3Zn DTPA or Na_3Ca DTPA were administered intravenously in isotonic saline. The diuretic Lasix, frusemide B.P., (0.2 ml, 8.9 mg kg^{-1}) was injected intravenously at intervals to promote a high urine flow to allow the collection of adequate volumes of urine for analysis by gel permeation chromatography.

The methods of pulmonary intubation, gel permeation chromatographic separation and radioactivity determinations are given by Stradling *et al.*, (2).

RESULTS AND DISCUSSION

After intubation into the lung, 0.22-1.2 μm diameter curium dioxide particles rapidly form particles of 0.001 μm in diameter (2). These particles, believed to be of the hydroxide, then diffuse passively to the blood probably through pores in the alveolar epithelium (3). In the blood intact 0.001 μm particles of CmO_2 combine with serum proteins. The protein-bound CmO_2 rises from 45% of the circulating radioactivity at 35 minutes after pulmonary intubation to >90% at 24 hours; the remaining activity is 0.001 μm particles.

0.001 μm particles will also combine with serum proteins *in vitro*. For example, when serum labelled for 24 hours was chromatographed on Sephadex G-200, Cm eluted with the α and γ globulins, and the transferrin and albumin fractions in about equal amounts. Negligible activity (< 1%) was recovered in the low molecular weight fractions where unbound particles or curium would elute. However,

if Na_3Ca DTPA or Na_3Zn DTPA is added at a concentration of 0.02 mg. ml^{-1} to the serum 6 minutes before the 0.001 µm particles the reaction between particles and proteins is inhibited and even after 24 hours 99% of the radioactivity eluted as intact particles. Similarly, intact 0.001 µm particles could be regenerated from protein-bound Cm by addition of Na_3Ca DTPA (2.5 $mg.ml^{-1}$). It is suggested that DTPA blocks receptor sites for the particles on the protein by a preferential binding process.

Previous work has shown that a major factor infuencing the urinary excretion of Cm following the intake CmO_2 into the lungs is the renal dialysis of 0.001 µm particles (2). The binding of particles to serum proteins may compete with this process. The above studies in vitro suggest that either Na_3Ca DTPA or Na_3Zn DTPA could maintain these 0.001 µm particles in the blood for long enough to permit the quantitative urinary excretion of Cm. The effect of administering Na_3Ca DTPA or Na_3Zn DTPA to rats exposed to CmO_2 suspensions is shown in Table 1. If the concentration of Na_3Ca DTPA or Na_3Zn DTPA in the blood is maintained above 0.002 $mg.ml^{-1}$ (Expt. 2), by administering 0.28 $mg.kg^{-1}$ body weight initially followed by injections of 0.14 $mg.kg^{-1}$ at 30 minute intervals, then deposition of Cm in the skeleton and liver is markedly reduced. The interval between successive injections corresponds to the half time of DTPA in the blood (4). At higher concentrations (Expt. 3) Na_3Ca DTPA is still effective in minimising tissue deposition even when administered 2 hours after small particle suspension. In all of the experiments where Na_3Ca DTPA or Na_3Zn DTPA and Lasix were administered before the oxide suspension the Cm was excreted as 0.001 µm particles. When the oxide suspension was administered before the DTPA and Lasix the Cm was present in the urine as 0.001 µm particles and Cm citrate. The Cm citrate is probably formed from particles and citrate in the renal tubular fluid (2).

The experiments outlined above demonstrate that (i) DTPA is not chelating Cm but inhibiting a reaction between 0.001 µm CmO_2 particles and serum proteins (ii) Na_3Ca DTPA and the less toxic Na_3Zn DTPA are equally effective and (iii) to obtain efficient urinary excretion of Cm the concentration of DTPA in the blood must be maintained above about 0.004 $mg.ml^{-1}$. Animal experiments indicate that following an accidental intake of $^{244}CmO_2$ by man, about 90% of that fraction destined to translocate to blood would do so during the next month (5). Therefore, for DTPA therapy to be most effective it should be administered continually over this period at a constant rate of 14 $mg.kg^{-1}.day^{-1}$. This is within the dose range normally used in clinical practice (6).

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Table 1. Injection schedules, tissue distribution and excretion of ^{244}Cm after administering ^{244}Cm oxide

Expt. No.	Injections	Injection schedule[a]		Tissue distribution and excretion[b] (%)			
		$mg.kg^{-1}$ body wt	t min	Lungs	Liver	Skeleton	Urine
1	Lasix	8.9	-5 ; 120, 210	11.1	15.6	33.5	32.4
2	$Na_3ZnDTPA$	0.28;0.14[c]	-5;25,55,85..235	13.1	0.5	1.5	79.8
3	$Na_3CaDTPA$	14;7[c]	120;140,180...360	12.2	0.3	1.5	83.5

[a] Suspension of 0.001 µm diameter particles 100 µl, 500 Bq administered by tracheal intubation at zero time. The injection times shown for DTPA are relative to this labelling. The amount of DTPA administered in the first injection is twice that administered in subsequent injections. Lasix 0.2 ml, 8.9 $mg.kg^{-1}$ administered intravenously at -5, 120 and 210 minutes except experiment 3 where these times are relative to the first injection of DTPA.

[b] Values expressed as a percentage of initial lung burden; animals killed 240 min after initial injection. Remainder of ^{244}Cm present in blood, kidneys and gastro-intestinal tract and contents. No faeces were passed during the course of the experiments.

[c] To convert to $mg.ml^{-1}$ of blood divide by 70 (7).

The metabolic data were closely similar when Na_3 CaDTPA or $Na_3ZnDTPA$ were administered by the same injection schedule.

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TOXICITY OF INHALED $^{238}PuO_2$ I. METABOLISM[1]

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This paper reports the results of a study of the fate of inhaled $^{238}PuO_2$ in Beagle dogs. It complements a study on the relationship of biological response to radiation dose after inhalation of aerosols of $^{238}PuO_2$ (4). The aerosols used for the inhalation exposures were monodisperse (containing particles of only one size) or polydisperse. The use of monodisperse aerosols makes it possible to study the effect of particle size on the biological effectiveness of plutonium for various end points and to study the effect of particle size on the deposition and subsequent redistribution of plutonium in the body.

Other investigators (5) have shown that ^{238}Pu translocates more rapidly from the lung than does ^{239}Pu after inhalation of the dioxide form. This increased translocation may be due to a specific activity dependent breakup of the PuO_2 particles that results in more rapid dissolution or direct translocation of very small particles (1,7). In the present study, it was shown that ^{238}Pu was translocated relatively slowly up to about 100 days after inhalation, but the translocation rate increased more than twofold thereafter. A study of autoradiographs of the lungs of dogs that inhaled monodisperse aerosols revealed the presence of particle fragments in the lung. This indicated that the change in translocation was due to the breakup of particles as suggested.

MATERIALS AND METHODS

Young adult Beagle dogs received inhalation exposures to one of three sizes of a monodisperse aerosol or to a polydisperse aerosol of $^{238}PuO_2$ designed to produce an initial burden in the pulmonary region of 2.6 kBq per kg of body weight. Periodic excreta collections were made and analyzed radiochemically for Pu-238 content. Dogs were serially sacrificed after exposure. Their lungs were inflated, fixed and sampled for autoradiography. The remainder of the lung and other major organs taken at necropsy were analyzed radiochemically for Pu content. Samples taken for autoradiography were embedded, 5 μm thick sections obtained and autoradiographs were made.

Each dog's initial lung burden was calculated by summing the total excreta with the total activity in the tissues at sacrifice. The pulmonary retention of plutonium and its build-up in liver and skeleton were characterized by functions of exponentials. Radiation doses to lung, liver and skeleton were calculated by integration of the fitted curves. The number of particles, fragments and single

This research was performed under U.S. Department of Energy Contract No. EY-76-C-04-1013 in facilities fully accredited by the American Association for the Accreditation of Laboratory Animals.

tracks in a lung section autoradiograph were counted using an Olympus BHC microscope with darkfield illumination at 100X. Detection of fragments in lung depended on the uniform appearance of the autoradiographic images of the particles of a monodisperse aerosol. A set of concentric tracks (alpha star) which had an appearance similar to that of a alpha star in an animal sacrificed shortly after inhalation exposure was considered a particle. An alpha star with fewer tracks was called a fragment. Single tracks were either isolated alpha tracks or tracks in a group that did not originate from a common point. The diameter of a spherical $^{238}PuO_2$ particle which would be expected to have a given number of tracks was estimated (6).

RESULTS

The distribution of ^{238}Pu in lung, liver, skeleton and tracheobronchial lymph nodes of Beagle dogs sacrificed at various times after exposure is illustrated in Figure 1. Only the values for the

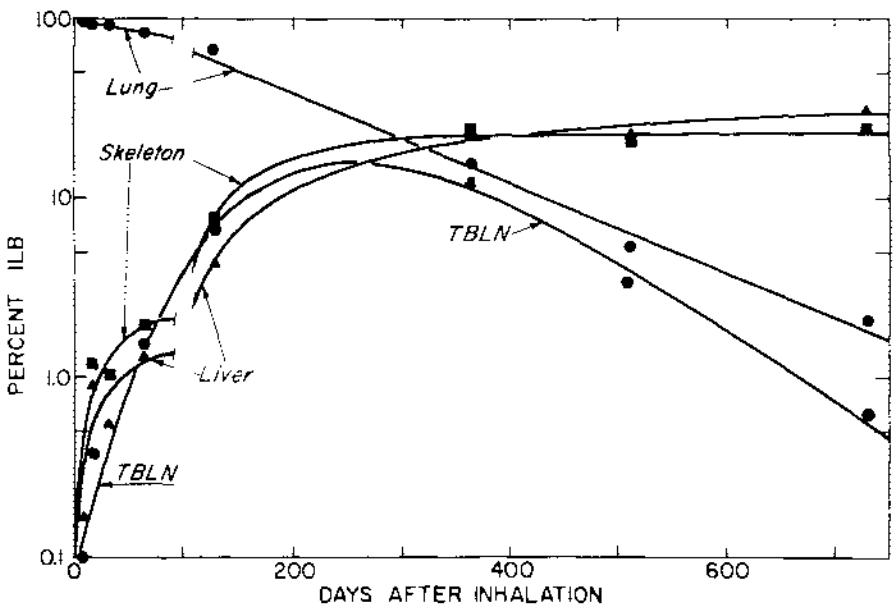

Figure 1. Distribution of ^{238}Pu in Beagle dogs following inhalation of a 1.4 μm AD monodisperse aerosol of $^{238}PuO_2$.

1.4 μm aerodynamic diameter monodisperse aerosol are shown in Figure 1. The retention and translocation of the other aerosols were similar except that the polydisperse aerosol translocated slightly more rapidly at early times after exposure. Up to 64 days after exposure, plutonium is cleared from the lung at a rate which would result in half of the material being cleared by 310 days after exposure. At later times this rate of clearance increased so that half of the

material present at 100 days after exposure was cleared by 220 days after exposure resulting in a clearance half-time of 120 days.

As a result of the increased translocation from lung, the doses to liver and skeleton at 720 days after exposure were each about 10% of the dose to lung at that time and were increasing rapidly.

The fraction of the alpha activity in the autoradiographs which was considered particles, fragments or single tracks is given in Figure 2. These data are also for dogs that inhaled a 1.4 μm aerodynamic diameter monodisperse aerosol. Fragments ranged in size from

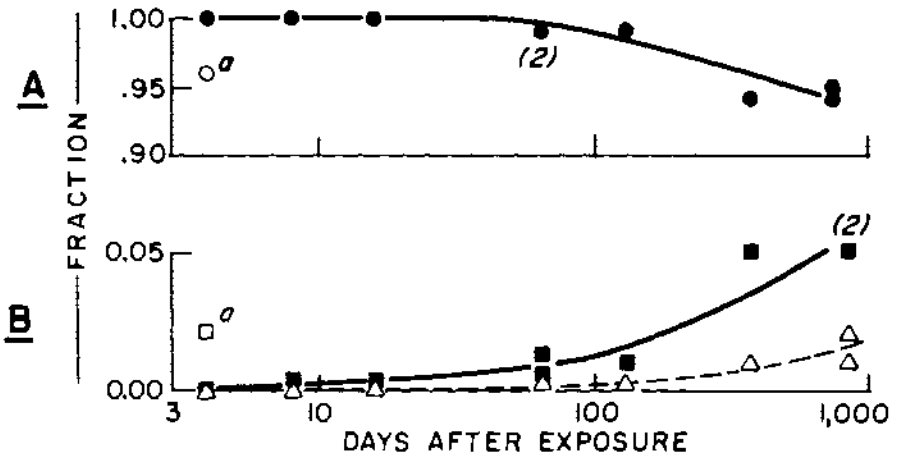

Figure 2. Fraction of activity from inhaled $^{238}PuO_2$ particles in lung which in in (A) particles or in (B) fragments producing one (□——) or more (△ - - -) tracks.

1 track per fragment to about 100 tracks with an average size of about 7 tracks if single tracks were not included in the average and about 1.1 tracks with single tracks included.

DISCUSSION

The rate of clearance from the lung at times after 100 days was more than twice that at times before 100 days. Increased translocation of plutonium from the $^{238}PuO_2$ particles in the tracheobronchial lymph nodes is illustrated by the decrease in the amount of Pu found in these nodes at times beyond 1 year after exposure. While urinary excretion of Pu began to increase at early times after inhalation, the peak level was not reached until after 100 days after inhalation. These data suggest that there was a definite change in the nature of the $^{238}PuO_2$ particles at about 100 days after inhalation. Autoradiographic analysis of the Pu in the lung revealed an increased number of fragments beginning at about this time. These fragments were somewhat larger than those observed by Fleisher and Raabe (1) in the

in vitro "dissolution" of $^{239}PuO_2$ particles stored dry for 3.75 years. The fragments observed in their study were all less than about 9 nm in diameter and hence could, in large part, be transferred directly from lung to blood (6). All or most of the fragments observed as single tracks in this study may be fragments larger than 10 nm in diameter, or those fragments which cannot be directly translocated to the systemic circulation (2,6). Thus, the fragmentation appears to have caused increased Pu translocation because of increased surface area (3) or direct translocation of extremely small particles into the systemic circulation (6).

CONCLUSIONS

There was breakup of $^{238}PuO_2$ particles deposited by inhalation in the lungs of Beagle dogs that resulted in less focal irradiation of the lung and in increased translocation of plutonium from the lung to other organs after about 100 days after exposure. This conclusion has several implications for the assessment of hazards following inhalation of $^{238}PuO_2$. First, in experiments using inhaled $^{238}PuO_2$, data must be obtained over a long time period, preferably at least 2 years after inhalation, to assess accurately the radiation dose to lung, liver and skeleton. Second, the increasing urinary excretion of ^{238}Pu with time following intake by inhalation must be considered when using urinary excreta data to assess the quantity of material present in lung following any inhalation incident. Third, because of the differences in translocation of Pu isotopes and the increased dispersion of ^{238}Pu in the lung, $^{238}PuO_2$ inhalation incidents must not be evaluated using factors derived from $^{239}PuO_2$ studies or incidents. Finally, the designation of organs at risk after inhalation of $^{238}PuO_2$ may require some reassessment.

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PROPOSED RETENTION MODEL FOR HUMAN INHALATION EXPOSURE TO $^{241}AmO_2$

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Human exposures to ^{241}Am have been reported for four cases with measurements of lung retention near the exposure time (1-3), and five cases with long-term measurements of skeleton retention (4-6). These data were used to evaluate a model of ^{241}Am dissolution and retention developed using data from inhalation exposures of dogs to $^{241}AmO_2$. In several of the reports on human inhalation exposure, discrepancies have been shown with predictions of the Task Group Lung Model (1,3,4). The dissolution and retention model used in this paper takes into account the effects of particle size, distribution of particle sizes, and density of particles on lung retention. It is shown that the proposed dissolution and retention model is consistent with human inhalation exposures to ^{241}Am.

MATERIALS AND METHODS

The dissolution and retention model was developed using data from inhalation studies in Beagle dogs exposed to one of three sizes of monodisperse aerosols (0.75, 1.5, and 3.0 μm aerodynamic diameter) or a polydisperse aerosol (1.8 μm activity median aerodynamic diameter) of $^{241}AmO_2$ (7). Animals were sacrificed in pairs from 8 to 730 days after inhalation exposure to determine the organ retention and distribution patterns. Metabolic data from the studies using monodisperse aerosols were used to evaluate the effect of particle size on retention. Model parameters derived from the studies using monodisperse aerosols were used for modeling the study using polydisperse aerosols with adjustments only being made for the rate of mechanical clearance from lung to gastrointestinal tract. Model parameters from the study using polydisperse aerosols in Beagle dogs were compared with the human exposures in this paper.

The dissolution and retention model in Figure 1 described the lung as consisting of three regions – the nasopharynx, tracheobronchial and pulmonary – each cleared by competing pathways of mechanical clearance of particles to the gastrointestinal tract or dissolution and absorption into the general circulation. Dissolution and absorption was modeled as occurring through a dissolution pool and a compartment for the fraction of dissolved ^{241}Am bound locally to lung constituents. The locally bound ^{241}Am represents ^{241}Am seen as a diffuse distribution of single alpha tracts on autoradiographs of lung from the Beagle dog studies (7). Dissolution of the particles was described by equations developed by Mercer (8) who assumed the

This research was performed under U.S. Department of Energy Contract No. EY-76-C-04-1013 in facilities fully accredited by the American Association for the Accreditation of Laboratory Animals.

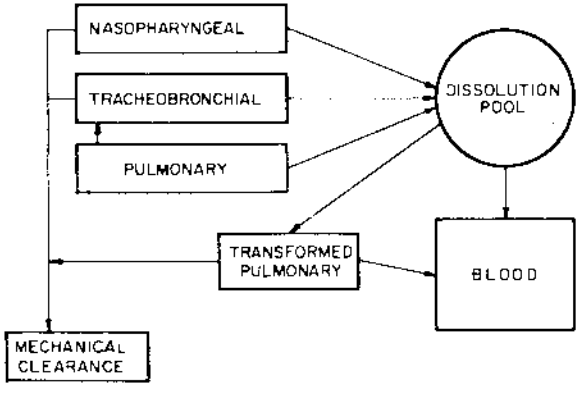

Figure 1. Dissolution and retention model for lung showing mechanical clearance pathways on the left to the gastrointestinal tract and dissolution pathways (described by equations of Mercer (8)) on the right from the 3 regions of lung to the dissolution pool. Other pathways for skeleton, liver, kidney, and soft tissue were modeled as exchanging with the blood compartment.

rate of dissolution is proportional to surface area of deposited particles. These equations take into account the distribution of particle sizes, density of particles, and shape of particles. The dissolution and retention model forced a total materials balance and used first order rate constants to describe transport, uptake and retention. In contrast, the Task Group Lung Model used by ICRP 30 (9) describes the clearance of lung as single exponential rates from subpools of the 3 regions of the lung. Also in contrast to ICRP 30, the model of retention in organs took into account redistribution of the absorbed ^{241}Am by representing the skeleton, liver, kidney, and soft tissue as two compartments in series exchanging ^{241}Am with the general circulation.

RESULTS

Predictions of lung clearance by the dissolution and retention model are shown for two particle sizes in Figure 2. Also shown are four cases of human inhalation exposures with early lung retention data (1-3) and the lung clearance of ^{241}Am as a Class W compound predicted by ICRP 30. Figure 3 shows predicted uptake by skeleton for three different particle sizes using the dissolution and retention model and for ^{241}Am as a Class W compound predicted by ICRP 30 (9). In applying the ICRP 30 lung model, it was assumed there was no absorption from the nasophyaryngeal and tracheobronchial regions since absorption from these regions appears to be low.

DISCUSSION

As can be seen from Figure 2, the dissolution and retention model can account for differences in early lung clearance observed in

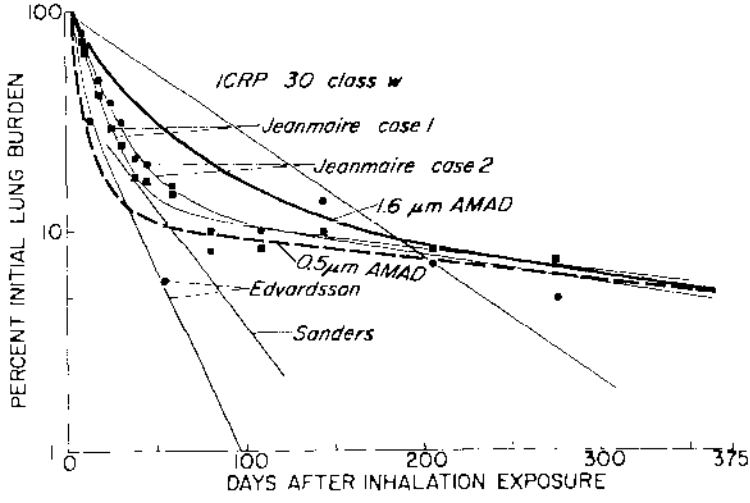

Figure 2. Comparison of early clearance from the lung for four human exposure cases (1-3), predictions of the dissolution and retention model, and predictions of ICRP 30 for ^{241}Am as a Class W compound (9). Lung retention is expressed as percent of the 4-day lung burden. Predictions of the dissolution and retention model are for a 0.5 um activity median aerodynamic diameter (AMAD) aerosol (dashed line) and a 1.6 μm AMAD aerosol (heavy solid line) with a geometric standard deviation of 2 and density of 8 g/cm^3.

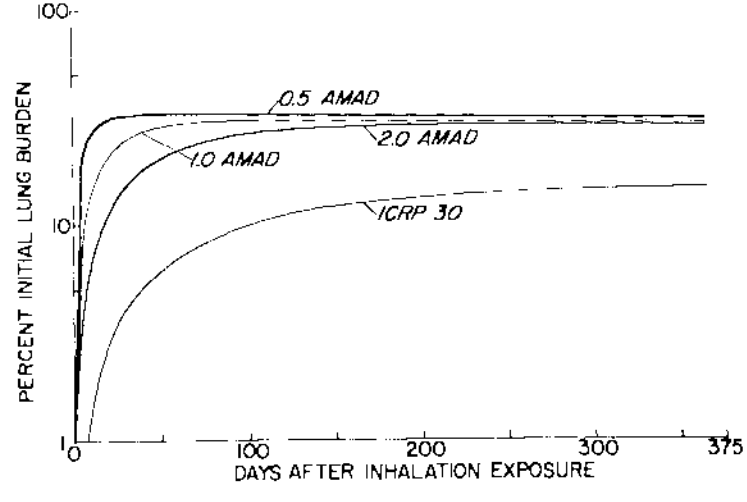

Figure 3. Predictions of skeletal uptake and retention from the dissolution and retention model for three particle sizes (geometric standard deviation of 2 and density of 8 g/cm^3) compared with the predictions of the ICRP 30 model of ^{241}Am as a Class W compound.

the human exposure cases by considering a range of activity median aerodynamic diameters from 0.5 μm to 1.6 μm. The surface area solubility rate constant was the same as that observed in dog studies (1.5 x 10^{-6} g/cm^2/day) and in *in vitro* solubility studies of AmO_2 (7). The particles were assumed to be spherical; however, this is not an essential assumption since changing the shape factor for irregular particles will have the same effect as changing the median particle size (8). Although characteristics of the aerosols were not reported for the human exposures, the parameters used to describe the aerosols in the calculations are typical of aerosols characterized in industrial facilities (10).

Comparison of the dissolution and retention model predictions with human inhalation exposure cases in which only long-term retention data were available for skeleton show reasonable agreement. Some discrepancies exist between the calculated half times of skeleton retention in humans of 17 years (5), 28 years (5), and 100 years (4,6), and the dissolution and retention model prediction of 10 years. These discrepancies probably occur because of a wide age range in human exposure cases (6 years to adults) and because the data used in developing the dissolution and retention model extends to only 2 years after inhalation exposure, making prediction of the long-term half-life in skeleton uncertain.

As shown in Figure 2, clearance of ^{241}Am from lung for the human exposure cases was more rapid than predicted in the ICRP 30 model with ^{241}Am as a Class W compound (9). The ICRP 30 model also fails to predict the presence of a long-term retained fraction as observed in several of the human exposures (1,4,5,6). Figure 3 shows the ICRP 30 model underpredicts the long term skeletal burden by a factor of 2 because the fraction of ^{241}Am absorbed from lung is underpredicted (4,7). Also the rate at which ^{241}Am accumulates in skeleton is underpredicted because ICRP 30 predicts a slower rate of absorption of ^{241}Am from the lung than was observed in the human exposures.

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TOXICITY OF INHALED $^{238}PuO_2$ II. BIOLOGICAL EFFECTS IN BEAGLE DOGS

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Plutonium-238 is produced in nuclear reactors using ^{235}U fuel. It is used as a fuel for space nuclear auxillary power units and as a power source in cardiac pacemakers.

The most likely route of entry of ^{238}Pu into the body during many accidents is by inhalation. Because of its high specific activity, local dose around particles of ^{238}Pu can be high and the question of homogeneous versus non-homogeneous dose to lung and its influence on biological effects becomes important. To study that question, the use of particles all of the same size (monodisperse) is necessary.

Dogs serially sacrificed after inhalation of $^{238}PuO_2$ had a significant amount of ^{238}Pu translocated to bone. Similar findings with significant numbers of bone tumors were found in another study (2).

MATERIALS AND METHODS

Seventy-two, 1 year old Beagle dogs, 36 males and 36 females, were given a single, nose-only exposure to an aerosol of 1.5 μm AD particles and an additional 72 dogs were given an exposure to 3.0 μm AD particles of $^{238}PuO_2$. Each study had 6 desired activity levels: 0.56, 0.28, 0.14, 0.07, 0.03 and 0.01 μCi per kg body weight; 12 dogs per activity level (Table 1). An additional 24 control dogs were exposed only to the aerosol generation solution. Methods for the preparation of monodisperse aerosols and for inhalation exposure of dogs have been described (1,3). The $^{238}PuO_2$ particles were tagged

TABLE 1. Experimental design.

Parameter	1.5 μm (AD)	3.0 μm (AD)
Physical size, μm	0.44	0.96
pCi per particle	4.9	51
Local dose rate, rads/day	280	3100
Number of particles, range	$2x10^4$ to $1x10^6$	$2x10^3$ to $1x10^5$
Fraction of lung irradiated	$9x10^{-4}$ to $5x10^{-2}$	$8x10^{-5}$ to $5x10^{-3}$
Initial lung burden, nCi	100 to 5600	100 to 5600
Avg. lung dose rate, rads/day	0.3 to 15	0.3 to 15

This research was performed under U.S. Department of Energy Contract No. EY-76-C-04-1013 in facilities fully accredited by the American Association for the Accreditation of Laboratory Animals.

with a gamma-emitting radionuclide, ^{169}Yb. Periodic whole-body counts of the ^{169}Yb tag were performed after exposure for the calculation of an initial lung burden (ILB). Medical examination of the dogs was daily observation, annual physical and radiographic examination, and semi-annual blood cell counts and serum chemistry tests. Sick dogs were examined and tested to establish a diagnosis. A few dogs died from their illness but most were euthanized. A necropsy examination was performed on all dogs and tissues were evaluated both histologically and radiometrically.

RESULTS

Initial lung burdens (ILB) ranged from 0.005 to 2.2 μCi/kg and 0.008 to 2.2 uCi/kg for dogs exposed to 1.5 um AD and 3.0 um AD particles, respectively.

The first biological effect observed was a lymphopenia. It was observed in all the dogs that died or were euthanized and occurred from 60 to 1200 days after exposure (90% of the dogs were diagnosed within 180 days). A 60% incidence of leucopenia was also noted.

Radiation pneumonitis with pulmonary fibrosis was found in dogs dying from 536 to 1213 days after exposure (Figures 1 and 2). The disease was characterized by a progressive and restrictive pulmonary disease. It was recognized clinically from 38 to 375 days before death, except 2 dogs died suddenly. About 80% of the dogs dying later with lung or bone tumors had histologic evidence of radiation pneumonitis and fibrosis.

Lung tumors were the primary disease at death in 4 dogs dying from 1107 to 1417 days after exposure (Figures 1 and 2). The tumors were in the peripheral portion of the lungs and were classified as adenocarcinomas or bronchioloalveolar carcinomas. They were distributed among all lung lobes and did not metastasize to organs outside of the thoracic cavity.

Bone tumors were the primary disease in 24 dogs euthanized from 1125 to 1918 days after exposure (Table 1). These osteosarcomas were located in the axial skeleton, pelvis or the proximal ends of the humerus or femur and one in the tibia. Some tumors (20%) metastasized to the lungs. Because these tumors caused paralysis or other serious locomotor problems, the dogs were euthanized from 4 to 156 days after the first observed clinical signs. Because the dogs were euthanized, survival time was slightly underestimated.

DISCUSSION

The initial lung burdens of ^{238}Pu in these two studies represent a continum of activity levels from very low to high levels. The dogs were exposed to $^{238}PuO_2$ from 1200 to 2100 days ago and only dogs with high lung burdens have shown biological response.

The earliest response, lymphopenia, was probably due to the irradiation of lymphocytes as these passed through the lung. The leucopenia, which occurred later than the lymphopenia, was possibly related to the accumulation of plutonium in the endosteum and subsequent irradiation of the bone marrow.

Radiation pneumonitis was the earliest cause of death. Seven dogs died due to radiation pneumonitis and no additional deaths from

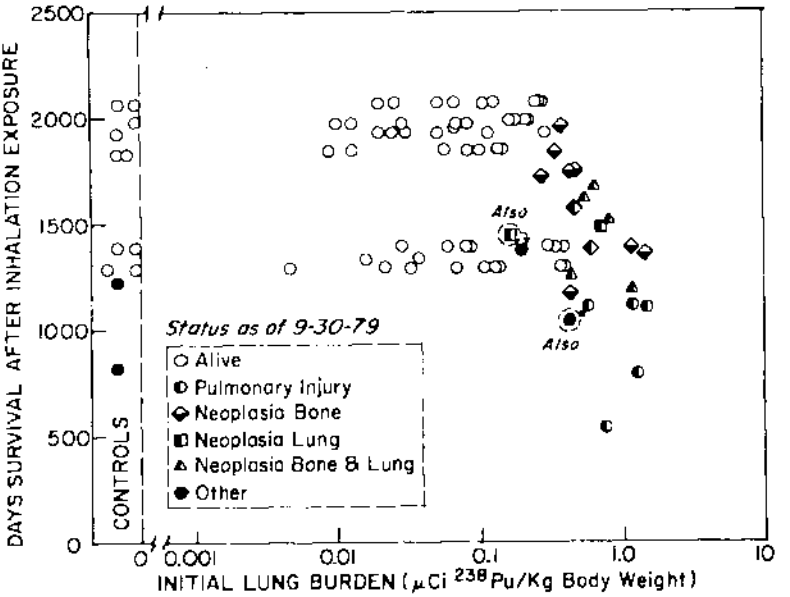

Figure 1. Survival time plotted vs. initial lung burden and major disease at death for dogs that inhaled 1.5 μm AD $^{238}PuO_2$ particles.

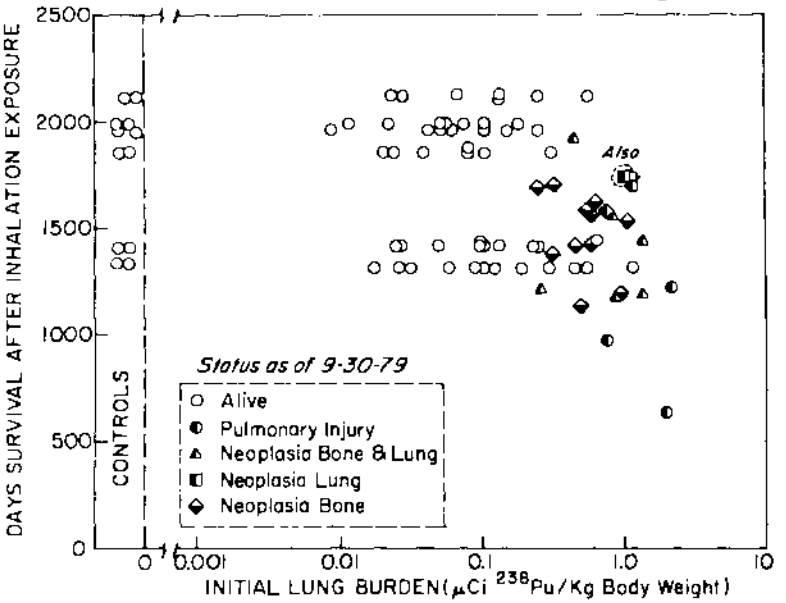

Figure 2. Survival time plotted vs. initial lung burden and major disease at death for dogs that inhaled 3.0 μm AD $^{238}PuO_2$ particles.

this cause are expected. Beagle dogs that inhaled polydisperse aerosols of $^{238}PuO_2$ had similar results with deaths from radiation pneumonitis occurring out to 3 years after exposure (2).

Lung tumors were observed beginning at 1107 days after exposure. This was earlier than the time of appearance of lung tumors with polydisperse aerosols of $^{238}PuO_2$ in Beagle dogs (2). A high incidence of lung tumors was observed in rats exposed to $^{238}PuO_2$ (4). The lobar distribution of primary lung tumors has been random.

The leading cause of death in the ^{238}Pu exposed dogs was osteosarcomas. These tumors occurred as early as 1161 days after exposure. In intravenous injection studies in Beagle dogs, osteosarcomas were found at about the same time in dogs injected with $\sim$ 1.0 μCi of ^{239}Pu/kg body weight (5). In that study, tumors doubled in size about every 12 days. That suggested that tumors were initiated about 1.3 years before death. In this study, osteosarcomas appeared earlier for a given dose than in the injection studies. This may be due to the continuous dose to the bone surface from the continuous translocation of Pu from lung to bone. Bone tumors occurred somewhat later in studies in Beagle dogs exposed to polydisperse $^{238}PuO_2$ aerosols (2). Bone tumors were not observed in rats exposed to $^{238}PuO_2$ polydisperse aerosols. This may reflect differences in the bone metabolism of plutonium between dogs and rats. In injection studies, the sites of tumor formation (axial skeleton, pelvis and the proximal end of the humerus) agreed with those in this study. These were found to be areas with the higher trabecular bone turnover rates (5).

No clear biological response differences are evident to date between the dogs exposed to 1.5 μm and 3.0 μm particles of $^{238}PuO_2$. So far, the lung and bone have been equal targets for response in the dogs exposed to the 1.5 μm particles and bone the primary organ in the dogs exposed to 3.0 μm particles. This may be related to the more uniform radiation of the lung with the 10 times higher number of 1.5 μm particles compared to the 3.0 μm particles. The average dose to organs is comparable for the two particle sizes to 1500 days after exposure (absorbed alpha dose to 1500 days: lung, 700 rads, liver 230 rads, skeleton 100 rads).

The development of dose-response curves based on local dose as well as total organ dose is expected as this study continues. Observation of each surviving dog will continue with particular concern for late effects at low dose levels.

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THE ACCURACY OF A ROUTINE PLUTONIUM IN LUNG ASSESSMENT PROGRAMME

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Systems for the detection of the radioisotopes of the transuranic elements in the human lung have been developed in several laboratories. Such systems have been in use for incident assessment since the early 1970s. There is justifiable criticism against the use of such systems for routine assessment programmes in that the minimum detectable activity (MDA) for pure plutonium 239 is such that any routine programme, at the best will produce ambiguous results and at the worst may be positively misleading. This short paper, by presenting a summary of our routine programmes over the past few years shows that meaningful conclusions can be made for the technologically important cases of reactor grade plutonium with MDAs of less than 6 nCi of alpha emitters in the lung and attempts to demonstrate that meaningful conclusions are still possible for most subjects in the case of pure plutonium 239.

THE SYSTEMS

Present systems for the assessment of insoluble particulates of plutonium in the lung rely on the detection of soft X-rays (@ 17 keV) external to the chest using phoswich detectors and proportional counters. Such systems have been described by several authors and will not be redetailed here.

SOURCES OF ERROR

For a measurement on a specific individual the statistical counting errors can be assessed together with the errors in predicting subject background and a calibration error. The sources of error and their approximate magnitudes are listed in Table 1.

Calibration error is not usually included in the overall assessed error and its relative importance (~30%) has been discussed elsewhere and will not be expanded here. For a 'standard man'* a typical standard deviation for pure plutonium 239 in the lung would be approximately 14.5 nCi giving an MDA of 29 nCi on a 2 σ criterion.

The use of single valued MDAs for such work is misleading as they depend critically on the subject's body size and a range of values between 4 nCi and 300 nCi is more illustrative.

An alternative approach is to study the distribution of 'observed' lung contents in several populations. This approach is followed here.

POPULATION OF "NORMALS"

A population of 305 male radiation workers with no history of plutonium work was studied to ensure that we would return nil lung contents for such a group. The distribution of excess counts (observed - predicted) was normal with a mean of -0.01 cpm and a standard deviation of 0.72 cpm. The 'standard' man calibration is 55 cpm/μCi plutonium 239 in the Winfrith system, ie the standard deviation of

TABLE 1. Sources of Error (approximate values for Standard Man*)

Source of Error	Pu239 equivalent
1. Counting statistics	11 nCi
2. Subject background	5 nCi
3. Body build correction	20%

*'Standard' man, approximates to ICRP standard men - a chest wall thickness of 25 mm is chosen.

this group would be approximately 13 nCi Pu239 equivalent. For high burn up plutoniums (see below) the calibration figure is of the order of 250 cpm/μCi alpha, giving a standard deviation for such a group of approximately 3 nCi.

ISOTOPIC COMPOSITION OF FUELS

Plutonium, as a potential nuclear fuel, is chemically extracted from high burn up uranium fuels when there are appreciable quantities (by weight) of the higher plutonium isotopes Pu240, Pu241, Pu242. When the mixture is expressed in terms of alpha activity, or X-ray activity, there are also appreciable amounts of Pu238 and Am241.

The X-ray/alpha emission of such fuels are usually of the order of 0.1 (cf Pu239 at 0.045) with Am241 alpha contents of between 3% and 8%. There are three main advantages in assessing such mixtures compared with the case of pure plutonium 239.

(a) The increase of specific X-ray emission.
(b) The independent assessment of Am241 via its 60 keV gamma ray.
(c) The contribution of degraded 60 keV radiations in the 17 keV band.

Point (c) may need a little amplification. Fig 1 shows the absorption curves of an Am241 impregnated lung in a realistic chest phantom with muscle equivalent overlying tissue.

The 60 keV band shows the exponential absorption characteristic of that energy but the 17 keV band has a two component structure; that of true absorption (see the Pu238 line) and that of degraded 60 keV radiations. Thus for fuel with an appreciable Am241 content there is an additional component in the 17 keV band which is not as readily absorbed as true 17 keV X-rays. For obese subjects this component predominates when the Am241 contents are above 5% by alpha and hence reduces the large predicted errors on such subjects (see below). Table 2 lists typical values of system responses for three subjects of differing body size.

All the above argument presumes a knowledge of the isotopic composition and also assumes that Am241 will behave in the lung with the same characteristics as the plutonium particulate. The latter point is certainly not true if either the material is solubilized in the lung or if there has been a long time lapse since intake (years).

RESULTS

(a) A population of 448 plutonium workers is shown in Fig 2. The

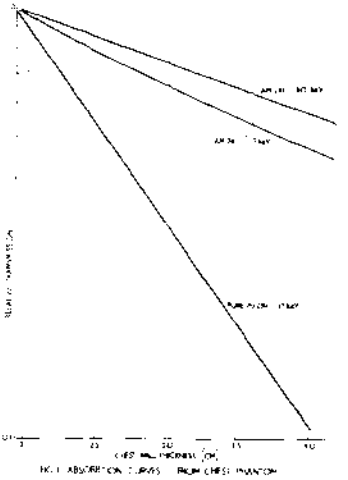

TABLE 2. Predicted system responses (from 17 keV energy band)

Composition of Mixture (% by Weight)	Subject	Chest Wall Thickness (mm)	Response (cps/μCi alpha)				Standard Deviation (nCi)	
			Pu239	Mixture				
			Xrays	Xrays	γ	Total	Pu239	Mixture
Pu238 0.1 Pu239 78.5 Pu240 18.0 Pu241 3.0 Pu242 0.4 Am241 0.2	A	18	110	237	157	394	7	2
	B	25	55	118	136	254	14	3
	C	45	8	17	81	98	96	8

isotopic composition is unknown and is taken as pure plutonium 239. The results are expressed in nCi Pu239.

The standard deviation of the population at 15.7 nCi is comparable with that of a single measurement on a standard man. For any individual it would be obviously unwise to deduce the presence or absence of plutonium at levels below ½ MPLD (8 nCi) although repeat measurements and probability analysis can help. Conclusions can be drawn as to the status of the whole group. Fig 2 also splits this data in two groups - one of body build thinner than the standard man (σ ~4.5 nCi) and one group more obese than the standard man (σ ~21 nCi).

We can now draw conclusions as to the status of the individuals within Group A using 9 nCi as a crude investigation level. Group B still presents problems as to whether specific individuals do contain plutonium and these individuals must be studied by bioassay, via urine and faecal samples. Some guidance may be obtained from Am241 in lung contents which is always measured simultaneously with the plutonium in lung assessment. The mean Am241 lung contents for subjects in Group B was 0.04 nCi with a standard deviation of 0.10 nCi.

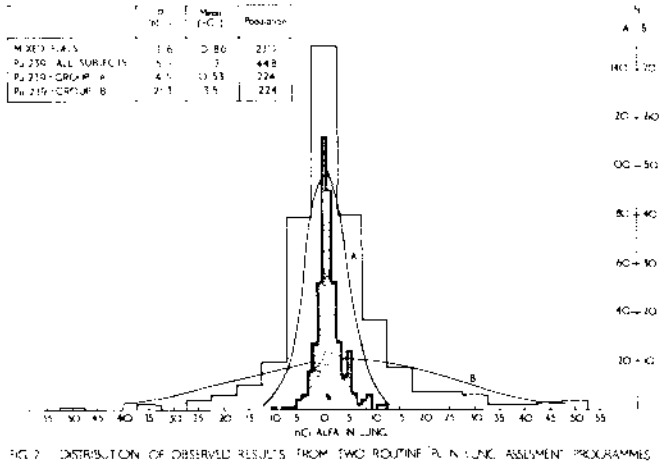

FIG 2 DISTRIBUTION OF OBSERVED RESULTS FROM TWO ROUTINE PU IN LUNG ASSESMENT PROGRAMMES

(b) A population of 230 plutonium workers where the isotopic composition, although variable, was known and was of high burn up is also summarised by Fig 2. The standard deviation of the group is now only 3.6 nCi. For such a group one can deduce the presence of plutonium with better than 96% confidence at the 8 nCi level. Because of the relatively high Am241 contents there is no longer the large difference between body builds.

CONCLUSIONS

If the isotopic composition of the plutonium is known and if this is of high burn up, a routine plutonium lung assessment programme can be used with MDA below 8 nCi alpha activity in lung. Such a programme in conjunction with routine bioassay and a defined procedure of monitoring after suspected incidents (1) is the basis of the Winfrith internal dosimetry system.

If the isotopic composition is not known, and cannot be "bracketed", any routine programme is of more doubtful use. It can be used to confirm the status of a group as a whole and can be used for individuals of a subgroup thinner than standard man with an approximate MDA of 9 nCi Pu239. For the more obese individual, the limitations of direct lung monitoring with no prior knowledge of the isotopic composition of the contaminant, are sadly obvious. For such individuals, decisions as to their 'plutonium' status are still predominantly based on biological monitoring.

Any subject who, as a result of a routine lung count, gives indications that plutonium may be present in the chest region is remonitored. His whole history is re-examined together with personal air sampling and biological monitoring results. It is on a combination of all such methods that a conclusion as to his plutonium 'status' is made.

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RETENTION AND EFFECTS OF ^{239}Pu IN THE TREE SHREW (TUPAIA BELANGERI)

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For the improvement of the evaluation of risk to man condiserably more information on the metabolism and effects of Pu-239 is needed in primates. The tree shrew (Tupaia belangeri) is considered to be the most primate-like non-primate or the most primitive of the living primates (e.g. 1). They are as small as rats, relatively easy to breed and maintain and can live from 8 to 14 years. One of the aims of this pilot study is to compare the retention of Pu-239 in tupaias with that in rodents and larger primates in order to evaluate the usefulness of tupaias for toxicological studies.

METHODS

The animals used were 19 female tree shrews, 8-18 months old, (150-200 g) from the breeding stock of the Institute for Animal Physiology, University of Bayreuth. Plutonium-239 citrate (The Radiochemical Centre, Amerham, UK), 0.5 uCi/kg, in essentially monomeric form was injected intramuscularly. The animals were sacrificed at the time intervals indicated in Fig. 1. The radioactivity remaining at the injection site and in the organs was determined by liquid scintillation counting (2). The distribution of the nuclide within the different parts of the skeleton was also measured. Thus, the data given in Fig. 1 for skeleton were not calculated from the activity in the femur but represent measured data for the whole skeleton. The data in Fig. 1 are expressed as a percentage of the dose absorbed from the injection site up to the time of sacrifice. The weight of the animals was followed continuously and the organs were carefully inspected at sacrifice. Sections for histological examination were prepared and hematological and chemical tests with blood were performed by routine methods.

Cells from different organs (muscle, kidney, spleen, thymus, and blood leukocytes) were established in tissue culture as described previously. The cell-free supernatants from these cultures were assayed for reverse transcriptase as indication for the expression of retroviruses as described previously (3).

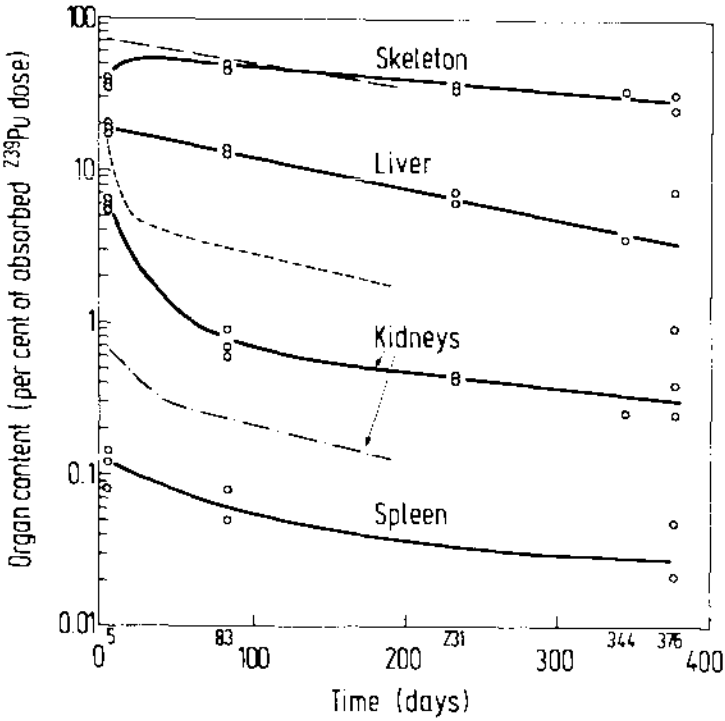

Figure 1. Retention of i.m. injected Pu-239 in tupaia belangeri (tree shrew). Each point represents one animal. Dotted lines are from rats (4).

RESULTS AND DISCUSSION

The retention of Pu-239 at the injection site decreased from ∿ 15 % at day 5 to ∿ 5 % at day 83, this value remained virtually constant thereafter. The retention in the organs is shown in Fig. 1, in which the corresponding retention functions for rats (4) are represented as dotted lines. As can be seen, the initial deposition in skeleton and liver is not very different from the well known picture in rats. The half life in tupaia skeleton is somewhat longer (one year as compared to 215 days) which may be due to the different biological age of the skeleton at the time of injection. During the first year, the elimination of Pu-239 from the liver can be described by a single exponential with a half life of 150 days, whereas ∿ 75 % of the initially deposited nuclide are eliminated from the rat liver with a half life of < 10 days. Initial deposition in the kidneys is much higher than in rats but the elimination rates are similar.

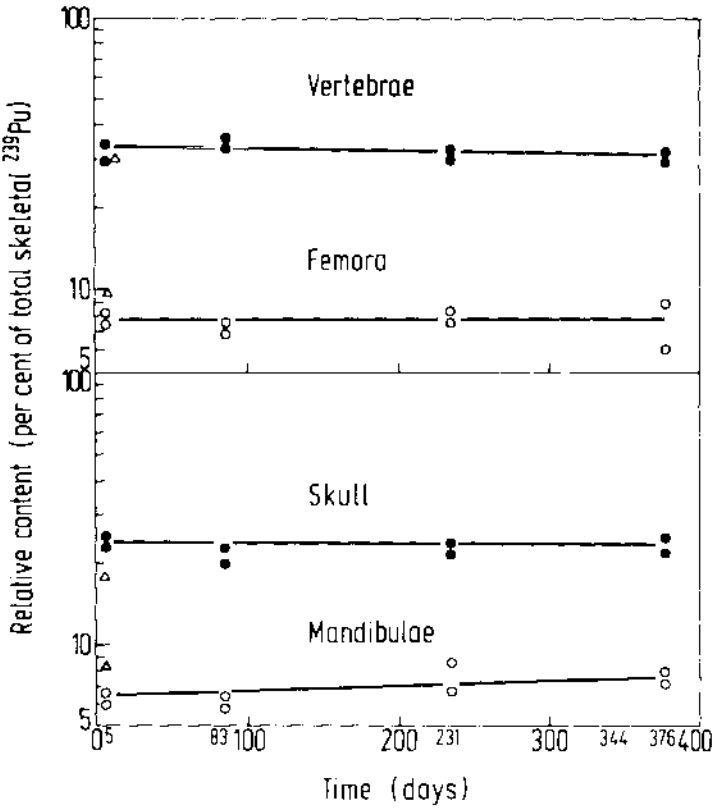

Figure 2. Retention of Pu-239 in different bones of the tupaia (tree shrew) skeleton expressed as a fraction of Pu-239 in total skeleton. Triangles are data from rats for Am-241 (5).

The retention in four bones is presented in Fig. 2 which also shows data for rats given Am-241 (5), (data for Pu-239 were not available). The fraction deposited in the different bones remained constant during the first year after injection, indicating a low rate of bone remodelling. The factor for calculating total Pu-239 in skeleton from activity in one femur was 27.

No statistically significant change in weight and no gross signs of toxicity were observed during the first year, except for one animal, which died for unknown reasons. No macroscopic or histological changes in the organs have been observed (till d 231). Hemogl., hematocrit and erythrocyte counts remained normal. There was a rise of glutamate-pyruvate- and oxalate-transaminase as well as of alkaline

phosphatase in the three animals sacrificed at day 376. A total of 36 different cell cultures (from nine animals) were tested and found to be negative for retroviruses.

This study provides further information to suggest that the retention of plutonium in rat liver is exceptionally short when compared to other animal species, hamster, primate, dog. However, if the tupaia is also regarded as a lower primate, the rate of elimination of transuranium elements by sub-human primates tested so far is still much faster than that predicted for human liver (ref. in 6,7). These findings strenghten our view that the realistic assessment of the risks to man from deposited transuranium elements must be based on a clear understanding of the biochemical mechanisms underlying the deposition and elimination of these elements in various animal species including man.

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THE EFFECT OF OXIDATION STATE ON THE ABSORPTION OF INGESTED OR INHALED PLUTONIUM

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Larsen and Oldham found that chlorine, at the concentrations found in municiple water supplies, can oxidize quadrivalent plutonium to its hexavalent state (1). Since studies in this laboratory had shown one thousand times more $^{239}Pu(VI)$ was absorbed than $^{239}Pu(IV)$ (2), they suggested that the maximum permissible concentrations (MPC), apparently based on data from Pu(IV), should be lowered.

Our initial experiments performed with $^{238}Pu(VI)$ nitrate did not support those earlier results. This suggested that the conditions of fasting and oxidation used in those studies may have been responsible. Absorption of plutonium either by gavage or inhalation was compared in fasting and nonfasting rats to determine if the intestinal contents influence absorption of plutonium that was injected intragastrically or swallowed as a result of clearance from the lungs.

METHODS AND MATERIALS

Wistar female rats weighing about 200 g received plutonium nitrate (pH2) by gavage or by nose-only exposure from a nitric acid aerosol generated by a Lovelace nebulizer (3). Fasted rats were deprived of food 18 hours before Pu gavage and for 72 hours following it. Animals exposed by inhalation were fasted either before, or both before and after treatment with Pu. Excreta was collected daily from gavaged rats for four days.

All animals were killed five days after treatment. Femurs were removed from all animals and the total skeletons analyzed from many with which a femur factor was derived to determine total bone Pu content. The skin and GI tracts were discarded. Carcass values were determined by a summation of bone and soft tissue, excluding the liver and lung values. High lung values in gavaged animals indicated poor injections and the data from these animals was rejected.

The ^{239}Pu was purified by anion exchange on a Dowex MSA-1 resin and oxidized to its hexavalent state by passing a stream of O_3, O_2 and Ar through a 0.4M HNO_3 solution for six hours. It was shown to be 100% $^{239}Pu(VI)$ by spectrophotometric analysis. Solutions that included the holding oxident $K_2Cr_2O_7$ were made 0.015 M by dilution. Gavaged animals received 1.0 ml of a solution containing 0.5 mg ^{239}Pu, 97% of which was filterable through a 0.01 μm filter at the time of treatment. The dose administered by gavage was 30 μCi and by inhalation 5 μCi.

Plutonium was analyzed by a modification of the Keough and Powers method (4). Carbon-free aliquots in mixture of 1.0% boric acid were mixed in a scintillation solution containing 1,4 bis-2-[5 phenyloxazolyl] benzene, 2,5,-diphenyloxazole (PPO), Triton X-100 toluene and water.

RESULTS

The data obtained by Weeks et al (2) after intragastric administration of $^{239}Pu(VI)$ are shown in Figure 1 along with data that we obtained simulating conditions used by those investigators. Other

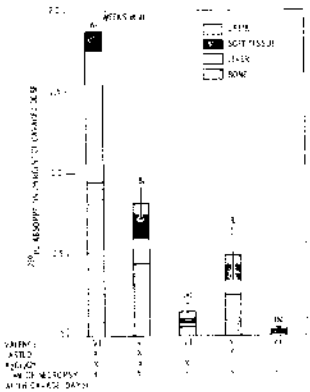

Figure 1. Absorption of $^{239}Pu(VI)$ by fasted and unfasted rats after intragastric administration of ^{239}Pu nitrate.

groups are included to show the effect that ad libitum feeding and an absence of the holding oxident, $K_2Cr_2O_7$ had on ^{239}Pu absorption from the GI tract. The fasting period lasted from 18 hours before gavage until 72 hours thereafter. Our absorption data under those conditions amounted to about half that of Weeks, possibly because their solutions were more acid, pH 1 versus pH 2, and their $K_2Cr_2O_7$ may have been more concentrated. Feeding reduced absorption 18-fold and the combination of feeding and elimination of dichromate from the solution reduced it about 26-fold.

Results obtained by exposing groups of rats to aerosols of either $^{239}Pu(IV)$ or $^{239}Pu(VI)$ (Figure 2) indicate that there was increased retention of plutonium after exposure to $^{239}Pu(VI)$ in comparison to $^{239}Pu(IV)$. The absence of food either before, or both before and after the inhalation exposure had no effect on the amount of ^{239}Pu retained by the liver and carcass.

DISCUSSION

Plutonium in its hexavalent state may under certain conditions be more readily absorbed from the GI tract. However, the experimental conditions in which increased uptake occurred, i.e. fasting for 90 hr, high acidity (pH 1-2) and a dose of 130 uCi/kg, are unlikely to occur in human exposure. The oxidation state may also influence absorption after inhalation because of the large fraction entering the bowel as a result of swallowing Pu cleared from the

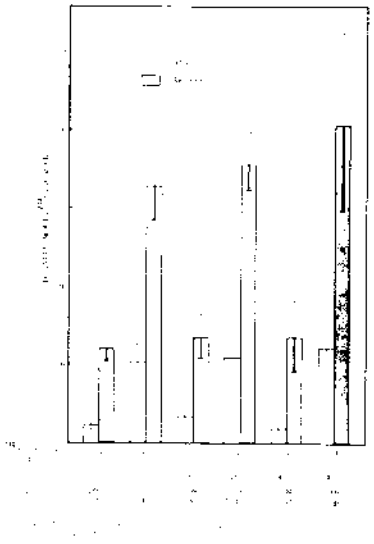

Figure 2. Retention of ^{239}Pu by fasted and unfasted rats after inhalation of either $^{239}Pu(IV)$ or $^{239}Pu(VI)$ nitrate.

lung. Although the Pu dose inhaled was lower than the dose gavaged (25 μCi/kg) some of it probably entered the stomach in its hexavalent state. Absorption from the GI tract was not higher than when $^{239}Pu(IV)$ was inhaled, even when the intestinal contents were depleted by fasting. The absence of an effect of food deprivation suggests that the increased absorption of plutonium was due to translocation of $^{239}Pu(VI)$ from the lung and not to an increase in transport across the GI tract.

This data support the observation made in the gavage studies that the MPC for drinking water, apparently based on the absorption of Pu(IV), is adequate for Pu(VI) ingestion in quantities that may be expected in the environment. Absorption of Pu(VI) from the lung, however, is higher than that of Pu(IV).

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PULMONARY CARCINOGENESIS FROM PLUTONIUM-CONTAINING PARTICLES

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The potential pulmonary effects of inhaled plutonium have been summarized in recent years (1,2) and the need for further animal research has been emphasized. The results presented here are the outgrowth of an effort to assess the tumorigenesis of focal plutonium sources in the Syrian hamster respiratory tract (3,4). Because localized radiation (hot spots) in bone was known to be more tumorigenic than diffuse radiation, it was thought that this same phenomenon may prevail in the lung tissue. Based primarily upon studies with beta irradiation of the skin (5) models were developed as guidelines for our "hot particle" research.

MATERIALS AND METHODS

To create well-controlled plutonium sources localized in specific numbers in the Syrian hamster lung it was decided to use the intravenous (IV) route of administration. Ceramic spherical particles of zirconium dioxide were manufactured such that fixed small amounts of plutonium could be homogeneously incorporated in them, to control the strength of each particle's radiation field (6). The particles were uniformly 10 μm in diameter (6) and were injected into the jugular vein (7), after which they lodge in the lung capillary bed. A ^{57}Co tag was also added so that retention characteristics could be determined through periodic whole-body counting.

The aerosol particles for inhalation (INH) studies were of two different chemistries. In the earlier work it was decided to nebulize a mixture of the ZrO_2 sol used in the hot particle microsphere manufacture; into the sol was incorporated the desired amounts of ^{238}Pu or ^{239}Pu and ^{57}Co. The sol was nebulized (8) and the droplets passed through a heating column at $\sim 900^{\circ}C$. The animals were exposed in a nose-only setup (9). The resulting aerosol was polydispersed with a aerodynamic mass median diameter of $\sim$ 1.8 μm and a geometric standard deviation of $\sim$ 1.8. The second type of aerosol was from plutonium dioxide generated in a similar fashion, but the starting material was a suspension of $^{239}PuO_2$ in distilled water. The aerosol samples for analysis were collected with cascade impactors (10) or electrostatic precipitators (11).

The Syrian hamsters were allowed to live out their lifespan and were sacrificed only when moribund. They were necropsied as soon as feasible, and routine gross and microscopic pathological examinations were performed (12, 13).

RESULTS

Data on the tumor incidences in all three (IV and INH) studies are presented in Tables 1-3. The histology slides for the PuO_2 studies have not been completely read so the available results only are presented in Table 3. The types of tumors are indicated in Tables 2 and 3, primarily to show the ratios of adenomas to adenocarcinomas.

TABLE 1. Pulmonary neoplasm incidence following intravenous injection of microspheres in Syrian hamsters

No. of Spheres Per Hamster	Lung Burden (nCi)	No. of Hamsters	Fraction of Lung Irradiated (%)	No. of Tumors	Lung Tumor Incidence (%)
CONTROL	0	521	0	3	0.6
2360	140	68	1	0	0
10 900	97	17	5	0	0
58 800	120	160	28	19	12
312 000	130	25	80	2	8

TABLE 2. Pulmonary neoplasm incidence following inhalation of Pu-ZrO_2 aerosol particles by Syrian hamsters

Initial Lung Burden (nCi)	No. Of Hamsters	Tumor Incidence (%)	Adenoma	Adeno-Carcinoma	Squamous Cell Carcinoma
0	144	0.7	1	0	0
6	40	5	2	0	0
8	43	12	5	0	0
76	50	28	11	6	0
87	50	40	12	8	0
101	44	50	10	9	3

DISCUSSION

It is obvious that plutonium alpha irradiation distributed focally is not tumorigenic (Table 1). When less than 5% of the pulmonary tissue receives the radiation dose, there are no observed

tumors at death. More diffuse irradiation does lead to the formation of tumors.

TABLE 3. Pulmonary neoplasm incidence following inhalation of PuO_2 aerosol particles by Syrian hamsters

Initial Lung Burden (nCi)	No. of Hamsters	Tumor Incidence (%)	Fraction Slides Read*		
			Adenoma	Adeno-Carcinoma	Undifferentiated Tumors**
0	50	0	0	0	0
40	63	4	1/23	0	0
96	66	13	6/54	0	1/54
110	60	7	1/46	1/46	1/46
144	65	16	7/49	1/49	0

* All animals in this study have not been processed; hence, the incidences are based upon fewer than the total exposed in each group.

** Carcinomas and Sarcomas

Inhalation studies are much more productive in the induction of lung tumors, as shown in Table 2. A trend of increased tumor incidence with increasing initial lung burden is obvious. There is also an apparent trend from adenoma induction to more invasive types of tumor, with increasing initial lung burden. The same dosage-incidence trend may be forthcoming in the PuO_2 studies (Table 3), but more data await analysis. A currently unexplained effect is the apparently greater tumor yield produced by Pu-ZrO_2 compared to PuO_2. Averaging the last 3 dose groups of the latter gives 12 ± 3% tumors from a mean lung burden of 117 nCi while averaging the last 2 dose groups of the Pu-ZrO_2 aerosol gives 45 ± 6% from 94 nCi. Particle size and residence time do not account for any difference.

SUMMARY

Plutonium administered as an alpha radiation source to the respiratory tracts of Syrian hamsters has resulted in various incidences of neoplasia. Adenomas are the primary lung tumor observed, but adenocarcinomas are also prevalent.

ACKNOWLEDGEMENT

Many individuals in the Toxicology Group played an important role in this work, but G. A. Drake and J. E. London have carried on the experimentation and collection of data throughout the studies.

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SUBCELLULAR DISTRIBUTION OF ^{239}Pu IN THE LIVER OF RAT, MOUSE, SYRIAN AND CHINESE HAMSTER

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It is well known that the biological half life of transuranium elements in the liver of mammalian species varies from a few days to many years (ref. s.1). However, the reasons for these differences are unknown. The aim of our studies is to elucidate the biochemical mechanisms responsible for these species differences as a part of our attempt to improve risk estimation following incorporation of transuranium elements into man through a profound understanding of the biochemistry of these elements. We have chosen rats and mice as models for a short and two hamster species as models for a long biological half life of Pu-239 in liver.

METHODS

The animals used were adult females of the following strains: Rats (Heiligenberg strain), mice (NMRI), Syrian hamsters (commercial strain) and Chinese hamsters (Breeding stock of Institute for Zoology, Darmstadt). Radiochemically pure Pu-239- and Fe-59-citrate injection solutions were injected i.v. (5-10 µCi/kg Pu-239, 50 - several hundred µCi/kg Fe-59). On the sixth day 750 mg/kg of the non-ionic detergent Triton WR1339 were given i.p. and sacrifice took place on the tenth day. Triton WR1339 causes a shift of the density of the lysosomes from $\sim$ 1.2 to $\sim$ 1.1 g/cm^3 and any lysosomally-associated material can be recognized by a parallel shift. Control animals received 0.9 % NaCl. After differential centrifugation of liver homogenates, a fraction designated MLP was obtained, which contained most of the formed cell elements except the nuclei. Aliquots of this MLP fraction were centrifuged in a linear sucrose density gradient for 4 or 16 hours at 88 000 av. g. Radioactivity and marker enzymes (s. Figures) were determined in all fractions obtained after isopycnic centrifugation. Details of the methods are described elsewhere (2).

RESULTS AND DISCUSSION

The MLP fraction contained more then three quarters of the total cellular radioactivity. The results obtained by isopycnic centrifugation of that fraction are shown in Figs. 1 and 2. Certain results are common to all species: The extent to which the profile of the lysosomal marker, acid phosphatase, is shifted to light densities by injecting Triton WR1339 is very similar. The profile of glutamate dehydrogenase (mitochondria) is not influenced by Triton WR1339 and

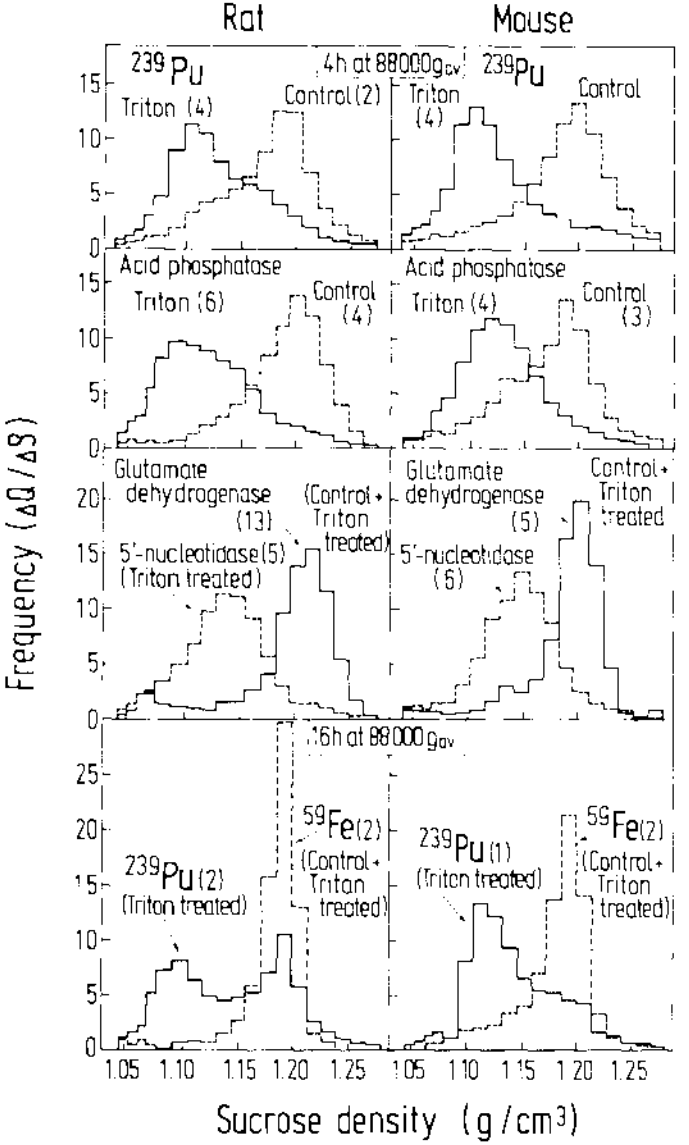

Figure 1. Distribution of radioactivity and marker enzymes (s.text) after centrifugation of the MLP fraction from liver cells in a sucrose density gradient. ΔQ: fractional amount of constituent found in that section; Δs: density increment from one fraction to the other. The area under each section is proportional to the fractional amount and the total area is one. Values in brackets represent number of experiments.

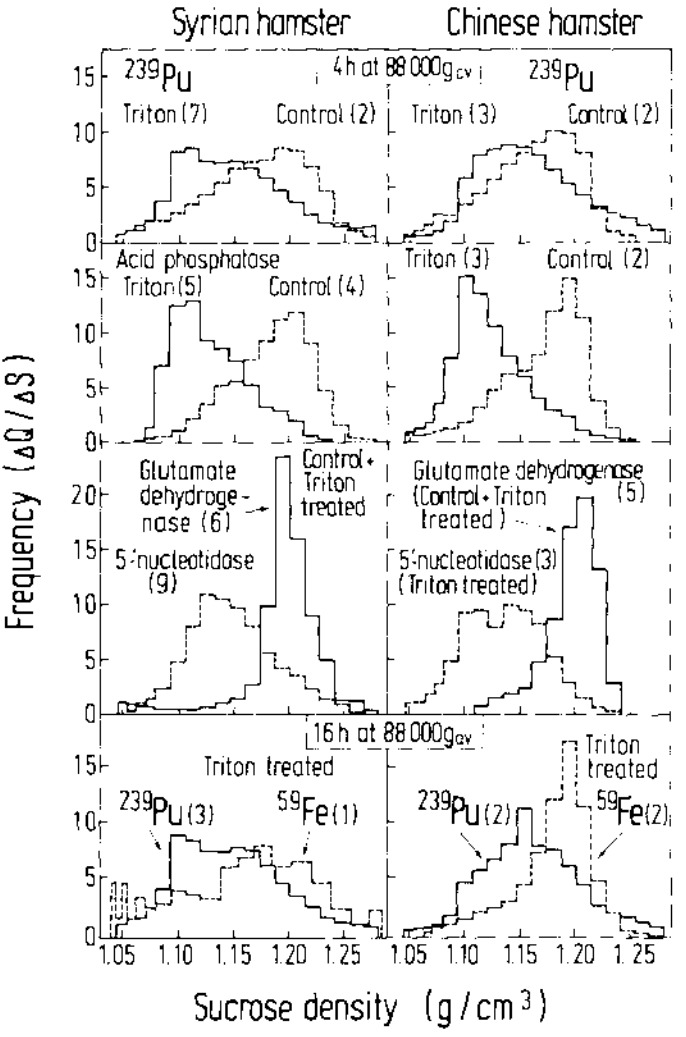

Figure 2. Distribution of radioactivity and marker enzymes after centrifugation of the MLP fraction from liver cells in sucrose density gradients. For presentation s. Fig. 1.

data for controls and treated animals could be combined. On the other hand, the Pu-profiles are considerably broader and the fraction which is unequivocally shifted is considerably smaller in both hamster species as compared to rats and mice (upper row of Figures). In Chinese hamsters, the peaks of Pu-239 and acid phosphatase after Triton treatment are not even congruent; a distinct shoulder at $\rho \sim 1.5$ g/cm^3 is visible in the Pu profile for Triton treated Syrian hamsters. With a centrifugation time of 16 instead of 4 hours (lower row of Figures) the profiles of the marker enzymes remain at virtually the same density. Again clear discrepancies between the Pu-profiles of the various species occur. In rats, and to a lesser degree also in mice, a second peak develops at the same density at which Fe-59 equilibrates. The profile for Pu in Chinese hamsters becomes unimodal with a peak at ~ 1.5, where a minimum exists in the corresponding profile for rats. The clear relationship to the Fe-profile seen in rats was not observed in Chinese hamsters. At this point it should be mentioned that we have seen only minor, or even no, differences between the Fe-profiles from control or treated animals, which, therefore, could be combined for presentation. There is no reason to assume any important association of Pu with plasma membranes, represented by 5'-nucleotidase, in rats, mice and Syrian hamsters. However, the role of the plasma membrane needs to be clarified by further experiments, especially in Chinese hamsters.

If the parallelism between the shift of Pu-239 and acid phosphatase is taken as a measure for the extent of lysosomal binding, there is clear evidence for association with these organelles for rats, mice and to some degree, also for Syrian hamsters. However, the data for Chinese hamsters are equivocal, at least at the tenth day, and it is possible that neither lysosomes nor iron-binding proteins are major sites of fixation in this species. It is interesting to note that the indications for lysosomal association are unequivocal in both rats and mice, the species with rapid Pu elimination, whereas these organelles appear to become less important in the species with longer retention, Syrian and especially Chinese hamster. However, investigations of the changes with time in the subcellular distribution and on the exact nature of the binding components for Pu-239 in the livers of all species, but especially of Chinese hamsters, are needed before further conclusions can be drawn.

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EUROPEAN INTERLABORATORY TEST PROGRAMME FOR LUMINESCENCE DOSEMETER SYSTEMS USED IN ENVIRONMENTAL MONITORING

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INTRODUCTION

A necessary task in the field of environmental monitoring is the measurement of small doses in the range of the natural radiation level. In contrast to personal dosimetry, the demands placed on the application of TLD systems in environmental monitoring include longterm exposure periods, high accuracy of dose measurement, low fading and long-term reproducibility.

With respect to the high accuracy of dose reading there is a need to establish the dosimetric properties of each dosemeter system applied in the laboratory. This was the reason that an interlaboratory test programme for TLD and RPL was prepared which command the same kind of experiments for all participants. Laboratories with longterm experience in personnel monitoring participated at the first experiment [1]. In the European intercomparison programme of dosemeter systems for environmental monitoring 1978/79 more than 17 laboratories participated with 45 dosemeter systems.

The results of the intercomparison programme allows to compare the dosimetric properties of different dosemeter systems applied today in personnel and environmental monitoring and reflects more or less the actual state of the art.

STANDARD TEST PROGRAMME

The dosimetric properties of a dosemeter system are given by the selected combination of reader, detector and read-out procedure. Differences in the dosimetric properties of the systems resulted from

- the type of reader and the individual properties of the photomultiplier component in the reader,
- the dosemeter material, the activator, the matrix material, the form, thickness and mounting of the detector, as well as the history of the batch,
- the read-out procedure, in particular the heating cycle and the maximum heating temperature which have been selected,
- the thermal treatment of the detector prior to measurement (preheating) or the thermal treatment used prior to re-use (annealing).

Any change in the above-mentioned parameters can change the dosimetric properties of the entire system. Even when reader types, detectors and read-out methode are identical dosemeter systems can differ in sensitivity up to a factor of 40, and in zero dose reading up to a factor of 10.

The basic experiments of the standard test programme includes

the examination of the following quantities

- the dark current reading α_D and the zero dose reading α_U defined as the dose reading during read-out without dosemeter or with the unirradiated dosemeter, respectively,
- the dose at the lower detection limit D_{LDL} definded as three times the standard deviation (3s-value) of α_U,
- the relative standard deviation s(D) as a function of exposure, in a range of 1x to 1000xD_{LDL}
- the uniformity of the detector sensitivity within a batch,
- the long-term stability of the dosemeter system during ten days,
- the long-term change in the zero dose reading α_U,
- long-term change of dose reading due to fading during a ten days storage at 70°C.

For all characteristic quantities to be examined, the test programme provides measurement runs to be carried out with a batch of 10 dosemeters, taking 10 reading for each dosemeter or repeating the measurement cycle ten times (e.g. Annealing, pre-heating, read-out, irradiation). Bearing in mind the objective of obtaining comparable results, the relevant dose range has been related to a multiple of the lower detection limit D_{LDL} of the individual dosemeter system.

With respect to the interpretation of the results the reader type and the consecutive system number of all systems participating in the intercomparison are presented in Table 1. More data will be published [2].

RESULTS

The frequenty distribution of the batch uniformity presented in Fig. 1 shows that the detector response of most of the TLD batches scatters by more than ±3%. A higher reproducibility of dose measurement can be expected only on the bases of a calibration exposure in order to establish the individual response of each TLD detector.

Fig. 2 shows the mean value of the zero dose reading $(\alpha_D+\alpha_U)$ and 3s-value of zero dose reading α_U for the different dosemeter systems. In practice the value $(\alpha_D+\alpha_U)$ is found to be the dose reading contribution in mR which will be subtracted from the dose reading. Most of

Table 1: Reader types

READER TYPE	NUMBER
HARSHAW 2000 A-B	4
2000 B-C	5
2000 C	3
2000 P	1
2271	1
Pitman Inteco	12
Studsvik 1313 B	5
Risø	3
Saphymo	4
National LD 605A	2
LD 710P	1
Kyokko TLD-1200	1
CRIP/TLD 048	2
TND Automat	1

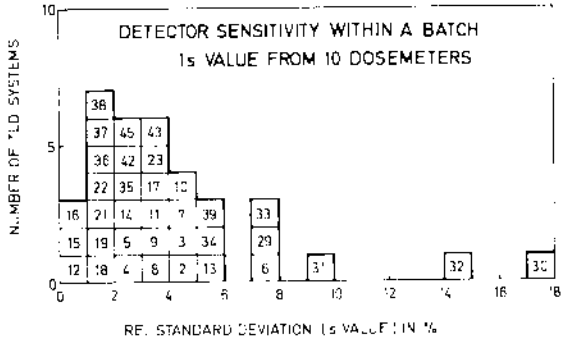

Fig.1 Batch uniformity of different TLD systems (No of system)

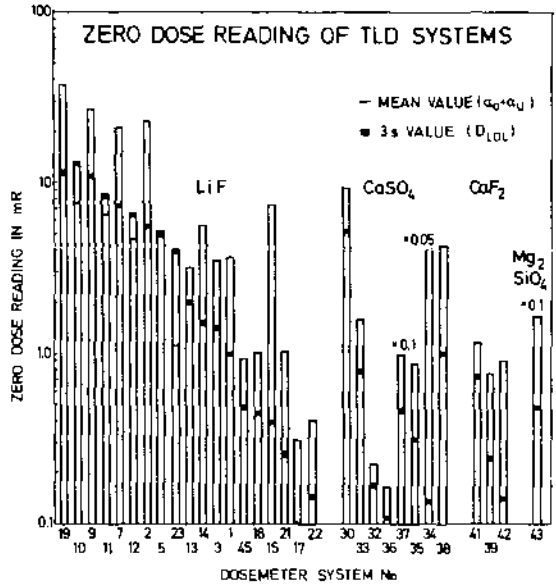

Fig. 2 Zero dose reading of TLD systems (No of system)

the LiF systems show ($\alpha_0+\alpha_U$) values between 3 and 10 mR, CaF_2 dosemeter systems in the order of 1 mR compared to $CaSO_4$ dosemeters between 0.08 and 0.2 mR.

Taking into account the individual response of each detector or a batch calibration the reproducibility of dose reading is presented in the Fig. 3 for an exposure to 30 mR -which is approximately the half-value of the annual dose from the natural radiation background- and for an exposure to 100 x D_{LDL}.

In the higher dose range the standard deviation (2s-value) of most of the TLD systems has been found to be lower then ±6%. With respect to an application in environmental monitoring, on the other hand, at 30 mR 9 different TLD systems show a reproduciblity of ≤4%, one system better than 2% (2s value).

Due to the subtraction of the zero dose reading the relative standard deviation is found to be high for low exposures. The relative s-value decreases as a function of exposure reaching a constant value at higher exposures.

For LiF and $Li_2B_4O_7$ systems a constant reproduciblity starts in the exposure range 10 mrem to 100 mrem, for $CaSO_4$ dosemeter systems already for exposures between 1 mrem to 10 mrem. A high sensitive dosemeter system, on the other hand, is not a guarantee of high reproducibility in the dose range of 100 mrem.

CONCLUSION

The main objective of the interlaboratory test programme is to calibrate and establish a dosemeter system for an application in environmental monitoring. On the basis of an extended experimental programme the standard test serves as a supplement to other calibration

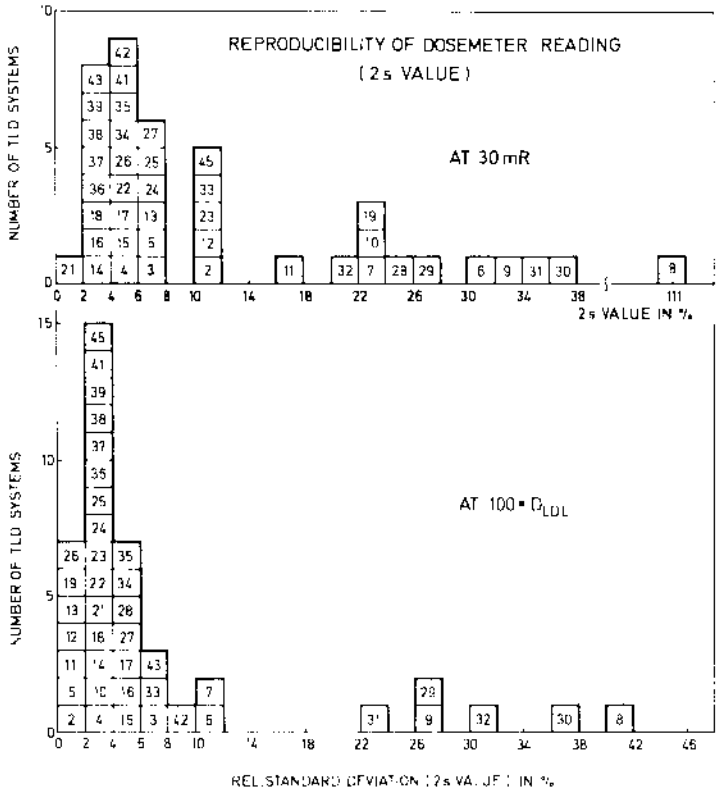

Fig. 3 Reproducibility of TLD systems (2s-value of 10 dosemeters)

exposures or intercomparison experiments in the environment [3]. The results of the standard test programme should be a help for each participating laboratory in finding out whether the properties of the dosemeter system result in a sufficient accuracy of dose reading and fulfill the recommended figures given by national standards and regulations [4, 5].

The study has been sponsored by the Commission of the European Communities.

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Proc. 5th Int.Conf.Luminescence Dosimetry Sao Paulo (1977) 365
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SIMULTANEOUS SENSITISATION AND RE-ESTIMATION IN THERMOLUMINESCENT LiF

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There is a continuing need for thermoluminescent (TL) phosphors of high sensitivity and approximate tissue equivalence for a variety of dosimetry applications in radiological protection. Lithium fluoride (TL) has been widely used for many years but has relatively low sensitivity for some applications such as environmental, skin and interface dosimetry. It is possible however to increase the sensitivity by a process of radiation sensitisation.

RADIATION SENSITISATION

It has been known for some time that if LiF is subjected to radiation doses of approximately 10^3Gy and a subsequent thermal anneal (300°C for 1 hour for example) the sensitivity is increased by a factor of up to six (1). Unfortunately this process of radiation sensitisation produces a high residual background signal which precludes its use for low dose measurements. Mayhugh and Fullerton (2) suggested a modified procedure for sensitisation to overcome this problem which involved an annealing procedure which incorporated simultaneous UV exposure and heat treatment. Their study did not however contain an analysis of the statistical fluctuations in the remaining background, which dictates the minimum detectable dose.

RE-ESTIMATION

In addition to the measurement of low doses it is often desirable to be able to re-estimate (confirm) the measured dose, perhaps because of instrument malfunction or suspected over exposure of personnel. Re-estimation involves the transfer of tightly bound trapped electrons, which remain after the initial read out, to the less stable TL traps. This is normally achieved by subjecting the phosphor to UV radiation, often at elevated temperatures. Bartlett and Sandford (3) studied the re-estimation capability of sensitised LiF using both conventional and UV/thermal annealing procedures and concluded that simultaneous sensitisation and re-estimation was not possible. Their data extended only to doses of 1Gy and used lower UV intensities than in the study reported here. This paper outlines and extends our previous studies of radiation sensitisation and re-estimation (4, 5) and shows that the two processes are definitely not incompatible.

SIMULTANEOUS SENSITISATION AND RE-ESTIMATION - THEORY AND PRACTICE

In an attempt to understand the reason for the effectiveness of UV/thermal annealing procedures an absorption spectra study was carried out using Harshaw ^{7}LiF chips (0.9 mm thick). Figure 1a illustrates typical spectra for chips which have received sensitising doses of 2 x 10^{3}Gy and then been subjected to conventional and UV/thermal annealing. Following the conventional anneal a broad absorption band remains at about 225 nm which has been associated with a $Mg^{++}-F$ centre pair - a so-called Z_3 centre (6). The reduced intensity of this band following the UV/thermal anneal is quantitatively correlated with the reduction in background which this treatment provides. The removal of this band is associated with only a small reduction in sensitisation, as pointed out also by Stoebe and Watanabe (7). Sensitisation is presumably associated with deep traps which can only be observed by absorption measurements below 200 nm.

Figure 1b illustrates the changes in absorption spectra which take place following re-estimation. After an initial read out the Z_2 band ($Mg^{++}-F^{-}$) at about 310 nm is considerably reduced. Following re-estimation these centres are repopulated at the expense of F, and possibly Z_3, centres.

Since the radiation sensitisation and UV re-estimation are associated with different trapping centres the two processes should be mutually compatible. Figure 2 shows the results of simultaneous sensitisation and re-estimation with Harshaw ^{7}LiF chips (0.9 mm thick). For sensitised chips the statistical fluctuation in the re-estimation background is such that the minimum re-estimatable dose (2σ) is $\sim$ 0.2Gy as compared to $\lesssim$ 0.01Gy for unsensitised LiF chips. The high UV re-estimation background may be due to remaining Z_3 centres. This limitation is not prohibitive since re-estimation is most important for the confirmation of high doses. It is possible that the background signal may be reduced by a judicious choice of the UV wavelength for both the annealing and re-estimation exposures. The intense 254 nm line of the low pressure mercury lamp has been widely used, in studies such as these, purely as a matter of convenience and is unlikely to be the optimum choice. Our work is continuing in this direction with the use of a high intensity UV monochromator facility.

Figure 3 shows the dependence of re-estimation (expressed as a percentage of the initial read out signal) on UV exposure time, temperature and UV intensity. The complex relationship between these parameters explains why so many different values of re-estimation have been reported in the literature. It is clear too that if the results of different workers are to be compared it is necessary to measure UV intensities as accurately as possible. This is not an easy task in the 200-250 nm region. Calculated intensities, based on manufacturers specifications are very unreliable. Calibrated thermopiles and similar detectors, together with filters to cut out stray light, are essential for reliable UV dosimetry.

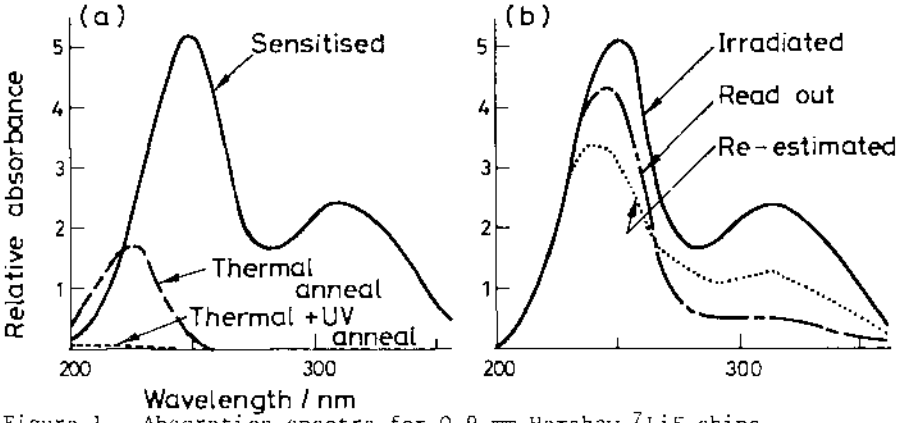

Figure 1. Absorption spectra for 0.9 mm Harshaw ^{7}LiF chips. (a) Sensitised at 2×10^{3}Gy ——; sensitised + thermal anneal (300°C, 1 hour); sensitised + simultaneous thermal/UV anneal (300°C, 6 mWcm^{-2} of 254 nm UV, 1 hour).----. (b) Unsensitised, Read out at 260°C, re-estimated at 150°C for 2 minutes using 5 mWcm^{-2} of 254 nm UV.

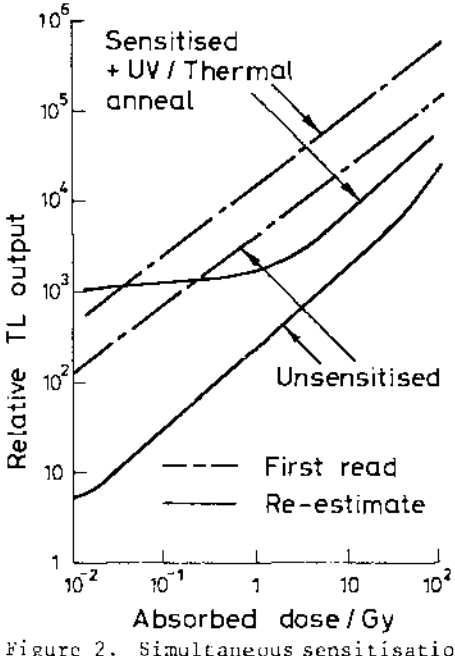

Figure 2. Simultaneous sensitisation and re-estimation in ^{7}LiF Harshaw chips. Sensitisation data as Fig. 1, re-estimation, 5 mWcm^{-2}, 100°C, 5 min.

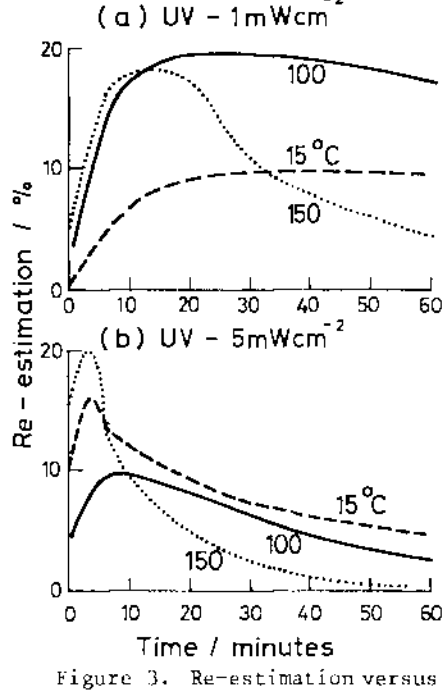

Figure 3. Re-estimation versus UV (254 nm) exposure time for ^{7}LiF Harshaw chips.

SUMMARY

Radiation sensitisation is a useful technique which can reduce the minimum detectable dose for LiF (TL) by a factor of 3, and retain the possibility of dose re-estimation. Further work is necessary however to reduce the background signal associated with re-estimation. The smallest dose which can be re-estimated with sensitised ^{7}LiF chips (0.9 mm thick) is about 0.2Gy at the present time.

ACKNOWLEDGEMENT

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SKIN DOSE ASSESSMENT IN ROUTINE PERSONNEL BETA/GAMMA DOSIMETRY

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1. INTRODUCTION

The International Commission on Radiological Protection (1) recommends a tissue depth of 5-10 mg cm^{-2} as the most appropriate depth for skin dose assessment. An ideal dosemeter for measurement of skin dose should therefore either show a response which is directly a measure of the average dose absorption in a layer of skin at a depth between 5 and 10 mg cm^{-2} or it should be able to give information about type and energy of radiation enough for converting dosemeter response to true skin dose. For penetrating radiation, e.g. gamma photons, the routine monitoring systems practiced today generally measure skin dose with satisfactory precision, e.g. to within ± 20% for energies ranging from 10 keV to 2 MeV. For more shallowly penetrating radiation, e.g. β-particles, which have ranges that are less than or comparative to the thickness of the dosemeters mostly applied for practical routine dosimetry considerable underestimation of skin dose may occur. Thin dosemeters with high efficiency for beta dosimetry are available (2), however practical difficulties in handling very thin dosemeters combined with a low sensitivity and therefore a high minimum detectable dose make these unattractive for routine dosimetry purposes. New developments in TLD techniques in recent years (3,4,5) have given practical dosemeters with a skin-dose equivalent response to beta exposures over a wide range of energies. These have so far mainly found application for extremity dosimetry.

The present work has studied possible improvements of existing TLD routine monitoring systems for obtaining a satisfactory skin dose assessment without changing the original capabilities of the system. The following three alternatives were studied:

1. Inclusion of a supplementary skin dosemeter in the TLD-badge.
2. Introduction of a second photomultiplier in the read-out chamber to obtain a simultaneous two-side reading of the TL-dosemeter.
3. Application of a boron diffused thermoluminescent surface layer of LiF TL-dosemeters for skin-dose assessment.

2. INCLUSION OF A SUPPLEMENTARY SKIN DOSEMETER IN THE TLD BADGE

The production of skin-dose equivalent, low-transparent, sintered LiF- and $Li_2B_4O_7$:Mn dosemeters by mixing the material with graphite has been reported earlier (3). Table 1 shows dimensions and gamma-response data measured from a recently produced series of $Li_2B_4O_7$:Mn dosemeter with graphite contents from 0 to 20%.

Table 1. Dimensions and ^{60}Co γ-response data of graphite mixed $Li_2B_4O_7$:Mn dosemeters

Graphite Content %	Dimension		Relative sensitivity to γ-expos.	Instrument background eqv.mR	Instrument + dosemeter background eqv.mR	Threshold detection limit mR
	Diameter mm	Thickness mg.cm^{-2}				
0	4.45	154	100	5.5	4.0	< 1
4	4.46	165	6.1	58	73	< 200
10	4.46	161	3.0	118	146	< 500
20	4.46	157	1.4	230	288	<1000

The threshold detection limit has been calculated according to ISO standard (6). The measured beta response data of various TL-dosemeters shown in fig. 1 clearly illustrate the amount of underestimation of skin dose, which may occur when standard TL-dosemeters are used to measure dose from low-energy beta particles. It can be seen further that the provision of a personnel badge with an additional graphite mixed (about 5% graphite) $Li_2B_4O_7$:Mn dosemeter, capable of estimating any skin dose with a threshold detection limit of 100-700 mrad will introduce essential improvements for assessment of doses to the skin from low-energy beta emitters.

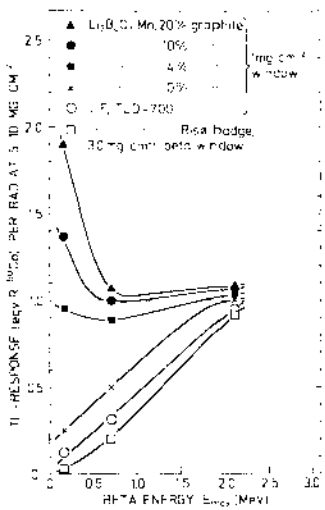

Fig. 1. Beta energy response curves of various TL-dosemeters.

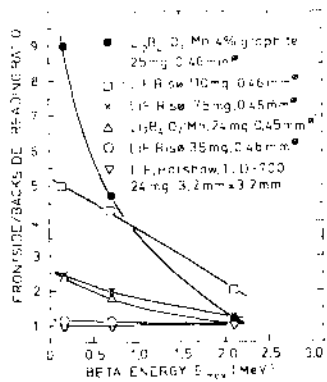

Fig. 2. Frontside/backside reading ratio for various TL-dosemeters as a function of beta energy.

3. TWO SIDE READING OF TL-DOSEMETER

A simultaneous separate reading of the TL-emission from each side of an irradiated dosemeter offers some possibility for registrating the attenuation of the radiation when passing through the

dosemeter mass and thus to get some information about the amount of low-energy radiation contributing to the dose. A two-side reader, based on hot nitrogen gas heating and provided with two photomultipliers looking at each side of the dosemeter during read-out has been described earlier (3). The ratio between front- and backside reading is a complex function of dosemeter thickness and optical transparancy, type and energy composition of radiation and light reflection characteristics of read-out chamber. Fig. 2 shows measured data from dosemeters with varying thickness and optical transparancy. By correlating the data from Fig. 2 with corresponding measured dose/dosemeter response conversion factors one can evaluate expressions which convert the front/backside reading ratios, R(Emax), to dose/dosemeter response conversion factors. An example of this, using data from $Li_2B_4O_7$:Mn, is given in fig. 3, which shows that the expression $R(Emax)^{0.63 \cdot R(Emax)}$ is an extremely good estimate of the corresponding dose/dosemeter response conversion factor. It is obvious that the highest accuracy of dose estimation by this method is obtained when only a single beta emitter is involved. For mixed energies the method underestimates doses from low-energy radiation f.example a skin dose received 90% from ^{147}Pm and 10% from ^{90}Y will be underestimated by 5%.

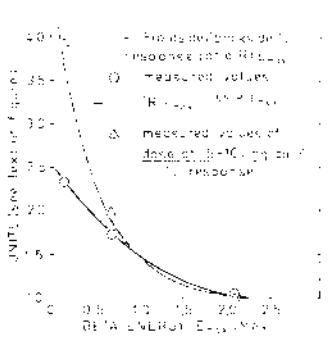

Fig. 3. Relationship between frontside/backside reading ratio R(Emax) and dose/dosemeter response conversion factor for $Li_2B_4O_7$:Mn

4. BORON DIFFUSED THERMOLUMINESCENT SURFACE LAYER OF LiF TL-DOSEMETERS

A new high temperature glow peak produced only in a thin surface layer of LiF TLD-100 chips by diffusion of boron into the LiF material has been applied recently for dosimetry of electrons of energies from 0.1 to 30 keV (7). Experiments with diffusion of boron into sintered LiF dosemeters produced in our laboratory have given diffused surface layers with a new glow peak at 340°C and with a thickness sufficient for application for personnel beta dosimetry. The height of the 340°C surface layer glow peak compared to that of the original 240°C peak representing the total dosemeter thickness is a measure of dose contribution from low-energy radiation. Fig. 4 illustrates the build-up of the new glow peak relative to the original peak 5 and fig. 5 shows the variation of the peak height ratio of the two peaks with beta energy. The threshold detection limit of the 340°C peak for application for skin-dose measurements is approx.

100 mrad.

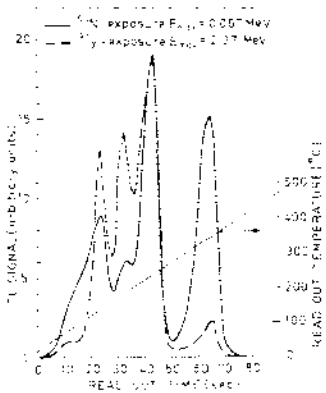

Fig. 4. Glow curves of boron diffused LiF from exposures to a low- and a high energy beta emitter.

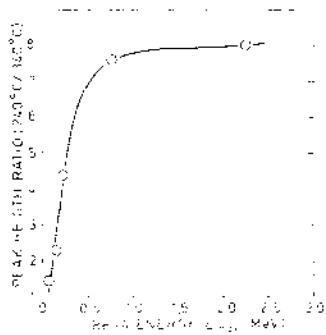

Fig. 5. Ratio of 240°C/340°C peakheigth as a function of beta energy.

5. CONCLUSION

The present work outlines three alternative methods by which substantial improvements of the capabilities of existing routine monitoring systems for skin dose assessment can be obtained. The introduction of a supplementary skin dosemeter may be an attractive method for systems with badges that have a capability for an additional dosemeter already built-in. The two-side reading method has limited possibilities because of reduced accuracy for mixed radiation and technical difficulties in using it for TLD systems with pinnchet heating. The use of a boron diffused LiF layer for skin dose assessment seems to be most attractive method since the only modification needed here is replacement of a dosemeter. However the study of this method is so far only in a preliminary stage and further investigations are needed to prove its practical applicability.

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ELECTRET DOSIMETRY

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Electrets are the electrical equivalent of magnets : they create a permanent electric field in their surrounding. This analogy, however, is formal since the physical processes involved in electrostatics and magnetism differ widely. More specifically, monopolar electrical charges do exist, whereas the magnetic properties of matter arise from dipoles only. The application which is presented in this paper is a direct consequence of the fact that ionizing radiations in a gas produce electrical charges which are acted upon by the permanent electric field of the electret. This concept, termed electret dosimetry, will be exposed after a brief, general introduction to electrets. Practical devices will subsequently be described, and the prospects of this new field will be outlined.

REVIEW OF THE ELECTRET STATE

When a dielectric plate is subjected to an electric field, a macroscopic polarization arises from three main processes :

- dipoles tend to orient themselves parallel to the electric field,
- ions migrate through the material, with a velocity which is directly related to their mobility and to the transport mechanisms involved,
- electric charges are transferred from the electrodes into the dielectric plate. They may be trapped, or they may migrate under the influence of the local electric field.

Once the electret is formed, the polarizations created by these various processes decrease in time with various time constants. In most cases, the polarization due to injected charges is the most permanent one. For this reason, several methods have been developed to inject charges in a controlled fashion into an insulating dielectric. Electrets may, for instance, be manufactured by Corona discharges in the gas surrounding the electret. The electric field accelerates the electrons or ions of the plasma and injects them into the dielectrics. This is a very convenient way to manufacture electrets with very long lifetimes, exceeding tens of years.

The most widely used electret-forming materials are polymer films. They are inexpensive, and they may be manufactured in various lengths, widths and thickness.

BASIC PRINCIPLES OF ELECTRET DOSIMETRY

The basic idea behind the use of electrets in dosimetry is straightforward ; it was proposed as early as 1920[1] : if an electret surrounded by an ionizable medium, such as air, is subjected to ionizing radiations, the charges created in the air will be attrac-

ted by the electret and be injected into it. Therefore, the apparent charge of the electret will decrease with increasing radiation dose. This idea, however, did not lead to any practical device through :

i) a lack a reproducibility of the phenomena : the materials used, mainly organic waxes, and the manufacturing methods, which were essentially empirical, could hardly lead to industrial applications,

ii) the absence of convenient and accurate methods for taking advantage of the charge decay.

The situation of electrets has changed drastically during the last few years, and a very large amount of research has brought the electret to an industrial stage. The Laboratoire d'Electricité Générale de l'E.S.P.C.I. has used its expertise in the field of electrets to design various prototypes of electret dosimeters, some of which are now under development by the Société BEFIC in France.

1) The electret ionization chamber

The basic equations governing the behavior of an electret ionization chamber have been described by various authors[2]. We summarize them briefly, with reference to Figure 1 : an electret of thickness d_2 and permittivity ϵ_2, bearing a charge Q, is separated from two plane parallel electrodes by two air gaps of permittivity ϵ_1, of thickness d_1 and d_3. The total amount of charges created by a dose D in the ionization chamber is known to be : $n = \alpha D(d_1 + d_3)S$ where S is the area of the chamber and α is of the order of 3.5×10^{-4} Cb/m^3. Therefore, neglecting edger effects, the change in the total charge of the electret will be given by : $\Delta Q = \alpha D(d_1 + d_3)S$.

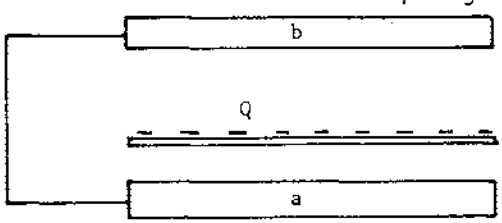

Figure 1

The decrease of the charge of the electret is linear with the dose, which is a very attractive feature. In pratice, several possibilities will be discussed below to take advantage of this charge decrease.

2) Principles of practical electret dosimeters

In order to take advantage of the decrease of the charge of the electrets, mechanical or electrical effects may be used. Mechanical effects have been used to make alarm dosimeters, which have been described in a previous paper[3].

In order to perform a real measurement of the dose, electrical effects are apt to be more accurate and reliable than mechanical ones. Several embodiments for electret dosimeters may be considered. The principle of one of these will be discussed with reference to Figure 2 : the electret lies between two parallel plates. A grounded shutter may be moved to screen electrode b from the electret. When the shutter is away from electrode b, a charge Q_b is present on this

electrode, given by : $Q_b = Qd_1/(d_1 + \varepsilon_1 d_2/\varepsilon_2 + d_3)$.

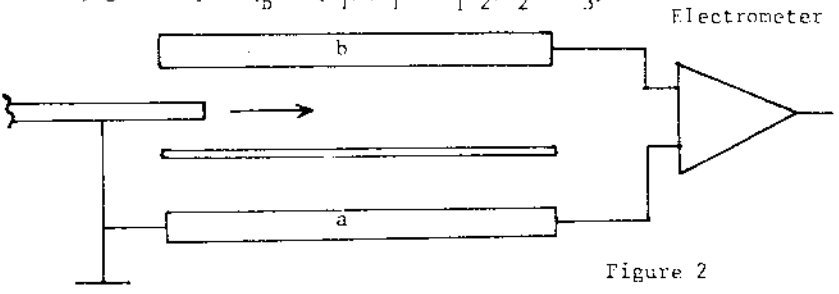

Figure 2

When the shutter is moved to screen electrode b, the charge Q_b drops to zero ; therefore, the value of the charge $- Q_b$, flowing through the electrometer, is displayed. Thus, in such a device, the charge measured when the shutter is moved is proportional to the charge of the electret. Therefore, the variation of the measured charge between two measurements is proportional to the dose, according to the relation : $\Delta Q_b = - \alpha D S d_1 (d_2+d_3)/(d_1+\varepsilon_1 d_2/\varepsilon_2+d_3)$.

Assuming d_2 to be much smaller than d_1 and d_3, the sensitivity of such a dosimeter is given by : $\Delta Q_b/D = \alpha S d_1$ Cb/Rad.

A remarkable feature of such devices is that the sensitivity depends on d_1 only and is independent on d_3. Conversely, the dynamic range D_o, which is the dose necessary to cancel out the charge of the electret, depends on d_3 : $D_o = Q_o/\alpha S(d_1+d_3)$, where Q_o is the initial charge of the electret. As a typical examples, an electret dosimeter embodying a 10 cm^2 electret with an initial charge of 5×10^{-9} Cb, with thicknesses d_1 = 0,4 cm, d_3 = 0,6 cm, has a sensitivity of $1,4 \times 10^{-12}$ Cb/mRad and a dynamic range of 1,4 Rad. Considering the fact that a sensitivity of 10^{-12} Cb is easily achieved by modern electrometer amplifiers, an overall sensitivity of 1 mRad can be expected. Smaller dimensions may be used when such a high sensitivity is not required. The linearity is excellent over the whole range. The energy response of such a device is flat down to 80 keV. A twofold increase in sensitivity, occurring at very low energies, can be easily reduced or compensated.

CURRENT DEVELOPMENT AND PROSPECTS OF ELECTRET DOSIMETRY

As can be seen from the above descriptions, electret dosimeters will be very simple and rugged. Two kinds of electret dosimeters may be considered : direct-reading dosimeters and indirect-reading dosimeters.

Direct-reading dosimeters are self-contained dosimeters incorporating the electret in its ionization chamber, the movable shutter and all the electronic components necessary to perform the measurement of the charge and display the dose. The use of large scale interation circuitry allows to pack all these functions in a small space. Moreover, CMOS circuitry and liquid crystal displays permit a very low power consumption. A direct-reading dosimeter for use in nuclear plants has been developed : apart from the above features, additional circuitry has been included in order to allow a non-contacting transmission of data to computer terminals at the entrances or exits of

critical zones. Such a dosimeter incorporating all of these functions weighs about 300 g and has external dimensions of : 14 x 7 x 2,5 cm.

Indirect-reading dosimeters are dosimeters which do not display the dose directly : an external dose reader must be used in order to get the necessary information. Such electret dosimeters are considerably simpler than the former since the dosimeter does not incorporate any electronic components ; they are extremely light weight and compact. If the required sensitivity is, for instance, 100 mRad, the overall dimensions can be 1 cm x 1 cm x 0,1 cm.

Both these types of electret dosimeters can be made with widely varying sensitivities, dynamic ranges, and functions. Since the principle and basic equations are extremely simple, the geometric and electrical parameters may be easily varied to meet the specific needs of the potential users.

CONCLUSION

The present paper has outlined the fundamental concepts and applications of electret dosimetry. The main advantages of electret dosimeters on most existing radiation monitoring devices are the following :

- the electret behaves as an electrically readable film : the dose can be read electrically without destroying the information carried by the film itself,
- the dose measurement does not depend on the reliability of the electronic components present in the dosimeter : should any failure occur, the information is still present on the electret and may be read by any other electret dosimeter,
- the principle of electret dosimetry is extremely simple and allows it to be adapted to all kinds of needs,
- the linearity is excellent.

Therefore, at this stage of development, electret dosimetry seems an extremely promising new method in radiation protection.

This work has been supported in part by a grant from the Fondation Jean Langlois.

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DEVELOPMENT OF NEUTRON DOSIMETERS FOR FAST AND EPITHERMAL NEUTRONS

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In recent years great effort has been devoted to the development of personnel neutron dosimeters. Attempts to replace the NTA film, which has several shortcomings, by polycarbonate foils have been successful (1-3) in the high energy region above 1 MeV. The polycarbonate foil dosimeters are based on the elastic interaction of the incident neutrons with the carbon and oxygen nuclei of the polycarbonate molecule. The damage sites are revealed by the electrochemical etching (ECE) technique (4,5). Albedo dosimeters composed of ^{6}LiF and ^{7}LiF thermoluminescent chips, are used in several laboratories (6) for the epithermal region, above the Cd cutoff. These dosimeters are strongly energy dependent and sensitive to γ fields. Quite recently a new type of plastic, CR-39, has been developed (7,8) and has been found to be useful in the detection of low energy neutrons down to 200 keV. This sensitivity is based on damage sites produced in the plastic by recoil protons. It creates new possibilities for the production of neutron dosimeters which are very sensitive in the energy range of 200 keV to 14 MeV. However, the energy dose dependence of these dosimeters must be investigated carefully since it might decrease dramatically at lower energies.

DESCRIPTION OF DOSIMETER

The study presented here is an attempt to obtain , in one dosimeter, sensitivity to the energy region from 1 eV up to 14 MeV. The dosimeter is composed of three separate parts as shown in Fig. 1. The first two parts are a ^{10}B miniature spectrometer for the energy region 1 eV to 30 keV and a ^{10}B albedo detector for determining the dose in the energy range 30 keV to 1 MeV; both are in contact with a polycarbonate foil (or CR-39 plastic). The third part is a plastic detector, which is either a bare polycarbonate foil or CR-39 material covered with a thin polyethylene proton radiator for the high energy region 1-14 MeV.

The ^{10}B miniature spectrometer

The ^{10}B spectrometer (9) consists of three ^{10}B layers of the following thicknesses: 60, 360 and 1,500 mg/cm^2. The layers are 6 mm diam cylinders, prepared by pressing ^{10}B powder (enriched to 92%). The thinnest layer is attached to a 1 mm thick Al disk in order to obtain rigidity. The three layers are mounted on the

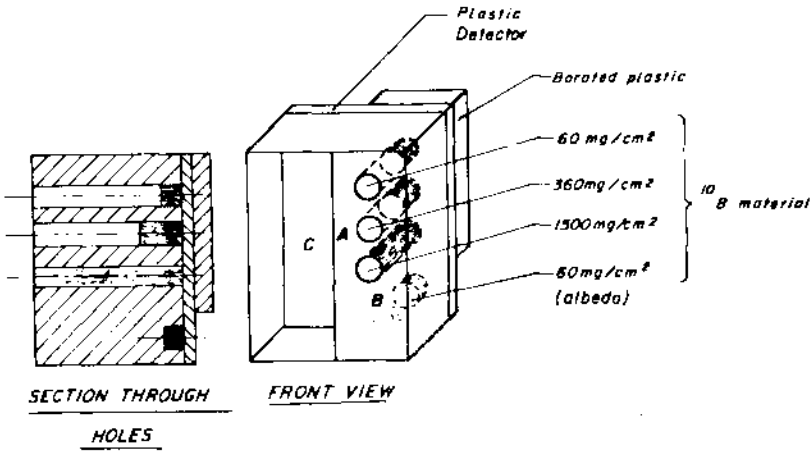

Fig. 1. The combined dosimeter for the 1 keV to 14 MeV energy region. A - ^{10}B spectrometer; B - ^{10}B albedo detector; C - bare plastic detector, made of either polycarbonate foil or CR-39 material with a polyethylene radiator.

plastic detector. The whole assembly is covered on all sides except the front by Cd and borated plastic.

All three layers are thicker than the maximum range of the emitted α (or ^{7}Li) particles. Therefore, the number of particles reaching the plastic detector depends on the number of neutrons which are not absorbed and reach the narrow region (~1 mg/cm^2) in contact with the detector and on the $^{10}B(n,\alpha)^{7}Li$ reaction and energy loss in this narrow region. The dependence of the number of α particles in each ^{10}B layer on energy was described by us in a previous publication (9). At low energies α particles emerge mainly from the thinnest layer, but at high energies they emerge equally well from the three layers. It was found that, up to 30 keV, there exists a linear combination D of the number of α particles emerging from each layer, which is proportional to the dose and almost independent of energy. D is given by:

$$D = K\ (N_1 + 4N_2 + 36N_3) \qquad (1)$$

where N_1, N_2 and N_3 are the number of α particles from the respective layers.

The following assumptions were made in calculating the number of α particles and the coefficients of the above linear combination:

1) The neutrons are incident normal to the surface of the layers.
2) The neutrons are not moderated in the ^{10}B layers.
3) All α particles emerging from the ^{10}B layers which have energies greater than 0.2 MeV produce damage sites in the plastic detector.

4) The $^{10}B(n,\alpha)^{7}Li$ cross section is isotopic in the center-of-mass system.

The albedo ^{10}B detector

It is suggested that, in the energy range between 30 keV and 1 MeV, the ^{10}B spectrometer described above should be used in conjunction with an albedo type dosimeter consisting of a 60 mg/cm^2 ^{10}B layer in contact with the plastic detector and unshielded toward the body. The number of α particles, N_4, emerging from this ^{10}B layer due to the interaction of thermal neutrons backscattered from the body, was calculated using the results of Alsmiller and Barish (10). N_4 is greater than N_i (i=1- 3) in the energy range mentioned above and the ratio $R = N_4 / \sum_{i=1}^{3} N_i$ can be used to determine the effective energy between 30 keV and 1 MeV. The dose can then be determined.

EXPERIMENTAL RESULTS

^{10}B spectrometer

Calibration experiments were performed on a ^{10}B spectrometer in which the ^{10}B layers were mounted on cellulose nitrate of the LR-115 type. The spectrometer was calibrated with 24 keV monoenergetic neutrons from a ^{124}Sb-Be source, polyenergetic neutrons from a ^{252}Cf source, and ^{252}Cf neutrons moderated through 5 and 10 cm thick cylinders of H_2O and D_2O. The efficiencies of the ^{10}B layers are given in Table 1.

TABLE 1. Efficiencies of the ^{10}B spectrometer for different neutron spectra and the ratio of response of the thinnest to the thickest ^{10}B layers. N_i is given in (α/cm^2)/100mRem

Source/Moderator	N_1	N_2	N_3	N_1/N_3
^{124}Sb-Be	4020	3560	2370	1.7
^{124}Sb-Be/5 cm H_2O	7500	4380	2080	3.6
^{252}Cf	58	58	58	1
^{252}Cf/5 cm H_2O	310	270	170	1.8
^{252}Cf/10 cm H_2O	830	500	190	4.4
^{252}Cf/10 cm D_2O	530	420	240	2.2
^{252}Cf/5 cm H_2O + 10 cm D_2O	910	480	260	3.5

The proportional factor K given in Eq. 1 was extracted using the results of the ^{124}Sb-Be irradiation and found to be equal to $\sim$0.1 mRem/(α/mm^2). The ratio of the number of α particles emerging

from the thinnest layer to those emerging from the thickest layer changes with the energy of the incident neutrons. The largest ratio was obtained using moderators composed of 5 cm H_2O combined with 10 cm D_2O or 10 cm H_2O. Good agreement was obtained between the experimental results and our theoretical calculations for the ^{124}Sb-Be source. Two dimensional transport calculations are now in progress for the other types of irradiations listed in Table 1.

Bare polycarbonate

The polycarbonate material, 370μm thick, was calibrated using monoenergetic neutrons of $^{7}Li(p,n)$, D(D,n) and T(D,n) reactions and polyenergetic neutrons of ^{252}Cf and ^{239}Pu-Be sources. The efficiencies for ^{252}Cf neutrons and for neutrons above 3 MeV were found to be (32±4) and (60±6) pits/(4 cm^2 - 100 mRem), respectively. The average background is (6±2) pits/4 cm^2.

DISCUSSION

The combination of the ^{10}B spectrometer, the albedo ^{10}B layer and the polycarbonate (or CR-39) foil for fast neutrons may make it possible to determine the dose in the entire energy range between 1 eV and 14 MeV. The plastic foils will be electrochemically etched to reveal the damage sites. The response of the ^{10}B albedo detector and that of CR-39 to different α-particle energies are now being studied.

ACKNOWLEDGEMENT

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ARRANGEMENT OF A TLD SYSTEM TO MEASURE THE DOSE TO PATIENTS UNDERGOING IRRADIATION

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SUMMARY

A dosimetric system has been developed to measure the doses to patients undergoing irradiation for medical purposes. This system is based on $Li_2B_4O_7$:Mn, TLD-800, in ribbon form, only recently manufactured by Harshaw Chem. Co.. The main advantages of TLD-800 are its equivalence tissue, its single glow-curve, its very easy annealing procedure. The problems investigated are: annealing procedure, linearity, energy response and fading characteristics.

SAMPLES AND READER-SYSTEM

The samples of $Li_2B_4O_7$:Mn studied are ribbons, measuring 3.2x3.2x0.9 mm^3, and a density of 2.4 gm/cc. The effective Z of borate, for photoelectric absorption, is 7.4 . After exposure, the samples were readout in a Harshaw thermoluminescent reader-system, model 2000 B+C, with a constant flux of nitrogen (about 5 l/min).

ANNEALING

The first problem investigated was the annealing procedure to be used on $Li_2B_4O_7$:Mn ribbon, in order to eliminate the effects of previous exposure.

The TLDs, six chips, were annealed for 10 minutes at various fixed temperatures between 200°C and 300°C, with 25°C steeps. After cooled to room temperature in air, three chips were irradiated at 20 rads and readout. The other three chips were readout for background check. The results are shown in fig.1. Each point of the graph is based upon nine measurements (3 readouts on each of 3 TLDs). It can be seen that the best reproducibility (lowest S.D.) is achived at 275°C, whereas the absolute value of TL output is maximum at 275°C. The whole experiment was repeated at higher dose levels, with practically the same results. Previously irradiated chips were annealed at 275°C for times varying between 10 min and 30 min, with 10 min steps. The results are shown in fig.2. The lowest S.D. is at 10 min. The optimum annealing to produce high sensitivity TLD-800 is 10 min at 275°C.

DOSE RESPONSE

Fig.3 shows the shape of the glow-curve at various dose levels. Fig.4 shows the response of borate, peak area, exposed to ^{60}Co. The linearity index, starting around 10 R, is about 0.85 (1).

ENERGY RESPONSE

The energy dependence of the response of TLD-800 is shown in fig.5. The energy response is lightly different from J.J.Thompson et al. data (2), which used borate in powder form.

FADING

No evidence of thermal fading was found during a short time of storage (10 days). Experiments are being run to test the fading effect over longer times.

CONCLUSION

The range of sensitivity of lithium borate is such that its most use is in clinical-therapy dosimetry rather

than in health physics. The results of this preliminary investigation have demonstrated that manganese-activated lithium borate ribbon is satisfactory for measuring high doses in some clinical applications like in therapeutic linear accelerator.

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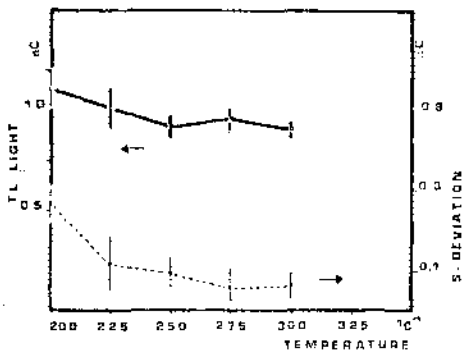

FIG. 1. ANNEALING: DETERMINATION OF OPTIMUM TEMPERATURE AT 10 MINUTES

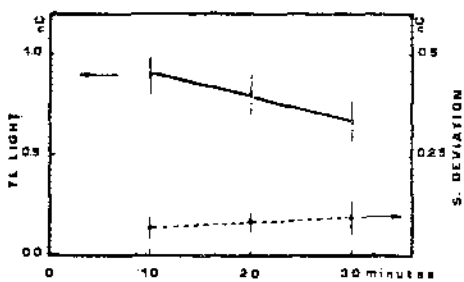

FIG. 2. ANNEALING: DETERMINATION OF OPTIMUM DURATION AT 275 °C

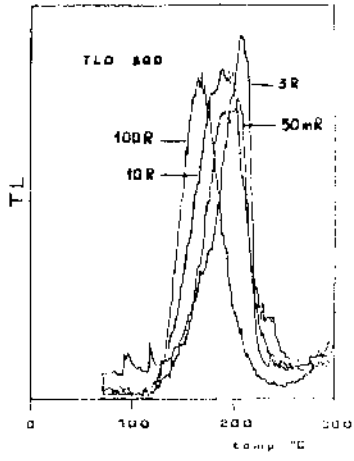

FIG 3. TYPICAL GLOW CURVES AT DIFFERENT DOSE LEVELS

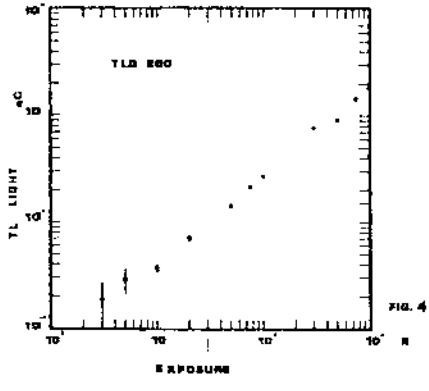

FIG. 4

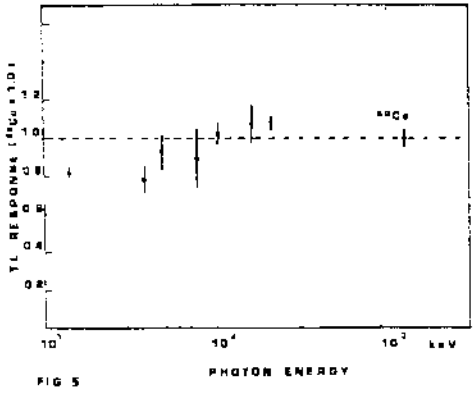

FIG 5

POLYURETHANE AS A BASE FOR A FAMILY OF TISSUE EQUIVALENT MATERIALS

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INTRODUCTION

Two very important parameters must be considered when selecting and using tissue equivalent materials. First of all, the tissue being simulated must be specified. The human body contains a range of tissues and tissue compositions from the low density volume of the lung with a specific gravity of about 0.3 to solid bone with densities more than six times greater. When the tissue is not specified, reference is usually being made to lean muscle which has been specified in a formal way by the ICRP (1). The second parameter that must be carefully specified is the type of radiation being used and the interaction parameter in question. Material that is "tissue equivalent" for the linear transmission of x-rays may be a poor simulant for neutron dosimetry applications. In the first case, the density, effective atomic number, and photon attenuation coefficients are important parameters, while for neutron interactions, the elemental composition, with particular attention to hydrogen and nitrogen may be important. In the case of high energy neutrons, the relative carbon and oxygen compositions of the simulating material become significant.

It is also important to remember that human tissue is not necessarily uniform or homogenous. Layers or striations of fat through muscle, for example, lend a great deal of variability from one tissue volume to the next. In addition, differences between people make precise definition of tissue composition nearly impossible.

Over the years, a number of formulations have been used to simulate the tissue of most general interest - lean muscle. These include, but are not limited to: Temex, a depolymerized rubber (2); Mix-D, a paraffin polyethylene based material (3); rando muscle equivalent material, an epoxy based simulant and Shonka A-150 muscle equivalent plastic (4) formulated for dosimetry applications with an emphasis on neutron interactions. Most of these formulations have been selected to simulate only one tissue. However, White has adopted the use of epoxy formulations as a basis for accurate simulation of a wide variety of human tissues including lung (5).

In the last few years we have been involved in fabrication of a realistic torso phantom for calibration of counters used to measure internal depositions of transuranic isotopes. Since these isotopes are primarily measured via low energy L X-rays, the important characteristic we needed to simulate was low energy photon ($\geq$13 keV) attentuation. In addition, it was important to simulate not only muscle, but adipose, cartilage, bone, and lung to adequately represent the torso morphology adequately. For various reasons, including the need for ruggedness and flexibility in the finished torso and organs, none of the available simulations were fully satisfactory for our purpose.

POLYURETHANE

Our search for a plastic material that could be used as a basis for simulating the wide variety of tissues needed for our application, led us to the selection of polyurethane. Polyurethane has a density in the range of 1.03 to 1.08g cm^{-3}, is flexible, easily molded in a variety of shapes. The material we use has a composition of 9.4% hydrogen, 64.2% carbon, 3.1% nitrogen and 23.3% oxygen, however a variety of other suitable commercial polyurethanes are available. The casting procedures used for polyurethane make it relatively easy to add a wide variety of materials that can be used to adjust the density and effective atomic number to simulate the tissues in question. Finally, special polyurethane formulations are available that can be used to make foamed shapes with densities in the range of 0.2 to 0.3 g . cm^{-3} - suitable for mock lungs.

Like epoxy, polyurethane is fabricated from two basic components. Our procedure involves including additives such as calcium carbonate, urea, polyethylene, and acrylic microballoons into one component of the polyurethane before casting. In low concentrations (a few per cent) of these materials, only vigorous mixing is needed to achieve a homogenous casting. In high concentrations, such as that needed for rib bone or a general purpose muscle simulation, we find that availability of a roll or ball mill, similar to those used for making paint are important to achieve the necessary homogeneity. Once the proper additives have been mixed in component A of the polyurethane, the formulation is completed by combining with component B and a trace quantity of catalyst used to help the polyurethane set properly. This mixture should then be evacuated in a large container to eliminate trapped air bubbles following the mixing process. After evacuation, the mixture is poured into the molds that have been prepared for the desired shapes, and the tissue equivalent casting is left to cure and harden for a period of about 24 hours.

One requirement of the phantom project was that we be able to incorporate transuranic isotopes uniformly through the lungs and other organs to simulate internal deposition for calibration of whole body counters. We can do this easily with polyurethane by dissolving the radionuclide in nitrate form, together with a small amount of lanthanum nitrate carrier, in acetone. This solution mixes well with the polyurethane before casting and results in highly uniform organ labelling.

SPECIFIC FORMULATIONS

During the last three years, we have developed polyurethane based formulations that are primarily designed to represent the low energy photon transmission properties of muscle, adipose, cartilage, bone and lung. The data and criteria of Newton and White (6) have provided guidance in developing a formulation of many of our materials. Our original muscle equivalent formulation was actually chosen to simulate the transmission of 17.2 kV x-rays from Pu through water rather than muscle. The differences between tissue and muscle are not great for our purpose, and water is a readily available, precise standard for attenuation. It is this formulation (LLLM1), that has been used in the phantoms that we have fabricated so far.

Recently, we have used criterion of Newton and White to refine formulations of some of the more important tissues for future applications. In particular, we have extended our formulation inventory to include a more accurate simulant of the ICRP muscle, with closer attention to reproducing the elemental concentrations of hydrogen and nitrogen that are important for neutron dosimetry applications (Table 1). It is possible using this formulation to cast a solid sphere of muscle equivalent material that would be useful as a physical representation of the ICRP-30 cm spherical phantom for calibration purposes.

Table 1. Polyurethane Tissue Equivalent Formulations

LLLM1 95.7% Polyurethane, 4.3% $CaCO_3$
LLLM2 66.4% Polyurethane, 25.0% polyethylene, 5.6% $CaCO_3$, 3.0% urea
LLLA1 99.4% Polyurethane, 0.6% acrylic micro-balloons
LLLR1 67.0% Polyurethane, 33.0% $CaCO_3$
LLLL1 96.2% Polyurethane, 3.8% $CaCO_3$

Table 2 shows a comparison of tissues and simulants for dosimetry applications. The electron density - No, and the average atomic number - $\bar{Z}$ have been calculated from the formulas of Spiers (7).

$$No = 6.023.10^{23} \sum \frac{Pi\ Zi}{Ai} \quad \text{and} \quad \bar{Z} = \left[\frac{6.023.10^{23}}{No} \sum \frac{Pi\ Zi^{3.94}}{Ai} \right]^{\frac{1}{2.94}}$$

Where Pi is the weight fraction of the ith element
Zi is the atomic number of the ith element
and Ai is the atomic weight of the ith element.

Table 2. Calculated Radiation Interaction Related Properties of Tissues and Simulants

Material	Density $g \cdot cm^{-3}$	$\mu\tau$ at 17.2 keV cm^{-1}	10^{23} No $\cdot gm^{-1}$	$\bar{Z}$	Neutron Kerma[a] (Ref 8) 10^{-10} rad $\cdot cm^2$ thermal	1.05 MeV
Muscle(1)	1.04	0.872	3.31	7.46	0.279	24.8
Temex(2)	1.01	0.677	3.27	7.07	0.00898	22.5
Mix-D(3)	0.97	0.726	3.40	6.95	0.00585	30.6
Rando Muscle	1.00	0.730	3.27	7.39	0.336	21.2
Shonka(8)	1.12	0.756	3.30	6.86	0.279	23.6
MS 20(5)	1.00	0.861	3.24	7.47	0.144	19.5
LLLM1	1.08	0.815	3.27	7.14	0.182	20.7
LLLM2	1.05	0.862	3.31	7.25	0.278	23.7
Adipose(1)	0.92	0.410	3.38	6.23	0.0687	27.9
AP6(5)	0.92	0.405	3.25	6.30	0.190	20.2
LLLA1	0.93	0.401	3.29	6.26	0.253	22.5
Rib Bone(6)	1.27	3.85	3.21	10.77	0.296	17.7
RB2(6)	1.26	3.88	3.20	10.77	0.242	15.8
LLLR1	1.33	3.96	3.20	10.67	0.169	15.9
Lung(1)	0.26	0.218	3.31	7.46	0.279	24.8
LN1(6)	0.26	0.222	DATA	NOT	AVAILABLE	
LLLL1	0.28	0.193	3.27	6.97	0.287	21.4

Newton and White (6) point out that the linear photo-electric attenuation coefficient - $\mu\tau$ - for low energy photons is a particularly important parameter in simulating tissues for x-ray applications. Therefore, we have included calculated values of $\mu\tau$ for the 17.2 keV uranium L X-ray from the decay of plutonium.

Neutron dosimetry applications and specification of the ICRP-30 cm phantom have generated a specific interest on our part in simulating the tissue compositions used for that application. In this case, not only photon attenuation but elemental composition is of importance. Table 2 also shows a comparison of the ICRP reference muscle with tissue simulants. It is important to note, that, although the A-150 plastic is not necessarily an adequate tissue simulation for low energy photon applications, it is still probably the best material available for use in ion chambers or other devices requiring a conducting material.

SUMMARY

We have used polyurethane as a base material for a wide variety of tissue simulating applications. The technique in fabrication is similar to that of epoxy, however, the end products are generally more flexible for use in applications where flexibility is valuable. The material can be fabricated with relatively small laboratory equipment. The use of polyurethane provides the dosimetrist with the capability of making specific, accurate, on-the-spot tissue equivalent formulations to meet situations which require immediate calibration and response.

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DEVELOPMENT OF A PERSONNEL NEUTRON DOSIMETER/SPECTROMETER

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Introduction

Of the single component personnel neutron dosimeters used today, only ^{237}Np fission track detectors have an energy response that adequately matches the ICRP dose equivalent conversion curve over the range of energies encountered in personnel monitoring [1,2]. Albedo detectors are severely over-sensitive to low energy neutrons, while the threshold detectors -- NTA film, ^{232}Th track etch foils, electrochemically etched polycarbonate and CR-39 -- are insensitive to the low energies that comprise a significant fraction of the neutrons present in heavily shielded fission facilities, power reactors, etc. Moreover none of the personnel dosimeters, including neptunium, can provide spectral information.

Two component dosimeters have been used to improve dose equivalent response and provide some spectral information[3,4]. These feature different types of detectors that have complimentary energy response characteristics (albedo + a threshold detector). At the Lawrence Livermore Laboratory, we have a wide range of neutron spectra -- highly moderated reactor neutrons, spontaneous fission neutrons with little or no moderation, alpha-neutron sources, 14 MeV neutrons and even neutrons from a 100 MeV electron linear accelerator. Under these conditions (not unusual at research laboratories), even two component systems have important limitations.

Dosimeter/Spectrometer

These considerations have led us to development of a four component dosimeter/spectrometer-(DOSPEC) - an albedo detector[5] that contains samples of polycarbonate[6], carbonate (CR-39)[7], and cellulose nitrate (Kodak Pathe LR115)[8] track etch material. The polycarbonate samples are etched electrochemically, CR-39 is pre-etched in KOH before electrochemical etching, and the LR-115 is etched conventionally in a hot caustic bath. We count all of the track etch samples optically using a microscope or a microfiche reader. The electrochemical etching enlarges tracks in the polycarbonate and CR-39 samples so that they are very visible under low magnification. A red dye has been added to the thin cellulose nitrate layer of the LR-115. Tracks which penetrate the film are visible as pinholes, particularly when illuminated with contrasting blue or green light. The $^{6,7}LiF$ crystals used for albedo dosimetry are read using conventional TLD techniques.

The components of the DOSPEC were selected because they have significant differences in neutron energy response characteristics (Figure 1). The albedo detector is sensitive to the intermediate range below the 100 keV threshold of the CR-39. CR-39 registers recoil protons, thereby providing needed sensitivity in the range from 100 keV to the energy threshold of polycarbonate -- 1.5 MeV. Polycarbonate responds by detecting the heavier recoils -- carbon, nitrogen and oxygen in the plastic itself. LR-115 also registers heavy recoils, but the particle range required to make a penetration

in the 12 μm sensitive layer results in a different energy response. We have arbitrarily chosen to break the energy range above 1.5 MeV into two bands - 1.5 to 6.0 MeV and 6.0 to 18 MeV.

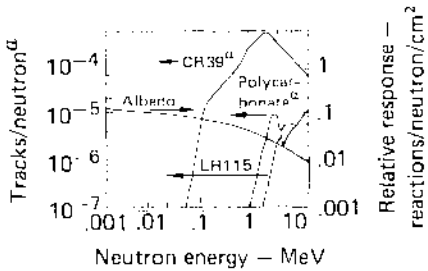

Figure 1. Relative responses of DOSPEC components - CR-39, polycarbonate, Kodak LR115 track etch detectors and TLD albedo. (a. Sensitivity of CR-39 and polycarbonate is the sum of tracks on both sides of the sample.)

DOSPEC Evaluation

The use of DOSPEC requires that we first assign a response for each component detector to each energy bin. We have calculated the spectrum weighted or effective responses for the four components using the responses shown in Figure 1, and a number of neutron spectra from a computer library of measurements, calculations and published data. The means of these values are presented in Table 1. This table represents a matrix that can be used as input for one of the 4 x 4 matrix operation programs now available on programmable hand calculators.

Table 1. Mean Spectrum Weighted Responses of DOSPEC Components

	Energy Bands			
	Thermal to 0.1 MeV	0.1 to 1.5 MeV	1.5 to 6.0 MeV	6.0 to 17.9 MeV
Dose Equivalent Conversion - 10^{-9} rem/n · cm^{-2}	1.6	23.	40.	41.
Relative albedo Response[a] - 10^{-8}	1.07	0.48	0.24	0.13
CR39[b] - 10^{-6} Tracks/Neutron		130.	330.	130.
Polycarbonate[b] - 10^{-6} tracks/neutron			4.6	5.4
LR115 - 10^{-6} tracks/neutron			1.0	4.4

a. equivalent gamma rad/neutron cm^{-2}
b. total on both surfaces.

We have now made measurements with the DOSPEC units by exposing them, on the front of water filled phantoms to previously measured neutron fields in our calibration facility[10] - unmoderated ^{252}Cf, $^{238}PuBe$ and D,T neutrons, as well as ^{252}Cf neutrons moderated by spheres of aluminum, D_2O and H_2O. The DOSPEC measured spectral distributions from these exposures are shown in Table 2, together with reference values from multisphere spectrum measurments[10]. (We used activation foils to determine the 14 MeV spectra from the D,T generator.) The dose equivalent values for DOSPEC were calculated from the spectra in Table 2. The dose equivalent response of an albedo dosimeter (one component of DOSPEC) has been included for comparison.

Table 2. Results of DOSPEC Measurements of Calibration Facility Neutron Fields.

		Neutron Fluence $10^6 \cdot cm^{-2}$				
		$2.5 \cdot 10^{-8}$ to 0.1 MeV	0.1 to 1.5 MeV	1.5 to 6.0 MeV	6.0 to 18 MeV	D.E. Rem.
Bare ^{252}Cf	(a)	9.1	14.2	15.	1.8	1.00
	(b)	3.3	21.0	9.5	1.4	0.92
	(c)	Albedo Component				1.00
20cm Al[d]	(a)	18.	24.	8.7	0.81	1.00
	(b)	12.	25.	6.3	-0.20	0.83
	(c)	Albedo Component				1.61
25cm D_2O[d]	(a)	250.	9.0	9.3	2.0	1.00
	(b)	330.	13.	5.9	2.4	1.15
	(c)	Albedo Component				22.6
25cm H_2O[d]	(a)	51.	12.	14.	2.1	1.00
	(b)	78.	13.	11.	2.3	0.95
	(c)	Albedo Component				5.8
Bare $^{238}PuBe$	(a)	2.2	3.0	17.	4.7	1.00
	(b)	13.	-17.	20.	4.9	0.63
	(c)	Albedo Component				0.70
D-T Neutrons	(a)	0.0	0.0	4.4	14.	0.74
	(b)	6.5	-7.6	4.2	18.	0.73
	(c)	Albedo Component				0.40

a) Measured with multisphere spectrometer
b) Measured with DOSPEC
c) Determined from albedo dosimeter, normalized to bare ^{252}Cf
d) Radius and material of sphere used to moderate ^{252}Cf neutrons.

The data in Table 2 show some differences between the DOSPEC values and the measured spectra (including some negative values). However, the important fact is that DOSPEC does detect prominent spectral features of the neutron field: large, low-energy contributions - Cf in D_2O and H_2O; shifts due to metal scattering - Cf in Al and unmoderated; significant high energy component - PuBe and 14 MeV. The DOSPEC provides important and otherwise unavailable data information for correctly assessing the dose equivalent (D.E.). The value of this information is demonstrated by a large reduction in the D.E. error when compared with the data from the albedo component alone.

It should be understood that, in practice, we do not process every component of every dosimeter issued to our workers. The polycarbonate and LR-115 are less sensitive than the albedo or CR-39 components. If the estimated dose as determined from the automated TLD albedo detector is less than 100 mrem, only the CR-39 will be read and DOSPEC evaluated using only the albedo and CR-39 components. Moreover, many people work in areas where not all four components are required. LR-115 represents the high energy component of the spectrometer, and probably has little value when neutrons above 10 MeV are not a problem. On the other hand, if there is a facility that generates significant levels of neutrons with energies above 20 MeV, the use of the LR-115 could be extended with the high energy radiator techniques adopted at CERN[11].

Summary

We are in the process of adding a few, inexpensive plastic track etch components (CR-39, Polycarbonate, LR-115) to our albedo dosimeters. This addition will significantly extend our personnel neutron measurement capability. The resulting assembly - DOSPEC - provides information necessary to infer a four energy group spectrum. This information should significantly reduce the expected error in personnel dose assessment, and help clarify the ambiguity that results when personnel must work in more than one neutron producing facility during the monitoring period.

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NEUTRON SENSITIVITY OF GEIGER-MÜLLER PHOTON DOSEMETERS FOR NEUTRON ENERGIES BETWEEN 100 keV and 19 MeV

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The two detector method is often applied for dosimetry in mixed neutron-photon radiation fields. For instance rem counters are used to measure the neutron component of the dose equivalent rate, $\dot{H}_n$, and energy compensated Geiger-Müller (GM) counters [1] to measure the photon component, $\dot{H}_g$.

In this paper we investigate the response of GM counters to neutrons. In a mixed field, the counting rate of an energy compensated GM counter may be written

$$\dot{R} = \dot{R}_n + \dot{R}_g + \sum_i \dot{R}_i = \overline{a}_H \cdot \dot{H}_n + \overline{b}_H \cdot \dot{H}_g + \sum_i \dot{R}_i \qquad (1)$$

Here, $\dot{R}_n$, $\dot{R}_g$ and $\sum\dot{R}_i$ are the counting rates due to prompt neutrons, to photons and to neutron activation of the GM counter, respectively. The quantity $a_H(E)$ is defined as the dose equivalent sensitivity to monoenergetic neutrons of the energy E, and $\overline{a}_H$ as its average value in the neutron spectrum. It is assumed in the following that, in addition to $\dot{R}$, the quantities $\dot{H}_n$ and $\overline{b}_H$ are measured or known from other experiments. In order to evaluate the desired quantity $\dot{H}_g$ from eq. (1), $\overline{a}_H$ and $\sum\dot{R}_i$ must be known. The following can be written

$$\overline{a}_H = \left[\int_0^\infty a_\varphi(E) \cdot \frac{\partial \varphi}{\partial E} \cdot dE\right] \cdot \left[\int_0^\infty d_H(E) \cdot \frac{\partial \varphi}{\partial E} \cdot dE\right]^{-1} \qquad (2)$$

where φ is the flux density of neutrons, $d_H(E)$ the fluence to dose equivalent conversion factor [2], and $a_\varphi(E)$ the neutron flux sensitivity which can be expressed by

$$a_\varphi(E) = b \cdot d_K(E) \cdot k_\mu(E) \qquad (3)$$

Here, b is the absorbed dose sensitivity of the GM counter for ^{60}Co gamma radiation, $d_K(E)$ is the neutron fluence to absorbed dose conversion factor, and $k_\mu(E)$ the ratio of the neutron absorbed dose sensitivity to the photon absorbed dose sensitivity [3]. The values of $d_K(E)$ are practically identical with the kerma factors for ICRU muscle tissue tabulated in appendix A of [3].

In order to evaluate $\bar{a}_H$ from eq. (2) and (3), besides the neutron spectrum the k_u value of the GM counter has to be known.

MEASUREMENT OF THE k_u VALUE

The k_u values of two commercially available energy compensated GM counters, ZP 1100 (Valvo, Hamburg) and MX 163/PTFE (Alrad Instr. Egham, Surrey, England) were measured by the time-of-flight (TOF) technique [4]. The MX 163/PTFE was a GM counter with a Teflon (PTFE) sleeve between the energy compensating filter and the counter tube. In general the counter axis was perpendicular to the neutron beam. The measured energy dependence is shown in fig. 1 for both GM counters. Our TOF data for the ZP 1100 including the data from [4] (=PTB-data) are connected by the solid line. The upper value at E = 2.5 MeV was measured by a different technique [5,6] and shows satisfactory agreement with the TOF value. There is also good agreement between our ZP 1100 and our MX 163/PTFE data below 5 MeV. At 15.5 MeV the MX 163 value is considerably lower than the ZP 1100 value. Such an effect is qualitatively to be expected, since in the case of the ZP 1100, recoil protons enter the active counter volume if $E \gtrsim 6.5$ MeV.

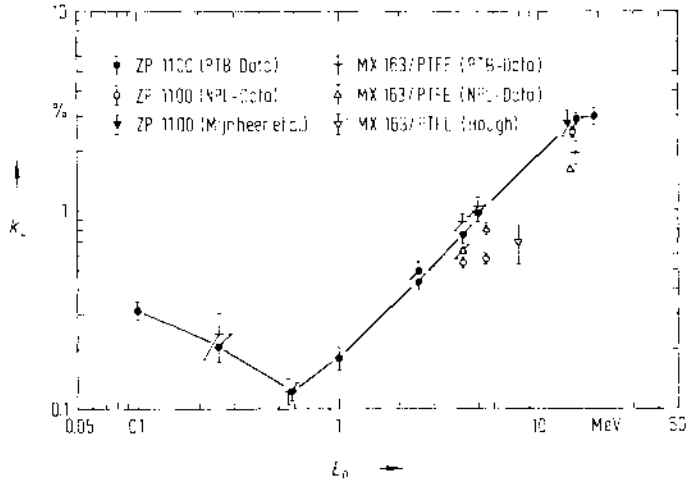

Figure 1. k_u values as a function of the neutron energy for two different GM counters of type ZP 1100 and MX 163/PTFE, as measured by different authors [7, 8, 9]. PTB data refer to this paper and data from [4].

Other authors' data are also included in fig. 1 [7, 8, 9]. In some cases they measured the k_μ value with the counter axis parallel to the neutron beam (e.g. [7] at 4.2 and 5.5 MeV). The TOF technique was only used in our laboratory. In order to compare the different techniques, we measured the data points at 4.2 MeV shown in fig. 1 with the counter axis parallel to the neutron beam.

In the energy range around 15 MeV there is a satisfactory agreement between the different data. In the energy range between 4 and 8 MeV, however, a considerable spread of data is observed. In this energy range, measurements are particularly difficult because in order to produce these neutrons, incident deuteron beams with comparatively high energies are used. They produce a high background of gamma radiation in the targets by nuclear reactions such as (d,p) and (d,n). In the TOF technique which was used in our experiments, the counts from the GM counter due to this gamma background could be clearly separated from the counts due to neutrons [4].

ACTIVATION BY NEUTRONS

In order to measure the activation of the ZP 1100 with mean lives larger than about 1 min, it was irradiated with 15.5 MeV neutrons. After irradiation, the GM counter was put into a room with low radiation background where the time dependence of the counting rate was measured. Activations with mean lives $\tau \approx 200$ s, 640 s and 3.71 h were measured. Comparing the activations in GM counters with and without the energy compensating tin filter, it is concluded that the activation of the tin filter is small compared with the activation in the intrinsic GM counter tube. The $\tau = 3.71$ h activation was assigned to the reaction $^{56}Fe(n,p)^{56}Mn$. For an infinitely long irradiation, the ratio

$$\dot{R}_n^{-1} \cdot \sum_{i=1}^{3} \dot{R}_i = 7.6\ \%$$

was determined for the three activations mentioned above.

The contribution of short-lived activations with 20 ns $\lesssim \tau \lesssim$ 1 min could be estimated from the background in the TOF spectra to be

$$\dot{R}_n^{-1} \cdot \sum_{short} \dot{R}_i \leq 3.4\ \%$$

DISCUSSION

The parameters given above are required for the use of GM counters in mixed neutron-photon fields. In the dosimetry for radiation protection, the short-lived

activations probably do not present a serious problem because they practically add to the prompt neutron response. A problem, however, may result from the long-lived activation with $\tau = 3.71$ h because it may introduce a complicated time dependence of the GM counter response.

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MICRODOSIMETRIC APPROACH FOR LUNG DOSE ASSESSMENTS

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1. INTRODUCTION

The initial phase of every biological radiation induced reaction is characterized by primary physical energy deposition mechanisms at the cellular level. This energy absorbed by radiation sensitive sites within cells causes chemical and biological alterations, resulting finally in macroscopically observable effects.

The basis for the determination of the energy distribution of inhaled nuclides in the lungs is the detailed knowledge of the microdistribution of the deposited radionuclides in relation to the irradiated biological target. In particular in the case of short-range alpha radiation with highly localized energy deposition over several cellular diameters any radiobiological reaction depends on the superposition of this microdistribution with the distribution of the radiation sensitive cells.

2. METHODOLOGY

Conventional lung dosimetry is based primarily on the ICRP compartmental lung model as proposed by the ICRP Task Group on Lung Dynamics (3). According to the theory of compartmental systems, this model neglects the fine structure of nuclide deposition, the specific location of radiation sensitive targets and the non-uniformity of energy deposition along particle tracks through different cells. Therefore, a dosimetric method for inhaled alpha-emitting nuclides was developed, taking into account these above inhomogeneities. These cellular dose calculations are based on randomly selected tissue slices and LUMC - a computer lung model for inhalation of radioactive aerosols in the human respiratory tract (1).

The basic approach of this microdosimetric method is to superimpose alpha particle tracks onto magnified images of selected tissue slices on the monitor of an electronic image analyzer (Quantimet-720). Using adaptive pattern recognition methods the different cells in the lung tissue can be identified and their distribution within the whole organ determined. The probability of track-cell-interactions as well as the energy deposited there allows the calculation of cell-specific doses. Integration over all cellular doses results again in an organ dose, but now taking into consideration the inhomogeneity of both absorbed energy and lung tissue.

This microdosimetric concept is applied to the inhalation of very soluble short-lived radon decay products as well as of highly unsoluble PuO_2 particles. With regard to the nuclide distribution within

the respiratory tract these nuclides represent both extreme cases of more homogeneously located radiation sources as well as very localized "hot spots". The methodology developed is, however, not restricted to lung tissue, but is also applicable to other organs and tissues of the human body for dose calculations in practical health physics.

3. NUCLIDE DISTRIBUTION

Deposition and retention of inhaled particles in the human lung are functions of various superimposed anatomical, physiological and aerosol-specific parameters. For the simulation of their mutual influence on the resulting particle distribution a multiparameter deposition and retention model for inhaled alpha-emitting radionuclides, called LUMO, was developed. The anatomical basis for this computer model is the Weibel model A for regular dichotomy (7). These anatomical data, together with physiological parameters, such as respiratory frequency and tidal volume, show a significant dependence of age, sex and physical activity, influencing particle deposition to a large extent (1, 2). The surface activities of the nuclides in the single generations of the lung model, resulting from activity deposited, radioactive decay, mucus transport and other clearance mechanisms are calculated by solving a linear differential equation system, assuming first order kinetics in the lung model (2).

4. AUTOMATED PATTERN RECOGNITION OF LUNG TISSUE COMPONENTS

This investigation has been performed on 1 µm-thick lung tissue sections, obtained from adult Sprague-Dawly laboratory rats. The goal of the following analysis of the tissue sections was to discriminate between the various tissue components, such as specific lung cells and background. By definition this can be treated as a classical pattern recognition problem, that is to recognize certain pictorial subsets, such as surface epithelial cells, endothelial cells, or septal cells by quantitative image analysis. For example in order to recognize septal cells automatically a set-transformation with osmiophilic cytosomes, characteristic for septal cells, has been used. These cytosomes appear as dense and round bodies 0.2 to 1 µm in diameter on unstained, but OsO_4-incubated, semi-thin Epon sections within a medium light-optical resolution (300 x magnification). The 2-dimensional section shows clusters of 2 to 5 cytosomes for each septal cell (Fig. 1). After the cytosomes are segmented by thresholding a television frame, they are transformed iteratively by adding a "structuring element", e.g. an octagon to each cytosome (6). The algorithm stops when all cytosomes are "closed", giving a particle count of 1 instead of originally 2 or 5. This procedure detects automatically areas with cytosomes and discriminates the cytosomes from background particles (e.g. dust) and other cells.

5. SIMULATION OF ALPHA PARTICLE TRACKS IN TISSUE SECTIONS

For the simulation of α-particle tracks the idea of Boolean schemes, originated by G. Matheron (4), has been used. To simulate random set realizations, such as alpha particle tracks in lung tissue, we start with a "diffuse point process" in E_2 (Euclidian space of 2-

Fig. 1 Lung tissue section with recognized cytosomes

dimensions), e.g. the stationary Poisson point process with intensity θ, where the number of points within a bounded set A, N(A) is Poisson-distributed. The Boolean scheme can now be realized by implanting non-stationary, compact, convex random-sets (primary grains), such as line segments, in each Poisson-point realization within $|A|$ ($|A|$, Lebeque measure of A). For the present work such Boolean realizations have been used to simulate alpha particle tracks in lung tissue. Thereby advantage can be taken of some interesting properties of this scheme:

a) Stability of the model under sections: Alpha particle tracks are naturally occurring in 3-dimensions, but we can only observe their 2-dimensional sections. Fortunately it is possible that a Boolean scheme of 3-dimensions yields 2-dimensional sections, which are again Boolean schemes.

b) Indefinitely indivisible model: An increase in average lung dose always results in an increase of the same family of microdose distributions and does not change the nature of the underlaying process (fundamental Poisson property).

The simulation of highly localized alpha particle tracks, typical for PuO_2 "hot spots" or radon decay product concentrations at bifurcations of airways, may be gained as follows: Starting with a parent Poisson process of intensity θ, we can generate new points using some distributions D and a strongly concentrated frequency function f. This procedure, where the parent Poisson-points are the centers for new satellite points, leads to the center-satellite process of Neyman (5), with highly concentrated and clustered point-distributions. If a needle with a random-variable length from a uniform probability distribution and random orientation over the unit-circle is taken, a segment process is obtained, which is very similar to "hot spot"-caused track distributions (Fig. 2).

6. INTERACTION OF ALPHA PARTICLE TRACKS WITH LUNG TISSUE COMPONENTS

After automatic recognition of the most important tissue compo-

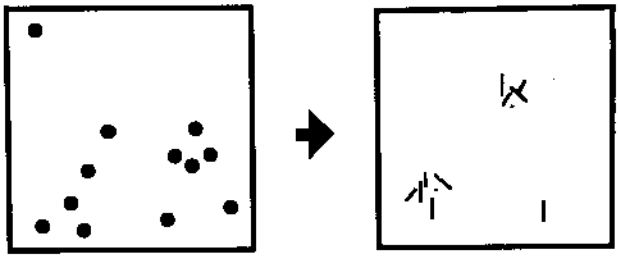

Fig. 2 Parent poisson process $\theta.|A|=2$ and its Boolean center-satellite scheme

nents and generation of alpha particle tracks by use of the Boolean scheme, both realizations can be intersected simultaneously. This leads to the consideration of conditional probabilities and their conditional distribution functions. If the random variable X is defined as the area of intersection between cell type I and alpha particle tracks originating from a Boolean scheme with intensity θ, the conditional distribution function of track intersection length under the condition of a given Boolean realization with $N_B=\theta.|A|$ has to be calculated. Under this condition a microdose for cell type I on its estimated distribution function can be assessed directly, i.e. the percentage of cell type I cells hit by tracks with a certain intensity. According to the stopping power function for alpha particles the intersection area has to be weighted by the energy loss along the track intersection length.

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CALIBRATION AND APPLICATION OF THE MULTISPHERE TECHNIQUE IN NEUTRON SPECTROMETRY AND DOSIMETRY

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During production of radionuclides with the cyclotron of the Eindhoven University of Technology high fluxes of energetic neutrons are generated. For reasons of practical radiation protection we paid attention to neutron detection techniques for dosimetry purposes. Since the biological effect of neutron exposure strongly depends on energy, the detection method must submit information on the neutron energy spectrum.

MULTISPHERE TECHNIQUE

The detection method is known as the multisphere technique (1). A neutron detector is placed in the centre of a sphere of moderating material. In our case the detector is a cylindrical ^{6}LiI(Eu) scintillation crystal (4 mm high, 8 mm diameter) which is optically coupled to a photomultiplier (RCA 6199) by a perspex light pipe (170 mm long, 20 mm diameter). Mainly thermal neutrons are detected in the crystal via the ^{6}Li(n,α)t reaction (Q = 4.8 MeV). The thermal cross section for this reaction is about 1000 barn and decreases strongly with increasing neutron energies ($\sigma \approx 1/\sqrt{E}$). So fast neutrons must be moderated to increase their detection probability.
Measurements were carried out with the bare crystal and with 15 polyethylene (ρ = 940 kg/m^3) spheres with different diameters ranging from 0.05-0.45 m (2-18 inch). Suppression of γ-rays with energies upto 4 MeV is achieved with pulse height discrimination.
The relation between the measured count rate T_i of detector i with detector cross section σ_i and the spectral distribution of the neutron flux density ϕ_E is:

$$T_i = \int_0^\infty \sigma_i(E)\ \phi_E(E)\ dE \qquad i = 1, \ldots\ldots 16 \qquad (1)$$

Relative detector cross sections (also called detector efficiencies) for the different sphere diameters are given by Nachtigall (2). To check these theoretical curves, calibration measurements were carried out for 9 different neutron energies chosen logarithmically equidistant in the energy range from 100 keV to 4 MeV. The mono-energetic neutrons were produced with a Van de Graaff accelerator.
Theoretically the product $T_i r^2$ of the detector count rate T_i and the distance r from the point source equals a constant K_i (inverse square law: $T_i = K_i/r^2$) which is charateristic for a specific source-detector combination. In practice scattered neutrons contribute to the count rate. This contribution varies with distance. The expression for the count rate T_i is:

$$T_i(r) = K_i/r^2 + S_i(r) \qquad i = 1, \ldots\ldots 16 \qquad (2)$$

To simplify the problem we only used measurements for r > 0.5 m. For smaller distances correction factors must be introduced in equation (2). It was experimentally shown that the contribution to the count rate by scattered neutrons by good approximation can be described with $S(r) = a+b/r$.

UNFOLDING PROGRAM

To unfold the neutron energy spectrum from measurements with the 16 detectors, the SAND-II unfolding program (3) was used. This program was developed for measurements with activation detectors. The program applies a non-linear adjustment to the input spectrum at each iteration step. Using the analogy between activity and count rate on one hand and cross section and efficiency on the other the SAND-II program is applicable in our case (5). The set of equations (1) is approximated by

$$T_i = \sum_j \sigma_{ij} \Phi_j \qquad \begin{matrix} i = 1, \ldots 16 \\ j = 1, \ldots 620 \end{matrix} \qquad (3)$$

in which Φ_j is the flux density in $m^{-2}s^{-1}$. The values of σ_{ij} were experimentally determined for the energy range of 100 keV - 4 MeV and were partially derived from the relative values given by Nachtigall for the non-measured energy ranges.

RESULTS

For testing purposes, the multisphere method in combination with the SAND-II unfolding program was applied to known neutron energy spectra. A 11 GBq Am-Be source and a 20 GBq ^{252}Cf source were used. The measurements took place in a concrete hall (dimensions 14 × 18 × 10 m). The detector was placed in the centre of that hall and the source could be moved over a horizontal rail at the same height. The count rates were measured at 10 different distances varying from 0.5 m to 3.5 m. For each detector and both sources the scattering parameters a and b as well as the K-value were determined with a least squares method. The resulting K-values were used as input values in the spectrum calculations. The result of the SAND-II unfolding method of the measurements with the bare Am-Be source, this means corrected for the contribution of neutron scattering, is shown in fig. 1 (curve K). A known Am-Be spectrum (5) was used as input spectrum. The same has been done for the bare Cf-source (4). When determining actual neutron spectra inclusive scattering, the measured count rates for the Am-Be source at several distances from the source were used as input values. With respect to the input spectrum a thermal Maxwell energy distribution was assumed; for intermediate energies (0.5 eV - 100 keV) proportionally to $E^{-0.7}$ proved to give the best results. The unfolded actual spectra at 1 m and 3 m distance from the Am-Be source are also shown in fig. 1. It is quite obvious that fast neutrons obey the inverse square law. For energies below 100 keV the occurrence of scattered neutrons is clear. In addition the calculated integral flux density Φ_{tot} and the mean energy are given in the table.

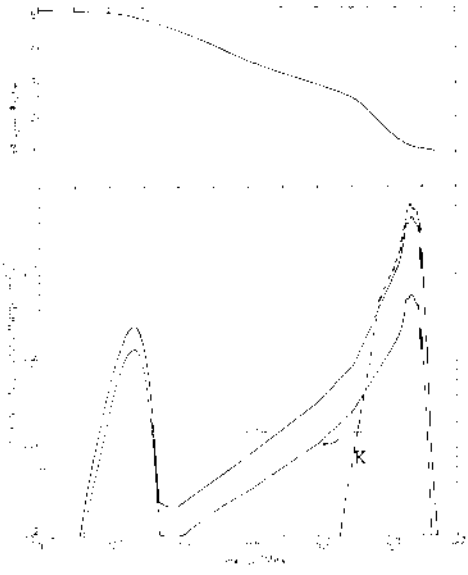

Fig. 1. The flux density per unit lethargy as a function of the energy for the ^{241}Am-Be source. The measurements were performed at 1 m and at 3 m. The spectrum for the bare source is given (curve K) and the uppermost curve gives the ratio of the flux density at 3 m multiplied by a factor 9 to that of 1 m.

FLUENCE TO DOSE EQUIVALENT CONVERSION FACTORS

For dosimetry purposes the effective energy E_{eff} is a more important quantity than the mean energy $\bar{E}$ because the neutron fluence to dose equivalent conversion factor can be strongly energy dependent. The neutron fluence to maximum dose equivalent conversion factors $\hat{h}(S)$ are defined by the ICRP (6). When the neutron energy distribution is known, the values for $\hat{h}(E_{eff})$ can be calculated for that specific source with the relation:

$$\hat{h}(S_{eff}) = [\sum_i \hat{h}_i(E)\ \Phi_i(E)]/\Phi_{tot} \qquad (4)$$

in which $\Phi_i(E)$ is the flux density in the energy range i between the energies E_i and E_{i+1} and Φ_{tot} is the integral flux density. The values of E_{eff}, corresponding with $\hat{h}(E_{eff})$ can be derived from the defined ICRP curve (6). Values for E_{eff} and $\hat{h}(E_{eff})$ were calculated for the bare Am-Be source and the bare Cf source (see Table). Our values for $\hat{h}(E_{eff})$ are in agreement with literature: we found 3.7 10^{-14} Sv.m^2 for Am-Be and 3.0 10^{-14} Sv.m^2 for Cf; respective literature values are 3.62 10^{-14} (7) and 3.39 10^{-14} (8). The quantities E_{eff} and $\hat{h}(E_{eff})$ are also calculated for the actual Am-Be spectra at 1 m and 3 m distance from the source. As is shown in the table the reduction in $\hat{h}(E_{eff})$ when changing from 1 to 3 m distance is more than follows from the decrease in $\bar{E}$. This is due to the fact that the relative contribution of low energetic scattered neutrons, increases with distance.

Table. Some results of the spectrum analysis and some derived dosimetric quantities. K in column II means that K-values were used as starting values for unfolding and T stands for use of count rates. N.B. 1 Sv.m² = 38 10^{11} mrem.h^{-1}.n^{-1}cm^{-2}s^{-1}.

Source	Remarks	ϕ_{tot} [10^6m^{-2}s^{-1}]	$\bar{E}$ [MeV]	$\bar{h}(\bar{E})$ [10^{-16}Sv.m^2]	E_{eff} [MeV]	$\bar{h}(E_{eff})$ [10^{-16}Sv.m^2]
^{241}Am-Be	K	0.642	4.26	4.1	1.86	3.7
^{241}Am-Be	T(1 m)	0.727	3.98	4.1	1.45	3.5
^{241}Am-Be	T(3 m)	0.125	2.90	4.0	0.77	2.7
^{252}Cf	K	2.35	1.97	3.8	0.89	3.0
^{123}I	T(A)	1780	0.56	2.2	0.10	0.56
^{123}I	T(B_1)	28.0	0.69	2.5	0.11	0.62
^{123}I	T(B_2)	4.30	1.21	3.4	0.14	0.76
^{123}I	T(C)	4.31	0.074	0.44	0.022	0.17

The detection system was applied in practical radiation protection measurements during the production of the radionuclide ^{123}I. Results under 4 different conditions are presented in the table. The difference between condition B_1 and B_2 is shielding with a 0.2 m paraffin layer. The detailed description of measurement positions is given elsewhere in this proceedings (9).

FINAL REMARK

In practical circumstances when significant neutron scattering occurs, application of the fluence to dose equivalent conversion factor at mean energy results in a considerable over-estimation of the maximum dose equivalent.

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A MICROCOMPUTER CONTROLLED THERMOLUMINESCENCE DOSIMETRY SYSTEM

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INTRODUCTION

Using a microcomputer, we have developed an automatic thermoluminescence dosimetry system for personal dosimetry and thermoluminescence detector (TLD) research. Process automation, statistical computation and dose calculation are provided by this microcomputer. Recording of measurement data, as well as dose record keeping for radiological workers is carried out with floppy disk. The microcomputer also provides a human/system interface by means of a video display and a printer.
The main features of this dosimetry system are its low costs, high degree of flexibility, high degree of automation and the feasibility for using in routine dosimetry as well as in TLD-research. The system is in use for personal dosimetry, environmental dosimetry and for TL-research work. Because of its modular set-up several components of the system are in use for other applications, too. The system seems suited for medium sized Health Physics groups.

FEATURES

Some of the benefits of a local personal dosimetry system by authorized health physicists are:
- the time interval between exposure and dose determination is short; in case of extreme radiation hazards this may be very important.
- the measurement data can be interpreted very well because of the knowledge about the working conditions and radiation qualities.
- if necessary the composition of the TLD-badge can be modified to fit in with the working conditions, by changing the filtertype and the TL-material.

The dosimetry system described here, is designed to function as a stand alone system. The automated TLD read-out and the complete data processing has been worked out locally. Other main features are:
- automatic dosimeter badge identification.
- reliability in read-out operation with built-in self-checking functions and error messages.
- possibility of simultaneous dose determination in mixed fields.
- the initiation of a number of read-out cycles is easy and personnel not familiar with computers is able to operate the system.
- switching from dosimetry to TLD-research takes little effort.

SYSTEM SET-UP

In 1978 we started developing our dosimetry system. To reduce the time for development, we used commercially available components. The modular set-up of the system is sketched in fig. 1. The thermo-

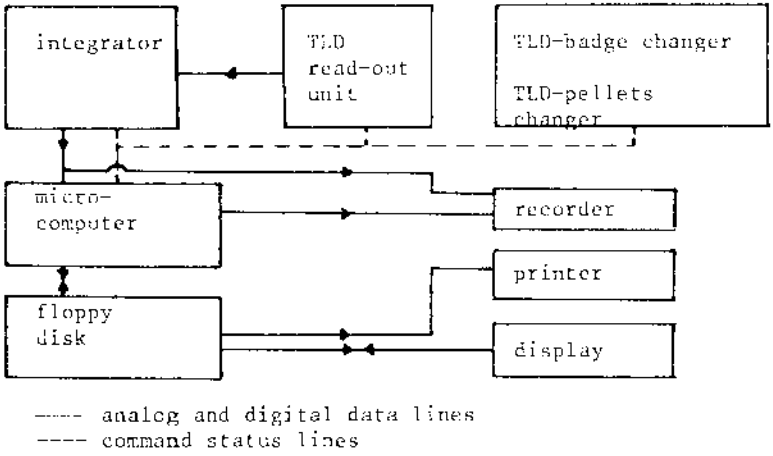

Fig. 1

luminescence analyzer consists of the Harshaw TL-reader (2000-A) and the integrating pico-amperemeter (2000-B). We have chosen a non-automatic analyzer because of the relative low system flexibility of the available semi-automatic systems. The read-out component has been completed with an external TLD sample changer. The TL-signal is detected by a photomultiplier tube and amplified and then integrated. These components are connected with the microcomputer which has full control over all the active and moving system parts. The differential as well as integrated TL-signal are connected with the A/D-converter and a TTY-interface to the computer. The temperature window by which the current integrator is switched, is set manually or by means of two variable DC-levels, available on D/A-convertors in the computer. In all cases of active control a feedback signal is added to provide a so called self-control feature.
Nowadays the complete software is written in BASIC. Although programming in BASIC is not the most advanced way, in view of the large occupied memory space and the slow execution time, we used BASIC instead of an assembly language for the following reason. Learning in BASIC is really easy. Therefore not only program development time can be short, but afterwards non-specialized personnel is able to update parts of the program. As soon as the system fulfils our expectations some parts will be transferred into assembly to speed up the system. During execution the program is stored in RAM-memory and directly accessible to the operator for changing program lines or variables in case of function errors or control problems. The measurement and the calculated data are stored in the computer memory and are also stored immediately on a floppy diskette. If problems occur and the program has to be aborted there will be no loss of information.
For heating of the TLD a so called planchet heater is used. The sample heating can be controlled hardware by the Harshaw 2000-A after manual presetting or software by the microcomputer. Especially for TLD-research purposes this feature saves time, since it is possible to program more than one heating function for groups of TLDs in a batch (100 pellets). When glow curve characteristics are investigated the digitalized TL-

signal is connected to the computer I/O and interpreted directly by software and stored on floppy diskette for further data processing. The microprocessor in the dual drive floppy disk provides its own intelligence and is used for storage and retrieval of software, experimental data as well as results in personal dosimetry. For communications actions a fast printer and a video display (CRT) are connected. The CRT also has direct access to the computer through the floppy disk.
The frame of the TLD sample changer consists of a modified sample changer of Nuclear Enterprises (type PSC 1) and a modified X-Y plotter. The plotter pen has been replaced by a vacuum needle, which can be moved in directions parallel and perpendicular to the X-Y plotter plateau. Putting it upside down, this component provides a transport of the single TLD-pellets. For personal dosimetry we use Studvic TLD-badges, containing four TLD-pellets per badge. The badge has identification numbers in decimal form as well as in punched binary form for automatic registration. The badges are piled up in the sample changer and transported mechanically one by one to the read-out component. In front of the heating planchet the badges are opened automatically. The pellets are transported from badge to planchet and vice versa with the vacuum needle. The moving parts are driven by steppermotors and a synchrone motor. The air pressure in the vacuum needle is controlled by air valves. By means of an air pressure sensitive relay the transport of the TLD-pellet is checked. For easy programming of the transport functions, the TLDs are for research purposes positioned in a batch on equidistant places (10 by 10 matrix).
It may be emphasized that during processing the TLD-pellets are not exposed to light and that the badge is not opened by the operator. At no time the pellets are touched by bare hands or tweezers.

SYSTEM PERFORMANCE AND APPLICATIONS

A main feature of the system set-up is its high degree of flexibility, owing to the modular way of hardware and software constructions. In future it is possible to change parts by more sophisticated components or add other functions. For example the nitrogen flow is at present only controlled, later on N_2-heating can be built in. After changing some parameters in the transport functions different types of research batches may be used. A β-source can be mounted for calibration purposes or measurement of irradiation damages in the TL-material in a continuous way.
As mentioned before, there may be defined a number of heating functions within one research batch read-out. To eliminate the time influence, the TLD-pick-up sequence can be don randomwise, calling a small subroutine.
In the research mode a standard initiation procedure consists of:
- inputting the read-out sequence by entering numbers or ranges of numbers corresponding with the positions on the matrix. TLDs may by read out more than once.
- inputting data to define groups of TLDs on which statistical calculations will be done. Groups may be nested in other groups. The maximal number of groups is 15. The statistical data output per group consists of the mean, the absolute and relative standard deviation, a group number identification and the number of TLDs on which the calculation has been worked out.
- inputting calibration data. The preselected research batces can have

different sensitivities for radiation. Giving some TLD-pellets well-known doses, it is possible to calculate the dose-TL-signal relation of that particular batch. The identification numbers of the TLDs and the calibration doses have to be entered.

After a complete batch read-out the function between the netto TL-signal and the absorbed dose is calculated by fitting a straight line with the method of least squares. The calculated coefficients are used to compute the absorbed doses of the TLDs on that particular batch. These parameters, the correlation coefficient, the standard deviation of the multiplication coefficient and the degrees of freedom are outputted to inform the operator about the data accuracy.
More sophisticated statistics such as variance-analysis methods are restricted by computer memory. However the data are stored on diskette and can be transmitted to other computer systems. The modular hardware construction is important in case of serious system errors. Within 15 minutes the non-automatic read-out component is disconnected from the other parts and can be used manually. Selecting components we took into account the possibility for multifunctional application. This reduced the specific system costs.

OTHER APPLICATIONS OF SYSTEM COMPONENTS

As argued before the specific system costs were restricted by the modular set-up. Other applications of main system components are:

- Record keeping of radioactive sources. A file system has been designed, containing all relevant data of radioactive sources, in use or used in the Eindhoven University of Technology. Some examples of stored data which are directly accessible to the operator are: nuclide names, irradiation qualities, availability, manager, location and physical forms.
- In gamma and neutron spectrometry the spectra measured by a 4000 channel analyzer can be stored and modified in the "intelligent" floppy disk to fit in with the I/O-specifications of a mainframe computer system (Burroughs B7700) for spectra unfoldings.
- The microcomputer is used for calculations and model simulations, at present for example in risk analyses in radiation protection.
- To support the clerical staff we developed a text editing system by using the microcomputer and its peripherals.

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SPARK COUNTING TECHNIQUE WITH AN ALUMINIUM OXIDE FILM

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We have tried to use aluminium dioxide film as neutron detector film with spark counter. The merits of this method are that 1) aluminium dioxide is good insulator, 2) arbitrary thickness of the film can be made, 3) chemical etching of the thin film is dispensable.

Preparation of aluminium dioxide film: An aluminium plate (30 x 40 x 0.5 mm^3) was treated with dilute NaOH solution and HNO_3 solution before electrolysis. A nickel plate as a cathode and the aluminium plate as an anode were immersed in 15% sulfuric acid solution for oxidation of aluminium. Electric current density was 10~20 mA/cm^2 and the temperature was 12 ^{o}C. Electrolysis was made for 10~30 minutes. After electrolysis aluminium plate was rinsed with distilled water and kept in boiling water for about 30 minutes for hole sealing. Thickness of aluminium dioxide film was calculated with the weight of aluminium plate before and after electrolysis.

Neutron irradiation: An aluminium dioxide film deposited on an aluminium plate was attached with a 100 μg uranium target of 2.4 cm diameter electrodeposited on a stainless steel plate. This pair was fixed with scotch tape and irradiated with thermal neutron flux (5.7×10^5 $n/cm^2.sec$) produced by Kinki University's reactor UTR-Kinki (1 watt).

Spark counting: After neutron irradiation the aluminium dioxide plate was separated from uranium target plate and attached with an aluminized polyester sheet. As shown in Fig.1 high voltage was applied between the aluminized sheet electrode (cathode) and the aluminium plate electrode (anode), which was the base of aluminium dioxide film. Sparked pulses were counted with a usual scaler.

Results: Aluminium dioxide thickness depends upon time and temperature of electrolysis and sealing time.

Time of electrolysis: The relation between time of electrolysis and thickness of aluminium dioxide deposited at 12 ^{o}C is shown in Fig.2. Aluminium dioxide thickness was proportional to the time of

electrolysis until 60 minutes. Thereafter it gradually reached saturation near 120 minutes.

Temperature of electrolysis: The relation between temperature of electrolysis and thickness of deposited aluminium dioxide at electric current density of 20 mA/cm^2 for 30 minutes at 12~30 $^{\circ}$C is shown in Fig.3. The thickness of aluminium dioxide deposited reached 7~8 μm until 20 $^{\circ}$C and it rapidly decreased as temperature rose. Above 30 $^{\circ}$C aluminium dioxide film once formed was solved into the solution.

Sealing: After electrolysis aluminium dioxide was immersed in boiling water for hole sealing. Thickness of aluminium dioxide formed seemed to be independent to sealing time. The thickness increased about 1.5 times that before sealing as shown in Fig.4.

Electric resistance: The relation between electric resistance of aluminium dioxide film and sealing time is shown in Fig.5. Water vapour (100 $^{\circ}$C) sealing instead of boiling water was also examined. Electric resistance of aluminium dioxide depended on the thickness of the film. Above 7 μm of thickness electric resistance was almost constant value of 15 MΩ. Circuit tester (MΩ range) was used for measuring electric resistances. The infinity sign "∞" in the ordinate in Fig.5 means no deflection of indicating needle.

Sparking characteristics: Sparking characteristics, the relation between applied voltage and spark counts, were examined for aluminium dioxide films of several thickness as shown in Fig.6. The figures in parentheses show aluminium dioxide thickness in μm. High voltage was changed from 300 to 500 volts. There were short plateaus at 410~430 volts. Spark counts increased as thickness of aluminium dioxide film decreased. Higher voltages than plateau voltages were liable to cause multiple sparks.

Circuit constants: Sparking characteristics were investigated at various electric resistances and capacities in the sparking circuit in Fig.1. Fig.7 and Fig.8 show the relations between spark counts vs. electric resistances and that between spark counts vs.capacities, respectively, at applied voltage of 420 volts. Three curves correspond to three different uranium amounts of targets attached to aluminium dioxide films of about the same thickness at neutron irradiation. From these results 250 kΩ and 200 pF were adopted as suitable

resistance and capacity for routine measurements.

Spark counts vs. thickness of aluminium dioxide film: The relation between spark counts and thickness of aluminium dioxide film is shown in Fig.9. Dotted line shows spark counts of a detector irradiated without uranium target. No spark counts were observed in case of the thickness above 7.3 μm. Spark counts increased as the thickness decreased, however, if aluminium dioxide film was too thin, minute scars on the surface of the film were liable to spark.

Linearity: The relation between spark counts and uranium amounts of targets used at neutron irradiation attached with aluminium dioxide detector of about the same thickness is shown in Fig.10. Spark counting efficiency was calculated to be about 8.5%, which was lower than that of usual polycarbonate detectors.

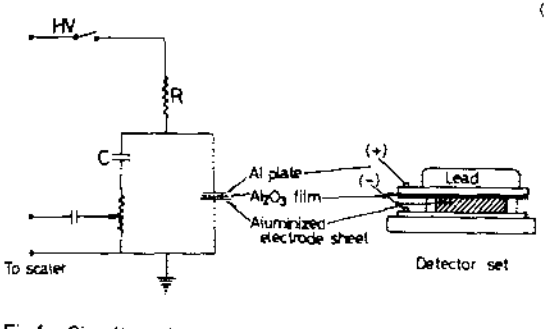

Fig.1 Circuit and arrangement of a spark counter.

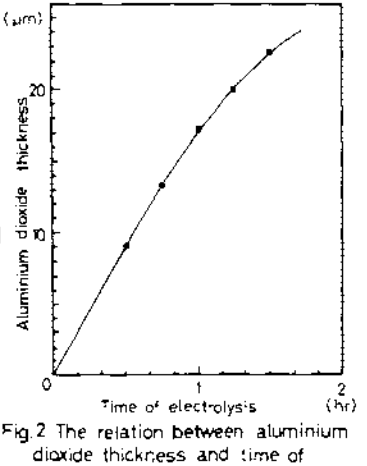

Fig.2 The relation between aluminium dioxide thickness and time of electrolysis.

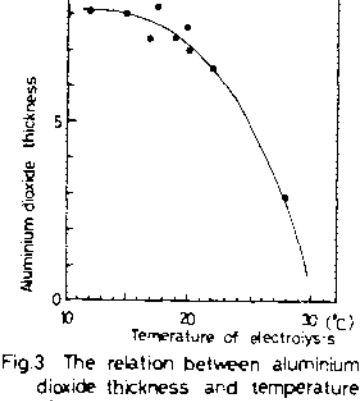

Fig.3 The relation between aluminium dioxide thickness and temperature of electrolysis.

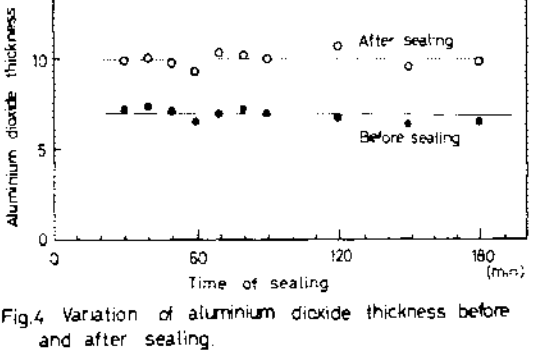

Fig.4 Variation of aluminium dioxide thickness before and after sealing.

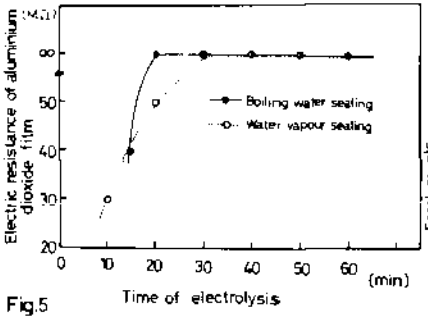

Fig.5 The relation between electric resistance of aluminium dioxide and time of electrolysis.

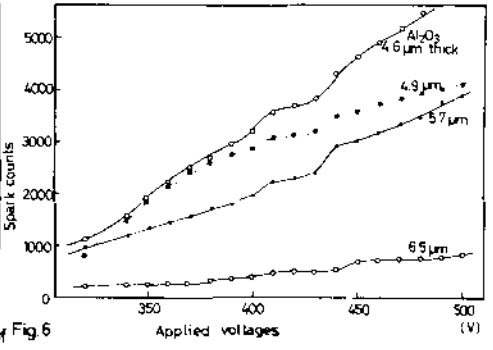

Fig.6 The relation between spark counts and applied voltages

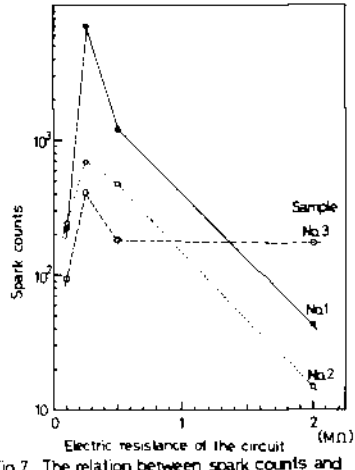

Fig.7 The relation between spark counts and electric resistance in sparking circuit.

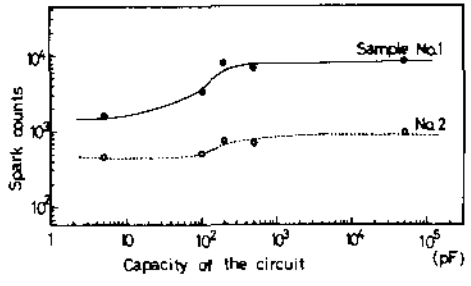

Fig.8 The relation between spark counts and capacity in sparking circuit.

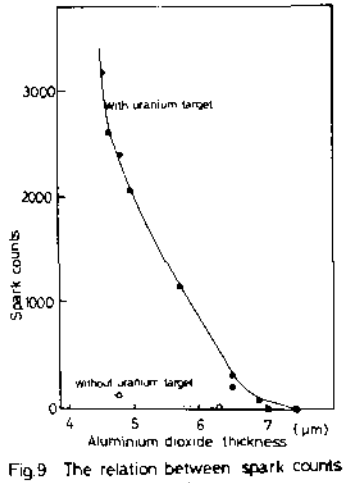

Fig.9 The relation between spark counts and thickness of aluminium dioxide.

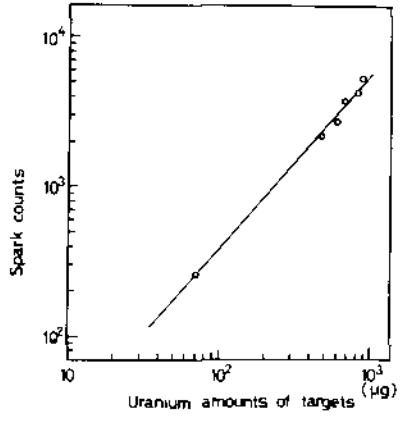

Fig.10 The relation between spark counts and uranium amounts of targets attached with aluminium dioxide detector.

RECENT DEVELOPMENT OF FLUORO-GLASS DOSIMETER IN JAPAN

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In most of the developing countries there are no centers for the calibration of instruments and sources. They have to ask the developed countries to calibrate the secondary standards (instruments and radiation sources) and then use these standards for the calibration of laboratory and/or field instruments. Naturally,such a practice leads to an increase of expences for radiological measurements. It might be possible that there would be a decrease in the accuracy of measurements because of the lack of proper operating knowledge by local personnel and changes in the calibrated instrument during shipment, in addition to the various inconveniences involved in the shipment of radiation sources and primary instruments for calibration. Therefore, it was proposed in 1968 to carry out the assessment of the accuracy and the reliability of the radiation protection measurements in Member States laboratories by mailing TOSHIBA radiophotoluminescent glass dosimeters. For this purpose a postal radiophotoluminescent glass dosimetry program was initiated in 1969-1970 by the IAEA. The TOSHIBA dosimeter glass pieces sent to the participating Member States laboratories for irradiation were returned to the IAEA and read out with a single reader,TOSHIBA Type FGD-6, to achieve a high degree of precision and avoid various sources of error. The results obtained in the past were considered extremely useful for the participating laboratories. For some time,this program was interrupted, but after the Three Mile Island Accident,the importance of Intercalibration Program is emphasized in some developing countries.

When silver activated phosphate glass is exposed to ionizing radiation, a stable luminescent center appears in it. When it is excited by ultraviolet light,the glass emits an orange luminescence having a peak wavelength of 500 to 750 nm. This is called radiophotoluminescence(RPL). Since the quantity of the luminescent emission is proportional to the radiation dose absorbed by glass, its application to the glass dosimeter has been developed. This was first applied to dosimeter by J.H.Schulman and others in the U.S.A. using radiophotoluminescence based on silver activated phosphate glass. Later it was improved several times by R.Yokota,Y.Nishiwaki,T.Omori and others in Japan, to develop the glass with a predose at 1/300,three times sensitivity,and little energy dependence. Special glass for neutron dosimetry was also developed.

The new TOSHIBA dosimeter system is not significantly affected by reading operation and enables repeated measurement and recording of the integrated dose. (Table-1,Fig-1, Fig-2).

Table-1. Glass Composition. (wt %)

	FD-1	FD-3	FD-3L	FD-4	FD-5	FD-6	FD-7
Li	3.67	3.58	Li 3.62	3.48	-	-	-
Na	-	-	-	-	8.94	6.60	11.00
P	33.45	34.53	34.52	34.04	33.13	33.16	31.55
O	53.71	53.51	53.49	52.74	51.34	51.39	51.16
Al	4.65	5.11	5.11	4.97	6.08	5.49	6.12
Ag	3.68	3.27	3.27	4.77	0.52	1.41	0.17
Mg	-	-	-	-	-	1.95	-
B	0.85	-	-	-	-	-	

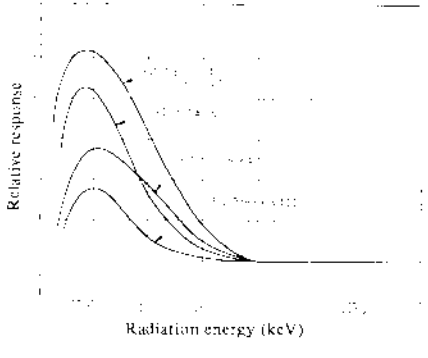


Fig-1. Energy dependence of different kinds of glass.

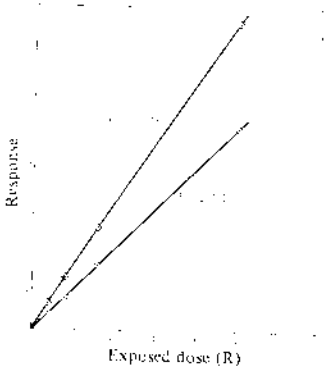
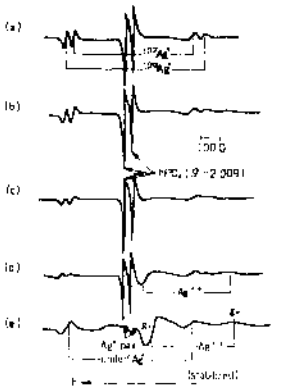


Fig-2. Dose characteristics of FD-3,FD-7.

The ESR spectral change after the $AgPO_3$ 1.07% doped with its composition of $LiPO_3$ 80% and $Al(PO_3)_3$ 20% is exposed to ^{60}CO gamma-ray radiation at 77K is shown in Fig-3. The structure of metaphosphate glass shares the oxygen with a PO_4 tetrahedron on its top and is in chain combination. When the PO_4 tetrahedron is exposed to radiation, it loses its electron and is stabilized,then seizes positive holes (hPO_4). Simultaneously with this, the Ag^+ ion in glass seizes a single electron and changes into the Ag^0 ion.



1.5×10^4 r exposure (at 77K)

1.4×10^4 r exposure (at 77K)

Item(b) is retained for 15 minutes at 195K

Item(c) is retained for 60 hours at room temperature.

Item(d) is retained for 360 hours at room temperature.

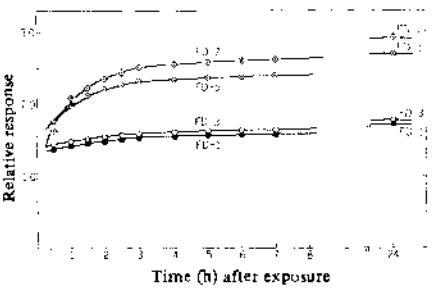
Fig-3. ESR spectra of Li-Al-metaphosphate glass

Once the positive hole has been seized by the PO_4 tetrahedron, hPO_4 is assumed to move to the Ag^+ ion due to the lapse of time and heat treatment, and then generate Ag^{++} ion. The Ag^{++} is also assumed to form a more stable Ag^0 pair. Build-up indicates that the quantity of luminescence increases and is stabilized with time and temperature, and sufficient build up is required in order that these Ag^0, Ag^0 pair and Ag^{++} can function as the stable luminescent center of the dosimeter glass.

The build-up time depends on basic glass composition, mainly on silver concentration. The smaller the silver concentration in glass is, the more hPO_4 is generated, and the build-up time is required to accelerate the reaction

$$Ag^+ + hPO_4 \longrightarrow Ag^{++}$$

The build-up curves of different kinds of dosimeter glass are shown in Fig-4.



Note here that the build-up rate of the FD-7 containing only about one twentieth Ag concentration of the FD-3 which contain comparatively more Ag concentration is almost equal to that of FD-3 in three hours after exposure. This is the reason why the latter is the Li glass with tight

Fig-4. Build-up characteristics of dosimeter glass.

structure,but the former is the Na glass allowing the Ag and Ag^{++} ions to be apt to move in glass and the RPL build-up to be rapidly completed. Also,the higher retaining temperature is,the more rapidly the RPL build-up is completed. When glass is heated at 100 to 120°C for 10 minutes,the build-up is immediately completed and the value with 1.1 times the normal value is obtained.

In order to improve the glass dosimeter, we tried to use the UV pulse (3371Å) by N_2 gas laser.
The pulse duration is 15 n sec and peak power is about 5KW. N_2 gas pressure is 6 Torr. Both ends of the laser tube have silica glass window with Brewstar angle.

The luminescence decay time of the predose and the oil or other organic surface contaminants is about ten times shorter than that of the RPL of the glass exposed to the ionizing radiation. Therefore, the signal due to radiation exposure can be measured separately by a delay circuit. With this method, the sensitivity can be greatly improved and it is possible to measure 1mR without being disturbed by a high predose or a surface contamination.

One of the difficulties was that the potential to be applied to the N_2 gas laser is high and the power source is large and expensive, but we succeeded in developing a small N_2 gas laser for this purpose.

In addition to the personnel dosimeter for relatively low dose range 1-100mR, a small portable reader for a relatively high dose range 0.5~1000R was developed as a personnel emergency dosimeter for accident and civil defence purposes. In this type of reader, a Xenon flash lamp is used as a source of UV excitation and a solar photocell with a high sensitivity for infrared is used as a sensor.

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TSEE DOSIMETER FOR GAMMA-RAYS AND FAST NEUTRONS USING CERAMIC BeO

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The dosimeter applying thermally stimulated exoelectron emission (TSEE) has been expected since Kramer[1] reported the linear relation between dose and TSEE. Its merits are high sensitivity, small size, and sensitivity only on surface. However it has not been used practically, since TSEE depends on the content of impurity, method of sample preparation, and surface condition.

Several different materials such as LiF, BeO, $CaSO_4$, CaF_2, $Li_2B_4O_7$ were examined by measuring TSEE glow curves in our laboratory. Of them ceramic BeO with special treatment was found to have high sensitivity and sufficient reproducibility when used as a dosimeter. In this paper are described method of sample preparation, TSEE measuring system, and characteristics of the dosimeter exposed to γ-rays and fast neutrons.

EXPERIMENTALS

Ceramic BeO called "Beryllia K-99" manufactured by "NGK Insulators Ltd"-Japan is 99% purity and in the disk form of 12 mm in diameter and 0.5 mm thick. Na^+ ion doping was made first by heating the disk at 1,400°C for 2 hours, cooling, then immersing in 1 M Na_2SO_4 solution, drying until Na_2SO_4 crystallized on the surface and finally heating at 900°C for 2 hours. The disk was placed in a recess, 12.5 mm in diameter and 0.5 mm deep, in a graphite holder as a dosimeter, whose outer dimensions were 15 mm diameter by 1 mm thickness. Before use, the dosimeter was annealed at 550°C for over 30 minutes to eliminate the effects of background radiation and mechanical excitation. Heatings were all done in air by an electric furnace.

For fast neutron measurement, a difference method[2] was used. Ten dosimeters of same sensitivity were selected and put in an aluminium holder. Five of them were each covered with a 2 mm thick polyethylene disk as a recoil proton radiator and another five each with a 2 mm thick polytetrafluoroethylene (Teflon) disk as a reference.

TSEE measurement of the dosimeter was made with a windowless gas flow G-M counter. The counting gas was Q-gas consisting of 99% helium and 1% isobutane. During the measurement, the dosimeter was heated up linearly at heating rate 1.5°C/sec from 100°C to 500°C. The pulse rate of TSEE from the dosimeter was recorded on an X-Y recorder

and also on a punched tape through a variable time-base counter every 5 seconds.

Responses of the dosimeter to γ-ray exposure and fast neutron dose were obtained by using radiation sources of 20 mCi of ^{60}Co, 1 Ci of ^{239}Pu-Be and 1 Ci of ^{241}Am-Be. The exposure to ^{60}Co source was measured by a Victoreen R-meter calibrated by "Electrotechnical Laboratory"-Japan. Total neutron fluence rate of ^{241}Am-Be source was subjected to international neutron source intercomparison at LRL-Berkeley in 1970. The fast neutron dose was calculated from fluence based on kerma factor of muscle[3] for soft tissue. The average neutron energy of the sources was assumed to be 4.5 MeV.

RESULTS AND DISCUSSION

Ceramic BeO without the above treatment shows low sensitivity to radiations, poorly reproducible glow curves, and spurious TSEE bursts of hundreds counts. In some ceramic BeO disks, successive measurements result in a glow peak at heater temperature of 280°C, which grows large with the number of measurements up to several. This may be related to the properties of ceramic BeO; this is, high specific resistance causing ion accumulation on the surface during measurement, or pyro-electricity.

Figure 1 shows a TSEE glow curve of ceramic BeO dosimeter with the above treatment for γ-ray irradiation, together with the background curve. The glow curve has two peaks, a higher one at 365°C and a lower one at 420°C. The sensitivity expressed as the number of exoelectrons counted per unit exposure is improved up to 1.2×10^4

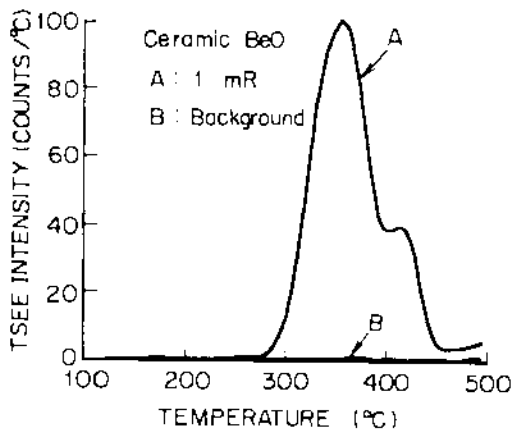

Fig. 1 TSEE glow curve of ceramic BeO dosimeter irradiated 1 mR of ^{60}Co γ-rays, together with background curve. Temperature is measured under a graphite sample holder.

counts/mR. The scattering in sensitivity of ten dosimeters chosen from the twenty for a 10 mR exposure is standard deviation ± 3.7%, while the reproducibility of a dosimeter exposed to 1 mR in five measurements is ± 2.0%. Fading of the radiation induced signals is not observed after keeping at 150°C for 1 hour. Sensitivity of some dosimeters gradually decreases with time in leaving them in the atmosphere of the room. By re-treatment, their sensitivity is recovered.

The TSEE response versus exposure is shown in Fig. 2.

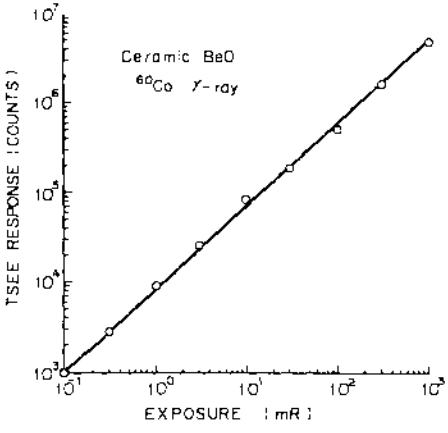

Fig. 2 TSEE response versus exposure.

It is linear on a log scale over the measured range from 100 μR to 1 R. In the figure, response R_g (counts) of the dosimeter for exposure X (mR) can be expressed by $R_g = 8.96 \times 10^3 \times X^{0.93}$. The lower detectable limit of exposure for the dosimeter is well below 10 μR, assuming that the significant difference between the net count of TSEE and the background count of the counter is more than three times the standard deviation.

The glow curves covered with polyethylene and Teflon disk respectively, irradiated by fast neutrons are similar in shape. But the former shows a higher peak due to the recoil proton component. Neutron response of the dosimeter is obtained by subtracting the reading with Teflon-covering from that with polyethylene-covering. The response of a group of the ten dosimeters previously selected and irradiated to a given dose is given by first expressing the individual mR equivalent calibrated by ^{60}Co γ-rays, then averaging each five dosimeters, and finally taking the difference. The TSEE response versus fast neutron dose is shown in Fig. 3. It is also linear on a log scale over the measured range from 500 μrad to 70 mrad, and the

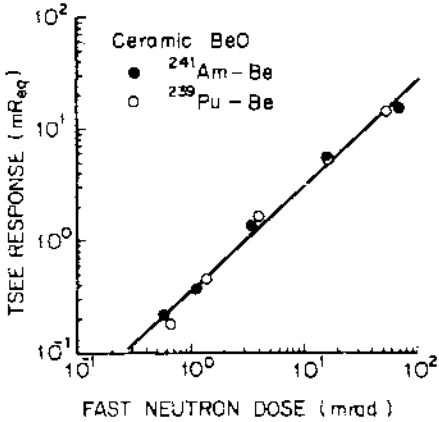

Fig. 3 TSEE response versus fast neutron dose. Response is average value of five pairs of dosimeters, and expressed by mR equivalent of ^{60}Co γ-ray exposure.

results with the two different neutron sources are in agreement. In the figure, TSEE respons R_n (mR_{eq}) of the dosimeter for D_n (mrad) can be expressed by $R_n = 0.37 \times D_n^{0.95}$. In this equation, the response for TSEE from 1 mrad of fast neutron dose is equivalent to that for 0.37 mR of γ-ray exposure. The lower detectable limit of fast neutron dose in a mixed field of γ-rays and fast neutrons is 14 μrad in 10 μR background, and 100 μrad in 1 mR, if the dosimeters are well calibrated and the TSEE response deviates only with statistical error of counting.

As seen above, the ceramic BeO dosimeter has outstanding characteristics such as suitable temperature of glow peaks, high sensitivity, and stability. Efforts are now in progress to develop dosimeters of high reliability in various circumstances, make number of dosimeters having the same characteristics, and attain higher sensitivity to radiation dose.

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DOSIMETRIC APPLICATIONS OF CELLULAR ELECTROPHYSIOLOGICAL CHANGES UNDER HIGH- AND LOW-LET IRRADIATION IN HEALTH PHYSICS[1)]

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1. INTRODUCTION

The knowledge of the relationship between the physical dose (tissue absorbed energy) and an observable biological effect is of great importance in view of the need for a sensitive biological dosimeter in radiation protection. The primary interaction of ionizing radiation with any form of living matter occurs at the cellular level. Partially disregarding the complexity of the target system many efforts have been made in molecular radiobiology to study changes induced by radiation in nucleic acids and proteins. However, the cellular component with the highest hit probability, the membrane, has only recently gained the necessary attention, e.g. in the structural-metabolic theory (1). Cell metabolism is highly dependent on its capability to maintain differences between extra- and intracellular ion-concentrations due to the functional fine structure of its membrane, characterised e.g. by specific bioelectrical membrane properties, such as transmembrane resting potential (MRP).

2. MATERIALS AND METHODS

Human embryonic lung cells (Flow 2002) were grown as monolayers in plastic petridishes, using conventional cell culture techniques. Some experiments were also carried out with transformed lung cells (WI38 SV13, subline 2 RA), which were grown in the same way. The petridishes were incubated at 37°C. Only cells in logarithmic growth phase were used.

The schematic set-up for the measurement of MRP in individual cells is shown in Fig. 1. The system consists of recording microelectrode, reference electrode, headstage with motorized micromanipulator, high-input impedance electrometer amplifier and phase-contrast microscope with automatic camera-system. MRP was measured with non-polarizable, 3 M KCl-filled glass micropipette electrodes, having a resistance between 15 and 20 MΩ. This recording electrode was mounted in a AgCl/Ag half-cell and connected to the headstage, housing a FET-input operational amplifier, capacitance compensation components and an internal driven shield. The output of the headstage was in on-line with the signal input of an electrometer amplifier (Transidyne, USA).

[1)] This research was sponsored by the International Atomic Energy Agency, research contract no. 2055/RB.

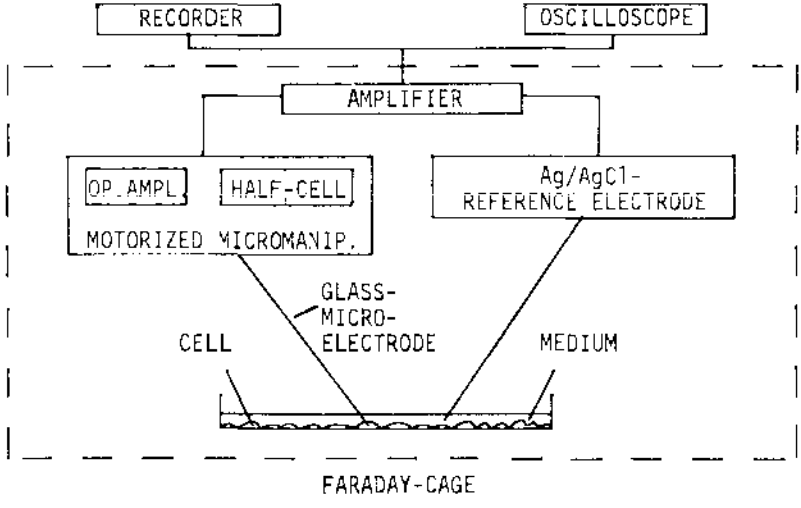

Fig. 1 Experimental set-up used for MRP-recordings from human cells

Amplifier output was connected to a dual-beam oscilloscope and strip-chart-recorder. A sintered AgCl/Ag electrode provided the ground reference to the preparation. For impaling of the cell a micromanipulator with both mechanical as well as three motorized drives and remote control unit was used. In order to approximate physiological conditions all intracellular measurements were carried out at 37°C and pH=7.2.

Cells were irradiated as monolayers in petridishes, covered by a thin layer of medium. For low-LET irradiation a Co 60 source (activity 37 GBq) was used to apply a dose rate of 20 mGy/min; for high-LET irradiation cells were exposed to a Cf 252 source (neutron flux 7.9×10^6 n/s) at the average exposure rate of 1 n/cell. MRP-values of the unirradiated cell were recorded for about 2 minutes, then the same impaled cell was irradiated either with gamma or neutrons.

3. RESULTS

Altogether 60 experiments have been evaluated so far. The change of the MRP for an unirradiated control cell in dependence of time after insertion of the microelectrode is shown in Fig. 2 (curve A). The MRP declines steadily to approach zero, indicating cell death due to membrane damage from the electrode impalment. For comparison, the complete set-up with both electrodes in the medium, but not impaling a cell, was irradiated under the same conditions with gamma-rays and neutrons. However, the influence was negligible and resulted in a DC-offset of only $\pm$ 0.5 mV (Fig. 2, curve B). Fig. 3 (curve A) represents the temporal MRP changes before and during gamma-irradiation of a normal lung fibroblast (Flow 2002). Within seconds after the onset

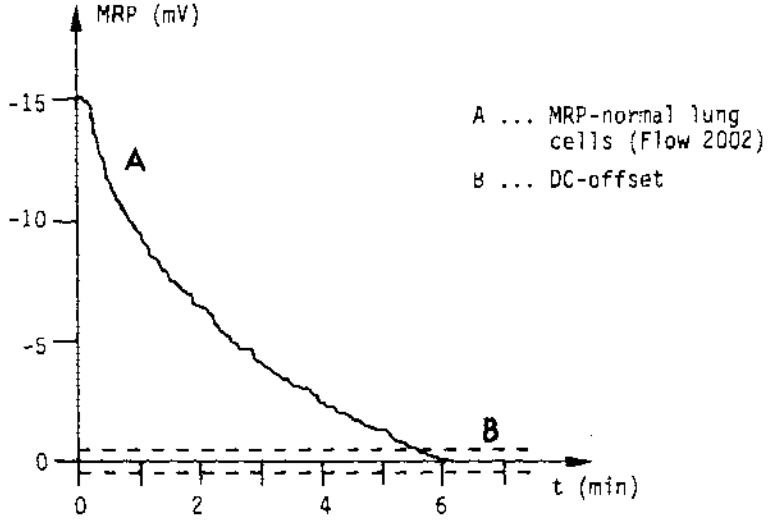

Fig. 2 Temporal MRP changes of unirradiated controls (A) and DC-offset of γ- and n-irradiated set-up (B).

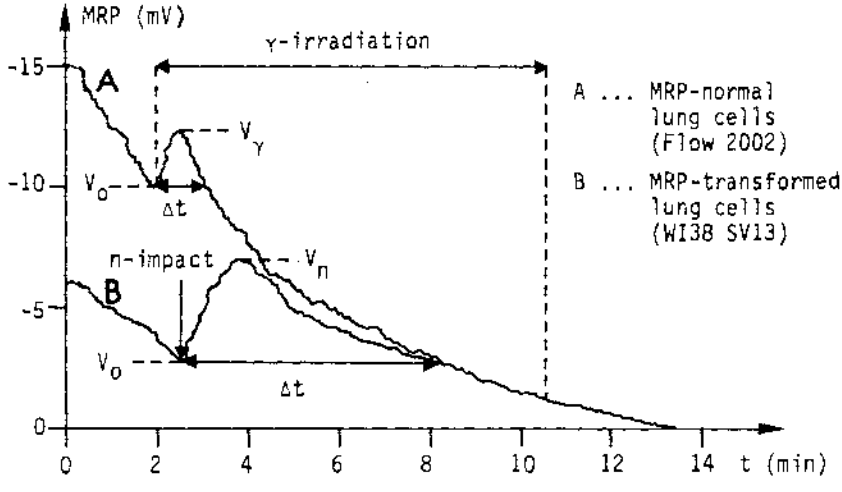

Fig. 3 Temporal MRP changes of γ-irradiated normal cells (A) and n-irradiated transformed cells (B)

of irradiation MRP starts to increase and reaches a maximum value (V_γ) within about 30 s. V_γ is more than 25% higher than the corresponding value of the unirradiated cell (V_0). During this time interval the cell has received a dose of only 10 mGy. Then the MRP declines to zero delayed, but in a similar way as the control in Fig. 2. Curve B in Fig. 3 shows the influence of n-irradiation on the cellular MRP in the case of a transformed lung cell (WI38 SV13). These cells show generally lower MRP-values than normal lung cells. Similar to gamma-irradiation n-traversal caused an increase of the MRP up to V_n, more than 150% above the comparable V_0-value. The peak width Δt is considerably larger for n- than for γ-irradiation.

4. DISCUSSION

It could be demonstrated that both, human normal and transformed lung cells show a pronounced temporary hyperpolarization of the transmembrane resting potential within seconds after low-level in vitro-irradiation. This cellular response varies in intensity and duration in dependence of the linear energy transfer of the irradiation mode applied. Many investigations have dealt with the influence of radiation on active bioelectric properties of excitable cells (nerves, muscles), mostly applying high dose values. 10% MRP-hyperpolarization above normal value was also observed during X-irradiation of frog skin with 7.5 kR (3), resp. of rabbit leucocytes with 10 rd (2). However, to our knowledge no data are available on continuous MRP-measurements before and during low-level radiation exposure of non-excitable human cells.

The MRP is caused by the difference between intra- and extracellular ion distribution, specific permeability for different ions and surface charges. The ions produced by the low-level irradiation cannot account for the observed increase in MRP (see also Fig. 2, curve B). However, the observed hyperpolarization could be caused by either an increased active transport due to an enhanced activity of the sodium pump, or by an increase, resp. decrease in the permeability for Cl^--ions, resp. Na^+-ions. All mechanisms mentioned can be interpreted as a feed-back response of the cellular system under the influence of low-level irradiation. At present different specific test-methods are being developed to elucidate the contribution of the described possible mechanisms to the observed hyperpolarization. Summarizing it can be concluded that MRP-measurements represent a highly sensitive test-system for correlating tissue absorbed low-level dose with radiation induced effects.

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ELECTROCHEMICAL ETCHING CR-39 FOILS FOR PERSONNEL FAST NEUTRON DOSIMETRY

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The registration of neutron-induced recoil tracks in plastics has become of particular interest for neutron dosimetry, since Sohrabi(6) has shown that these tracks can be enlarged up to macroscopic sizes by electrochemical etching(7).

Polycarbonate foils, which have been used so far for the electrochemical etching (hereafter ECE), are insensitive to neutrons with energies below about 1.5 MeV, and register heavy recoil tracks only, i.e. carbon and oxygen recoils(5). However, the dose delivered to biological systems is essentially due to proton recoils and thus an ideal neutron dosimeter should be capable of recording proton tracks(3).

Therefore, the discovery that the thermosetting polymer, known as CR-39, is an efficient proton detector represents an important novelty(1,2). In addition to this property, CR-39 foils have excellent morphological characteristics for the electrochemical etching(8,9).

In order to develop a personnel neutron monitor based on CR-39 foils, the registration characteristics of these detectors have been analysed both by chemical and electrochemical etching.

MATERIALS AND METHODS

Among the different types of CR-39 samples analysed, we have chosen the foils from American Acrylics, that are protected on both surfaces by 60-micron-thick polyethylene coatings, because of their satisfactorily low background(9). These coated foils have been irradiated at Livermore with normally incident neutrons of energies from 0.05 to 18.3 MeV. The irradiated samples were then etched and evaluated both at Livermore and Casaccia.

The chemical etching was made with 6 N KOH in water at 60 °C, while the electrochemical etching was obtained

using the same etchant for 5 hours at 25 °C and a.c. electrical fields of 30 kV/cm at a frequency of 10 kHz.

In this paper the registration efficiency is expressed as the number of tracks (or number of ECE-spots) for both surfaces per cm^2 per rem. In the case of chemical etching, recoil tracks from neutrons with energies less than 0.5 MeV are difficult to distinguish from background and considerable observer errors may be present. Therefore, we have used average values of track countings obtained by three different people.

CHEMICAL-ETCHING RESPONSE

The energy response for chemically etched CR-39 foils is shown in Fig. 1 for two different etching times of 5 and 10 hours respectively. The track registration efficiency is only slightly changed by doubling the etching time. Both curves of Fig. 1 drop considerably as the neutron energy increases up to 0.5 MeV. This drop in efficiency is mainly due to the decrease in the (n-p) cross-section and in the neutron fluence per unit dose. Above 1 MeV the registration efficiency rises to a maximum between 2 and 3 MeV and finally decreases appreciably at higher neutron energies. While the increase can be attributed to carbon and oxygen recoils which become detectable above 1 MeV, the rapid drop at the high energy end is due to the decrease of the neutron cross-sections of all reactions involved.

ELECTROCHEMICAL-ETCHING RESPONSE

The dotted line of Fig. 2 shows the registration efficiency of ECE-spots in CR-39 versus neutron energy(4). It can be seen that, under the particular electrochemical conditions mentioned above, neutrons with energies below 1 MeV are not detected and the response of CR-39 resembles that of polycarbonate(5). This indicates that tracks from proton recoils are not registered. As shown in Fig. 2, the registration efficiency changes remarkably if CR-39 foils are chemically etched for 5 or 10 hours prior to the electrochemical etching. These changes occur essentially within the first 5 hours of chemical etching.

The background of CR-39 foils electrochemically etched corresponds to a dose-equivalent of 20 mrem of 200 keV neutrons and about 5 mrem of 2 MeV neutrons.

Recent preliminary data from Livermore indicate that, with a pre-etching of only two hours, the peak of the registration efficiency above 1 MeV disappears, while the response below 1 MeV is not altered. The great differences in response among the CR-39 foils pre-etched for 0, 2 and 5 hours can be attributed to the different track formation times(8,9), which depend on the energy and type of the recoil particles. Finally, all the above data suggest that by properly adjusting the extent of pre-etching, the ECE-registration efficiency might becomes independent on the neutron energy.

DISCUSSION AND CONCLUSIONS

Among all the existing track detectors, whether they are nuclear emulsions or solid dielectrics, the CR-39 polymer is unique in recording nuclear tracks with energies below 0.5 MeV.

The chief disadvantages of the chemically-etched dosimeter are its high dependence on neutron energy, the tedious microscope examination required and the difficulties in track recognition (expecially at low energies). These drawbacks are eliminated by enlarging the chemically-etched tracks by ECE processes.

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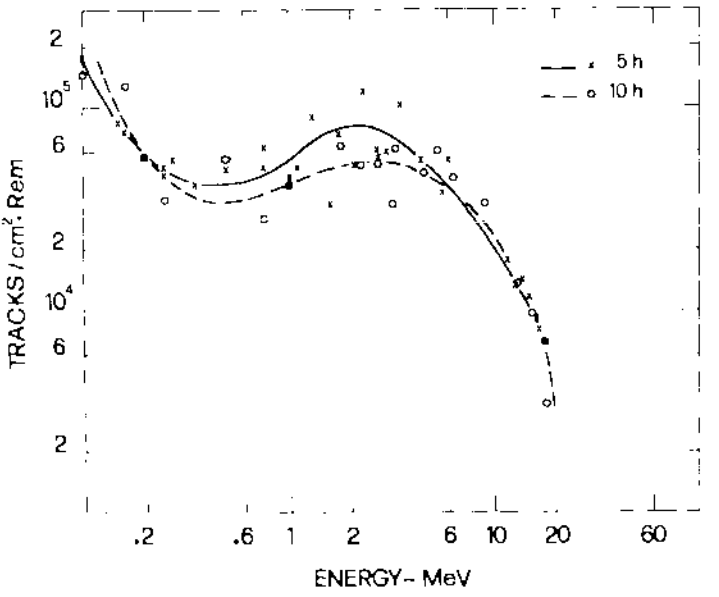

Figure 1. Chemical-etching efficiency vs energy.

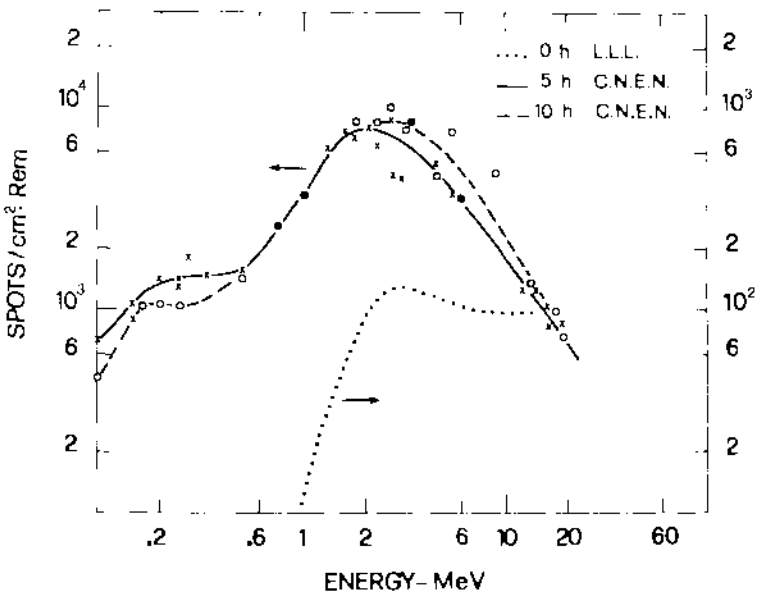

Figure 2. Electrochemical-etching efficiency vs energy.

CLASSIFICATION OF LiF-DOSIMETERS USING THE RATIO OF PEAK HEIGHTS

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INTRODUCTION

During the past decade thermoluminescence (TL) has received wide application in radiation protection and health physics. Because of its tissue equivalent response LiF has become a preferred substance as a detector material (1). The most widely used and best studied LiF dosimeter materials are the commercial products TLD-100 and TLD-700. When TLD-700 is irradiated at room temperature and subsequently heated to 220 °C five glow peaks are observed which are conventionally labelled 1 - 5, as shown in Fig.1.

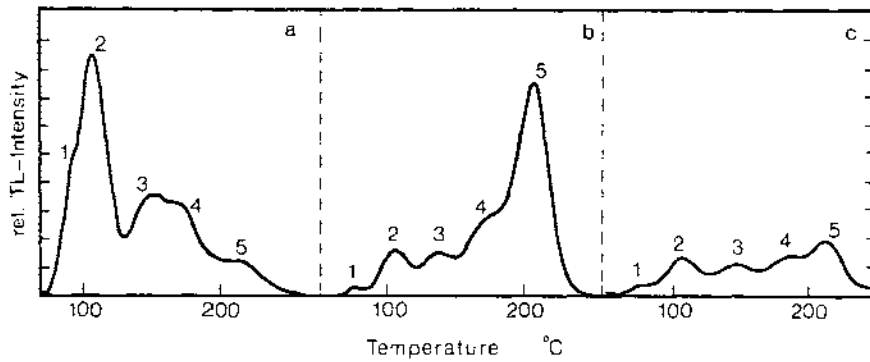

Figure 1. Glow curves of LiF:Mg, Ti specimen doped with various amounts of Mg and Ti.

Glow peak 4 and 5 near 200 °C are suitable for dosimetry in the dose range between 10^{-5} Gy (1 mrad) and 10 Gy (10^3 rad) because of their prominence and their stability at room temperature. For re-use LiF (TLD-100) must be standardized by a pre-irradiation heat treatment (1 h at 400 °C followed by a fast cool to room temperature).

Additional to the influence of heat treatments on the shape of the glow curve (2) there is a marked dependence of the glow curve's shape on the concentration of the activator impurities Mg and Ti. In a study of TL in samples from molten LiF and single crystals doped with Mg and Ti, we obtain different structured glow curves due to various concentrations of Mg and Ti. To describe the glow curve structure, which determines the quality of LiF for dosimetric application, we used the ratio of peak 5 height to peak 4 height and the ratio of peak 5 height to peak 3 height.

EXPERIMENTAL

For the preparation of samples from the melt the following procedure was used: Various amounts of MgF_2 and TiF_3 were added to LiF powder (suprapur, Merck) in a graphite crucible, which was heated up to 950 °C in an argon atmosphere. Then the furnace was switched off and the melt cooled within 30 minutes to room temperature. Afterwards the melt was crushed to powder and sieved. From this powder pellets were pressed. Additional single crystals were prepared using the Czochralski technique. For irradiation gamma rays from a ^{60}Co source and 30 kV-rays from a molybdenium tube were used. All samples were annealed at 400 °C for 1 h and afterwards cooled rapidly to room temperature by placing them on a copper block. After irradiation the glow curves were recorded (heating rate 2 °C/s) with an EG & G reader.

RESULTS

A careful study of glow curves from several specimen from the melt and from single crystals revealed strong differences in the glow curve structure. If the Mg-concentration (ppm by weight) was low (Mg/Ti = 1 ... Mg/Ti = 15) compared with the Ti-concentration glow curves similar to that shown in Fig.1a could be observed. This glow curve is typical for a material with "poor" dosimetric properties, because the contribution of glow peak 2 and 3 is greater than that from the main dosimetric glow peaks 4 and 5. If the ratio of concentrations Mg/Ti was about 30 we found dosimetric glow curves structured like that known from TLD-100 (Fig.1b). With increasing Mg-concentration the glow peaks broadened and peak 4 and 5 decreased to about the same height as peak 2 and 3 (Fig.1c). By using the ratio of peak 5 height to peak 4 height (or height ratio of peak 5 to peak 3) to describe the glow curve structure, we found a relation as shown in Fig.2. A classification in 3 categories (I, II, III) seems advantageous. Dosimeters in category II are of interest for dosimetric application.

DISCUSSION

Glow curve shape and TL-sensitivity determine the quality of LiF for the application in dosimetry. The glow curve of a material with high quality consists of low peaks 2, 3 and prominent peaks 4, 5 (high peak height ratio). So the peak height ratio allows,as a simple measure for dosimeter quality,classification of different commercial available LiF dosimeters. In different batches of the same material various dosimetric properties can be measured. We observed for example an outstanding quality improvement in charges of TLD-100 hot extruded chips (6.3 x 6.3 x 0.9 mm) produced between 1978 (peak 5 height/peak 4 height = 1.75) and 1979 (peak 5 height/peak 4 height = 2.1).

LiF of high dosimetric quality is obtained if the optimal Mg-concentration (which is determined by the peak of the curve in Fig.2) is added to the material with optimal Ti-content. The optimal Ti-content determines the TL-sensitivity of LiF (3) and the optimal Mg-concentration is necessary for a suitable dosimetric glow curve shape.

In each LiF:Mg, Ti material one can find in principle a relation between glow curve structure and activator concentration like that shown in Fig.2 but the position of the optimum is between Mg/Ti = 20 to Mg/Ti = 40. The actual position depends on the concentration of hydroxyl ions (OH^-) in LiF (4).

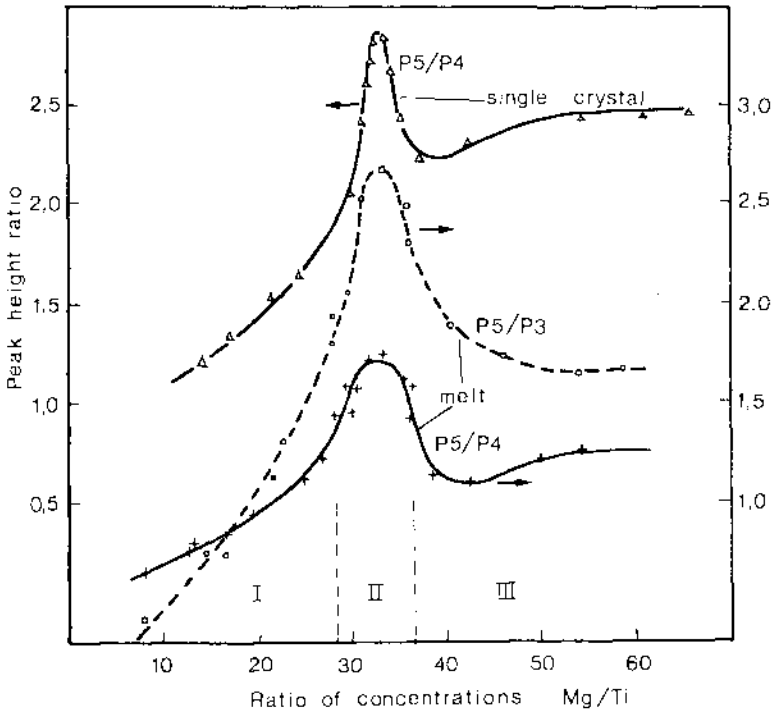

Figure 2. Peak height ratio depending on the concentrations (ppm) of Mg, Ti.
(P5/P4 ... ratio of peak 5 height to peak 4 height,
P5/P3 ... ratio of peak 5 height to peak 3 height)

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ENQUETE SUR LA QUALITE D'EXPLOITATION DES DETECTEURS GAMMA AU GERMANIUM

Estimation du taux de survie des détecteurs Ge : Li

Groupe de travail des spécialistes en spectrométrie γ de la Commission d'Instrumentation de la Radioprotection du CEA (FRANCE)
présenté par ZERBIB Jean Claude - CEA/SACLAY

Un fichier de détecteurs gamma semi-conducteurs Ge : Li ou Ge intrinsèques, a été constitué par un groupe de travail (GT) après enquête auprès des différents laboratoires. Cette étude regroupe 228 équipements installés en France et en Belgique.

Les informations prises en compte sont :

- l'année d'acquisition et le constructeur,
- le type et le volume du détecteur,
- son efficacité, sa résolution et le rapport pic/compton,
- le matériau constituant la fenêtre,
- le type et le volume du cryostat, les conditions de vide,
- les anomalies et incidents,
- le nombre de retraitements,
- la durée de vie.

Trois autres renseignements ont été également recensés pour aider le groupe de travail dans d'autres travaux tels que :

a) le diamètre extérieur du capot en vue de définir des géométries "standard" de grand volume (enveloppant le détecteur).

b) la tension de polarisation

c) la distance "fenêtre d'entrée du capot - surface du détecteur" (7 à 9 mm pour la moitié des détecteurs).

En partant de l'historique de chaque détecteur, les auteurs ont cherché à répondre à plusieurs questions telles que :

- espérance de vie des détecteurs,
- causes d'incidents et de décès,
- influence de l'origine du détecteur (constructeur)
- influence du type et du volume utile

Les résultats détaillés de cette étude font l'objet d'une publication qui paraîtra dans le premier semestre de 1980 dans un rapport CEA.

Il faut noter que la méthode d'interprétation des informations liées à la durée de vie suppose que les détecteurs sont tous de fabrication identique de 1965 au 31 décembre 1977, date de clôture du fichier.

Les résultats suivants ont été obtenus :

DUREE DE VIE MEDIANE (c'est à dire l'âge auquel la moitié des diodes est décédée) est de 6,1 ans - fig. 1 - pour une probabilité de 95%.

L'AGE MOYEN DES DIODES DECEDEES (33% du total) est de 3,9 ans.

C'est une estimation pessimiste car toutes les diodes ne sont pas décédées - fig. 2 - Pour les diodes les plus récentes, 41 détecteurs achetés au cours des deux dernières années, nous ne comptons qu'un seul décès.

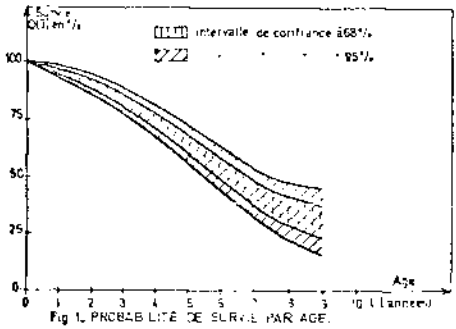

Fig 1. PROBABILITÉ DE SURVIE PAR AGE.

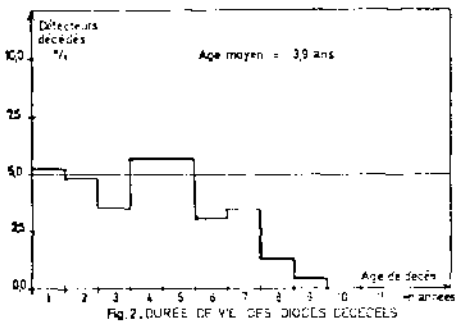

Fig.2. DURÉE DE VIE DES DIODES DÉCÉDÉES

DIODES DECEDEES - Elles sont au nombre de 76 soit 33% des détecteurs ; 24 ont eu au moins un retraitement, 8 en ont subi plusieurs. Le bilan des causes est :

- 36% par évolution des caractéristiques,
- 24% par défaut de vide
- 16% de décès accidentels
- 10% reformées pour non rentabilité

à ceci s'ajoute 14% de décès divers.

DIODES EN VIE - Elles sont au nombre de 152, soit 76% du parc total ; 95 n'ont subi aucun incident. Sur les 57 autres, 40 ont été retraitées dont 15 plusieurs fois. La fiabilité des cryostats à sortie verticale est supérieure à celle des autres dispositifs.- voir tableau 1 -

CAUSES D'INCIDENTS - Elles sont provoquées par défaillance du cryostat ou par un manque d'alimentation. Lorsque l'évolution est progressive, le suivi de la résolution et de l'efficacité est un bon indicateur d'incident.

RETRAITEMENTS - Des informations nous manquent pour évaluer l'espérance de vie du détecteur après retraitement. On note cependant que sur 64 détecteurs retraités (28% du total), 23 (soit un sur dix installations) ont dû l'être plusieurs fois. Il est important de ne pas oublier qu'au coût de l'opération (15% à 20% du prix d'achat) s'ajoutent les frais d'immobilisation et de réétalonnage de la chaîne.

EVOLUTION DES DETECTEURS

Le volume couramment utilisé est passé de 30 cm^3 à 60 cm^3, mais l'efficacité de détection ne dépend pas que du volume [17]. Les figures 3, 4 et 5 montrent que l'augmentation du volume, l'amélioration de la résolution et du rapport pic/compton ont été accompagnées par une augmentation de la tension de polarisation.

Les détecteurs Ge : Li coaxiaux sont les plus nombreux - figure 6 - mais ils seront dans l'avenir supplantés par les Ge Hyper purs qui représentent 42% des diodes acquises en 1977 contre 10% en 1976 et 1975. Sur les 17 détecteurs de ce type recensés, un seul est décédé. L'échantillon est trop petit et trop jeune pour se prêter à une analyse statistique.

TABLEAU 1

TYPE DE CRYOSTAT	SIGLE	NOMBRE DE DETECTEURS			RAPPORT DECEDES/ TOTAL(%)
		NOMBRE	EN VIE	DECEDES	
Canne Haute Droite		113	90	23 (20%)	23
Canne haute Coudée		34	23	11 (32%)	
Canne Basse Droite		4	3	1 (25%)	
Canne Basse Coudée		11	3	8 (73%)	50
Canne Basse Coudée Droite		5	4	1 (20%)	
Canne Omnidirectionnelle		3	1	2 (66%)	-

Conclusion : Pour un parc important, le renouvellement des diodes est à prévoir sur 6 ans. Au niveau d'un laboratoire, il est prudent de calculer l'investissement sur 4 ans. En améliorant l'emploi et la maintenance des détecteurs, on réduit tant les risques de retraitement que de décès.

$$x_x x_x x_x x_x x$$

/17 Groupe de travail spectrométrie γ des spectromètres équipés de détecteur à semi-conducteur - Rapport CEA / 4919 (1978)

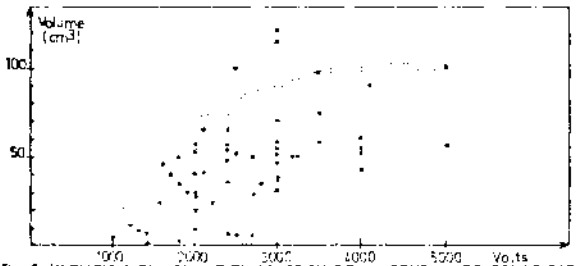

Fig 3_VARIATION DU VOLUME EN FONCTION DE LA TENSION DE POLARISATION.

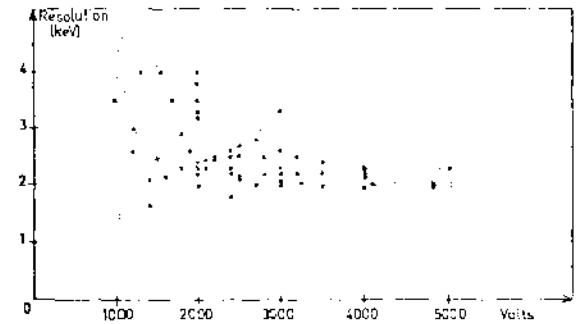

Fig.4_VARIATION DE LA RESOLUTION EN FONCTION DE LA TENSION DE POLARISATION.

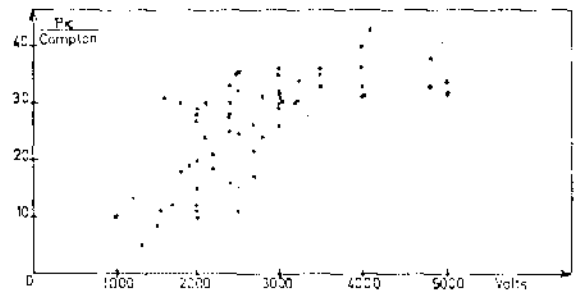

Fig 5_VARIATION DU PIC/COMPTON EN FONCTION DE LA TENSION DE POLARISATION

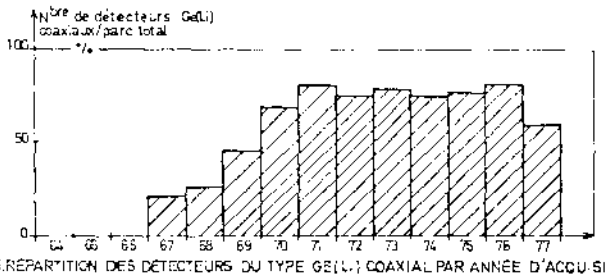

Fig.6.REPARTITION DES DETECTEURS DU TYPE GE(Li) COAXIAL PAR ANNEE D'ACQUISITION.

SURVEYING AND ASSESSING THE HAZARDS ASSOCIATED WITH THE PROCESSING OF URANIUM

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INTRODUCTION

The mining and milling of uranium covers the physical mining process, followed by the extraction of the uranium to a final concentrated uranium compound, usually ADU or oxide. Further processing and conversion of the uranium to other chemical forms or to the metal is normally associated with the nuclear industry and such plants are also located separately from the mining industry. It is generally accepted that exposure to radon constitutes the main hazard in mining operations whereas exposure to dust constitutes the main hazard in milling operations (1). The monitoring of radon and exposure during mining operations have received, and is still receiving, extensive attention. Uranium processing plants associated with the nuclear industry are controlled according to the norms applicable at nuclear installations, including full radiological surveying and monitoring. At uranium plants associated with the mining industry it is found that radiological surveying is infrequent if done at all. Assessment of the working environment is based on a limited air sampling program. Biological monitoring by means of urinassay, sometimes performed on selective sampling of the work force is used to assess the exposure of the work force (2, 3). Reliance is placed on good housekeeping and visual observation is used to judge the effectiveness of measures taken to prevent the spread of material.

The toxicology of uranium and the hazards associated with uranium processing have been extensively documented. A comprehensive résumé on occupational health experience with uranium was presented at a conference in Arlington, Virginia, USA in 1975 (4). Exposure due to the uptake of natural uranium depends on the solubility of the compounds in the body fluids. For soluble compounds the chemical toxicity to the kidney limits the uptake of material to 2,5 mg per day by inhalation, or 150 mg in two days by ingestion (5). For insoluble compounds the radiation exposure to the lung becomes the limiting factor. The permissible lung burden for long-term exposure can be calculated to be 26 mg (6). Since the original method of monitoring urinary excretion for assessing exposure due to the uptake of uranium (7), bioassay methods have been extended (8). There is, however, a lack of information on epidemiological data on human exposures.

This paper reviews the involvement in terms of radiation protection of several natural uranium processing facilities over a number of years in South Africa. The extent of the involvement differed appreciably for different facilities, and also changed in time. An effort will be made to use the large amount of data available from surveys to relate acceptable environmental conditions to exposure standards and to indicate how contamination control results in a de

crease of internal exposures.

THE SURVEYING OF PROCESSING FACILITIES

The survey information used in this analysis consists of routine static air sampling data and routine surface contamination smear data. Special survey data obtained during abnormal operating conditions or accidental releases to evaluate such situations, have not been included. The longer term consequences of such abnormal conditions are however reflected in the routine data. The frequency and extent of the routine surveys varied from time to time and were determined by prevailing circumstances. It is assumed that the routine data are a true reflection of the environmental levels over a period of time. It must be appreciated that due to such uncertainties as the representativeness of the static air sampling data, the selected frequency of surveying, and accidental releases, this assumption may not be true at any particular time.

The only method available for assessing personnel exposure over the period under review was bioassay by urinalysis. The routine urine sampling results are used as an indicator of personnel exposures. No effort is made to relate this to actual dose commitment, chemical or radiological. Where personnel exposures could be attributed to accidental releases which were not reflected in the routine surveys, the data were excluded from the analysis. As in the case of the survey routines, the frequency of urine sampling was determined by prevailing circumstances.

The collection of survey data refers to three different sets of circumstances:

Study A (9): In the early sixties a facility was operated for pilot plant work on the extraction of uranium from ore, and conversion to oxides, fluorides and metal. In 1964 health physics coverage was provided on a part-time basis, and this evolved to full-time coverage from 1967 onwards. Whereas the previous period included infrequent surveys, the latter period included regular routine surveys. A routine urine sampling program was undertaken from the beginning of operations, with a sampling frequency of once a month. Urinalysis results are available for the whole period under review.

Study B: In the middle seventies several pilot plants for experimental work on the conversion of uranium to oxides, fluorides and metal started operating. Health physics assistance was provided from the design stage of the plants and full coverage and surveys were provided from the onset of operations. Frequent and regular routine surveys were initiated. A monthly urine sampling program was initiated. As circumstances dictated and the work load increased, the frequency of urine sampling was increased to once a week at some of these plants. The feed materials to these plants are normally ADU.

Study C: Since the late sixties a number of surveys have been made on request at several uranium plants associated with the mining industry. These surveys do not constitute anything like routine coverage and represent infrequent samples taken at different plants. During any one survey an effort was made to obtain representative samples of the working environment of that particular plant. The surveys cannot be considered comprehensive as only a few of the existing plants were covered. Although routine urine sampling programs are

in existance, the results were not available to the author. The operations at these plants include extraction of uranium from the ore and concentration to ADU or calcining to the oxide.

ASSESSING THE HAZARDS ARISING FROM URANIUM

Considering the uranium series it is seen that although the parent emits only weak gamma-rays, and in itself presents a small external exposure risk, the daughter products do present an external exposure risk. During extraction of uranium the daughter products (with radium-226 the main hazard) are separated and these can be present in the waste stream, collected somewhere in the process, and may even to a smaller extent be carried through the product stream. After separation the first two daughters of uranium build up again to reach 50 % of equilibrium in about 24 days. The alpha contamination from uranium as well as the beta contamination from these daughters should therefore be considered in any uranium processing plant. The exposure due to external radiation will not be considered in this paper.

The working environment is judged against Derived Working Levels (DWL). For airborne material the (MPC) for uranium for a 40 hour week is used (10), namely 7×10^{-11} Ci/m^3 or 200 µg/m3 (11). For surface contamination the following DWL's were used (12):

Uncontrolled areas : 10^{-5} µCi/cm^2 for alpha and 10^{-4} µCi/cm^2 for beta.

Controlled areas : 10^{-4} µCi/cm^2 for alpha and 10^{-3} µCi/cm^2 for beta.

The relaxation by a factor of 10 for controlled areas was not used.

The exposure of personnel was judged against two urinary levels of uranium. The first level is considered as the minimum significant level of 10 µg/ℓ and serves to indicate a positive exposure. The second is the maximum permissible level of 100 µg/ℓ for uranium excretion.

RESULTS

Study A: For the purpose of comparison two periods are considered and totalled, i.e. before 1967 and after 1968. The surface contamination levels as they were found to exist in 1967, are used to describe conditions prior to routine health physics coverage, and levels at the beginning of 1969 to describe conditions after contamination control had been implemented. A survey consisted of 20 smear points and 8 air sampling points. A total of about 1 800 urine results over the whole period was used to describe personnel exposures. The results are tabulated in Table 1. The percentage of smears (either alpha or beta activity) that exceeds the DWL for controlled areas is shown together with the percentage of urine samples exceeding 10 µg/ℓ, those exceeding 100 µg/ℓ and the maximum average alpha air activity. (Subsequent to the publication of reference 7 a collection efficiency of smear sampling of 10 % has been defined as realistic. The smear results in reference 7 has been corrected accordingly).

TABLE 1

Period	% smears > DWL	% urine > 10 µg/ℓ	% urine > 100 µg/ℓ	Max. av. air activity in DWL
Before 1967	10	45,6	0,4	0,43
After 1968	0	9,3	0,1	0,1

Study B: The available data was considered for periods of time according to the sampling periods for urinalysis, i.e. for monthly urine sampling a period of one month was defined, and for weekly urine sampling a period of one week was defined. The data was furthermore divided into two groups as determined by the smear results over the period considered:

Group 1: All smear results during the period defined show levels less than DWL for controlled areas (alpha or beta contamination).

Group 2: During the period defined smear results showed one or more values in excess of the DWL for controlled areas (alpha or beta contamination).

Data from a total of 3 400 urine samples, 40 000 smear samples and 1 600 air samples are used in the analysis.

The results are given in Table 2. In each group the percentage of smear samples in excess of DWL (uncontrolled areas for group 1 and controlled areas for group 2), the percentage of urine samples in excess of 10 µg/ℓ, the percentage of urine samples in excess of 100 µg/ℓ, the percentage of air samples in excess of the appropriate DWL, and the maximum average air activity is given.

TABLE 2

Group	% smears > DWL	% urine > 10 µg/ℓ	% urine > 100 µg/ℓ	% air > DWL	Max. av. air activity in DWL
1	9,9	9,8	0,1	1,8	1,8
2	3,3	19,9	0,7	4,2	15,3

Study C: The results from the individual surveys were totalled in two groups. It was found that in those plants access to stages subsequent to the precipitation of uranium (be it mechanical or chemical) should be and, in some cases, was controlled. All stages preceding the precipitation of uranium (i.e. leaching, gravitational settling, extracting and ion exchange) can be considered uncontrolled areas. In practice, analytical laboratories are also in uncontrolled areas. The results are therefore divided into:

Group 1: Uncontrolled areas

Group 2: Controlled areas

A typical survey consisted of 40 to 60 smear samples and 10 to 16 air samples.

Results are given in Table 3. For each group the percentage of smears showing levels in excess of the appropriate DWL, the per-

centage of air samples in excess of the DWL and the maximum air contamination level is given.

TABLE 3

Group	% smears > DWL	% air > DWL	Max. air activity in DWL
1	14	0	0,1
2	22	30	3 000

The highest air activities were measured at specific points where, during normal operations, release of uranium was taking place. These points were defined as : ADU filtration at installations where these were not enclosed, oil-fired calciners and drum loading areas. Excluding these particular samples, the highest air activity was found to be 0,3 DWL.

CONCLUSIONS

It must be emphasized that the conclusions are of a general nature. A number of complicating factors are not represented. At different plants the uranium compounds differed largely and the uptake and excretion patterns differed widely. Effects of abnormal releases and of undesirable operating practises were not defined, although they may have been included in the routine survey results.

The survey data totalled over periods of bioassay show the relationship of higher exposures during larger contamination of the working environment. Air sample data, especially grab sampling, do not show a clear relationship.

In study A the improvement in exposure after reduction of surface contamination levels correlates well with the surface smear results. The air sampling, performed on a grab basis, does not correlate to the same extent. In study B higher exposures correlate clearly with higher contamination levels of the working environment. The air sampling was more continuous and therefore more descriptive of the air contamination than in study A. In total they also show the correlation of higher average air samples to higher exposures. However, over shorter periods they do not necessarily correlate with higher exposures - see for example the maximum averages recorded in the two groups. It is believed that this does not indicate exposure by ingestion instead of by inhalation, but rather points to the unrepresentativeness of static air sampling.

The Derived Working Levels used for surface contamination are illustrated to closely resemble levels that give rise to significant personnel uptakes, as judged from urinalysis. The use of an in vivo method of dosimetry is under way to further investigate and assess exposure of personnel. It is, however, concluded that in the light of uncertainties in personnel dosimetry, the lack of epidemiological data and to keep exposure to as low a level as is reasonably achievable, the DWL's above are realistic for control on personnel exposures. The relaxation by a factor of 10 is not advisable.

Although a DWL for uncontrolled areas (which is equivalent to 3 mg/100 cm^2 for uranium) is considered to give a visible indication of uranium contamination, study C shows that visual control is not truly representative of the control of material, and accordingly also not of the risk of exposure involved. Infrequent grab air sampling only is not adequate for assessing the hazard. Surface contamination surveys, together with frequent air sampling, are required for uranium processing plants. Proper area control and personnel access is indicated.

ACKNOWLEDGEMENTS

The author would like to thank past and present personnel of the Health Physics and Safety Subdivision of the Atomic Energy Board who were involved in the radiation protection program. In particular a word of thanks to Mr. J. Constantino, Health Physics Supervisor, and Mr. J.F. Beyleveld who assisted greatly in compiling the survey data.

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ACCIDENT IN A BWR OFFGAS CHARCOAL TREATMENT SYSTEM

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To date, the charcoal treatment system is the most widespread technique used to reduce airborne releases from the BWR main condenser offgas pathway to "near-zero" figures. The physical process involved is a selective adsorption of fission noble gases onto the charcoal, causing delay and eventually radioactive decay of most radionuclides. As a matter of fact, you can achieve delay times of a good deal of days for Xenon and a few days for Krypton, depending on their adsorption properties. The overall decontamination factor, defined as the ratio of inlet to outlet concentration, can range from 100 to 10000, according to several parameters. Under these conditions, the radiological impact of the mixture discharged can become almost negligible. Figure 1 illustrates a schematic diagram of a BWR charcoal treatment system showing the most important equipment involved.

Up to now, the offgas treatment design has been basically focused on the decontamination factor needed to comply with existing regulations, in terms of individual and collective doses to the public, during normal plant operation.

On the other hand, a possibility of accidents concerning the spillage of charcoal mass (e.g. due to an earthquake) has to be considered. Owing to the presence of thousands or even tens of thousands of curies adsorbed on the charcoal mass, the potential risk to the public in case of desorption may be considerable.

HISTORICAL BACKGROUND

As nearly all of the noble gas charcoal treatment concepts, even this one has been deeply investigated by Underhill (1,2). In the meantime the U.S. Nuclear Regulatory Commission (NRC) assigned a study on the subject to the Battelle National Laboratory (BNL) (3).

The U.S. Regulatory Guide (R.G.) 1.45 (4) has set up analysis criteria: equipment seismic class had to comply with the 500-mrem limit dictated by R.G. 1.29. According

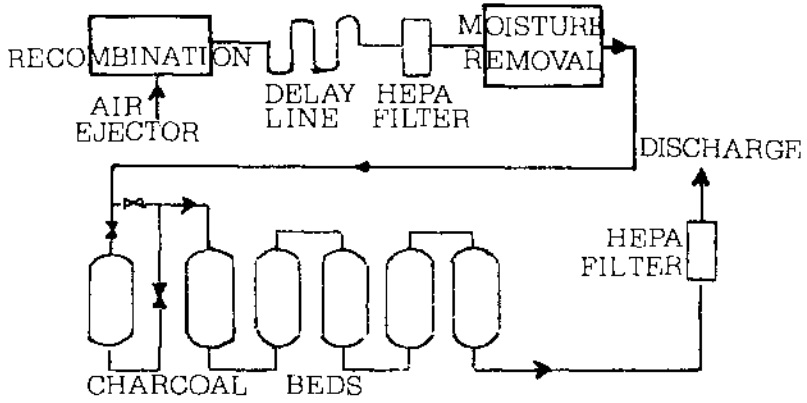

Figure 1. Flowsheet for an offgas ambient-temperature charcoal treatment system.

to the present NRC position, seismic criteria have to be applied to tank support elements and offgas building (5). General Electric's statement claims that calculated site boundary doses result anyway below the regulatory limit (6).

In the framework of the Alto Lazio nuclear power station licensing procedure (Preliminary Safety Analysis Report), the rupture accident has been assessed with the aim of deciding whether seismic class had to be prescribed for charcoal tanks or not. Seismic class was eventually chosen: R.G. 1.98 hypotheses were adopted for calculation.

DESIGN AND OPERATION FEATURES

This paper gives indicative results arising from Underhill's hypotheses (2) that appear more realistic than those of R.G. 1.98: especially the assumption that the whole noble gas inventory is released further to the accident looks overconservative. The most relevant parameters affecting the accident impact can be related to: 1) Offgas mixture conditions (inlet radioactivity, mixture type) 2) Treatment plant variables (transport time in the delay line, air flowrate, charcoal temperature and humidity, number of beds, bed mass, charcoal diffusivity) 3) Accident conditions (offgas production history, depth of spilled charcoal, mixing coefficient) 4) Site conditi-

ons (site boundary distance, wind speed, meteorological class, building cross-section, time elapsed after the accident) 5) Regulatory position. (7)

The two most interesting parameters to be assessed are the air flowrate and the charcoal depth after the accident. The combination of both gives rise (Fig.2) to "safety" and "non-safety" areas, having established the 500-millirem limit in between.

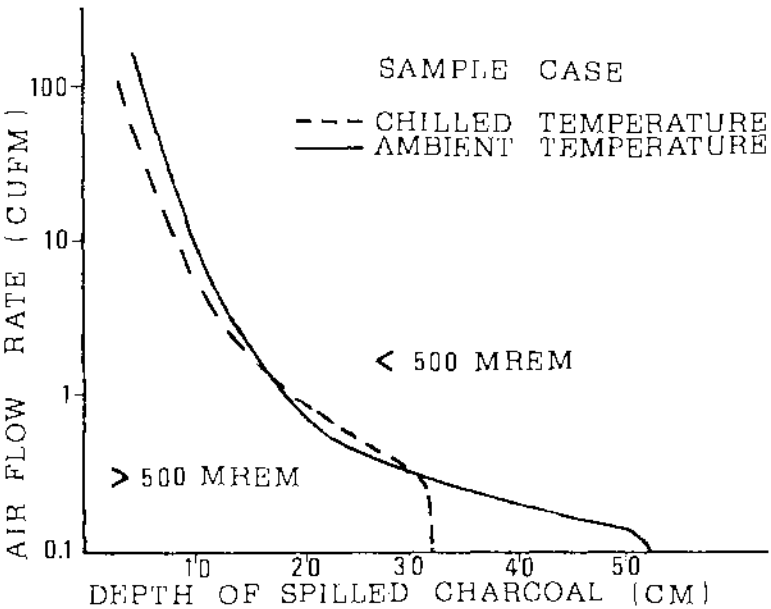

Figure 2. Effect of air flowrate and charcoal depth in complying with the 500-millirem limit.

The relevance of the stream flowrate can be readily seen; thus, ensuring a minimal flow will be enough to respect the above limit. This idea physically means spreading the inlet activity along several charcoal beds instead of storing it up on the first one: the concentration gradient inside the charcoal diminishes and so does - under the same conditions- the motive power for airborne post-accident transport. Such an evaluation could drive designers to install an extra air injection line, to be used when carryover air is insufficient and/or radioactivity noticeable. However, since an increased flowrate adversely affects the charcoal residence time, a few ensure beds could become a necessity in order to retain the increased activity transport during normal operation.

The layer depth further to the accident is of the utmost concern: a few centimeters can be enough in order to ensure that the majority of noble gases are retained and then decayed before escaping into the environment. A possible suggestion could be that a layout criterion should avoid the spillage of the charcoal on a vast surface.

Besides, a seismic design - at least for the first-in-line bed where most activity accumulates - could be very effective in order to greatly reduce any risk.

CONCLUSION

The BWR offgas charcoal treatment system appears a powerful device for reducing doses to the public during normal operation; however, the presence of relevant activities stored up on the charcoal can be a source of risk in case of a spillage accident.

The most interesting solutions to cope with this problem can be applied through both design and operating features. Among the design features, an adequate seismic class and/or a "tight" layout seem worthwhile, among the operation features, a minimal air flow could prevent "steep" activity gradients inside the beds.

Last but not least, the rupture accident analysis cannot be separated -at a design stage- from other design aspects of the offgas treatment system: any modification will have impact on other design decisions.

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TRITIUM CONTROL AT THE TRITIUM SYSTEMS TEST ASSEMBLY*

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The Tritium Systems Test Assembly (TSTA) is a computer-controlled facility designed to mock up full-scale the deuterium-tritium fuel cycle of next-generation tokamak fusion test reactors. Such reactors will use or build on the experience and technologies of the TSTA and other engineering facilities. The TSTA is presently under construction at the Los Alamos Scientific Laboratory and will be fully operational in 1982.

DESCRIPTION OF THE FACILITY

The TSTA will consist of a gas loop (Fig. 1) and associated control and safety support systems. The loop will include a mock-up torus vacuum chamber followed by cryogenic vacuum and impurity removal systems, a cryogenic hydrogen isotope distillation system, and gas analysis, transfer, mixing and injection systems. The gas loop will be designed to handle a flow of up to ∿360 moles of DT per day. This flow will provide operating experience on a scale that is equal to or greater than that of any of the cycles currently being planned for advanced reactors. To accomplish this will require an on-site tritium inventory of ∿150 g (1.5 MCi).

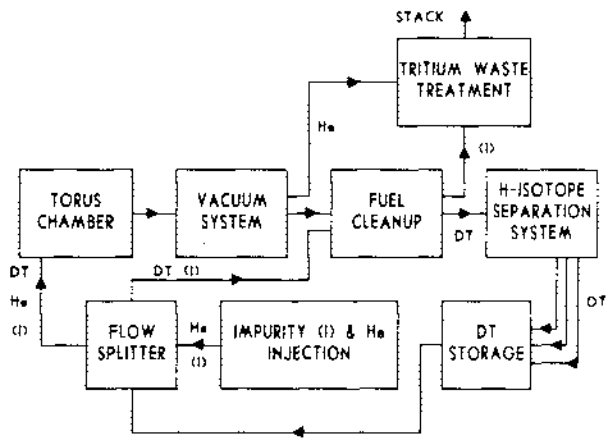

Figure 1. Simplified flow diagram of the TSTA. Note that DT also includes D_2 and T_2.

*Work performed under the auspices of the United States Department of Energy.

The facility is being constructed in an existing steel-frame structure with concrete floor and concrete block walls. The building consists of a large central experimental room (volume: $\sim 3\ 10^3$ m^3) where the tritium-handling systems will be located, and adjacent areas for the data-acquisition and control computer, mechanical and electrical support equipment, laboratories, shops, and offices.

The main experimental room and two adjacent laboratories where tritium will also be handled are on a separate, once-through ventilation system (4 m^3/s) designed to maintain this area at approximately 0.2 torr negative pressure. The exhaust air will be routed to the outside through a 30-m stack. The remaining rooms will be maintained at 0.2 torr positive pressure by a separate, partly-recirculating ventilation system. The two zones will be connected through air locks which will be maintained at local atmospheric pressure.

An artist's concept of the facility is shown in Fig. 2. Running the length of one side of the main experimental room are two pits 1.5 m deep. Overhead, about 4.5 m above the pit floor, are two steel mezzanines. With the exception of the torus and vacuum systems, all of the tritium handling components of the fuel loop will be located either on the mezzanines or in the pit areas.

DESCRIPTION OF SAFETY SYSTEMS

Two of the principal objectives of the TSTA are to demonstrate that the fuel cycle of a large scale fusion reactor incorporating

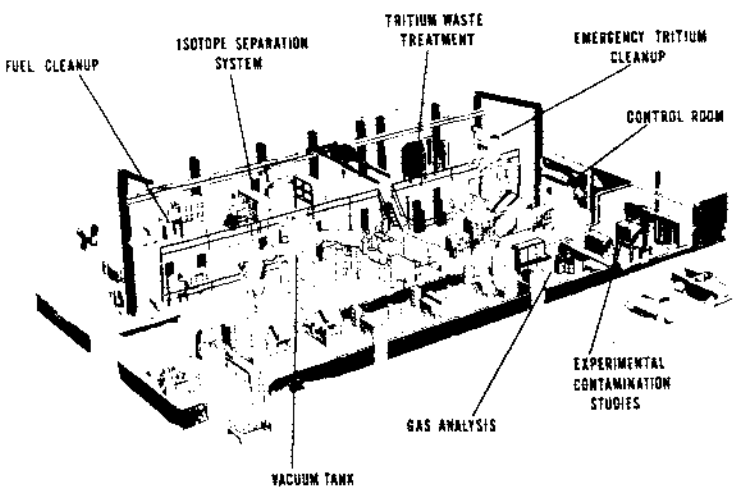

Figure 2. Artist's concept of the TSTA.

large quantities of tritium can be operated reliably and safely. With regard to safety, both for workers and the public, two design goals have been set, that of keeping TSTA personnel exposures below 1 rem/yr and that of maintaining stack effluents below 200 Ci/yr. These goals are the result of attempting to keep all potential exposures to tritium "as low as reasonably achievable." To accomplish these objectives, the TSTA will incorporate a number of safety features.

The first of these is multiple containment. All of the system components which may contain tritium will be doubly contained either by inherent design or by being placed in nitrogen-filled gloveboxes. Exceptions include the torus chamber, some of the associated vacuum system components, and the tritium cleanup systems. Interconnecting lines transporting tritium between sub-systems will be double-walled with one end of the secondary space open to a glovebox.

The TSTA will have two tritium cleanup systems, an on-line system, referred to as the Tritium Waste Treatment System (TWT) and an Emergency Tritium Cleanup System (ETC) for processing tritium releases into the main experimental room. The TWT is a moderate-size batch-type cleanup eystem with a maximum processing capacity of 32 ℓ/s. Routine tritiated effluents from all of the system components, pumps, etc., as well as releases into gloveboxes can be processed by this system. It incorporates a large receiving tank (5.7 m^3) kept below atmospheric pressure, pre-heater, precious-metal catalyst bed, compressors, high pressure receiving tanks, and molecular-sieve beds. The ETC is a 660-ℓ/s system similar in principle to the TWT but with two main differences: it is a continuous system (as opposed to a batch system), and is designed so that as much as 85% of the moisture in the air may be removed by refrigeration with the remainder removed by molecular-sieve beds. The result is that much smaller beds are required for efficient drying of the processed air. The condensed water can be placed on smaller beds for shipment or disposal or kept as water for recovery of the tritium.

Tritium instruments will help monitor the performance of the TWT and the ETC. All but one will be ionization-chamber monitors with the exception being a scintillation flow-cell monitor in the condensed-water line of the ETC. Glovebox atmospheres will be monitored with instruments equipped with screen-walled ionization chambers to eliminate the need for air pumps, which are often troublesome. The chambers will be covered with felt which will serve as crude dust filters. Seven tritium ionization-chamber instruments will monitor the air of the main experimental area with each instrument monitoring 2-3 points simultaneously. The ventilation exhaust duct will have its own monitor (with added integrating capability) which will be located about 5 seconds in transient time before the air at the monitor reaches a duct isolation valve. Stack monitoring will be performed by a similar ionization-chamber instrument in addition to a bubbler-catalyst-bubbler passive integrating system. The condition of each instrument will be routinely monitored by the computer.

A tritium release into a glovebox will be detected by its monitor, which will be set to alarm at either of two preset levels. The lower-level alarm will prompt only an investigation; with a

higher-level alarm, the glovebox atmosphere will automatically be flushed to the TWT. If any one of the room, duct, or stack monitors alarms at its high trip level, the ventilation intake and exhaust duct valves will automatically isolate the room. If a second such instrument similarly alarms, the ETC will automatically begin processing the room air.

For a major release of $\sim 10^6$ Ci of tritium into the room, cleanup of the air is expected to take about 24 hrs. Barring any leaks by permeation or through cracks, the total amount released to the environment should be under 5 Ci.

Experience has shown that at facilities such as the TSTA, the most common tritium exposures result from handling contaminated components during installation, maintenance, or removal of equipment. To minimize this source of exposure, much of the maintenance will be performed either within the gloveboxes where the equipment is located, or in a special glovebox dedicated to this function. Extensive use will also be made of specially designed flexible plastic enclosures equipped with gloves, and plastic suits supplied with breathing air. To eliminate the amount of tritium that would otherwise be released to the room (and might have to be processed by the ETC), the air within these plastic enclosures will be processed by the TWT or a small transportable scrubber if the level of contamination warrants it.

To minimize the risk of a significant loss of tritium to the environment should a release to the experimental room occur during a power failure, two supplementary power supplies will be installed: a diesel-powered 750 kVA emergency motor-generator set and an uninterruptable, battery operated, power supply. All tritium monitors will be on the battery supply as will be the data acquisition and control computer. In addition to supplying power to the ETC, the emergency generator will provide power to the battery supply, all pumps, valves, and other equipment necessary for a partial, temporary shutdown of TSTA. All valves and control equipment are designed to "fail safe" and in the event of complete loss of power, the TSTA would shut down in a safe manner.

Chronic releases to the environment will be under 200 Ci/yr. This will result in a maximum dose at the site boundary of less than 1 mrem/yr and a total local population dose of under 0.2 man-rem/yr.

Major accidental releases to the environment have also been analyzed. One accident scenario that was studied because of the proximity of the facility to the local airport is an aircraft accident involving penetration through the roof of the TSTA building. In the scenario, the inventory of the cryogenic isotope separation system (100 g) is released, oxidized by the accompanying fire and lost to the environment through the roof. With the plume rise from the heated gases, the resultant dose to an exposed person at the site boundary (0.4 km distant) is 1 rem. An unlikely loss of the entire TSTA inventory, oxidized and stacked, would result in a dose of less than 5 rem at the same boundary point. Releases of this magnitude to the environment as a result of damage caused by natural phenomena (tornados, earthquakes) are considered highly unlikely because of the infrequenny at Los Alamos of euch phenomena of the severity required to cause the necessary damage, and because of the safety features innorporated into the design of the facility.

TRITIUM CONTROL FIELD STUDY AT AN 'OPEN-CONCEPT' CANDU (PHWR) STATION

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The CANDU nuclear power generation concept is that of a pressurized heavy water reactor fuelled by natural uranium in the form of uranium dioxide.

Within the reactor core, neutron activation converts some of the deuterium in heavy water to the radioactive isotope of hydrogen, tritium. At present the average tritium concentrations in the moderator and heat transport liquids at the Bruce 'A' Generating Station are 7 Ci/Kg and 0.3 Ci/Kg respectively. In the absence of tritium removal measures, it is anticipated that the moderator equilibrium concentrations could be 6 to 10 fold higher, while those of the heat transport would have increased 7 fold.

There are two areas of interest: (1) occupational exposure (2) environmental emissions

Both of these occur as a consequence of heavy water escape from systems. Tritium control is therefore directly related to heavy water escape and, it is an essential component in the design and operation of CANDU reactors for both safety and economic reasons.

TRITIUM CONTROL CONCEPTS

The design features in support of tritium control can be considered as conceptual barriers which prevent or minimize occupational exposure to and/or environmental emissions of tritium. They are illustrated in Figure 1.

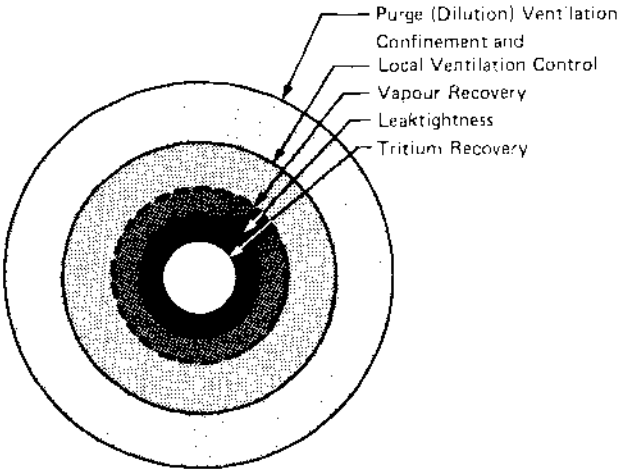

FIGURE 1 IDEALIZED CONCEPTUAL BARRIERS FOR TRITIUM CONTROL

The illustrated sequence corresponds to their relative capabilities in controlling environmental emissions and occupational exposures. The essential action and effectiveness of each barrier is summarized in Table I.

Tritium recovery is the most fundamental method available to reduce tritium concentrations in the heavy water. Thus, the consequences of the tritiated heavy water escape could be mitigated at the source. The subsequent barriers Leak Tightness, Vapour Recovery, Confinement, etc. are progressively less effective in the order shown.

THE BRUCE 'A' "OPEN-CONCEPT"

The design of the Bruce 'A' CANDU is such that all 4 reactors share a common containment structure. In addition, a single building shell without major internal dividing walls, surrounds the containment structure, the active equipment Confinement rooms, the Reactor Auxiliary Bays, and the Turbine areas. These "Open Concept" features are illustrated schematically in Figure 2.

TABLE I

Conceptual Tritium Control Barriers

Barrier Name	Barrier Action Reduces: Occupational Exposures	Environmental Emissions
Tritium Removal	Yes	Yes
Leaktightness	Yes	Yes
Vapour Recovery Confinement and Local Control	Yes	Yes
Ventilation	Yes	No
Purge Ventilation	Yes	No

TABLE II

Tritium Concentration Design Targets

Area	Design Target (MPC_A)
Turbine Areas	$\leq$ 0.01
Reactor Auxiliary Bay Areas	$\leq$ 0.1
Confinement Rooms	$\leq$ 1

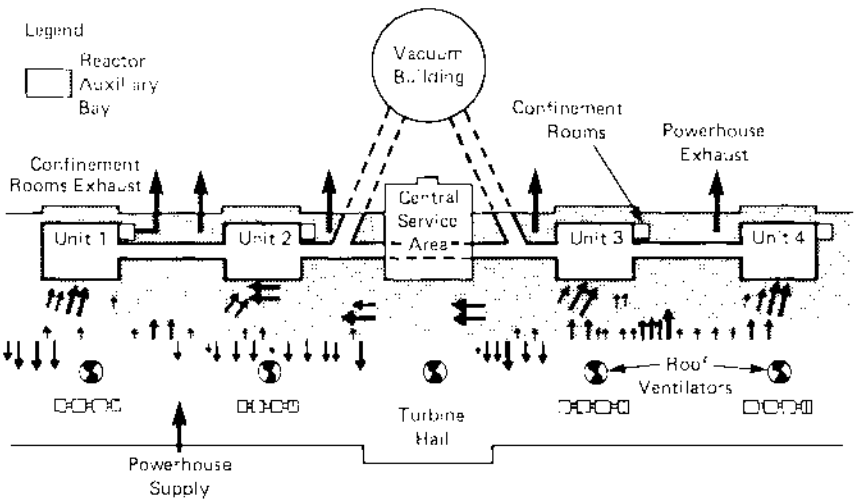

FIGURE 2 BASIC VENTILATION LAYOUT AND AIR VELOCITIES

BRUCE 'A' TRITIUM SOURCE POTENTIALS AND CONTROL

Most of the heavy water systems are located within the reactor containment structure. As such, they are inaccessible during operation. The design and operation of the containment structure prevents the escape of tritiated water vapour. This, coupled to a closed cycle vapour recovery system, ensures that the containment is a minimal source of environmental emissions.

Ventilated plastic suits are generally used for work in this area to control occupational exposure to tritium. For these reasons the study did not address tritium control within the containment structure.

Some heavy water containing equipment (usually for low pressure service) with a defined on-power access requirement, is located outside the containment. Most of this equipment, however, is located in Confinement Rooms which function as an additional barrier, preventing the spread of contamination into the surrounding reactor auxiliary bay area. The Confinement Rooms are ventilated to maintain desirable air quality, by means of purge ventilation. They are not currently serviced by vapour recovery systems. Vapour leakage within them is therefore exhausted to the environment via the filtered and monitored exhaust. Again, ventilated protective clothing may be used in these areas, when required.

Outside Confinement, i.e. within the Reactor Auxiliary Bay areas are located those systems that are/were deemed to possess a wholly insignificant vapour leakage potential. In this area, contamination control is achieved by means of purge ventilation, exhausting via a monitored, non-filtered exhaust.

Occupational exposure control is obtained both by this ventilation purge and at the source by leak tightness maintenance. Environmental emission control is provided by leak tightness only.

MOTIVATION AND SCOPE FOR THE FIELD STUDY

The tritium control field study was motivated by a requirement for design feedback on the performance and adequacy of the tritium control barriers engineered into Bruce 'A'. The gathered information is to be applied to the tritium control engineering for Bruce 'B' and Darlington 'A', two new stations presently in their design and construction phases.

The scope of the study was fourfold:

(1) Identify and quantify the tritium sources.

(2) Assess the adequacy of the ventilation system performance including confinement capabilities, capture and flushing of airborne contamination as well as determining air flow directions, velocities and volumes of exhausted air.

(3) Determine the present levels of contamination throughout the plant's accessible areas and estimate the associated maturity environmental emission and occupational exposure consequences.

(4) Make recommendations to ensure that both environmental emissions and occupational exposures remain below their maturity operating targets.

METHODOLOGY FOR THE FIELD STUDY

As a first step, the study defined the airborne tritiated vapour concentration limits which would have acceptable maturity environmental emission and occupational exposure consequences (summarized in Table II). These limits were arrived at by considering emission targets, exhaust flow rates, and station area occupancy factors.

In order to identify and quantify tritium sources, a technique was developed that made possible finding and measuring heavy water leakage-rates as low as 10 milligrams/day from individual valves or fittings. A portable infra-red spectrometer was the core of this technique.

The assessment of the ventilation system performance was performed by means of a propeller-type anemometer, backed up by smoke tests, and mappings of ventilation velocities on station layout grids. The airborne contamination assessment was done by long term sampling using some 50 molecular sieve tritium traps distributed throughout the study area. These measurements were repeated during three separate one week sampling periods. The station logbooks prior to and during these periods were reviewed to confirm that normal operating routine prevailed at these times.

STUDY FINDINGS

Source Identification: Approximately 6% of the process system components of one unit were leaktested. Although the selection of individual components was random, the selection of types and classes was not. A marked preference was given to component types and classes which in the past had exhibited leaktightness failures. The data should therefore be interpreted with caution. Of those tested, 70% were found to leak less than 0.1 g/day. Only 10% leaked the in excess of 1.0 g/day. However, this 10% contributed 95% of the assessed total leakage.

Ventilation Performance Assessment: The smoke tests gave qualitative indications that in the large open Reactor and Turbine areas the purge ventilation flows were frequently in a random direction. This was confirmed by the quantitative anemometer readings. The turbulences were mostly induced by thermal currents producing "weather-like" systems of drafts and air currents with velocities nearly an order of magnitude greater than net design purge ventilation flow. A typical flow velocity distribution pattern is illustrated in Figure 2.

Airborne Contamination Assessment: The findings of this assessment are summarized in Figure 3. While the average concentrations do not exceed design targets, local concentrations exceed targets in some cases.

CONCLUSIONS

The earlier identified absence of vapour recovery in the Confinement areas implies that for these areas, environmental emission control currently depends entirely on leaktightness (see also Fig. 1: Bruce GS 'A'). The extension of leaktightness maintenance will have a particularly significant impact on occupational exposures in areas having inadequate purge ventilation flows. Figures 4 and 5 plot the potential development of the occupational and environmental consequences as a function of time to 1988 for a hypothetical leaktightness deterioration of 10% per year. The projections include the expected buildup of system tritium concentrations assuming no "tritium recovery" is practiced.

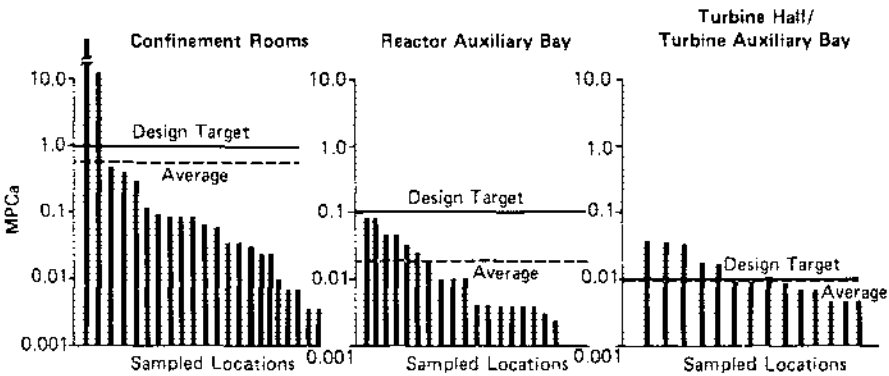

FIGURE 3 AVERAGE TRITIUM CONCENTRATIONS

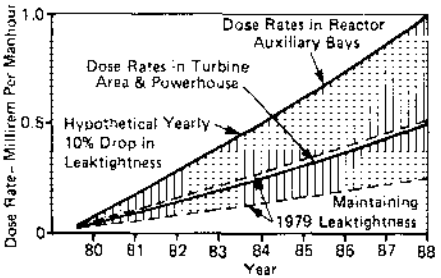

FIGURE 4 OCCUPATIONAL CONSEQUENCES PROJECTED TO 1988

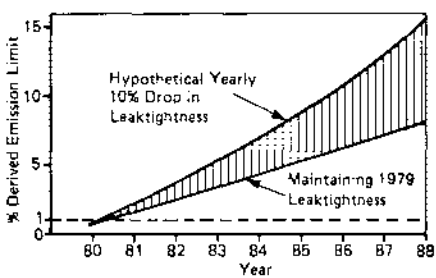

FIGURE 5 ENVIRONMENTAL CONSEQUENCES PROJECTED TO 1988 – ASSUMING NO TRITIUM RECOVERY

RECOMMENDATIONS

The recommendations were formulated to be applicable to both existing stations as well as to those under design and construction.

Leaktightness: it was recommended that leak performance data be compiled for all types and classes of components so that statistically derived leakage rates can be assigned during the design phases and give direction to tritium control engineering.

Vapour Recovery: It was recommended that all components and collections of components with the potential for contributing chronically 10^{-5} of the Derived Emission Limit or greater to the station tritium emissions be located in an enclosure serviced by the tritiated vapour recovery systems. This likely will require a significant extension of both Confinement enclosures, and drying capacity.

Confinement and Local Control Ventilation: It was recommended that all components and collections of components with the potential for contributing chronically greater than 0.1 MPC_a to the tritium contamination levels in the surrounding occupational area, either be located inside Confinement, or be subjected to local control ventilation.

Purge Ventilation: It was recommended that purge ventilation not be relied upon as a method for controlling airborne contamination in large open areas containing thermal sources.

Tritium Recovery: A decision to implement a tritium recovery program to process system heavy water will be dependent on its cost effectiveness relative to other control options, e.g. advanced dryers, increased component leaktightness, and extended confinement.

SUMMARY

An assessment was made of the future tritium control requirements for a CANDU "Open Concept" Pressurized Heavy Water Nuclear Generating Station. Data was gathered. Projections were made. Recommendations focussing on achievable improvements were formulated. A work program implementing these recommendations is presently underway.

BIOLOGICAL EFFECTS OF INHALED RADIONUCLIDES: SUMMARY OF ICRP REPORT 31

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A report on the biological effects of inhaled radionuclides was prepared by an International Commission on Radiological Protection Task Group charged with evaluating the hazards associated with inhalation of plutonium and other radionuclides.[1] The Task Group enumerated the biological responses, identified tissues and cells at risk, derived risk coefficients for inhaled radionuclides from animal experiments for comparison with human data, and determined an Equal Effectiveness Ratio for alpha emitters relative to beta gamma emitters.

BIOLOGICAL EFFECTS

Since there are no human populations (other than uranium, fluorspar, and other miners who worked in mines containing high concentrations of radon decay products) that have shown health effects that can be associated with radionuclide inhalation, it was necessary for the Task Group to use data from animal experiments to describe the biological effects. Radionuclides deposited in the respiratory tract are either "insoluble" (not readily translocated to other tissues or excreted) or "relatively soluble" (more readily translocated to other tissues or excreted). Biological effects resulting from the inhalation of radionuclides depend upon the distribution and retention of these radionuclides in the body and upon the doses to the tissues irradiated.

Life-Span Shortening

The Task Group compared the mean or median survival time of animals exposed to radionuclides with those of appropriate controls. Only data from alpha emitter experiments were used because of the paucity of beta-gamma emitter data. Figure 1 shows the shortening of life span in animals that inhaled PuO_2. At deposition doses below about 0.01 μCi/g lung, the shortening of mean life span was less than about 10%. Several experimental groups had mean life spans greater than control groups. At doses above about 0.01 μCi/g lung, life-span shortening increased with dose. The dose causing a 50% reduction of mean life span was between 0.05 and 0.1 μCi/g lung. Similar results were obtained for soluble alpha emitters; the dose causing a 50% reduction of mean life span was about 0.1 μCi/g lung.

Pathologic and Clinical Responses

The shortening of life span following the inhalation of radionuclides was accompanied by and/or caused by certain pathologic changes, some of which were reflected in clinical signs and symptoms. Some of these effects appear to be nonstochastic, i.e., the degree of effect, rather than its occurrence, is a function of dose; examples are lymphocytopenia, respiratory insufficiency, pulmonary and lymph node fibrosis, and cellular metaplasia. Other effects, such as pulmonary and bone neoplasia, are stochastic since the probability of occurrence is related to dose. Since a full spectrum of doses has not been investigated for any inhaled radionuclide in any animal species, it is not possible

[a]This summary was prepared under contract EY-76-C-06-1830 with the U.S. Department of Energy.

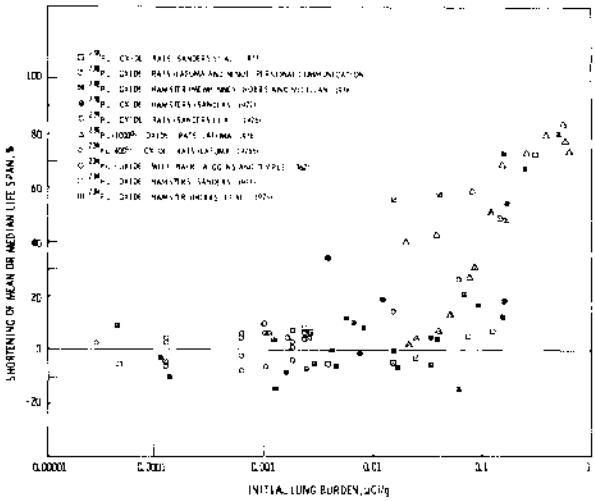

FIGURE 1. Shortening of Life Span by Inhaled PuO_2[a]

to know the lowest doses at which any biological effect *will occur* nor even the highest dose at which it *will not occur*. However, the available data were used to base judgments about the lowest dose range at which given nonstochastic effects are likely to be seen. For stochastic effects, such as death causing neoplasia, estimates of the frequency of occurrence were developed.

Hematologic Effects. Hematologic effects occur after inhalation of radionuclides because of the irradiation of the hematopoietic tissue to which the radionuclides are translocated and deposited, and because of the direct irradiation of blood circulating in lungs, liver, and perhaps lymph nodes containing the radionuclide. Leukocytopenia was observed in animals after inhalation of soluble forms of alpha emitters or beta gamma emitters. The most consistent hematologic response observed in dogs after inhalation of PuO_2 and insoluble beta-gamma emitters was lymphocytopenia. The Task Group concluded that lymphocytopenia is probably a nonstochastic effect, and that it probably can be detected after pulmonary deposition of ≥0.0005 μCi PuO_2/g lung. Much higher doses of inhaled insoluble beta-gamma emitters are required to cause a detectable lymphocytopenia, probably on the order of >0.01 μCi/g. Since both the magnitude and time of onset of lymphocytopenia are dose dependent, lymphocytopenia caused by these low doses would not occur until long after exposure to the radionuclide and might be marginally detectable.

Nonneoplastic Lesions. Respiratory insufficiency (characterized by increased respiratory rate, decreased arterial P_{O_2} and increased P_{CO_2} caused by diffuse fibrosis of the lungs) and alveolar edema may cause death within a month after inhalation of large amounts of alpha or beta-gamma-emitting radionuclides. Death from respiratory insufficiency and fibrosis may also result after exposure to much lower doses and may occur in rodents after many months and in dogs after several years. Respiratory insufficiency is the major cause of death in those groups of animals that showed a greatly reduced mean life span, Figure 1.

The available data from several animal species indicate that respiratory insufficiency and possibly death resulting from extensive fibrosis in the lungs caused by radiation from inhaled radionuclides are nonstochastic processes and might be expected to occur after alveolar deposition of >0.01 μCi alpha emitters/g lung and of >0.5 μCi beta

[a]See ICRP Publication 31 for references.

gamma emitters/g. If the retention times of the radionuclides in the lungs are short, larger quantities of the radionuclides would have to be deposited in the lungs to cause these lesions and early death.

Lymph nodes containing radionuclides may be severely damaged. The primary lesions are characterized by lymphadenitis and fibrosis with partial to complete depletion of germinal centers. Primary neoplasia does not appear to occur in lymph nodes of experimental animals that have inhaled alpha emitters or beta-gamma emitters. The dose-effect relationship for nonneoplastic lesions in the tracheobronchial lymph nodes is poorly known. Although lesions have been described in rodents, most of the dose-effect data from experiments with dogs suggest that fibrotic and/or atrophic lesions in the tracheobronchial lymph nodes are nonstochastic responses and that the dose at which these responses may be expected to begin to be observed is an alveolar deposition of about 0.001 μCi/g lung. This applies to insoluble alpha emitters such as $^{239}PuO_2$. In the case of more soluble alpha-emitting compounds, which are not as readily accumulated in the tracheobronchial lymph nodes as $^{239}PuO_2$, larger amount would be required to produce lymph node lesions. The dose of insoluble beta-gamma emitters required to produce such lesions is not known, but is probably greater than 0.05 μCi/g lung.

Neoplastic Lesions. Pulmonary neoplasia has been identified in miners of uranium and other substances as a consequence of inhaling alpha-emitting radon daughter radionuclides. Although pulmonary neoplasia has not been reported in human beings who have inhaled other radionuclides, such as the transuranic elements, it has been well demonstrated in experimental animals as a potential consequence of inhaling several different alpha-emitting and beta-gamma-emitting radionuclides.

In Figure 2 the incidences of lung cancer are plotted against the initial lung burdens of soluble alpha-emitting radionuclides. In rats, initial lung burdens above 0.001 μCi/g had an increasing probability of causing lung cancer. The low incidences at high doses reflected the shortened life spans due to deaths from causes other than neoplasia which prevented the full lung cancer potential from being expressed. At initial lung burdens below about 0.001 μCi/g, none of the experiments showed statistically significant increases in lung cancer, although several lung cancers occurred.

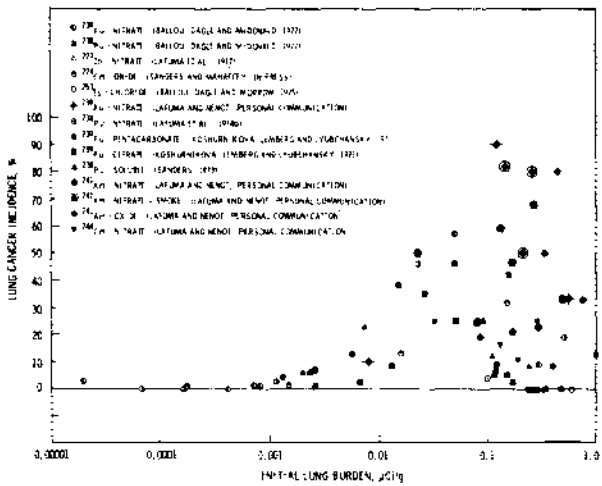

FIGURE 2. Relationship Between Incidence of Lung Cancer in Experimental Animals and Initial Lung Burdens of Inhaled, Relatively Soluble, Alpha-Emitting Radionuclides[a]

[a]See ICRP Publication 31 for references.

Like the data for relatively soluble alpha emitters, the lung cancer incidence for insoluble PuO_2 increased markedly at initial lung burdens above 0.001 μCi/g. Low-dose experiments are still in progress in which groups of dogs were exposed to initial lung burdens as low as 0.00003 μCi $^{239}PuO_2$/g lung and 0.000016 μCi $^{238}PuO_2$/g lung. In several experiments at different laboratories it was observed that hamsters were relatively insensitive to the induction of lung cancer. Thus, comparing species susceptibility to lung cancer caused by inhaled PuO_2 at the dose range where data exist (~0.01 μCi/g initial lung burden), beagle dogs were more sensitive than rats, which were more sensitive than mice, which were more sensitive than hamsters.

In studies of beta-gamma-emitting radionuclides deposited by inhalation or by intratracheal injection, initial lung burdens above 0.1 μCi/g led to increasing incidences of lung cancers. There are no data at lower doses.

While recognizing the possible shortcomings of using experimental animal cancer incidence data, the Task Group believed that, in the absence of a human data base, a quantitative descriptive model for radionuclide carcinogenesis in the lungs based on the animal data could be useful for risk assessment. Since a model could not be devised from hypotheses of the mechanism(s) of induction of cancer by inhaled radionuclides, the Task Group chose the logarithmic probit model usually employed in dose-response analysis and contrasted it with the linear model usually used to reflect conservatism. The incidence values for the linear regression model were weighted on a basis of the number of observations.

The original data for alpha emitters, uncorrected for control mortality, and the fitted functions (heavy solid lines) in Figure 3, suggest that insoluble alpha emitters were slightly more effective than soluble alpha emitters in causing pulmonary cancer in experimental animals.

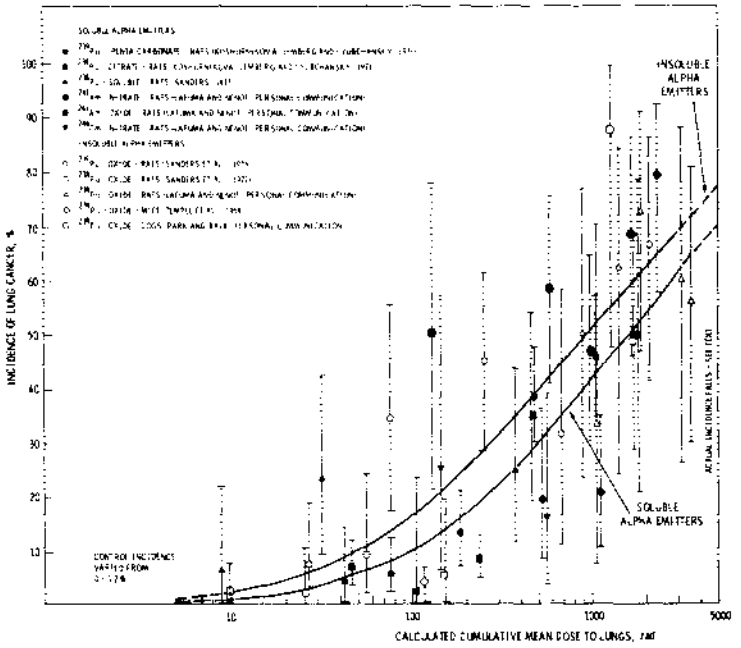

FIGURE 3. Relationship Between Incidence of Lung Cancer and Alpha Dose to Lungs from Inhaled Soluble and Insoluble Alpha Emitters: Probit Analysis[a]

[a]See ICRP Publication 31 for references.

Both the linear and probit models gave an adequate description of the alpha-emitter incidence results over the range of observed doses, Figure 4. However, the linear model used to describe the beta-gamma-emitter incidence data gave unrealistically high projected results at low or zero doses, Figure 5. Although the probit model gave more realistic values at low doses, the slope of the line was very shallow.

The improved Mantel-Bryan procedure also was used to obtain lung cancer risk estimates based on data from animal experiments. The Task Group compared these estimates with published estimates obtained by other methods (usually by linear extrapolations) from both experimental animal data and from limited human data, the latter mostly from external radiation exposures, Table 1.

The risk coefficients obtained by the Mantel-Bryan procedure and by extrapolating the fitted probit models are lower than those obtained from the fitted linear model and other published linear models using animal or human data. This means that for these experimental data, linear models yield a more conservative estimate of lung cancer risk than the other models. The applicability of these lung cancer risk coefficients to human beings who have inhaled or may inhale radionuclides is not known. However, the Task Group believed the risk estimates calculated from available animal data, summarized in Table 1, are supportive of the ICRP decision in Publication 26 (1978) to use a risk factor for the lungs of 2×10^{-3} Sv^{-1} (20×10^{-6} rem^{-1}).[2]

Comparison of the risk estimates obtained by analysis of all the alpha-emitter data, 25 and 36×10^{-6} rad^{-1}, with the risk estimate of 0.84×10^{-6} rad^{-1} for beta-gamma emitters gave an Equal Effectiveness Ratio of about 30 for inhaled alpha-emitting radionuclides. Thus, the experimental animal data tend to support the decision by the ICRP to change the recommended quality factor from 10 to 20 for alpha radiation (ICRP, 1977).[2]

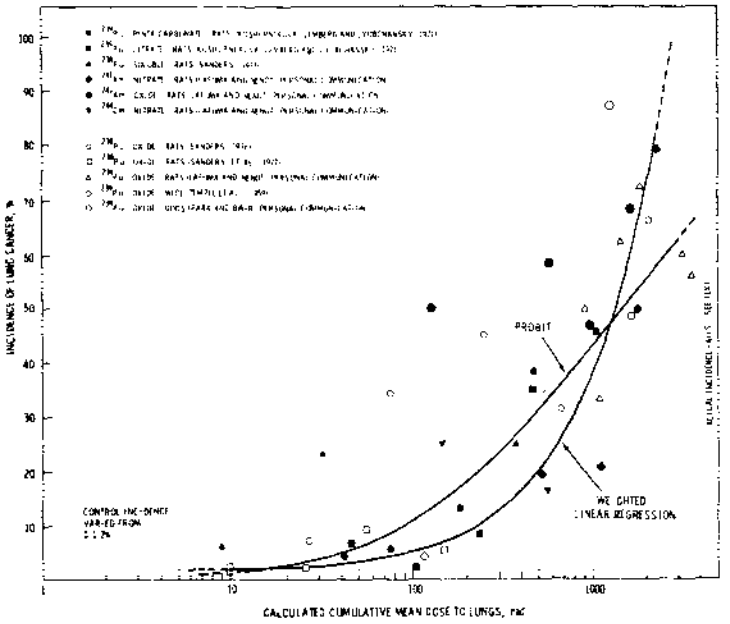

FIGURE 4. Comparison of Weighted Linear Regression and Probit Models as Descriptors of Alpha-Induced Animal Lung Cancer Data[a]

[a]See ICRP Publication 31 for references.

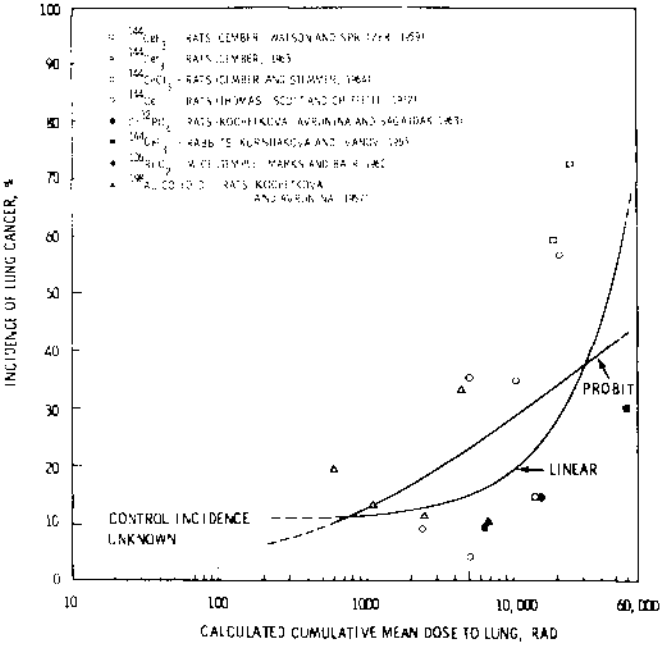

FIGURE 5. Comparison of Weighted Linear Regression and Probit Models as Descriptors of Beta Gamma Induced Animal Lung Cancer Data

Extrapulmonary Lesions. Various tumors in tissues other than those of the respiratory tract have been observed in animals after inhalation of alpha-emitting or beta gamma emitting radionuclides. These have occurred mostly in animals that have inhaled relatively soluble radionuclides which translocated from lungs to other tissues in the body. It is well documented that alpha-emitting radionuclides such as radium deposited in bone tissue can cause osteosarcomas in human beings as well as in experimental animals. Thus, bone neoplasia can be expected to be a potential consequence of inhaling alpha-emitting radionuclides if sufficient quantities are translocated to bone. In experimental animals, bone neoplasia has occurred at dose levels of about 0.01 μCi/g lung of inhaled soluble plutonium and other transuranic elements. Inhaled beta gamma-emitting radionuclides such as ^{90}Sr and ^{144}Ce, which deposited in skeleton, also have been shown to cause skeletal neoplasia in experimental animals.

With the more highly transportable transuranic elements such as einsteinium, curium, and americium, the incidences of extrapulmonary and extraskeletal cancers were increased in experimental animals. These included kidney and bladder carcinomas and thoracic and abdominal lymphoreticulosarcomas. Although many of the inhaled radionuclides studied in experimental animals translocated to liver, relatively few liver cancers were reported. Since these usually occurred at long times after exposure it was concluded that liver cancer could be one of the predominate late consequences of inhaling radionuclides.

Numerous neoplasias were observed in the nasal cavities of animals that inhaled alpha emitters such as radon and uranium and beta-gamma emitters such as ^{91}Y, ^{144}Ce, and ^{90}Sr, probably as a result of continuous irradiation of the nasal epithelium.

All of these neoplasias are associated with tissues and organs in which inhaled radionuclides are deposited or accumulated following translocation from the respiratory tract. However, the fact that a tissue accumulates radionuclides does not necessarily

TABLE 1. Summary of Risk Coefficients for Radiation Induced Lung Cancer

Animal Species	Model	Risk Coefficients (cases of lung cancer per million animals or persons per rad) Insoluble	Soluble	all	Beta-Gamma Radiation	Reference[a]
Rodents and dogs	Improved Marshall-Byrom	70	20	25	0.84	ICRP-31
Rodents and dogs	Probit	65	20	36	-	ICRP-31
Rodents and dogs	Linear		-	365	-	ICRP-31
Rats	Linear	1600	800	1250	-	Bair and Thomas, 1976
Dogs	Linear	-		650[b]	-	Bair and Thomas, 1976
Man	Linear	-	-	400[b]	20	Thorne and Vennart, 1978
Man	Linear	-	-	500[b]	25	MRC, 1975
Man	Linear	-	-	200		Mays, 1976
Man	Linear	-	-	400[b]	20	BEIR, 1972
Man	Linear	-	-	200-800[b]	10-40	UNSCEAR, 1972
Man	Linear	-	-	500[b]	25-50	UNSCEAR, 1977
Man	Linear	-	-	400[b]	20	ICRP 26, 1977

[a] See ICRP Publication 31 for references.

[b] Values converted to rad from rem on basis of a Q factor of 20 for alpha radiation

indicate a susceptibility to cancer induction. For example, the thoracic and, to a lesser extent, hepatic lymph nodes have been shown to accumulate concentrations of radionuclides that exceed by many times the concentrations retained in lungs or that occur in other tissues. This is especially true for insoluble compounds. Primary neoplasia of thoracic and hepatic lymphatic tissue has not been reported in any of the experiments with inhaled alpha- or beta-gamma emitting radionuclides. In life-span studies with beagle dogs that inhaled $^{239}PuO_2$ or insoluble ^{144}Ce particles, metastases of primary lung cancers were found in thoracic lymph nodes, as were occasional lymphangiosarcomas and several hemangiosarcomas. There were no other lymph node cancers. Thus, lymph nodes in experimental animals appear to be much less susceptible to cancer induction than other tissues in which inhaled radionuclides are deposited or accumulated. Further, inhalation of radionuclides is not known to be related to the induction of lymph node tumors in any human being. These observations influenced the ICRP's decision to consider the lymph nodes with the lungs as one composite organ for radiation protection purposes.[3]

CELLS AND TISSUES AT RISK

High lung burdens of inhaled radionuclides result in profound structural and functional changes in which the pulmonary capillary endothelial cells are the most prominent cells at risk. Pulmonary carcinogenesis is the most serious effect of low doses of inhaled radionuclides. The cells at risk are the precursor cells and basal cell layers of the respiratory tract epithelia. The Task Group considered the possibility that certain types of neoplasia induced by inhalation of radioactive material could be related to the presumed cell(s) of origin, and the cell lines at risk, but recognized the types of neoplasias produced may also depend on the pattern of spontaneous tumor development in a given species and strain. For instance in uranium miners small cell carcinomas, associated with lower radiation exposures, and epidermoid carcinomas, associated with higher initial expostures, appeared to originate in the larger proximal bronchi. In animals after inhalation of radionuclides, adenocarcinomas and epidermoid carcinomas appeared to originate in peripheral regions of the lungs. If the types of neoplasia observed after inhalation of radionuclides do suggest the cell line at risk, basal cell layers and Kulchitsky cells of bronchial epithelia may be at risk in uranium miners who develop undifferentiated small cell and epidermoid carcinomas. For adenocarcinoma in experimental animals, the cells at risk appear to be bronchiolar and, possibly, type II pneumocytes or Clara cells.

The occurrence of hemangiomas in lungs of animals that have inhaled radioactive particles suggests that the endothelial cells of pulmonary capillaries are also at risk. At risk also are the cellular constituents of bronchial and tracheobronchial lymph nodes, which may accumulate large doses or radiation. These include the T and B lymphocytes, germinal center cells, plasma cell precursors, endothelial cells and possibly reticulum cells, histiocytes, fibroblasts, and other mesenchymal elements. Endosteal bone tissue, hematopoietic marrow, and liver and spleen tissue may also be at risk from radionuclides translocated from the lungs. Lymphocytes appear to be at high risk from inhaled radioactive particles because blood lymphocytopenia is among the earliest and most sensitive changes observed in experimental animals. Since inhaled radionuclide-induced lymph node tumors as well as lymphomas and leukemias have been very rare in animal experiments, nothing can be said about tissues and cells at risk for these types of neoplasia. The possibility of genetic damage to germ cells after inhalation of radionuclides was not excluded by the Task Group but was not addressed because of the lack of data.

HOT PARTICLES

The Task Group believed that knowledge about the behavior of inhaled alpha-emitter particles in the lungs and the interaction of alpha irradiation with cells is inadequate either to support or completely deny the hot particle theory concerning the induction of lung cancer. Animal experiments indicate that the lung cancer risk associated with inhaled plutonium particles in quantities that could be distributed in hot spots may be slightly greater than for more soluble and, therefore, more diffusely distributed alpha emitters. Other experiments with plutonium microspheres clearly showed that "diffuse"radiation sources in the lungs of hamsters were much more likely to cause both malignant and benign lung tumors than highly localized sources. The Task Group concluded that the risk of lung cancer from inhaled radioactive particles will be greatly overestimated if based on hot particle concepts.

OTHER FACTORS

The Task Group considered possible modification of the effects of radionuclides by inhalation of other potentially damaging agents. Only a few animal experiments have addressed the question of combined effects of inhaled radionuclides and air pollutants or smoking. Results of these few studies are inconclusive. Therefore, such factors had to be ignored in addressing the objectives of the report. However, it was stressed that the possibility that the effects of inhaled radionuclides could be greatly influenced by smoking, and air pollutants should not be ignored in protecting human beings from airborne radionuclides.

ACKNOWLEDGEMENT

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AN APPROACH TO THE DERIVATION OF RADIONUCLIDE INTAKE LIMITS FOR MEMBERS OF THE PUBLIC*

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This paper describes a systematic approach to the development of radionuclide exposure limits for members of the public--an approach which starts with the occupational ALI (Annual Limit on Intake) and applies an adjustment factor, separately derived for each radionuclide. It seems expedient to utilize the extensive body of data and calculations relevant to occupational radionuclide limits, as collected in ICRP Publication 30 (1), rather than to attempt the derivation of public exposure limits from first principles. This also follows the past practice of deriving limits for external whole-body exposure of members of the public by applying a factor (usually 1/10) to the occupational limit for external exposure.

Lower exposure limits for members of the public, as compared to radiation workers, have been justified for such reasons as the following: (a) the radiation worker receives a specific benefit in the form of wages and other career satisfactions that is not received by the involuntarily exposed member of the public; (b) radiation workers constitute a population less susceptible to damaging effects, because of their age and general state of health; (c) radiation workers constitute a much smaller population, which is important from genetic considerations; (d) radiation workers are a more carefully monitored and medically treated population; (e) the timespan of occupational exposure is only a fraction of the lifespan--a minor fraction for the case of genetically significant exposure. Whatever numerical factor may be justified by these arguments, the external whole-body exposure limit for a member of the public may be defined by the following equation:

$$H_{L_p} = F_p \cdot H_{L_w}$$

where H_L is the annual dose equivalent limit for the member of the public (p) or the worker (w) and F_p is the *general population exposure reduction factor* justified by the cited arguments.

The considerations that lead to F_p for external exposure also apply to the derivation of intake limits for radionuclides. An additional factor is required, however, relating to specific interactions between radionuclide and individual, which differ for the worker and the member of the public. Radionuclide intake limits for members of the public may then be defined as:

$$(ALI)_p = F_p \cdot F_j \, (ALI)_w$$

*Work supported by U.S. Department of Energy Contract EY-76-C-06-1830.

where F_j is the *radionuclide-specific exposure adjustment factor*. The derivation of radionuclide intake limits for the member of the public then becomes a problem of developing appropriate values for F_j.

I am assuming that a single limit for a member of the public is desirable. One might set limits applicable to any number of special categories of exposed persons, but for practical purposes it would seem necessary to control to a single limit, chosen to provide an acceptable apportionment of dose among all these special categories. In practice, of course, this $(ALI)_p$ will not be employed directly, but will be used as a basis for derived limits, which will control the amount of radionuclide to which the member of the public has access.

Some of the more important factors, varying with age, state-of-health, etc., that might be involved in the development of values for F_j, include the following: (a) radiosensitivity factors altering risk; (b) morphological factors altering dose; (c) metabolic factors altering retention, distribution, and absorption from gastrointestinal tract or lung; (d) environmental factors altering biological availability; and (e) a physical half-life factor that might alter all of the above. I would like to illustrate this approach by applying it to the derivation of exposure limits for plutonium.

TABLE 1. Derivation of Member-of-the-Public Radionuclide-Specific Exposure Adjustment Factor for ^{239}Pu Ingestion

Parameter	Factor for: Infancy	Lifespan
Organ size	0.1	1
Food consumption	10	1
Food selection	5	1
Enhanced gastrointestinal absorption	0.001	0.7
Distribution and retention in: Bone (2x for infant), Liver (0.5x for infant), G.I. tract (100x for infant)	0.6	1
Environmental availability	1	0.2
Fraction of committed dose received	40	1
Overall value of F_j	0.12	0.14

Table 1 summarizes adjustments required for the ALI for ingestion, considered on a lifespan basis. Also shown are the adjustments required if only the first year of life is considered, as a check on the adequacy of protection during this critical period. There is no unusual factor of radiosensitivity relating to plutonium; any general effect of age on radiosensitivity is presumably included in the general population exposure reduction factor, F_p. Because of the smaller size of infant organs, the same level of intake would result in a higher dose rate in the infant than in the adult. For purposes of this illustration, a factor of 0.1 is assumed to equalize the infant and adult dose. Over a lifespan, this factor is insignificant. Food consumption by the infant is smaller than that assumed in deriving the ALI_w, by an assumed factor of 10. Because the infant's food is likely to be of lower than average plutonium content, e.g., milk and canned foods rather than leafy and root vegetables and seafoods, a factor of 5 is introduced for food selection.

A major factor of 0.001 is required to correct for the enhanced gastrointestinal absorption of plutonium by the infant, as suggested by studies in miniature swine (2). Applying this factor of 0.001 to 1/50 of the lifespan and correcting for the lower rate of intake during the period of infancy, the overall lifespan effect requires adjustment by only a factor of 0.7.

From animal data it appears that the very young deposit more plutonium in bone (by a factor of 2 we will assume) and less plutonium in liver (0.5 assumed) (3), and retain plutonium for a much longer time in the gastrointestinal tract (a probably overly conservative factor of 100 assumed) (3). When converted to weighted dose equivalent according to ICRP practices (1), these altered distribution and retention parameters result in about a 60% increased effective dose equivalent, adjusted for by the factor of 0.6. A 5-fold enhanced environmental availability of plutonium is assumed throughout the lifespan; this factor is not applied to the infant because it seems unreasonable to add this to the already conservative assumption of a 1000-fold increased absorption from the infant gut.

Because the ALI limits dose commitment rather than current dose, and since only about 1/40 of the committed dose is actually delivered during the first year of exposure, adjustment by a factor of 40 is required if one wishes to evaluate only the period of infancy.

Multiplied together, these factors lead to a value for F_j of 0.12 for infancy and 0.14 for lifespan. A factor of 0.1 should therefore afford protection for the first year of life as well as for the adult. If a general population dose reduction factor of 0.1 is assumed, we have a final ALI for a member of the public, which is one-hundredth of the occupational ALI for ingestion.

TABLE 2. Derivation of Member-of-the-Public Radionuclide-Specific Exposure Adjustment Factor for ^{238}Pu Inhalation

Parameter	Factor for: Infancy	Lifespan
Organ size	0.1	1
Volume inhaled	10	1
Fraction deposited and/or retained	2	1
Enhanced gastrointestinal and/or pulmonary absorption	0.2	1
Distribution and retention in: Bone (2x for infant), Liver (0.5x for infant), G.I. tract (100x for infant)	0.6	1
Fraction of committed dose received	40 (W)* 4 (Y)*	1
Overall value of F_j	10 (W)* 1 (Y)*	1

* Compound solubility class (T)

Table 2 summarizes a similar derivation for inhaled plutonium, differentiating, where necessary, between Class W and Y compounds (T). Again, there is no radiosensitivity adjustment. It is assumed that smaller lung size is exactly compensated by a reduced volume inhaled. Based on limited experimental animal data (4,5), alveolar deposition

and/or retention is assumed to be lower by a factor of 2 in the infant. Enhanced absorption from the gastrointestinal tract during infancy is a less significant factor for inhaled plutonium, since a substantial fraction is directly absorbed from the lung. The situation is also complicated by the lack of data on possibly enhanced infant absorption from the lung. These interactive factors were lumped and conservative adjustment factors assumed. Distribution parameters for the infant are the same as those assumed for the ingestion case and result in a similar adjustment. There is no correction for environmental availability, since this is primarily a food-chain phenomenon.

The correction for the fraction of committed dose received during the first year is about a factor of 40 for Class W compounds. It is only about a factor of 4 for Class Y compounds, because something approaching 25% of the weighted committed dose equivalent will be delivered to the lung during the first year following exposure.

Multiplied together, these factors lead to F_j values of 1 for lifespan exposure and for Class Y infant exposure, indicating that no further adjustment is required beyond that provided by the general population dose reduction factor, F_p. For Class W compounds the infant is relatively overprotected by limits based on lifespan exposure.

Let me emphasize that my purpose in this presentation has been to illustrate an approach to systematic consideration of radionuclide exposure limits for members of the public, and not to suggest specific limits for plutonium. A more definitive analysis might well lead to different numbers, but I hope the exercise has shown how such analyses could be conducted for all radionuclides.

As a member of ICRP Committee 2 on Secondary Limits, and of the U.S. NCRP Committee 57 on Internal Emitter Standards, I have benefited from much discussion in these committees on the subject matter of this paper. Views expressed in this paper should not, however, be interpreted as reflecting official ICRP or NCRP opinions.

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ARE WE AT RISK FROM LOW LEVEL RADIATION - DNA REPAIR CAPACITY AS A PROBE OF POTENTIAL DAMAGE AND RECOVERY

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Can the question of risk from low level radiation be answered by an experimental molecular approach? Can we assess accurately biological damage caused by low level radiation? Is there a threshold ? Is there a way to predict future genetic or late somatic expression of damage? What are the processes that occur in a cell and lead, following assaults by radiation or chemicals, towards mutagenicity and possibly carcinogenicity? What is the main reason for differences in radiosensitivity between cells which are otherwise equal?
This last question was answered in the case of bacteria first, when differences in survival from ultraviolet radiation was shown to be dependent on the existence of enzymatic repair systems which are lacking in radiation sensitive mutants (1, 2, 3). This discovery, soon followed by finding the existence of similar enzymatic repair mechanisms for mammalian cells and for assaults also by ionizing radiations as well as chemicals, opened new prospects for approaching by similar ways the problems of risk from radiation and the raging debate on the question of threshold.

Biological dosimetry and risk estimation may be approached from two different angles: one is to look at damage already inflicted by radiation, and expressed in a biologically important molecule, such as the chromosome, at such low doses that the damage is not apparent in the normal functions of the cell. The other is more directly related to risk evaluation, that is to look for a cellular system which will indicate the potential risk of a radiation assault. Cytological techniques have been developed to enable observing chromosome aberrations in lymphocytes, the most sensitive human cell, and determine in this way the dose of radiation. Aberrations are however the end product of a series of events occuring in the irradiated cell, furthermore they can be easily and accurately seen only following a dose of about 20 rads or more. The biochemical systems responsible for repair of damage to DNA are controlling the ultimate fate of radiation damage, be it specific products (4) or strand breaks (5), and thus are determining both the number of aberrations as well as the radiosensitivity of the cell. Indeed, excision repair and/or postreplication repair function in most normal human cells, and can repair most types of damage to DNA. An impaired activity of any of the enzymes of repair results in lack or reduced capacity of repair. This may result in extreme sensitivity of such cells to radiation, and such a situation has been found to exist in humans carrying genetic auto-immune diseases. Cells from patients with the disease Xeroderma pigmentosum (XP) show extreme sensitivity to UV light, and patients with the disease

Ataxia telangiectasia (AT) show extreme sensitivity to X-rays. These cells may be considered as the human mutants, identical in response to the bacterial mutants which show radiation sensitivity, in comparison to their wild type strains. Impairment in excision repair of UV-irradiated DNA is clearly implicated in XP, and similar impairment is suspected in Ataxia (6,7) and in several other diseases (although the existence of variants with normal DNA excision repair activity sheds doubt whether this is the only factor involved in the appearance of the diseaee),such as Fanconi's anemia, chronic lymphocytic leukemia and more. An interesting feature of these diseases is characterized in their "chromosome breakage syndrome" and their predispoeition to cancer. Several recent reports indicate also an impairment in mitogen-induced transformation of lymphocytes of patients with these diseases. The possibility of a link between chromosome aberrations lymphocyte transformation and DNA repair was recognised by us (8) and led to a decision to develop a biochemical method, based on measurement of DNA repair capability, which would enable predetermination of radiation sensitivity, and a more important feature - an indication of inherent sensitivity which might be expressed ultimately in the future - when the cell or organism might be faced with a situation in which its repair ability will have to function to its full capacity.

EXPERIMENTAL APPROACH AND RESULTS

The aims and steps of these studies have been previously formulated (9) and described (10). We have already shown that induced transformation of human lymphocytes is somewhat impaired following acute radiation dose, in cases where the person from whom the lymphocytes have been drawn was chronically exposed previously to low level beta (tritium) radiation (8). Induced transformation is, however,a black box and not a natural event in the life cycle of the lymphocyte. A clear and exact biochemical reaction was needed and none is better than the study of DNA repair itself. DNA repair may be followed by employing the technique of repair synthesis, whereby the amount of incorporation of labelled thymidine into cellular DNA is measured under conditions which do not permit the normal semiconservative DNA synthesis to occur. The commonly used method of treatment with hydroxyurea results in still too high a background and is subject to increasing criticism as affecting repair. A novel new method was therefore developed in our laboratory in which the cells are treated with trioxalen (trimethylpsoralen, TMP) and near UV light (NUV), bringing about an almost complete cessation of semiconservative DNA synthesis, 99.5 to 99.8% inhibition (Heimer, Kol and Riklis, in preparation). This treatment was previously successfully used by us to study gene expression and the control and regulation of inducible enzymes (11). Now it enabled the accurate measurement of incorporation of labelled thymidine (^{3}H-TdR) into DNA, following assaults by radiation or chemicals, indicating that repair synthesis is occuring.

Details of the experimental procedures are described elsewhere (11, and, Heimer, Kol and Riklis, in preparation; Kol, Heimer and Riklis, in preparation for detailed results). Briefly, it involves irradiation of cells with near UV light for 3-5 minutes in the presence of 5×10^{-6}M TMP, incubation for 3 hrs, assault by far UV, gamma radiation or a chemical (MMS), incubation with ^{3}H-TdR for 90 min., washing, TCA precipitation and counting in liquid scintillation counter. The method has been found suitable for human fibroblasts, human breast cancer cells, Chinese hamster cells and human lymphocytes. The number of counts incorporated is many thousands above background, indicating repair synthesis following the acute assault. A linear increase of counts is observed up to a certain radiation dose, this being designated as the repair capacity of the cell, given by absolute numbers when the number of counts incorporated while repairing a high dose damage is divided by the counts incorporated following the lowest acute dose or no irradiation-control (Fig.1). The counts were taken up into double stranded DNA, as shown by the S1-nuclease method (13) indicating true repair replication. Thus, the 'repair capacity" is a true measure of the capability of the cells to repair damage.

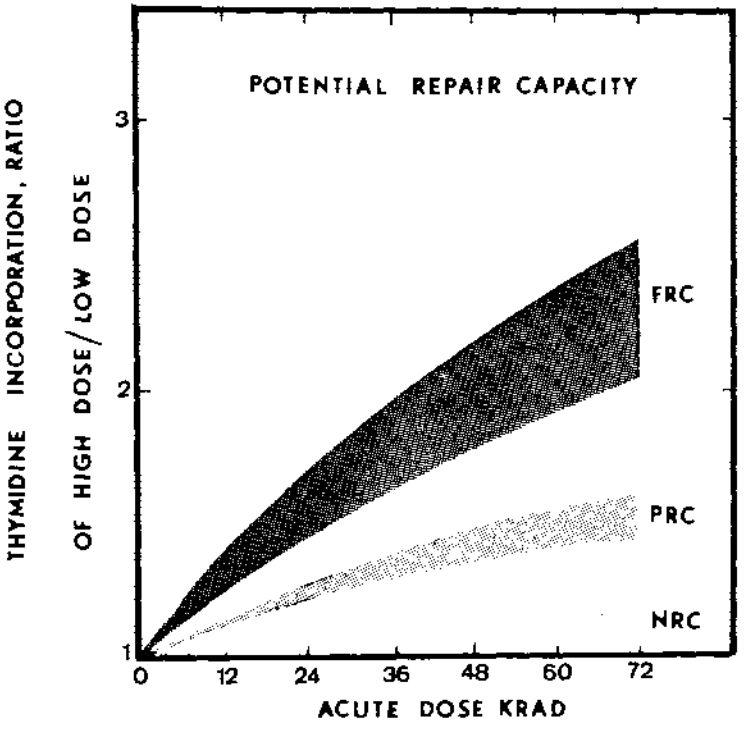

Figure 1. Potential Repair Capacity. Number of counts incorporated per 10^6 cells following each dose is divided by cpm/10^6 of control cells. FRC=full repair capacity; PRC=partial repair capacity; NRC=no repair capacity (12). A hypothetical graph.

A most significant result was obtained in showing that chronically irradiated Chinese hamster cells, with 20 rads per day for 3 months showed the same repair capacity as controls. Their survival was also not affected although their growth rate was slower.

DISCUSSION

Repair synthesis as measured by thymidine incorporation does not indicate the type of repair which has taken place, the exact nature of it is yet to be determined. A more important question to be asked is whether this repair is error-free or error-prone, namely, can we determine whether the cell is fully functional and not mutated. This is clearly a difficult problem, but attempts are being made to answer it by following the performance of an inducible enzyme, Ornithine decarboxylase (14), which if induced only after a radiation assault and if it functions to full capacity, indicates the wholesomeness of the DNA template.

Since the psoralen + NUV method of repair capacity measurement is applicable for lymphocytes, it can easily be used for a world-wide interlaboratory comparative study, in which the range of 'repair capacity' of normal healthy humans will be established, and average figures will be determined for fully repairing cells (FRC), partially repairing cells (PRC) and non-repairing cells (NRC). A threshold for radiation damage will be between the PRC and the NRC regions. Individuals with no repair capacity (NRC), are 'at risk' from any level of radiation above background, while the general public, showing normal repair capacity, may be considered safe from the effects of low level radiations.

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QUATRE OBSERVATIONS DE TRAITEMENT CHIRURGICAL DE BLESSURES CONTAMINÉES PAR DES RADIONUCLÉIDES.

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INTRODUCTION

A travers quatre cas concrets survenus dans des Etablissements Nucléaires Industriels différents, il nous est apparu utile d'évoquer les problèmes médicaux liés au traitement de blessures contaminées.

Les enseignements tirés de ces cas permettent de dégager les points essentiels concernant l'organisation de la sécurité, les moyens thérapeutiques et les méthodes de mesure.

I - DESCRIPTION DES CAS OBSERVES

Observation n° 1 -

Au cours d'un nettoyage de l'intérieur d'une boîte à gants fortement contaminée en 239 Pu, un agent s'est légèrement coupé l'auriculaire droit à travers les gants de protection.

Après application immédiate d'une solution de DTPA, le blessé est dirigé vers le Service Médical de l'établissement. Il existe une plaie linéaire de 3 mm de longueur et de 3 à 5 mm de profondeur. Une mesure avec la sonde α donne 10 chocs/seconde. On pratique une première excision superficielle et on fait saigner la plaie sous DTPA. Les compresses sont comptées et indiquent une contamination profonde. On décide alors de pratiquer une injection intra-veineuse de DTPA (1 gr), suivie d'un comptage à la sonde X. Le résultat du comptage permet d'évaluer la quantité de Pu incluse à 200 nCi. On décide alors de faire appel au chirurgien pour pratiquer une excision. Les comptages X effectués extemporanément pendant l'intervention permettent de déterminer l'importance des excisions à réaliser : quatre excisions successives sont ainsi effectuées, jusqu'à une activité résiduelle acceptable, évaluée à 14 nCi.

Compte tenu du traitement par le DTPA, l'étude de l'excrétion urinaire et fécale de la première semaine conduit aux conclusions suivantes :

- la quantité de Pu qui a diffusé de la plaie vers le sang pendant la période comprise entre l'incident et l'intervention chirurgicale est restée très faible.
- dans l'hypothèse où tout le Pu résiduel finisse par diffuser dans les liquides extracellulaires, la charge corporelle qui serait acquise après l'élimination complète du Pu de la blessure sera au maximum celle restant dans le doigt après l'excision des tissus, soit 14 nCi dont environ 6 nCi dans l'os.

Observation n° 2 - JL.M.

Un ouvrier travaillant en vêtement pressurisé au démantèlement d'une boîte à gants se blesse sur une arête de matière plastique à 10 h. Il s'agit d'une plaie linéaire de 6 mm, profonde de 1 mm, de l'éminence thénar gauche.

Conformément aux consignes en usage, le blessé, après déshabillage et isolement de la blessure par l'agent de radioprotection de l'atelier est conduit à l'infirmier du bâtiment. Ce dernier constatant une contamination (sonde α) procède à une décontamination. A la fin du traitement le comptage superficiel est négatif, mais le blessé est adressé au service médical de l'établissement pour mesure de contrôle, faisant appel à une sonde X. Cet examen met en évidence au niveau de la blessure une contamination profonde (Pu 239 : 3,86 nCi). Un traitement chélatant (DTPA local et général) est immédiatement institué. La décision est prise d'exciser la plaie. Le comptage effectué en fin d'intervention montre que 1,7 nCi (44 % de l'activité initiale) a été enlevé. (2 h 30 après l'accident).

Compte tenu de la nature chimique du contaminant (Nitrate de Pu 239) et du traitement déjà institué, il est décidé de ne pas augmenter l'excision. Un traitement chélatant itératif est mis en oeuvre et la victime placée sous surveillance radiotoxicologique.

En moins de deux mois, les mesures urinaires deviennent négatives. La partie transférable du radionucléide peut être considérée comme éliminée de la blessure.

Observation n° 3 - M.C. J. Pierre

Piqûre des 2° phalanges des index droit et gauche par deux brins d'un cable métallique effiloché contaminé par des produits de fission.

Contrôle : 150 chocs/seconde en β sur chaque doigt. On pratique sans succès une décontamination cutanée. Un examen au microscope ne permet pas de repérer les points de pénétration des brins de cable . On gratte superficiellement l'épiderme sans obtenir de modification de l'activité.

L'examen anthroporadiamétrique du corps entier (mains exclues maintenues derrière la tête) met en évidence 5,8 nCi de 106 Ru. Une mesure simultanée des 2 doigts donne 19 nCi.

24 heures après : Contrôle des doigts à la sonde β : même résultat que la veille. L'existence ce jour d'une légère réaction inflammatoire localisée, permet de visualiser sous microscope les points de pénétration des brins de cable.

Un grattage et une excision très superficielle amènent une diminution de l'activité : 120 chocs/seconde à gauche, 60 chocs/seconde à droite.

48 heures après : Pas de modification par rapport à la veille. Etant donné le radionucléide en cause et l'activité mesurée, la contamination interne de cet agent pouvait être considérée comme négligeable et l'irradiation localisée en $\beta\gamma$ d'un niveau très bas.

Cependant, la persistance de cette contamination localisée rendant difficile les contrôles systématiques lors du travail en zone contrôlée, la décision d'excision cutanée a été prise.

Dans un premier temps, des comptages successifs à travers des écrans en aluminium percés de trous de diamètres décroissants (de 20 mm à 2 mm) ont permis de localiser très précisément les zones à exciser.

L'intervention (six jours après l'incident) permet :
- l'ablation de fragments dermoépidermiques en quartiers d'oranges dans les zones repérées,
- l'examen spectrométrique γ a montré que l'activité mesurée antérieurement "in vivo" au niveau des doigts se retrouvait au niveau des fragments excisés :
  5 nCi provenant du doigt droit,
  9,9 nCi provenant du doigt gauche.

Lors de la reprise du travail, l'anthroporadiamétrie de contrôle montrait une courbe normale. Les prélèvements d'urines et de selles effectués après l'incident avaient par ailleurs été négatifs.

Observation n° 4 - J.R.

J.R. 45 ans, ouvrier chimiste, en changeant en boîte à gants un élément de canalisation ayant contenu du 239 Pu irradié se blesse au pouce droit à 11 h.

Il existe une plaie de 1,5 cm de longueur au niveau de la face palmaire, 2ème phalange, du pouce droit, avec une contamination superficielle importante décelée à la sonde α.

Après examen par l'infirmier du bâtiment, la victime est transférée au service médical. Après un essai de décontamination suivi de lavage de plaie au DTPA et injection de DTPA on décide de pratiquer une excision, compte tenu de la première évaluation de la contamination sous jacente (239 Pu et 241 A ; 18 nCi au niveau de la plaie). Après l'excision, terminée 4 h 30 après l'incident, le comptage de la région contaminée est négatif, et le comptage du lambeau excisé fait apparaître 36 nCi de 239 Pu et 31 nCi de 241 Am.

Les examens radiotoxicologiques des excretas de surveillance sont toujours restés négatifs.

Cette observation a fait l'objet du film qui va être projeté.

II - ENSEIGNEMENTS TIRES DE CES OBSERVATIONS

II-1 Organisation de la sécurité

II-1.1 Personnel et installations spécialisés

La présence d'un agent de radioprotection dans l'atelier et éventuellement d'une infirmerie de bâtiment pour les premières mesures et les premiers soins est bénéfique.

En outre, le service médical de l'Etablissement doit posséder les installations nécessaires pour pouvoir effectuer des interventions chirurgicales avec contrôle pré, per et post opératoires des lésions contaminées. Ces interventions nécessitent la disponibilité d'une équipe multidisciplinaire associant chirurgiens et radiotoxicologues.

II-1.2 Consignes sur les lieux de travail

Dans tous les lieux de travail où sont manipulés des radionucléides doivent exister des consignes indiquant la conduite à suivre à différents niveaux :

- appel immédiat de l'agent de radioprotection par l'agent blessé ou par ses compagnons de travail, quelle que soit l'importance de la blessure,
- lavage immédiat de la plaie à l'eau ou par des chélateurs si nécessaire (terres rares, transuraniens),
- mesure rapide α ou β de la contamination superficielle et évacuation vers le Service Médical.

II-1.3 Formation du personnel

Les personnels d'exécution ainsi que ceux des services de sécurité doivent être informés sur la nature des risques et sur la conduite à tenir. Des exercices périodiques permettent de s'assurer que les consignes sont correctement comprises et exécutées.

II-2 Moyens thérapeutiques

Lorsqu'existe une présomption de contamination par des terres rares ou des transuraniens il est indispensable de procéder rapidement à une injection de chélateur (par exemple DTPA). Ce traitement général permet d'éviter la fixation dans les organes du radionucléide qui diffuse dans le sang et permet de procéder sans précipitation aux mesures et interventions nécessaires.

L'exploration des plaies, la recherche éventuelle de petits frangments radioactifs sont facilitées par l'utilisation de microscope et de micro instruments chirurgicaux (micro instrumentation type chirurgie de l'oreille).

Toute intervention chirurgicale nécessitée par une contamination doit être aussi limitée que possible. Il serait aberrant d'aboutir à des interventions mutilantes pour éliminer complètement des contaminations résiduelles peu importantes.

II-3 Méthodes de mesure

II-3.1 Mesure de l'activité dans la plaie

La mesure de l'activité β et surtout α est insuffisante elle doit être doublée d'une mesure systématique des rayonnements γ ou X associés permettant de déceler la contamination profonde. Si le radionucléide contaminant est inconnu il est nécessaire de faire appel à des mesures spectrométriques.

II-3.2 Mesure de la contamination interne

Il est indispensable de surveiller par des examens radiotoxicologiques des excrétas et/ou anthroporadiamétriques l'efficacité de la thérapeutique et la contamination résiduelle.

RADIOLOGICAL CONSEQUENCES OF THE THREE MILE ISLAND ACCIDENT

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SITE AND ENVIRONS

The Three Mile Island Nuclear Station (TMI) is located on an island in the Susquehanna River approximately 14 km southeast of Harrisburg, Pennsylvania. The station is operated by a private utility, the Metropolitan Edison Company, and consists of two reactors, Unit 1, a 2535 megawatt (thermal) pressurized water reactor (PWR), and Unit 2, a 2772 megawatt (thermal) PWR. Unit 1 went into commercial operation on September 2, 1974 and Unit 2 went into commercial operation on December 30, 1978, approximately 3 months prior to the accident.

Three Mile Island is one of a number of islands in the Susquehanna River. It is located approximately 275 m from the east bank of the river and approximately 2 km from the west bank. Several private residences are located along the east shore within 0.8-1.2 km of the reactor buildings. Approximately 200 summer cottages are located on the nearby islands. Goldsboro, a community of approximately 900 people, is situated approximately 1.9 km west of the site and Middletown (approximately 10,000 people) is located 4.0 km to the north. Major population centers in the area are Harrisburg (~70,000 people) which is 14 km NW and York (~50,000 people) which is 21 km South. There are approximately 2,000,000 people residing within 80 km of the TMI site.

THE ACCIDENT

Three Mile Island Unit 2 was operating at 97 percent (916 MWe) of its licensed power level on the morning of March 28, 1979. At 0400 a series of events resulted in a substantial loss of primary coolant and the reactor's core being partially uncovered for several periods during the next 16 hours. High cladding temperatures resulted in metal-water reactions between the zirconium fuel cladding and the water (or steam). Oxidation and failure of the cladding resulted, releasing substantial quantities of fission products into the coolant and production of hydrogen. A primary coolant sample collected on March 29 shows the degree of the fission product contamination (Table I).

RADIOACTIVE MATERIALS RELEASED: PATHWAY AND QUANTITY

The fission products released to the coolant were transported to the auxiliary building in the primary coolant through the normal coolant purification system. The noble gas radionuclides and a fraction of the radioiodines were stripped into the gas phase and leaked into the supporting equipment buildings. Ventilation air transported these gases to the auxiliary building stack (10 feet below the top of the containment building) through high efficiency particulate filters (HEPA) and a charcoal absorber.

Although substantial noble gas activity was released, estimates range from 2.4-14 MCi, the water-to-air partition process and the filters reduced radioiodine release, estimated to be 15 Ci. The distribution of the noble gases released is shown in Table II, along with the core inventory of these radionuclides at the time of the accident.

TABLE I. The Major Radionuclides in a Sample of Reactor Coolant Taken on March 29, 1979.*

Nuclide	Half Life	Coolant Concentration uCi/cc
Iodine-131	8 d	1.3×10^4
Iodine-133	20.8 h	4.6×10^4
Cesium-134	2 y	6.3×10^1
Cesium-136	13 d	1.8×10^2
Cesium-137	30 y	2.8×10^2
Barium-140	12.8 d	21.0×10^1
Strontium-89/90	50 d/29y	5.3

*Reactor coolant sample taken at approximately 1700 on March 29. Sample was analyzed by the Bettis Atomic Power Laboratory, Pittsburgh, Pa.

TABLE II

Radionuclides Released to the Environment as a Result of TMI-2 Accident.

RADIONUCLIDE	HALF-LIFE	QUANTITY IN CORE AT TIME OF SHUTDOWN (Curies)	QUANTITY RELEASED ESTIMATED (Curies)	ESTIMATED FRACTION OF TOTAL RELEASED
Xe-[illegible]	2.E hours	6.92×10^7	3.75×10^5	0.15
Xe-133	5.2 days	1.42×10^8	1.58×10^6	0.63
Xe-133m	2.2 days	2.11×10^7	2.25×10^5	0.09
Xe-135	9.1	3.31×10^7	3.0×10^5	0.12
Xe-135m	15.3 min.	2.60×10^7	2.5×10^4	0.01
I-131	8.0 days	6.55×10^7	35	*

* On an estimated fractional basis of total nuclides released, iodine-131 was very small.

Almost all (99%) of the noble gas emissions occurred in the period from March 28 until April 1 and 70% of these releases occurred within the first 36 hours. Radioiodine releases persisted until the end of April due to evaporation of liquids in the auxiliary building and degeneration of the charcoal filter performance.

Releases of fission products in liquid effluents were very small and consisted primarily of radioiodine and cesium-137. The total activity released in liquid effluents during the first three months following the accident was 0.23 Ci of iodine-131 and 0.24 Ci of other radionuclides.

RADIOLOGICAL MONITORING RESULTS

The U.S. Nuclear Regulatory Commission requires all reactor licensees in the United States to have an environmental monitoring program. In addition, each reactor is to have an emergency plan. Once it was realized that significant radiological releases might occur, the licensee dispatched teams to determine radiation levels offsite, particularly in the anticipated plume direction. Several State and Federal agencies also responded to the emergency and established environmental monitoring programs, sampling air, milk, water, vegetation, foodstuff and deploying additional thermoluminescent dosimeters (TLD's). The U.S. Department of Energy (DOE) used helicopters for tracking and measuring the activity in the plume. Metropolitan Edison used helicopters for monitoring on the site. During the 3 months following the accident, several thousand sample analyses were performed by the Commonwealth of Pennsylvania and several U.S. Federal agencies including the Department of Health, Education and Welfare, Bureau of Radiological Health and the Environmental Protection Agency.

Environmental

As a result of increasing in-plant radiation levels, beginning around 0700 on March 28, monitoring teams were dispatched to make radiation measurements outside the plant both onsite and offsite. Initial measurements made onsite starting at 0748 and offsite starting at 0832 were less than the minimal detectable level of the instruments (1 mR/hr). Radiation levels first began to increase at 1020 on March 28 when onsite monitoring teams detected exposure levels of 3 mR/hr. The instruments used for the offsite survey were Geiger-Muller detectors and ion chamber (RO-2) survey type instruments. Many of the reported readings were open window measurements and reported as β,γ-mR/hr, which is an undefined exposure rate. Where "β,γ" readings are known, they are so indicated. The instruments were not calibrated against a beta source, nor were they calibrated for an immersion situation. What the influence is on the total reading of the beta component is not known. These levels generally increased over the next 12 to 13 hours. Peak onsite radiation exposure rates of 300-365 (β,γ) mR/hr were reached between 2130 and 2330. Offsite radiation exposure rates were generally very low (maximum of 3 mR/hr). A reading of 50 mR/hr measured along the east river bank at 1548 was the highest reported offsite exposure rate. Noble gas emissions continued to be high until late in the morning of March 29. A reading of 30 mR/hr was recorded in Goldsboro (1.9 km WSW) at 0600 on March 29.

The maximum onsite dose rate on March 29 was 150 mR/hr (β,γ) at 0532. During the remainder of March 29, onsite levels were generally less than 10 mR/hr and offsite levels less than 1 mR/hr and did not exceed 2 mR/hr. Wind direction throughout the night of March 28-29 was generally in a northwesterly direction (toward Harrisburg). During the afternoon of March 29, a helicopter above the stack measured 3 R/hr (β,γ), 400 mR/hr gamma.

A second period of noble gas emissions occurred on March 30-March 31. This release resulted from intentional venting of the waste tanks in the auxiliary building required to reduce excessive pressure buildup in the tanks. Onsite exposure rates associated with this release reached a peak of 110 mR/hr. The highest offsite levels were 5-15 mR/hr at a point approximately 1.6 km to the south. However, a helicopter reading taken ~40 meters above the stack was 1.2 R/hr (β,γ) at 0800 hours. The reading could not be repeated, indicating a probable puff release.

Radioiodine Analyses

Offsite radioiodine was detected in analyses of milk samples collected for the first seven days following the accident with 68 positive iodine-131 results out of 264 samples collected. The concentrations ranged from 1 to 41 pCi/l (the 41 pCi/l was in a sample of goat's milk, which was not used for human consumption). In the subsequent 2 weeks, only 8 out of 80 samples taken by the U.S. Food and Drug Administration yielded positive results. Concentrations ranged from 15-36 pCi/l.

Initial measurements of airborne radioiodine concentrations made using portable air sampling equipment having charcoal adsorption cartridges were reported as 10^{-9}-10^{-10} μCi/ml at Goldsboro at 0900 and 0940 on March 28. However, laboratory gamma spectrometric analysis of the second cartridge by the Pennsylvania Bureau of Radiation Protection showed that this activity was primarily due to xenon-133 and xenon-135 and that actual radioiodine concentrations were less than 10^{-11} μCi/ml. The highest reported offsite radioiodine concentrations and measurable deposition occurred in mid-April in conjunction with replacement of the effluent filters in the auxiliary building, onsite 4 x 10^{-10} μCi/ml, offsite 1 x 10^{-10} μCi/ml.

In Plant

The highest radiation levels encountered by Met Ed personnel were in the auxiliary and fuel handling buildings. Radiation levels in excess of 1000 R/hr were measured during the first days of the accident at entrances to the cubicles containing tanks of primary coolant. General area radiation levels in these buildings ranged from 5 R/hr to 100 R/hr. Radiation levels in the reactor control room and other areas were generally low, less than 0.5 mR/hr. Due to the airborne activity (noble gases) in the Health Physics Control Station, counting and gamma spectrometry facilities had to be evacuated.

Population Exposure

The ground surveys that were performed and the analyses of local foods indicated that there was no measurable deposition of radioactive materials released from TMI. Of primary concern, however, was the need to assess the dose to the population and evaluation of the potential long-term consequences.

As part of Metropolitan Edison's environmental monitoring program, 20 TLD stations both on and offsite were located around the site at the time of the accident at distances up to 22 km. In addition, ten stations had a quality control TLD of a different type. Commencing on March 31, the Nuclear Regulatory Commission (NRC) placed an additional 37 TLD's around the TMI site. These were analyzed daily for a period of one week and at longer intervals thereafter.

The first evaluation of the population dose was performed by Battist, et al[2], using the TLD's in place at the time of the accident and those subsequently placed by the NRC. This was accomplished by an interpolation equivalent to plotting the measured doses for each sector on logarithmic coordinate graph paper and joining the measured values by straight line segments. The intersection of each line segment with a standard distance for the grid was taken as the dose at that distance. In instances where the net dose calculated for a location was not greater than zero, this method could not be used. In such cases, linear interpolation was used to estimate the doses at standard distances.

Doses at distances beyond the outermost dosimeter or within the innermost dosimeter were estimated by extrapolation using the assumption that the dispersion in a sector is proportional to distance to the (-1.5) power. A DOE analysis concludes that their airborne measurements and the TLD data suggest a more rapid decrease of exposure with distance, more consistent with an exponential function or a power function with an exponent of (-2). The (-1.5) power assumption is therefore conservative, yielding a higher collective dose.

Doses for the standard distances in sectors in which no measurements were made were estimated by interpolating linearly between the dose values of the adjacent sectors for which measured data were available.

The mean dose within each sector segment was estimated by weighting the dose, H(r), by the area within the sector

$$\bar{H} = \frac{\int_{r_1}^{r_2} H(r)\,r\,dr}{\int_{r_1}^{r_2} r\,dr}$$

where $\bar{H}$ is the mean dose, H(r) is the dose as a function of distance, r, and r_1 and r_2 are the inner and outer radii of the sector segment, respectively.

The collective dose for each sector segment is the product of the corresponding mean dose and the population in that sector. The sum of the collective doses for all sector segments and periods is the total collective dose for the entire assessment area for the total period under consideration.

Utilizing the available TLD data, a range of collective dose equivalent estimates were determined. These values ranged from 1600 to 5300 person-rem, with the most probable value being 3300 person-rem. This range resulted from different sets of dosimeter data used in individual determinations.

The highest value, 5300 person-rem, was the result of including all of the NRC and Metropolitan Edison dosimeters. However, the first day's set of NRC TLD data contained several inconsistencies. Later analyses[3] indicated that these dosimeters were most likely exposed prior to deployment and that no controls were included to evaluate these effects. Use of the Metropolitan Edison dosimeters, including the quality control badges, resulted in a collective dose equivalent of 3300 person-rem. The other two values, 2800 and 1600 person-rem, were obtained by using the TLD data within 10 km of the plant; first including all of the TLD data, and later by excluding the NRC TLD data.

A second evaluation of the TLD data was performed by Auxier, et. al.[3] Their estimate of the collective dose equivalent was 2800 person-rem. Taking into account the shelter factor for the low photon energy of xenon-133 reduced the collective dose equivalent estimate to 2000 person-rem.

The agreement between these independent analyses is quite good, and the collective dose equivalent is in the range of 1600-3300 person-rem. Attempts were also made to determine the collective dose equivalent using meteorological dispersion calculations.

In-Plant Exposures

Although high radiation fields existed in the auxiliary building, and several entries were made, only three individuals exceeded NRC's quarterly whole-body exposure limits of 3 Rem. The exposures were 4.1, 3.6, and 4.2 Rem, respectively. In total, during the seven-month period following the accident, only seven individuals received doses in excess of 3 Rem. The total collective occupational exposure through September 30 was approximately 1200 person-rem.

In August 1979 several workers were contaminated by beta activity when working in contaminated areas. Extremity exposures were high, approximately 40-50 Rem, due to residual contamination. No whole-body exposures in excess of regulatory limits were reported.

DISCUSSION

Although the accident at TMI Unit 2 was the most severe reactor accident to date, the release of several megacuries of radioactive noble gases resulted in a relatively small population exposure estimated to be in the range of 1600 to 3300 person-rems, as determined from TLD measurements. The sparseness of the data and the extrapolation of individual dosimeter results to assess the dose to the population in a large sector contribute to the uncertainty. However, the continual low offsite exposure readings, lack of residual ground activity and other dosimeters placed in the environs of the site by Federal agencies all tend to confirm that the population dose could not have been significantly different than that defined above. The maximum individual offsite dose was stated to be less than 100 mrem in the Ad Hoc group study,[2] and about 50 mrem by Auxier, et.al.[3]

In-plant personnel exposures have been maintained at reasonable levels. The fact that only three overexposures were recorded on the first two days of the accident is remarkable in view of the high radiation fields that existed. However, the cleanup operations could result in a significant collective worker dose unless significant health physics control is exercised.

The defense-in-depth concept under which nuclear plants are designed worked well in practice. Radiological releases were quite small in view of the magnitude of the fuel damage. The containment building, requirement of filtered pathways, and backup systems all functioned to minimize the potential radiological consequences. However, the accident indicated that better health physics instrumentation and personnel training is required to obtain more meaningful survey results and to control in-plant exposures.

HEALTH EFFECTS

As a result of the radiation exposure to the offsite population within 50 miles of the TMI site, the projected incidence of fatal cancer is less than 1; and fatal plus non-fatal cancers is less than 1.5, with zero not excluded. This is to be contrasted to the nearly 541,000 cancers (325,000 fatal and 216,000 non-fatal) expected in this population over its remaining lifetime, not related to the TMI accident.

The additional lifetime fatal cancer risk to the individual receiving the maximum probable dose offsite (less than 100 mrem) is about 1 in 100,000. The additional risk of fatal cancer to an individual receiving the average individual offsite dose (1.4 mrem) is about 1 in 5,000,000. The risks of non-fatal cancer induction are the same as those for fatal cancers.

The additional cancer risks due to internal irradiation and skin irradiation are very small compared to the above values and can be regarded as being included in the values presented above for whole-body gamma irradiation. Even if the cancer risks defined above were to be expressed, the resultant cancers would not be detectable among the population in the vicinity of TMI. (Note that zero additional incidence is not excluded.)

The whole-body external occupational exposure of 1,000 person-rem has potential total cancer risk of less than 0.5 (zero not excluded). The risk to the maximally occupationally exposed individual (4.1 rems) is about 1.2 in 1,000 for both fatal and non-fatal cancers.

The potential incidence of genetically related ill-health is considerably smaller than that of producing a fatal or non-fatal cancer. This risk is estimated to be about 0.002 cases per year, and about one case per million live births for all future human existence. This contrasts with an estimated 3,000 cases pwer year of genetically related ill health among the offsprings of the population in the vicinity of TMI based on present birth rate (28,000 births per year), and not related to the TMI 2 accident.[5]

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Protective Action Guides: Theory and Application
Lessons from the Three Mile Island Accident

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During the Three Mile Island Nuclear accident, the Food and Drug Administration (FDA), Department of Health, Education and Welfare (HEW), fulfilled its traditional role in insuring a safe food supply, and together with other elements of HEW, provided advice and assistance in the response to health concerns. This paper will present two aspects of FDA's role: 1) actions relative to the use and availability of potassium iodide (KI) as a thyroid-blocking agent; and 2) protective action guidance (PAG's) relative to human food and animal feed.

KI AS A THYROID BLOCKING-AGENT

By a notice in the Federal Register of December 15, 1978[1], entitled "Potassium Iodide as a Thyroid-Blocking Agent in a Radiation Emergency," the FDA did the following:

1) In the interest of public safety it requested submission of New Drug Applications (NDA's) for potasium iodide in oral form for use as a thyroid-blocking agent in a radiation emergency.
2) It announced the availability of labeling guidelines for potasium iodide for such use.

Many organic and inorganic drugs were considered for emergency use as thyroid-blocking agents. Potassium iodide was chosen over the other drugs because of: 1) the high degree of blocking achieved; 2) the rapidity of onset of action; 3) a long high duration of its blocking effect; and 4) its relative safety Although potassium iodide acts on the thyroid in several ways, its usefulness for this purpose ia primarily predicated on its ability to aaturate the iodide transport system, effectively abolishing entry of radioiodine. Complete blocking is achieved, that is, over 90 percent of radioiodine is blocked by the recommended oral administation of 130 milligrams of KI. This is equivalent to 100 milligrams of iodide. The recommended dose for infants under one year of age is 65 milligrams. Onset of block ing occurs within 30 minutes. Therefore, administration before or immediately after initial exposure yields the best results. However, substantial benefit can still be achieved if potassium iodide is given within three or four hours after exposure. The duration of time that a blocking agent is required is not likely to exceed 10 daya.

As with any drug, certain cautions have to be recognized. The number of reports of adverse reactions for the use of potassium iodide when it is given in greater doses than for blocking and over a longer period of time has been very low. The risk is judged to be low for the short-term use of potassium iodide in a radiation emergency.

The Federal Register notice offers no guidance as to when to use potassium iodide as a blocking agent. Other sources of guidance are available. The National Commission on Radiation Protection and Measurements Report No. 55[2] suggests that potassium iodide should be considered for blocking at a projected or anticipated thyroid dose from radioiodine of 10 - 30 rads to the thyroid.

The Three Mile Island episode graphically demonstrated the need for effective planning for the use of potassium iodide as a component of public health emergency reponse capability. The drug was not available for mass distribution in the proper dosage forms at the time of the Three Mile Island accident. FDA had not as yet received nor approved an NDA for potassium iodide as a thyroid-blocking agent. To meet the emergency, FDA arranged for the manufacture of a supply of potassium iodide before any decision to employ the drug had been made. Pennsylvania Officials had a plan for distribution of the supplies of KI and patient information. The Three Mile Island emergency was unique in that FDA supplied the drug. The Agency does not stockpile nor will it be a source of the drug in the event of other accident situations.

Among significant factors which influence the decision as to whether or not KI should be employed as a thyroid-blocking agent in a radiation accident are:

1. The efficacy of potassium iodide as a thyroid-blocking agent depends on the pathway of radiation exposure.
2. There is an absence of Federal guidelines for the use of potassium iodide as a thyroid-blocking agent.
3. The area and population potentially affected by the release of radioactivity are important to the decision as to when and how to use potassium iodide.
4. There are logistical problems of storage and distribution because of the need for prompt administration in the event of an accident.
5. Alternative protection actions must be considered instead of, or in concert with, the use of potassium iodide.

Discussion of these problem areas requires more space and time than are available here, and is the subject of a paper in preparation[3].

PAG's FOR HUMAN FOOD AND ANIMAL FEED

"Proposed" Protective Action Guidance for the food pathway also appears in the Federal Register of December 15, 1978[1]. These recommendations are for use by appropriate State and local agencies in response planning for radiation emergencies in the event of an accident resulting in the contamination of food or

animal feed by radioactive substances. Events which may result in radiation emergencies include, but are not limited to, accidents at nuclear facilities, transportation accidents, and fallout from nuclear devices. Although such incidents could lead to a general contamination of the environment, the FDA's recommendations are limited to the food pathway.

The proposed protective actions are intended for implementation within hours or days from the time an emergency is recognized, and the duration should not exceed a month or two. Other Federal agencies are responsible for guidance in the event of exposure to the population from pathways other than for food, and for action of a longer duration.

The FDA protection action guidance does not imply an acceptable radiation dose from food containing radioactivity during normal peacetime conditions. Rather, their purpose is to reduce or avoid further radiation dose to the population via the food chain in the event of a radiation accident. These recommendations are not a license to needlesssly permit environmental levels of radiation to rise.

Protective action guides or PAG's define the projected dose commitment to individuals in the general population that warrants protective actions following the release of radioactive material. Projected dose commitment, is defined as the dose commitment that would be received in the future by individuals in a population group from s contaminating event if no protective action were taken. In other words, the protective action guidance is based on anticipated or projected doses. The purpose of the PAG's is to provide guidance in order to prevent additional radioactive contamination from entering the human food chain and to reduce or avoid future radiation doses to the population after an accidental contaminating event.

Two protective action guidance levels have been proposed. They are:

1. The Preventive PAG is applicable to situations where protective actions causing minimal impact are justified. These protective actions prevent or reduce concentrations of radioactivity in food. The preventive PAG is 1.5 rem projected dose commitment to the thyroid, or 0.5 rem projected dose commitment to the whole body, bone marrow, or any other organ.
2. The Emergency PAG is applicable to incidents where protective actions of high impact are justified because of the greater projected health hazards. Levels at which food should be isolated from commerce are appropriate at the emergency PAG level. The emergency PAG is 15 rem projected dose commitment to the thyroid, or 5 rem projected dose commitment to the whole body, bone marrow, or any other organ.

A practical means of employing the PAG's is through the use of

derived response levels. Derived response levels refer to the activity of a specific radioactive substance per unit weight or volume of food, or animal feed, which corresponds to a particular numerical PAG limit previously mentioued. Response levels have been calculated based on recent metabolic and agricultural models. Specific derived response levels are given in the FDA recommendations for radioactive substances which are thought to be relatively abundant under emergency conditions, easily enter the food chain, and are taken up and retained by the human body. These radionuclides are iodine-131, cesium-137, strontium-90, and strontium-89. Derived response levels are given specifically for initial deposition on pasture, concentration in forage and in milk, and total intake. Variations in the basic model permit calculation of derived response levels for different food products and mixtures of radionuclides.

A large number of milk samples were collected following the Three Mile Island accident from farms and dairies in the vicinity of the accident site. Based on the peak concentration of iodine-131 detected in these samples, the dose to the thyroid of an infant drinking one liter of milk daily for the eutire time during which positive milk samples were found, would be 0.005 rem over the lifetime of the individual[4]. The maximum iodine-131 levels detected at Three Mile Island were thus 300 times smaller than the preventive action level.

Two prohlems which arose during the Three Mile Island accident relative to the PAG's were their applicatinu over an extended period of time and their effect on the "marketability" of food. The PAG's are intended for use up to one or two months. They include a derived response limit for total radionuclide intake which may be used as a basis on which to evaluate chronic intake during this period.

Producer's general experience with food contaminated with other materials indicates the general public's reluctance to accept "contaminated" products. This prohlem involves puhlic perception of radiation risks and governmental credibility. The issuance of PAG recommendations should influence public perceptions of radiation risks in a manner to encourage rational action relative to maketing and acceptance of foods.

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SOME RADIATION PROTECTION IMPLICATIONS OF THE THREE MILE ISLAND INCIDENT

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In many cases in the past, the design of Nuclear Power Plants (NPP) in the area of system safety and accident analysis has considered the radiation protection aspects mainly under conditions of normal operation including anticipated operational occurences. The Regulatory Guide 8.19 which was published recently by the USNRC (7) provides an example.

The Three Mile Island (TMI) incident and the recovery operations which followed it, focus attention on additional aspects which require careful considerations in the design of NPP:

(a) Safety related systems should be available for operation without any delay, even though a significant amount of fuel failures have already occurred in the reactor core.
(b) The operating personnel should be able to conduct recovery operations without undue exposures even when an accident has resulted in some fuel damage.
(c) Radiation-protection instrumentations should provide useful information even in the case of accidents which produce high radiation fields in the area of their reading.

High radiation fields that can be expected during severe accidents have been considered in the past mainly with respect to radiation measuring instrumentation (2,8) and radiation qualification of components in the containment (3). Only conditions of large-break LOCA (loss of Coolant Accident) were used to estimate the radiation fields (2). The desgin of shielding to allow access to safety systems and other vital equipment was in many cases based only on normal operating radiation levels (4).

In the next sections we discuss these safety problems, the reccommendations (5,6) made by the Nuclear Regulatory Commission (NRC) and by the Kemeny Commission (1) to treat these problems and arrive at some additional conclusions and suggestions.

SOME SAFETY PROBLEMS REVEALED IN THE TMI RECOVERY OPERATIONS

Our review of the recovery operations which followed the accident (1,5,6) revealed some safety problems. These safety problems may be divided into four general groups:

(a) Delay or prevention of the use of safety systems important to the recovery operations:

The Decay Heat Removal System is the main system planned for operation to maintain the reactor in the cold-shutdown mode; however, it was realized during the accident that this system is not sufficiently leak-tight for use with highly radioactive primary water.

Therefore, the main Reactor Coolant Pumps were used in their place even though they were not intended for cold shutdown operation. Thus, further contamination of the Auxiliary Building and radioactive fission product release to the environment were prevented.

During the accident it was required several times to enter the Auxiliary Building to align valves, start pumps or acquire samples from the containment atmosphere. Several of these operations were delayed or were not completed. The radiation fields within that building only allowed for a short stay of several minutes, which were insufficient to complete the required assignments.

(b) Undue exposures of operating personnel:

The review of the recovery operations reveal that some operations are needed more frequently than others. Among these are:

- Reactor Coolant sampling for boron analysis
- Operation of equipment from radwaste panels
- Change of filters
- Surveillance of equipment, monitors and instrumentations.

These operations have resulted in personnel exposures approaching the quarterly dose limit at each entry, and resulted in high extremity doses. Such high doses may be warranted in nonroutine one-time assignments and should be avoided in operations required on a frequent basis.

(c) Failure of radiation-protection instrumentation and monitors to provide correct information.

Radiation-protection instrumentation and monitors have been designed mainly to control normal operation including conditions of anticipated operational occurrences. Postulated accidents were also considered, but all protective systems were assumed to perform successfully and to reduce the fission product release. The TMI experience reveals many cases of monitors which were driven out of range. It also points out cases of monitors which measured the background radiation created by large amounts of noble gases rather than iodine or particulates being released through them. In the last case the monitors were exaggerating the actual release.

It should be pointed out that the measuring equipment operated successfully from the electro-mechanical point of view in most cases, i.e., the equipment was available and redundant equipment reached their set points within several minutes.

(d) Failure of evaluating radiation protection measurements, alarms and other information to determine the actual reactor situation:

In spite of much instrumentation going out of range, there were quite a number of high radiation alarms, high level measurements by the containment dome monitor (which did not go out of range) or by operators surveilling the Auxiliary Building. In addition the gaseous effluent monitoring system was indicating high effluent discharge (exaggerated by radiation from the noble gases). These measurements could be related to core conditions and fuel failuers in the core. During the TMI incident the above information was not correctly interpreted to indicate that significant fuel failure was taking place in core. It was rather explained as steam generator leakage to containment atmosphere combined with some steam generator tube failures. This indicates a need for improved training of personnel to distinguish radiation feilds indicating abnormal occurrences from normally encountered fields.

RECOMMENDATIONS MADE BY INVESTIGATING COMMITTEES

Several investigations into the TMI incident were performed. The NRC investigation (5,6) and the Kemeny Commission investigation (1) resulted in some recommendations related to radiation protection from the system safety point of view.

With respect to the safety problem (a) above, it is recommended by the NRC (5,6) to improve the integrity of systems outside containment likely to contain radioactive materials. No design improvements are required for, at least, the short term. The recommendations call for implementation of all practical leak reduction measures for the systems and performance of leakage rate tests on a periodic basis to keep the leakage rate at a constant level.

With respect to safety problem (b) above, it is recommended by the NRC (5) to perform a design review of the radiation fields and the shielding in the spaces arround systems that may contain highly radioactive materials. The design review should identify vital areas and equipment required during post-accident recovery operations. Measures to be taken to provide adequate access to vital areas should include post-accident procedural controls, permanent or temporary shielding and when required also redesign of facilities, components or systems. A quantitative source-term is suggested for the design review, i.e., the Regulatory Guide 1.3 source term (9).

With respect to safety problem (c) above it is recommended by the NRC (5) that no high range radiation monitors for noble gases in plant effluent lines and in the reactor containment be installed. In addition instrumentations for the monitoring of radioiodine and particulate effluents under accident conditions would also be provided.

With respect to safety problem (d) above it is recommended by the NRC (5,6) to improve post-accident sampling capability. A design and operational review of the reactor coolant and containment atmosphere sampling systems should be performed to determine the capability of personnel to obtain a sample within an hour under accident conditions, without incurring a radiation exposure exceeding the quaterly dose limit to whole body or extramities for radiation workers. Timely information from such samples can be important for an early understanding of core conditions.

The Kemeny Commission recommendations (1) are more qualitative in nature. They call on the NRC to include as part of its licensing requirements plans for the mitigation of consequences of accidents, including the cleanup and the recovery of a contaminated plant. The Kemeny Commission urges correcting inadequacies in equipment required for the mitigation of the accident (i.e., safety problem (a) above). It recommends that consideration should be given to overall gas-tight enclosure of systems processing highly radioactive water during the accident and the recovery phases. It urges improvements of radiation monitors and the provision of the capability to take and quickly analyze samples of containment atmosphere and reactor coolant.

SUMMARY

This paper presents four safety problems and the recommendations made by the NRC and the Kemeny Commission to treat them. It can be seen that the recommendations respond adequately to the safety problem mainly, in the above mentioned problems (b) and (c).

In its recommendations the NRC provides a source term to allow for more quantitative design review and for determination of cases which require some improvements. It is suggested here to add two supplemental steps to this quantitative approach:

(a) The use of specific scenarios of postulated accidents to determine the required recovery operations, which are the vital plant areas for post accident access and the required systems and equipment during the recovery phase. In particular, such scenarios may include, in additions to a large break LOCA, the small break LOCA, an ATWS event, a steam line break case and a control rod ejection followed by a small break LOCA.

(b) The use of a dollar value per man-rem as a criterion for determination when design imporvements are required rather than procedural controls.

The design review of the NPP will therfore include the assumption of a Reg. Guide 1.4 source term and a specific postulated accident scenario. Design improvements which have the potential for dose reduction both to personnel or to the population, would be judged by their dollar value per man-rem reduced. The criterion may be a lower dollar value then used for ALARA purposes today.

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ELECTROMAGNETIC POLLUTION OF THE ENVIRONMENT

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The recent development of the radar and telecomunication via satellite devices have imposed to the attention of the researches the necessity to solve the problem regarding the protection of radiofrequency electromagnetic and microwave radiations.

The use of high power devices have besides stimulated many interests in the research of the possible biological effects from human exposition to the electromagnetic radiations in acute or chronic form.

The present research deals with national situation concerning the ambient pollution by the radiofrequency electromagnetic waves and the microwaves (1)(2)(3).

The experimental part is limited from 0.5 to 500 MHz. This one is divided into section:

- devices numbering: broadcasting, telecomunication, television, air traffic control, air and shipborne navigation, meteorology, metallurgy, food processing, etc.;
- environmental tests: rural zones, towns, industries.

In this report are shown the results of measures that have been effected in Livorno (Fig. 1), to make out the real entity of electromagnetic pollution, in order to intervene if it will be necessary.

METHODOLOGY AND MEASUREMENT

Livorno has a surface of 104.6 km^2, a population of 177523 persons, a density of population of 1697 persons/km^2, medium altitude on the sea-level is 3 m, but some places have a highter position, as Valle Benedetta (338 m), Montenero (293 m), Quercianella (91 m), Castellaccio (300 m), Collinaia (52 m), Monterotondo (149 m).

Livorno is endowed with several factories and an industrial part that is characterized by an uninterrupted flux of ships.

According to the distribution of the installations we have divided the territory in three areas (Fig. 2 and Table 1):

- area A, including the port in which are situated about thirthy stations for marine support and aid, working in HF, UHF and VHF band; their maximum power is 150-400 W and they have an inconstant time of work.
  You can find them both on earth and on boats;
- area B, including Castellaccio and the places near there, where you can find 6 relay stations in VHF band for the comunications of power 10 W and 6 relay stations with parabolic antennas in UHF band of power is 1 kW and they work almost uninterrupting;
- area C: in this area all the broadcasting and television stations of

the town, public and private (transmitters and relay stations) are grouped and they have a power with fixed time of work.

About 12 km far from the town in Coltano there is another station with remarkable power, in Monte Serra; about 35 km from Livorno, there are antennes with great power for broadcasting, television and telephony.

The stations of O.M. and C.B. are not individualized.

We have choosen 50 test points (Fig. 3 and Table 2):

- area of the port: 6 stations;
- central area: 6 stations;
- peripherical area: 8 stations;
- area of Montenero: 5 stations;
- area of Castellaccio: 10 stations;
- the surroundings: 15 stations.

We have used the Electric and Magnetic Field Sensor TE 307, by Aeritalia (Torino), maximum range 10 V/m, the measure the electric field. The RAHAM, by General Microwave, has been used to measure the density of power.

RESULTS AND DISCUSSION

The results of our measurements are showen in Table 3.

The following areas can be individualized (Fig. 4):

- area 1 - including the area of the port; obtained results: 0.1÷0.2 V/m and 0.05÷0.3 mW/cm^2;
- area 2 - Castellaccio - obtained results: 1÷5 V/m and 0.2÷0.5 mW/cm^2;
- area 3 (coinciding with area C): obtained results: 0.2÷1 V/m and 0.05÷1 mW/cm^2;
- area 4 - it is the central area, excepted area C: obtained results: 0.05÷0.2 V/m and 0.03÷0.1 mW/cm^2;
- area 5 - it is the peripherical area: obtained results: 0.05÷0.2 V/m and 0.05÷0.1 mW/cm^2;
- area 6 - including the surroundings: obtained results: 0.1 V/m and 0.05÷0.1 mW/cm^2;

CONCLUSIONS

- The stations of area around the port have an influence that can't be measured on the electric field and the density of power in the central area.
- The stations of area C have a greater influence on the central area.
- According to our expectations, the stations of area B give the greatest contributions to values of the electric field and of the density uf power.
- In the peripherical area and in the surroundings the electric field and the density of power have the same values of the central area (area 4).

Making a comparation between the obtained results and the maximum values that have been fixed in the national rules we can believe that this are near these maximum values expecially regarding to the density of power.

TABLE 1 - Distribution of the installations

Area A : the port
Area B : Castellaccio
Area C : Montenero, Castellaccio, Cavour and Roma square

TABLE 2 - Measurement stations

AREA	STATION N°
The port	1-2-3-4-5-6
Central	7-8-9-10-11-12
Peripherical	13-14-15-16-34-35-36-37
Montenero	17-18-19-20-21
Castellaccio	22-23-24-25-26-27-28-29-30-31
Surroundings	32-33-38-39-40-41-42-43-44-45-46--47-48-49-50

TABLE 3 - Results of the measures

STATION n°	ELECTRIC FIELD V/m	E.M. DENSITY mW/cm^2	STATION n°	ELECTRIC FIELD V/m	E.M. DENSITY mW/cm^2	STATION n°	ELECTRIC FIELD V/m	E.M. DENSITY mW/cm^2
1	0.2	0.05	18	0.1	0.1	35	0.1	0.05
2	0.2	0.1	19	0.2	0.15	36	0.1	0.05
3	0.2	0.1	20	0.1	0.05	37	0.2	0.05
4	0.2	0.3	21	0.2	1	38	0.1	0.1
5	0.2	0.3	22	2	0.2	39	0.1	0.1
6	0.1	0.1	23	0.2	0.15	40	0.1	0.05
7	0.1	0.1	24	0.8	0.21	41	0.1	0.05
8	0.05	0.05	25	1	0.2	42	0.1	0.05
9	1	0.1	26	2.4	0.2	43	0.1	0.05
10	0.5	0.05	27	3.4	0.25	44	0.1	0.05
11	0.2	0.03	28	4	0.4	45	0.1	0.05
12	0.2	0.15	29	4.4	0.5	46	0.1	0.05
13	0.1	0.15	30	5	0.5	47	0.1	0.05
14	0.2	0.1	31	1	0.1	48	0.1	0.05
15	0.05	0.05	32	0.2	0.1	49	0.1	0.05
16	0.1	0.05	33	0.1	0.05	50	0.1	0.05
17	0.8	0.1	34	0.1	0.1			

The measurements are been executed in collaboration with the Italian Navy.

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Fig. 1: The position of Livorno.

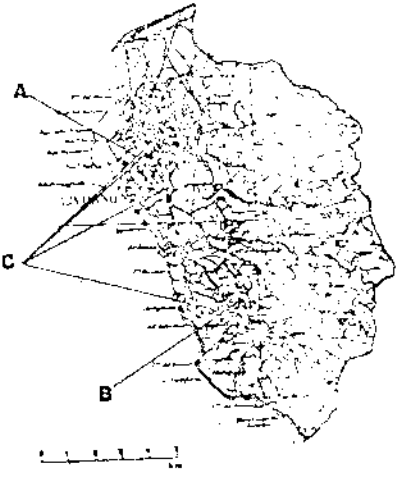

Fig. 2: Distribution of the installations.

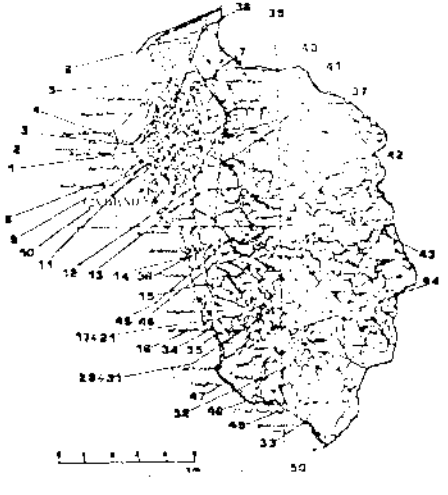

Fig. 3: Measurement stations.

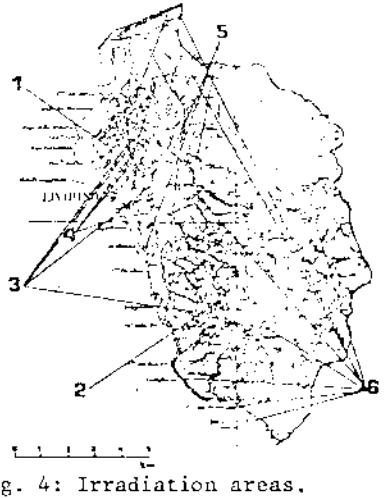

Fig. 4: Irradiation areas.

THE EVOLUTION OF NON IONIZING RADIATION PROTECTION STANDARDS IN NORWAY

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Members of the Norwegian Committee on Non Ionizing Radiation Protection. [x]

In Norway a Committee[x] was set up in 1975 to review the risks associated with the use of non-ionizing radiation in Norway. The Committee's mandate is also to recommend amendments to the existing standards on non-ionizing radiation protection. For a small country with limited resources for research on biological effects of radiation the Committee soon decided that its recommendations had to be based on research mainly performed in other countries.

However, such a decision does not solve all the problems. As long as disagreement exists among competent international sources any national standard may be open to question. Some approaches to non-ionizing radiation protection standards are discussed in this paper.

COMPARISON IONIZING-NON IONIZING RADIATION

Concern for potential non-ionizing radiation hazards appears to be universal. But, to date, there seems to be a universal lack of agreement as to the nature and the severity of the hazards involved.

For ionizing radiation, however, the situation is different. Most countries have long regulated the use of ionizing radiation through legislation, and only minor differences exist from country to country in this legislation. The main reason for this is the fact that the fundamental concepts of risk are internationally accepted. Several international bodies are involved in radiation

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The Norwegian Non-Ionizing Radiation Hazard Committee comprises the following members: Dr. F. Devik (Chairman, pathology), Dr. T. Hvinden (physics), Dr. H.H. Tjønn (occupational medicine), Dr. P. Syrdalen (ophthalmology), Dr. F. Storm Davidsen (military medicine), T. Liholt (telecommunications and broadcasting), H. Aamlid (radiation physics), G. Saxeböl (radiation physics) and M. Brady (microwave biological effects).

protection and they all take advantage in the basic recommendations issued by the International Commission on Radiological Protection (ICRP). The early identification of ionizing radiation hazards and the resultant early establishment of ICRP has for many countries, especially smaller countries, been of fundamental value in legislative work. The resultant situation today is that most groups in society - workers, politicians, employers and legal authorities - respect and accept the recommendations of the ICRP. Therefore the evaluation of ionizing radiation hazards and the relevant questions of risk are identified and treated similarly in different countries.

For non-ionizing radiation hazards, the non-existence of an international body capable of working out and recommending standards such as the ICRP for ionizing radiation causes both international and national difficulties. No one country, or groups within a country, can set or use standards with confidence when the international picture of the subject seems confused.

Additionally, an important difference exists in the nature of risk associated with non-ionizing radiation when compared to ionizing radiation. The presence of a stochastic effect of ionizing radiation which has given rise to the concept of the linear dose-effect relationship for protection purposes is well established. With respect to non-ionizing radiation most effects are considered non-stochastic i.e. there is a threshold above which an effect will occur and below which the effect will not occur. It should be expected that this nature of the dose (or irradiation)- effect relationship would make the setting of standards more simple and make the struggle for international agreement more easy. However, this is not always the case. While the risk and hazards and guides for protection standards are relatively uniform for the optical region of the spectrum, the existing protection standards for microwaves and radio-frequency show discrepancies both in principles and figures.

NORWEGIAN POSITION

In Norway the State Institute of Radiation Hygiene under the Ministry of Health and Social Affairs is the competent authority for protection against ionizing radiation as well as non-ionizing radiation. Many countries have organized the radiation protection system in a different way. This difference in administration may cause problems as far as international cooperation is concerned. For instance the IRPA will have problems in collecting the international expertise in non-ionizing radiation simply because these experts are not members of the national radiation protection societies.

The general radiation protection Law in Norway which was passed by the Norwegian Parliament in 1938 opens

possibility for the Government to implement this Law also for non-ionizing radiation such as light, ultraviolet radiation, diathermy, short waves, etc. This widening of the 1938-Act was made by regulations issued by the Ministry of Health and Social Affairs in 1976 about the same time as the multidisiplinary Committee on NIR was set up. Working under the auspices of the State Institute of Radiation Hygiene the Committee decided to evaluate an approach to protection against non-ionizing radiation based upon the same philisophy as for ionizing radiation.

RISK PHILOSOPHY

Any standard concerning a hazard must be built on a philosophy of acceptable risk. Because of difference in physical properties and biological effects the spectrum is divided into two parts: the optical and the radiofrequency.

The risk and hazard questions associated with the optical spectrum, i.e. wavelengths between 100 nm and 1 mm, seem to be relatively uniform. For instance the standards for exposure to lasers have reached a high degree of international agreement . Standards and recommendations have been proposed by international bodies such as IEC and WHO. The ultraviolet part of the optical spectrum is a region of growing interest. One reason for this is that the number of skin cancers including malignant melanomas in many countries has shown a steady increase over the last 10-20 years. Medical experts feel confident that the increased exposure to sunshine for tanning purposes have influence upon the observed increasing number of cases. For UV-radiation the erythema curve is the main basis for safety standards. So far the evaluation of standards has not been influenced by the acceptance of the carcinogenic risk. In Norway the NIR-Committee has recommended that all sunlamps intended for total body exposure should be equipped with a warning label reading: "By increased UV-exposure the cancer risk may increase". The argument for the NIR-Committee to propose this recommendation was the principle of optimization used by the ICRP: "All exposures shall be kept as low as reasonably achievable, economic and social factors being taken into account". The Committee found that this part of the basic philosophy of ICRP should also be valid for non-ionizing radiation especially when such radiation can be associated with risk for stochastic effects.

APPROACH TO SAFETY STANDARDS FOR MICROWAVE AND RF

The NIR-Committee started its work on microwaves and radiofrequency by stating that the general exposure limit of 100 W/m^2 being inofficially used in Norway for the frequency range 300 MHz - 300 GHz should not be lowered unless clear biological evidence was found. However, by

taking the ICRP principle of optimization into account, the Committee has found that 100 W/m^2 is an unnecessary high limit for whole body all day occupational exposure. On the other hand the Committee of course recognizes the need for time limited or partial body exposure to higher levels. Accordingly the Committee now discusses the implementation of a new limitation concept which will start off with a general and lower exposure limit for the general public. Exposure to higher levels may take place for radiation workers under controlled conditions. This includes principles such as time limitation and measurements of exposure levels. The exposure limit for the general public being discussed at the moment is 10 W/m^2 for the frequency range 10 MHz - 300 GHz. This limit is also thought to be implemented for full day occupational exposure. The Committee recognizes the need for additional exposure limits for partial body exposure. However, the Committee strongly feels that a set of regulations should be as simple as possible. Unnecessary frequency dependent specifications should be avoided. One item that will be seriously discussed in the Committee is the need for differentiation in standards between radiation workers and the general public. The following way of thinking will support the idea of such a differentiation even if only non-stochastic effects of radiation are considered. As the number of irradiated individuals increases the mean value of the threshold-effect will be found more exact. However, the scattering of the results will increase and the slope of the envelope curve (or ED_0-curve) will decrease and the ordinary S-curve will be streched out and show a more linear form. We may come to the conclusion that this concept should be realized in the regulations. Following this way of thinking the exposure limits for large population groups may be set lower than exposure limits for full time radiation workers. The idea is to take care of the varying biological sensitivity to radiation in the population.

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A Recommended Permissible Environmental Standard for Microwave and Radiofrequency Radiation

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Non-ionizing radiation in the radiofrequency and microwave region of the electromagnetic spectrum has received comparatively insubstantial investigation and environmental regulatory attention. With the increasing utilization of the radiofrequency spectrum in recent decades, including radio and television broadcasting, radar for commercial transportation control and military observation, consumer products such as microwave ovens, diathermy equipment employed in medical therapy, and other applications, there has been a surge of warranted concern and interest in public health control of possible harmful biological effects of non-ionizing radiation in humans.

In June 1978, in my capacity as Director of the Bureau for Radiation Control of the New York City Health Department, I recommended to the Board of Health of the City, adoption of a regulation to the Health Code which would set maximum permissible levels for potential exposure to microwave and radiofrequency radiation to members of the general public in uncontrolled or unregulated areas.

The overall framework of the recommended regulation was derived from these main considerations.

(1) Biological and clinical effects have been exhibited on a sufficiently broad scale in laboratory experiments and clinical observation to demonstrate the potential for physiological impairment in humans from microwave/radiofrequency radiation from various power or energy density levels. Physiological effects conclusively or almost certainly demonstrated in animal experiments include cataractogenesis, hormonal alterations, chromosomal anomalies, and hematological changes. Less certain but probable effects include central nervous system impairment and mutagenesis. In the realm of conjectural but possible effects, warranting however the most careful public health scrutiny, are mutagenic, oncogenic and teratogenic influences.

(2) The very approximate threshold for some observable effects of varying public health implications appears to be between one and 10 milliwatts per cm^2 for noncontinuous exposures. Of course, a microwave or radiofrequency induced physiological perturbation may not be clinically significant in the sense of hazard or impairment.

(3) Also legally licensed mobile units employed by police, fire and emergency response departments have a potential for exposing individuals in the public environment to hundreds of milliwatts/cm^2 in the microwave and ultra high frequency range. Presumably these transmissions are of an intermittent and non-sustaining nature as far as individual radiation exposures are concerned. Their aggregate public health impact deserves careful future scrutiny.

Biological Effects and Clinical Observations for Microwave and Radiofrequency Radiation

It should be emphasized at the outset that the frequency region of interest i.e. 30 KHz-300 GHz is in the non-ionizing portion of the electromagnetic spectrum. Unlike x-or gamma radiation typical of the ionizing region of the electromagnetic spectrum, microwave/radiofrequency radiations are relatively low energy in the quantum sense and incapable of separating bound electrons from atoms and molecules.

The biological modes of action may be characterized in three very general ways:

(1) Macroscopic heating or hyperthermia of a whole living organism or substantial part thereof resulting in the sustained elevation of temperature producing reversible, irreversible or partly reversible biological changes.

(2) Macroscopic heating of individual cells or very small sections of an organ producing biological changes of various persistence without perceptible temperature rise in the macroscopic sense.

(3) Non-thermal or only partly thermal effects relating to interaction of impinging electromagnetic radiation with the electric or magnetic fields of living tissues or cells.

Very approximately, mode one, microscopic heating or hyperthermia, would be associated with power densities in excess of 10 milliwatts/cm^2. The second mode, microscopic heating, would identify with power densities roughly in the range of one to 10 milliwatts/cm^2. The third or athermal mode, would be related to biological effects produced below one milliwatt/cm^2 or in the microwatt/cm^2 power density range.

In Table 1 are listed the principal representative biological effects purported to have been observed in various biota by investigators as the result of microwave or radiofrequency exposure. Also included are representative effecta alleged to have been seen clinically or by epidemiological inference in human beings. Most such reports have addreased observations in workers occupationally connected with microwaves or radiofrequency radiation and information on the frequency range or power density is not available at all or incomplete.

Measured Background Environmental Levels for Radiofrequency Radiation

In one survey performed by the United States Environmental Protection Agency (EPA) covering the radiofrequency ambient environmental bands between 46 MHz and 900 MHz, the power densities encountered generally fell into the range of between 0.001 and 1 microwatt per square centimeter ($\mu W/cm^2$) with a median value of between 0.02 - 0.03 $\mu W/cm^2$. Forty locations were surveyed in the metropolitan New York area. The range of ambient values encountered in this region from a minimum .000068 $\mu W/cm^2$ (Tottenville, Staten Island) to a maximum of 4.6 $\mu W/cm^2$ (Mount Pleasant Street, West Orange, New Jersey). The average for these forty observed locations was 0.22 uW/cm^2 including the two maximum values. If the two maxiumum values (i.e. 4.6 $\mu W/cm^2$ and 1.9 $\mu W/cm^2$) are omitted, the average for the remaining 38 sites is 0.069 $\mu W/cm^2$.

Table 1. Biological and Clinical Effects Observed from Exposure to Microwaves or Radiofrequency Radiation.

In Non-humans (biological effects)

Chromosomal anomalies

in Chinese hamster and drosophila melanogaster; also breaks in human lymphocytes in culture; frequency 5 to 40 megahertz(pulsed).

Mutagenesis

in Swiss male mice; DNA changes; high mutagenicity index;in sperm; exposure ranges 50 milliwatts per square centimeter at 17 gigahertz. Also, basic changes in cell structure and density of bacteria between 10 to 50 milliwatts per square centimeter at 50 to 90 gigahertz.

Teratogenesis

in mealworms; preceded presumed thermal lethality at 9 to 10 gigahertz with power density not reported but total power of 20 to 80 milliwatts.

Behavioral impairment

in rats; adverse motor coordination and balance;induced "docility"; exposure ranges between 0.4 and 2.8 milliwatts per square centimeter at 1.3 to 1.5 gigahertz.

Neuroendocrine and hormonal alterations

in rats and dogs; transient changes ascribed to temperature increase. Exposure 20 to 60 milliwatts per square centimeter for 30 to 60 minutes at 2,450 megahertz.

Prenatal impairment of body and brain weight

in rats; 10 milliwatts per square centimeter at 2,450 megahertz for 5 hours daily for 17-day gestation period.

Blood-brain barrier alterations

in hamsters; 10 milliwatts per square centimeter at 2,450 megahertz.

Central nervous system influence

in chicks; 1 to 2 milliwatts per square centimeter at 147 megahertz increase in calcium ion release.

Mortality

in rats, rabbits, dogs; effects dominantly thermal;typical exposure ranges 40 milliwatts per square centimeter for minutes to hours,2800 megahertz; various wavelengths between 1 millimeter to 10 centi - meters.

In Humans(biological or clinical effects)

Cataractogenesis and other ocular effects

Epidemiological studies among microwave workers show increase in lens opacities and retinal lesions;frequency range, power density incompletely reported.

Central nervous system influence

Auditory nerve response not necessarily hazardous.

Oncogenesis

Speculative; epidemiological suggestion of carcinogenesis in North Karelia region of Finland.

Biochemical imbalances

Many studies but exposure parameters not reported.

Subjective physchological complaints

Wide variety of alleged clinical effects between 0.01 to 10milliwatts per square centimeter.

Hematological changes

Many blood changes cited; main instability in leukocyte indices.

These measurements were all done at street level. Total field strength measurements reported by the EPA yielded maximum levels for New York City as follows:

	Total Field Strength ($\mu W/cm^2$)
World Trade Center	
Outdoor Observation Deck	6.8
Indoor Observation Deck	1.2
Empire State Building	32.50
Pan Am Building	10.3

Two locations, one in Miami, Florida (2 Biscayne Boulevard) and one in Chicago (Sears Tower) yielded higher field strengths, 96.85 $\mu W/cm^2$ and 65.73 $\mu W/cm^2$ respectively.

Conclusion

The thrust of the evidence would indicate an occupational permissible level for sustained working exposures (viz. in excess of 0.1 hours) of 500 $\mu W/cm^2$ i.e. about ten percent of the approximate midpoint of the 1-10 mW/cm^2 potentially hazardous interval.

Public health prudence, in the absence of more definitive research to the contrary, would dictate that unregulated or uncontrolled areas which are available to men, women and children by virtue of residence, recreation or general public access maintain a microwave/radiofrequency power density environment not exceeding ten percent of the indicated occupational permissible level, or 50 $\mu W/cm^2$. This reduction is in sound recognition that the public environment is not subject to the same presumed level of biomedical surveillance and detailed health and safety awareness as the occupational workplace.

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INSIDIOUS OCULAR EFFECTS OF LASER RADIATION

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Most safety codes for lasers emitting in the wavelength band 400-1400 nm are based on threshold studies for a single event thermal lesion, where ocular damage is related to wavelength, retinal image size, pulse duration and energy density. Current codes of practice are derived from a 50% damage probability for different laser systems producing diffraction limited image sizes.

Noell in 1965 (1) discovered that the rat retina could be damaged by exposure to moderate light sources. Marshall (2) showed that pigeons exposed to moderate white light luminances suffered cone loss; he continued his work with fish (3) where he was able to selectively damage specific cones responding to one primary colour, by illuminating the aquarium with monochromatic light. Harworth & Sperling (4) in their behavioural studies produced temporary and permanent colour blindness in monkeys following exposure to intense spectral sources. Ham (5) in his studies showed that retinal damage thresholds decreased for short wavelengths. Zwick (6) exposed 2 monkeys to very low luminances of argon laser irradiation on a hemisphere and was able to show that photopic visual function was substantially depressed and that recovery was minimal over a 12 month period.

Concern is now felt that current codes of practice may be inadequate to protect individuals exposed, for long periods, to subthreshold levels of laser irradiation, particularly so with lasers emitting at the blue end of the spectrum.

Many workers involved in activities such as research, holography, data processing and laser light shows are exposed over long periods to subthreshold laser irradiation. This has caused us to investigate visual function in one such group of workers who have been exposed to argon laser irradiation over a 2-year period in the development of a laser scan visual flight simulator.

MATERIALS AND METHODS

The laser scan, at present, employs two argon lasers of 5 watts nominal output working multimode. The laser camera illuminates a terrain model Fig 1 and is normally operated at 500 milliwatts. The information from the model modulates a laser projector which has operated for 90% of the time at 1 watt output and for remaining 10% of the time at 4 watts output. The display is presented on a hemisphere, the luminance of which has varied between 3.5-7.0 cd/m^2. The field of view to the observer is 180° in the horizontal and 60° in the vertical. The resolution of the display is 5,280 television lines and the colour is blue-green. Production systems will be in full colour, this will be achieved by separately modulating the blue and green lines of the argon laser and by adding red from a krypton laser.

Eight workers have been monitored and their exposure times for viewing the display and for being in the vicinity of the lasers are given in Table I.

As evidence from the literature suggests that cones are primarily at risk, the measures of visual performance have mainly concentrated on photopic function. The tests were:-

1. Clinical eye examination.
2. Liminal brightness increment for white and blue light at photopic and scotopic luminances.
3. Colour vision testing with Farnsworth-Munsell 100-Hue Test. Illumination of 10 Lux.
4. Perimetric field of view measurements for 2 mm white and blue stimuli.
5. Macula thresholds for white and blue light stimuli.
6. Central visual fields for white and blue light stimuli
7. Dark adaptation curves for white and blue light stimuli, following light adaptation.

The tests 5, 6 & 7 were made on a calibrated Friedmann Visual Field analyser. The unattenuated luminances of the white and blue light sources were 3.0 and 0.1 photopic nit seconds respectively.
The blue filter used in these tests was an Ilford 623, a spectrophotometric trace of which is shown in Fig 2.

TABLE I. Estimated exposure times, of workers exposed to argon laser light.

SUBJECT	AGE	HOURS LOOKING AT HEMISPHERE DISPLAY	HOURS WORKING IN LASER ENVIRONMENT
A	47	0	0
B	44	2	100
C	40	3	1
D	32	3	0
E	45	1	1
F	25	300	120
G	40	150	10
H	32	20	100

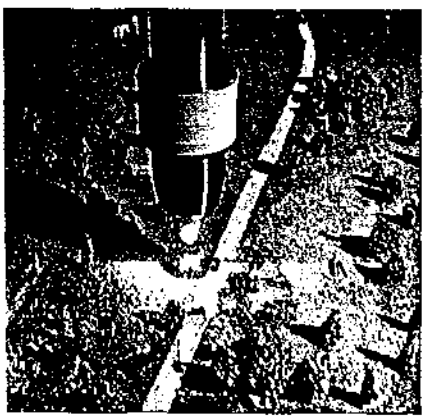

Fig I. View of laser camera and terrain model.

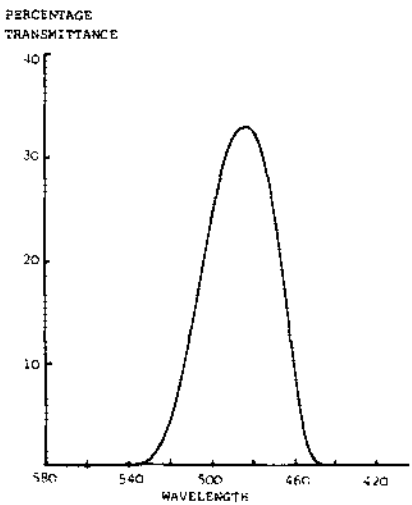

Fig 2. Spectrophotometric trace of Ilford 623 filter.

RESULTS

The clinical examination included an ocular history, tests of pupillary function, visual acuity estimates on a Snellen chart under white and blue illumination with a refraction where necessary, macula function tests using an Amsler grid, a fundoscopic examination and an examination of the anterior segment of the eye with a slit lamp. These tests did not reveal any pathology which could be attributed to work in a laser environment. The expected addition of a -0.50 sphere was necessary to restore visual acuity under blue illumination.

Colour vision testing with the Ishihara pseudo-isochromatic plates did not show any deficiencies. Colour vision testing with the Farnsworth-Munsell 100-Hue test under low illumination did reveal some error scores, particularly in the blue-green and purple hue regions.

TABLE 2. Farnsworth-Munsell 100-Hue test

SUB-JECT	HUE ERROR SCORES				TOTAL ERROR SCORES
	610-570 nm	570-500 nm	500-470 nm	470-630 nm	
	RED-YELLOW	YELLOW-GREEN	GREEN-PURPLE BLUE	PURPLE BLUE-RED PURPLE	
A	4	8	4	0	16
B	0	4	12	0	16
C	4	15	32	8	59
D	4	0	16	4	24
E	4	8	24	20	56
F	0	0	0	0	0
G	0	4	4	0	8
H	0	8	4	4	16

Perimetric assay did not show any loss of peripheral field for white or blue stimuli.

The log densities for the white and the blue macula thresholds are the maximum densities at which no error was made for ten consecutive stimuli. The log densities for the central fields are the maximum densities at which the central field was full for the white and the blue stimuli.

TABLE 3. Central fields and macula thresholds.

SUB-JECT	AGE	NORMAL LOG DENSITY FOR AGE (WHITE)	LOG DENSITY FOR FULL FIELD				MACULA THRESHOLDS. LOG DENSITIES.			
			WHITE		BLUE		WHITE		BLUE	
			Rt.	Lt.	Rt.	Lt.	Rt.	Lt.	Rt.	Lt.
A	47	1.8	1.8	1.8	1.2	1.2	2.0	2.2	1.0	1.2
B	44	1.8	1.8	1.8	1.0	1.2	2.2	2.2	1.0	1.0
C	40	2.0	2.0	2.0	0.8	1.0	2.4	2.4	0.8	0.8
D	32	2.0	2.0	1.8	1.0	1.0	2.6	2.4	1.2	1.2
E	45	1.8	1.8	1.8	1.0	0.8	2.6	2.4	1.0	1.0
F	25	2.0	2.0	2.0	1.2	1.2	2.4	2.4	1.2	1.2
G	40	2.0	2.0	2.0	1.2	1.2	2.4	2.4	0.8	0.8
H	32	2.0	2.0	2.0	1.0	1.0	2.8	2.8	1.2	1.0

The results of the liminal brightness increment measures showed the expected increase in contrast threshold $(\frac{\Delta I_s}{I})$ at the scotopic luminances for both white and blue light.

TABLE 4. Liminal brightness increment

BACKGROUND LUMINANCE cd/m^2		ΔI+I cd/m^2 mean	I cd/m^2 mean	$\Delta I/_I$% RANGE	$\Delta I/_I$% MEAN
3.2 x 10^{-3}	BLUE	3.46	3.18	2.8-14.0	8.76
SCOTOPIC	WHITE	3.49	3.19	3.7-14.0	9.49
5.0 x 10^{-2}	BLUE	5.15	4.96	2.0-7.6	3.92
PHOTOPIC	WHITE	5.17	5.00	1.2-5.6	3.42

Dark adaptation curves after white light adaptation did not show any significant departures from normal for white and blue stimuli. The scotopic portion of the curve was not continued beyond 20 minutes.

DISCUSSION

This is a preliminary survey into the visual function of workers exposed to long term argon laser irradiation and as yet it has not been possible to demonstrate any visual decrement which could be, directly, attributed to work with lasers. Paradoxically the workers with the longest exposure times performed as well or better than those with minimal exposure. The decrements found were those which would be expected in any random group of varying age. The survey is limited, it involves eight workers of whom only half have been exposed for a significant period. It is intended to repeat the tests of visual performance at six monthly intervals as exposure times increase.

It will be understood that by the nature of the work exposure times are approximate, particularly with the variable exposure incurred when working in the laser environment. The exposure time for looking at the hemisphere display is more precise, as is the luminance. Should visual decrements develop it would be valuable to correlate these with display times over the six monthly intervals.

The dosimetry of laser exposure in man can never be as precise as that in experimental animals. It is considered, however, that the subtle changes in vision which may occur with long term low level laser irradiation may only be detected in man.

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SECONDARY STANDARD DOSIMETRY SYSTEM WITH AUTOMATIC DOSE/RATE CALCULATION

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INTRODUCTION

In view of the increasing requirements for standardization measurements in radiation dosimetry a versatile and automated secondary standard instrument*) has been designed for quick and accurate dose/rate measurement in a wide range of radiation intensity and quality for protection- and therapy level dosimetry. The system is based on a series of secondary standard ionization chambers /1/ connected to a precision digital current integrator /2/ with microprocessor circuitry for data evaluation and control. Input of measurement parameters and calibration factors stored in an exchangeable memory chip provide computation of dose/rate values in the desired units.

IONIZATION CHAMBERS

Secondary standard ionization chambers require excellent reproducibility and long-term stability of the sensitive volume. Therefore graphite is generally used as the wall material in combination with Al collecting electrodes to achieve a flat energy response. This compensation however is only valid for free-air measurement and may introduce significant errors when used in-phantom due to the inhomogeneous construction materials /3/.

In contrast the described chamber design uses walls and electrodes made from Polyacetal resins $(CH_2O)_x$. This material provides superior mechanical properties assuring the necessary long-term stability of dimensions and a most suitable chemical composition. By choosing the proper mixture of Polyacetal with Polytetrafluoroethylene (PTFE) and small additions of higher Z-material such as CaO, the chambers can be made virtually tissue-, water-, or air-equivalent as desired.

In order to achieve electronic equilibrium for photon energies above 1 MeV the wall thickness has to be at least 2 mm. For soft X-rays the absorption in the wall is compensated due to a thin vacuum-deposited layer of Al on the inner wall surface. In this way the energy response is within $\pm$ 2% between 0,02 - 1,2 MeV without any additional build-up caps etc.

*) International patents.

The ionization chambers (see fig. 1) are tailored to the different applications. For radiation protection measurements at low doserates a large spherical air-equivalent chamber of 10 l volume can be used down to environmental levels. An internal check-source of ^{241}Am can be introduced into the center of the chamber through a hollow axial tube protruding from the stem to the other pole of the spere. For high doserates in therapy level dosimetry a small water- or tissue-equivalent thimble chamber of 1 cm^3 volume has been designed which can directly be put into a water phantom. For the intermediate doserate range a 100 cm^3 spherical chamber can be used. In addition a backscatter chamber for soft X-ray therapy measurements is in preparation.

ELECTRONIC CIRCUITRY

The basic components of the electronicsystem are shown in the simplified block diagram fig. 2.

The ionization chambers are connected to a MOS/FET electrometer amplifier through a series of reed switches (R1-R3). The exchangeable measuring capacitor (100 pF - 100 nF) which determines the range of measurements is normally shorted and the input grounded. During a measurement cycle R2 is closed and R1/R3 opened. The ionization current generates an increasing voltage signal at C, which is measured by an automatic TOWNSEND-balance circuitry consisting of a 5 digit dual slope DVM integrator with compensation by a feedback amplifier system. With an offset current of less than $10^{-15}A$ ionization currents in the range of $10^{-12}A$ to $10^{-7}A$ can be measured within $\pm$ 1% error.

The system is controlled by a microprocessor central processing unit (CPU) chip containing a 1 K x 8 bit EPROM and 64 byte RAM with 6 MHz quartz clock and internal timer/counter. An additional arithmetic processing unit (APU) performes all calculations. A 320 byte RAM is used as a data buffer controlled by the CPU. Up to 100 chamber calibration factors (for 10 chambers at 10 qualities) and 10 capacitance values (C) are stored in an exchangeable memory chip (2 K x 8 bit EPROM). This reusable chip is loaded after calibration and exchanged with each new chamber.

The measurement parameters (atmospheric pressure, temperature, radiation quality and number of cycles) are manually set on BCD-thumb wheel switches on the front panel. Preset dose values and additional calibration factors not contained in the memory can be manually selected if required.

The LED-display contains the voltage signal (5 digits),

integrating time (4 digits), dose/rate (4 digits) with 5 prefix-symbols and the unit of measurement (Gy, R,/h, /min,/s).

With a built-in miniaturized alphanumeric printer (16 characters/line) dose/rate, meanvalue, standard deviation, time, calibration factor and capacitance value are recorded.

The ionization chambers and measurement capacitors are identified by encoding resistors contained in the connectors. The programmable high voltage supply (0 - 2 kV,2 mA) is automatically set to the correct chamber high voltage by the CPU as a function of the decoded chamber number.

DISCUSSION

The system described is designed for secondary standard measurements in protection- and therapy level dosimetry. It covers a wide range of measurement between 1 µR and 100 kR (0,2 nC/kg - 20C/kg) with proper chamber and capacitance and automatically calculates dose/rate due to its microprocessor circuitry. The ionization chambers provide excellent long-term stability and energy response and can be used with internal check sources to test validity of calibration. The system is a useful tool particularely for daily measurements in a secondary standard dosimetry laboratory or radiation therapy center.

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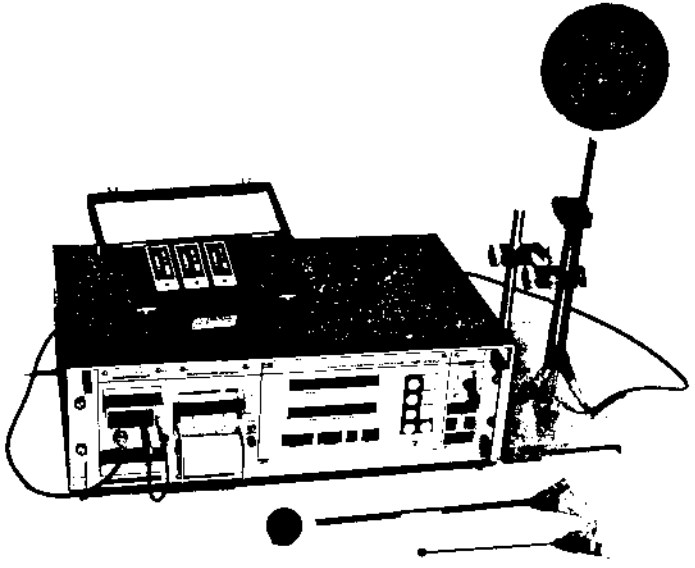

Fig.1 PHOTOGRAPH OF THE SECONDARY STANDARD SYSTEM

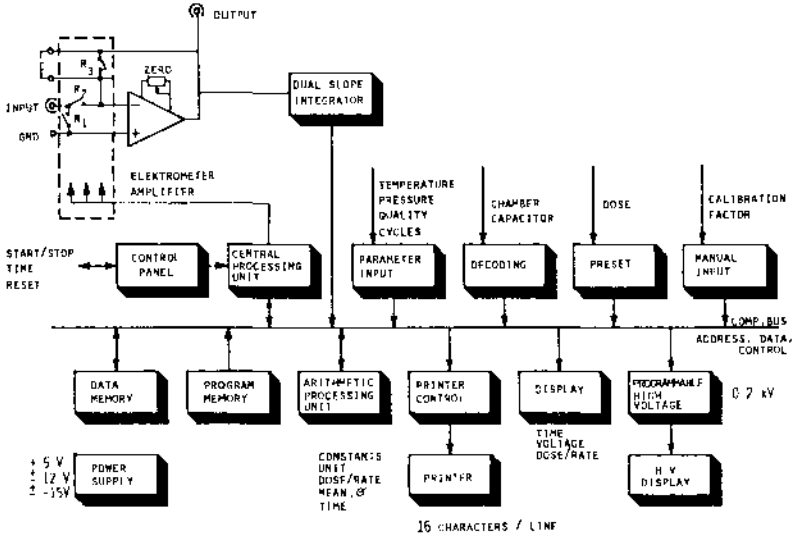

Fig.2 SIMPLIFIED BLOCK-DIAGRAM OF THE ELECTRONIC SYSTEM

A PERSONAL TRITIUM MONITOR

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A tritium monitor, similar in size to a normal gamma survey meter, is being developed as part of the systematic approach to providing an improved capability for measuring tritiated water vapour (HTO) near workers in CANDU nuclear power plants. The prototype is shown in Fig. 1.

Tritium, formed by neutron capture in deuterium, is present in the reactor heavy water so that small vapour leakages may result in unexpected airborne contamination in some working areas. The requirement is for a quick indication of concentrations of tritiated water vapour

FIGURE 1. Prototype Personal Tritium Monitor

in air that are the order of one maximum permissible concentration and above, wherever a worker happens to be. Sensitive methods have already been developed for sampling [1] and for monitoring with a centrally-located instrument [2]. However, size, mass and cost are more important than sensitivity in an instrument that is designed to meet this requirement as opposed to a central monitor. Tritium monitors, even portable ones developed and described to date, have not been such that they can be available in large quantities. The instrument described here is intended to complement other monitoring methods by allowing on-the-spot assessments of tritium hazards and is designed to meet the severe constraints posed by the needs for portability and availability.

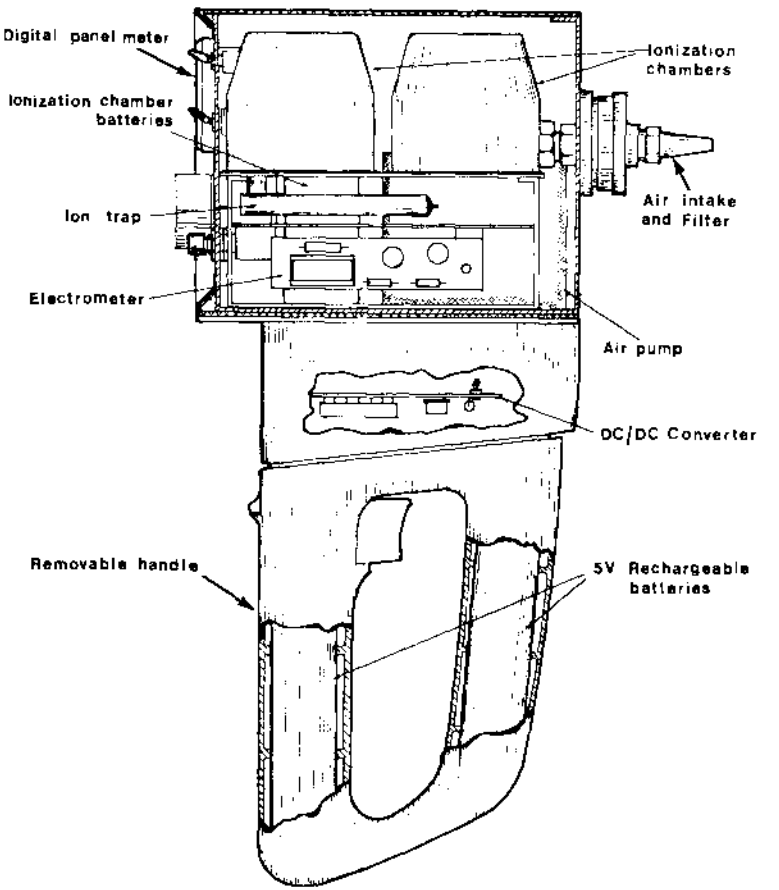

FIGURE 2. Mechanical assembly of the tritium monitor

As illustrated, the instrument is small enough to be easily handled and weighs only 2 kg. Only inexpensive, readily-obtainable mechanical and electrical components have been used in the instrument. Components need little machining and the assembly is uncomplicated.

The mechanical assembly is shown in Fig. 2. The tritium detector is an 80 cm^3 ionization chamber made from a nickel crucible (as used in chemical analyses) soldered to a fibre-glass printed circuit board on which guard rings and electrical connections have been etched and the copper nickel plated. No O-ring seals are used in the entire assembly which may be thrown away if it should be contaminated. Sample air is sucked through the molded plastic filter assembly and the cylindrical ion trap to the tritium ionization chamber by the loud-speaker coil-driven air pump [3]. A second ionization chamber, also 80 cm^3, is adjacent to the tritium chamber and, in the prototype shown, is sealed. The two ionization chambers are oppositely polarized so that the current from the sealed chamber partially cancels the background current in the tritium chamber caused by ambient gamma radiation. The electronics, on a 3 cm x 7 cm printed circuit board, are mounted under the ionization chamber assembly. The chamber and electronics can be removed as a single package for maintenance or replacement. The power supply is an inexpensive power handle, as sold for portable tools [4], that includes a rechargeable nickel-cadmium cell. The required voltages are derived in the DC/DC converter mounted in the mating assembly to the power handle to which the instrument is attached. The power handle may be detached for recharging whilst other handles are used. The switch on the power handle may be used to control intermittent operation of the pump.

The electrical schematic is outlined in block form in Fig. 3. The 4.2 V battery in the power handle drives the air pump directly and is converted to 5.6 V and 10 V for the electrometer and digital display meter. As shown, the major current drain from the power handle is the air pump. The battery lifetime with the current drain shown is about 5 hours. If an external (hand-operated) air pump is used then the battery lifetime is an order of magnitude longer.

The MOSFET input stage (3N155As, connected differentially as source followers) and one amplifier of a quad-amplifier DIP, LM324, comprise an operational amplifier with high input impedence and an effective feedback resistance of 4 TΩ. A smoothing circuit may be switched into the output stage to lengthen the electronic response time constant from less than one second to 3.3 seconds. The controls shown are on the monitor front panel; three additional positions on the slow/fast switch allow the battery voltages to be monitored.

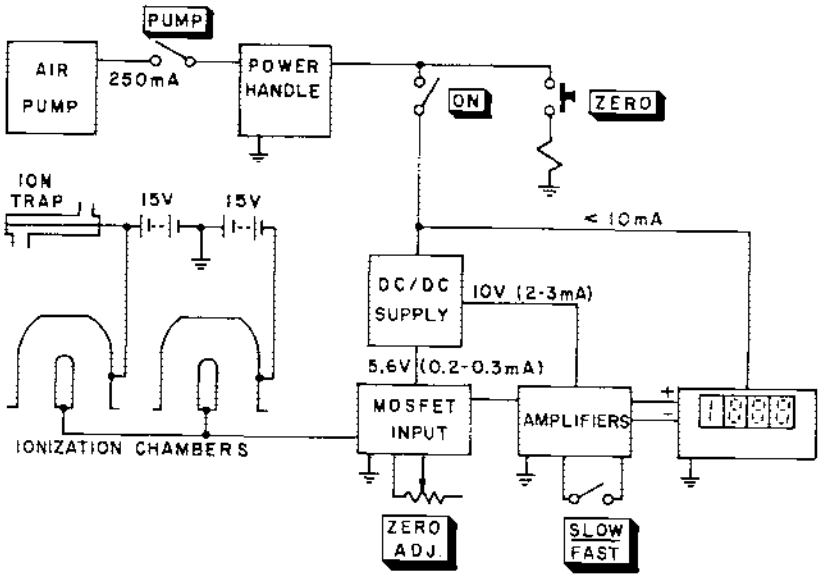

FIGURE 3. Block diagram of the electrical schematic of the tritium monitor

The sensitivity of the ionization chamber/amplifier combination is 3.2 mV/$(MPC)_a$. (The $(MPC)_a$ used is 10 $\mu Ci/m^3$ [0.37 MBq/m^3]). The input to the 3½ digit meter is scaled to give an output range from 1 to 1999 $(MPC)_a$.

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MEASUREMENT OF ABSORBED DOSE-RATE IN SKIN FOR LOW-LEVEL BETA-RAYS

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A new type of beta-ray absorbed dose-rate meter has been manufactured applying the detection method developed by K. Bingo et. al. (1) to evaluate the absorbed dose in skin at a depth of 7 mg/cm^2 lying above a contaminated sandy beach.

The instrument uses a plastic scintillator with 2.5 mm thick and a single channel pulse height analyzer(SCA) to obtain the best correlation between the instrument response and the absorbed dose-rate.

The absorbed dose-rates for beach sands were measured by the instrument and were compared with the calculated values by the computer code "BETA-SAND" developed by PNC and JAERI (2). With this instrument the absorbed dose-rate of about 2 μrad/hr can be measured by 60 minutes counting.

METHOD AND RESULTS

The beach sands were sampled at eight points around PNC Tokai-works and were dried at about 80°C by a drying oven. The concentration of potassium-40 in the samples were determined by a Ge(Li) spectrometer. The concentration of K-40 in the samples showed a range from 6.3 to 14.9 pCi/g-dry.

The schematic block diagram of the beta-ray absorbed dose-rate meter used in this experiment is shown in Fig. 1.

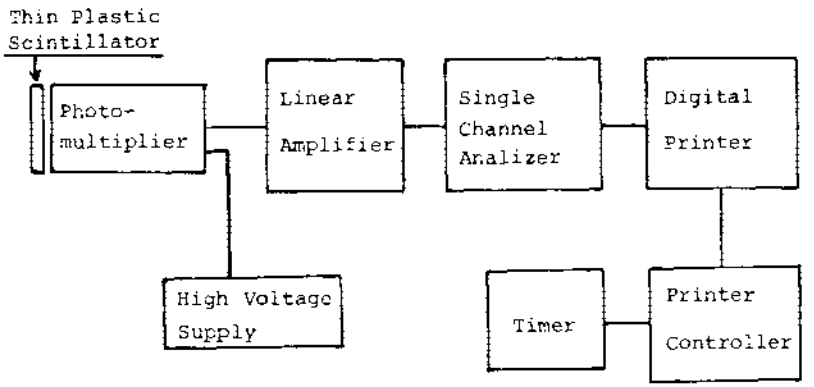

Fig. 1 Schematic Block Diagram of the Beta-ray Absorbed Dose-Rate Meter

The reference beta-ray sources shown in Fig. 2 such as Sr-90/Y-90, Cs-137, Pm-147 and Tl-204 were used to determine the discrimination level and the window width of the SCA. The absorbed dose-rates in skin above the sources were determined by the theoretical value calculated by W. G. Cross (3). The conversion factor obtained from the calibration was 0.4 μrad/hr per cpm for the maximum beta-ray energy ranging from 0.5 to 2.27 MeV. The beta-ray spectra of Cs-137 and Tl-204 measured by the instrument are shown in Fig. 3.

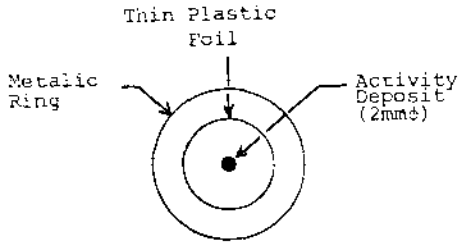

Fig. 2 Reference Beta-ray Source (LMRI made)

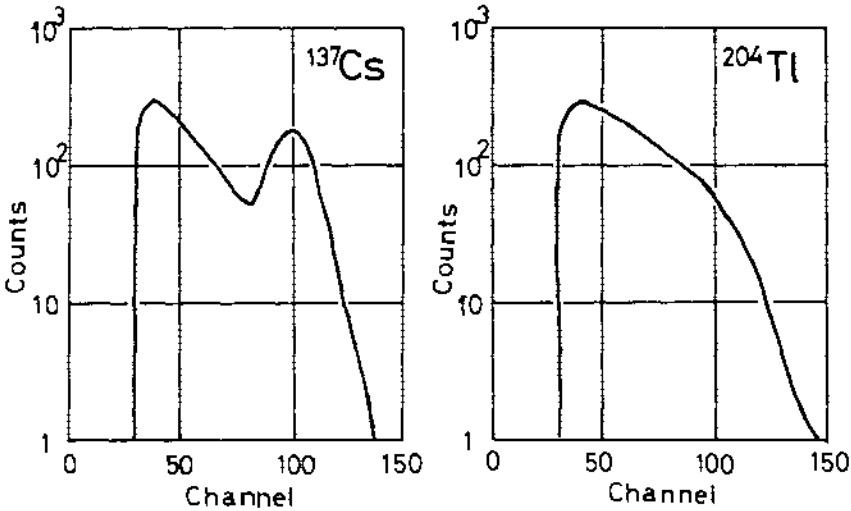

Fig. 3 Beta-ray Spectra of ^{137}Cs and ^{204}Tl measured by the Absorbed Dose-rate Meter

The absorbed dose-rates of the samples were measured by following steps;

(1). Counting the background (C_B) for 60 minutes,

(2). Counting the sample including the background (C_{BS}) for 60 minutes, and

(3). Calculation of the net counts (C_S)

$$C_S = C_{BS} - C_B$$

and the absorbed dose-rates (D)

$$D = (C_S/60) * (\text{conversion factor})$$

An experiment was done to know the relation between the absorbed dose-rate and the depth of sands. The sample prepared for the experiment was adsorbed K-40 on the sand particles. As shown in Fig. 4, the result presents that the absorbed dose-rate is independent with the depth of sands when it is greater than 0.4 g/cm^2.

The dominant natural radionuclide contributing to the absorbed dose is considered to be K-40 (4), so that we compared the measured values and the calculated K-40 dose-rates in the sands. The results of this experiment is shown in Fig. 5 and presents a good correlation.

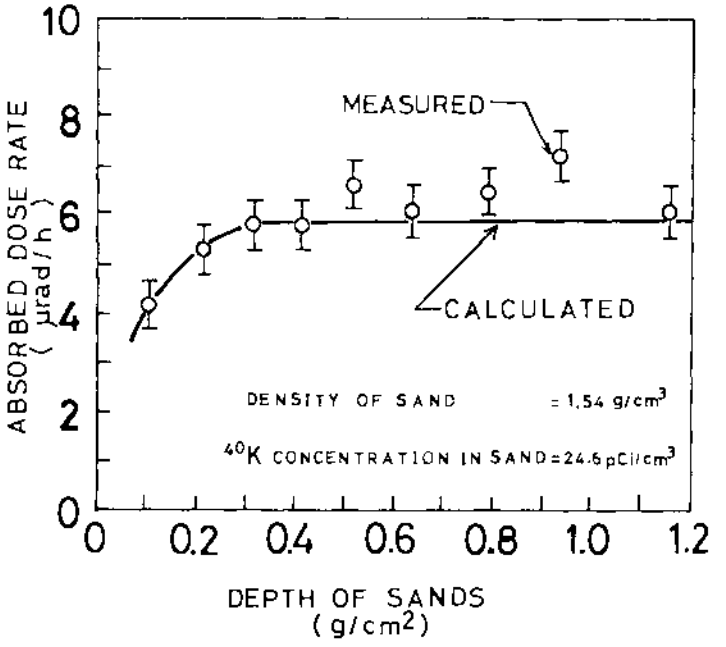

Fig. 4 Change of Absorbed Dose-Rate with Depth of Sands. "CALCULATED" is the Dose Rate by the Beta-ray from Potassium-40 Adsorbed on Particles of Sand.

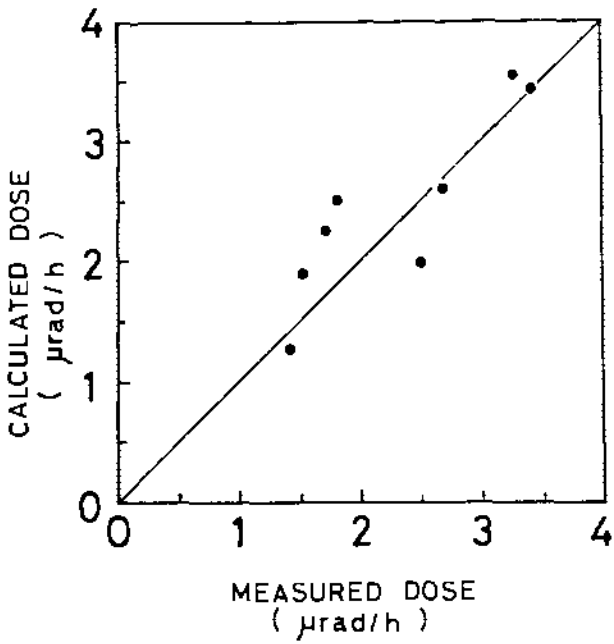

Fig. 5 Correlation between Calculated and Measured Absorbed Dose Rates. "CALCULATED DOSE" is the Dose Rate only by K-40.

SUMMARY

(1). This absorbed dose-rate meter has relatively high stability and sensitivity to measure skin doses.
(2). The measured absorbed dose-rate and the calculated dose-rate has shown a good correlation.
(3). Further study must be made on;
- the influence of the particle size distribution of sands,
- the influence of the water content in the sands, and
- the contribution of the natural radionuclides other than K-40.

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AVANTAGES PRESENTES PAR L'INTRODUCTION INDUSTRIELLE DE L'INFORMATIQUE DANS LA SURVEILLANCE CENTRALISEE DES NIVEAUX DE RAYONNEMENTS.

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L'expérience acquise au CEN/SACLAY depuis plus de vingt ans montre que dans les installations nucléaires importantes, la surveillance des risques d'irradiation et de contamination radioactives doit être permanente et centralisée. La technologie a, évidemment, considérablement évolué au cours du temps, et récemment, le Service de Protection contre les Rayonnements du CEN-SACLAY a défini un type de matériel entièrement nouveau, faisant notamment appel aux techniques numériques. Une réalisation pilote de 20 voies de mesure a été décrite lors de l'exposition NUCLEX 75 /1/ et les résultats expérimentaux obtenus par la méthode de traitement numérique pour le contrôle de la radioactivité de l'air ont été exposés lors d'un colloque de l'AIEA /2/. La présente communication concerne une réalisation industrielle de 120 voies de mesure assurant la surveillance effective de l'usine de préparation de radioéléments du CEN-SACLAY.

PRESENTATION DU MATERIEL.

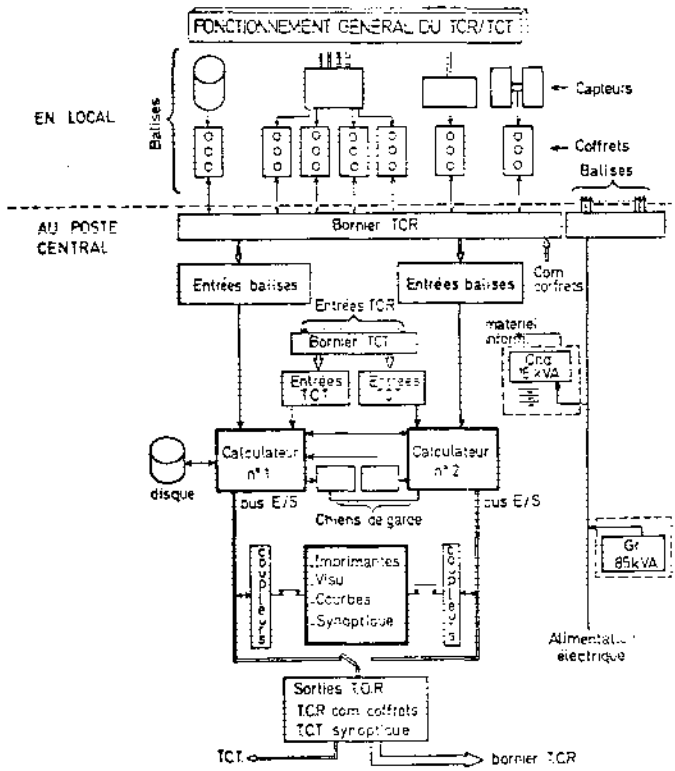
La figure jointe décrit le fonctionnement général de l'ensemble regroupant le "Tableau de Contrôle des Rayonnements (T.C.R.) et le Tableau des Contrôles Techniques (T.C.T.) de l'installation. Cet ensemble comprend :

- 62 voies de mesure pour le contrôle de la contamination de l'air par les aérosols.
- 54 voies de mesure pour le contrôle de l'irradiation externe.
- 4 voies de mesure pour le contrôle de la contamination de l'air par les gaz.
- 279 entrées de contrôles techniques (portes, niveaux de cuves à effluents, ventilations).

Le dispositif de mesure est bâti autour de deux calculateurs identiques, l'un d'eux ayant le rôle de calculateur de mesure. Tous les capteurs qui ont leur électronique propre, délivrent le même signal impulsionnel standardisé de telle sorte qu'une banalisation des voies de mesure est obtenue sur toute la chaîne, capteur exclu bien entendu. Au centralisateur, les informations d'entrée sont distribuées, en parallèle, aux deux calculateurs qui les traitent simultanément et de manière indépendante. Seul un système de surveillance mutuelle des calculateurs par "chien de garde" permet une commutation automatique à tout moment du calculateur maître (celui qui dispose des périphéries) vers le calculateur esclave.

/1/ H. JOFFRE - NUCLEX - Bâle, octobre 1975

/2/ H. VIALETTES and al. - IAEA-SM-217/13

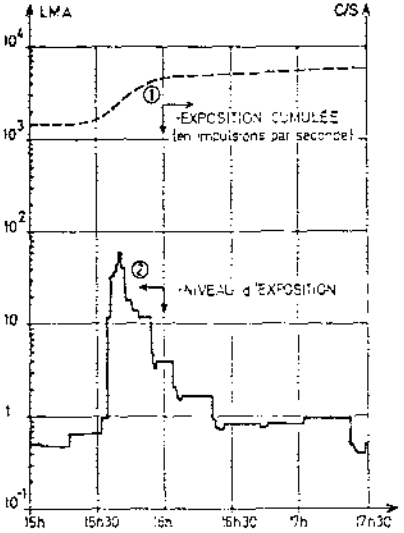


AMELIORATION APPORTEES PAR UN TEL SYSTEME.

Qualité de l'information : Les indications brutes des détecteurs sont converties, par le calcul, en niveaux d'exposition exprimé en nombre de LMA et en expositions cumulées exprimées en nombre de LMAh . Cette conversion tient compte de paramètres d'exploitation tels que facteur de correction de position du capteur, facteurs de rendement de filtration et de détection et facteur de radiotoxicité ; elle nécessite des algorithmes de traitements spécifiques pour chacun des deux types de voies de mesure ; elle aboutit dans tous les cas à des résultats homogènes et plus complets puisqu'elle permet d'exprimer d'une part le niveau instantané en rem/h et en CMA, et, d'autre part, l'exposition cumulée en rem et en CMAh. Grâce à ces résultats, la signalisation locale est relative soit à un débit de dose soit à une activité volumique dans l'air de sorte qu'elle est directement reliée, dans les deux cas, à une limite dérivée des recommandations de la CIPR,et qu'elle a donc une même signification vis-à-vis des risques pour les travail - leurs.

Dans le cas de la contamination atmosphérique, l'algorithme de calcul permet, à partir de l'activité déposée sur un filtre fixe, d'évaluer au mieux - c'est-à-dire en temps réel et compte tenu des fluctuations statistiques - les concentrations de ces radionucléides dans l'air. Par conséquent, le système décrit permet d'assurer le suivi de l'évolution d'une contamination accidentelle, en particulier

d'observer la phase de décroissance, sans qu'aucune intervention sur le dispositif de prélèvement ne soit nécessaire (figure ci-contre). Parallèlement, il est clair que le système informatisé est rapidement disponible et permettrait de mettre en évidence une deuxième bouffée de contamination, même à un niveau de quelques CMA, intervenant peu de temps après la première ;avec le système classique, au contraire, le comptage correspondant à cette deuxième bouffée risquerait de ne pas pouvoir être distingué des fluctuations statistiques du comptage correspondant à l'activité déjà déposée sur le filtre de prélèvement.



① Enregistrement d'un système classique
② Enregistrement du système informatisé

GESTION DES RESULTATS.

Outre les avantages bien connus qu'apporte l'utilisation de l'informatique dans la gestion en temps réel des résultats et dans leur archivage, la conception du système permet de ne pas encombrer, d'informations sans intérêt, les supports d'exploitation directe ou différée, comme l'étaient les rouleaux d'enregistrement des anciens systèmes. Pour autant, l'exploitant n'est privé d'aucune information grâce à des possibilités très riches de dialogue avec le système informatique mais surtout grâce aux trois dispositions complémentaires suivantes :

- la visualisation permanente des niveaux d'exposition (en clair et sous forme d'histogramme) et des états de toutes les voies de mesure.
- l'enregistrement permanent sur disque magnétique de toutes les informations des voies de mesure ; cet enregistrement permet de reconstituer intégralement l'historique de l'une quelconque des voies de mesure sur simple commande opérateur.
- le tracé en temps réel des courbes de variation des niveaux d'exposition par commande manuelle ou par commande automatique à la suite du dépassement d'un seuil de consigne.

SOUPLESSE D'UTILISATION ET FACILITES D'EXPLOITATION.

La standardisation des voies de mesure jusqu'au niveau du capteur permet de brancher, sans aucune modification mécanique ni électronique, n'importe quel type de capteur sur n'importe quelle voie. La seule intervention consiste à taper au clavier de commande la nouvelle identification de la balise de façon que ses informations soient traitées par le programme ad hoc.

D'autre part, grâce à la présence d'une source de contrôle de faible activité au niveau des capteurs, le contrôle de bon fonctionnement est global et permanent. Le traitement numérique prend en compte le bruit de fond introduit par cette source.

Fiabilité.

- La redondance des fonctions vitales (acquisitions, traitement et transmission des alarmes) et l'autocontrôle du système informatique en font un système sûr. De ce point de vue, l'arrêt d'un périphérique ou d'un calculateur ne perturbe en rien le système de contrôle continu des rayonnements ; en effet, le basculement d'un calculateur sur l'autre avec tous les périphériques est automatique, de même que le basculement des tâches d'une imprimante sur l'autre ; pour les consoles de visualisation, il est possible, sur simple commande opérateur, de basculer les informations de l'une sur l'autre, le changement de format étant automatique.
- En cas de panne sur le circuit d'alimentation générale, l'ensemble du système décrit ici est secouru par un groupe électrogène de 85 kVA, le matériel informatique étant lui-même secouru par un moduleur de 15 kVA.
- Sur une période de fonctionnement de 10 000 heures, on a observé une indisponibilité de 0,5 heures, ce qui confirme les calculs de fiabilité effectués par le fournisseur qui aboutissaient à 0,72 heures d'indisponibilité pour 10 000 heures de fonctionnement.

Prix de revient.

En janvier 1977, le système complet décrit ci-dessus a coûté, pour la partie TCR, 5 millions de francs se répartissant en 37 % pour les capteurs, 16 % pour les coffrets de signalisation, 31 % pour le centralisateur (matériel + logiciel), 16 % pour l'installation et la maîtrise d'oeuvre. Il est intéressant de noter que :

- ce coût est inférieur d'environ 30 % à une installation identique en version classique, sans traitement de l'information.
- par la réunion du TCR et du TCT, qui permet notamment une gestion commune, le prix de revient du TCT a été notablement diminué.

CONCLUSIONS.

Le contrôle centralisé des rayonnements par des méthodes de traitement numérique apporte des améliorations fondamentales pour la surveillance d'une installation nucléaire d'une certaine importance. Ces améliorations sont sensibles, tant sur le plan de la qualité de l'information, qui joue directement sur la radioprotection des travailleurs, que sur le plan de l'exploitation centralisée qui intervient directement sur la qualité des prestations de radioprotection. Etant donné qu'il est démontré que la fiabilité d'un tel système est excellente et que son prix de revient est sensiblement inférieur à celui d'un système classique donnant des informations beaucoup plus simplistes, il nous paraît clair que la généralisation s'impose pour toutes les installations.

Une variante de ce système existe en version décentralisée pour ce qui est du traitement de l'information et de la signalisation. Il est encore trop tôt pour la comparer valablement au système décrit ici, car elle n'a pas encore été appliquée à une réalisation vraiment industrielle et sa fiabilité n'a pas fait, non plus, l'objet des études mentionnées ici.

A MICROPROCESSOR BASED AREA MONITOR SYSTEM FOR NEUTRON AND GAMMA RADIATION

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At the MPI-Heidelberg two Tandem van de Graaff (6 and 12 MV)and one postaccelerator are in operation. The 10 MV postaccelerator which consists of independently phased spiral resonators is coupled to the 12 MV Tandem. Depending on the specific ions accelerated, their energy and current, the dose rates range from nondetectable values up to some 10 rem/h. High-energy neutrons contribute 80 to 90% of the stray radiation total rem dose and gamma rays make up the balance. The extreme dose rate situations request flexible radiation protection regulations. They have to ensure that the absorbed dose of the personnel is kept as low as achievable without restricting the experimental operations more than necessary and that at beamtimes with high dose rates accidental exposures are strictly avoided. Since the radiation regulations are basedon dose rate measurements of the area monitor system it has to operate in a wide dynamic range at a high reliability level.

The old system was no longer able to fulfil this requirement. There have been too many failures of the electronics. The choice for its replacement is a system consisting of individually microprocessor controlled area monitors which are connected to a microcomputer. A system witnout the monitor processors which would have been possible, too, was not chosen because of insufficient reliability. Failures of the central microcomputer would have blocked the whole monitor system. A conventional hardware solution would have been less secure and at least two times more expensive.

In Fig. 1 a block diagram of the complete new system is shown. The area monitors as well as the microcomputer have been developed on the basis of the 16 bit TMS 9900 microprocessor. The pulses of the neutron rem counter and of the gamma counter are each fed into 8 bit binary counters of the area monitor microprocessor system. Fig. 2 shows how the pulses are processed by the developed software. All operations are executed in the work space register of the TMS 9900. The overflow rate of the counters is stored in incremental registers. After 1 second the contents of the counters are read into one register and then added to the product of the overflow rate by 8 bit in another register. Now, the γ-count rate is normalized to give the same relative dose equivalent unit as the neutron count rate and finally converted into mrem/h. The γ- and neutron dose rates and their sum are stored in the memory. A comparison of the sum with two thresholds identifies three different dose levels. For failure surveillance the results of the γ- and the neutron measurement are integrated for a long enough time to judge from background statistics if there are too few counts. A subprogram converts the data from the hexadecimal into the decimal notation.

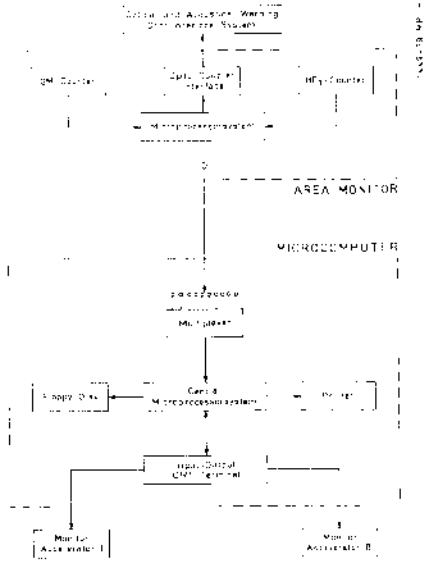

Fig. 1: Block diagram of the area monitor system.

Dose rate and failure indications are transmitted to different warning installations and to the door interlock system as shown in Fig. 1. All in- and output signals of the microprocessors are transmitted through opto-couplers. Each channel can process count rates up to 65000 per sec, which corresponds to about 30 rem/h for γ-rays and neutrons, respectively . The whole area monitor as encircled in Fig. 1 is housed in a box with a transparent front cover.

The dose equivalents of the individual area monitors are read by the central microcomputer via a multiplexer. Here, the data are processed for printing, display on TV monitors and recording on floppy disk. The printed data give information on the near past, whereas long time records are taken from the floppy disks. The TV monitors, with a turnover time of 10 sec, display the actual dose rate situation at the radiation protection office and the control rooms of the Tandems. Here the high dose level is also indicated on inspection panels of the door interlock systems. Fig. 3 shows the inspection panel of the 12 MV Tandem and the postaccelerator. 12 monitors survey this area. If a high dose threshold is surpassed, the red LED within the radiation sign lights up.The doors adjacent to that area can be locked either by switches on the console or automatically. Their state (open, locked and closed, locked but open) are also signaled by different LED. In the automatic mode the beam is interrupted if a door that should be

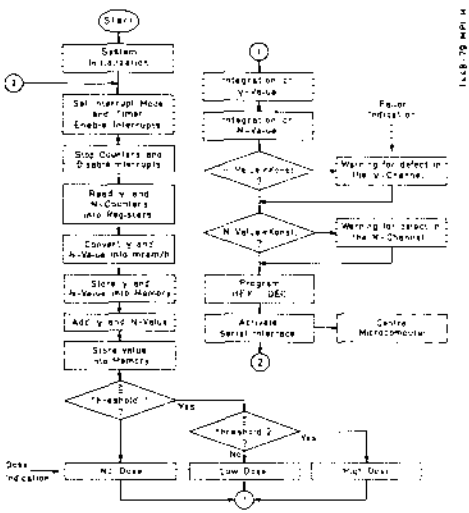

Fig. 2: Flow chart of the area monitor microprocessor program.

closed stays open for more than 2 minutes. In the experimental area, the low and high radiation levels are indicated by acoustic and light signals inside the individual target places and by a kind of traffic lights at the entrance doors to these places. Green light "no dose", yellow light "low dose", red light "high dose".

The microprocessor-based area monitor system provides high flexibility, which is desirable in view of the rather frequent changes in accelerator experiments.

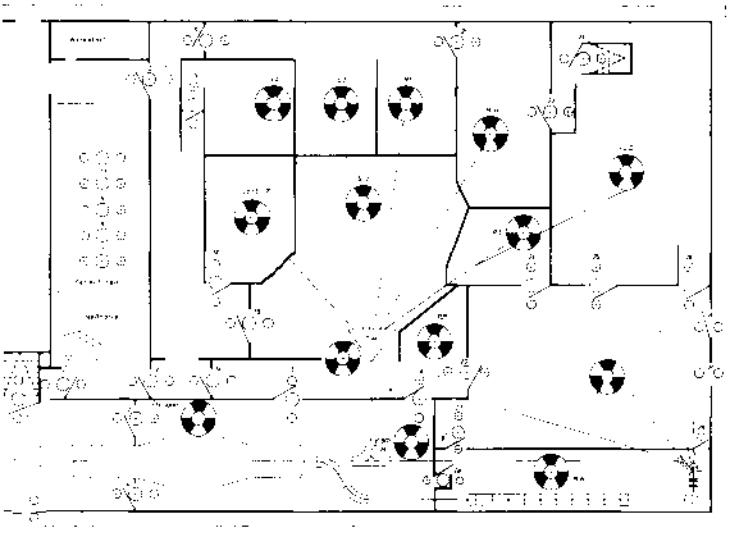

Fig. 3: Inspection panel of the MP Tandem.

ACKNOWLEDGEMENT

We are very much indebted to W. Heinecke and W. Schreiner for their help and fruitful discussions.

USE OF INFORMATIC FOR RADIATION CONTROL PANELS

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V.MAI - Merlin Gerin Provence (FRANCE)

Radiation control panels (R.C.P.) are systems which enable irradiation and contamination risks to be quantitatively determined and monitored.

Such systems automatically control the immediate activation of warning devices (audible and visible in the zones concerned so as to alert personnel at their working stations).

For a few years now, the CEA has been developing a programmed system generation of radiation control panels.

R.C.P. can be divided in to three main elements :

1°) a series of monitoring stations

Each station monotoring a zone consists of :

. a detector adapted to the characteristics of the radiation to be detected together with part of all of the associated electronics (power supply, amplifier...)
. an audible and visible alarm unit alerting personnel of the risks to which they are exposed.

Measured exposure levels fall into four scales as seen in the table of figure 1, which also indicates the corresponding visible and audible signals.

2° a central station

All the radioprotection data recorded converge into this station In general, an operator is posted here, whose responsibility is to monitor, and when necessary, record, the risks encountered at each individual station.

3° more or less sophisticated information processing facilities (between 1° and 2°)

A recent orientation in the design of the R.C.P. programmed system generation is to locate totally autonomous units in the various different zones. These units provide signals when given thresholds expressed in LMA or CMA are exceeded. In this way safety and availability are improved.

This unit, which together with it s detector constitutes an autonomously operating monitoring station, can be connected to a centralizing unit (e.g. minicomputer).

DESCRIPTION OF THE MICROPROCESSOR PROCESSING AND SIGNALLING UNIT (PSU)

The monitoring station of a zone figure 2 consits of :

- a detector and its associated electronics which delivers a standard pulse for all types of detector.
- a processing and signalling unit assuring the following functions :
  . acquisition of information detected by the detector
  . processing of this information to determine LMA dose rates, while taking into account parameters such as the radiotoxicity of the radioelement, which can be memorized in the unit

. generation of different output signals to be transmitted to the central station (100 to 200 meters)
. warning outputs for synoptic
. analogic dose rate output for recordings
. asynchronous line output for centralizer

- upon cyclic interrogation by minicomputer, the unit transmits :
  . dose rates
  . the threshold exceeded by the unit
  . the state of the unit
  . the memorized values of the thresholds and coefficients

The correct operation of each station is verified :

- the complete system, by measurements with a permanent low activity control source, which triggers correct functioning threshold
- the quasi-totality of the system with periodic tests (generator simulating levels)

An MC 6800 microprocessor is used.
A maximum of 8K REPROM and 2K RAM memories is available.

DESCRIPTION OF CENTRALIZE

At the central station, all or part of the following facilities are available :

- minicomputer which acquires (via asynchronous lines) informations from units for determining cumulative doses and different logs
- a detailed visual synoptic providing, for each station, the number of the threshold exceeded
- recorders

The last two devices are independent from the computer and can constitute :

. either, by themselves, the centralizer
. or, a back up system for the computer in the case of failure.

A first realisation of this system is being used to control an effluent treatment plant. 30 units are connected to a MULTI 6 minicomputer.

RADIATION LEVEL DISPLAY

4 Measurement scales (according to regulations)

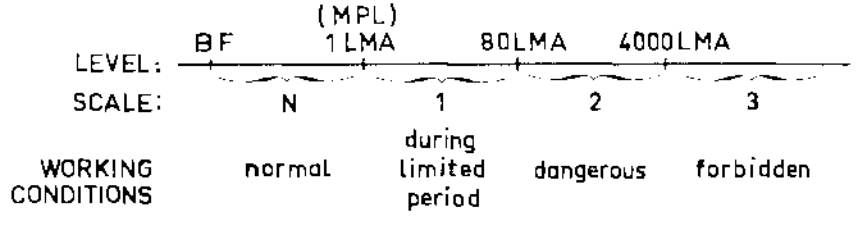

Fig. 1

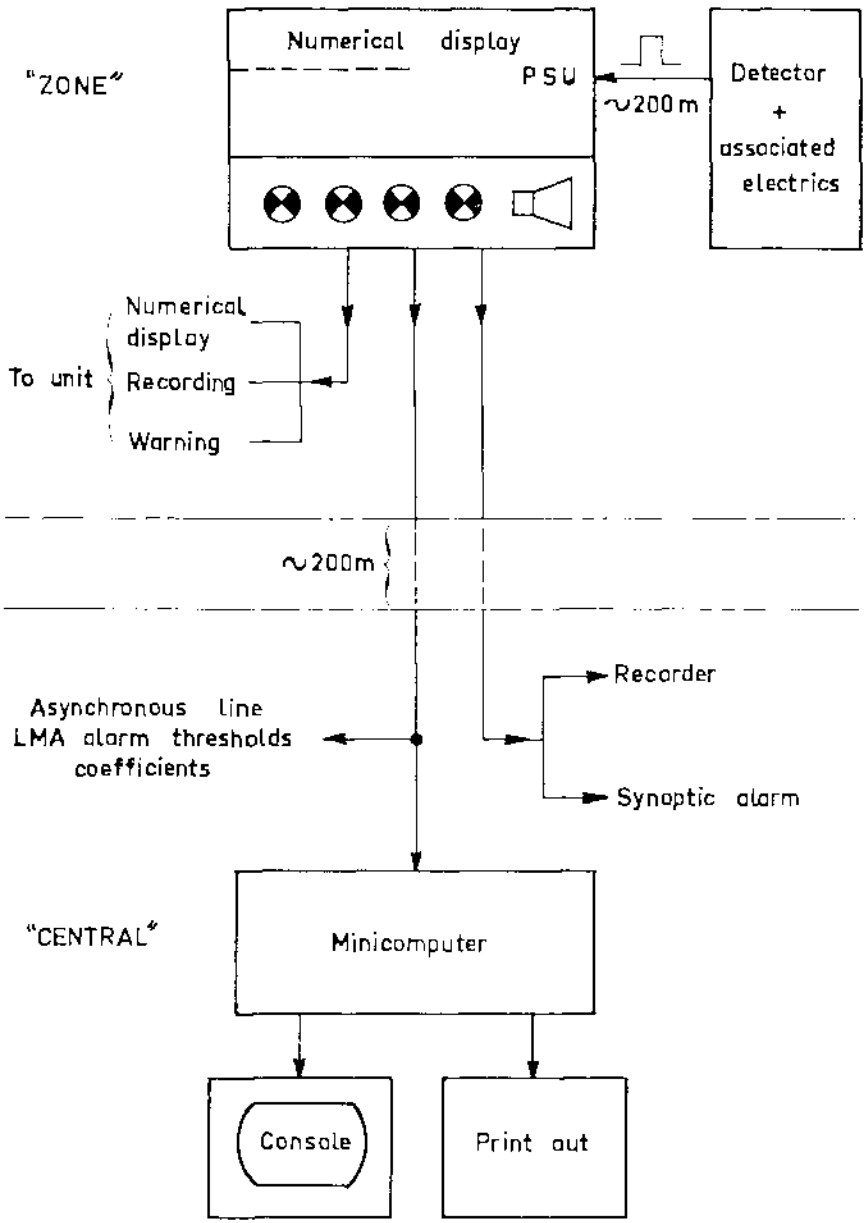
"ZONE"
Numerical display
PSU
~200 m
Detector + associated electrics
To unit
Numerical display
Recording
Warning
~200m
Asynchronous line
LMA alarm thresholds
coefficients
Recorder
Synoptic alarm
"CENTRAL"
Minicomputer
Console
Print out

Fig. 2

NUCLEAR SHIELDING ANALYSES FOR AN INTENSE NEUTRON SOURCE FACILITY

J. Celnik - Burns and Roe, Inc., USA

1. INTRODUCTION

This paper summarizes nuclear shielding analyses applicable to fusion, and fusion related, facilities. The analyses were performed during the design of an Intense Neutron Source Facility, to provide an experimental neutron irradiation facility yielding a neutronic environment similar to that encountered in a fusion power reactor.

The analyses included:

- bulk shield
- skyshine
- various generic and specific penetrations, single and multi-legged
- source cell door design.

All results are based on a neutronic environment generated by a D-T source yielding 3×10^{15} -14 MeV neutrons/second. Analyses included the effect of secondary gamma rays produced by the interaction of primary and scattered neutrons with air and the proposed shielding materials.

2. BULK SHIELD ANALYSIS

The adequacy of an eleven-foot concrete wall, composed of an inner layer of one-foot borated gypsum and ten feet of ordinary concrete to meet the design dose rate criteria was calculated using ANISN. The ANISN computer program calculates radiation transport in a one-dimensional geometry via the discrete ordinates method. This computational technique is commonly used for the solution of deep penetration problems.

To evaluate the computational uncertainty the results were compared using:

- different cross section data sets
- increasing the order of the Legendre expansion of the scattering cross section with a comparable increase in the angular quadrature representation.

Some highlights of the analyses are:

a) An outer wall of one-foot borated gypsum (to minimize activation) followed by ten feet of concrete yields a dose rate of 135 mRem/occupational year.

b) The total dose rate drops rapidly when the detector is placed off the source axis.

c) Use of the CASK 22 neutron-18 gamma ray coupled cross section data set underestimates the neutron leakage by a factor of about three and the total dose rate by more than a factor of two, as shown in Table 1.

TABLE 1: Dose Rates Outside 1' Borated Gypsum and 10' Concrete Wall

Cross Section Data Set	Dose Rate (mRem/hr) Neutron	Gamma Ray	Total
CASK (22N + 18G)	5.25-3	3.63-2	4.16-2
DLC-31 (37N + 21G)	1.55-2	7.30-2	8.85-2

d) Table 2 shows the effect of using a more detailed description of the anisotropic scattering, for an eleven foot concrete shield wall.

TABLE 2: Dose Rate Outside 11' Concrete Wall

Cross Section Data Set	Dose Rate (mRem/hr) Neutron	Gamma Ray	Total
P_3S_8 - DLC 31	1.31-2	7.80-2	9.11-2
P_5S_{12} - DLC 27	6.44-3	6.11-2	6.76-2

3. SKYSHINE ANALYSIS

Some highlights of the analysis include:

a) A source cell roof of eight feet concrete is required to reduce the skyshine dose rate contribution to acceptable levels.

b) For proper skyshine evaluation, the leakage spectrum on top of the source cell roof should be calculated using a two-dimensional discrete-ordinates code. A one-dimensional code will under-estimate the leakage and hence the skyshine contribution by about a factor of ten.

c) The 2-D leakage results can be used as the source for the full 3-D Monte Carlo analysis of the skyshine contribution.

4. PENETRATION ANALYSES

An investigation was done to estimate radiation streaming effects for a variety of penetration configurations. The results may be used to design general penetration layouts, which could then be analyzed in greater detail and accuracy. All analyses were performed with the Monte Carlo method. Preliminary analyses were done with MORSE CG program using the CASK cross section data set, with more detailed results obtained with the LASL MCNP program using the LASL recommended cross section data.

All penetrations analyzed were for a ten foot ordinary concrete wall. Some of the general conclusions are:

a) The line-of-sight uncollided dose rate dominates for penetrations directly facing the source;
b) The scattered dose rate is large (> 1 Rem/hour for a source of 10^{15}-14 MeV neutrons/second) even for a 12 cm penetration angled 45^{o} through a 10 foot wall;
c) Placement of large (two foot diameter) penetrations in an extreme corner of a 12 foot X 12 foot cell is not adequate to reduce streaming to acceptable levels;
d) Use of a multi-legged, non-coplanar design can reduce the exit dose rate, for a large diameter penetration, by several orders of magnitude;
e) Decreasing the penetration size from 2' X 2' to a 1' X 1' opening will decrease the dose rate at the end of the first leg by about a factor of 2, and decreases it by a factor of about 100 at the exit of the four-legged non-coplanar penetration;
f) The above comparison is also valid when an 18 inch (45.7 cm) radiation flux trap is included at the end of the first leg;
g) For some configurations, use of a flux trap will increase the total attenuation by about a factor of two;
h) When using a flux trap, the dose rate at a point on the outer wall surface opposite the first leg may be significantly higher than the dose rate at the exit face of the last leg. A magnetite concrete plug may be used to match the dose rate through the first leg with the dose rate at the penetration exit.
i) The dose rate at the penetration exit, for the multi-legged penetrations analyzed in this study, is due primary to secondary gamma ray leakage. Inclusion of an 8% borated polyethylene liner will decrease gamma ray leakage by a factor of about 100 and the total dose rate by a factor of about 15.

5. SOURCE CELL DOOR DESIGN

Access to the source cell is provided by a system of hydraulic doors. The inner door, adjacent to the cell, is composed of a one-foot liner of borated gypsum followed by 3'-3" of magnetite concrete. The outer door, consisting of 3' of magnetite concrete, contains a lead glass viewing window. The doors are stepped to minimize streaming between the doors and the wall in which they are located. In addition, a steel plate is placed in the cavity beneath the door to eliminate the potential for radiation scattering into the cavity and then re-emerging in front of the door.

TABLE 3: Results of Source Cell Door Analysis

Shield Configuration*	Total Dose Rate (mRem/hr)
Bulk wall (6'-6" mag. concrete)	0.8
Through both doors (6'-3" mag. concrete)	1.4
Through window (3'-3" mag. concrete + 3' Pb-glass)	9.0
Through alternate window design	5.5
Through window lining (3'-3" mag. concrete + 3' Fe)	18.5
Window lining shielded by oil layer	3.6

*All shield configurations included a liner of one foot of borated gypsum.

The results indicate that the dose rate behind the wall and behind the doors meet the dose rate criterion of 10 Rem/occupational year (equivalent to 5 mRem/hr). However, use of a "standard" lead glass configuration will lead to a significant hot spot. Use of an alternate lead glass window composition can reduce the radiation streaming to acceptable levels.

In addition, design of the window frame should include provision for inclusion of a one-foot oil layer in front of the frame to reduce neutron streaming.

6. CONCLUSIONS

This paper presents highlights of nuclear shielding analyses performed during the design of an intense neutron source facility. It is expected that the results, though preliminary in some areas, would be useful in the design of similar fusion-related facilities as well as in the conceptual design of a fusion power reactor complex.

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SPECTRA, DIFFERENTIAL ALBEDO AND SHIELDING DATA FOR BREMSSTRAHLUNG SCATTERED FROM COMMON SHIELDING MATERIALS

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INTRODUCTION

In the design of radiation shields for X-ray machines and electron accelerators not only the primary radiation but also the radiation scattered from irradiated material must be taken into account. The thickness of shielding barriers against secondary radiation can be calculated if the intensity and the quality of the radiation are known. For radiation protection purposes the scattered intensity is most conveniently described in terms of the differential albedo:

$$d\dot{J}_s = A_{JX}(E, \theta_o, \theta, \varphi)\, \dot{J}_o \frac{df \cos\theta_o}{r^2}$$

A_{JX} differential exposure albedo

$d\dot{J}_s$ scattered exposure rate at a distance r from the surface area df of the irradiated material

E energy of incident photons

θ_o angle of incidence; θ polar angle of exit

φ azimuthal angle of exit

θ_s scattering angle $\theta_s = \pi - \theta_o - \theta$ for $\varphi = 0$

In the case of very high energy photons it is more appropriate to replace the exposure by the absorbed dose in a tissue equivalent material.

Albedo data were measured for bremsstrahlung with maximum photon energies in the ranges from 100 keV to 400 keV and from 10 MeV up to 35 MeV. In the low energy region spectra of scattered radiation were measured with a semiconductor detector. From these spectra attenuation curves were calculated using known broad beam transmission factors for monoenergetic photons. For high energy bremsstrahlung (10 MeV to 35 MeV) the quality of scattered radiation was determined by direct attenuation measurements.

EXPERIMENTAL

Bremsstrahlung with maximum photon energies between 100 keV and 400 keV was produced by means of a conventional industrial 400 kV X-ray machine (inherent filtration: 4 mm Al). The beam profile of the primary radiation was defined by a series of matched lead collimators.

The size of the irradiated area df was determined by exposing an X-ray film on the surface of the scattering material. Ion chambers with a flat energy response were used for measuring the exposure rate of primary and scattered radiation. The differential exposure albedo of ordinary concrete, barytes concrete, iron and lead was determined for normally incident radiation ($\vartheta_o = 0$) as a function of the polar angle of exit ϑ as well as for the special geometry $\vartheta_o = 45^o$, $\vartheta = 45^o$, $\varphi = 0^o$.

High energy bremsstrahlung was produced using a 35 MeV travelling wave linear accelerator. The electron beam is deflected by an achromatic magnet system and focused on a tantalum bremsstrahlung target (7 mm thick) where it is completely absorbed. The cross section of the photon beam is defined by several steel collimators which are coaxially mounted in a channel through the concrete wall (3 m thick) between the accelerator hall and the experimental area. The dose rate of the incident and the scattered radiation is measured with ion chambers surrounded by a sufficient amount of tissue equivalent buildup material.

RESULTS AND DISCUSSION

Some selected albedo values are compiled in Fig. 1 to Fig. 3 and Table I for normally incident radiation.

In the low energy region there is a complicated relationship between the exposure albedo, the radiation energy and the scattering material (1), but for $\vartheta_o = 0$ the scattered dose rate always increases with increasing scattering angle ϑ_s (Fig. 1). Between 10 MeV and 35 MeV the differential albedo of materials with low mean atomic number (ordinary concrete, brick) decreases with increasing energy (Fig. 2). In the case of lead however (Fig. 3) higher scattering intensity is observed at higher energies because at high primary photon energies the main contribution to the albedo of high-Z material is due to the production of secondary bremsstrahlung which rapidly increases with energy (5).

Fig. 4 and Fig. 5 show photon spectra of low energy bremsstrahlung scattered from ordinary concrete. Whereas the quality of radiation scattered from low-Z material is determined by compton scattering, the spectra from high-Z materials (barytes concrete, lead) show a large contribution of characteristic X-radiation to the total scattered intensity. As an example calculated attenuation curves for low energy photons scattered from ordinary concrete are plotted in Fig. 6. They reasonably agree with measured values (3). Similar shielding data were calculated for several combinations of scattering and shielding materials.

TABLE I. Differential exposure albedo for low energy bremsstrahlung normally incident on common shielding materials ($\vartheta_o = 0$)

tube voltage U [kV]	ϑ_s [deg.]	scattering material		
		barytes concrete	steel	lead
100	110	$1{,}5\cdot10^{-2}$	$2{,}6\cdot10^{-3}$	$7{,}7\cdot10^{-3}$
	135	$2{,}5\cdot10^{-2}$	$5{,}2\cdot10^{-3}$	$1{,}5\cdot10^{-2}$
	160	$3{,}1\cdot10^{-2}$	$7{,}5\cdot10^{-3}$	$2{,}0\cdot10^{-2}$
200	110	$1{,}3\cdot10^{-2}$	$3{,}7\cdot10^{-3}$	$9{,}5\cdot10^{-3}$
	135	$2{,}2\cdot10^{-2}$	$6{,}8\cdot10^{-3}$	$1{,}7\cdot10^{-2}$
	160	$2{,}9\cdot10^{-2}$	$1{,}0\cdot10^{-2}$	$2{,}1\cdot10^{-2}$
400	110	$9{,}3\cdot10^{-3}$	$6{,}1\cdot10^{-3}$	$7{,}1\cdot10^{-3}$
	135	$1{,}6\cdot10^{-2}$	$1{,}1\cdot10^{-2}$	$1{,}2\cdot10^{-2}$
	160	$2{,}1\cdot10^{-2}$	$1{,}5\cdot10^{-2}$	$1{,}6\cdot10^{-2}$

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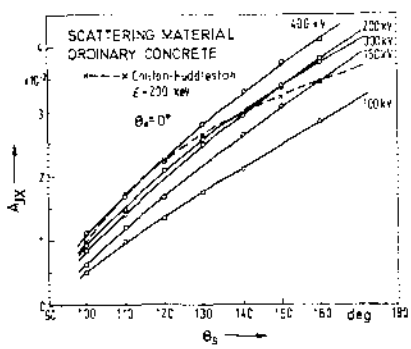

Fig. 1 Differential exposure albedo of ordinary concrete for low energy bremsstrahlung

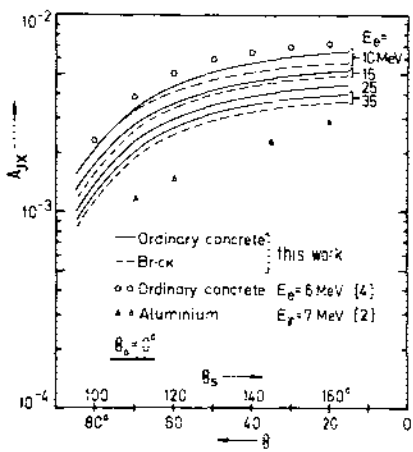

Fig. 2 Differential dose albedo of concrete and brick for high energy bremsstrahlung

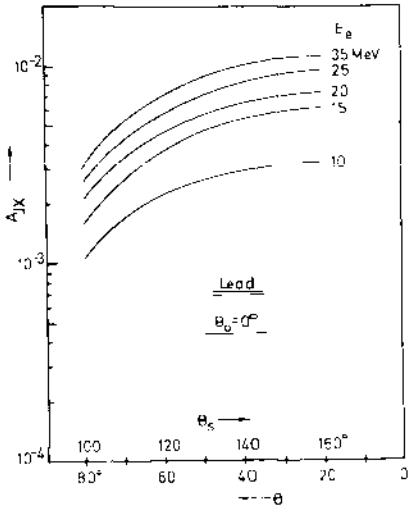

Fig. 3 Differential dose albedo of lead for high energy bremsstrahlung

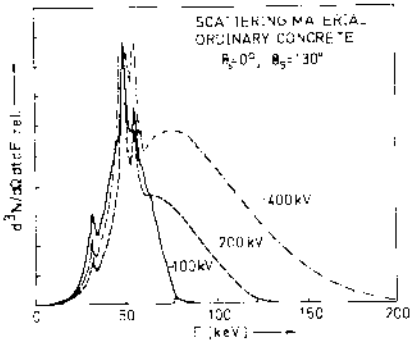

Fig. 4 Spectra of low energy bremsstrahlung scattered from ordinary concrete as a function of tube voltage

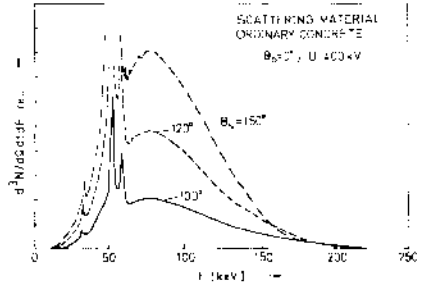

Fig. 5 Spectra of 400 kV bremsstrahlung scattered from ordinary concrete as a function of scattering angle

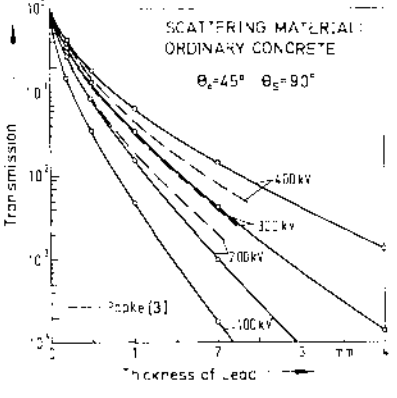

Fig. 6 Attenuation curves for scattered radiation. Scattering material: Concrete; shielding material: lead

"INTELLIGENT" RADIATION INSTRUMENTS

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Any activity involving the exchange, storage and analysis of information is likely to be strongly affected by the development of microprocessor technology. In the field of radiation protection we are concerned with information in the form of medical records, dose records, approved schemes of work and lists of classified workers. We are also concerned with the collection and examination of information from a wide range of radiation monitoring instruments. We are even concerned to an increasing extent with information being exchanged from one machine to another with only the most casual human interference. Further development of microprocessors is occurring and it seems possible that quite intelligent machines may be available in the next decade for such major tasks as the complete automation of nuclear waste management and perhaps also to assist in the eventual dismantlement of the present generation of nuclear power reactors as they come to the end of their useful life. In this paper a description is given of three straightforward applications of current microprocessor technology. These applications are characterised by the use of the microprocessor to impart a degree of intelligence to what would otherwise be conventional radiation detection techniques.

GEIGER COUNTER

We are all very well aware that the output from a geiger counter consists of a series of identical pulses. Furthermore, considerable background information with regard to the type of radiation sources involved is necessary to infer a value for the exposure dose rate from the observed counting rate. In the hands of an unskilled person a serious error can occur in estimating the radiation exposure. The arrangement described here turns over to the micro computer the task of computing the radiation dose from the observed counting rate.

Slide 1 is a schematic diagram of the system. Two identical geiger tubes are employed but in front of one of the tubes an absorber is placed which has the effect of causing the ratio of the counting rates on the two tubes to depend significantly on the effective gamma ray energy in the incident radiation. Each output pulse from either tube causes an "interrupt" to the main program on the micro computer which promptly postpones its main program and operates an auxiliary program which "services" the interrupt. The micro computer is programmed to keep a record of the number of pulses from each of the two geiger tubes and to form a ratio of these two numbers. The observed ratio in a particular situation is compared with stored data and in this way the incident gamma ray energy is estimated. The program then continues using additional stored data on the sensitivity of the geiger tube as a function of gamma energy to correct the counting rate (on the tube with no absorber) and thus obtain the required

dose rate. The dose rate thus obtained is then printed out in a convenient form together with auxiliary information which includes the maximum permissible exposure level, any special precautions which are relevant and the results of a check made by the micro computer into whether the entire system is functioning correctly.

SCINTILLATION COUNTER

The spectrum of pulses from a scintillation counter contains information concerning individual gamma energies. However, the cost of conventional multi channel pulse height analysers can be considerably more than the cost of the scintillation counter itself. It is possible to use a micro computer not only to carry out the function of obtaining the pulse height distribution but also to go a stage further and employ a pattern recognition program to identify the incident gamma ray energies and thus the radioisotope or isotopes involved.

Slide 2 is a schematic diagram of a typical arrangement. As each pulse emerges from the counter it is held for a period at its peak height and during this period an analogue to digital converter produces a binary code number proportional to the pulse height. This binary number is then latched on to an input port of the micro computer. At the same time the converter produces a control pulse which operates the interrupt request line on the micro computer. This interrupt causes the main program to stop temporarily and a service program comes into action. This service program reads the binary number at the input port and compares its value with a list of all possible values (channel numbers). When the matching stored value is found the content of a linked memory location is incremented by one. The interrupt terminates and the micro computer returns to the main program where it is ready for a further pulse. The contents of the linked memory locations form a conventional pulse height distribution. The period of counting is controlled by a timer in the micro computer. At the end of the counting period the micro computer changes to a program which is essentially a pattern recognition search using stored data on the pulse height distribution observed for different gamma ray energies. There are fairly strict limits on pattern recognition programs before the problem of identifying the gamma rays simply becomes too complex. However, in radiation protection situations it is usually the case that a limited number of radioisotopes are likely to be involved. For example, the laboratory concerned may use Cobalt-60, Caesium-137 and Iodine-125. Pattern recognition programs to identify these radioisotopes are quite straightforward.

The system described here can be left to itself to operate as a long-term monitor so arranged that it will generate, from within the micro computer, audible and visual warnings in a hazard situation. Thus, for example, it is possible to set the system to give priority to the detection of the radiation from Iodine-125 with its wellknown associated thyroid hazard.

MONITOR CALIBRATION BENCH

The calibration of radiation monitors is an important part of a radiation protection service. However, it is rather time consuming when carried out rigorously and there is a certain risk of radiation

exposure particularly in the calibration of monitors intended to measure high dose radiation. These practical difficulties are such that many radiation protection laboratories make only the most casual check on whether the monitor is performing satisfactorily.

The new legislation emerging in the European Community following the Euratom Directive and the response to the Directive from national governments emphasises the need for rigorous calibration of radiation monitors.

Slide 3 shows in schematic form an arrangement for monitor calibration based on micro computer control which is currently under construction in the authors laboratory. The arrangement is simple and straightforward. A large number of standardised radiation sources are mounted on a slim rod which passes between two heavily absorbing rods which have a small gap between them. The arrangement is such that only one source can be exposed at any one time within the gap. The radiation monitor to be calibrated is carried forward through a series of halt positions. At each halt the monitor output is read and stored together with a record of the monitor halt location and the source involved. After measurements have been made at each halt the source is changed by moving the rod and the set of measurements repeated. This procedure is continued until the output of the monitor has been recorded (for a group of sources relevant to that monitor) at all halt locations. The entire procedure is controlled by the micro computer which also stores the data and prints out the results in a very convenient form. Initially all of the sources are shielded by one of the thick absorbers so that the operator is not significantly exposed when placing the monitor to be tested in position. The procedure once started is fully automatic under the control of the micro computer which also provides an audible and visual warning while calibration is in progress.

A very important point is that since the procedure is under software control by a program stored in the micro computer it is quite easy to arrange to store a number of programs one for each type of monitor brought to the system for calibration. It is this flexibility coupled with the convenient facilities of the micro computer which renders it much more suitable than any other technique for this type of control. It is also the case that the cost of micro computer electronics is extremely low. The micro computer employed in all three applications described here cost only a few hundred dollars.

ON THE EXTENT OF EMERGENCY ACTIONS FOR THE PROTECTION OF THE PUBLIC AFTER ACCIDENTAL ACTIVITY RELEASES FROM NUCLEAR POWER PLANTS

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Emergency actions for the protection of the public are necessary only after those nuclear accidents which are termed "hypothetical" and which involve core meltdown and failure of the leak tightness of the containment. Such accidents have been analysed in the German Reactor Risk Study (1). The analysis showed, however, that only a small part of the core meltdown accidents analysed have the potential of releasing radioactive material into the atmosphere to an extent that - if they occur in combination with unfavourable meteorological situations - they require extended and immediate protective actions. Such unfavourable meteorological conditions mostly involve precipitation which is an effective means to wash out the radioactive material and to deposit it on the ground. For this reason the accident consequence calculation model applied in (1) has been analysed for the accuracy of the contaminated areas calculated.

AREAS COVERED BY PROTECTIVE ACTIONS

The protective action model applied in (1) has been developed in consistency with official German recommendations (2). It takes into account the specific problems of nuclear accidents (ground contamination, time scale of radioactive decay, efficiency of decontamination, etc.) as well as the high density of population in the F.R.G.. The areas are defined as follows, see fig. 1:

- area A, fixed size, evacuation in any case of core meltdown,
- areas B_1 and B_2, potential early fatalities (bone marrow dose due to external irradiation during 7 days exceeds 100 rad),
- area C, no early fatalities, but ground contamination too high for early decontamination (whole body dose due to external irradiation during 30 years exceeds 250 rad),
- area D, ground decontamination necessary and sufficient (whole body dose due to external irradiation during 30 years exceeds 25 rad).

Consecutive steps of the actions are:

- sheltering in buildings or basements in areas A and B_1,
- evacuation of area A after 8 hours,
- subsequent relocation of the population in areas B_1 and B_2, later relocation of the population in area C,
- immediate decontamination of area D,
- later decontamination of area A, B_1, B_2 and C,
- crop and milk interdiction.

ACTIVITY DEPOSITION MODEL

The external radiation from the ground is the main exposure pathway considered in the following investigation. The activity depo-

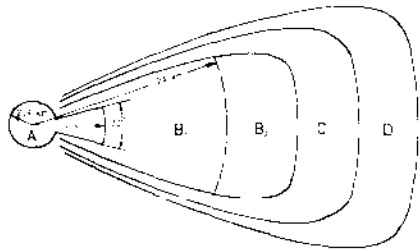

Fig. 1. Areas of protective actions.

sited on the ground is calculated according to conventional models (1). Dry deposition is characterised by the deposition velocity, wet deposition by the washout coefficient. The values of these parameters are choosen in accordance with (3), but the washout coefficient is assumed to depend on the precipitation intensity. The plume depletion is described by the "source depletion model:" the plume inventory is decreased by the deposited amount of activity.

Fig. 2 is a three-dimensional graph of the dose equivalent by external irradiation from the ground. The two pronounced peaks are mainly due to washout by rain during two separate periods of time. The graph is characteristic of the local dose rate distribution in cases where ground contamination is caused by washout.

The following feature of the consequence model tends to overpredict the area contaminated by washout: Activity concentration and ground contamination are calculated at discrete mesh points. The number of mesh points is limited for reasons of computer costs. At distances greater than about 20 km from the source the plume travel time from one grid point to the next exceeds one hour. If washout takes place during only part of that time, the activity being washed out is distributed nevertheless over the total interval. The resulting overestimate of the area and underestimate of the contamination would tend to sometimes overestimate and sometimes underestimate the areas B, C and D. It will be investigated below whether a serious error is involved by this feature of the model.

RESULTS

The area size distribution function of the B_1, $B_1 + B_2$ and C areas is shown in fig. 3. For this figure, the occurence of one of the accidents considered is assumed. The areas of most concern are B_1 and B_2, as here the external radiation is high enough to cause early fatalities, and therefore, the time available for relocation is limited. Such areas are mainly a consequence of a release characterised by the release category 1 (core meltdown and "steam explosion") or 2 (core meltdown and large containment leak). It should be added here that it is still questionable whether the release category 1 involving "steam explosion" is to be postulated in reactor risk assessments or not.

The calculation is based on the reference sample and on the test sample of weather sequences. A weather sequence contains the atmospheric transport and diffusion data (wind speed, diffusion category,

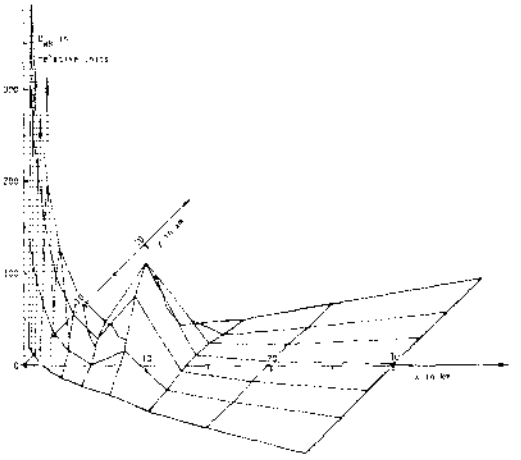

Fig. 2. Whole body dose D_{WB} due to external radiation from the ground.
Release category 2

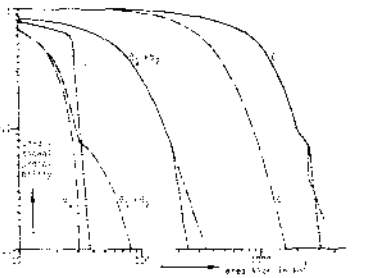

Fig. 3. Protective action area size distribution

release category 1
fine meshed grid
coarse " "
release category 2
coarse meshed grid

precipitation) of subsequent hours, starting at the presumed time of emission. These data are inputs to the plume dispersion calculation. The weather sequences chosen for calculation should adequately cover the variety of meteorological situations, but their number is limited. In (1) 115 reference weather sequences have been taken into account. To investigate the accuracy of the model, a test sample of 98 weather sequences has been chosen alternatively, each involving precipitation during at least one of the first hours. The area sizes have been calculated using two different spacings of the grid mesh-points:

a) fine-mesh grid (the travel time from one mesh pont to the next does not exceed one hour);
b) coarse-mesh grid (the reference spacing of 18 grid points up to a distance of 500 km).

The result is shown in tab. 1 and fig. 3. The area B_1 is not affected, as up to a distance of 24 km (size limit of area B_1) the reference grid is fine enough. The maximum of the areas $B_1 + B_2$ is overestimated by 45 % when using the coarse-mesh grid, whereas the average area $B_1 + B_2$ is slightly underestimated. The same is found for area C, but the overestimate is not as drastic. For area D the effect is negligible, since this area is mainly caused by dry deposition which is not affected by the grid spacing. In fig. 3 the distribution functions of the reference and the test sample (coarse-mesh grid) do not differ markedly.

Tab. 1. Emergency Action Areas

	A	B_1	$B_1 + B_2$	C
average area size				
fine-mesh grid	33 km^2	19.8 km^2	71.2 km^2	1101 km^2
coarse " "	"	"	66,4 "	1090 "
error by coarse grid	-	-	-7 %	-1 %
maximum area size				
fine-mesh grid	"	39.3 km^2	241 km^2	3180 km^2
coarse " "		"	349 "	3710 "
error by coarse grid	-	-	+45 %	+17 %
minimum area size				
fine-mesh grid	"	0	0	117
coarse " "	"	0	0	133

CONCLUSION

The models applied in reactor risk calculations are designed to give satisfactory answers in terms of the overall risk which is related rather to average than to maximum values of single parameters. Maximum values,however, are likely to be used as a guideline for emergency planning. It has been shown that the average areas covered by protective actions are only negligibly affected by model refinements, that the maximum of the area of fast relocation (area $B_1 + B_2$), however, is overestimated by a factor of about 1.5 unless a refined analysis is applied. Such methods will be studied in more detail in the second phase of the German Reactor Risk Study.

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AN APPLICATION OF COST-EFFECTIVENESS ANALYSIS TO RESTRICT THE DAMAGE CAUSED BY AN ACCIDENTAL RELEASE OF RADIOACTIVE MATERIAL TO THE ENVIRONMENT

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When an accidental release of radioactive material occurs, the mitigation of health effects in the exposed population can be achieved only by Remedial Actions (RA) applied to individuals or their environment. RA adoption should be based on a balance of the damage they carry and the reduction in the health effects they can achieve.

In this paper a "cost-effectiveness" analysis is performed by comparing the costs of RA with the monetary value of the collective dose avoided by them. The extent of the resulting damage is partly determined by the Intervention Level (IL) chosen for defining RA time and space features. In a general fashion, the higher the value of IL is, the smaller is the economic damage DCRA caused by RA, but smaller is the health damage DARA avoided by them If DWRA is the health damage in absence of RA and α is the monetary value of "health damage" ($ per men - Sv), the "total social damage" DT will be equal to αDWRA - DCRA - αDARA.

METHODOLOGY

1) The Effective Dose Equivalent (EDE) received or committed by the Reference Member of the Public (RMP) has been evaluated by accounting a) external exposure from the cloud and from the contaminated ground; b) internal exposure from material inhaled from the passing cloud. Collective EDE for different groups of RMP's has been evaluated on the basis of a commitment time of 30 years for EDE from inhaled material and up to ∞ for exposure to contaminated ground. The RMP is characterized by organ dose conversion factors as in WASH 1400 (1) and organ weighting factors as in ICRP 26 (2).

2) At every distance from the source RA begin at the same time TE after the onset of the accident and all the members of the public in the "Non-Stochastic Effects Area" (NSEA) and in the "Stochastic Effects Area" (SEA) are evacuated. The boundaries of NSEA and SEA are defined by a maximum value RLIM of the distance from the source and the following criteria: a) within NSEA the doses received during or committed for a time span of 30 days by the RMP exceed the following values: Total Body - 5 Sv; GI tract

= 35 Sv; Red Marrow = 5 Sv; Lung = 45 Sv; Thyroid = 250 Sv; b) within SEA the Evacuation Dose (ED) exceeds IL; ED has been defined as the EDE which will be received by RMP from TE to TE + 70 years if RA are lacking; in other words ED is the EDE which could be reduced by RA. After the passage of the cloud a location within NSEA or SEA is interdicted for the Interdiction Time (IT) required for reducing to IL the EDE which will be received by RMP in the 70 years following IT, owing to the exposure to contaminated ground. In SEA, IT cannot be less than the time IT0, required for decontaminating (if needed) and for surveying the area; in NSEA, a larger minimum value IT1 for IT is adopted, because we assume that the evacuees (which suffered NS effects) or other people in their stead cannot return back before IT1.

3) By summarizing, the economic consequences of RA have been evaluated as follows: a) property within NSEA and SEA are expropriated at TE; the economic charge has been computed as the difference between the market values at TE and TE + IT converted at the present worth at time TE; b) the evacuation of SEA is definitive if IT exceeds TEV, the mean time required for evacuees fromSEA to undertake a new work out of the interdicted area. During IT (or up to TEV) the earnings of the evacuees are secured by the community by means of subsides or services; c) decontamination is equally effective but its cost is a function of land use; d) only a fraction of the goods in the contaminated areas can be removed out of the area for being again utilized after decontamination. A "RA-tree" (Fig. 1) can be built by means of the options; b) - Interdiction, c) - Decontamination and d) - Goods Removal. The options which minimize the total costs of RA are supposed to be undertaken at every location.

RESULTS

The values of parameters for Reference Cases (RC) are in Table 1. The WASH 1400 accidents PWR2, PWR6and PWR 7 have been selected as large-extent, mean-extent and contained accidents, which last one approximates a Design Basis Accident. Figg. 2, 3 and 4 show the values of DCRA, αDARA and DCRA-αDARA as function of IL, expressed in a normalized manner as the ratio of total damage to the unitary value of the land around the plant. We could draw the following conclusions:

a) - There is a value IL0 of IL which optimizes the balance between DCRA and α DARA: for PWR2 and PWR6 the value of IL0 is about 0.1 Sv.

b) For PWR6 there is a value of IL1 of about 0.01 Sv below which DCRA is larger than αDARA. In other words, if IL is less than IL_1, the intervention enlarges, in a broad sense, the social cost of the accident.

c) For a contained accident like PWR7, no value of IL optimizes the balance between health and economic consequences, because DCRA is always larger than αDARA.

The optimization process is scarcely influenced by the values for the parameters of the model. In Fig. 5 the meteorological and the RA scenarios have been modified for PWR2 by changing, one at time, the wind speed, the dispersion coefficients, the deposition velocities and by putting TE = 0.

Also shown are the results for a value of α as low as 10^4 \$/man-Sv and for a "Developed area", whose features are between the RC and an area highly industrializated and with high density of tertiary activity. The strong influence of α on ILO and IL_1 is obvious, but we must notice how DARA is small compared to DWRA.

CONCLUSIONS

1 - The damage related to an accident in a NPP can be classified as follows: a) - Early Damage (t < TE); related to health effects from exposures before RA; it could be reduced only by ties on land usage around the plant, and cannot be avoided by RA; b) - Damage which can be controlled by evacuation and interdiction of the land (t > TE), related to health effects and to RA costs; it could be minimized by a value of ILO around 0.1 Sv (for PWR2 and PWR6),but the reduction of damage obtained by RA is very small (a few percent); c) - Damage which can be controlled by impoundment of agricultural products, not considered in this paper.

2 - If RA were managed only on the basis of the "cost-effectiveness" analysis as carried out in this paper (but we are aware that this cannot be true, mainly owing to the psychological factors which could enhance the reduction of the health effects), an Intervention Plan for reducing the "social damage" would be justified only for large and mean extent accidents. For large accidents the time and space features of RA could be so large that also the probability of the accident should be accounted for in planning the intervention.

REFERENCES

1 - WASH 1400 - Reactor Safety Study - 1975

2 - ICRP Publication 26 - 1977

Table 1 Values of the parameters for the Reference Case

TE(evacuation time) - 6 hours - 6 hours IC0 (Interdiction Time for SEA) = 60 days IC1 (Interdiction Time foe SEA) = 1 year TSV (for SEA) - 1 year Decontamination Factor DF = 99 % Accidents categories PWR2,PWR6,PWR7 (as in WASH 1400)	MIIM = 250 km Pasquill categorie D Wind speed u = 5 m/sec Noble gases = 0 Deposition Velocity Particulates - 0.1 cm/sec Halogenes - 1 cm/sec $\alpha = 10^{5}$ $/man-Sv

	Reference Case	Developed area
Value of property (M$/$km^2$)	0.75	50
Population Density (km^{-2})	10	500
Perishable goods	10%	10%
Real Estate	70%	70%
Chattel	20%	20%

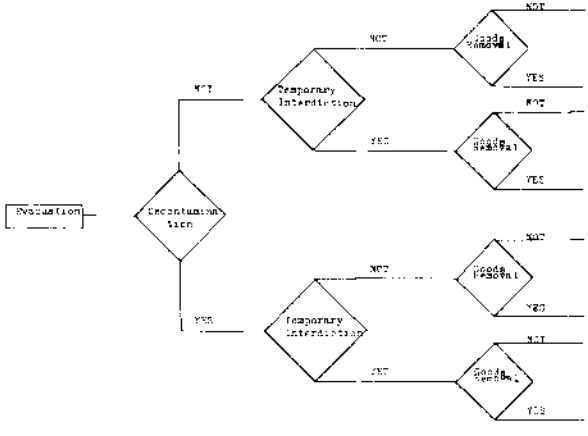


Figure 1 - Remedial Actions Tree.At every location are adopted the RA which minimize the total costs.

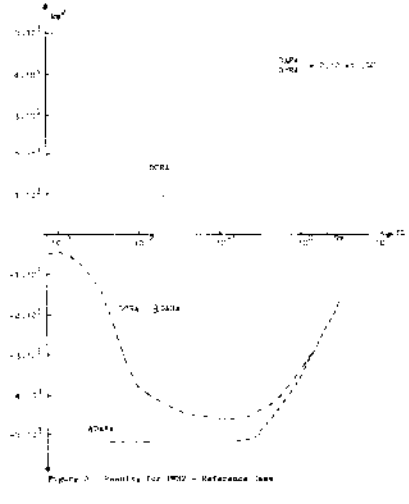
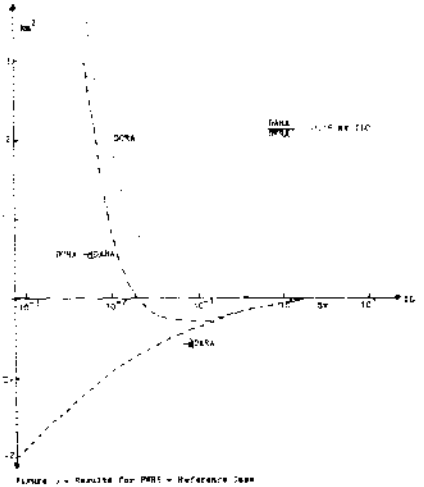
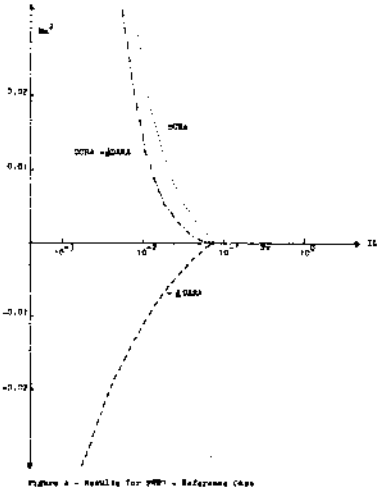
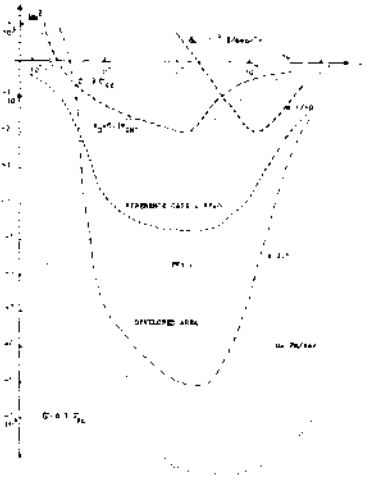


Figure 2 - Results for PWR2 - Reference Case







DISCUSSION OF AN ENVIRONMENTAL DOSE METHODOLOGY TO OBTAIN COMPLIANCE WITH DOSE LIMITS IN THE CASE OF POSTULATED ACCIDENTS

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INTRODUCTION

In the case of postulated accidents in nuclear facilities the Radiation Protection Ordinance of the F.R.G. demands sufficient protection of the public by technical design of the plant. Thus the worst case individual annual dose shall be kept within the given accident dose limits of 50, 150 and 300 mSv for total body, thyroid and bone respectively. These limits comprise all exposure pathways including the exposure via ingestion of contaminated food.

In order to achieve compliance with these demands, a practical methodology for the assessment of radiological impacts caused by postulated accidents has been developed. Thereby special attention is paid to the conservatism of its assumptions and its compatibility with the methodology of routine releases according to the existing regulations (1).

DOSE CONCEPT

The judgement of accidental releases concentrates on the radiological consequences of short-term emissions. Consequently the accident dose methodology is based on the concept of the committed environmental dose. Due to the equivalence between equilibrium dose and dose commitment based on the same boundary conditions (2), (3), (4), the calculated committed dose can directly be compared with the dose limits given as maximum annual doses in the German Radiation Protection Ordinance.

For the purpose of dose assessment it is significant to base the calculations on the three-compartment scheme of the ecosphere shown in fig. 1. Then one transfer function for each compartment can be defined which is determined by its output to input ratio. Due to the time dependence of the release rate in the case of an accident, the radionuclides undergo a time dependent transfer in the environment. Thus each of the transfer functions is also principally determined by the time dependence of the release rate. However, as our investigations concerning the nuclide dynamic in the environment have proved

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(5), the time dependence of the different ecosystems "atmosphere", "biosphere" and "man" can be temporally decoupled. Hence, each of the corresponding transfer functions A, B und D can now be determined irrespective of the time dependence of the output of the donor compartment as a peak response.

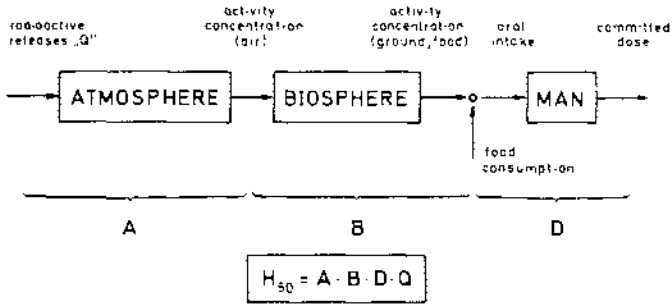

fig. 1

Due to this procedure the committed environmental dose for all exposure pathways"i"is now generally given by the following simplified formula

$$H_{50} = A_i \cdot B_{50,i} \cdot D_{50,i} \cdot Q \qquad [1]$$

where A is the atmospheric dispersion or deposition factor, B the transfer factor for the biosphere, D the dose conversion factor and Q the accidental source strength.

The dosimetric models on which the determination of the different transfer factors in equation [1] is based, are explained in more detail in (5).

DISCUSSION OF THE MODELS

The meteorological dispersion and deposition factors (A) represent the most unfavourable weather conditions, e.g. pessimistic wind velocities, high precipitation rates, etc., and thus lead to an upper limit of air and ground contamination.

According to the conclusions in (5) all dose conversion factors (D) derived for continuous releases can be applied to short-term releases as well.

In the case of external irradiation from the cloud or from the contaminated ground, the dose conversion factors are in fact "dose rate conversion factors" being de-

fined as dose rate to air concentration ratio, and thus do not depend on the type of releases.

For internal irradiation (inhalation and ingestion) the applicability of the dose conversion factors for continuous releases to short-term releases is due to the equivalence between the committed dose and the equilibrium dose concept.

The use of the proposed biosphere transfer factors (B) results in conservative doses for the corresponding exposure pathways.

This is valid for the inhalation exposure, particularly caused by the conservative assumptions for the respiration rates. In case of resuspension being considered, this causes an additional contribution to the conservatism of the dose.

For γ-ground irradiation the model represents the dose of a person exposed for 50 years on contaminated paved surface. Possible shielding or wash-off effects are neglected. This neglection causes an overestimation of the dose up to a factor of 7 (5).

In the case of ingestion there are various simplifying assumptions which give rise to different systematic errors in dose calculation. So the temporal decoupling of the compartments in fig. 1 may cause systematic underestimation of the dose up to 90 % (factor 10). Moreover the soil contamination for the milk and beef pathway may be underestimated up to 50 % (factor 2). The remaining assumptions, i.e. using soil-plant transfer factors for continuous instead of short-term releases with no distinction between vegetation period and the rest of the year, and the application of pessimistic consumption rates for the different food pathways result in reasonable overestimations of the dose between a factor of about 3 to 11. Hence the overall overestimation is a factor of 2 to 10.

This systematic error is valid only for activity uptake via root transfer. For the consumption of food directly contaminated by the deposition of radionuclides on the above surface parts of the vegetation, the systematic error is considerably smaller. Thus, due to the dominating importance of the latter type of contamination, the total overestimation of the ingestion dose does not exceed a factor of 3.

If one assumes administrative preclusion of food consumption after an accident, the calculable ingestion dose would then be reduced to values between 60 % for Sr 90 and 0,03 % for I 131 compared to the case without any restrictions.

CONCLUSIONS

The proposed methodology for dose assessments in the case of postulated accidents shows compliance with the

F.R.G. Radiation Protection Ordinance and has the following main characteristics

1) Due to the equivalence between the committed dose concept and the concept of equilibrium dose, the methodology shows sufficient compatibility with that for routine releases. This is valid not only for the dose calculation, i.e. the total ecosystem but also for each subsystem in the ecosphere. Therefore both the routine and the accident calculations can be based on the same ecological data and dose conversion factors.

2) The dose calculations for postulated accidents based on the ecological parameters which are determined for the equilibrium case result in conservative assessments.

3) The determination of one transfer function for each compartment ("atmosphere", "biosphere" and "man") supplies theoretical measures, which partly offer the opportunity of direct validation of the corresponding model by measurements following accidental releases.

4) For a planning engineer these measures, calculated and fixed beforehand, enhance the practicability of the methodology as a tool in decision-making. Using these measures dose assessments become simple and feasible without deeling with the radiological problems being involved.

5) The methodology results in considerable reduction of the computational efforts.

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PROTECTING FRONT-LINE SURVEY AND RESCUE TEAMS DURING EMERGENCIES

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Teams trained in First-Aid, Fire Fighting and Rescue are available from the shift personnel at British civil nuclear power stations to provide assistance during an emergency. Before the Emergency Controller can deploy these resources safely, he needs information about the situation on the plant. A reconnaissance team with monitoring instruments, protective clothing and self-contained breathing apparatus may first have to enter the accident zone before successful remedial action can be planned. Procedures are described which try to ensure the safety of personnel who have to approach unknown dangers.

The hazards which the reconnaissance team may encounter include:

1. Fire or high temperatures
2. Smoke or poor visibility
3. Concentrations of irrespirable gas (CO_2 at Magnox power stations)
4. Unsafe access routes due to building or plant damage
5. High levels of radiation or contamination.

In most of these accident situations the team may need to wear breathing apparatus, so the personnel involved must train regularly.

TEAM COMPOSITION

The minimum number for a team is 5 trained men, each equipped with breathing apparatus. One member should be a plant engineer with detailed knowledge of the conventional hazards of the site; a second should be used to making radiological measurements; a third should be trained in first aid.

In addition to breathing apparatus, protective helmet, ear defenders and a personal dosimetry pack, the team have found it advisable to take a radiation dose-rate meter, a gas detector, a walkie-talkie radio or jack-in telephone, first aid kit, torches, master keys, a guide line and a B.A. Control Board with clock and pen.

SELECTING THE INCIDENT CONTROL POINT

The Team Leader must use considerable discretion in approaching the unknown situation. When the accident has been located, the team establish an Incident Control Point in a safe place near to the boundary of the hazardous area, taking account of any forseeable short-term changes. Areas in the reactor buildings with easy access to stairs or lifts and equipped with telephones have been selected as potential control points. The Emergency Controller is informed of the situation and can send forward reinforcements together with a mobile trolley described later.

If the situation demands more than one access route to the area, a separate Incident Control Point is set up at each entry.

If the Control Point becomes untenable, teams already in the

hazardous area are withdrawn at once. When all are out, the Control Point can be rested at a safe location in clean air.

INCIDENT CONTROL POINT PROCEDURES

A trained man is nominated by the Team Leader to take charge of entry procedures. It helps if he puts on a distinguishing armband or helmet. He is referred to as the Control Officer and all personnel who enter the hazardous area must report to him first.

No man is allowed to enter or remain alone in an area of hazard and normally men work in teams of 2 to 4. The opportunity to save life may motivate a man to go beyond these procedures, but he should recognise the risk involved both to himself and to his colleagues.

Usually two men go forward to investigate the situation while two more are immediately available to help if required. A U.H.F. walkie-talkie radio is carried by one member of each team and a "talk through" facility at the Control Point enables the Emergency Controller and other teams to know what information is available.

If self-contained breathing apparatus is required, the tally from each set in use is marked with the wearer's name, the air cylinder pressure and the time of entry to the hazardous area. This tally is pushed into a slot in a Control Board faced with transparent plastic, so that the information cannot be obliterated inadvertently. Men are instructed to return at once to the Control Point if the warning whistle on the B.A. set sounds, indicating 10 minutes air reserve left. The safe working time is entered on the Control Board next to the tally, using initially a standard table and the clock provided. It can be updated if the team members can inform the Control Officer of the reading on the pressure gauges of their air cylinders, because the duration is very dependent on the physical exertion required. The destination or function may also be written next to the tallies. The Control Officer will arrange for each team to be replaced by fresh men at the appropriate time, until the task is completed.

On returning to the Control Point, each B.A. set wearer collects his tally and replaces it on the set when the air cylinder has been changed. If the team do not reappear within the expected time, the Control Officer will send in the reserve pair to locate and assist those missing. In addition, he will ask the Emergency Controller for another pair of men to be available to him before the reserve team are due out, in case further reinforcements are needed.

The Control Officer can cope with 10 to 12 B.A. set wearers at one time. If more need to enter simultaneously, a second Control Point should be considered.

The written record of personnel entries reduces the risk to those in a hazardous area because their safety does not depend on the memory of a man who could himself become a casualty.

USE OF GUIDE LINES

Guide lines are used at the discretion of the Team Leader. They ensure that a team can retrace their entry route to the Control Point even when visibility is nil. Each wearer of a B.A. set has a "personal line" secured at one end to the set harness. The free end has a snap hook which can be clipped over the guide line.

A main guide line consists of 60 meters of nylon cord carried in a cylindrical canvas container. One end of the cord is attached to the container and the line is laid up so that it pays out through a hole in the lid as the user carries it away from the fixed end.

The container is carried by the team or attached to the harness of a B.A. set. The last man in the team ties the line to convenient objects at sufficient intervals to keep the line off the ground. The knot must be easily untied and a slip knot is normally used.

Two main guide lines may start from one Control Point and these are then marked with an 'A' or 'B' circular tally.

Every 2½ meters along the guide line, two tabs are attached 15cm. apart. One tab has two separate knots while the other is unknotted and longer. The knotted tab is on the "way out" side of the plain tab.

The team leader may decide to start laying a main guide line after he has left the Control Point. In this case, the way back to the Control Point must be well defined and have good visibility.

Up to four branch lines may be attached to the main guide lines from one Control Point. These lines are marked where they start from the main guide lines by a rectangular tally containing 1, 2, 3 or 4 finger-sized holes, so that they can be identified by touch alone.

The Incident Control Point trolley described later carries six similar lines each in a separate container. When used, they function as a main guide line if attached at one end to the Control Point and as a branch guide line if attached as a spur to a main guide line. A person working at the far end of a branch line can be 120 meters away from his Control Point.

A personal line consists of 6 meters of lighter nylon cord secured to a pouch attached to the B.A. set. A 'D' ring is attached to the line 1¼ meters from the other end which has a snap hook to clip over a guide line. Normally this 'D' ring is secured inside the pouch so that the wearer is within 1¼ meters of the guide line. However, the personal line can be increased to 6 meters to allow the B.A. set wearer to extend his area of movement away from the guide lines.

After a guide line has been laid, all teams should attach the snap hook of their personal line to it or to the B.A. set harness of the man in front, with only the leading man coupled to the line.

Away from the Control Point, the personal line should only be unclipped from the guide line on two occasions. The first is when the team is transferring from a Main Guide Line to a Branch Guide Line, or returning. The second is to allow another team to pass. Out-going teams have precedence in the use of the guide line because their air supply may be low. The in-going team uncouple their personal lines and stand aside to allow the other team to pass.

If personnel do not return to the Control Point, a Rescue Team can find them by following the guide line and the personal line.

If a casualty is found but cannot be brought out, the line can be terminated there so that rescuers can go straight to the location.

Subsequent teams can move rapidly along a guide line knowing that hazards have been avoided.

CONTROL POINT TROLLEY AND EQUIPMENT

A 4-wheel trolley has provided a convenient mobile store for

Control Point equipment. It is small enough to be wheeled through doorways and into lifts, so that it can be brought to the location chosen for the Control Point. Its presence helps to remind personnel of the need to avoid cross-contamination by segregating clean and used equipment, active team members and reserves etc. The Oldbury trolley is about 115 cm x 65 cm x 100 cm tall and the stock of equipment shown in the table below has proved useful in a variety of exercises and simulated accidents.

Radiation Dose-Rate Meter	Contamination Monitor and Probe
Gas-in-Air Detector	Clean Protective Clothing (6 sets)
Filter Packs (6 off)	Battery Operated Air Sampler
B.A. Set & Spare Cylinder	Notice for "Empty Cylinders"
Wax Crayons (2 off)	B.A. Control Boards (2 off)
Telephone Extension Lead	Guide Line & Container (6 off)
Jack-in Telephone	Guide Line Tallies (1 set)
Communications Amplifier	Communication Headsets (2 off)
Communications Lead	U.H.F. Walkie-Talkie Radio
QFE Charging Unit	Dosimeter Issue & Record Forms
First Aid Kit	Hazard Warning Rope (2 reels)
Adhesive Tape (1 roll)	100 cm. wide Plastic Sheet (1 roll)
Plastic Bags (3 off)	Facemask Disinfectant (1 bottle)
Paper Sacks (3 off)	Paper Tissues (1 box)
Torches (6 off)	Chemi-Luminescent Light (6 sticks)
Floor Plans of Plant	Personal Line & Pouch (6 off)

DISCUSSION

For brevity, the preceeding sections have described equipment at Oldbury as examples, but the control principles can be used without such aids. The purpose-built B.A. control board is convenient to use, but a pencil or chalk on a wall or door could provide the essential information of who went where and at what time. The U.H.F. radio system by which each walkie-talkie receives all the messages transmitted reduces delays because trained men can anticipate developments, but the safety of team personnel is not reduced if communications are by telephone. The trolley has provided a useful focal point for the control procedures but other methods of personal discipline can be used to contain the hazard.

Plant employees who are also part-time firemen use similar techniques for non-radiological hazards, so they are not subjected to conflicting training methods on and off site. Reinforcements from full-time services can be deployed on site quickly because the control system in operation will be familiar to them.

These methods have been criticised for slowing down the first stages of rescue or remedial work in an emergency. While it takes time to lay out a guide line initially, subsequent movements in and out of the area are eased and the whole operation may be completed more quickly. During exercises, time scales tend to be shortened because participants do not expect to come to any harm. When the risks are real or the consequences of the accident are not yet known, personnel do not make snap decisions to send teams into areas of danger.

NATURAL BACKGROUND AS AN INDICATOR OF RADIATION-INDUCED CANCER*

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A recurrent theme in discussions about nuclear energy is the uncertainty regarding the effects of exposure to low levels of radiation. Statements on this subject are often misleading by implying a general level of knowledge approaching total ignorance. This paper attempts to place the problem in a probabilistic perspective.

Recent theoretical predictions on the effects of low-level ionizing radiation indicate induction rates as high as 8×10^{-3} cancers per man-rem.(1) Assuming this estimate is correct, then roughly half of all cancer incidence in the U.S.A. could be attributed to natural background radiation. Previous studies (2,3), however, indicate no correlation between cancer incidence and levels of background radiation.

Table 1 reviews some past estimates of radiation-induced cancer rates. These estimates pertain to effects of low-level radiation (sub MPC) whole body doses to large populations. The estimates shown were not all originally given in terms of cancers per man-rem. Some were calculated from doubling dose estimates, or were inferred from estimated effects of given population exposures. No judgment is made here as to the quality of these estimates or how they were derived. They are given to demonstrate their wide diversity.

Table 2 presents a summary of data on external background radiation and cancer incidence for the U.S.A. by state. The radiation values were taken from the USEPA (15), and were corrected for structural and body shielding according to the method suggested by the NCRP. (16) They are presented as corrected external background dose (CXBD) in mrem/yr. The cancer incidence values were taken from U.S. statistical abstracts. (17) For purposes of this analysis, a simplified but reasonable method was used to correct the raw data for age variation and other factors affecting the general state of health. The raw incidence data were normalized according to overall death rate, which itself is a function of the population age distribution and general state of health. The normalized cancer rate (NCR) was calculated by

$$NCR = \frac{\text{national death rate}}{\text{state death rate}} \times \text{state cancer rate.}$$

TABLE 1. Estimated radiation-induced cancer rates

Source	Date	Risk estimate (cancers/man-rem)	Source	Date	Risk estimate (cancers/man-rem)
ICRP: 8 (4)	1966	5.0×10^{-4}	Rasmussen (10)	1975	1.7×10^{-4}
Robison & Anspaugh (5)	1969	1.0×10^{-3}	NUREG-0216 (11)	1977	1.4×10^{-4}
Gofman & Tamplin (6)	1969	1.7×10^{-3}	Marcuso (12)	1977	6.8×10^{-3}
Otway (7)	1971	7.0×10^{-4}	Gofman (13)	1977	7.3×10^{-3}
Hull (8)	1971	1.0×10^{-4}	Tamplin & Cochran (1)	1977	8.0×10^{-3}
BEIR (9) Committee: Low	1972	8.8×10^{-5}	Morgan (14)	1978	6.0×10^{-4}
BEIR (9) Committee: "most likely"	1972	1.8×10^{-4}			
BEIR (9) Committee: High	1972	4.4×10^{-4}			

*Work performed under the auspices of the U.S. D.O.E., contract No. W-7405-Eng-48.

TABLE 2. Background radiation and cancer incidence

STATE	NCR[a]	CXBD[b]	STATE	NCR	CXBD	STATE	NCR	CXBD
Alabama	150.9	81	Louisiana	161.3	57	Ohio	171.4	87
Alaska	112.2	79	Maine	175.1	93	Oklahoma	159.3	83
Arizona	164.5	92	Maryland	186.7	71	Oregon	165.9	83
Arkansas	167.9	84	Massachusetts	183.8	84	Pennsylvania	171.2	75
California	180.2	68	Michigan	183.8	83	Rhode Island	192.9	78
Colorado	155.8	175	Minnesota	168.9	94	South Carolina	144.0	81
Connecticut	198.8	74	Mississippi	147.4	78	South Dakota	158.5	137
Delaware	196.3	74	Missouri	162.	79	Tennessee	157.3	85
Florida	183.2	70	Montana	162.6	119	Texas	163.5	60
Georgia	144.5	74	Nebraska	163.6	103	Utah	142.1	129
Hawaii	198.0	66	Nevada	170.7	102	Vermont	168.3	74
Idaho	152.6	115	New Hampshire	185.4	82	Virginia	170.8	76
Illinois	170.3	82	New Jersey	190.8	74	Washington	180.9	83
Indiana	165.0	76	New Mexico	147.1	139	West Virginia	152.1	83
Iowa	170.4	83	New York	184.0	82	Wisconsin	174.0	80
Kansas	163.7	83	North Carolina	150	89	Wyoming	143.3	175
Kentucky	161.5	79	North Dakota	165.5	92			

[a]Normalized cancer rate ($yr^{-1}/10^5$ population).
[b]Corrected external background radiation dose (mrem/yr).

Figure 1 presents this cancer and radiation data graphically. The data points present a picture similar to that previously shown by Frigerio (2), indicating a lack of correlation. Regression analysis of these data, assuming a linear relationship, indicates that the best-fit curve is described by the formula NCR = 190 - 0.27 CXBD, shown as a solid line in Fig. 1. The regression analysis reveals a somewhat negative correlation; however, the correlation coefficient, r = 0.39, indicates a very weak relationship.

Useful insight can nonetheless be derived by testing certain hypothetically assigned radiation-induced cancer rates against the

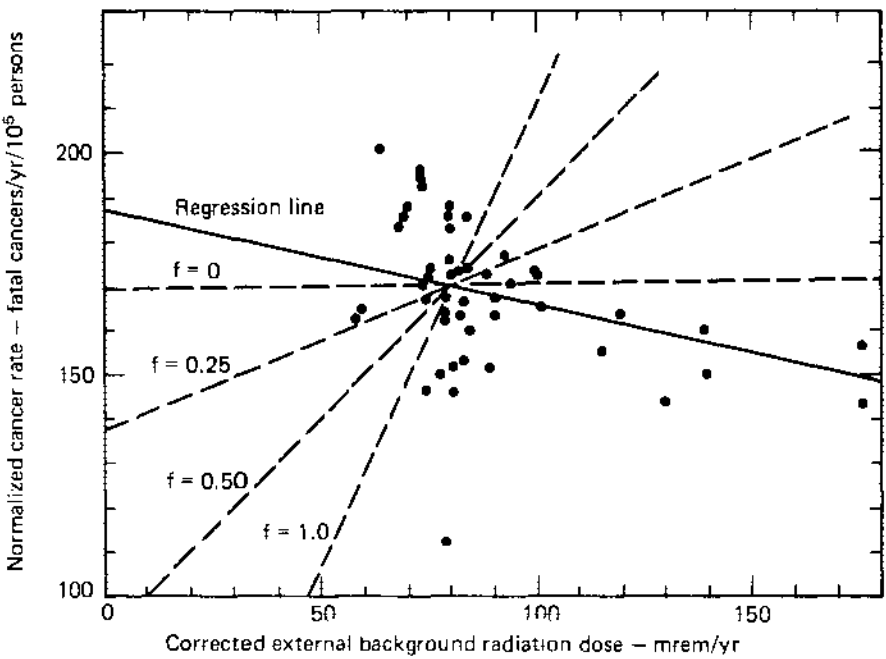

Fig. 1. Cancer incidence and background radiation.

data. This is done by determining mathematical relationships that (1) conform to the hypothesis and (2) best conserve the observed data. The hypotheses tested were:

1. The Fractional Hypothesis (a fraction, f, of all cancers are induced by exposure to external background radiation): NCR = 167(1 - f) + 2.1 f CXBD.
2. The Null Hypothesis (cancer and background radiation are unrelated): NCR = 167 (note: f = 0).
3. The Total Hypothesis (all cancers are induced by exposure to external background radiation and nothing else): NCR = 2.1 CXBD (note: f = 1.0).

These hypothetical relationships are shown as the dotted lines in Fig.1. In testing these hypotheses, the question is asked: Given that the hypothesis is correct, what is the probability that the actual data could have been observed?

The statistical method used for the calculation was the "t" test described by F.S. Acton (18), where:

$$t_{(n-2)} = \frac{(b - \beta)\ \sqrt{n - 2}\ \sqrt{Sxx}}{\sqrt{SSR}} .$$

Knowing "t" (at 48 d.f.), it is possible to determine the probability of a fractional cancer incidence (f) equal to or greater than that hypothesized. This relationship is given in Fig. 2.

To relate "f" to the radiation-induced cancer rate, assume all cancer incidence is caused by external background radiation, in which case, at steady state in the U.S.A.,

$$\frac{167 \text{ cancers/yr}}{10^5 \text{ people} \times 0.079 \text{ rem/yr}} = 2.1 \times 10^{-2} \text{ cancers/man-rem.}$$

If, for example, the induction rate value were in fact 8 x 10^{-3} cancers/man-rem, then f = 0.38. In light of the observed data, the probability of an induction rate of 8 x 10^{-3} or higher is approximately 7 x 10^{-5}, or about one chance in 14,000. From

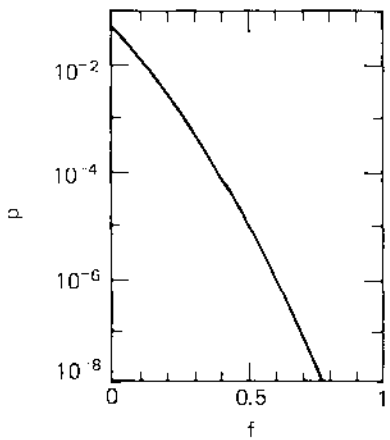

Fig. 2. Probabalistic asessment of radiation-induced cancer, where p is the probability that the indicated fraction or greater is valid, given observed data; and f is the hypothetical fraction of cancer incidence in U.S.A. attributable to background radiation.

this analysis it may be concluded that such high estimates of radiation-induced cancer rate, although not impossible, may certainly be considered highly improbable.

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TWO DECADES OF RESEARCH IN THE BRAZILIAN AREAS OF HIGH NATURAL RADIOACTIVITY.

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The geology of Brazil presents us with several areas of high natural radioactivity. Our university groups have identified and studied three natural theaters in a collaborative project over twenty years. The results were only recently presented in a symposium (1). The three areas are Guarapari, Araxa-Tapira, and Morro do Ferro.

GUARAPARI

With the slow weathering of the mountains that parallel the Atlantic coast, beaches have been formed with mottled patches of monazite sand. Monazite is a complex of rare earth phosphates with strong impurites of thorium and weaker uranium. Guarapari is a town of 12,000 in the monazite region.

The radiation levels were mapped. Patches on the beach showed levels up to 2.0 mR/hr; the streets averaged 0.09 mR/hr. A TLD survey of the population revealed that the dose rate ranged up to 2,000 mrem/year with a mean of 640 mrem/yr.

All the water and most of the food comes from normal areas. Any internal contamination might come from fine inhaled dust or from thoron and radon in the air. Whole body counting gave negative results, and analysis of placentae showed low values of internal contamination, as can be seen in Table 1.

The lack of reliable medical practise and records renders an epidemiological study impractical. With the relatively low dose rate and the small population, the somatic chromosomal aberrations in peripheral lymphocytes were selected as a biological parameter.

From the start a higher rate of chromosomal aberrations, especially the two-break type, were seen in Guarapari. A resumé of the data is given in Table 2. The number of aneuploid cells ($2n \neq 46$) and the chromatid type aberration are not considered as radiation induced, but are given as culture technique controls.

The double break phenomenon suggests an internal contaminant, an alpha emitter. Since tests for long-lived

body burdens are essentially negative, it is thought that chronic exposure to higher values of thoron and radon are responsible.

TABLE 1. Guarapari: ^{228}Th and ^{226}Ra in Human Placentae.

Local	N	^{228}Th (pCi/g Ca)	^{226}Ra (pCi/g Ca)
Controls	17	0.09 - 1.36	0.05 - 0.76
Guarapari	10	0.10 - 5.96	0.11 - 1.38
Araxa	5	0.19 - 2.63	0.07 - 0.44
Tapira	8	1.00 -26.70	0.3 - 14.50

TABLE 2. Guarapari: Individual Means and Standard Deviations for Cytogenetic Data.

	Guarapari	Control
No. Cells	66.55 ± 22.75	61.23 ± 10.72
No. Aneuploid Cells	3.45 ± 3.81	3.46 ± 2.67
Chromatid Aberrations	2.00 ± 2.54	2.23 ± 2.92
Deletions	0.65 ± 1.01	0.52 ± 0.90
Dicentrics	0.07 ± 0.28	0.04 ± 0.19
Rings	0.02 ± 0.15	0.00
Total no. Breaks	0.85 ± 1.20	0.57 ± 0.93

ARAXA-TAPIRA

IN this region in the interior State of Minas Gerais, the soil is naturally fertilized by uraniferous apatite, a phosphate mineral that serves as a source of radium-226. Here the concern is naturally with the food chain as a contaminant. The highest concentrations are found in the staple foodstuffs, manioc and its flour, as well as in potatoes and citrous fruit. Radium-228 content reaches 2,720 pCi/kg and radium-226 81 pCi/kg.

A house to house inquiry gave clear information on the dietary habits of 28 families. Certain poor families live almost entirely on the contaminated produce. As one moves from the hot spots the radium content becomes more diluted. The small group has daily intakes of the range 20-40 pCi of radium-226 and 120-240 pCi of radium-228. This would result in body burdens of the order of 280-560 pCi of radium-226 and 1680-3360 pCi of radium-228. This is below the sensitivity of any whole body counter in Brazil.

It was found that only a small number of people are

affected, and only 196 individuals of the 1670 were selected for further investigation.

MORRO DO FERRO

In the region south of the city of Poços de Caldas an alkaline plug thrust its way 400-600 meters above the surrounding land. The inner portion lowered, and became highly mineralized. In the center Morro do Ferro (Iron Mountain) rises 140 m. above its base. Across it two dikes of magnetite have penetrated, with mixtures of titanium and manganese. A great variety of rare earth oxidation compounds are found here with strong percentages of thorium oxide and traces of uranium.

The harsh face of the mountain offers poor soil for vegetation, and only low grade grazing grass grows. The mountain was mapped with ionization chamber and portable scintillometer. Some 42,000 m^2 show levels above 1.0 mR/hr and small patches are above 3.0 mR/hr.

While it was considered an ecological laboratory many measurements were made of the uptake by plants, and of the concentration of thoron and radon in rat holes and termite mounds. Some of the data are given in Tables 3 and 4. In his study of the exposure to local rats Drew found that the greatest dose was to the trachea-bronchi area with an average dose of 200 rads/year and a maximum ten times that. Takahashi conducted a cytogenetic study of the scorpion found on the mountain.

TABLE 3. Morro do Ferro: Range of Radionuclide Concentration in Plants.

	(pCi/kg)
Ra-228	169 - 10,303.
Th-228	68 - 2,200.
Ra-226	7 - 1,105.

TABLE 4. Morro do Ferro: Concentration of Thoron and Radon in Ground Holes

	(pCi/l.) Average	Range
Thoron	16,790.	285 - 55,400.
Radon	3,300.	5 - 27,400.

The research work on Morro do Ferro has recently taken a different direction. This mountain is conservatively estimated to hold 12,000 tons of thorium. In view of the almost identical chemistries of thorium and plutonium, the deposit is now regarded as an analogue for modeling the transport of plutonium over geological time, once a depository has been breached. The mineral is near the surface of the mountain, and is being washed by tropical rains i70 cm per year. The equivalent amount of plutonium is greater than that which will be produced by the reactors in the United States up to the year 2050. It is thought that the deposit is from 60 to 80 million years old.

The drainage pattern is straightforward. The rain penetrates the surface through the mineral, reaches the water table some 70 meters below, and runs off through springs at the foot of the mountain. The sediments carried off are being analyzed. The water and the solids in suspension are also being measured. Early measurements show about 0.3 pCi/l in the water.

In the future the plume will be studied in detail. A portable X-ray fluorescent spectroscope will follow the pattern of the distribution of thorium, rare earths and uranium.

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EVALUATION OF CANCER INCIDENCE FOR ANGLOS IN THE PERIOD 1969-1971 IN AREAS OF CENSUS TRACTS WITH MEASURED CONCENTRATIONS OF PLUTONIUM SOIL CONTAMINATION DOWNWIND FROM THE ROCKY FLATS PLANT IN THE DENVER STANDARD METROPOLITAN STATISTICAL AREA.

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A plutonium-processing plant in Jefferson County, Colorado has released transuranic nuclides in exhaust plumes since 1953 (1,2). In addition to leaks, small particles can migrate through banks of high efficiency particulate air filters by diffusion, a quality shared with other alpha radiation emitters (3,4). This results in a "dissemination of the finest radionuclide particles throughout the area over a radius of several miles from the plant site", and "once airborne, these smallest particles are not noticeably reduced in number by gravitational settling up to three miles from the apparent point of origin and presumably reached much farther afield" (5). Releases of Pu (Pu oxide) in exhaust from the plant (13 million m^3 daily from the main stack) ranged from an annual average concentration of 0.03 picocuries or 0.06 disintegrations per minute per cubic meter (pCi or dpm/m^3) in 1953 to 1.05 pCi or 2.33 dpm/m^3 in 1962 (6). Plutonium concentrations in air are consistently the highest in the U.S. Department of Energy 51 station worldwide monitoring network (7). Average daily concentrations of Pu in exhaust plumes were as high as 948 pCi/m^3, for the eighth day (12 mCi, or about 200 mg) after a fire in 1957 which burned out the filter system (8). There are no records of emissions for the seven day period during or after the fire, but those releases may have been 4 to 5 orders of magnitude greater. The releases of Pu and other transuranics in the fire represent the most important exposure to to the population near the plant during the period 1953-1971. The major route of exposure is the inhalation of airborne particles of Pu and other transuranium nuclides by people in the path of exhaust plume from the plant, and, for those living near the plant, the inhalation of Pu in resuspended surface dust. Work in progress confirms the presence of Pu from the facility, identified by isotope ratio, in persons in the area (9).

A preliminary study of leukemia and lung cancer deaths compared eight census tracts around the facility with 19 census tracts with a similar population in the relatively uncontaminated part of the county. A higher age-corrected leukemia death rate was noted in the contaminated area ($p=0.01$) and the age-specific (45-64 years) death rate from lung cancer was about twice as great as for the control area ($p<0.05$) (10,11). A preliminary study of congenital malformations coded at birth found a rate of 14.5 per 1,000 births for a large suburban city near the plant compared with a rate of 10.4 for the remainder of the county, and 10.1 for the state of Colorado, a difference of interest (12).

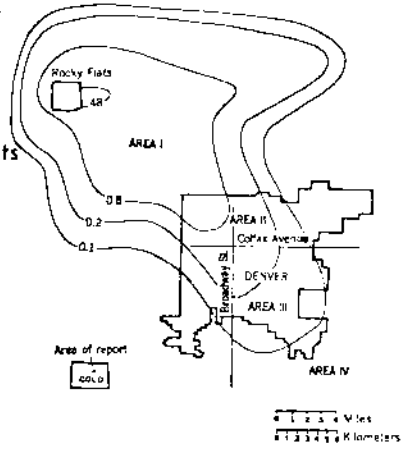
In order to determine if exposure of a large population to small concentrations of Pu and other transuranics had produced a measureable effect on cancer incidence, the following investigation was conducted.

The incidence of cancer for the period 1969-1971 for each site was calculated for Pu isopleth areas of census tracts with decreasing concentration of Rocky Flats Pu (identified by isotope ratio) in soil, determined by an area-wide survey

(core samples to a depth of 10 cm) carried out by the Atomic Energy Commission in the Denver area in 1972 (figure 1) (13,16). The position of the isopleths are approximate but useable in comparing the incidence of health effects between areas with decreasing environmental contamination around a point source of emission. The Pu content of soil was used as a surrogate measure of exposure through pathways other than those that originate from the soil (i.e. an indication of direction of exhaust plumes from the Rocky Flats plant since 1953). Census tracts divided by isopleths were counted in the area containing the major part. Age-specific cancer rates for whites, excluding persons with Spanish surname, (typical of the area near the plant) were calculated for the Denver SMSA, and sub-areas with similar population size and decreasing soil concentrations of Rocky Flats Pu were compared to the rates for the SMSA as a whole, yielding age-corrected cancer incidence rates per 100,000 per year, observed/expected case numbers, risk ratios and chi square values for males and females. Area I within the Pu concentration range 40-0.8 millicuries/km^2 (mCi/km^2), lies between 3 and 21 km from the center of the Rocky Flats plant site along the principal wind vector. Area II (0.8 to 0.2 mCi/km^2) extends from 21 to 29 km and Area III (0.2 to 0.1 mCi/km^2) from 29 to 35 km. Area IV, the unexposed population, had an age-adjusted cancer incidence virtually identical to that for the state for males, 269 and 270 per 100,000 respectively, and for females, 226 and 227 per 100,000 respectively (13). The risk ratio for Area IV was assumed to be 1.0 and the exposed populations (Areas I-III) were compared to Area IV.

Fig. 1

Denver area census tracts within isopleths for soil contamination with plutonium-dominant from the Rocky Flats plant



Isopleths in millicuries of plutonium per square kilometer

The total incidence of cancer for the period 1969-1971 is summarized in Table 1 by isopleth area of Pu concentration. Compared to males in the unexposed area, there was an incidence of cancer 8%* higher (i.e. a proportionate morbidity ratio of 1.08) for males in Area III, most distant from the plant, 15%** higher in Area II, nearer the plant, and 24%** higher in Area I, nearest the plant. The corresponding values for females (all higher than expected) were 4%, 5% and 10%*, and for both, 6%, 10%** and 16%**.

The total population exposed to low concentrations of Pu (Areas I-III) can be compared to the unexposed population (Area IV) in Table 2. The incidence of cancer of lung and bronchus is higher for males* and for both sexes together**. The leukemia incidence is higher for both sexes*, and the incidence of lymphoma and myeloma for males*. The incidence of cancer of the tongue, pharynx, and esophagus was higher for males**, females**, and for both sexes**, as was cancer of the colon and rectum (males*, females** and both sexes**). The incidence of cancer of the liver and "biliary" was higher for males**. Both sexes had a higher

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* critical X^2 value at 95% confidence level. **critical X^2 value at 99% conf. l.

incidence of cancer of the gonads (testis, 40/18** and ovary, 159/127**) with the greatest proportionate morbidity ratio for testis (2.22). This finding is consistent with the propensity of Pu to concentrate in gonads (17,18). The incidence for sites of cancer not listed ("other" in this table) was small in number for most sites but in toto was higher than the incidence in the unexposed population (males** and both sexes combined**).

Within the exposed area, the incidence of lung and bronchial cancer for males was about 33% higher** in Area I (a suburban area with more in-migration) than for males in uncontaminated Area IV (also predominantly suburban). This higher incidence persisted in Areas II and III (46%** and 13%, respectively). Females

Table 1

Census tract areas selected by decreasing soil concentrations of Rocky Flats plutonium, Anglo population size, median income and education, and total incidence of cancer, age-adjusted, for 46 sites, by sex, for the period 1969-1971 (a).

	Plutonium mCi/km²	Anglo population male	Anglo population female	median education years	median income	Incidence of cancer compared to unexposed population: Male cases obs/exp(b)	Male % (c)	Female cases obs/exp	Female %	Total cases obs/exp	Total %
Area I	48 - 0.8	75,280	78,920	12.04	$ 8,891	644/519**	+24%	636/581*	+10%	1280/1100	+16%**
Area II	0.8 - 0.2	90,300	103,900	11.85	6,367	1086/947**	+15%	1154/1100	+ 5%	2240/2047	+10%**
Area III	0.2 - 0.1	117,370	129,530	12.69	12,094	1078/1000*	+ 8%	1149/1109	+ 4%	2227/2109	+ 6%*
Areas I-III	48 - 0.1	282,920	312,350	12.22	8,668	2808/2466**	+11%	2939/2790**	+ 5%	5747/5256	+ 9%**
Area IV	< 0.1	210,670	213,190	12.97	8,055	1114	0	1260	0	2374	0

Table 2

Summary: Anglo cancer incidence by site and sex in the Denver metropolitan area over a period of three years (1969-1971) by areas of census tracts with and without plutonium soil contamination by the Rocky Flats plant (a).

	Areas I-III 50 - 0.1 millicuries/kilometer²						Area IV (unexposed)		
	Male		Female		Total		Male	Female	Total
Site	cases obs/exp(b)	pmr(c)	cases obs/exp	pmr	cases obs/exp	pmr	obs.	obs.	obs.
Lung and Bronchus	497/383*	1.30	128/120	1.07	625/503**	1.24	174	51	225
Other Respiratory	67/62	1.08	12/12	1.0	79/74	1.07	32	5	37
Leukemia	92/84	1.10	100/83	1.20	192/167*	1.15	45	38	83
Lymphoma, Myeloma	134/110*	1.22	109/123	0.89	243/233	1.04	59	56	115
Tongue, Pharynx, Esophagus	89/50**	1.78	41/17**	2.41	130/67**	1.94	24	7	31
Stomach	79/78	1.01	59/53	1.11	138/131	1.05	34	27	61
Colon, Rectum	379/333*	1.14	433/378**	1.15	812/711**	1.14	144	146	290
Liver and Biliary	52/31**	1.68	49/54	0.91	101/85	1.19	5	3	8
Pancreas	96/106	0.91	88/77	1.14	184/183	1.01	46	30	76
Testis	40/18**	2.22	---		---	2.22	13	-	13
Ovary	---		159/127**	1.25	---	1.25	-	63	63
Thyroid	22/28	0.79	80/71	1.13	102/99	1.03	18	42	60
Brain	40/48	0.83	39/34	1.15	79/82	0.96	27	20	17
Other	1221/1129**	1.08	1642/1570	1.05	2863/2699**	1.06	493	772	1265
All Cancer	2808/2466**	1.14	2939/2790**	1.05	5747/5256**	1.09	1114	1260	2374

(a) From the National Cancer Institute's Third National Cancer Survey: Incidence Data (21) (Calculated from age-adjusted rates, obtained by pooling the age-specific rates using the age distribution of the total 1950 population of the United States as a set of weights.) "Anglo" includes all whites except those with Spanish surname.

(b) $X^2 = \frac{(|\text{obs.} - \text{exp.}| - 0.5)^2}{npq}$ where n=population size, p=incidence of cancer, and q=1-p. The X^2 used with the variance = npq is a more conservative test than the Mantel-Huenszel X^2 (19)

* Critical X^2 value at a 95% confidence level. ** Critical X^2 value at a 99% confidence level.

(c) pmr = proportionate morbidity ratio observed/expected -1) x 100, compared to Area IV, the unexposed population.

did not have a higher incidence of lung and bronchial cancer near the plant, but did have a higher incidence in Areas II and III. A higher incidence (about 50% greater) of neoplasms of the nasopharynx and larynx ("other respiratory") was observed for males in Area I. The incidence of leukemia (all types) was higher for males in Area I near the plant (42%), but not for females, and higher for both males and females in Area III (9% and 58%**, respectively). The incidence of lymphoma and myelama was 40% higher for males, and 12% higher for females in Area I. The higher incidence persisted in Areas II and III for moles (29% and 12% respectively) but not for females. Cancer of the pancreas occurred with a greater than expected incidence for females in all three study areas (49%, 9% and 7% respectively) but not for males. There was a higher incidence of cancer of the thyroid for females in Areas I and II (33/26). The incidence of cancer of the stomach was higher for males in Area I, but not for females. The incidence of cancer of the colon and rectum was much higher for both males and females in Area I and II (42% higher** in Area I and 11% higher in Area II for both sexes). The incidence of cancer of tongue, pharynx, and esophagus was high for both sexes in all three study areas, especiolly in Area II (68/25* for both sexes).

Areas II and III include the Denver urban core, with the low income housing, lower educational and income level, and greatest air pollution (factors associated with a higher incidence of cancer) but have a lower incidence of cancer than Area I, a suburban area neor the Rocky Flats plant similar to Area IV, the unexposed area with the same cancer incidence as the state of Colorado. The consistency of the increase in incidence of all cancer and of certain categories of cancer with increasing concentrations of Pu in soil supports the hypothesis that exposure of the general public to low concentrations of Pu in the environment may have an effect on cancer incidence.

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HAZARDS OF RADON DAUGHTERS TO THE GENERAL PUBLIC

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Recent data from various countries in northern latitudes suggest that the most important source of public exposures to ionizing radiation may well be the inhalation of radon daughters which accumulate inside buildings; these exposures result in appreciable radiation doses to the bronchial epithelium and smaller doses to the pulmonary region of the lung, with little increment in radiation dose to other organs of the body (1).

The incidence of lung cancer resulting from inhalation of radon daughters by the general public is not known. Considerations based on vital statistics suggest that appropriate risk estimates for the general public probably lie in the range 0 to 200 fatal lung cancer per million working level months (WLM) (2,3). Risk estimates can be derived from analyses of data on excess lung cancer incidence observed in uranium miners who were in the past exposed to high concentrations of radon daughters. These data can be fitted by linear dose-response models, by "quasi-threshold" models involving, for example, a probit response on log dose (3) or by curvilinear relationships which exhibit a decreasing response per unit dose with increasing dose (4). The available data are not precise enough to distinguish between linear and "quasi- threshold" models as representing the most probable fit, but do indicate that the curvilinear relationship (4) is statistically much less probable than either of the first two models (3).

Assuming a linear dose-response relationship, and assuming that most of the fatal lung cancers induced in uranium miners occur between 10 and 25 years after the initial exposure to radon daughters (5), the available data from the U.S.A. and Czechoslovakia indicate about 100 fatal lung cancers per million·WLM in uranium miners, with a range from 50 to 200 in these estimates (3). The more limited data available from other countries including Canada (6) are not incompatible with this estimate.

If this risk estimate is directly applicable to the general public, and assuming that average public exposures due to accumulation of radon daughters in buildings are in the region of 0.08-0.16 WLM per year (1,7), then inhalation of radon daughters by the general public would be responsible for 8-16 lung cancers per million persons per year. This is equivalent to 3-6% of all lung cancers or about 0.8% of all fatal cancers in Canada in recent years. Other published risk estimates (1,4,7) would yield percentages which are considerably higher. However, even the lower numbers suggested above indicate that the public health hazards due to radon daughters are greater than those due to 0.1 rem whole body radiation per year either from natural background radiation (approximately 0.3-0.5% of all fatal cancers, based on the absolute risk model) or from medical diagnostic

procedures. This conclusion remains the same whether absolute or relative risk models should prove to be most appropriate. All of the percentages are increased 2-3 fold when the relative risk model is used (7,8), even though the loss of life expectancy is essentially the same with the absolute or relative risk models (7,9).

The concentrations of radon daughters in open air, inside buildings and in air entrapped in soil are usually in the region of 0.0006, 0.003-0.005 and 0.3-10 WL respectively (1,7,10). The primary sources of radon and thus of radon daughters inside buildings are the building materials themselves and radon which enters the building from the soil through various openings in the building foundations; dissolved radon in water from wells may also form an appreciable source in some cases. The concentration of radon daughters inside buildings is strongly influenced by the ventilation rate (2). The possible health hazards of decreased ventilation in buildings and the cost of heating the air required to provide extra ventilation during winter months in a cold climate have been calculated on the basis of the following assumptions: (a) An average of 100 cubic metres of enclosed building space per person. (b) Approximately 5,000°C-days of heating required, primarily over six months of the year, as is true for the Ottawa region in Canada. (c) Approximately $120 (U.S.) per person required at current 1979 prices to heat 100 cubic metre of air at a ventilation rate of one change of air per hour over 5,000°C-days. (d) An average exposure of 0.14 WLM per year at one change of air per hour (2), assuming that 80% of a person's time is spent inside the building. (e) An increase in radon daughter concentrations at decreased ventilation rates in proportion to the values calculated by Cliff (2). (f) A risk of 100 fatal lung cancers per million WLM. The results of this calculation are shown in Table 1.

Current ventilation rates in Canadian homes during winter months are estimated to be about 0.3 to 0.5 air changes per hour (11,12). The cost of heating air to provide extra ventilation during the winter months is in the region of one to ten million dollars (U.S.) per fatal lung cancer avoided (Table 1) based on the assumptions listed above. Factors that would affect this result are:

a) If a heat exchange unit were installed between the outgoing warm air and incoming cold air in a forced ventilation system, costs of heating this air could be reduced by 30 to 40 percent. Offsetting this saving would be the cost of the heat exchange unit.
b) The actual radon daughter concentration in a house with a "forced air" central heating system, common to many Canadian homes, could be much less than taken from Cliff (2) for the same radon output. The rapid mixing of air in such a forced air system would result in increased plate-out of radon daughters onto the walls of the heating ducts and the coarse filters in the system. Wrenn et al. (13) have shown that this effect can reduce the concentrations in air by up to a factor of 10 under appropriate conditions.
c) The assumption that a linear exposure response relationship will extend from the high exposures accumulated by miners down to the exposure and exposure rate applicable in most buildings may overestimate the actual risks. There is some evidence to suggest that very low concentrations of alpha-emitters may not produce cancers within the normal life-span (14).

TABLE 1. Estimated benefits and costs of increased ventilation in buildings

(i) Increase in ventilation rate (changes of air per hour).	(ii) Decrease in exposure to radon daughters (WLM) for the six month heating period	(iii) Fatal lung cancers avoided (per million persons).	(iv) Cost of heating air to provide increased ventilation (millions of dollars per million persons).	(iv/iii) Cost/benefit (millions of dollars per lung cancer avoided).
from 0.1 to 0.2	from 0.94 to 0.46	48	12	0.25
from 0.2 to 0.5	from 0.46 to 0.16	30	36	1.2
from 0.5 to 1.0	from 0.16 to 0.07	9	60	6.7
from 1.0 to 2.0	from 0.07 to 0.032	4	120	30.

d) Risk estimates derived from studies of uranium miners may not be valid for the general population due to differences in smoking habits, age distribution, and exposure to dust and other factors.

This paper has attempted to quantify the cost-benefit relationship for decreasing radon daughter exposures in Canadian homes by increasing ventilation rates. It has identified the main components in this relationship and pointed out the uncertainties associated with some of them. The uncertainties in the calculated values appear to be related primarily to the most appropriate risk estimates for inhalation of radon daughters and to actual radon daughter concentrations inside buildings at various ventilation rates.

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LUNG DOSES FROM RADON IN DWELLINGS AND INFLUENCING FACTORS

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In recent years there has been a growing interest in the naturally occuring radiation, and in the possible biological effects of small exposures. One of the most interesting problems in this field seems to be the radon concentrations in dwellings and the associated lung cancer hazards to the population, and in this paper we will discuss some factors that have influence upon the population doses from inhalation of radon and its decay products in dwellings.

RADON IN DWELLINGS AND INFLUENCING FACTORS

There are several sources of radon to the indoor air, i.e., radon from building materials and the ground, well water and natural gas used in the household. In Norway, well water and gas is used very infrequently, but in some other countries such as Finland, radon supplied from water is a large problem (1).

The activity concentration of radon inside a room under steady state conditions may be expressed as:

$$C = \frac{E \cdot F}{V \cdot \lambda_v} \qquad (1)$$

where: C = radon concentration (Bqm^{-3})

E = exhalation rate of radon ($Bqm^{-2}h^{-1}$)

F = area of the radon source (m^2)

V = volume of the room (m^3)

λ_v = ventilation rate (h^{-1})

Recently we measured the radon exhalation from some main building materials. In table 1, the radon exhalation rate pr unit radium activity concentration is shown for some typical walls.

Measurements of the radium concentration of building materials from the whole country indicated the following values: Concrete: 28 $Bqkg^{-1}$, Brick: 63 $Bqkg^{-1}$, LECA: 52 $Bqkg^{-1}$. For houses with walls of concrete, the ratio F/V may be about 1.8 m^{-1}. Using this value, we find that the mean radon concentration inside concrete houses in

Norway may be between about 50 and 90 Bqm^{-3} for ventilation rates between 0.3 and 0.5 h^{-1}. These ventilation rates may be commonly found in modern houses in the Nordic countries. Recent measurements in Norwegian houses indicated a mean value of 74 Bqm^{-3} (2) in concrete houses, which seems to be in good agreement with the values found by exhalation measurements. The measured values in brick and wooden buildings were 37 and 48 Bqm^{-3} respectively.

Table 1. Radon exhalation rate pr unit radium activity concentration for some walls

Material	Typical wall thickness (cm)	Exhalation rate ($Bqm^{-2}h^{-1}/Bqkg^{-1}$)
Concrete	20	0.50
Brick	20	0.20
LECA	20	0.26

The radon concentration inside rooms is highly dependent on the ventilation rate. The ventilation rate is dependent on factors such as wind speed and direction and the temperature, while the exhalation rate is dependent on the atmospheric pressure. The radon concentration in houses will therefore show large variations (2,3). An example of this is shown in figure 1.

DOSES

The doses to the respiratory system is dependent on the radon daughter concentration. There will not be equilibrium between the radon and its daughters because of the ventilation and deposition of daughter products on walls, furniture etc. Measurements (2) have indicated that an equilibrium factor of about 0.5 may be representative for Norwegian houses. The mean radon concentration in Norwegian houses is about 50 Bqm^{-3} (2), and this indicates that the radon daughter concentration is about 0.007 WL (Working Levels). For a person spending 80% of his time within doors, the annual exposure to radon daughters is about 0.3 WLM (Working Level Months).

There are large discrepancies in the dose estimates for the respiratory tract. The risk estimates for miners are all given pr WLM, so this concept seems to be more useful in evaluations of the biological effects of inhaled radon daughters. In a recent study (4), we discussed the possible lung cancer incidence in Norway from inhalation of radon daughters in dwellings. From the lung cancer statistics, and from modified risk factors we concluded that about 30 cases pr year and 10^6 persons may be due

to radon daughters in Norwegian dwellings. If we now use the ICRP concept of effective dose equivalent, this indicates that an exposure of 1 WLM in houses may be equivalent to an effective whole body dose equivalent of 10 mSv. If we adopt this, we may study the influence of reduced ventilation upon the population doses from radon daughters in dwellings. This is shown for a few ventilation rates in table 2. The values are calculated for concrete houses, using the exhalation rate given in table 1. The values are calculated for a radium concentration of 100 $Bqkg^{-1}$, which is proposed as an excempt limit by the OECD (5).

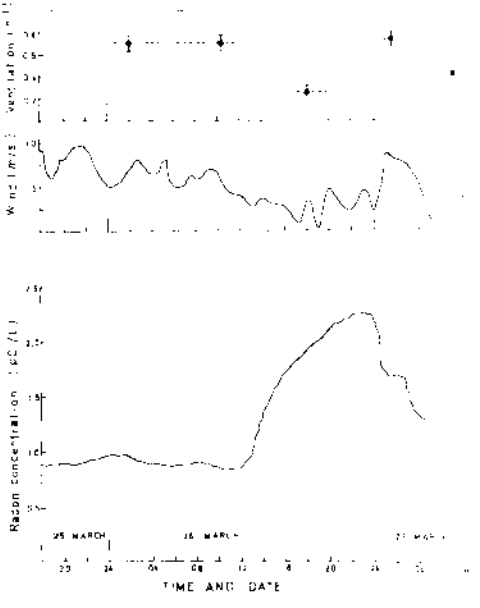

Figure 1. Measurements of wind speed, ventilation rate and radon concentration in a testhouse (ref. 2).

Table 2. The effective dose equivalent from inhalation of radon daughters in concrete houses for different ventilation rates. Radium concentration: 100 $Bqkg^{-1}$

Ventilation rate (h^{-1})	Daughter exposure (WLM/year)	Effective dose equivalent (mSv/year)
0.1	7	70
0.3	1.8	18
0.6	0.7	7
1.0	0.3	3

The population effective dose equivalent from inhalation of radon daughters in Norwegian houses today is about 3 mSv/year. This is the largest contributor to the population exposure, and the research on the radon problem should be given high priority among those concerned with the radiation protection of the population.

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ASSESSMENT OF BIOLOGICAL EFFECTS RESULTING FROM LARGE SCALE APPLICATIONS OF COAL POWER PLANT WASTES IN BUILDING TECHNOLOGY IN POLAND

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INTRODUCTION AND RISK ESTIMATES

The aim of this work is to evaluate radiation-induced remote biological effects to the population of Poland /34.2 millions in 1975/ due to the hitherto progressing and the expected development of the building technology. From among the harmful biological effects only severe genetic damage, leukemia and malignant tumors are considered. Serious genetic damage is considered in two groups. One of them comprises the effects which would be expressed over several generations following the irradiation of the parents. The other group consists of effects which would be manifested in the first generation after the exposure, thus being virtually caused by dominant mutations. The value of 10.5 severe genetic defects in the first generation and 300 over all generations of the progeny of irradiated parents per 1 rad per 10^6 births are accepted as genetic risk factors of chronic irradiation at low dose rates. The risk of induction of harmful somatic effects is evaluated in terms of absolute risk, i.e. as the difference between the risk of the irradiated and non-irradiated population. On the basis of the linear model of the dose-response curve, the absolute risk is expressed as a number of excess cancer inductions or deaths observed in a given population per year and rad or rem. For calculating the absolute risk factors the radiation risk estimates inserted in BEIR-72 and UNSCEAR-72,-77 reports are used. Risk coefficients are determined as average values weighted for products of numbers of irradiated persons times years of observations.

The risk factors for leukemia are estimated in two age groups: in children 0-9 years old /0.96 excess deaths per 10^6 man-rems per year/ and in older people /2.22 excess deaths per 10^6 man-rems per year/. In accordance to the BEIR-72 report the latent period is taken for

2 years and plateau region for 25 years.

Absolute risk of death from cancer /excluding leukemia/ also varies with age at irradiation. Two age groups are distinguished, viz. people younger than 20 and adults, with the respective risk coefficients of 0.73 and 4.20. Latent periods of 15 years and a plateau regions of 30 years are accepted for both groups.

As the concentrations of Rn-222 and its daughters in the indoor air are usually higher than the natural levels it is reasonable to make an evaluation of the lung cancer risk as a specific kind of risk in a population exposed to these excess concentrations. In this case the risk factor of 0.79, a 15-year latent period and a 30-year plateau region are accepted.

The risk of death from cancer, attributable to the prenatal irradiation of an embryo or fetus, is evaluated for the first 10 years of life. A risk factor of 23 deaths per 10^6 pregnant mothers at risk per year rem is used. No latent period and a plateau region of 10 years are assumed.

METHOD OF CALCULATION

The general structure of types of buildings in Poland in each decade since 1950 up to the end of 2010 is determined with the use of statistical data concerning building development and consumption of building materials. Distribution of the population between the particular types of buildings is assumed to be consistent with the presupposed building structure. For four main types of buildings the average additional dose equivalent rates of gamma radiation in air, soft tissue, bone marrow and gonads as well as dose equivalent rates of alpha radiation in bronchial epithelium are estimated and considered as representative for the whole population. The residence time coefficient 0.8, the tissue screening factors for gamma radiation and the equilibrium factor $F = 0.5$ for radon daughters are taken into account. The estimated mean excess dose equivalents in various tissues, weighted for the population distribution among the various types of buildings, are within the range of 12 to 32 mrem/year for gamma radiation and of 276 to 1108 mrem/year for alpha radiation.

The average countrywide dose rate inside buildings varies from one decade to another because of variations in the building structure and the widespreading use of technologies exploiting industrial wastes for the production of building materials. The actual dose cumulated by the population is the sum of individual absorbed doses and its effects depend upon the age distribution of the population and the durations of the latent and the elevated incidence perioda.

Somatic effects are calculated according to Johnson's method /1/ with the use of the above mentioned risk factors. As these values vary with the age of the irradiated persons, fractional and summarized risk factors are introduced for each 10-year periods. The appearence of the somatic effects of the irradiation extents for long periods which do not coincide with the calculation intervals. Therefore for each decade effective values of average annual dose equivalents to bone merrow, soft tissue and bronchial epithelium were calculated with allowance made for irradiation period and the plateau region. Basing on the above mentioned considerations and data the number of somatic effects occuring in a decade is calculated.

RESULTS AND DISCUSSION

Our assessments of risks indicate that the considerably growing use of power plant wastes for the production of building materials, creates an increased risk of death from neoplasms and genetic diseases. In the considered period /1951 - 2010/ the number of leukemias due to that reason are expected to increase more than twice /246 cases in the last decade/, the number of malignant neoplasms nearly three times /1049 cases in the last decade/ and the number of lung cancers nearly five times /8459 cases during 2001-2010/. By the same period only a relatively small increase of about 13 per cent should be observed in the serious genetic defects from the same cause. The malignant neoplasms in children caused by the excess irradiation of pregnant mothers during their stay inside buildings can be neglected as the irradiated population is small and the time of the irradiation reletively short.

According to the present evaluation, the serious somatic effects of the excess indoor irradiation, expressed in absolute numbers, amount in Poland through 1951 to 2010 to more than 31,000 cases. That figure comprised nearly 1000 leukemias and nearly 4000 other malignant neoplasms due to the elevated indoor irradiation, as well as more than 26,300 lung cancers caused by the excess concentrations of Rn-222 and its daughters in the air inside buildings. Deaths caused by the deleterious genetic effects during the same period are reletively small and should not exceed 260 cases in the first generation and 7,500 cases in the whole progeny.

Confrontation of the calculated number of these somatic effects with the death rate of neoplasms from all causes reveals the reletively high contribution of the indoor irradiation in the overall incidence of neoplasms. That contribution in the decade 1971-1980 should equal nearly 3 per cent of the cancers of the respiratory

system and 0.7 per cent of all neoplasms in Poland.

TABLE 1. The expected numbers of leukemias, malignant tumors and severe genetic demages induced by the excess gamma-ray doses and numbers of lung cancers induced by the excess alpha radiation doses to the inhabitants of various types of buildings in Poland.

Biological effect	Decade						Total
	1951 1960	1961 1970	1971 1980	1981 1990	1991 2000	2001 2010	
Leukemias	108	121	138	174	209	246	996
Malignant tumors /excluding leucemias/	383	454	560	703	768	1049	3,917
Lung cancers	1826	2258	2999	4416	6336	8459	26,294
Malignant neoplasms after irradiation of embryo or fetus	2	2	2	3	3	3	15
Genetic effects in the first generation	46	33	41	44	45	51	260
Genetic effects in the whole progeny	1300	951	1159	1254	1289	1469	7,422

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FAST NEUTRON DOSIMETRY USING $CaSO_4$:Dy TERMOLUMINESCENT DOSIMETERS

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INTRODUCTION

Sulphur as a threshold detector is widely used in fast neutron dosimetry to measure the activity beta of ^{32}P arising from $^{32}S(n,p)^{32}P$ reaction[1,2]. However, conventional pellet-activation techniques require sensitive radiation detectors for the measurement of induced activity in sulphur pellets[3]. Thermoluminescent dosimeters could combine in a single device the functions of an activation-pellet and a detector of radiation emited by itself. The use of activation of $CaSO_4$:Dy for detection of fast neutrons has been suggested by some autors[4] However, quantitative measurements have not been reported so far.

In this paper we describe the use of $CaSO_4$:Dy phosphor powder in fast neutron dose measurements using the activation of sulphur from the reaction $^{32}S(n,p)^{32}P$.

The thermoluminescence induced during the irradiation and also the thermoluminescence due to decay of the short-lived activation products, is erased annealing the dosimeters, after a post-irradiation time of 3 d.

MATERIAL AND METHODS

Samples of 30 $\pm$ 0.5 mg of $CaSO_4$:Dy powder prepared at the Instituto Nacional de Investigaciones Nucleares (ININ) of Mexico were used[5]. These dosimeters were irradiated in a mixed field of epicadmium neutrons and gamma rays in the tangencial west-2 beamport of the Triga Mark III Reactor at the Nuclear Center of Mexico. The dose range used was 10^{-2}-10^{2} Gy with an associated gamma dose in the range 10-10^{3} Gy. The fast neutron and gamma ray doses with which the $CaSO_4$:Dy was irradiated were measured with sulphur pellets[2] and $Li_2B_4O_7$:Mn phosphor[6] powder.

After post-irradiation time of 3d, the dosimeters were treated to 800°C of temperature during 1 h in order to erase all thermoluminescence produced during irradiation and also thermoluminescence induced by the decay of all the short-lived activation products.

After this treatment the dosimeters were stored in plastic capsules of 2 mm thickness at room temperature (21°C) in the darkness to allow the self-irradiation of

$CaSO_4$:Dy from ^{32}P beta particles.

The self-induced Thermoluminescence was measured at different intervals of post irradiation time; these measures give an estimation of the fast neutron dose to which the dosimeters were exposed.

The thermoluminescent readings were plotted as a function of the post irradiation time of 30 d as a function of fast neutron dose in gray.

RESULTS

The figure 1 shows the build-up of thermoluminescent response of $CaSO_4$:Dy powder as a function of the post-irradiation time. Each point on the plot was obtained by taking averages of ten readings. After a post-irradiation time of 30 d, the signal of the accumulated thermoluminescence produced by 2.2 x 1011 n/cm^2 was equal to 3 x 10^{-3} Gy of ^{60}Co equivalent gray.

The minimum detectable gamma dose with this phosphor is 3 x 10^{-6} Gy of ^{60}Co gamma rays(5), and this dose corresponds to a fast neutron dose of 9 x 10^{-4} Gy.

The figure 2 shows the calibration curve for fast neutron dose in gray as a function of accumulated thermoluminescence during a post-irradiation time of 30 d. This calibration plot corresponds to a straight line on full log paper which is a typical calibration plot for thermoluminescent dosimeters.

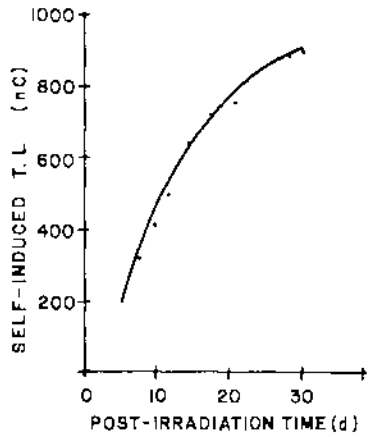

Fig. 1 Self-induced thermoluminescence in $CaSO_4$:Dy powder as a function of the post-irradiation time using 2.2 x 10^{-11} n/cm^2.

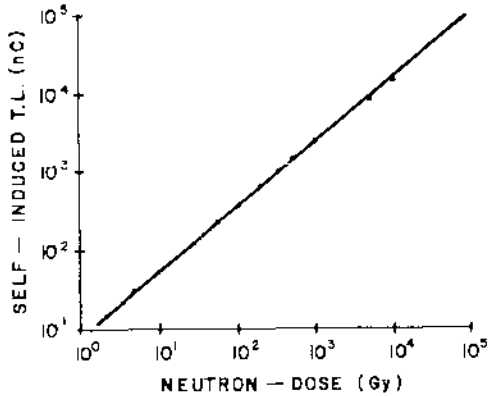

Fig.2.- $CaSO_4$:Dy powder calibration plot which was obtained after a post-irradiation time of 30 d. for epicadmium neutrons.

CONCLUSIONS

Three conclusions may be drawn from this work:

1.- Sulphur activation in $CaSO_4$:Dy thermoluminescent dosimeters is a useful method in fast neutron dosimetry.
2.- A calibration curve for fast neutron dose as a function of accumulated thermoluminescence during a post-irradiation time of 30 d, was obtained as a straight line in a full log paper in the dose range from 2 to 10^4 Gy.
3.- The minimum fast neutron dose measurable with $CaSO_4$: Dy powder as 9×10^{-4} Gy for measurements made after a post-irradiation time of 30 d.

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EMERGENCY PLANNING AND PREPAREDNESS: PRE- AND POST-THREE MILE ISLAND

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PRE-THREE MILE ISLAND - AN OVERALL PERSPECTIVE

Prior to the accident at the Three Mile Island nuclear generating station, radiological emergency response planning and attendant preparedness as it relates to nuclear facilities, was never in a position of high visibility within the nuclear industry or within the Federal, State and local governments in the U.S. Further, very few resources, in terms of personnel and funds, were devoted to it. There were a variety of reasons for this state of affairs.

First and foremost, were the two long cherished notions: (1), that nuclear facilities were designed, constructed and operated with such integrity, the chances of a serious accident occurring were extremely remote; and (2), that even if an accident were to happen, because of the integrity of design, construction and operation, any accident would have little effect in terms of offsite radiological consequences. Although the record of nuclear power safety is excellent in general terms, it is not flawless and we have been given some serious warnings.

The first of these two notions, that is 'chances" or "probabilities' of accidents happening, has, in my view and the views of others, been essentially 'knocked into a cocked-hat.' Two relatively serious events, in terms of 'chance", have occurred in large power reactor facilities in this country within the last four years: the serious fire at the Browns Ferry nuclear power facility and the accident at the Three Mile Island nuclear power facility.

The corrolary or second of these two notions, that is that little would happen in terms of offsite consequences, is to some measure still supported by the integrity of the facilities themselves. One cannot say too much with respect to the role and actions of operators and nuclear facility management during both of these events, except to say that tardy notification of offsite organizations occurred, some correct moves were made, but at the same time, many incorrect moves were also made. The point to be made here is that we were all very fortunate in both of these accidents in that offsite radiological consequences were either non-existent or relatively minimal. However, we came uncomfortably close in both of these accidents to potential consequences that could have caused grievous harm to individuals, our society, our environment, and our national energy program.

The warning has clearly manifested itself. Dr. Stephen Hanauer, of the NRC, who was the Chairman of the NRC Special Review Group (of which I was a member), which prepared the report (NUREG-0050) (1) concerning the fire at the Browns Ferry nuclear power facility, remarked at one point during that investigation, with words to the effect -- "Maybe it was like a mild heart attack -- it woke us up." We have had a second "mild heart attack" at Three Mile Island. So, it behooves all of us, industry, government and everyone else involved,

to learn from this experience because we may not get another chance to improve matters in the interim, should another accident occur -- especially a fast-breaking accident, as opposed to the drawn-out Three Mile Island event.

Other reasons for a relatively weak radiological emergency response planning and preparedness program with respect to the operation of nuclear facilities, are rooted in long-seated deficiencies in general emergency planning and preparedness programs at the Federal, State and local government levels in the U.S. Notwithstanding the massive Federal emergency operational response and industry response at Three Mile Island, advance emergency planning and coordination leaves much to be desired. Initially at Three Mile Island, coordination between Federal, State and local authorities, was a problem.

General emergency planning and preparedness at the governmental levels has suffered a period which can be best characterized as relative "benign neglect," ever since the end of World War II. Civil Defense or Emergency Services programs at the Federal, State and local government level have fallen into disarray and mediocrity due to fragmentation of efforts, lack of motivation, lack of effective leadership, inadequate attention, and inadequate funding. This is partially the reason why the new U.S. Federal Emergency Management Agency (FEMA) was established on April 1, 1979. FEMA brings together the major Federal agencies who have had responsibilities in civil preparedness, continuity of government during a national emergency, and disaster control and mitigation.

Any radiological emergency response planning and preparedness program that is mounted, must depend ultimately on an adequate general emergency planning base, at Federal, State, and local government levels. Efforts to build a proper radiological emergency response posture in support of these nuclear facilities, has suffered because one cannot build a "golden idol" on "feet of clay." If the base is defective, which it is, the idol will not stand for very long, if at all.

Adequate, well-conceived general emergency planning and preparedness at all levels of government, to cover the wide range of hazards in our technological society, is the key to an improved radiological emergency response planning and preparedness program. The NRC and other technical agencies must and will work with the new FEMA to improve this program.

POST-THREE MILE ISLAND - PROBLEMS AND PROGRESS

I have presented the overriding problem in my foregoing remarks. But, there are a number of specific problems related to radiological emergency response planning and preparedness. All of these problems existed before the accident at Three Mile Island, but the accident has speeded-up progress in these areas. There are many problems, but let me discuss five of the more salient ones:

1. An Adequate Planning Basis

What is an adequate planning basis for radiological emergencies at fixed nuclear facilities? This question, (rephrased as -- "What kind of an accident at a nuclear facility should we plan and prepare

for handling?") was essentially asked by many of the U.S. States and local governments, and their national organizations some years ago. This resulted in two Federal agencies, NRC and EPA, launching an effort to examine this question.

In August of 1976, a joint U.S. Nuclear Regulatory Commission/ U.S. Environmental Protection Agency Task Force on Emergency Planning was formally appointed to look into this matter. In December of 1978, after over two years of work, the joint NRC/EPA eleven-member Task Force unanimously concurred in and published its report, "Planning Basis for the Development of State and Local Government Radiological Emergency Response Plans In Support of Light Water Nuclear Power Plants" NUREG-0396/EPA-520/1-78-016. (2)

The "bottom line" on this Task Force report is, that there is no specific nuclear power plant accident that one can identify as being the accident for which plans and preparedness programs should be in place. Rather, the Task Force came down on the side of planning for consequences, with only minimal concern for the uncertainties of probabilities. And, to define an adequate, improved planning basis, the Task Force recommended that essentially generic Emergency Planning Zones (EPZs) be established around all nuclear power facilities in the U.S. The Task Force further determined and recognized that the U.S. Low Population Zone (LPZ) concept used for siting purposes had little real meaning in terms of offsite emergency planning and preparedness. The Task Force, in essence, rejected the concept of the "LPZ" for definitive and comprehensive emergency planning offsite. Further, the Task Force recognized the need to develop an emergency planning basis to address the so-called "Class 9" accidents, or accidents resulting in extensive damage to, or melting of, the nuclear fuel core.

This need for a capability to accommodate emergency situations beyond the so-called "design basis accidents" used in plant and site evaluation, makes generic rather than site specific areas appropriate. The Task Force decided that the establishment of Emergency Planning Zones (EPZs) of about 10 miles for the airborne "plume" radiological exposure pathway, and about 50 miles for the ingestion or food radiological exposure pathway would be sufficient to define the areas in which planning for the initiation of predetermined protective measures is warranted for any given nuclear power plant. The Emergency Planning Zone concept is illustrated in Figure 1.

As a side note and independent of the work of the NRC/EPA Task Force, the Swiss Federal Office of Energy, Nuclear Safety Division, was developing an Emergency Planning Zone concept very similar to the zones recommended by the NRC/EPA Task Force. The Swiss have 3 zones; an inner "Fast Alarm Zone" of 2 to 6 kilometers, a second zone of 20 kilometers (12.5 miles), and a third zone (for the ingestion pathway) with no radius prescribed.

Although not without some initial controversy and resistance from many quarters, the Task Force report is a major milestone along the way toward defining an adequate radiological emergency response planning basis. The report, and the recommendations contained in the report have been formally endorsed by the Commissioners of the U.S. NRC as of October 5, 1979, and were endorsed by the EPA Administrator on January 15, 1980. Plans are to establish these Emergency Planning Zones in the U.S.

2. Accident Assessment

Accident assessment has been, and continues to be, a problem area. Although defined as an essential emergency planning element in 1970 in the AEC (now NRC) emergency planning regulations 10 CFR 50 Appendix 'E' (3) for nuclear facility NRC licensees, and later in the former AEC's emergency planning guidance document for States and local governments, "WASH-1293" (now NRC publication "NUREG-75/111"), (4) much needs to be done to improve accident assessment, both onsite and offsite.

Steps are underway to improve this accident assessment capability. On the nuclear facility side, improved in-plant instrumentation specifically designed for assessing accident situations has been indicated and will now be required. On the Federal, State and local side, standardized offsite accident assessment techniques and systems need to be developed and improved, especially in the areas of coordination between agencies at all levels of government and in the evaluative/decisionmaking process. The coordination of accident assessment information must also be improved between the nuclear facility operator and the offsite agencies. Guidance concerning the types of emergency instrumentation which might be useful, and the acquisition of instruments and systems themselves, are needed in many localities.

Several programs are now moving to address these problems. Nuclear facility operators will be required to upgrade their emergency plans. Further, they will be required to implement the related recommendations of the NRC "Lessons Learned Task Force" (5) involving instrumentation to follow the course of an accident, and relate the information provided by this instrumentation to emergency action level guidelines (6) promulgated by the NRC. This will include instrumentation for post-accident sampling, high range radioactivity monitors, and improved in-plant radioiodine instrumentation since radioiodine can be a dominant radioisotope of concern in airborne radiological releases. The implementation of the "Lessons Learned" recommendation on instrumentation for detection of inadequate nuclear core cooling will also be factored into the emergency plan action level criteria.

Guidance in the area of radiological instrumentation and offsite accident assessment techniques for States and local governments, are being prepared by the Idaho National Engineering Laboratory under contract to the NRC. Plans are also afoot to test an inexpensive airborne radioiodine sampling and collection device, which together with an existing modified Civil Defense radiological instrument, has the potential to help provide quick, rough "go" - "no go" information to authorities responding to an accident in offsite areas where a radioiodine release may be the dominant radioisotope of concern in certain accidents. This portable device, invented and recently patented by researchers at the Brookhaven National Laboratory (7) under contract to NRC, is being independently evaluated by the Idaho National Engineering Laboratory. If the device passes muster, NRC has plans to put it into the existing inventory of civil defense radiological monitoring instruments currently available to State and local government personnel.

Recently, the Commission has approved relatively modest budget resources to allow us to proceed with a few "pilot-demonstrations" of

the Lawrence Livermore Laboratories (LLL) Atmospheric Release Advisory Capability (ARAC) system. The system, in its ultimate form, is capable of providing rapid atmospheric and radiological consequence assessment offsite, thus freeing nuclear facility operators and State and local organizations from laborious "1890"-type operations, with maps, plastic sheets, overlays, and grease pencils, which is the "State-of-the-art" in many nuclear power plants today.

ARAC was employed by the U.S. DOE response team, on an ad hoc basis at Three Mile Island. NRC intends to establish the first pilot-demonstration of ARAC in the State of New York by installing ARAC computer terminals and other hardware in the New York State Emergency Operating Center, and a local government Emergency Operating Center located near Consolidated Edison's Indian Point Nuclear Power Facility.

3. Training

Since March 1, 1975, the NRC with the assistance of other Federal agencies, has conducted formal training programs for Federal, State and local government personnel in both radiological emergency response planning and operations. The training programs have been well received and are of excellent quality, thanks to competent and dedicated faculty members. Much remains to be done in terms of retraining because of the high turn-over (roughly 10% per year) among State and local government personnel and also to keep pace with new developments in the emergency planning and preparedness area. NRC's plans are to continue to improve these training programs and to develop new ones where necessary. Nuclear facility personnel training must also be accelerated and improved as well.

Related to training, is the matter of standardized exercise-scenarios to test emergency plans. The NRC is developing exercise-scenarios to realistically test onsite and offsite emergency plans which should result in improving the emergency response capability at all levels of government.

4. Funding

Adequate funding for general and radiological emergency response planning and preparedness has been a problem at all levels of government; Federal, State and local. The funding problem is particularly acute at the local government level, where often many of the involved personnel are low-paid employees, part-time employees or volunteers with meager resources available to them. The funding situation needs to be improved. The amount of money required for a substantial improvement in the radiological emergency planning and preparedness effort, (as a sub-set of general emergency planning and preparedness), does not appear to be staggering. As a matter of fact, it is very small when compared to the investment made in a single nuclear power unit, of say, 1000 Megawatts-Electric, the gross cost of which today is well over the one billion dollar mark, in today's dollars, and we have some 70 nuclear power facilities licensed to operate in this nation today, and many more under construction.

Where can these funds come from? -- and more importantly -- where should they come from?

Dr. Stephen Salomon, an Environmental Economist of the NRC's

Office of State Programs, has recently completed a year-long study of this matter. His report, which was released in draft form as 'NUREG-0553" (8) in the spring of this year, one day before the Three Mile Island accident, examines this question of emergency planning funding in significant detail. His findings depict a wide range of funding situations, from relative 'affluence" -- to "abject poverty," -- concerning personnel and resources to do a proper job in this area, particularly at local government levels. Even where funding was adequate, in some cases there was no motivation or encouragement to spend funds on radiological emergency response planning and preparedness. These problems have at their roots, the individual, political, social, governmental and industrial perceptions of the relative safety of a high technology facility. Three Mile Island has changed a lot of heretofore complacent views.

But in those communities with little available to them to improve matters, the recognition of a need to do more does not always translate to, or result in, improvement. Help is needed. And, although the Federal government can and should provide some assistance, the nuclear industry has an obligation to provide financial assistance as well. Dr. Salomon's report, "Beyond Defense-in-Depth", NUREG-0553, was published as a final NRC staff report in October, 1979. The report should be useful to not only those of us involved in the regulation and management of the nuclear industry, but to the new U.S. Federal Emergency Management Agency (FEMA), and the Congress of the United States.

5. Emergency Planning Guidance

The accident at Three Mile Island, has in great measure, validated existing emergency planning guidance. Existing guidance on Protective Action Guides (PAGs) (9) (10) for radiological exposure needs to be completed by the U.S. Environmental Protection Agency and the U.S. Department of Health, Education, and Welfare, agencies charged with this responsibility. A Federal policy on the administration of radioprotective drugs, such as the use of potassium iodide as a thyroid blocking agent in some circumstances, needs to be developed by DHEW who is also charged with this responsibility. (11) (12) The NRC/EPA Task Force recommendations on the establishment of Emergency Planning Zones, must and should be quickly adopted. Specific technical guidance, such as emergency instrumentation and accident assessment guidance, needs to be developed. Guidance on interdicting or controlling the accidental radiological exposure to humans via domestic animals and agricultural products in the food chain, needs to be developed as well.

SUMMARY

The last bastion of the often quoted "Defense-in-Depth"concept against consequences of accidents at nuclear facilities, which has governed the development of commercial nuclear power for two-and-one-half decades, is a proper and effective emergency planning and preparedness program with respect to these facilities. This bastion, has not received the support which it deserves. Proper and adequate emergency planning can help alleviate many of the fears surrounding the safe operation of nuclear power facilities. This accident has given us a golden opportunity to improve things and we must not fail, collectively, to take advantage of it and to learn from it;to act on it. We are unlikely to have another chance to do so.

Figure 1. CONCEPT OF EMERGENCY PLANNING ZONES

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EFFECTS OF MICROWAVE RADIATION ON ENDOCRINE SYSTEM OF MOUSE

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Swiss male mouse, 1 or 2 months old were irradiated in an exposure chamber ($32,5 \times 32 \times 54$ cm) ; microwave frequency 2450 MHz ; time of irradiation : 1, 2, 3 or 4 hours during 5 days. The sacrifice of animals is performed by decapitation immediatly or 24 hours or 5 days after the last irradiation. The density power was calculed from the relationship

$$P = N \times S \times 10^{-2}$$

P : density power (watt)
N : number of animals in the exposure chamber
S : surface of an animal (cm^2)

Ambient temperature was maintened between 23-25°C. In any case a total of irradiated 40 mice and an equal number of non irradiated animals (controls) were studied.

When the mice were decapited immediatly after removal from the exposure cage, colonic temperature was measured within 60 s : we had never observed a variation.

After decapitation trunk blood was collected in iced tubes containing ethylenediamine-tetraacetic acid (EDTA), centrifuged at 2,500 rpm for 10-15 min in a refrigerated centrifuge, and the plasma was frozen at -23°C until it was assayed for hormones.

Testosterone assay. A modification of the radioimmunoassay method of Nieschlag and Loriaux (3) was used for the measurement of testosterone.

The intrassay coefficient of variation was less than 10%, as determined by assaying 10 ml samples of plasma. The base values

obtained from water (charcoal treated) were between 0 and 8 pg ; 10 pg was considered significantly different from 0.

Corticosterone assay. The plasma corticosterone level was determined by the competitive binding radioassay of Murphy (2). Rat transcortin was used as the binding protein. Test plasma was extracted with carbon tetrachloride followed by celite chromatography to remove other steroids which might interfere with the assay.

LH assay. Plasma luteinizing hormone assay was performed exactly as described in the radioimmunoassay kit supplied by the Rat Pituitary Distribution Program (NIAMDD). The second antibody used was antirabbit gamma-globulin purchased from Welcome laboratories (England). For the assays to be acceptable, the coefficient of variation for hormone concentration at the 50% level of bound radioactivity intercept was not greated than 6.0% in this LH assay.

ACTH assay. A rabbit antiserum against porcine ACTH (ACTH retard de poc Choay) obtained from Dr Depieds (1) was used. This antibody is produced chiefly against the biological fragment 1-24 ACTH. Buffer used in this procedure consisted of 1.5 ml 20% human albumin, 200 µl 1-2 mercaptoethanol, 1.0 ml zymofren at 10 000 U/ml (Specia), 0.02 M veronal (pH 8.6), and water to a final volume of 100 ml. This buffer had no inhibitory effect on the antigen-antibody reaction. The standard curve was done in triplicate using ACTH-free plasma from hypophysectomized animals. Assays of plasma were done in duplicate. After a 48 h incubation at room temperature, free ACTH was adsorbed with 50 mg of talc. The radioactivity was determined in the precipitate after centrifugation.

The statistical significance of differences among data from the different groups of animals was determined using Student's t-test.

The testosterone and LH levels in mice after various microwave exposures are shown in table I,corticosterone and ACTH levels in

table II.

The results of the experiments suggest that the mice adrenal axis is transitory stimulated during microwave exposure without regulation by a possible feed-back on ACTH secretion.

This endocrine perturbation seems to be a concomitant to increased testosterone plasma level. We have never observed perturbations concerning spermatogenesis.

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Table I

Time irradiation (hour/day)	Number of irradiation per day	Time between last irradiation and sacrifice	age of animals (day)	Plasma testosterone level (pg/100 ml)			Plasma GH level (pg/100 ml)		
				control		irradiated	control		irradiated
1	1	0	30	97,2 ± 2,1	↗	106 ± 2,0	94 ± 0,6	↗	106 ± 0,8
			60	310 ± 18,3	↗	[illegible] ± 14,0	87 ± 4,4	↗	117 ± 3,1
		24 h	30	55 ± 2,2	NS	56,7 ± [illegible]	92,8 ± 6,1	NS	100 ± 4
			60	297,6 ± 12,2	NS	281 ± 29	83,4 ± 0,8	↗	82 ± 2
		5 D	30	74,2 ± 4,1	NS	65 ± 3,1	101,4 ± 10,2	↗	96,6 ± 7,3
			60	315 ± 18,0	NS	308,8 ± 5,2	79 ± 1	↗	87 ± 22
2	2	0	30	76,4 ± 4,2	↗	100,6 ± 5,6	91,9 ± 1	↗	126 ± 2,7
			60	318,4 ± 8,4	↗	410 ± 20,7	92,4 ± 0,8	↗	101 ± 1
		24 h	30	56,4 ± 3,0	↗	76 ± 2,4	100 ± 9,3	↗	118 ± 2
			60	166,2 ± 11,1	↗	294 ± 6,9	79 ± 4,5	↗	94 ± 0,8
		5 D	30	[illegible] ± 5,0	NS	[illegible],6 ± 0,9	84,2 ± 0,9	NS	88 ± 2
			60	[illegible] ± [illegible]	NS	[illegible] ± 10,7	86 ± 2,4	NS	86 ± 2,4
3	3	0	30	[illegible],8 ± [illegible]	↗	[illegible] ± [illegible],5	66 ± 2,4	↗	124 ± 2,4
			60	[illegible],0 ± 1,3	↗	[illegible]29 ± 85,0	96 ± 5,3	↗	120 ± 3,1
		24 h	30	[illegible],6 ± 4,1	↗	121,4 ± 4,6	80 ± 2,7	↗	120 ± 1,1
			60	[illegible] ± 22,8	↗	[illegible] ± 5,0	77 ± 4,5	↗	120 ± 4,4
		5 D	30	[illegible],4 ± [illegible]	NS	[illegible] ± 5,7	100 ± 5,0	NS	[illegible]2 ± 4,8
			60	[illegible] ± 8,7	NS	[illegible] ± 11,2	75 ± 3,3	NS	80 ± 2
4	4	0	30	[illegible],2 ± 2,4	↗	[illegible]2 ± 16,5	61 ± 9	↗	104 ± 2,4
			60	[illegible]26 ± 2,9	↗	[illegible]75 ± 30,9	98 ± 6	↗	128 ± 2
		24 h	30	[illegible],8 ± [illegible]	↗	[illegible] ± 4,4	105 ± 2,2	↗	122 ± 5,8
			60	[illegible]2 ± 15,9	↗	[illegible] ± 3,7	99 ± 3,3	↗	120 ± 7,0
		5 D	30	[illegible],4 ± 5,[illegible]	NS	[illegible]37,2 ± 1,7	96 ± 9,79	↗	10[illegible] ± 2,69
			60	[illegible]82 ± 4,6	NS	315,4 ± 9,3	72 ± 2	↗	7[illegible] ± 5,3

Table II

Time irradiation	Number of irradiation day	Time between last irradiation and sacrifice	Age of animals (day)	Plasma corticosterone level (µg/100 ml).			Plasma ACTH level (ng/ml).		
				control		irradiated	control		irradiated
1	1	0	30	14,8 ± 0,5	→	18,6 ± 2,1	109 ± 3,7	→	124 ± 2,4
			60	15,2 ± 0,6	→	25,1 ± 2,0	75 ± 6	→	129 ± 4
		24 h	30	14,9 ± 0,8	→	19,6 ± 0,4	102 ± 2	→	122 ± 2
			60	2 ± 0,4	→	26,9 ± 2,9	85 ± 5,2	→	107 ± 8
		5	30	16,5 ± 1,2	NS	2,5 ± 1,1	108 ± 4,8	No	119 ± 5,5
			60	2,5 ± 0,7	NS	1,2 ± 0,3	97 ± 3,5	NS	104 ± 6
2	2	0	30	2,3 ± 0,5	→	8,4 ± 0,4	101 ± 4	→	122 ± 2
			60	2,2 ± 0,8	→	26,4 ± 1,1	80 ± 3,5	→	122 ± 3,3
		24 h	30	14,8 ± 0,3	→	25 ± 1,5	117 ± 4,8	→	124 ± 4
			60	18,6 ± 0,8	→	27,4 ± 1,6	58 ± 1,6	→	90 ± 3,6
		5 D	30	12 ± 0,3	NS	12,8 ± 1,2	86 ± 7	NS	84 ± 14,6
			60	12,5 ± 0,6	NS	12 ± 0,3	92 ± 8	NS	105 ± 6,3
3	3	0	30	11,2 ± 0,8	→	16,6 ± 0,7	120 ± 8,9	→	126 ± 4
			60	14,6 ± 1,3	→	18,0 ± 0,3	98 ± 6,1	→	122 ± 2
		24 h	30	13 ± 0,6	→	20,6 ± 1,6	128 ± 5,8	→	132 ± 3,2
			60	14 ± 0,7	→	25 ± 1,5	81 ± 0,6	→	92 ± 2
		5 D	30	11,2 ± 0,2	NS	12,2 ± 1,5	110 ± 3,6	NS	102 ± 11,2
			60	11 ± 1,5	NS	12 ± 0,5	94 ± 9,1	NS	96 ± 5,6
4	4	0	30	8,5 ± 0,1	→	20 ± 1	81 ± 2,3	→	108 ± 2,3
			60	2,4 ± 0,3	→	15,2 ± 0,5	65 ± 9,1	→	112 ± 5,6
		24 h	30	16,2 ± 0,7	→	31 ± 0,5	38 ± 4,8	→	96 ± 3,6
			60	12,6 ± 1,4	→	20 ± 2,3	45 ± 6,5	→	73 ± 9,5
		5 D	30	12 ± 0,3	NS	12,2 ± 0,3	76 ± 2,4	NS	78 ± 5,5
			60	12,2 ± 0,2	NS	12,6 ± 1,4	74 ± 7,4	NS	75 ± 5,8

RBE OF α-PARTICLES vs. β-PARTICLES IN BONE SARCOMA INDUCTION*

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^{226}Ra and ^{90}Sr were injected intravenously into 17-month-old beagles at the University of Utah (13) and 70-day-old CF1 female mice at the Argonne National Laboratory (2,3,4). Bone sarcomas, mostly osteosarcomas, were the main radiation-induced cancers (Tables 1 and 2).

^{226}Ra is an α-emitter and ^{90}Sr is a β-emitter. Both are bone volume seekers, so that the mean endosteal dose is roughly equal to the skeletal dose averaged over both bone and marrow (1,8). The average skeletal dose in rads was computed for ^{226}Ra including its retained α-emitting daughters (7,9), and for ^{90}Sr including its β-emitting daughter, ^{90}Y (7,10). The average skeletal dose was calculated at the assumed start of tumor enlargement, which was taken as 1 year before death in the beagles (10), 140 days before death with bone sarcoma in the mice injected with ^{90}Sr (10), and 100 days before radiographic appearance of the tumors in the mice injected with ^{226}Ra (9). Since the shapes of the retention curves for ^{226}Ra and ^{90}Sr are similar in beagles (and in mice), assumptions on the time span of the "wasted" radiation have little influence on the calculated RBE (11).

The relative biological effectiveness (RBE) of α-particles vs. β-particles in producing bone sarcomas was taken as the ratio of ^{90}Sr dose/^{226}Ra dose at a given level of incidence. The RBE progressively increased as the incidence decreased, reaching RBE = 26 at 8.7% incidence in beagles, and RBE = 25 at 7.7% incidence in mice (Table 3). The increase in RBE was largely due to the decreased effectiveness per rad of ^{90}Sr β-radiation at low doses and low dose-rates (10). Because of statistical fluctuations, the RBE's are not shown below an incidence of 7.5%, but the trends are compatible with even higher RBE's. In this experiment all of the mice have died. None of the beagles injected with ^{90}Sr are still alive. However, if future bone sarcomas appear among the 9 surviving beagles which received low levels of ^{226}Ra, the α-particle RBE at low doses will be increased above the values indicated in this paper. Of special relevance is the RBE at the low doses and low risks that are considered permissible for man. The ICRP recently increased their quality factor for α-particles up to 20 (5). But is that enough? Additional information should come in a few years from beagles injected with ^{226}Ra and ^{90}Sr at Davis, California.

The increase of RBE with decreasing dose seems a general property of densely-ionizing radiation. It also applies to the fast-neutron-induction of leukemia in people, chromosome aberrations in human lymphocytes, skin damage (human, rat, mouse, pig), breast tumors in rats, cataracts in mice, inactivation of intestinal crypt cells in mice, mutations in Tradescantia, and growth reduction in Vicia Faba (6,12).

*Research supported by the U.S. Department of Energy.

TABLE I. Bone sarcomas in *beagles* injected with ^{226}Ra or ^{90}Sr

Nuclide	Inj. μCi/kg	Yr. inj. to death	Injected dogs	Sar. dogs	Incidence (%)	Av. skel. rads 1 yr before death
^{226}Ra	10.4	2.86	10	9	90.0	13400
	3.21	4.13	13	12	92.3	5700
	1.07	6.12	12	11	91.7	2500
	0.339	10.05	13	5	38.5	1100
	0.166	9.40	14	1	7.1	447
	0.062	---	23(3)*	2	8.7	~ 210
	0.022	---	25(4)*	1	4.0	~ 74
	0.0074	---	10(2)*	0	0	~ 25
	0	---	44(14)*	0	0	0
^{90}Sr	97.9	3.40	14	8	57.1	10100
	63.6	5.82	12	8	66.7	9360
	32.7	9.98	12	2	16.7	7940
	10.8	12.27	12	0	0	2870
	3.46	10.79	12	0	0	798
	1.72	11.31	13	0	0	445
	0.57	12.93	12	0	0	143
	0	11.49	13	0	0	0

*Living dogs, as of 1 January 1980, shown in parentheses.

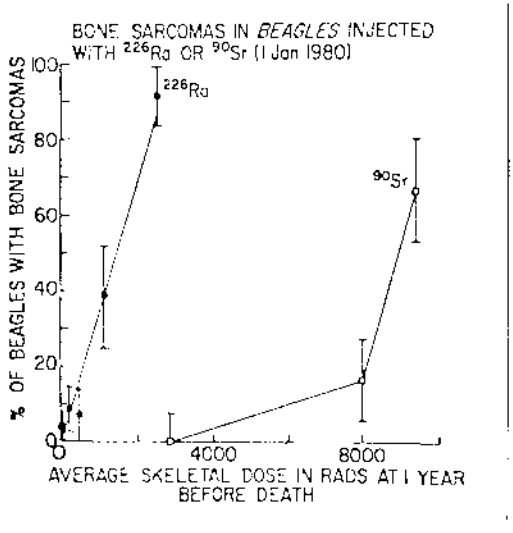

FIGURE 1. Bone sarcoma incidence in beagles. The response seems approximately linear up to 2500 rads from ^{226}Ra, but is strongly concave upwards for ^{90}Sr. Standard deviations in incidence are shown here and on Fig. 2.

TABLE 2. Bone sarcomas in *female CFI mice* injected with ^{226}Ra or ^{90}Sr.

	Inj. μCi/kg	Mice at 150 days Post Inj.	Sar. Mice	Inc. (%)	Bone Sarcoma Mice: Days, inj. to appear. (Ra) or death (Sr)	Bone Sarcoma Mice: Av. skeletal rads 100 d before appear. or 140 d before death
^{226}Ra	120	45	14	31.1	328	28900
	80	44	31	70.5	359	21300
	40	45	33	73.3	394	11800
	20	44	38	86.4	428	6420
	10	43	34	79.1	484	3640
	5	45	28	62.2	544	2040
	2.5	104	45	43.3	639	1190
	1.25	104	22	21.2	657	614
	1.00	239	56	23.4	643	480
	0.75	504	94	18.7	686	383
	0.50	683	80	11.7	655	244
	0.25	247	19	7.7	580	109
	0.10	252	5	2.0	853	62
	0.05	254	11	4.3	710	26
	0	521	6	1.2	730	0
^{90}Sr	2200	26	19	73.1	216	12000
	880	45	41	91.1	260	6630
	440	42	34	81.0	440	6300
	200	59	8	13.6	510	3310
	88	74	2	2.7	760	2090
	44	83	3	3.6	640	900
	8.9	104	0	0	---	172*
	4.5	119	2	1.7	600	87
	1.3	148	2	1.4	630	26
	0	149	2	1.3	550	0

*Dose for 8.9 μCi/kg level calculated at (600-140) = 460 days.

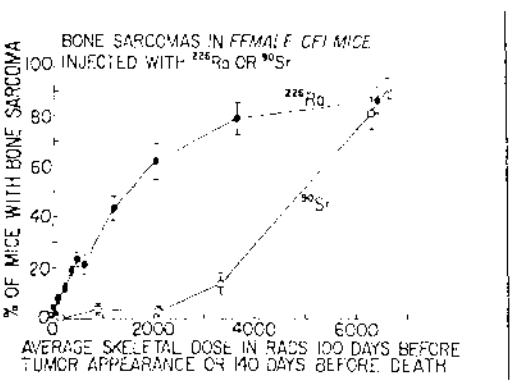

FIGURE 2. Bone sarcoma incidence in mice. ^{226}Ra is much more effective than ^{90}Sr at low doses, but at very high doses the effectivenesses converge.

TABLE 3. Bone sarcoma RBE of ^{226}Ra vs. ^{90}Sr.

Species	Incidence (%)	^{90}Sr β-particles (rads)	^{226}Ra α-particles (rads)	RBE (α vs. β)
Beagles	66.7	9360	1900*	5
	38.5	8600*	1100	8
	16.7	7940	480*	17
	8.7	5500*	210	26
Mice	86.4	6500*	6420	1
	81.0	6300	4400*	1.4
	79.1	6200*	3640	2
	62.2	5500*	2040	3
	43.3	4600*	1190	4
	21.2	3700*	614	6
	23.4	3800*	480	8
	18.7	3500*	383	9
	13.6	3310	280*	12
	11.7	3100*	244	13
	7.7	2700	109*	25

*Interpolated from curves on Figures 1 and 2.

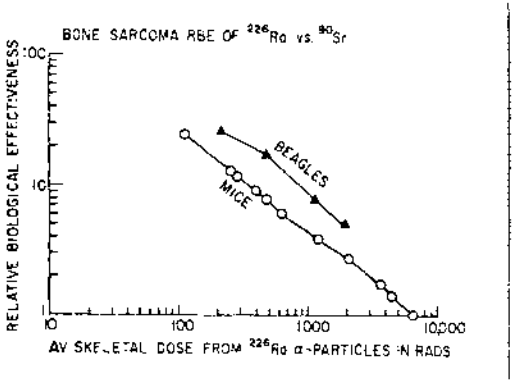

FIGURE 3. Relative biological effectiveness of α-particles from ^{226}Ra and retained daughters, relative to β-particles from ^{90}Sr and its daughter, ^{90}Y. The RBE increases as dose decreases, both in beagles and in mice. The effect is mainly due to decreased effectiveness per rad in bone sarcoma induction by β-particles at low doses and low dose-rates.

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Microwave/Radiofrequency Protection Standards: Concepts, Criteria and Applications*

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Electromagnetic energies i.e. 300 KHz to 300 MHz (Radiofrequency) and 300 MHz to 300 GHz (Microwaves) can produce biological effects or injury depending on power levels and exposure durations. There is thus a need to set limits on the amount of exposure to radiant energies individuals can accept with safety. Protection standards should be based on scientific evidence but quite often are the result of empirical approaches to various problems reflecting current qualitative and quantitative knowledge.

In considering standards, it is necessary to keep in mind the essential differences between a "personnel exposure" standard and a "performance" standard for a piece of equipment. An exposure standard refers to the maximum safe (incorporating a safety factor) level of power density and exposure time for the whole body or any of its parts. This standard is a guide to people on how to limit exposure for safety. An emission standard (or performance standard) refers not to people but to equipment and specifies the maximum limit of emission close to a device which ensures that likely human exposure will be at levels considerably below personnel exposure limits.

Basic Considerations Illumination of biological systems with MW/RF energy leads to temperature elevation when the rate of energy absorption exceeds the rate of energy dissipation. Whether the resultant temperature elevation is diffuse or confined to specific anatomical sites, depends on: the electromagnetic field characteristics and distributions within the body as well as the passive and active thermoregulatory mechanisms available to the particular biological entity.

Analysis of Literature Elucidation of the biologic effects of microwave exposure requires a careful review and critical analysis of the available literature. Such review requires differentiation of the established effects and mechanisms from speculative and unsubstantiated reports.

Although there is considerable agreement among scientists concerning the biological effects and potential hazards of microwaves, there are areas of disagreement. There also is a philosophical question about the definition of hazard. All effects are not necessarily hazards. In fact, some effects may have beneficial applications under appropriately controlled conditions. MW/RF induced

changes must be understood sufficiently so that their clinical significance can be determined, their hazard potential assessed and the appropriate benefit/risk analyses applied. It is important to determine whether an observed effect is irreparable, transient or reversible, disappearing when the electromagnetic field is removed or after some interval of time. Of course, even reversible effects are unacceptable if they transiently impair the ability of the individual to function properly or to perform a required task.

In an analysis of scientific literature to determine the probability of a biological response from exposure to a noxious agent, we must consider the consistency of experimental results claimed, both the nature of the response and the biological system involved, the ability to replicate the results of studies with consistency and whether the results claimed and observations reported can be explained by accepted biological principles.

Experiments with small animals, such as mice and rats, to evaluate the potential effects of MW/RF energy, must be carefully designed and performed. The responses may be the result of another, unrelated agent inadvertently introduced into the experimental design rather than the factor intended to be studied. The fact that a living organism responds to many stimuli is a part of the process of living; such responses are examples of biological "effects". Since biological organisms have considerable tolerance to change, these "effects" may be well within the capability of the organism to maintain a normal equilibrium or condition of homeostasis. If, on the other hand, an effect is of such an intense nature that it compromises the individual's ability to function properly or overcomes the recovery capability of the individual, then the "effect" should be considered a "hazard". In any discussion of the potential for biological "effects" from exposure to electromagnetic energies we must first determine whether any "effect" can be demonstrated; and then determine whether such an observed "effect" is "hazardous".

When assessing the results of research on biological effects of MW/RF exposure it is important to note whether the techniques used are such that possible effects of intervening factors i.e. noise, vibration, chemicals, variation in temperature, humidity, air flow are avoided and care is taken to avoid population densities that perturb the field to the extent that measurements become meaningless. The sensitivity of the experiment should be adequate to ensure a reasonable probability that an effect would be detected if indeed any exists. The experiment and observational techniques should be objective. Data should be subjected to acceptable analytical methods with no relevant data deleted from consideration. If an effect is claimed, it should be demonstrated at an acceptable level of statistical significance by application of appropriate tests. A given experiment, should be internally consistent with respect to the effect of interest. Finally, the results should be quantifiable and susceptible to confirmation by other investigators.

Principles of Biologic Experiments and Interpretation Proper investigation of the biologic effects of electromagnetic fields requires an understanding and appreciation of biophysical principles and "comparative medicine". Such studies require interspecies "scaling", the selection of biomedical parameters which consider basic

physiological functions, identification of specific and nonspecific reactions, and differentiation of adaptational or compensatory changes from pathological manifestations.

Scaling Much of the research on biological effects of microwaves has been done with small rodents that have coefficients of heat absorption, field concentration effects, body surface areas, and thermal regulatory mechanisms significantly different from man. Adverse reaction in animals does not prove adverse effect in man, and lack of reaction in animals does not prove that man will not be affected. Even closely related species can differ widely in their response. The literature is replete with "anomalous" reactions. Thus, results of exposure of common laboratory animals cannot be readily extrapolated to man unless some form of "scaling" among different animal species, and from animal to man, can be invoked in an accurate way to obtain a quantitatively valid extrapolation from the actual data observed.

The physical factors that must be considered include: frequency of radiation, intensity, animal orientation with respect to the source, size of animal with respect to the wavelength, portion of the body irradiated, exposure time-tensity factors, environmental conditions (temperature, humidity, air flow), and absorbed heat distribution in the body. In addition, variables such as restraint, metabolic rate, body ratio of volume/surface area, and thermoregulatory mechanisms will affect the biological response to microwaves.

The need for proper dosimetry in experimental procedures and the importance of realistic scaling factors required for extrapolation of data obtained with small laboratory animals to man are clearly required. Maximum absorption in man occurs at 80 MHz and falls off at higher frequencies. Formulas for scaling factors among species are available (1). Five milliwatts per centimeter square, 2450MHz, 5 mW/cm^2 exposure of a small animal such as a mouse or a rat, can result in a thermal effect that could influence the central nervous system and elicit behavioral and other physiologic responses in that animal, but not necessarily in a larger animal or man.

Epidemiology A number of retrospective studies have been done on human populations exposed or believed to have been exposed to MW/RF energies. Those performed in the U.S. (2,3) and Poland (4), have not revealed any relationship of altered morbidity or mortality to MW/RF exposure. Nervous system and cardiovascular alterations in humans exposed to microwaves has been reported in Eastern European literature (5,6,7). Most of the reported effects are subjective, consisting of fatigability, headache, sleepiness, irritability, loss of appetite, and memory difficulties. Psychic changes that include unstable mood, hypochondriasis, and anxiety have been reported. The symptoms are reversible, and pathological damage to neural structures is insignificant. There is considerable difficulty in establishing the presence of, and quantifying the frequency and severity of "subjective" complaints. Individuals suffering from a variety of chronic diseases may exhibit the same dysfunctions of the central nervous and cardiovascular systems as those reported to be a

result of exposure to microwaves; thus, it is extremely difficult, if not impossible, to rule out other factors in attempting to relate microwave exposure to clinical conditions.

Protection Guides and Standards: Exposure Standards The first standards for controlling exposure to MW/RF were introduced in the 1950's, in the USA and the USSR. The maximum permissible exposure levels proposed then have remained substantially unchanged, i.e., for continuous exposure these are respectively 10 mW/cm^2 and 10 μW/cm^2. Most countries that developed national standards based them on either the US (8) or the Soviet (9) values. Subsequently, however, some countries have proposed standards intermediate between these extremes.

Current U.S. Government Standards Include: Occupational Standard (general) - OSHA Standard, adopted in 1972, applies to employees in the private sector. An addendum, adopted in 1975, applies to work conditions particularly in the telecommunications industry. OSHA Standards are mandatory for federal employees including the military. Maximum permissible exposure limit is 10 mW/cm^2, for durations greater than 6 min, over the frequency range 10 MHz-100 GHz.

Product Emission Standard - "Radiation Control for Health and Safety Act of 1968" (PL 90-602), administered by HEW/FDA (BRH), provides authority for controlling radiation from electronic devices. BRH microwave oven standard, effective October, 1971: Ovens may not emit (leak) more than 1 mW/cm^2 at time of manufacture and 5 mW/cm^2 subsequently, for the life of the product--measured at a distance of 5 cm and under conditions specified in the standard.

Additionally, there are non-goverment organizations which develop recommended standards and safety criteria, e.g.: American National Standards Institute (ANSI) - A voluntary body with members from government, industry, various associations and the academic community which develops consensus standards (guides) in various areas. ANSI issued a nonionizing radiation safety standard in 1966 with maximum permissible exposures of 10 mw/cm^2, as averaged over any 6 minute period, for frequencies from 10 MHz to 100 GHz which was essentially adopted by USHA. This standard was reviewed and reissued with minor modifications in 1975. ANSI must review and withdraw, revise or reissue ANSI Standards every five years. Presently Subcommittee 4 of ANSI Committee C-95, which deals with hazards to personnel is reevaluating ANSI's radiofrequency exposure standard for adoption in 1980. The recommendations, based on frequency dependence and specific absorption rates (SAR) state: For human exposure to electromagnetic energy of radiofrequencies from 300 KHz to 100 GHz, the radiofrequency protection guides, in terms of equivalent plane wave free space power density, and in terms of the mean squared electric (E^2) and magnetic (H^2) field strengths as a function of frequency, are:

Frequency (MHz)	Power Density (mW/cm^2)	E^2 (V^2/m^2)	H^2 (A^2/m^2)
0.3 - 3	100	400,000	2.5
3 - 30	$900/f^2$	$4{,}000\ (900/f^2)$	$0.025\ (900/f^2)$
30 - 300	1.0	4,000	0.025
300 - 1500	f/300	4,000 (f/300)	0.025 (f/300)
1500 - 100,000	5	20,000	0.125

Note: f is the frequency, in megahertz (MHz)

For near field exposure, the only applicable radiofrequency protection guides are the mean squared electric and magnetic field strengths given in columns 3 and 4. For convenience, these guides may be expressed in equivalent plane wave power density.

For both pulsed and non-pulsed fields, the power density and the mean squares of the field strengths, as applicable, are averaged over any 0.1 hour period and should not exceed the values given in the Table. For situations involving exposure of the whole body, the radiofrequency protection guide is believed to result in energy deposition averaged over the entire body mass for any 0.1 hour period of about 144 joules per kilogram (J/kg) or less. This is equivalent to a specific absorption rate (SAR) of about 0.40 watts per kilogram (W/kg) spatially and temporally averaged over the entire body mass. This recommendation will no doubt be adopted with only minor modifications if any.

The National Institute for Occupational Safety and Health (NIOSH) is developing a criteria document with recommended standards for occupational MW/RF exposures, which is, except for certain modifications, comparable to that recommended by ANSI.

The Radiation Protection Bureau of Health and Welfare Canada is considering "Emission and Exposure Standards for Microwave Radiation". The maximum permissible levels (MPL'S) are 1 $mW\text{-}hr/cm^2$ average energy flux for whole body exposure as averaged over an hour and a maximum exposure during any one minute of 25 mW/cm^2 for occupational settings. The MPL's would apply for the frequency range of 10 MHz-300 GHz. No distinction is made between CW and pulsed waveforms. There is no lower MPL for the general population.

The State Committee on Standards of the Council of Ministers of the USSR has promulgated "Occupational Safety Standards for Electromagnetic Fields of Radiofrequency (GOST 12.1.006-76)," effective January 1, 1977. It specifies the maximum permissible magnitudes of voltage and current density of an EM field in the workplace. It does not apply to personnel of the Ministry of Defense. Maximum permissible RF fields in the workplace must not, during the course of the workday, exceed;

P (mW/cm^2)		E (V/m)	H (A/m)	Frequency Range
Stationary Source (See Note 1)	Rotating, Scanning		5	60 KHz-1.5 MHz
		50		1.5 MHz-3.0 MHz
		20		3.0 MHz-30 MHz
		10	0.3	30 MHz-50 MHz
		5		50 MHz-300 MHz
0.01	0.1	(entire workday)		300 MHz-300 MHz
0.10	1.0	(2 hr. period during workday)		
1.00	(20 min. period during workday)			

Note 1: Also applies in environments with ambient temperatures above 28°C and/or in the presence of X-ray radiation, except, under these conditions, the maximum during a 20 minute period is restricted to 0.1 mW/cm^2.

There is some indication that the USSR Ministry of Health has endorsed guidelines for maximum exposure limits for the general population which stipulates the maximum allowable levels of electromagnetic energy in human dwellings or in areas of human dwellings, as follows:

P ($\mu W/cm^2$)	E (V/m)	Frequency Range
	20	30-300 KHz
	10	300 KHz-3.0 MHz
	4	3.0-30 MHz
	2	30-300 MHz
5		300 MHz-300 GHz

In 1977, the Polish Ministries of Work, Wages and Social Affairs and of Health and Social Welfare promulgated a change in the Polish Standard for occupational exposure. The change extends the frequency range down from 300 to 0.1 MHz;

Hazardous Zone II	Hazardous Zone I	Intermediate Zone	Safe Zone	Frequency Range
Tp	Tp	Tp	Tp	
0	250 A/m 1000 V/m 150/E	40/H 10 A/m 70 V/m	Entire 8 hr workday	2 A/m No limit 20 V/m	0.1-10 MHz
	300 V/m 3200/E2	20 V/m		7 V/m	10-300 MHz

Tp = Permissible time of exposure/workday (minutes).
E = Electric field (volts/meter).
H = Magnetic field (amps/meter).

In 1976, the Swedish National Board for Industrial Safety, promulgated a nonionizing radiofrequency standard (Worker Protection Authority Instruction No. 111) effective January 1, 1977. This regulation applies to all work which may involve exposure to radiofrequencies between 10 MHz and 300 GHz. The instruction specifically excludes applications involving the treatment of patients. Maximum permissible exposures (as averaged over a six minute period) are:

Power Density	Frequency Range		
5 mW/cm^2	10 MHz	to	300 MHz
1 mW/cm^2	300 MHz	to	300 Ghz

The maximum permissible momentary exposure is 25 mW/cm^2.

Emission Standards The best known emission standards concern the maximum permissible leakage from microwave ovens. The Canadian standard (10,11) restricts the maximum leakage to 1 mW/cm^2 at 5 cm from the oven (consumer, commercial and industrial). The U.S. standard (12) specifies a maximum emission level at 5 cm of 1 mW/cm^2 before purchase and 5 mW/cm^2 thereafter which is consistent with standards for the general population in the USSR and Poland (13). The standard applies to domestic and commercial ovens, but not to industrial equipment. This has been adopted in Japan and most of Western Europe.

Conclusion

International cooperation in the development of compatible standards should be encouraged. Towards this end the International Radiation Protection Association (IRPA) charter was broadened in April 1977, to include nonionizing radiation. IRPA has cooperated with the World Health Organization (WHO) in the preparation of a criteria document which is scheduled for 1980. The European Regional Office (ERO) of the World Health Organization is presently preparing a manual on health aspects of exposure to nonionizing radiation. The manual is intended to provide guidance in nonionizing radiation protection and to summarize international experience in the field. Among the topic areas to be included are health aspects of ultraviolet, optical, infrared and laser; microwave RF and ELF fields; ultrasound; licensing, legislation and regulations.

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BETA RESPONSE OF TL PERSONNEL AND ENVIRONMENTAL SURVEILLANCE DOSE METERS.

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INTRODUCTION

Conceptional practice in TL personnel and environmental monitoring is based on separate estimation of the socalled skin dose and penetrating dose. The magnitude of skin dose usually to be assessed behind a layer of 5 to 10 mg/cm^2, is essentially influenced by the correct determination of beta radiation, if present. However, only few data are available on the beta response of TL detectors. For reasons of pronounced absorption and scattering of beta particles in matter, the beta energy and detector thickness are decisive for the response. Since detectors commonly used for skin dose assessment are extended in thickness up to 1 mm and covered with plastic foils of up to several 10 mg/cm^2 for protection purposes, serious underestimation of beta doses to the skin may occur. In order to provide basic data on this subject extensive investigations were made.

CALIBRATION

For assessment of the beta field parameters the GSF beta calibration set-up has been used which is based on a commercial PTW extrapolation chamber. The extrapolation chamber allows the absorbed dose in a solid medium to be determined from the ionization in a small gas-filled cavitity inside the medium by extrapolation to zero volume. The measuring devices for determination of ionization current and chamber volume were of high metrological quality. The chamber factors were determined for three beta energies using a beta secondary standard elaborated by PTB (Table 1)/1/. The measuring quantity is absorbed dose in tissue/2,3/. All irradiations were performed in free air and source distances of 11 cm, 20 cm or 30 cm depending on source type (see table 1). There is a 2π field geometry. Compensation filters of perforated plastic foils provide for field homogeneity within a diameter of 11 cm. Reproducibility of irradiation geometry was around 0,01 cm by means of an optical bench system. The reliability of the measuring device was checked by taking up depth dose profiles and comparing them with data from PTB. Excellent agreement was

TABLE 1: Properties of the beta secondary standard at GSF (Data for 27.7.1978)

Radioactive source	Activity mCi	$t_{1/2}$ a	E_{max} MeV	$\bar{E}$ MeV	$\dot{D}_{tissue}$ (d=0) rd/h	Calibration distance cm
Pm-147	14	2.62	0.225	0.06	0.161	20
Tl-204	0.5	3.78	0.763	0.24	0.082	30
Sr-90/Y-90	2.0	28.5	0.546	0.8	0.849	30
Sr-90/Y-90	50.0	28.5	2.274	0.8	18.731	30
					208.120	11

found even for the low-energy Pm-147 (Fig. 1). With the described facility the TL dose meters have been calibrated. The TL readout values were interpreted in terms of absorbed dose in soft tissue of cobalt-60 gamma radiation under CPE conditions.

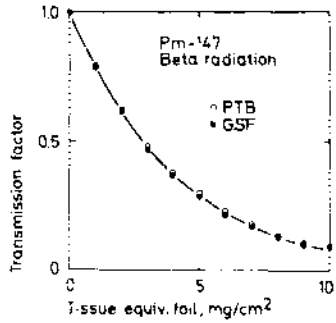

Fig. 1 Comparison of the PTB and GSF central axis beta transmission factors for tissue-equivalent foils. Source Pm-147.SSD 20 cm. Large field geometry.

RESULTS

Table 2 shows the relative energy dependence of the beta response of different TL detector materials each 1mm in thickness. The detectors were exposed free-in-air without any cover. The response of the three detector materials behaves similarly according to similar electron density. From the results, the TL detectors underestimate the beta doses by a factor 2 in case of Sr-90/Y-90, by a factor 5 in case of Tl-204 and by a factor 20 to 30 in case of Pm-147.

In table 3 the relative beta response of TL detectors with comparable effective atomic numbers and density are listed as a function of detector thickness and beta energy. Only the thin-layer detector of $CaSO_4$: Tm provides a dosimetry acceptably independent of the beta energy with a tolerable dose underestimation of about 40% in case of Pm-147.

TABLE 2: Relative response of different TL detectors as a function of beta energy. Detector thickness 0.9 mm. Readout calibration absorbed dose in soft tissue for Co-60 CPE. Beta dose 1 rad in soft tissue.

Detector material	ratio indicated dose/nominal dose Sr-90/Y-90	Tl-204	Pm-147
LiF: Mg, Ti	0.5	0.2	0.03
LiF: Mg, Ti+graphite	0.5	0.2	0.05
CaF_2: Dy	0.5	0.2	0.03

TABLE 3: Relative response of high-Z TL detectors as a function of detector thickness. Parameter beta energy. Readout calibration absorbed dose in soft tissue Co-60 and CPE. Beta dose 1 rad in soft tissue.

Detector material	Detector thickness, mm	ratio indicated Sr-90/Y-90	dose/nominal dose Tl-204	Pm-147
CaF_2: Mn	0.90	0.5	0.2	0.03
CaF_2: Mn	0.25	0.7	0.5	0.08
$CaSO_4$: Tm	0.03	1.0	1.0	0.60

The beta transmission factors for hostaphane foils up to 20 mg/cm^2 are given in table 4 as a function of foil thickness and beta energy. The measurements were performed with the extrapolation chamber. All data given are extrapolated to zero thickness of the entrance foil of the chamber. The results provide information about the effect of cover material to TL detectors for purposes of detector protection or dose assessment to the basal layer of the skin. As a result, buildup is found for Sr-90/Y-90 while there is attenuation for Tl-204 and that particularly pronounced for Pm-147. It is worth mentioning that for Pm-147 only 15% of the beta radiation is transmitting a 7 mg/cm^2 tissue layer.

TABLE 4: Beta transmission factors for hostaphane foils as a function of foil thickness. Beta dose 1 rad in soft tissue. Large field geometry.

Area weight mg/cm^2	Transmission factor Sr-90/Y-90	Tl-204	Pm-147
4	1.018	0.97	0.33
7	1.023	0.94	0.15
10	1.025	0.91	0.08
20	1.028	0.75	0

DISCUSSION AND CONCLUSION

In cases where TL dose meters based on photon calibration are applied for skin dose estimation resulting from beta radiation, the detector thickness must be adopted to the beta energies of interest. Definitly, the commonly used 0.9 mm thick detectors will seriously underestimate beta doses up to a factor 5 for Tl-204 and, hence, do not seem appropriate for the measuring task. On the other hand, taking the thin-layer detector $CaSO_4$: Tm as an example it becomes evident that energy-independent beta dosimetry even down to Pm-147 with its mean energy around 60 keV is possible with TLD and that behind skin layers $\geq 2 mg/cm^2$. This finding has been confirmed by the results of national and international intercomparisons with unknown beta energies and doses. As is shown in table 5, maximum uncertainties were within a margin of ± 20% or 13% excluding 2 NPL values) on a 95%

confidence level and that based on home calibration. More recent studies concern low-Z thin-layer detectors which estimate beta and equally phonton doses energy independently. The results will be published elsewhere.

TABLE 5: Beta intercomparison results based on GSF calibration (+NPL calibration)

Beta source	nominal dose, mGy	ratio evaluated dose/nominal dose GSF/LMRI	GSF/NPL	GSF/PTB
Sr-90/Y-90	1.6			0.94
	2.43		1.15	
	19.4		1.06	
	20.0			1.06
	30.9	1.01		
	51.5	1.02		
	49.8		1.06	
	103.1	0.99		
	120.0			1.05
Tl-204	2.12		0.94	
	2.4			0.88
	12.0			1.03
	16.0			1.01
	16.9		1.00	
	33.8		1.04	
Pm-147	1.6			1.00
	2.26		0.93	
	5.0			0.92
	6.02		0.78(0.96+)	
	8.13			0.92
	18.4		0.78(0.96+)	
mean value		1.00	0.97(1.01+)	0.98
stand.dev.		0.01	0.13(0.07+)	0.07

Doubtlessly, beta dosimetry plays up to now a minor role in personnel monitoring and environmental surveillance. The reason is not at all a neglegible importance of beta radiation from the point of view of radiation protection, but should be seen in the more difficult measuring technology compared with photon dosimetry. The presented results on beta dosimetry with TL detectors promise valuable improvement in this field, in future.

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5th INTERNATIONAL CONGRESS OF THE INTERNATIONAL RADIATION PROTECTION ASSOCIATION

BOOK OF PAPERS

VOLUME III

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JUSTIFICATION AND OPTIMIZATION IN RADIATION PROTECTION

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1. INTRODUCTION

1. To meet the objectives of radiation protection the ICRP (1) has recommended the use of a system of dose limitation composed of the following requirements: 1) Justification of practices involving radiation exposures; 2) Optimization of the level of protection for such practices; 3) Individual dose limitation. The third requirement is individual-related, and is the continuation of previous recommendations limiting the risk to individuals from exposure to radiation. The first two requirements, on the other hand, are source-related. They apply even if all individuals are so well protected that their risk is negligible, requiring that the radiation detriment from a given source be reduced by increasing protection to the optimum level, and that the practice (with its remaining radiation detriment) be justified by benefits.

2. The ICRP has recommended the use of cost-benefit analysis in the assessment of justification of practices involving radiation exposures and in the optimization of radiation protection (1). The concept of "net benefit" from the introduction of a practice involving radiation exposures was defined in that publication in symbolic form as:

$$B = V - (P + X + Y)$$

where B is net benefit from the introduction of the practice, V is the gross benefit, P is all production costs excluding protection costs, X is the cost of achieving a selected level of protection and Y is the cost of detriment associated with that level of protection.

2. QUANTIFICATION OF THE RADIATION DETRIMENT

3. The application of cost-benefit analysis requires the assignment of quantitative values to X and Y in all cases, and for some applications to V and P. While P and X costs are readily expressed in monetary terms, V may contain components difficult to quantify. The quantification of Y, the cost of the radiation detriment, is regarded as the most problematic and the most controversial of the quantifications. Nevertheless, it is essential for the application of cost-benefit analysis to radiation protection.

4. Optimization of protection takes place in a region of low individual doses, always smaller than a fraction of the dose limits. Therefore, only the induction of somatic and genetic stochastic effects of radiation will contribute to the deleterious health consequences, as non-stochastic effects would be totally prevented. In order to deal with the risk of stochastic effects, the ICRP uses the quantity:

$$H_E = \sum_T w_T H_T$$

where H_E is a sum of weighted organ dose equivalents, called the "effective" dose equivalent", w_T is a factor representing the fraction of risk resulting from tissue T when the whole body is irradiated uniformly, and H_T is the dose equivalent in tissue T. The recommended values of w_T are given in ICRP publication 26; and additional value for skin exposures has also been provided by the ICRP (2).

5. The "detriment" in an irradiated population group is defined as the expectation of the harm incurred, taking into account not only the probabilities of each type of deleterious effect but also the severity of the effects. If P_i is the risk of suffering the effect i, the severity of which is measured by a factor g_i, then the detriment G in a group of N persons is $G = N \sum_i P_i g_i$.

6. For stochastic effects it is assumed that increments of risk are proportional to increments of dose. Then p_T, the probability of suffering a stochastic effect in tissue T can be taken to be proportional to the average dose received in that tissue

$$p_T = r_T H_T$$

r_T being a risk factor per unit dose equivalent. When this is substituted into the equation for detriment, the detriment of one person is given by

$$G_1 = \sum_T r_T H_T g_T$$

Several approaches are possible to quantify the severity factors g_T (3). For radiation protection purposes it could be assumed as a first approximation that the detriment is dominated by the induction of fatal malignancies and of servere genetic effects in the first two generations, assigning a severity factor of one to all these effects. In this case, the effective dose equivalent would be proportional to the individual health detriment because

$$G_1 = \sum_T r_T H_T = R H_E,$$

where R is the total risk for whole body irradiation, and $w_T = \frac{r_T}{R}$. The value of R is taken to be 1.65×10^{-2} Sv^{-1} (3).

7. This first approximation, however, neglects the contribution to the detriment of subsequent generations after the second, and of non-fatal malignancies, which are not taken into account in the definition of effective dose equivalent. The contribution of subsequent generations to the detriment could be roughly taken into account by adding a term $w_{gon} H_{gon}$ to the effective dose equivalent. Non-fatal malignancies could probably be neglected when compared to fatal malignancies. Several attempts to quantify their contribution, which are very controversial, support the idea of neglecting such contribution to the detriment.

8. The detriment is an extensive quantity. The detriment from a given source is therefore the summation of the detriments of all individuals irradiated by the source, either at present or in

the future. It follows (4) that the detriment from the source k, G_k, is given by

$$G_k = R \sum_i N_i \bar{H}_{E,i} = R.S^c_{E,k}$$

where R is the risk factor for whole body irradiation, N_i is the number of individuals receiving an average effective dose equivalent $\bar{H}_{E,i}$ from the source, and $S^c_{E,k}$ is the collective effective dose equivalent commitment from the source.

9. As the detriment is an expectation of death (and of serious genetic harm), the assignment of a cost to the detriment involves some valuation of human life. In fact, countless policy decisions affect the incidence of death and none tries to minimize this incidence regardless of cost. Implicit in any of such decisions, therefore, is some valuation of human life.

10. A key feature of the modern approach for taking account of life in cost-benefit analysis is that it does not value life as such, but only changes in the probability of death. Being the detriment a mathematical expectation of death, the assignment of a cost to the detriment would fit well with the quoted approach. As the detriment is proportional to the collective effective dose equivalent commitment, the problem reduces to the assignment of a monetary value to the unit of collective effective dose equivalent. Obviously, this assignment is a value judgement rather than a scientific determination. It has been attempted by assigning values to the increased probability of death, or by observation of the values society actually is willing to pay to reduce exposures in given practices.

11. With the first approach, values ranging from 20 to 200 dollars per man rem can be deduced from assessments of "cost of a statistical life" and a risk of 1 to 2 10^{-4} per rem. The second approach gives somewhat higher values for a man rem, up to a few hundred dollars. A value of about 100 to 200 dollars per man rem seems to be adequately representative, and could be used for planning purposes in those cases where the competent authority has not yet established the value to be used.

12. For the purpose of cost-benefit analysis in radiation protection, therefore, the cost of the detriment can be expressed as:

$$Y = \alpha S^c_E$$

where Y is the cost of detriment, α is the monetary cost assigned to the unit of collective effective dose equivalent and S^c_E is the collective effective dose equivalent commitment associated with the level of protection under consideration.

13. Problems associated with costs and detriments occurring over different time periods are frequent, especially when a practice leads to environmental contamination by long lived radionuclides and therefore to subsequent exposure in future populations. The concept of collective effective dose equivalent commitment allows the calculation of detriment in these cases giving the same weight to present and future detriments, which is not the usual practice in

other types of human judgements, which involve the traditional economical technique of discounting.

14. However, on ethical grounds it has been argued that discounting perhaps be properly applied within the time period of one generation, but that it should not be applied when a substantial part of the detriment will occur in future generations. Some have also expressed the opinion that it is not valid to discount the cost of the detriment (even if manifested in the future) committed from one year of practice, because only the present decision was relevant and the future harm was unavoidable. However, it would be legitimate to discount the cost of the detriment committed successively year after year of the practice.

3. OPTIMIZATION

15. A basic requirement of radiation protection is that all doses should be kept "as low as it is reasonably achievable", taking into account social and economical considerations. This requirement is usually called "optimization" of radiation protection and consists in reducing the collective dose (and thus the detriment) to a value such that further reductions are less significant than the additional efforts required to achieve such reductions.

16. Optimization, therefore, consists in an interplay of the cost of protection and the cost of the remaining detriment, in such a way that

$$X(w) + Y(w) = \text{minimum}$$

where X is the cost of protection, and Y is the cost of the radiation detriment, both at a level of protection represented by w (e.g., shielding thickness, ventilation rate, alternative options of protective equipment, etc.). It should be noted that w, and X(w) and Y(w), can in some cases be continuous, while in other cases they take only discrete values. It is obvious that the selection of the optimum pair of values for X and Y, would maximize the "net benefit" from the introduction of the practice, as defined in paragraph 2.

17. Some of the technical difficulties of optimization are related to the boundary condition introduced by the dose limits. As the limits apply to the combined exposure from all sources (except those specifically excluded), it is necessary to use a fraction of the limit as a boundary condition for the optimization of a given source. It is not the purpose of this paper to review the criteria to set such source upper bound, L, but to show its use as a boundary condition.

18. In the ideal optimization case, there is only one exposed group of individuals and one protection parameter or a simple set of protection options. Additionally, a basic requirement of this ideal case is the existence of a quantitative relationship between the collective effective dose equivalent commitment S, and the maximum annual effective dose equivalent, H*, such as H* = f (S). Taking the detriment to be proportional to the collective dose (paragraph 12) and using the symbols defined previously, optimization in the ideal case can be expressed as the set of conditions (4)

$$(1)\quad X(w) + \alpha\, S(w) = \text{minimum}$$
$$(2)\quad f(S) \leq L.$$

19. The minimum for the first expression, usually called the objective function, can be obtained by differentiation and making the result equal to zero:

$$\frac{dX}{dw} = -\alpha \frac{dS}{dw}\ , \text{ or as usually presented,}$$

$$\left(\frac{dX}{dS}\right)_{S_o} = -\alpha$$

The optimized value S_o correspond to a given optimum protection parameter w_o and a given protection cost, S_o, because X can be expressed as a function of S, the function being called the constraining function.

20. The optimized value S_o must, however, comply also with the second condition of paragraph 18, namely the limit equation $f(S_o) \leq L$. Therefore, optimization is achieved at a value of collective effective dose equivalent commitment, S_o, such that

$$\left(\frac{dX}{dS}\right)_{S_o} = -\alpha$$

provided that $f(S_o) \leq L$, and at a value $S_o = f^{-1}(L)$ in all other cases.

21. Examples of application of this procedure of optimization have been published for radiation shielding and for ventilation design in installations handling radioactive materials, in uranium mines and in buildings (in relation to radon) (4)(5) (6). In many other practical cases of optimization assessments, the changes in protection levels are achieved in finite increments, both X and S being discrete instead of continuous variables. The decision of going from a level of control A to a more expensive level of control B would be taken if

$$-\frac{X_B - X_A}{S_B - S_A} \leq \alpha$$

Examples of application of this step by step procedure have been published relating to the control of release of radioactive effluents (4) (5) (6).

22. When exposures from a given source or practice can be regarded as composed of contributions of subsystems, each requiring appropriate protection measures, optimization implies that

$$\sum_j (X_j + \alpha\, S_j) = \text{minimum}$$

where X_j is the cost of protection of sub-system j, S_j is the collective effective dose equivalent commitment resulting from sub-system j when its cost of protection is X_j, and α is the monetary value per unit collective effective dose equivalent.

23. Optimization procedures in this situation can be complicated. In one case, however, the constraining functions in the optimization procedures can be readily established, namely when the sub-systems j are independent, in the sense that the control in one of them does not influence the collective effective dose equivalent commitments from the others (4). In this case, differentiating the objective function with respect to each S_j and making each result equal to zero, the following set of equations are obtained for j = 1, 2, ... n

$$\frac{dX_j}{dS_j} + \alpha = 0$$

because for all X_i and S_i where $i \neq j$, the derivatives are equal to zero ($\frac{dX_i}{dS_j} = 0$ and $\frac{dS_i}{dS_j} = 0$), due to the independence of the sub-systems.

24. As individual annual doses should not exceed the operational limit, a further set of equations (limit equations) are obtained (4)

$$f_j (S_j) \leq L$$

It follows from both sets j of equations that the optimization of control can be obtained by optimizing each independent sub-system taken separately. Similarly, the optimization for the combined exposures from several installations at a given site can be obtained by optimizing separately the protection at each installation, provided the condition of independence applies.

25. In cases where the sub-systems are not independent, optimization procedures can be difficult. The protection to be optimized can conceptually be divided into sub-systems while the exposed group can be conceived as composed by sub-groups. After establishing the Objective functions, Constraining functions and Limit equations, if the number of sub-systems and sub-groups is small, the solution can be obtained analytically (7). However, in most cases, the number of variables will not be too large and programming or direct search methods will have to be used (8).

26. A word of caution is necessary presenting the quantitative techniques of optimization. It should be recognized that optimization of radiation protection, as optimization in engineering in general, is basically an intuitive process (8). The quantitative techniques discussed above are a substantial aid to the process of optimization, but are not the complete process itself.

4. JUSTIFICATION

27. The justification of a proposed practice or operation involving exposure to radiation could be determined by consideration of the

advantages and disadvantages to ensure that there will be an overall net advantage from the introduction of the practice. Justification assessments would be required to decide the introduction of a given practice or to select one among many options. The first type of decision is really a particular case of the second, one of the options being not to change the present situation.

28. The decision among several options, the first being not to introduce any new practice, could conceptually be based on a cost-benefit analysis, as indicated in paragraph 2. The basic notion in the application of cost-benefit analysis to such decisions is very simple: a course of action is taken if the resulting net benefit exceeds those of the next best alternative, and not otherwise. Calling the options $i = 1, 2, \ldots n$ and noting with $i = 0$ the decision to introduce no change, then the options would be increasingly justifiable at increasing positive values of the net benefit B_i

$$B_i = (V_i - V_o) - (P_i - P_o) - (X_i - X_o) - (Y_i - Y_o)$$

where the symbols have the same meaning than in paragraph 2.

29. The justified option, B_j, would then be such that

$$B_j = \max (B_i)$$

In practice, the existence of intangible costs and benefits in many cases makes the analysis subjective. However, relative assessments comparing the justification of alternative procedures are simpler, because the same gross benefit is involved. It is apparent from the equations, that in the very simple case of only two options, differing only in the level of protection, the justification assessment becomes identical to optimization.

30. Acceptance of a practice or the choice between practices will depend on many factors, only some of which being associated with radiation. The role of radiation protection in justification procedures is to ensure that the radiation detriment is taken into consideration, and that the comparisons between practices are made after having applied the procedure of optimization to each of them.

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THE COST OF OCCUPATIONAL DOSE

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INTRODUCTION

The current system of dose limitation recommended by ICRP (1) is such that compliance with dose equivalent limits is a necessary but not sufficient criterion for radiological protection; the emphasis is instead on the concepts of justification and optimisation. The result is that the 'as low as reasonably achievable' (ALARA) principle has become a primary objective for radiation protection when dealing with justified sources of exposure.

For the vast majority of day to day problems concerning occupational exposure, the ALARA principle can be satisfied by using the intuitive judgement of operational health physicists. A formal analysis employing the cost benefit techniques suggested by ICRP will not be warranted by the scale of the problem. However in circumstances where there may be potentially large exposures and various possible ways to reduce them, the use of such an analysis can be a useful input into the required decision making. The cost benefit technique can, in theory, identify optimum exposures ie, the level of exposure below which further reductions would not be justified. Nevertheless, in order to perform the analysis in practice, a monetary valuation of radiation exposure is required so that the cost of detriment, Y, can be made directly commensurable with the cost of protection, X. The problems of assigning a cost to health detriment for public exposure have been examined by the authors elsewhere (2). In this paper some aspects of the corresponding costing for occupational exposure will be discussed.

THE VALUATION OF DETRIMENT

The ICRP have defined the health detriment from radiation exposure as a mathematical expected value ie, a summation of the product of the frequency of (stochastic) health effects and weighting factors for their severity. Assuming a linear dose response relationship for the stochastic health effects and the homogeneity of risk and severity factors in populations it is possible to show that the health detriment is proportional to the collective dose equivalent, S (3). There is therefore a simple proportional relationship between collective dose and the number of predicted health effects. However, to establish a relationship between the cost of the health detriment, Y, and collective dose, a separate judgement is required. It has generally been assumed that Y is also simply proportional to collective dose, ie. $Y = \alpha S$ where α is "the cost of the man Sv" in £ man-Sv^{-1}. The use of this relationship implies a single monetary valuation of stochastic health effects independent of the level of individual risks involved. Without questioning the assumption of proportionality between dose and health effects, it is important to note that this does not automatically lead to a proportional relationship between Y and S; other relationships are possible and the

appropriate choice is a matter of judgement.

One alternative is to use a cost benefit approach to valuing risk changes (4) which explicitly considers the size of population at risk and the significance of the risk increment to individuals. This leads to a variable value for α for public exposure which increases with increasing per caput dose (2). Such an approach to the functional form of α will tend to concentrate (limited) protection resources in areas of high individual risk; it can therefore be shown to be consistent with equity considerations and has a strong intuitive appeal. The use of a variable value in optimising occupational protection could be justified on the same criteria, although there may be other criteria that need to be considered, both in general and on a case by case basis.

IMPLICATIONS FOR OPTIMISATION

As previously stated the choice between a fixed or a variable α to convert health detriment into monetary terms is a matter of judgement. Nevertheless this judgement can be shown to have important implications for the optimisation of occupational exposure which arises from a common set of operational conditions; namely where the principal mechanism for controlling individual exposure is to vary the number of workers, N, employed on a specific task. Typically one may assume that increasing the number of workers will reduce average individual doses, H, for example by reducing the average time necessary for each worker to spend in radiation areas. However it would appear that this increase in the number of workers will be accompanied by a general increase in doses resulting from non-productive work (5). In the previous example this might arise during the entry and exit from radiation areas. Increasing the number of workers will, in general, tend to increase both the total time required to complete any given task and the total non-productive dose. Assuming that there is a fixed dose associated with the task itself, this will typically lead to an increase in collective dose, S. In order to fulfil the ALARA principle under these conditions, it is necessary to assess what is the optimum exposure to be associated with the task.

In accordance with the formal optimisation procedure recommended by ICRP, the solution to this problem is that at which the sum of the protection costs and the detriment costs (X + Y) for each feasible level of manpower, is minimised. If a fixed value for α is employed in the analysis then Y must be at a minimum for the option which results in the lowest collective dose. On the basis of the general assumptions outlined above, this will occur where the minimum number of workers are assigned to the task and the average individual dose is at its highest (within the constraint of the dose limits). Moreover, as the costs of protection will generally increase if there are more workers requiring, for example, specialist training or protective equipment for the task, this option is likely to also minimise X, and will therefore appear to be optimum. Thus whenever these general assumptions concerning N,H,S and X apply to actual operational conditions, the optimisation procedure will consistently advocate options characterised by the smallest collective dose and the smallest feasible number of workers and will

involve the highest resultant average individual exposure. Indeed if the required data validate these relationships between N,H,S and X for a given situation, then the analysis itself is unaffected by the precise value assigned to α.

The results of an optimisation using a variable value for α provides a significant contrast. Even where the same postulated relationships between N,H,S and X hold, the increasing valuation of α with increases in average individual dose precludes any automatic relationship between reductions in detriment costs and reductions in collective dose. Thus while the minimum S and highest H option will still minimise X, it may no longer minimise Y. The optimisation of any given task will therefore be crucially dependent on the specific relationship arising between N,H,S and X and the numerical relationship between α and H.

SUMMARY AND CONCLUSIONS

The implementation of the ALARA principle within the workplace is intended to oppose the attitude that a worker's dose limit represents a constraint on a 'resource' which until reached, can be fully utilised in an arbitrary manner. Thus while the method for formal optimisation is based on the parameter of collective dose, the ICRP suggest that individual exposure at or near the limit is only acceptable if justified by "a careful cost benefit analysis" (1). Such an analysis will generally consider the relationships between the number of workers involved, the costs of their protection and the resultant individual and collective doses. Moreover, it seems likely that the optimum solutions will be significantly influenced by the manner in which collective doses are converted into monetary terms. A fixed conversion factor will ignore individual dose levels, and in attempting to reduce both protection and detriment costs will tend to increase average individual doses. A variable conversion factor which increases with increasing average individual doses will, in contrast, explicitly account for the distribution of individual doses in the assessment of detriment costs, will discriminate against high individual doses, and will tend to select options on a more case by case basis. Such an approach is fully in the spirit of the ALARA principle as applied to all exposures, both individual and collective.

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OPTIMISATION ET CONTROLE DES REJETS RADIOACTIFS DES CENTRALES A EAU PRESSURISEE DU PROGRAMME ELECTRONUCLEAIRE FRANCAIS.

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Les dernières recommandations de la CIPR[1] proposent d'introduire dans le processus de décision concernant la fixation des niveaux de protection dans les installations électronucléaires, la notion d'optimisation.

Dans l'état avancé de développement de certains programmes électronucléaires nationaux, et c'est le cas en particulier pour la France, les choix de radioprotection ont été arrêtés antérieurement et indépendamment de règles ou de directives issues plus ou moins directement de l'esprit de la CIPR. L'analyse de ces choix selon la démarche méthodologique préconisée par la CIPR n'est cependant pas dénuée de tout intérêt.

L'analyse des divers systèmes de traitement des effluents liquides et gazeux équipant actuellement les installations doit en effet permettre de situer les niveaux atteints en matière de protection par rapport à ce qu'il convient d'appeler les niveaux ALARA.

L'étude présentée ici concerne une telle analyse pour les systèmes de traitement des effluents radioactifs des centrales à 300 MWe à eau pressurisée caractéristiques du programme électronucléaire français.

METHODE ET CADRE D'ANALYSE.

Afin de pouvoir comparer et classer les différents systèmes de traitement, il est nécessaire d'évaluer l'impact économique et sanitaire associé à chaque système. Ces évaluations ont été effectuées à l'aide d'une série de modèles reposant à la fois sur des données réelles et sur des hypothèses[2].

Les quantités rejetées de radionucléides dans l'environnement sont évaluées à partir des hypothèses concernant le fonctionnement du réacteur en marche normale et les efficacités théoriques des systèmes de traitement des effluents liquides et gazeux, telles que les admettent les exploitants dans les études préalables à la mise en route des réacteurs. (Rapport de Sûreté - Demande d'Autorisation de Rejets).

Le tableau 1 précise les principales caractéristiques de fonctionnement des réacteurs à 300 MWe et les activités rejetées après traitement. Le tableau 2 donne pour les effluents traités le type de traitement associé, chaque traitement étant caractérisé par un facteur de décontamination (FD) et/ou une durée de stockage (D).

Pour l'évaluation des effets sanitaires associés aux rejets, un site particulier dans la vallée du Rhône, jugé représentatif de l'implantation actuelle des centrales françaises, a été retenu. Les concentrations puis les doses ont été calculées pour les principaux radionucléides (à l'exception du tritium pour lequel aucun traitement

n'est prévu) et les voies d'exposition généralement retenues dans ce genre d'analyse. En ce qui concerne le site il s'agit de données réelles concernant les conditions météorologiques et la répartition géographique des populations et des productions agricoles.

TABLEAU 1. Caractéristiques principales du réacteur et du site.

REACTEUR	. Puissance électrique nominale	1 300 MWe
	. Durée de fonctionnement	8 000 h
	. Taux de rupture de gaine	0,25 %
	. Fuite primaire totale	14 kg/h
	. Fuite primaire-secondaire	4 kg/h
REJETS	. Total gazeux	15 000 Ci
	dont iode	0.9 Ci
	. Total liquide hors tritium	7 Ci
	dont iode	2 Ci
SITE	. Population (R ≤ 100 km)	$2\ 10^6$ hab.
	. Débit du fleuve	1 600 m^3/s

TABLEAU 2. Systèmes de traitement et voies d'effluents associés.

VOIE D'EFFLUENT	Symbole	TRAITEMENT
Drains + Effluents chimiques	DR	Evaporation thermique + stockage (FD = 100 D = 5 j)
Laverie	LV	Stockage (D = 30 j)
Dégazage eau primaire	TEG	Stockage sous pression (D = 60 j)
Fuites bâtiment réacteur	BR	Piégeage des iodes en balayage continu (FD = 10)

L'indicateur de dose retenu est l'engagement d'équivalent de dose effectif collectif dans un rayon de 100 km autour de l'installation. Le passage aux effets est effectué en utilisant le coefficient proposé par la CIPR de $1.65\ 10^{-2}$ effet (cancers mortels aux organes autres que la peau) par Homme-Sievert.

L'indicateur économique qui caractérise chaque système de traitement est le coût total actualisé, c'est à dire la somme actualisée sur 20 ans du coût annuel d'exploitation d'un système et de l'annuité d'amortissement de l'année correspondante. (Le taux d'actualisation retenu est de 9 %, les coûts exprimés en francs 1978). Le coût d'exploitation annuel ne tient pas compte du coût de traitement des déchets. Quant aux coûts d'investissement ils prennent en compte les coûts directs (matériel, montage, étude et transport) et indirects (intérêts intercalaires et frais d'ingenierie). Il est à noter que le système de traitement des eaux primaires et le système de traitement des purges des générateurs de vapeur sont considérés comme des systèmes de production indispensables au bon fonctionnement du réacteur et qu'en conséquence leur coût ne figure pas dans les coûts de protection.

RESULTATS

Le tableau 3 synthétise les résultats. Les systèmes sont classés dans l'ordre des rapports coût-efficacité croissants : l'efficacité étant définie par le nombre d'effets évités par le recours à un système.

TABLEAU 3. Coûts et efficacité des systèmes de traitement des effluents.

SYSTEME	COUTS (10^3 FF) Inv.	Expl.	Tot. actual.	Effets évités(*)	ΔC/ΔE
TEG.	1 290	52	1 440	3.1	4.6 (2)
DR.	4 480	350	6 750	2.3 (-1) **	2.9 (4)
NR.	100	36	440	4.8 (-4)	9.1 (5)
LV.	300	88	1 160	1 (-4)	2.9 (6)

* 1,3 GWe x 20 ans

** 2.3 (-1) = 0,23

Pour l'analyse coût-bénéfice, conformément à la présentation préconisée dans la publication 22 de la CIPR[3], il convient d'exprimer l'efficacité en effets résiduels. En l'absence de traitement l'ensemble des rejets conduisent à 3.5 effets pour 20 ans de fonctionnement du réacteur. Les valeurs de l'Homme-Sievert retenues sont respectivement de 400 FF et 4 000 FF. L'analyse de la Figure 1 montre que pour ces valeurs qui peuvent être considérées comme extrêmes, le niveau de radioprotection atteint se situe au-delà de la valeur qui minimise le coût social de radioprotection.

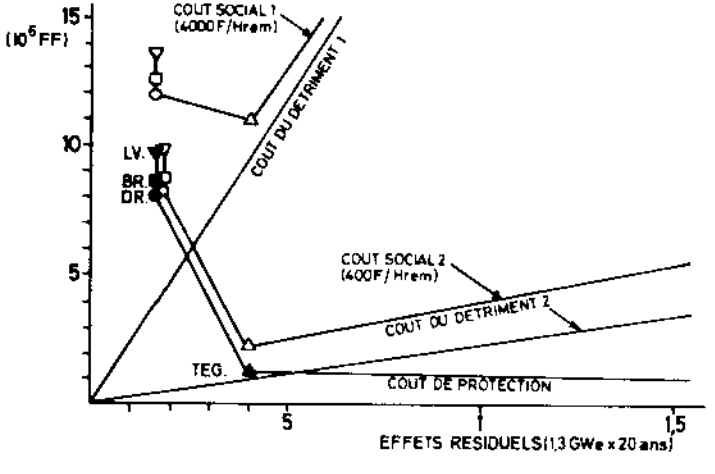

FIGURE 1. Analyse coût-bénéfice.

CONCLUSION

L'analyse a posteriori des choix de radioprotection permet comme on l'a montré, de situer voire de justifier ces choix. Elle permet d'autre part de donner les éléments nécessaires (le coût marginal de l'effet évité en particulier) pour une comparaison avec d'autres technologies présentant des risques en marche normale (énergie conventionnelle, industrie chimique, etc...).

C'est cependant dans une perspective décisionnelle que la méthodologie présentée s'avère la plus conforme aux recommandations de la CIPR : les résultats présentés démontrant que dans le domaine de la gestion des effluents des installations nucléaires, cette méthodologie est tout à fait pratiquable.

Il convient néanmoins de manier ce type d'analyse avec précautions. En particulier la seule dimension du risque prise en compte est ici le risque collectif, il est donc nécessaire de s'assurer par ailleurs que les limites de doses individuelles sont bien respectées. De ce point de vue des méthodes décisionnelles multidimensionnelles pourraient apporter un éclairage plus synthétique sur le problème de la gestion du risque radiologique.

D'autre part les données sur lesquelles reposent ces études sont caractérisées par une grande variabilité qu'il s'agit d'évaluer par le recours à une analyse de sensibilité approfondie, d'où la nécessité d'élaborer des outils adéquats(4).

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OPTIMISATION DE LA PROTECTION ET EVALUATION DU RISQUE
LE CAS DES ACCIDENTS DE TRANSPORT DE MATIERES DANGEREUSES.

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Cette étude se propose de mettre en oeuvre les principes d'optimisation recommandés par la CIPR dans les publications n° 22 et 26, et traite du cas des risques encourus par le public du fait des accidents de transport de matières dangereuses.

L'objectif principal de l'étude est la rationalité des mesures de protection . Plusieurs sous-objectifs doivent être atteints préalablement :

- Analyse du processus accidentel
- Evaluation du risque sanitaire
- Evaluation économique des mesures de protection supplémentaires.

METHODE

Modèle physique d'évaluation du risque -

Ce modèle vise à calculer pour un programme de transport donné, le nombre de colis de matières dangereuses ouverts et le nombre d'effets sanitaires associés pendant une période de temps donnée.

L'évaluation du risque s'effectue en quatre étapes faisant appel à différentes méthodes mathématiques, statistiques ou physiques :

- Définition de l'ensemble des scénarios d'accidents probabilisés.
- Identification et détermination des indicateurs de gravité.
- Evaluation du nombre de colis ouverts étant donné un programme de transport.
- Nombre d'effets sanitaires dûs à un relâchement de matières dangereuses.

Modèle d'aide à la décision

La mise en oeuvre du modèle d'aide à la décision s'effectue suivant l'organigramme représenté sur la figure 1. Le choix d'un certain nombre de mesures visant à améliorer le niveau de protection du public est fait en tenant compte des contraintes de faisabilité, quelles soient de nature institutionnelle ou socio-économique.

L'analyse de cet ensemble de mesures sur la base de leur coût et de leur efficacité vis-à-vis d'un indicateur pertinent (risque résiduel) peut alors être réalisée. Par ailleurs, on peut comparer les mesures qui auront été jugées efficaces avec d'autres mesures affectant le risque dans divers secteurs industriels.

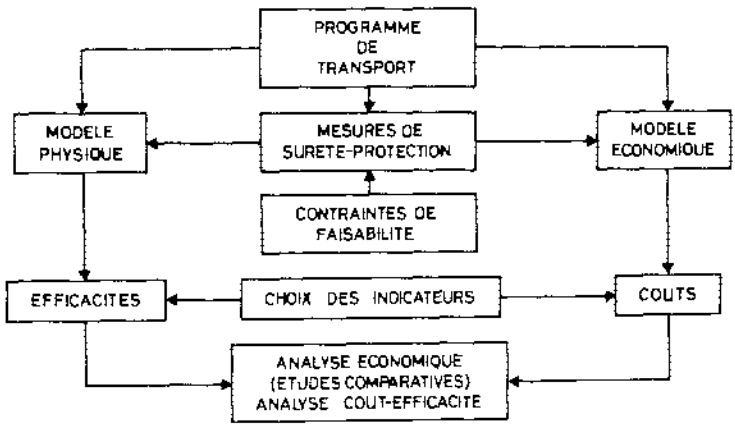

Figure 1. Organigramme fonctionnel

APPLICATION

Le programme de transport.

Le risque associé au transport de gros conteneurs est calculé à titre d'exemple pour un trafic de 100 000 véhicules x km dans le cas où le mode de transport est le chemin de fer. En faisant l'hypothèse d'un système de transport conforme aux pratiques en vigueur actuellement, notamment en ce qui concerne les mesures de protection, on a les résultats suivants quant au niveau de risque. Les effets sanitaires sont estimés à partir d'un relachement d'HF.

TABLEAU 1. Risque associé au programme de transport.

	Probabilité d'accident	Nombre d'ouvertures attendues	Risque résiduel (Nombre d'effets sanitaires)
Chemin de Fer	$.8\ 10^{-2}$	10^{-4}	$2.5\ 10^{-5}$

Les options candidates

Cinq mesures de base sont prises en compte, susceptibles d'affecter le risque de différentes façons.

- M - Morcellement des expéditions (1 citerne par wagon au lieu de 2 ou 3).
- Q - Adoption de citernes renforcées pour l'ensemble des expéditions.
- D - Utilisation de train direct ne passant pas en triage.
- I - Contournement des grandes agglomérations.
- V - Limitation de la vitesse à 40 km/h.

Une option de protection est une combinaison compatible de ces cinq mesures, c'est ainsi qu'une quinzaine d'options sont candidates y compris la mesure "OPO" consistant à ne rien faire.

RESULTATS

Le classement coût-efficacité

Les options candidates sont symbolisées par les lettres identifiant les mesures de base qui les constituent. Le classement coût-efficacité, qui est indépendant du risque associé à l'option "OPO", puisqu'il ne tient compte que du risque résiduel après application de l'option et du surcoût associé est présenté figure 2 et explicité dans le tableau n°2.

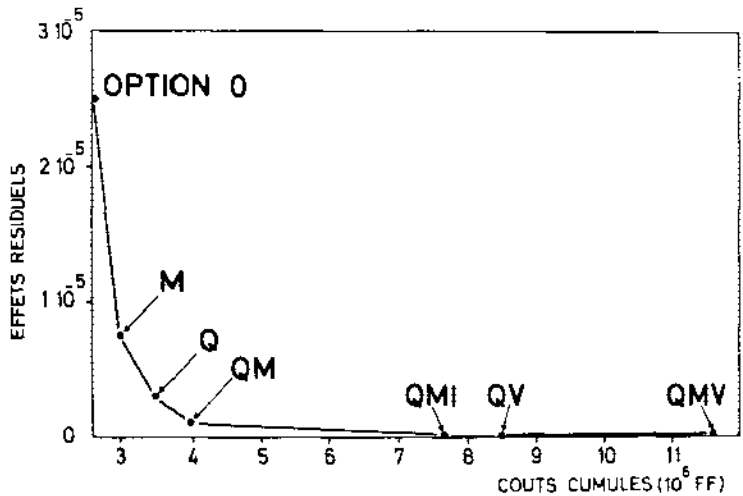

Figure 2. Courbe coût-efficacité

TABLEAU 2. Classement coût-efficacité

Classement	Option	Δ Coût (FF)	Δ Efficacité	ΔC/ΔE
1	M	$4.8\ 10^{4}$	$1.7\ 10^{-5}$	$2.7\ 10^{9}$
2	Q	$4.4\ 10^{4}$	$4.5\ 10^{-6}$	$9.8\ 10^{9}$
3	QM	$4.8\ 10^{4}$	$2.1\ 10^{-6}$	$2.3\ 10^{10}$
4	QMI	$3.7\ 10^{5}$	$7.4\ 10^{-7}$	$5.0\ 10^{11}$
5	QV	$8.3\ 10^{4}$	$1.1\ 10^{-7}$	$7.9\ 10^{11}$

Il apparaît ainsi que la mesure la plus coût-efficace est le morcellement des expéditions.

Comparaison avec d'autres mesures de protection.

On peut comparer les résultats obtenus ici avec ceux obtenus lors d'une étude coût-efficacité de certains systèmes de sûreté (ECCS = Emerging Core Cooling System,...) sur les réacteurs PWR /1/. Le tableau suivant n° 3, montre une relative cohérence entre les rapports coût-efficacité.

TABLEAU 3. Comparaison des rapports coût-efficacité

	ΔE (morts)	ΔC (FF)	ΔC/ΔE
PWR : ECCS	11	$7.5\ 10^{6}$	$6.8\ 10^{5}$
: Recombineur d'H_2	$?\ 10^{-5}$	$2\ 10^{5}$	$1\ 10^{10}$
Transports : Option M	$1.7\ 10^{-5}$	$4.8\ 10^{4}$	$2.7\ 10^{9}$

CONCLUSION

La démarche proposée ici pour appliquer les recommandations de la CIPR semble déboucher sur des résulats concrets pouvant se traduire par une meilleure rationalité des choix de protection.
D'autres approches ayant les mêmes objectifs, tellesque les méthodes multicritères peuvent être envisagées et compléteront la méthodologie proposée.

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APPLICATION OF THE PRINCIPLES OF JUSTIFICATION AND OPTIMISATION TO PRODUCTS CAUSING PUBLIC EXPOSURE

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INTRODUCTION

Radioactive substances have been used in some consumer products for many years; other uses may be developed in the future. The possibility that the public might be unjustifiably exposed to radiation from these products is a reason for the development of systems of control whereby such products are required to be approved by national authorities before they can be supplied to the public. The basis of these controls should be the application of the ICRP principles of justification and optimisation of protection together with consideration of individual doses in comparison with dose limits. In this paper we explore the difficulties and judgements which are involved in applying these principles to radioactive consumer products. These problems should be explicitly recognised in the process of establishing radiological protection criteria for use by national authorities, or indeed, before attempting to harmonise international practices and thereby facilitate international trade.

JUSTIFICATION

ICRP has recommended cost benefit analysis as the 'ideal' mechanism for determining the acceptability of a proposal involving exposure of people to radiation, and have expressed the principles of justification and optimisation in mathematical notation (1). The aim of cost benefit analysis is to identify all the positive and negative aspects of a proposed practice, to quantify them in a common unit, usually money, and thereby determine whether the proposal provides a net benefit to society as a whole. In view of the quantitative nature of the technique, it might be regarded as a sufficient basis for objective decision making. This however is not the case. The major advantage of its application, in theory or practice, lies not in the removal of subjective judgements but rather in requiring their explicit recognition. It is therefore valuable in assisting the development of consistent policy criteria, but of itself does not necessarily provide a simple and unequivocal solution.

The analysis initially draws attention to the distribution of doses and benefits between both those involved in supplying the products and the public. However, frequently the only concern of the national authority is consumer protection as separate regulations cover occupational exposure. In defining the scope of the analysis the national authority is therefore likely to restrict its attention to estimating the net benefit to the public and ignore the occupational aspects.

The measure of net benefit to be attributed to the consumption of a product can, in the theory of cost benefit analysis (2), be derived from what is known as the "consumer surplus". This repre-

sents the sum of the maximum additional amounts of money that consumers would be prepared to pay for the product (ie above that actually paid). This is a measure of the net value to the consumers from their own viewpoints. However in practice, this perceived net benefit from a product is likely to be distorted both by advertising and misinterpretation (or ignorance) of the associated hazards. Indeed, it is partially because of this inability of consumers to correctly assess these radiological hazards that national authorities need to exercise control over such hazards. Thus even where the product buyers are the sole recipients of the benefits and the radiological detriment from a product, this measure of net benefit cannot automatically be assumed to reflect public preferences.

In most cases radioactive consumer products will expose non-users as well as users to radiological hazards, and the protection of these individuals provides a further reason for the control of such goods. However, if money is the common unit to be used in cost benefit analysis then this detriment will also need to be expressed in monetary terms. The difficulties and judgements of applying cost benefit analysis to value radiological detriment in this manner are comparatively well known and are described elsewhere (3). Once this conversion has been made, then for any product a measure of the net benefit to the public as a whole could be obtained, e.g. by subtracting the detriment cost from the consumer surplus. Even so the demonstration of an overall positive net benefit is not a sufficient criterion for the purposes of justification without some reference to the scale of the individual doses, particularly to non-users. Moreover, compliance with the ICRP dose equivalent limits may not be considered sufficient, and some lower maximum acceptable dose level may need to be specified using subjective judgements.

Quite apart from the subjective judgements involved in integrating the results from cost benefit analysis into a decision making framework, the technique requires a substantial amount of data. It requires realistic (as opposed to cautious) assessments of the dose distributions from normal use, misuse and accidents, and uncontrolled disposal, as well as extensive market research to obtain data on the likely demand for the product. The generation of these data, even with products already on the market, would involve considerable costs in terms of time and effort. These costs will normally be out of all proportion to the scale of the radiological problems associated with such products.

OPTIMISATION

Many of the comments made above are, of course, equally relevant for the application of the principle of optimisation to radioactive consumer products. This may be illustrated by reference to a worked example recently carried out to develop a standard for the quality control of gaseous tritium light sources (GTLSs) intended for use in liquid crystal digital (LCD) watches (4). Three manufacturers were involved in the study and each was already undertaking some form of quality control to detect inadequately sealed GTLSs. The changes in costs and benefits resulting from changes to their procedures were analysed. The following is a summary of the findings :

1. The analysis was only 'objective' in the numerical optimisation of parameters and the results were in general different for each manufacturer, depending on the relative cost effectiveness of their quality control procedures. In deriving a standard to be applicable to the whole industry subjective judgements were therefore required. One such judgement was that those parameters which had only a minimal effect on radiation exposure would be ignored.
2. While the primary concern was consumer protection the analysis focused on changes in manufacturing costs as a parameter reflecting changes in price and net benefit to consumers. In one case implementation of the derived optimum quality control parameter would have resulted in reduced costs to the manufacturer but increased detriment to the public. As it is not possible to ensure that the cost savings would be passed onto the public, it was considered undesirable to adopt this optimum parameter. However, in situations where implementation of the derived optimum quality control parameter would have resulted in reduced detriment to the public albeit at higher costs to the manufacturer, it was considered reasonable to adopt the optimum parameter; the manufacturer could then recover the higher costs by increasing the price for his GTLSs.
3. The results were very sensitive to both the monetary value assigned to the unit of collective dose equivalent and the assessed dose per unit of tritium evolved.
4. The effort involved in what initially appeared to be a relatively straightforward analysis was substantial. It was however, considered to be worthwhile in this instance as it highlighted some of the practical problems involved in optimisation and as there was the possibility of very wide scale distribution of LCD watches with GTLSs.

CONCLUSIONS

It may be concluded that the role of quantitative cost benefit analyses in determining the acceptability of consumer products is likely to be limited. A more pragmatic scheme is likely to be sufficient. For justification purposes, one such scheme could involve the restriction of doses to individual users and non-users perhaps in relation to a qualitative description of the benefits. For optimisation purposes, a more practical approach could again be adopted involving, for example, the selection of the least hazardous radionuclide in its intrinsically safest physical and chemical form (5,6). Such a basis for the control of radioactive consumer products would substantially reduce the effort required by the national authority. However if this control is to be applied in a consistent manner, it requires the development of specific criteria which will inevitably incorporate subjective judgements on, for example, acceptable levels of exposure. The explicit recognition of the judgements involved will clarify the development of such criteria both nationally and internationally.

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SETTING STANDARDS FOR TRIVIAL CONCENTRATIONS OF RADIOACTIVITY

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1. INTRODUCTION

Radioactivity is an integral part of the environment. The quantities produced and released to nature by man-made processes is but a small fraction of that allready present. Life has evolved more or less acceptably in an environment of natural radiation the levels and variation of which is known with considerable accuracy and the effects of which has been better studied than any other natural phenomenon. In spite of this, society is reluctant to consider man-made radioactivity in the same manner as natural radioactivity and to accept that the effects from these sources are no different.

No material is totally devoid of radioactivity and it is clearly not the intention of any waste management practice that every material should be treated as a potential radioactive pollutant. In the legal profession there is a saying De Minimus non curat lex, meaning the law does not care for or take notice of very small or trivial matters. A similar philosophy should hold true for the radiological protection profession and the saying can be construed to mean that a quantity of radiation or an amount of radioactivity that is so low that it can not be distinguished from natural radiation should be regarded as trivial. Radiological protection should not be concerned with trifles.

The principles of radiation protection as is incorporated in the recommendations of the ICRP, requires, among others, the balancing of costs and detriments of any practice involving radioactivity. In waste disposal practices the detriment due to radiation exposure is closely related to the total quantity of radioactivity while the cost is a function of the total mass of the waste to be disposed. If follows therefore that there can well be an optimum concentration value below which control is not cost effective.

Radioactive wastes are often classified into three categories i.e. those requiring complete isolation from the environment over extended periods (thousands of years) through methods such as deep geological burial; secondly, isolation over shorter periods through methods such as shallow land burial - the period being that over which institutional control can be exercised and during which time the radioactivity will decay to trivial amounts; and finally wastes containing trivial amounts of radioactivity that would be exempt from special regulation and disposed of as normal waste into sewage systems, onto garbage dumps or into the ocean.

A trivial level of radioactivity is that below which considerations other than those of radioactivity becomes of overriding importance and where the management should be concerned with these properties rather than with the radioactive content of the waste.

A need exists to come to an agreement on a level or concentration below which the radioactive content can be considered trivial. Various approaches to the setting of radiation standards have been discussed, i.e. natural variation in radiation doses (1), relating a trivial dose to an insignificant risk (2).

2. EXISTING STANDARDS FOR TRIVIAL AMOUNTS OF RADIOACTIVITY

An NEA/OECD survey among its member countries of threshold values below which solid materials or effluents will be regarded non-radioactive showed that the value of 0,002 μCi/g are the most widely recognised limit. This value has its origin in the IAEA Basic Safety Standards of 1967 (3), where it is specified that only radioactive substances at concentrations exceeding 0,002 μCi/g or solid natural radioactive substances at concentrations exceeding 0,01 μCi/g shall be notified or registered by the competent authority. In the same publication maximum permissible activity values for exemption from notification for individual nuclides are given; these vary between 0,1 μCi and 1 000 μCi. The basis for these values is not explained.

Other values used are 10^{-5} μCi/g by the Radiochemical Inspectorate in the UK for man-made radioisotopes and is also in the USA by the Environmental Protection Agency (EPA) for transuranic nuclides. In the Netherlands, sources of radiation representing total activities between 0,1 and 100 μCi, depending on the radioactivity of the nuclide, can be disposed of as solid waste, probably in accordance with ref. 4 above. Others use a fraction of the ICRP maximum permissible concentration values in air or water as threshold values, below which the material is regarded as non-radioactive.

In order to be consistent, a basis should be found which can be used for the calculation of threshold values for all radionuclides and for all different disposal practices of radioactive material. Dose is the fundamental criterion for protection of man and therefore the correct basis to use when determining effects which can be considered trivial. An obvious approach is to define a certain trivial dose limit the detriment of which can be considered to be negligible and its contribution indistinguishable from the natural dose variation.

3. BASIS FOR THE CHOICE OF A TRIVIAL DOSE

3.1 Variation in natural radiation dose

The main contributors to the natural radiation dose are cosmic radiation, terrestial sources of which there are two important components, i.e. external gamma radiation and internal radiation from inhaled ^{222}Rn and its daughter products and ^{40}K.

Cosmic radiation dose varies with latitude and altitude. The dose is about 32 mrem/a at sea-level and mid-latitudes. The dose is highest at the poles and decreases by about 10 % at 40° latitude and by 25 % at the equator. The decrease is equivalent to 1 mrem/a for every 10° decrease in the latitude. The increase with altitude is around 2 mrem/a for every 100 m.

The radiation dose contribution from the three most important terrestial sources are 5,5 mrem/a from 1 ppm U_3O_8 in the soil, 1,9 mrem/a from 1 ppm ThO_2 and 13 mrem/a from 1 % of potassium in the soil. The large variations in soil composition give rise to whole-body radiation dose range for 95 % of the population living in "normal" areas of between 26 and 61 mrem/a (4).

Potassium is a major element present in the human body. The radiation dose from the natural radioactive isotope ^{40}K to various body organs varies with age, sex and natural concentration of potassium in the body; typically, gonads 9-21, lung 10-24 and red bone-marrow 16-38 mrem/a.

The variation in the dose to the lungs due to natural ^{222}Rn is even larger and can vary between 4 and 400 mrad/a (4).

These natural variations in the radiation dose to man of tens of mrem per annum is a fact which society accepts and do not consider when choosing a domicile.

Adler et al (1) makes a strong plea for using a standard deviation in natural radiation levels as a basis for setting standards from radiation resulting from human endeavours. This approach is based on the fact the human race has a long history of being exposed to these levels and that no correlation between these variations and any detriment can be shown. The value of the standard deviation in natural background is of the order 20 mrem/a in the USA.

3.2 A comparison of risks

All human activity entails a risk of same kind or another. The annual risk of death from natural causes in the prime of life is about 10^{-3} and never less then 10^{-4} (2). As it is assumed that the risk to health corresponds linearly with the dose received, a trivial dose would be that which corresponds to an insignificant level of risk to the recipient.

Through an analysis of voluntary and involuntary risks Webb and McLean (2) arrived at a value of 10^{-6} as an annual level of risk which is not taken into account by individuals in arriving at decisions as to their actions and which is therefore negligible. This corresponds to the value considered by USEPA as not being of regulatory concern. By applying the linear dose-risk relationship a value of 10 mrad/a is proposed by Webb and McLean as an insignificant or trivial dose to the individual. The risk from this dose has been compared to smoking 1½ cigarettes, drinking ½ liter of wine, travelling 16 km on a bicycle or 80 km in a car or 300 km by air.

4. THE APPLICATION OF A TRIVIAL DOSE

Dose is not a quantity of direct practical value when considering the disposal of radioactivity. It is necessary to convert dose into a quantity which can be measured such as a specific concentration or a rate of disposal.

4.1 Assessment of release rates

Depending on the particular practice i.e. atmospheric, aquatic or terrestrial disposal, a dosimetric model, realistic in terms of the possible pathways to man, should be used to derive trivial quantities of radioactivity. Two important aspects i.e. multiple sources of exposure and the dose commitment due to the continuous nature of the practice must be considered to ensure that the continuation of this practice over many years and from many sources will not result in the tri-

vial dose limit being exceeded in any individual. The allocation of a percentage of the trivial dose to a particular practice can be used to resolve the first concern. Webb and McLean (2) propose that an annual dose commitment of 0,1 mrad be used and considers it very unlikely that any individual will receive a dose in excess of 10 mrad from the various practices that have been exempted in this manner.

An assumption as to the time span over which the practice will continue is needed so as to ensure that the dose commitment from future practices will not add up and exceed the trivial dose as defined. For short-lived nuclides the replacement value which is equal to the decay in one year can be used. For the long-lived nuclides this approach becomes very restrictive and it would be more realistic to assume a finite time during which the practice will continue, for instance the time period for which the production of nuclear energy is expected to continue.

Assessments of this nature depend on the characteristics of the particular environment and requires realistic models, site specific data on transfer parameters through the environment and knowledge of the critical pathways to exposed groups. The result of the assessment specifies a release rate which is a quantity that can not always be controlled and is difficult to administer. In practice it may be necessary to convert the release rate to a concentration on the basis of the expected total mass of waste to be disposed.

4.2 Specific activity

A method which avoids these problems and allows for the calculation of trivial concentrations of radionuclides for general application under all circumstances and at all sites is one based on controlling the specific activity i.e. the ratio of the radionuclide to its stable isotope expressed as pCi/g stable element in the waste material.

The activity of a particular radionuclide that will result in a specified dose, i.e. the trivial dose to a particular organ or tissue, can easily be inferred from ICRP recommendations (5). By expressing this activity as a ratio to the total mass of the corresponding stable element normally present in the organ or tissue, the trivial specific activity for a particular nuclide in the critical organ is found.

The principle underlying this approach is that biological species do not distinguished between the radioactive and stable form of the same element. If therefore the radioactive and stable nuclides are in the same physical-chemical form this ratio will not be increased in its passage from the waste itself through the biological food chain to man. The specific activity will remain constant in the unlikely event of the waste being the only source of the particular element to man or, more likely, decrease as mixing occurs with the vast quantities of the stable element which may be present in similar physical-chemical form. There is thus virtual assurance that the dose to man will be within the trivial dose value.

Despite the apparent attractiveness, there are reservations to which the method is subject. Firstly, the radionuclide and its stable element must be either in the same physical/chemical form, displaying the same biological availability from the outset or must equilibrate before the waste comes into contact with more of the radionuclide and its element. Secondly, the method is limited in its application to situations where internal pathways are critical and to those nuclides which have a stable element analogue. Finally, it cannot be used where the critical organ is the gastro-intestinal tract. With the introduction of the weighted whole body dose concept of the ICRP this latter problem might be overcome; however, the ICRP has now also refrained from specifing body or organ burdens of radioactivity on which, together with the stable element content of body organs, calculation of values of specific activity depends.

5. CONCLUSION

The ICRP system of dose limitation implies that a level of radioactivity exists below which the material should be considered for properties other than its inherent radioactivity. In waste management practice there is a need for a basis on which such a decision can be made. A trivial dose level from which trivial concentrations can be assessed appears to be the logical approach. A value of 0,1 mrad/a for a trivial dose commitment can be substantiated. The application of the specific activity approach may prove valuable for setting trivial concentration levels to wastes resulting from neutron activation processes such as, for example, structural components from reactor cores.

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INTERNATIONAL ELECTROTECHNICAL COMMISSION (IEC) AND RADIATION SAFETY REQUIREMENTS FOR MEDICAL X-EQUIPMENT

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INTRODUCTION

The International Electrotechnical Commission (IEC) came into being in 1904. Its object is to facilitate the coordination and unification of national electrotechnical standards.

In 1966, the Technical Committee No. 62 of the International Electrotechnical Commission was established. Its first meeting was held in May 1968. The scope of the Technical Committee No. 62 of the IEC was defined as follows: "To prepare international recommendations concerning the manufacture, installation and application of electrical equipment used in medical practice. This also concerns surgery, dentistry and other specialities of the healing art".

The major task of IEC TC 62 is to reach an international concensus on the requirements for manufacture, installation, use and maintenance of electrical equipment used in medicine. One on the most important aspects is safety of the patients and users of the equipment. This means protection against electric hazards, radiation hazards, mechanical hazards and hazards that have to do with the particular kind of effects an electrical equipment is intended to produce.

The work of IEC TC 62 started with X-ray equipment because an urgent need for international harmonization in this field was apparent.

IEC TC 62 Sub-committee B deals with X-ray equipment operating up to 400 kV and accessories, and Sub-committee C with high energy equipment and equipment for nuclear medicine.

LIASONS WITH INTERNATIONAL ORGANIZATIONS

The Central Office of the IEC in Geneva, Switzerland, maintains liason with the following international organizations in the field of electrical equipment in medical practice:

ICRP	International Commission on Radiological Protection (particularly with Committee 3 on External Exposure)
ICRU	International Commission on Radiation Units and Measurements
IAEA	International Atomic Energy Agency
IFMBE	International Federation for Medical and Biological Engineering
IFIP	International Federation for Information Processing (particularly with Committee 4, Information Processing in Medicine)
WHO	World Health Organization
SIC	Société Internationale de Cardiologie
FDI	Fédération Dentaire Internationale
OIML	Organisation Internationale de Métrologie Légale
ISO	International Standards Organization, particularly with Committees:
	/TC 42 Photography
	/TC 85 Nuclear Energy
	/TC106 Dentistry
	/TC121 Anaesthetic equipment and medical breathing machines
	/TC150 Implants for Surgery

IEC PUBLICATIONS ON RADIATION PROTECTION

In 1973 IEC Publication 407: Radiation Protection in Medical X-ray equipment 10 kV to 400 kV was published. In 1975 this was supplemented by publication 407A: Radiation protection in dental X-ray equipment.

At the present time a new publication 601, supplement A to part 1 is in preparation to be published. It deals with the radiation safety of medical X-ray equipment and is a part of publication 601 that deals with the general philosophy of the safety of electrical medical equipment.

The purpose of this paper is to present some main features of this new standard.

SCOPE OF THE NEW STANDARD

This document concerns the radiation protection of diagnostic X-ray equipment including computed tomography (CT) and dental equipment. High voltage generators and X-ray therapy equipment are covered by separate standards.

CATEGORIES OF EQUIPMENT

All diagnostic equipments are devided into categories according to the imaging arrangements as follows:

Radioscopy and radiography with spot film device

Radiography
- image receptive areas, FID or both variable
- image receptive area and FID fixed

Radioscopy
- direct
- indirect

Special purpose X-ray equipment

Special purposes X-ray equipment are listed separately:
- mass chest survey
- transportable, radiography
- transportable, for indirect radioscope
- mammography
- therapy simulator
- reconstructive tomography
- diagnostic, above 200 kV
- dental, with intraoral receptor
- dental, panoramic tomography
- dental, panoramic radiography with intraoral X-ray tube
- cephalometric radiography

OBJECT OF THE STANDARD

The object of this standard is to protect both patient and staff from unwanted radiation. The philosophy of ICRP has been followed as closely as possible.

TECHNICAL ASPECTS

In the following some technical aspects of this new standard are discussed.

Filtration

Recommendation of ICRP are respected. However, the user may remove or add special filters.

Alignment of radiation beam

Limitation of the radiation beam to a minimum is an important factor affecting both the image quality and the radiation dose. Therefore, exact requirements concerning the alignment of the X-ray field are given.

Leakage radiation

KERMA in air from leakage radiation: max 0.87 mGy (=100 mR) in one hour is allowed at 1 m distance from the focal spot.

Focal spot to skin distance

This is limited to minimum 0.38 m.

Radioscopy

During radioscopy maximum KERMA rate allowed is 50 mGy (=5.7R) per minute. The apparatus shall be operable only on this condition.

Material between the patient and image receptor

Maximum absorption values are specified as follows:

Cassette holder Film changer Spot film device	max 1.0 mm Al equivalent
Patient support	max 1.5 mm Al equivalent
Cradles	max 2.0 mm Al equivalent

Normal operation location

In this document models are given for calculating the radiation dose received by the operator of the equipment assuming a normal operation location for each type of X-ray equipment.

CONCLUSIONS

Recommendations of the ICRP have been used as a basis for setting up the requirements of the IEC standards according to which compliance of an equipment or device can be tested and verified.

The new standard is of basic importance for the manufacture and installation of X-ray diagnostic equipment. Its aim is to ensure adequate protection of the public as well as personnel using ionizing radiation.

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THE LABORATORY APPRAISAL OF IONISATION CHAMBER SMOKE DETECTORS

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A paper at the Paris conference in 1977 described the approach which the National Radiological Protection Board (NRPB) was to adopt in the United Kingdom with regard to the radiological testing of products that irradiate the public (1). The paper summarised early results of tests on ionisation chamber smoke detectors (ICSDs). Since that time, there has been a steady supply of detectors for evaluation. At the time of writing 47 detectors have been received, most of which utilised americium-241 foil sources. One detector received utilised krypton-85 and some utilised radium-226 foil sources. The test programme which was originally proposed in 1976 (2) has been modified in the light of experience, and the results have formed part of the input to the Nuclear Energy Agency (NEA) in the drafting of recommendations on these devices (3). These were published in 1977 and the NRPB evaluation procedure follows that described in the document. The present paper describes some of the Board's experimental methods and experiences and its interpretation of some of the NEA criteria. The evaluation consists of four sections; visual inspection, dose rate measurements, surface contamination measurements and destructive testing. These will be discussed in turn.

The visual inspection is intended to detect any shortcomings in design, with particular regard to access to the source(s). The recommendations require that under normal conditions of use, direct access to the sources shall be impossible and that the construction of the ionisation chamber for single-station ICSDs shall be sufficiently tamper-proof. While none of the detectors examined so far permit direct access to the sources, several have been considered to be insufficiently tamper-proof. The interpretation of this term within the UK is that the source(s) should only be accessible by means of a special tool or by damaging the detector. For example, the securing of the ionisation chamber by the use of plain screws would be considered unacceptable although the use of special screws would be acceptable. The use of rivets, solder, glue or certain types of plastic clip would be considered acceptable if their removal constituted significant damage to the device. Construction is of particular importance in the domestic situation since there can be no statutory control over use.

The requirement of the recommendations with regard to dose rate is that irrespective of the radionuclide the dose equivalent rate at any accessible point may not exceed 1 μ Sv h^{-1} (0.1 mrem h^{-1}) at 0.1m from the surface of the device. Dose equivalent rates from those detectors containing radium-226 have been measured directly using a calibrated GM counter, and that from the detector utilising krypton-85 was measured using thermoluminescent dosimetry. However, for detectors which utilise americium-241 there is a significant contribution to the dose equivalent from low energy x-rays. In consequence the total dose equivalent rate is dependent upon the materials which shield the source, the methods of construction and the internal dimensions of the device. It is therefore necessary to measure the

photon spectrum using a Si(Li) detector, in order to calculate the total dose equivalent rate. The assessed dose equivalent rates for different types of detector which have the same source activity vary by a factor of three. Most of the detectors examined have dose equivalent rates of less than 10% of the recommended maximum, the two exceptions being the ones with the most active sources. The detectors which contained activities of radium-226 in quantities comparable with those used for americium-241 sources exceeded the limit as did detectors utilising krypton-85. The dose equivalent rate at 10cm from the surface of most americium-containing detectors is below the detection limit. In consequence measurements are routinely made at 0.05m and the results extrapolated to 0.1m. Some of these results are compared in Table 1.

TABLE 1. Dose rates at 0.05m from outer casing

Nuclide	Activity kBq	Dose rate $\mu Sv\ h^{-1}$
Am-241	33.3	6.0×10^{-3}
	37.0	8.5×10^{-3}
	33.3	6.1×10^{-3}
	37.0	1.0×10^{-2}
	37.0	1.3×10^{-2}
	33.3	1.8×10^{-2}
	29.6	2.4×10^{-2}
Ra-226	37.0	1.5+
	1.85	1×10^{-1}
Kr-85*	18500	2×10^{-1}

The standard deviation in the results for Am-241 is 1.2×10^{-3} $\mu Sv\ h^{-1}$

* Dose rate measured at 0.1m from the surface of the device
+ β dose rate 2.0 $\mu Sv\ h^{-1}$ at 0.05m

The preferred method of surface contamination assessment is the wipe test, which permits several components of each detector to be independently checked. The wipes are performed using an alcohol-moistened cotton wool swab. A comprehensive wipe-testing programme results in a large number of samples, and for this reason the activity transferred to each swab is measured by liquid scintillation counting, which allows an automatic throughput of a large number of samples with a detection limit of 0.2 Bq (5 pCi). The results obtained on the inactive areas have almost invariably been below the limits of detection.

Wipe tests also play a major part in the integrity assessment during the final part of the evaluation, the destructive testing programme. Where possible, the source and holder are wiped separately both before and after the test, but where dismantling might

invalidate the subsequent test, only the post-test wipes are carried out. Most of the tests (impact, puncture, drop, pressure, temperature, vibration) have produced few problems with regard to wipe testing, although they were designed to simulate conditions of normal use and credible abuse.

The current test programme differs from the earlier proposals (2) in several aspects. It was found that the sulphur dioxide corrosion test was unrealistically severe. The same conclusion was reached by NEA and the test was not included in the recommendations. However, it was apparent that in tests such as corrosion and fire, there was a likelihood of inactive deposits on the source. A policy of carrying out two consecutive wipe tests was therefore adopted for these tests. In addition, in the case of corrosion tests, activity was frequently found in the liquid beneath the sample, indicating that wipe tests alone were an insufficient criteria of leakage.

The earlier programme contained two fire tests - one at 600°C to simulate a domestic fire and one at 1200°C to simulate a hot industrial fire.

With the 600°C experiment it was possible to define a pass-fail criterion based on leakage and wipe tests and to identify material incompatibility problems. The 600°C test continues to provide the most interesting results. The use of a closed flow system and representative samples continues to give reproducible results. Standard counting methods are used to determine total leakage, and further qualitative information about source behaviour is obtained by alpha spectrometry and autoradiography. Alpha spectra in particular correlate well with the results of wipe testing. The results of 600°C fire tests are summarised in Table 2. The data serves to illustrate the effects of different holder materials, methods of fixing the source and different types of plastic. The range of results quoted refer to the first wipe taken.

TABLE 2. Summary of the results of the 600°C fire test

Holder material	Range of activity transferred to wipes, Bq	Comments
Stainless steel	0.3 - 370	No dispersion
	3589	Plastic housing contains fire retardant
Aluminium	0.6 - 163	No dispersion
Tin plated	148 - 3145	Extensive dispersion in tube and some in filters
Brass	555 - 12950	Extensive dispersion throughout apparatus
Source soldered on brass holder	81	Activity found in the vapour trap

The higher temperature in the 1200°C test was sufficient to melt the source and cause some dispersion of activity within the combustion tube. Several experiments were performed at the higher temperature, whereupon it was decided to curtail the test since no adequate pass-fail criteria could be defined. The NEA included a test at this temperature, defining a criterion in terms of activity which escapes from the combustion tube. This test is being incorporated into the NRPB programme and those samples which were subjected to it proved satisfactory in terms of the NEA criterion. Activity was detected remote from the combustion tube in only one instance, when 28 Bq was found in the vapour trap.

At the time of writing, the Board continues to act in an advisory role with regard to consumer products, although it is likely that this will change in the future. Nevertheless, manufacturers and distributors have willingly submitted samples for evaluation, normally prior to distribution within the UK, were equally willing to make modifications as a result of the Board's findings, and were agreeable to the results being published in a recent Board report (4).

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THE DEVELOPMENT OF THE AMERICAN NATIONAL STANDARD, "CONTROL OF RADIOACTIVE SURFACE CONTAMINATION ON MATERIALS, EQUIPMENT AND FACILITIES TO BE RELEASED FOR UNCONTROLLED USE."

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In January, 1976, the Health Physics Society Standards Committee (HPSSC) submitted to the secretariat of the American National Standards Institute (ANSI) Committee N13 (Radiation Protection) the completed standard, "Control of Radioactive Surface Contamination on Materials, Equipment, and Facilities to be Released for Uncontrolled Use," with the recommendation that it be forwarded to the Board of Standards Review (BSR) of ANSI for final processing and publication.

CHRONOLOGY

This standerd had gone through the many steps required in the development of a standard under ANSI. The first was the appointment of a chairman in September, 1971, with the charge to organize a subcommittee on surface contamination under HPSSC to develop the standard. The subcommittee held its first meeting in July, 1972 at the annual meeting of the Health Physics Society to discuss the approach to the standard. Four months later, the second meeting of the subcommittee was held at offices of the Atomic Energy Commission in Washington, D.C. The subcommittee first met with reguletory personnel of the AEC for an exchange of views and a review of their experience and policies on contamination control. This was followed by a session to prepare an outline for submission to ANSI as the first step in the development of an ANSI standard. The outline attempted to cover all the basic considerations in hazard evaluation and control of surface contamination and as a result was quite ambitious. Drafts prepared by the subcommittee members were reviewed at the third session during the annual meeting of the Health Physics Society in June, 1973, and led to a first draft of a standard one month later. This draft emphasized basic properties and data on surface contaminated equipment rather than specific conditions for release of contaminated equipment. It appeared to be of limited practical use and was rewritten as a performance standard. The approach was based in part on a position paper prepared by the Standards Committee of the Southern California Chapter of the Health Physics Society. The draft was submitted to ANSI N13 and N42 (Nuclear Instruments) for comment in September, 1973. The subcommittee continued to work on the standard during the same period, in particular on limits for surface contamination. It met at the Atomic Industrial Forum office in New York City in December, 1973 to consider the comments. The Atomic Industrial Forum at that time served as the secretariat for the development of standards under N13. Comments were reviewed

and voted upon, and those accepted were incorporated into the standard. All persons who submitted comments were informed of the subcommittee action and reasons. The revised standard was approved unanimously hy the subcommittee, and sent to HPSSC in July, 1974 for transmittal to ANSI. That same month, HPSSC submitted the standard to ANSI N13 for letter ballot action. The standard was sent out by ANSI for letter hallot action in September, 1974. The returns included three negative hallots. Two of these were resolved by changes in the standard. The third, which was cast because the limits were felt to be too low, could not be resolved. All action on the balloting was completed by the end of 1975 and in January, 1976, the standard was sent by HPSSC to the ANSI N13 secretariat (which by that time had been transferred from the Atomic Industrial Forum to the Health Physics Society) with the request to send it to BSR for final processing. The voting action reported was 28 affirmative and 1 negative. Three committee members did not respond despite follow up.

The standard provided criteria for the release for uncontrolled use of materials, equipment and facilities contaminated or potentially contaminated with radioactivity. Permissible contamination limits were specified as well as methods for assessing the levels of contamination. While more precise phrasing was given in the standard, the limits applied essentially to the following categories of radionuclides.

GROUP	TOTAL dpm/100 cm^2	REMOVABLE dpm/100 cm^2
1. Long lived alpha emitters except natural uranium and thorium	100	20
2. More hazardous beta-gamma emitters	1000	200
3. Less hazardous beta-gamma emitters	5000	1000
4. Natural uranium and thorium	5000	1000

The condition for placing a radionuclide in Group 1 was that the nonoccupational maximum permissible concentration in air (MPC_{air}) applicable to continuous exposure of members of the public be 2×10^{-13} Ci/m^3 or less and the nonoccupational MPC_{water} be 2×10^{-7} Ci/m^3 or less. (There had been considerable discussion on whether to use 4×10^{-13} or 1×10^{-13} for the MPC_{air} and 2×10^{-13} was adopted as a compromise. The value chosen determined whether all, or only a portion of the more hazardous alpha emitters would be in Group 1.) The upper limits for Group 2 were 1×10^{-12} Ci/m^3 air and 1×10^{-6} Ci/m^3 water. Acceptable sources for MPC values were those published by ICRP, NCRP, or NRC. The standard specified that the levels could be averaged over 1 m^2 provided that the maximum activity in any area of 100 cm^2 was less than 3 times the limit value. The criteria for the standard put some beta emitters in Group 1 and ^{210}Po in Group 2, but otherwise, the breakdown was as shown above.

As the standard was being processed through the final stages prior to promulgation as an official standard, the chairman of N13 changed his ballot from affirmative to negative after concluding that the alpha limits were too low and not readily measurable with state of the art detectors. It was sent out for reballot to ANSI N13 in September, 1976. During this period, the limits were adopted in

Regulatory Guide 1.86 of the Nuclear Regulatory Commission and the standard was sent out to various laboratories of the Energy Research and Development Administration (ERDA) for implementation on a "trial and use" basis. In April, 1977 ANSI N13 transmitted the standard to BSR with a report of 20 affirmative votes, 3 negative votes, and 4 unreturned ballots. This was followed by a period of public review and comment which ended in August, 1977. Because of the unresolved negative votes, HPSSC requested that it be issued as a Draft Standard for a one year trial and use period. This period began in January, 1979. The limit of 100 dpm/100 cm^2 for the most hazardous group of alpha emitters was changed in the Draft Standard to "non detectable," with an accompanying footnote stating that the instrument utilized for the measurement was to be calibrated to measure 100 pCi of any Group 1 contaminants uniformly spread over 100 cm^2. The total activity limit for Group 2 beta or gamma emitters was also changed to read "non detectable" with an accompanying footnote that the instrument used for the measurement was to be calibrated to measure 1 nCi of any group of beta or gamma contaminant uniformly spread over an area equal to the sensitive area of the detector. The limits for removeable contamination were unchanged. The change in the wording addressed the concerns of those who did not want to have to account for a specific limit which they felt could not be measured accurately.

COMMITTEE DELIBERATIONS

At its first meeting in 1972, the subcommittee made plans to produce a standard that would serve as a source document rather than simply as a control document. As such, it would provide basic information on the nature of contamination, transport through the environment, and resultant doses. The standard would also deal with the determination of the contamination potential from both the history of use and the interpretation of monitoring results; criteria for release, test instrumentation and procedures; and decontamination methods. Other proposed sections were cost-benefit analyses and an annotated bibliography. The result would be an authoritative treatment of exposure risks associated with given types of surface contamination that would provide regulatory agencies with the guidance needed to set numerical limits for contamination levels in specific cases. Additional views on the features of a standard were obtained from representatives of regulatory agencies. Numbers proposed for contamination limits should be related to dose; the relationship could be based on experience factors. Healy's "decision level" approach was recomended for consideration. The need to keep levels as low as reasonably achievable(ALARA) should be incorporated. Did ALARA require some decontamination in all cases where contamination was found, or only at levels which were above the limits? It was possible that the requirements of industry, such as the photographic industry, could be limiting rather than the hazard to people. Should the levels depend on the number of people at risk? The standard should be readily incorporated into normal practice by industry. It should develop as a consensus rather than as a decree. Survey techniques presented should be adequate and clear. But how specific should the standard be? Should the number of measurements, or wipes be specified? Was it really desireable to present numerical limits?

Shouldn't one rather rely on best decontamination practice as shown by experience? The committee was advised to take a fresh look at setting limits without being influenced by the old numbers and to also determine what was as low as practicable.

It was obvious that the initial goals required a much greater effort than the committee could undertake. Also, the highly technical nature of the proposed approach to contamination evaluation would turn every case into a research project. In the end, the committee came to the conclusion that a workable standard had to be a performance standard. It was not possible to present a truly representative contamination level-dose curve. The standard would have to present specific limits and test procedures that could allow for uniform survey techniques. Technical analyses could be presented as a backup but not as a substitute for limits.

The following considerations were the basis for the preparation of the standard:

(1)The relative hazards of surface contamination produced by different radionuclides were given by the MPC's in air and water.

(2)For practical purposes, radionuclides were assigned to a small number of groups with given contamination limits, but this did not preclude the use of a graded scale based on the MPC's in air and water.

(3)A contamination reference level of 1000 dpm/100 cm^2 for ^{90}Sr was set as the basis of assigning limits to radionuclides presenting an ingestion hazard and other radionuclides were grouped based on the values of their MPC_{water} relative to ^{90}Sr.

(4)The contamination limit for ^{239}Pu was chosen as the basis for assigning limits based on MPC_{air} to radionuclides presenting an inhalation hazard. Values of 100 and 200 dpm/100 cm^2 were considered and 100 was adopted in the standard.

(5)An upper limit to surface contamination (i.e. the limit for the least hazardous group) was set on the basis of practicability of achievement and control rather than on MPC. Values of 2000-10000 dpm/100 cm^2 were considered, and a value of 5000 selected for the standard.

(6)Values of maximum scanning speeds for survey instruments were specified to provide the needed detection sensitivity. However, when contamination was detected, survey instruments had to be held stationary when recording readings.

Suggested values for removeable contamination varied between 10 and 20 percent of the total level. A value of 20 percent was adopted in the standard.

The current trial use of the standard should result in useful comments and documentation on the practicability of cleaning equipment to the limits in the standard, and on detecting those limits. It should be noted that the promulgation of the standard does not preclude release at levels above the limits. However, appropriate controls and restrictions would have to be observed until and if the limits in the standard could be satisfied.

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MEASUREMENT TECHNIQUES

A SIMPLE METHOD FOR COLLECTION OF HTO FROM AIR

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About 50 plants process luminous paint in Switzerland. The contamination of the air by tritium constitutes a major problem in these plants. 250 workers in these plants are professionally exposed to tritium and have an average concentration of about 7 µCi/lt tritium in the urinary excretion corresponding to annual doses of 1,2 rem. Furthermore watch factories and stores might have locals where the stock of watches or dials with luminous paint might also lead to an excessive content of tritium in the air. This contamination of the air is due to the fact that luminous paint loses annually 5 to 10 % of the tritium to the environment. The tritium in air is in the form of HTO.

It was therefore desirable, to adoperate a rapid and practical method to check the HTO content in air in order to estimate the tritium intake by workers. Ideally this method should cover the range of three interesting limiting values:

10 µCi m^{-3}, 300 mCi m^{-3}, 30 nCi m^{-3}.

These three values correspond to the upper limit of concentration for working places of professionally exposed persons, for places accessible by single persons of the population and for places with general access, respectively.

So far 3 methods are described in the literature (1), which allow to measure in the range stated above:

- Freezing or condensing the atmospheric moisture
- Bubbling the air through water
- Absorbing the atmospheric moisture

We describe now an alternative method which also covers the interesting measuring range. It is less sensitive than the methods mentioned above, but it is much faster and simpler and therefore it was adopted as our standard method.

MEASURING METHOD

If a liquid scintillation counter is available the rest is very simple. One goes to the place where the measurement of the content of HTO in the air has to be made, opens a soft plastic bottle and squeezes it several times in order to get the original air out and the environmental air into the bottle. Afterwards one pours

quickly some water into the bottle and the bottle is immediatedly closed. Of course the water was prepared beforehand and brought along in a small flask in order to avoid contamination. The HTO and other soluble gases containing tritium will go from the gaseous to the liquid phase in the bottle. At home this liquid will be poured into the vial and mixed with the cocktail for liquid scintillation counting. the determination of the efficiency of the liquid scintillation counter is the only step which requires calibration.

CHOICE OF BOTTLE SIZE AND AMOUNT OF ADDED WATER

A first choice which has to be made is the size of the bottle which serves for the collection of air. To increase sensitivity one would choose the size of the bottle as large as possible. However, there are some limitations:

- The amount of water in the vapour phase and the water remaining on the walls of the bottle should be small compared to the amount of added water.
- One should be able to squeeze out a considerable fraction of the bottle.
- As one bottle for each sample is needed, the size should not be unpractically large.

We decided to use half litre bottles, although for conditions where increased sensitivity is needed, larger bottles could be used. We found that the volume of the bottle is actually 2 % larger than the nominal value. When squeezing the bottle for air sampling, each time about 40 % of the volume is exchanged. After squeezing the bottle 10 times, less than 1 % of the original air remains in the bottle.

The subsequent addition of 10 ml of water serves to absorb the HTO into the water phase. This process can be accelerated by agitating. But there is no problem as the time constant for diffusion over a distance of 10 cm in air is of the order of 10 min. Usually the bottle is left 24 hours before pouring the water into the counting vial. When this is done, some water is lost on the wall of the bottle. For the half litre bottle this loss amounts to 90 µl with a standard deviation of 30 µl. As we were adding 10 ml of water, this loss can be neglected. Also the proportion of HTO remaining in the gaseous phase will be negligible. Even the amount of the added water needs not to be exactly known.

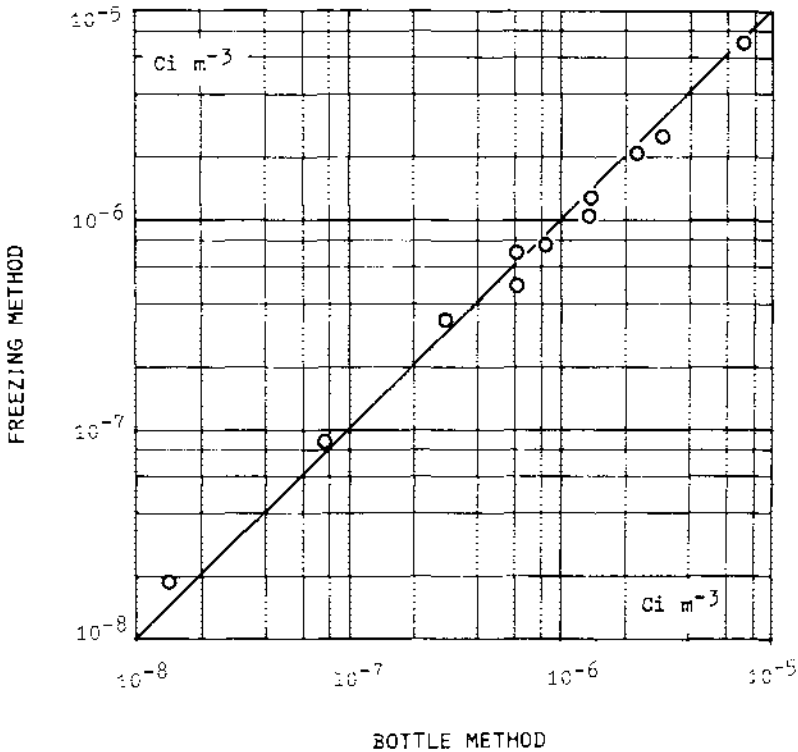

Figure 1

Agreement between bottle method and freezing method.

When mixing 10 ml water with 10 ml of Insta-Gel we find in our laboratory a counting efficiency of 25 %. this efficiency is calibrated and controlled by the measurement of the external standard ratio. The background count is 25 cpm. For 20 min. of counting time the limit of detection is about 3 cpm. This corresponds to

400 Bq m^{-3} or 10 nCi m^{-3}.

RESULTS

An alternative method to measure HTO in air was used to check the bottle method. We collected frost on a can filled with crushed dry ice, scraped the frost off and measured its tritium concentration in the liquid scintillation counter (2).

Simultaneously the humidity had to be measured in order to calculate the concentration of tritium with respect to the air. Fig. 1 shows good agreement between the two methods.

DISCUSSION

The freezing and the bottle method are both absolute methods, the precision being given by the calibration of the liquid scintillation counter. However, the freezing method involves more measurements, of which the most critical is the determination of the humidity of the air. On the other hand, the freezing method is nearly three orders of magnitude more sensitive than the bottle method.

The bottle method proved its reliability and simplicity during one year of use. One needs only two minutes to take an air sample, while with the freezing method it takes 20 min. However, one precaution must be taken: The person sampling the air should not be contaminated with tritium, otherwise the measurements can be much too high. This happens if one first looks around in a laboratory where luminous paint is processed and thereafter measures the tritium outside the plant. The freezing method is less vulnerable in this respect as the sampling takes much more time and one can go away during this time. The freezing method may be the method of choice for measuring tritium in the environment. The bottle method is adequate for measurements in laboratories or for a quick check in the environment in order to decide whether the concentration is above or below the legal limit.

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A PROCEDURE FOR ROUTINE RADIATION PROTECTION CHECKING OF MAMMOGRAPHY EQUIPMENT

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In Sweden, screening for mammography is only permitted in demonstration projects in a few parts of the country including Gävle. After evaluation around 1983 it is possible that screening might be widely used at the 60 installations where mammography equipment is available. Should this occur, standardized checking of the equipment will be an essential radiation protection task.

As radiation protection physicists can be consulted fairly simply at all equipment sites, we wanted to design a checking system that could easily be handled by them on-the-spot. The system should include adequate instruction about criteria for remedial action.

Techniques in Sweden are fairly standardized and the main difference between screening and clinically indicated examinations is in the number of projections, where screening normally employs only the oblique one. Almost all mammography units have molybdenum anodes, automatic exposure control and 0.03 mm Mo filter. Only low-dose film-screen systems are used. Potential differences range from 24 kV to 37 kV with a tendency towards the lower end, and typical tube charges range from 5 mAs to 200 mAs.

RADIATION PHYSICS PROPERTIES OF BREASTS

The distribution of breast thickness in screening situations measured in three locations with the following result:

	Number of measurements	Breast thickness, mm min	max	mean
Location 1	111	15	75	41
Location 2	413	15	80	48
Location 3	486	15	95	50

In all locations the relative standard deviation of the breast thickness was about 27%. The differences in mean thickness are thus significant at the 99% confidence level.

For the estimation of energy imparted to the total breast, the breast area was derived from the exposed x-ray films for 200 films at one location. To a good approximation, the area was proportional to the breast thickness, being 150 cm^2 at 50 mm breast thickness. The relative standard deviation of the area for a given thickness was about 30%.

We have also looked at the attenuation properties of breasts compared with those of water, polymethylmetacrylate and 85% solution of ethyl

alcohol in water. The following results were obtained for the average ratio of the tube charges required to trip the automatic exposure control for breasts of a given thickness and for polymethylmetacrylate of the same thickness:

Breast thickness, mm	30	50	70
Tube charge ratio breast/polym.			
Location 1	1.35	0.9	0.4
Location 2	1.0	0.8	0.6
Location 3	1.3	1.1	0.9
Mean	1.17 ± 0.15	0.91 ± 0.15	0.60 ± 0.3

The uncertainly interval assigned to the mean covers the full range of variation between the three locations. This variation may have been due to different screen-film systems having different energy dependence, and to the employed tube potential differences. An error of 10-20% may have been caused since the cassette used at the phantom measurements did not represent exactly the mean of all cassettes used in the breast exposures.

Smaller breasts approached water in linear attenuation properties, larger breasts approached alcohol/water solution. Polymethylmetacrylate seemed the best approximation when the entire range of breast thicknesses was considered, but obviously it can overestimate the breast dose by more than a factor of 2 at large breast thicknesses.

RECOMMENDATIONS

The recommended checking method includes the use of the Kodak mammography image quality phantom (3) placed above 40 mm of polymethylmetacrylate. Reference pictures will be included in the set of recommendations, and it is recommended that remedial action be taken if the actual image quality subjectively appears to be significantly poorer than that of the reference film. This qualitative recommendation has been thought to be sufficient since the high professional competence in diagnostic radiology in Sweden makes poor image quality rather unlikely.

Radiation dose is to be based on phantom measurements using 50 mm polymethylmetacrylate. Detailed instructions are given for the measurements using either thermoluminescent LiF dosimeters (4 dosimeters per measurement), ionization chambers or a specially designed plastic scintillator (1) with diameter 25 mm and height 50 mm. The exposure at the phantom surface should be multiplied by the following conversion factors to give mean absorbed dose in the breast:

Potential difference, kV	25	28	31	34
Conversion factor, mGy/R	1.19	1.28	1.34	1.39

According to a preliminary recommendation, a mean breast dose of 2 mGy indicates a need for remedial action. Suggestions will be given for countermeasures.

DISCUSSION AND CONCLUSIONS

Since the range of potential differences is rather small, the TLD, ionization chamber and plastic scintillator methods are about equally accurate and expected to yield the same alleged mean absorbed dose to the breast within about ± 10%. A larger range of potential differences might be handled most accurately using the plastic scintillator. The three month reproducibility (double standard error with a given exposure) will according to long term tests be about ± 3% with the ionization chamber and plastic scintillator (mean of two measurements) and considerably higher with TLD (mean of 4 dosimeters).

The day-to-day variations of the x-ray equipment for a given exposure setting may give doses outside a range of ± 10% from the mean. In practice it is difficult to make the dose with the cassette used at phantom measurements represent the average dose for all cassettes within less than ± 10%.

The country-wide consistency may thus be rather poor for the reasons just discussed. In national dose intercomparisons a standards laboratory could provide an assessment of "true" dose using the methods recommended. Several centers would then be asked to irradiate the phantom as if it were a 50 mm breast and state the mean breast dose as assessed by them using a recommended method. It is believed that this dose might be up to ± 30% off from the "true" dose. This variability would mainly be due to errors associated with the automatic exposure, and similar variability would be inherent in the normal breast exposures.

In addition, there are the problems of representativity. Results obtained at any installation using the phantom measurements are expected to represent the mean for a screened population within -30% and +50% for the mean breast dose, and within -40% and +60% for the mean energy imparted to the breast. These errors are mainly due to the previously discussed variations in radiation physics properties of breasts. They have little relevance for the levels selected for remedial action but should be kept in mind when it comes to assessment of population risk. The thickness 50 mm was selected as the best compromize with representativity criteria for mean energy imparted and mean breast dose. A thickness of 40 mm would have been a much poorer compromize and 60 mm a somewhat poorer. If representativity is sought only for the mean energy imparted to the breast as compared with the mean breast dose, a thicker phantom results.

Measurements by one team employing the recommendations at about half of the 60 Swedish mammography installations (2) gave a mean breast dose of 1.6 mGy, a minimum of 0.5 mGy and a maximum of 4 mGy. About one fifth of all installations had doses above 2 mGy, mostly because of different film-screen systems, sub-optimal developing procedures or unusally high optical density of the films.

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HEALTH PHYSICS ASPECTS OF THE IN-VIVO ANALYSIS OF HUMAN DENTAL ENAMEL BY PROTON ACTIVATION.

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INTRODUCTION.

The technique of fluorine analysis by low energy nuclear reaction has often been described (1, 2) and the application to fluorine concentration determination is now routinely used in various laboratories. Usually the sample is placed in a chamber and maintained under vacuum. This technique was adapted to non-vacuum analysis (3) allowing the handling of the samples at atmospheric pressure, and thus permitting in-vivo analysis of human teeth.

As described in a previous paper (4), the protons, having an initial energy of 3 MeV, emerge in air through a tantalum window in which they lose part of their energy; the distance in air between the foil and the impact point on the surface of the tooth being 0.8 cm. The residual energy of the proton beam is 2.725 MeV at the surface of the enamel and the beam diameter is 2 mm.

When the beam crosses the tantalum foil, X-rays and γ-rays from Ta(p,p'γ) reaction are produced. Other nuclear reactions take place when the protons interact with fluorine, phosphorous, sodium etc... contained in the tooth enamel. A typical spectrum is shown in Fig.1.

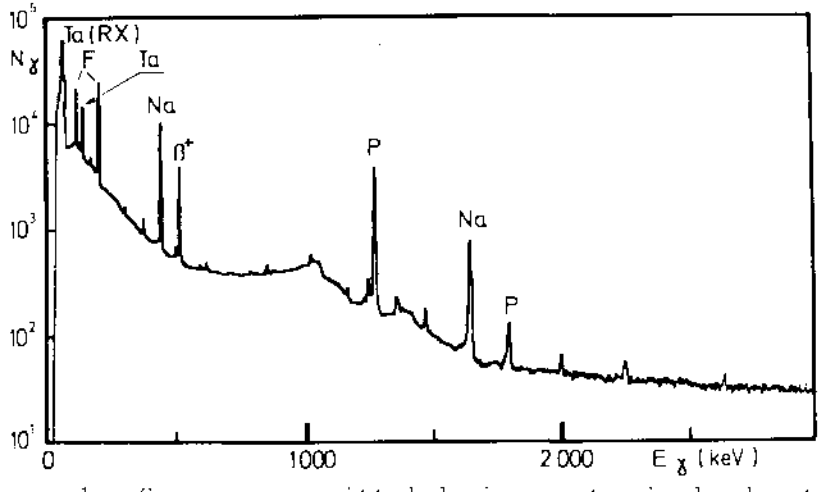

Figure 1 : Gamma-rays emitted during proton bombardment of human teeth and recorded with a Ge(Li) detector.

These γ-rays are used for the determination of the absolute concentrations of these elements in the surface of the tooth. Each measurement takes 1 min, with a beam intensity of 20 nA, which corresponds to a power dissipation of 0.055 W. No heat can be felt by the patient during the bombardment.

The technique of fluorine measurement by prompt activation allows repeated determinations on the same area of tooth enamel both before and after topical application of fluoridated compounds.

EXPERIMENTAL PROCEDURE.

In tooth enamel, ionization, produced by charged particles losing their energy, is very intense over a short range, 30-40 μm. This process can produce local destruction of the enamel and emission of soft X-rays that are completely screened by the tooth and the surrounding frame; hence they are not observable. Neutrons are not observed because the proton energy is below the neutron threshold for the majority of nuclei present in the teeth. Any dose rate is thus due mainly to hard X-rays and γ-rays induced in atomic and nuclear reactions. Due to the short irradiation time, as mentioned above, the dose delivered to the patient is very small : a few μrad. Preliminary measurements have been taken during a 60 min.run made on an extracted tooth to obtain the order of magnitude of the dose around the irradiated tooth. Neutron dose rates were measured with a neutron REM counter (Nuclear Enterprise type NM1), and the dose rate was less than the minimum observable, i.e. 0.1 mrem/hr. To estimate the absorbed dose of gamma-rays, LiF chips were placed around the mouthpiece surrounding the tooth, and an ionization chamber of 500 cm^3 with a tissue equivalent wall of 300 $mg.cm^{-3}$ (Babyline type Nardeux) was used 5 cm behind the beam spot. The dose rate observed with the ionization chamber was less than 1 mrad/hr, but too small to be measured with the LiF chips, indicating that the neutron and γ-ray dose delivered in the mouth area is very small.

To estimate more exactly this very small radiation rate we decided to use $CaSO_4$:Dy chips in longer irradiations on an extracted tooth and to determine the doses to the skin and inside the head, a phantom skull was obtained. This phantom contains a human skull embedded in a tissue-equivalent medium contoured to human features.
This phantom is cut into 2.5 cm slices (Fig.3), and each slice contains drilled holes, arranged in a matrix, which are capable of holding the TLD dosimeters.

Figure 2 shows the energy dependance of the $CaSO_4$:Dy teflon disks (0.4 mm thick), including a correction to take into account the energy spread of the gamma-ray spectrum.

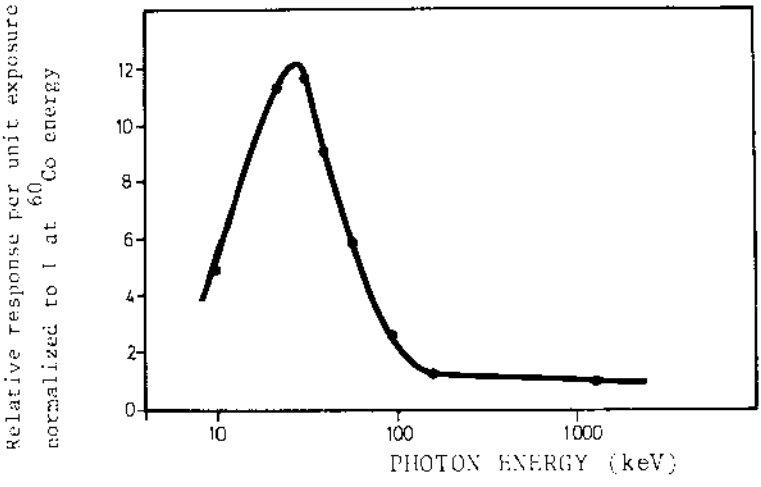

Figure 2 : Energy dependance of the $CaSO_4$:Dy teflon disks

Table 1 summarizes the percentage of the total photon yield for various energy intervals.

TABLE 1 : Percentage of photons emitted during the fluorine determination versus energy.

Energy (keV)	Mains origins of the photons	Percentage normalized to 100 for the entire spectrum.
10-100	Ta X-rays and Compton γ-rays	97.2 %
100-200	γ-rays from Na and F	1.6 %
200-1000	γ-rays from Na and β^+ annihilation	0.5 %
1000-2000	γ-rays from Na and P	0.4 %
2000-8000	high energy γ-rays from F	0.3 %

The calibrated TLD dosimeters were placed in the matrix at appropriate levels of the head which was then placed in the same position as the patient for an 8 hr irradiation under identical experimental conditions.
For each slice, a map of the exposure data was obtained. This data is presented in Figure 3, where the doses (in mrad) correspond to those which would be received after a 1 min. irradiation time.

SUMMARY.

The health physics aspects of the in-vivo analysis of human dental enamel by fast proton activation have been studied. Doses to the head including surface skin and in-

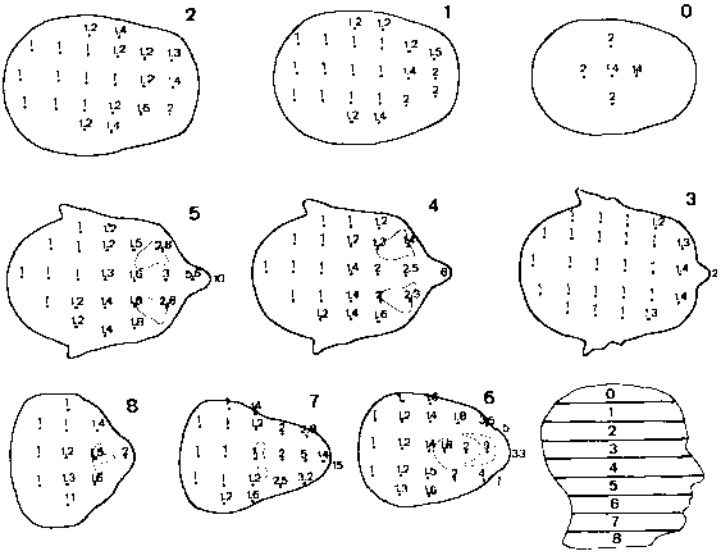

Figure 3 : Doses, in micro-rad, received by the patient during the in-vivo analysis (corresponding to 1 min. irradiation time).

terior regions of the brain and mouth were determined using TLD materials in a phantom head. The maximum dose to the head occurs on the right part of the lips and is of the order of 50µrad. In the mouth, behind the tooth analysed, the average dose is 10µrad, and the average over the whole head is of the order of 1µrad; this is twice the dose received by personnel who spend the same period of time in the counting room a considerable distance from the beam line.

As shown in a previous paper (4) no residual destruction has been found in the first 30µm of the enamel, which is the total range of the 2.7 MeV protons penetrating hydroxyapatite. Indeed, even more than 2 years after in-vivo analysis, there is no apparent destruction of tooth enamel.

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A RELATIVELY FAST ASSAY OF Sr-90 BY MEASURING THE CHERENKOV EFFECT FROM THE INGROWING Y-90

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In the year 1978 alone, over 30 publications were registered dealing with the biological effects and radiation hazards due to the intake and adsorption of Sr-90 (1). Some of these publications include experimental results obtained from radioanalyses of Sr-90, which by themselves are mainly based on measurements of the short-lived, separated daughter nuclide Y-90. A delay of 10 to 15 days is often necessary prior to the radioassay, in order to insure a large enough ingrowth of Y-90 ($T_{\frac{1}{2}}$ = 64.1 h). This time interval can be considerably reduced if the assay is carried out by Cherenkov counting of the ingrowing yttrium. Cherenkov radiation is produced in aqueous solutions by high energy β- particles and has been successfully used for the determination of P-32 (2), Na-24 (3), K-40 (4) and Rb-86 (5). This work is an improvement of a procedure on the assay of Sr-89 and Sr-90 in presence of each other, in which the equilibrated Y-90 is separated from the parent before counting; this part of the procedure is superfluous if only Sr-90 is present (6).

METHOD

The Cherenkov effect induced by the 2.27 MeV β-emitter Y-90 can be measured with good efficiency using conventional liquid scintillation counters (6). The weak contribution of the soft β-s from the parent Sr-90 is eliminated by suitable adjustments of the gain setting and the base-line window opening in the counting channel. Any detectable activity in a freshly separated Sr-90 sample is thus due solely to the ingrowing Y-90. The activity of the strontium is calculated by use of the following expression :

$$\mu Ci(t_o) = \frac{cpm\ Y\text{-}90\ (t_b) \times \exp \lambda_{Sr}(t_a - t_o)}{y_{Sr} \times \varepsilon_Y \times 2.22 \times 10^6 \{1 - \exp[-\lambda_Y(t_b - t_a)]\}} \qquad [1]$$

where : t_o = the reference time of sampling;
t_a = the final Sr/Y separation time;
t_b = the Y-90 counting time;
λ_{Sr} and λ_Y = the decay constants of the two radionuclides ($\sim$0.0243 y^{-1} and 0.0108 h^{-1} respectively);
y_{Sr} = the radiochemical yield of strontium;
ε_Y = the counting efficiency of Y-90.

THEORETICAL ASPECTS (summarized from 7,8)

Cherenkov radiation is produced when charged particles (in our case β-s) pass through a transparent medium at a velocity greater

than that of light in the same medium. This radiation is emitted at an angle θ with respect to the direction of the inducing particle according to (7) $\cos\theta = 1/\beta n$, where β is the velocity ratio v/c and n the refractive index of the medium. For relativistic electrons, β is related to the energy E (keV) of the particle, by (7)

$$\beta = \left\{1 - \left[\frac{1}{(E/511) + 1}\right]^2\right\}^{\frac{1}{2}}$$

. The threshold condition for Cherenkov radiation is $\beta n = 1$, so that if n for water equals 1.332 then β must be > 0.751. This lower energy threshold is 263 keV, below which no Cherenkov effect occurs. However only β-emitters above 1 MeV give sufficiently high yields to be of any practical significance (8).

Cherenkov radiation is one-directional , weak in intensity and has a continuous spectrum mainly in the ultraviolet, with only a small portion extending into the visible region (7).

RESULTS AND DISCUSSION

All the measurements were conducted with 20 ml aqueous solutions in standard polyethylene counting vials. Two optional liquid scintillation spectrometers were used, the Packard Tricarb Model 3390 and the older Model 3214; the working conditions for both instruments are summarized in table 1.

TABLE 1. Working conditions for the Cherenkov counting of Y-90

	Tricarb 3390	Tricarb 3214
Gain %	40	30
Window opening	100 - 1000	50 - 1000
Counting efficiency	$\varepsilon = 0.43$	$\varepsilon = 0.30$
Background	9 - 11	11 - 13

The data in table 2. are a measure of the feasability of assaying a sample before the Sr/Y equilibrium is reestablished. It gives the magnitude of the experimental error that can be anticipated if the Y-90 counting is carried out at various time intervals after the final separation and purification of strontium. Activities as low as 10^{-3} µCi per vial can be measured with reasonable accuracy after only 2 days following the separation. Lower activities require longer periods of time for the Y-90 ingrowth, but even for 10^{-5} µCi 5 days seem to be sufficient.

The radiochemical yield was determined with the help of the γ-emitting Sr-85. Additions of up to 2000 dpm were not detectable at our working conditions. With activities higher than .01 µCi larger aliquots of the marker Sr-85 are desirable (6).

The lower limit of detection is about 3×10^{-6} µCi at conditions of secular equilibrium (= 19 d), which in practice amounts to 2 or 3 cpm above background. An assay of such very low activities requires long counting times (about 500 counts per sample), if the measurements are to be statistically significant. In

addition, too early counting of such samples would introduce an error due to an increase in the activity of the ingrowing Y-90, during the measurement itself. Very low activities should therefore be assayed only after the Sr/Y equilibrium has been reestablished.

TABLE 2. Estimated overall errors for Sr-90 activities, measured at various time intervals after the separation from Y-90
(counting times per sample : 100 minutes or less)

Y-90 ingrowth time ($t_b - t_a$) (days)	± 95% confidence limit, in % of activity : .01 - .15 (μCi)	.003 - .009 (μCi)	8×10^{-5} - .001 (μCi)	10^{-5} (μCi)
< 1	30.	-	-	-
1 - 2	4.4	8.1	-	-
2 - 3	4.5	5.9	34.	-
3 - 5	5.6	5.7	23.	-
5 - 7	6.0	7.7	9.0	15.

RECOMMENDED PROCEDURE (short summary)

1. About 5 to 10 mg of inactive carrier and a known activity of Sr-85 tracer (10^{-3} - 10^{-5} dpm) are added to the original sample. The strontium is separated and purified by a standard radiochemical procedure.

2. The aqueous sample is transfered into a polyethylene counting vial and made up to the desired volume (20 ml). It should be ready for measurement 24 hours after the final Sr/Y separation step. The gain settings and window openings on the liquid scintillation spectrometer should be arranged in such a way as to eliminate the contribution of the parent nuclide Sr-90; only the ingrowing Y-90 is counted.

3. If necessary, the measurements are repeated on consecutive days until the desired precision is achieved. The radiochemical yield is determined by comparison with a blank Sr-85 sample.

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DECONTAMINATION OF TRITIATED WATER SAMPLES PRIOR TO TRITIUM ASSAY

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The liquid scintillation assay of tritiated water samples in radiation protection control is often inaccurate and unreliable due to the presence of other radionuclides. A simple and efficient separation procedure based on distillation and recovery of the tritiated water was published not long ago (1). In the work presented here various aspects of the above method were investigated, with the view of improving the purification and decontamination of the sample from interfering nuclides. The technique is especially useful in the tritium assay of liquid waste effluents from hospitals, biological and veterinary institutes, as well as in the production facilities for labelled compounds.

METHOD

Measured aliquots of tritiated water samples, containing other additional radionuclides, are introduced into one of the distillation apparatuses shown in the illustrations. The larger apparatus (fig.1) consists of a test tube for 25 ml samples and is provided with a small aluminum condenser. The sample is heated to boiling in a glycerol bath and the distillate recovered into a 20 ml plastic vial. About 300 mg of a hold-back carrier are added prior to the distillation; it consists of a dry mixture of $AgNO_3$ (40%), NaI (20%), CuS (10%), anhydrous Na_2CO_3 (10%) and anhydrous $Sr(NO_3)_2$. CuS cannot be exchanged with sodium thiosulfate, which forms a soluble complex with AgI, but $NaHSO_3$ can be used. Salts containing crystallization water must be absent.

A device for smaller samples is useful when only limited amounts of tritiated water are available, or if volatile compounds of radioiodine are present (fig. 2). The glass-tripod with a 4 ml cup on top is placed into a 100 ml beaker and covered with a watch-glass (2); the water sample at the bottom of the beaker is slowly evaporated on a sand bath at 70°C and in presence of the hold-back carrier, until the cup is filled with a sufficient amount of distillate.

In both apparatuses most of the interfering nuclides are completely removed by the hold-back carrier, except for some traces of radiocesium and radioruthenium, which sometimes appear in the distillate. They are removed by shaking the sample with (< 100 mg of) $K_2Co[Fe(CN)_6]$ before or after the distillation, and discarding the solid residue (3).

The sample should be distilled to dryness whenever time and the presence of foreign nuclides permit it. The separation lasts up to one hour with the larger apparatus and about 4 hours with the smaller one. Twelve samples can be easily separated in one run. The

purified sample is now mixed with the liquid scintillator and assayed.

RESULTS AND DISCUSSION

The method was tried out on aqueous solutions of sixteen of the most common nuclides (in ordinary water, no D-3 !), in order to determine their "decontamination factors". The following tracers (mostly as chlorides or nitrates) were used: Am-241, C-14 (as carbonate or uridine), Cl-36, Co-60, Cs-134, Fe-55, mold-198, Hg-203, I-125, Mn-54, Na-22, Ru-103, Sr-85, Te-99, Th-230 and Zn-65; their activities ranged from 10^{-5} to 1.1 uCi. They were subjected to the distillation procedure either individually or in prepared mixtures. As expected, most of the radionuclides were absent from the distillate except for some traces of radioiodine, Hg-203 and Ru-103; decontamination factors of these nuclides are presented in table 1.

TABLE 1. Decontamination factors for I-125, Hg-203 and Ru-103; R = cpm(distillate)/cpm(sample)

Nuclide	sample (ml)	distillate (ml)	hold-back carrier	R
I-125				
0.3 uCi	10	5	Hamiltons (1)	4×10^{-5}
"	10	4	" without CuS, but with $Na_2S_2O_3$	10^{-3}
"	10	4	Hamiltons, no CuS, but with $NaHSO_3$	7×10^{-5}
I-125				
0.05 uCi	25	first 8	Hamiltons	3×10^{-4}
"	25	second 8	"	4×10^{-4}
"	25	last 9	"	0.034
Hg-203				
0.02 uCi	18	first 15	"	2×10^{-5}
"	18	last 3	"	10^{-4}
Ru-103				
0.02 uCi	20	first 7	"	0.027
"	20	second 7	"	7×10^{-3}
"	20	last 6	"	0.026

All the activities were measured by liquid scintillation counting using the Instagel scintillator. A Packard Tricarb Model 3390 spectrometer was used, at optimum gain settings and window openings.

OPTIMAL FIGURES OF MERIT FOR TRITIUM COUNTING

The "figure of merit" (FM) in tritium assay is obtained by multiplying the sample volume with the counting efficiency ε. The highest FM with polyethylene vials (20 ml) were with 7 ml of sample and 9 ml of Instagel (ε = .18); with the small (7 ml) vials FM was highest for a 1:1 ratio in a 5 ml total volume (ε = .16).

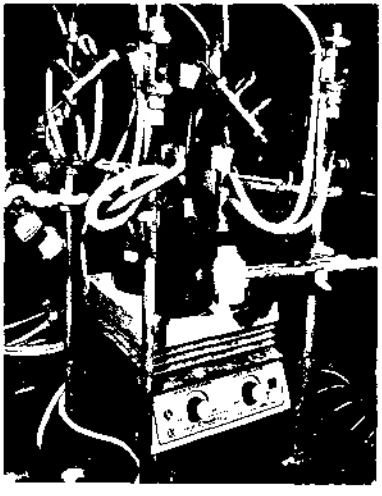

Figure 1.

Figure 2.

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EVALUATION OF THE SPECTRAL DISTRIBUTION OF X-RAY BEAMS FROM MEASUREMENTS ON THE SCATTERED RADIATION

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INTRODUCTION

The knowledge of photon or energy flux density distribution with respect to quantum energy is of certain importance when X-ray beams are produced by tubes supplied with voltages below 100 kV. The reason for this is threefold: 1) the interaction parameters show a high energy gradient below 100 keV, 2) secondary electrons produced by photons fall in an energy range which seems very effective in radiobiological damage (1), 3) beams of such kind are by far the largest source of irradiation for man (radiodiagnostic uses). In these cases, however, spectral distribution measurements present difficulties arising from the very high flux densities attained during the current pulse. Therefore, a number of contrivances have been proposed in order to make the nuclear spectrometers usable. The most important of these are: a) a very narrow beam collimation, b) a large distance between the focal spot and the detector, c) a very low current through X-ray tube.

PRINCIPLES

None of the above mentioned methods is free of criticism. For this reason a different approach was regarded as worthy of more detailed study. A very partial experiment on this spectrometry method is described by Greening (2). The principle of the method is based on the advantage which can be gained from the physical attenuation which the beam undergoes in its partial scattering over a thin sheet in order to solve the difficulty raised by high density fluxes. Therefore, a calculation procedure is required, suitable for the evaluation of the radiation incident on the scatterer from the instrumentally measured distribution. On the grounds of both theoretical and experimental considerations it may be inferred that as far as such kind of beam spectral analysis is concerned the most appropriate direction for measuring the scattered radiation involves crossing the beam axis at about $\pi/2$.

The evaluation of the true spectrum incident on the scatterer needs a procedure complicated on the one hand for the coexistence

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of coherent and incoherent scattering and on the other hand for the presence of distortions caused by collateral physical effects. The exact solution with respect to the first problem can be obtained by measuring spectra of the same beam as scattered by two thin metallic sheets made of pure elements of different atomic number. An alternative procedure which appears equally satisfactory and much simpler both in theory and in experience, requires the measurement of the spectrum scattered by only one scatterer provided it is made of a pure element of a very low atomic number for which the interference between coherent and incoherent scattering is pratically avoided. The collateral effects are due partly to the response function of the detector and partly to distortions in the beam energy distribution caused by a number of unavoidable attenuators. There are no less than ten corrections to be considered.

Even if all of these effects are taken into account in the calculations, which is possible, the above mentioned spectrometry technique has a cost which is low and acceptable only for quantum energy below about 100 keV. In fact, apart from the resolution loss connected to the finite aperture of the beam which can be rendered negligible by a suitable choice of geometrical arrangement, that resolution loss coming from the broadening of the energy interval of the reconstructed discrete spectrum in comparison with the direct one should be pointed out. Indeed it is possible to show that the reconstruction necessarily involves the incoherent scattering which changes the energy interval breadth.

RESULTS

Two distributions are used in order to check the theory: a) the first one is the highly filtered spectrum produced by a Be window tube supplied with a high voltage of 60 kVp, an average current of 4 mA and filtered by an additional 0.5 mm Cu, b) the second one is the spectrum emitted by the same tube supplied with a high voltage of 30 kVp, an average current of 3 mA and filtered by the Be window 0.5 mm thick. While the first irradiation technique allows the comparison of the reconstructed spectrum with the direct one, the second technique requires a theoretical evaluation of the spectral distribution for comparison.

Being practically indistinguishable the results obtained from the measurement done with only one Be scatterer or from the two in which a Be and an Al scatterer are used respectively, no graphs comparing such results are given. In figure 1 the spectral distributions of the 60 kVp beam are drawn. The continuous line represents the spectrum measured on the direct beam at a suitably large focus to detector distance. The dotted line is the reconstructed spectrum. Besides the above mentioned corrections, in this example special atten-

tion must be paid to the difference in the dead time resulting from the two measurement conditions. The two spectra are normalized to the same area. Some differences are present and their amount is significant. This preliminary result, however, is still under investigation and a better understanding of such small discrepancies should be achieved very soon. In order to complete the evaluation of this X-ray spectrometry method a check is required at the lowest limit of the

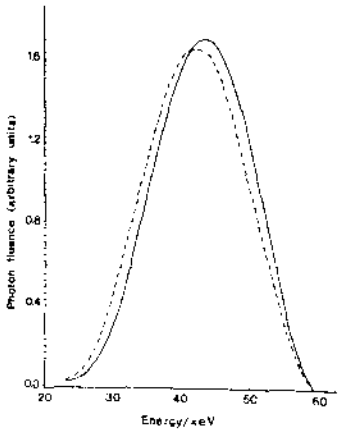

Fig.1: Distributions, twice smoothed, obtained at 60 kVp nominal high voltage: ——— primary spectrum, ----- reconstructed spectrum.

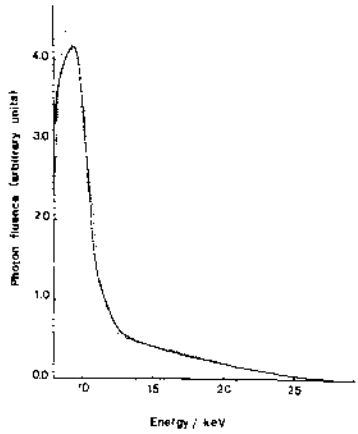

Fig.2: Distributions once smoothed, obtained at 30 kVp nominal high voltage: ——— calculated spectrum, ----- reconstructed spectrum.

pratical energy range. The second irradiation technique satisfies this requirement with its large, low energy interval. The very high flux density and the presence of very low quantum energies do not allow the measurement on the direct beam. The comparison is then possible only by calculation of the spectral distribution. This is achieved by means of the procedure described by Birch et al. (3) for the continuous spectrum and by means of the one worked out by Casnati et al. (4) for the characteristic lines. The experimental reconstructed distribution (dotted lene) is compared with the theoretical spectrum (continuous line) in figure 2. The last one clearly includes the con-

volution with the response function of the spectrometer distorted for the scattering dependent energy interval change. Normalization criterion is like the one used for 60 kVp spectra but applied to the continuous component alone. Comments on this result are quite similar to those relevant to figure 1.

CONCLUSION

On the ground of the results so far gained the scattered radiation spectrometry seems a promising technique for a better description of low energy X-ray beams as their interaction properties and their large diffusion would need.

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EFFICACITE DE COMPTAGE DE GELS SCINTILLANTS

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Nos travaux antérieurs (1) sur des gels scintillants contenant du dioxanne, PPO, Diméthyl POPOP, et du naphtalène et dont la viscosité était augmentée par addition de quantités croissantes de Cab-o-sil, avaient mis en évidence une augmentation parallèle du rendement de scintillation. Les résultats obtenus en présence d'inhibiteurs chlorés (CCl_4, $CHCl_3$) ont permis de conclure que la silice ne modifie pas l'efficacité du transfert d'énergie. Par contre l'augmentation du rendement de scintillation serait due à l'augmentation du rendement quantique de fluorescence de PPO quand la viscosité du milieu devient plus élevée. Pour confirmer cette hypothèse nous avons étudié une autre série de gels scintillants de même composition hormis la silice et de viscosité croissante, obtenus par addition au liquide de base de différentes quantités de HP_{55} (hydroxypropylméthyl-phtalate de cellulose), seul produit parmi les nombreux testés, qui conduisait à un milieu macroscopiquement homogène. D'autre part, nous avons élargi notre travail à d'autres gels de silice scintillants, différents par la nature du soluté primaire utilisé.

TECHNIQUES EXPERIMENTALES

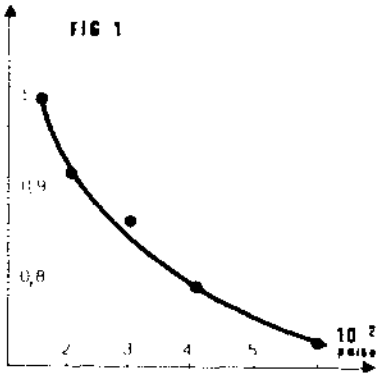
HP_{55} de poids moléculaire moyen égal à 20 000 daltons est fourni par la Shin Etsu chemical company. Toutes les autres conditions expérimentales ont été rapportées par ailleurs (1, 2).

RESULTATS ET DISCUSSION

- Etude de gels scintillants à base de HP_{55}.

Les premiers essais effectués à concentration de $HP_{55} < 10^{-4}$ M ne produisent pas de variation détectable de la viscosité, ni de variation du rendement de scintillation lors d'irradiation γ. Pour des concentrations plus élevées ($0,1\ 10^{-3}$ M < HP_{55} < $1,5\ 10^{-3}$M), qui conduisent à une phase d'aspect gélifié, nous obser-

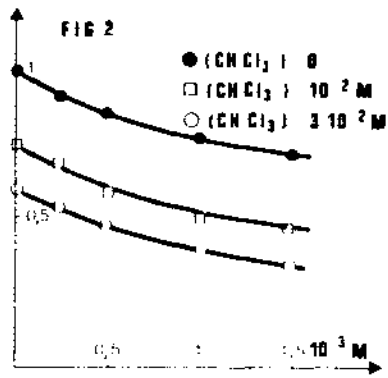
vons (figure n° 1) une diminution du rendement de scintillation, alors qu'à viscosité identique, la silice provoquait une augmentation de ce paramètre. Les densités des deux milieux étant très voisines, cette différence ne peut être imputée à une différence d'absorption du rayonnement ionisant.



Rendement de scintillation = f (viscosité)

Pour tester l'activité de HP_{55} sur les mécanismes de transfert d'énergie dans le gel, nous avons mesuré les variations du rendement de scintillation en présence de $CHCl_3$ ou de CCl_4. Pour des concentrations fixes de ces inhibiteurs chlorés comprises entre 0,5 et $3\ 10^{-2}$ M, les courbes de rendement de scintillation en fonction de la concentration molaire en agent gélifiant sont parallèles entre elles, leurs pentes négatives sont identiques pour une même concentration de HP_{55}. (exemple figure n°2). De même les courbes des rendements de scintillation en fonction de la concentration en inhibiteur chloré, pour différentes concentrations de HP_{55} (non présentées) sont parallèles. Il n'y a donc aucune compétition entre HP_{55} et les dérivés chlorés en ce qui concerne la capture d'énergie par voie ionique ou excitonique, résultat identique à celui obtenu avec la silice.

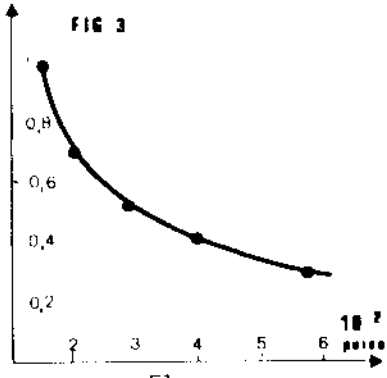
Nous avons alors étudié la fluorescence du naphtalène, de PPO et de diméthyl POPOP en solution dans le dioxanne en présence de HP_{55}. L'absorption lumineuse de ces composés à leur maximum d'excitation n'est pas modifiée, de même que l'allure de leur spectre d'émission. Par contre, l'intensité de fluorescence au maximum d'émission normalisée par rapport à celle obtenue en l'absence de HP_{55} est nettement diminuée

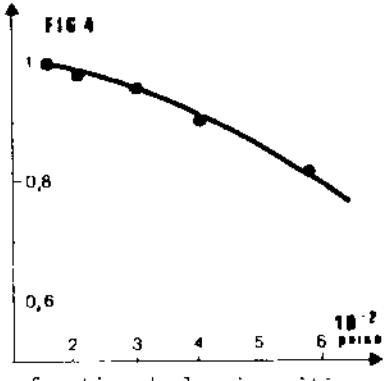


Rendement de scintillation = f [HP_{55}]

dans le cas du naphtalène (figure 3), de façon moins importante pour PPO (figure 4) alors quelle ne varie pas pour diméthyl-POPOP.

Nous avons conclu précédemment (1) que la modification de viscosité du milieu était responsable des

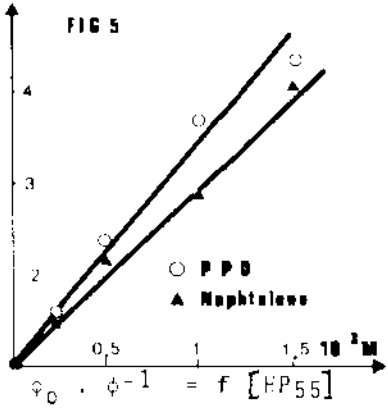




Fluorescences normalisées en fonction de la viscosité

naphtalène (λ_{ex}=322nm, λ_{em}=370nm) PPO (λ_{ex}=322nm, λ_{em}=370nm)

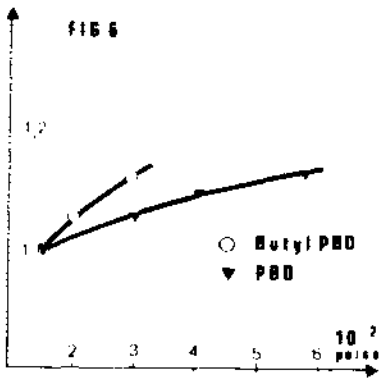
modifications du rendement de fluorescence de PPO. Il est alors logique de perser que cet effet persiste en présence de HP_{55} pour une viscosité identique. Il faut donc en conclure que l'inhibition observée dans ce cas est en fait le résultat de deux mécanismes qui s'oppose : une exaltation de la fluorescence de PPO due à la viscosité et une inhibition propre à HP_{55}. Cette inhibition se manifeste aussi bien sur le naphtalène que sur PPO mais elle est apparemment plus importante sur le premier car il n'y a pas alors d'effet d'exaltation de fluorescence par viscosité. Les variations des rapports $\phi_0 \cdot \phi^{-1}$ de fluorescence en présence respectivement de silice et de HP_{55} sont rapportées sur la courbe 5, en fonction de la concentration en HP_{55}. Les résultats sont du même ordre de grandeur pour les deux composés. Les droites obtenues par regression linéaire présentent une pente trop élevée pour pouvoir être considérées comme des droites de Stern-Volmer. HP_{55} inhiberait la fluorescence du naphtalène ou de PPO par un mécanisme de quenching statique. Ainsi l'étude de gels scintillants contenant



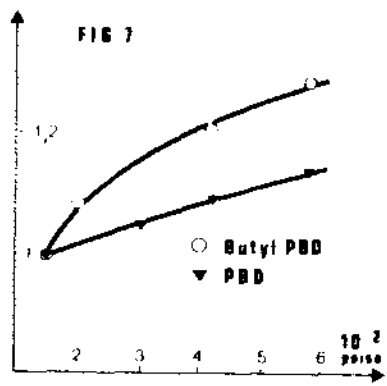
HP_{55}, qui certes sont moins intéressants du point de vue pratique que ceux à base de silice du fait de leur moindre rendement de scintillation, nous ont permis de confirmer l'importance de la viscosité. Les valeurs plus élevées de ce paramètre augmentent le rendement quantique de fluorescence de PPO et ainsi le rendement de scintillation. Il convient donc, dans la pratique, d'opérer les mesures de radioactivité sur l'échantillon à doser comme sur la gamme d'étalonnage en maintenant la viscosité constante pour éviter toute erreur systématique.

- Etude de gels de silice scintillants.

L'augmentation du rendement de scintillation par addition de silice avait été rapportée par Germai(3,4) dans le cas de PPO. Nous avons voulu prolonger ce travail aux autres solutés primaires couramment utilisés et nous avons donc étudié toute une gamme de gels de silice scintillants, différents uniquement par la nature du soluté primaire. Aucune augmentation du rendement de scintillation n'a été observé dans le cas de αNPO, αNPO, PPO, CPO et PPD lors de l'augmentation de la viscosité du milieu par addition de silice et parallèlement ces solutés ne présentent pas de variation de rendement de fluorescence dans ces conditions. Par contre PPD et Butyl PPD se comportent comme PPO (figures 6,7) et ces 3 solutés primaires sont donc les plus intéressants pour la



Fluorescence normalisée= f(viscosité)



Rendement de scintillation= f(viscosité)

fabrication des gels scintillants. On comprend dès lors leur emploi très fréquent à cet usage qui apparemment ne résultait que de constatations empiriques.

Remerciements : nous remercions M.BOUTEILLE pour son aide technique.

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STUDY AND MEASUREMENT OF THE ATMOSPHERIC POLLUTION BY ^{85}Kr

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The continuation of the total release of ^{85}Kr in the reprocessing plants, for the expected growth of nuclear power up to the year 2000, would cause the largest specific nuclear background increase, for the next decades. This irreversibel pollution would yield a significant β-radiation skin dose to the world population, which justifies retention techniques on cost-benefit grounds (1). This conclusion could be invalidated by application of ICRP 26. However, the actual insufficient knowledge on skin cancer rise at low dose, necessitates a conservative policy. Further research is necessary, and should take synergism between ionizing and ultra-violet radiation (2) into account. Secondly, the micro- and macro-climatological consequences of aerosol distrurbancies by ^{85}Kr should be investigated. Our laboratory has started a research project on this subject.

Since different reprocessing centres are sited in Western Europe (Fig.1), the follow-up of ^{85}Kr activity is very important in this area. In our laboratory, a method, similar to the E.P.A. method (3) for measurement of low ^{85}Kr activities in air samples, has been worked out successfully. The measuring cycle has been calibrated. The results of a large number of measurements performed in Ghent during 1979, are presented and discussed.

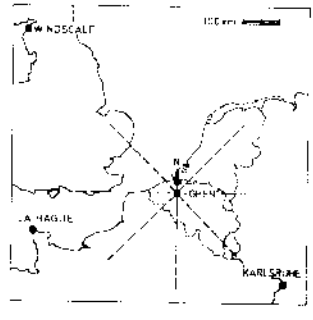

Fig.1

SAMPLING AND COUNTING METHOD

The set-up for sampling the air, chromatographic separation of Kr, and condensation in a scintillation vial (a), has been reproduced in Fig.2. About 1 m^3 of air is sucked to and absorbed on an activated charcoal trap (IV), passing cooling trap I, for water removal, molecular sieve MS II, for H_2O and CO_2, and a cooling trap III for final CO_2 removal. Immersion in liquid nitrogen is applied for any cooling. Then the charcoal trap IV is heated to 100°C and purged with helium, such that all gases are transferred to MS V. This 5Å molecular sieve is 1.5 m long, with 6.5 mm inner diameter. After removal of the cooler, trap V is eluted in a helium carrier flow, to a thermal conductivity detector. The sequence of gases, Kr being identified with a mass spectrometer, is the same as in (4). The chromatographic resolution between Kr and methane is

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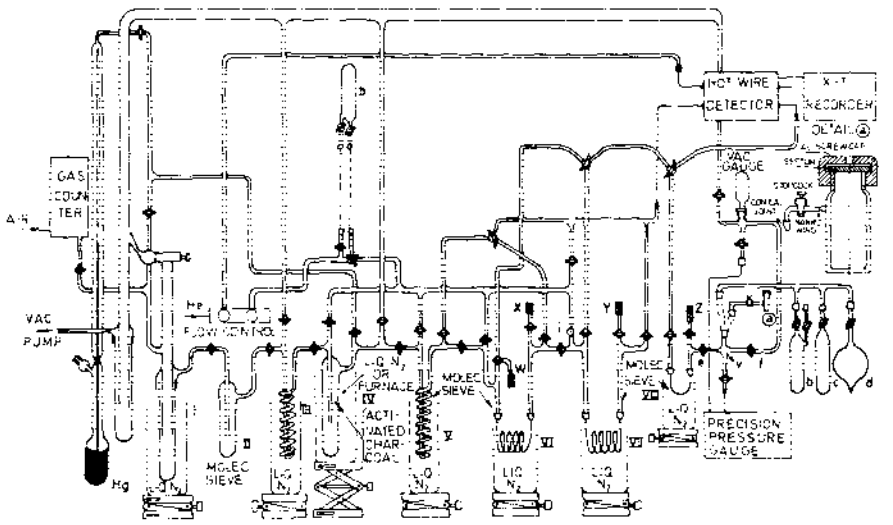

Fig.2 : Set-up for measurement of ^{85}Kr activity in air

about 1.5. Switching of the taps allows to direct the Kr fraction to trap VI, the residual gases being vented (W). This procedure is repeated for transfer to traps VII and VIII, thus augmenting the purity. Traps VI and VII, 3 m long, 2 mm I.D., contain MS 5Å, 60-80 Mesh. Helium is pumped off from the cooled trap VIII, but there remains a residual He pressure, which has to be corrected for (see Calibration Method). The gases on trap VIII are expanded at room temperature to a cross of glas tubes (V), with precisely known volume. The pressure is measured with a precision pressure gauge (0 to 19 psi, Texas Instr.). The Kr gas is then condensed in a scintillation vial and the narrowing is fused. The Instafluor scintillator is injected through a Teflon coated septum. Counting the ^{85}Kr activity (with efficiency 72%), and calculating the Kr mass, from the known temperature, pressure and volume, allows to determine the ^{85}Kr concentration relative to the known amount of Kr in air (1.14 cm^3/m^3). The Kr recovery varies between 55 and 70%.

CALIBRATION AND ERROR DISCUSSION

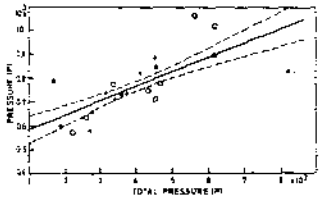

Fig.3 : He pressure versus total (He + Kr) pressure

The calibration is performed with a ^{85}Kr source with known activity (8% accuracy, all uncertainties given for 99% confidence level). A fraction of this source is introduced in the scintillation vial for the determination of the counting efficiency. The total dilution uncertainty is 2.8%. The He residual pressure (Fig.3) on trap VIII has been found to be dependent on the Kr content, probably through covering of the pores in the molecular sieve by Kr. The net residual

He pressure, together with the Kr recovery, have been measured by mixing a known mass and activity of ^{85}Kr from bottle 5 (Fig.1) in the He stream, and directing this stream to trap V and consequently through the same path as for a atmospheric measurement. The ^{85}Kr partial pressure is derived from the measured activity. The sets of measured He pressures, obtained by substraction, have been reproduced on Fig.3, as a function of the total pressure. A straight line fitting allows to correct the total pressure in an atmospheric measurement with an uncertainty of about 2.2%. The computed error on the measured amount of Kr + He, due to errors on temperature, pressure and volume, is 1.7%. The scintillation counting error is 9.7% for normal counting rates, about twice the scintillation background. The total uncertainty is 13% (99% conf. level).

The installed equipment thus allows measurement of the atmospheric ^{85}Kr concentration, within quite narrow error limits. The accuracy of the scintillation measurement has been verified by counting the activity of Kr from a bottle, bought in september 1971. The gas is expanded in glas tubes V and in the scintillation vial, at the same pressure existing in the vial in case of atmospheric measurements. The measured activity, corrected for decay, is 0.46 Bqm^{-3}, which is in good agreement with the values measured by Stevenson (3) in the same year, ranging from 0.43 to 0.50 Bqm^{-3}.

MEASUREMENTS AND DISCUSSION

In the course of the year 1979, a large number of atmospheric measurements have been performed, which are reported on Fig.4. The base line of the activity is rather constant, and equals about 0.70 Bqm^{-3}. There is thus a strong increase of the ^{85}Kr background, from 0.46 Bqm^{-3} in 1971 to 0.67 Bqm^{-3} in september 1977 (measured with a gas bottle of that year) and 0.70 Bqm^{-3} in 1979. This increase must be essentially related to the world reprocessing of nuclear fuel in this period. There is only a small increase from 1977 to 1979, and no significant change of activity during 1979. Our measurements will be continued in the next years, to allow monitoring of the evolution.

On Fig.4 a number of peak measurements can be observed,which can be related to the transport of released activities in reprocessing

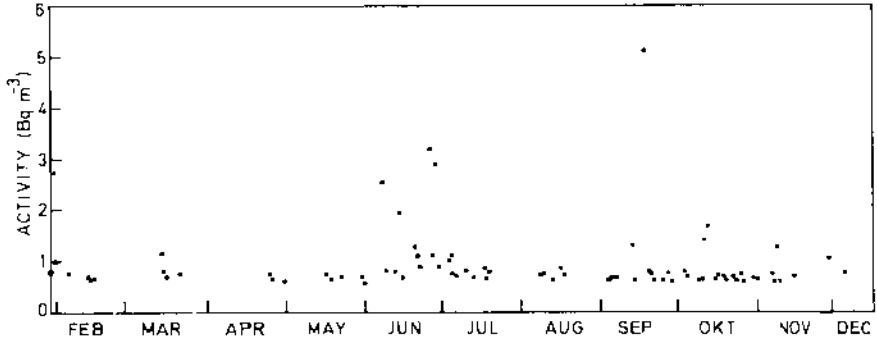

Fig.4 : Measured ^{85}Kr activities in 1979

centres, under favourable meteorological conditions. In regard of the predominant S-W wind direction, during measurement of the increased activities, their origin should be situated in La Hague. Pilot calculations of the meteorological transport, and realistic estimations of the releases, indicate that this correlation is justifiable. Thus, if the detailed source data would be available, which is only the case for Karlsruhe, these measurements of the ^{85}Kr activity would become important for the study of meteorological transport on mesoscale, ^{85}Kr serving as a tracer.

CLIMATOLOGICAL IMPACT OF ^{85}Kr

It is well known (5) that a number of radio-chemical reactions, in particular with SO_2, lead to formation of aerosols from the gas phase. Reaction vessels (20 l), filled with pure air and known concentrations of SO_2, NO_2 and H_2O, are being irradiated with U.V. light and with a γ-source. The total particulate and SO_4^{2-} concentrations are measured in function of the different parameters. In the near future, experiments will be performed in large volumes (m^3) filled with ^{85}Kr at realistic concentrations. Aerosol distribution measurements will be used for the assesment of transformation rates of the precursors. Finally the effect of homogeneous nucleation induced by ^{85}Kr on ambient air will be measured, and its relevance for smog formation and acidification of the environment will be evaluated.

ACKNOWLEDGEMENTS

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NEUTRON SPECTRA AND DOSE EQUIVALENT INSIDE REACTOR CONTAINMENT*

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INTRODUCTION

The purpose of this study is to measure, characterize and evaluate neutron radiation dose equivalent rates and neutron energy spectra at selected commercial nuclear facilities where operating plant workers may be exposed to neutron radiation fields. Improved understanding and control of occupational neutron exposure should result from this study.

Neutron exposures to operating plant workers have not been observed on dosimeters in the past due to the lack of sensitivity of the personnel dosimeters used and the energy ranges suspected to be present in commercial nuclear facilities, particularly PWR plants. Most of the facilities presently use nuclear track emulsions to detect fast neutrons. The emulsions and most of the newer dosimeters based on track-etch techniques are not sensitive to the neutrons in the energy ranges of the leakage spectra which may be present in the commercial power reactor plants. Average energies are expected to be below 500 keV.

Recently, albedo type neutron dosimeters have become available for use at these facilities and relatively large neutron dose equivalents are being observed, especially when workers enter containment during full power operation.

At the present time, the albedo type of personnel dosimeter is the only available dosimeter which seems to have adequate sensitivity for neutrons in this energy range.

This study was designed to provide measurement data from a minimum of six nuclear power sites, which were selected to include reactors manufactured by each of the four U.S. NSSS vendors and nuclear plants with at least four different architect-engineers. The measurement data include: 1) a determination of neutron dose and dose equivalent rates inside and outside of containment; 2) neutron spectral and flux distributions at selected locations both inside and outside of containment; 3) special monitoring with currently available neutron dosimeters, i.e., albedo, film and fission fragment, at each site; and 4) correlate instruments and dosimeter data to flux and spectral distributions to determine their proper response and interpretation. Lawrence Livermore Laboratory personnel have assisted with measurements at two of the sites and analysis of multisphere data taken at all the sites.

To accomplish this study, tissue equivalent proportional counters (TEPCs) were used to measure the neutron dose and dose equivalent rates.

* Work performed on this project was sponsored by the U.S. Nuclear Regulatory Commission.

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At the same time, currently used neutron survey instruments, such as the Snoopy, were used for dose equivalent rate measurements. The multisphere technique and other spectrometers were used to estimate the neutron spectrum at selected locations inside the outside containment and were supplemented by TEPC data. Currently available dosimeters were then placed on phantoms to provide data which can be used to establish their capability for adequate personnel dosimeter measurement requirements as stated in Regulatory Guide 8.14.

The personnel dosimeters used were obtained from vendors who normally supply this service to licensees and include film, TLD and track etch detectors. We used an improved version of the Hanford Multipurpose Dosimeter (MPD). A combined track-etch-albedo neutron dosimeter developed by Hankins and Griffith of Lawrence Livermore Laboratory (1) was also used.

Neutron calibration exposures were conducted with a bare Cf-252 neutron source, and with spheres of Al, H_2O and D_2O surrounding the Cf-252 source. Neutron monitoring instruments, which use moderated BF-3 detectors, tend to read high in the moderated Cf-252 spectra when they are calibrated with the bare Cf-252. The tissue equivalent proportional counter correctly measures all absorbed dose and dose equivalent for event sizes larger than about 5 keV/μm. Albedo TLD dosimeters were exposed on phantoms to the various Cf-252 source configurations. When calibrated with bare Cf-252, the albedo dosimeters read a factor of 2 to 3 high for the moderated Cf-252 sources.

EXPERIMENTAL RESULTS

The multisphere spectrometer system consists of five polyetnylene spheres of various sizes plus a bare and cadmium covered neutron detector. For a measurement point, the thermal neutron count rate was measured using a ^{6}LiI(Eu) detector in seven different configurations. A spectrum unfolding computer program, known as LOUHI (1), is used to determine the neutron spectrum from each set of data. Response functions for each of the detector geometries are included in the computer program. The response functions for each of the sphere sizes,7.6, 12.7, 20.3, 25.4 and 30.5 cm diameter, were calculated by Sanna (2).

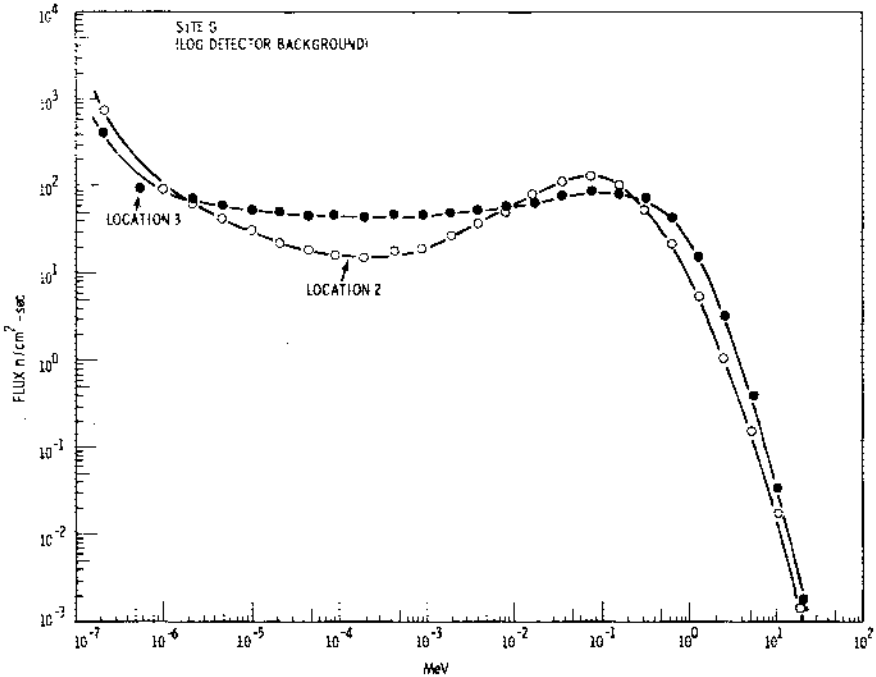
Typical results for spectra measured inside containment at a PWR nuclear power plant are shown in Figure 1.

Because of the "low" resolution of the multisphere system, the unfolded spectra show no sharp peaks or edges. The computer program provides 26 neutron energy groups from thermal to about 20 MeV.

The differential flux, integral flux, integral dose equivalent, energy band width, and flux density are provided for each energy. Kerma rate, dose rate and average energy for each spectrum is also given. The average energy for the spectrum is 62 keV at Location 3. The range of average energies for all locations at all the PWR sites was from about 10 keV to 90 keV. Readings taken with standard neutron survey instruments at the same locations as the multisphere system show a response about a factor of 2 higher than the multisphere.

Tissue equivalent proportional counters (TEPCs) were used at several locations inside containment to measure absorbed dose directly for comparison with "Snoopy" and Rascal monitoring instruments. At most locations a 12.7 cm diameter spherical counter filled to 1 μm equivalent size with tissue equivalent gas was used.

FIGURE 1. Typical Spectra Inside Containment at a PWR

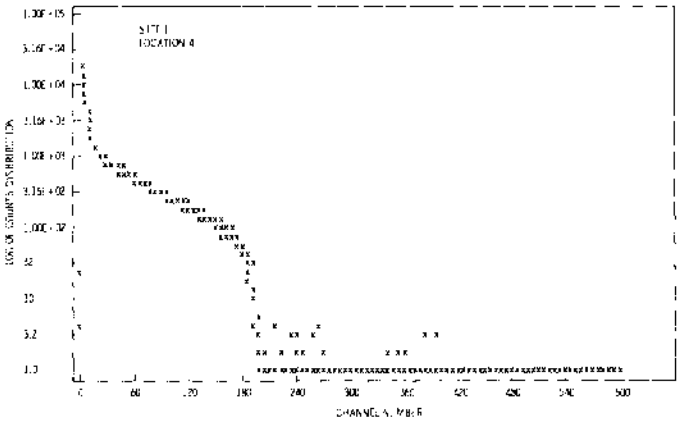


A typical event size spectrum from the TEPC is shown in Figure 2. Data are shown as they appear on a multichannel analyzer for Location 4 at Site I. The sharp drop in the number of events per channel at about channel 200 corresponds to 28 keV/μm, the maximum energy loss a proton recoil can have in the TEPC.

This drop point is often called the proton drop point and is used to calibrate the measurement system. Absorbed dose is obtained by multiplying the number of events at each energy by the energy and integrating over the spectrum.

The absorbed dose rate for the spectrum in Figure 2 is 0.51 Gray/hr. The quality factor determined by Rossi (3) analysis is 10.8. In this case the analysis of the data for quality factor includes all events greater than 5 keV/μm. At this location the dose equivalent is about 560 mrem/hr. Measurements with monitoring instruments gave readings of 1300 to 1600 mrem/hr.

FIGURE 2. Typical Event Size Spectrum from TEPC



Both the multisphere system and the TEPC indicate that the monitoring instruments are a factor of 2 or so high in the well-scattered neutron fields inside containment.

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DOSIMETRY OF CRITICALITY ACCIDENTS USING ACTIVATIONS OF THE BLOOD AND HAIR*

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INTRODUCTION

The evaluation of the dose that a person received in a criticality accident can be difficult. Most accidents have occurred when the person was not wearing nuclear accident dosimetry and since the NRC no longer requires these dosimeters, future dose evaluations may have to be based on body activations and gamma-to-neutron dose ratios. To aid in a dose evaluation we have compiled in a table (available from the author) the results from numerous criticality accident studies using 10 different critical assemblies, each with different neutron leakage spectra. There are several problems involved in applying these results accurately, the most significant problem being the determination of the configuration of the fissile material at the time of the accident. Other problems include a lack of information concerning the location, orientation, and possible shielding between the person and the accident assembly.

DOSIMETRY STUDIES

The literature contains a number of criticality accident dosimeter studies,[1] including a mock-up study of the accident which occurred in the Y-12 facility at Oak Ridge, Tennessee and four studies sponsored by the International Atomic Energy Agency. Also studied were the Health Physics Research Reactor (HPRR) at the Oak Ridge National Laboratory and five different types of critical assemblies at the Los Alamos Scientific Laboratory.

These dosimetry studies were made with reactors and critical assemblies having extremely diverse configurations. Schematics of these assemblies, drawn to scale in Fig. 1, show major features that are important in modifying the neutron leakage spectrum. Leakage spectra vary greatly, and as a result the blood or hair activations for the same neutron dose will be very different from each assembly.

SODIUM ACTIVATION OF BLOOD AND ^{32}P ACTIVATION OF THE HAIR

The most useful activations of the body for dose estimation following a criticality accident are those of the blood and the hair.

Activation of ^{23}Na in the body is determined by counting the ^{24}Na produced in about 20-cm^3 of blood with a NaI or GeLi detector. The 1.369-MeV gamma ray is used to determine the ^{24}Na activity in μCi of ^{24}Na per mg of ^{23}Na. The probability of neutron capture by sodium in the human body is fairly constant for neutron energies from thermal to 5 MeV,[2] but the kerma dose delivered by

*Work performed under the auspices of the U. S. Department of Energy by the Lawrence Livermore Laboratory under contract No. W-7405-ENG-48.

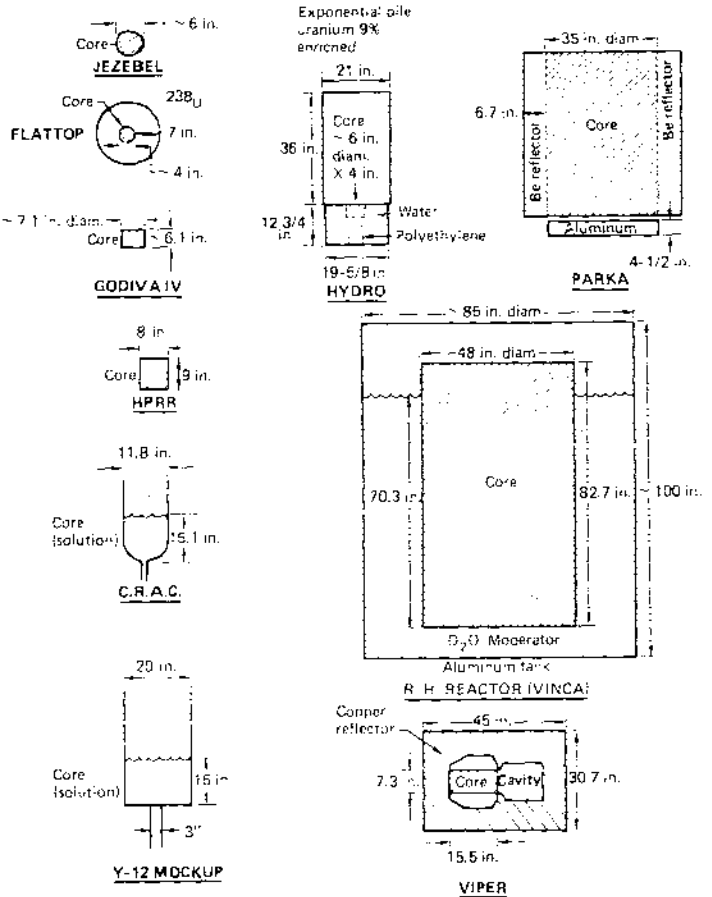

Figure 1. Schematic of the critical assemblies and reactors used in dosimetry studies (drawn to scale), showing the core configuration and components that affect the leakage neutron spectrum.

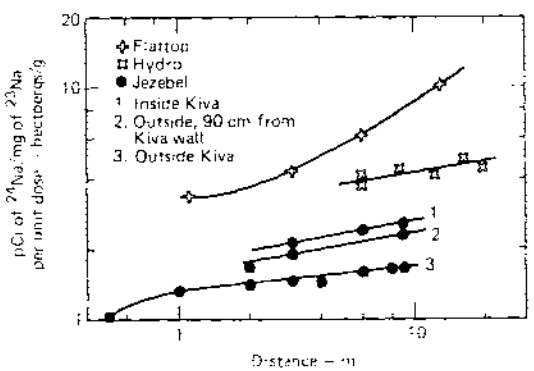

Figure 2. Blood sodium activation per hectoerg/g of neutron dose as a function of distance from the assembly. Jezebel results with assembly located inside, outside, and near the outside wall of a building are given.

neutrons is predominantly from fast neutrons. Consequently, ^{24}Na activation in the blood is not proportional to the neutron dose.

Figure 2 shows blood sodium activation from the Jezebel assembly as a function of the distance from the assembly, when the assembly is (1) in a building (kiva), (2) outside, and (3) near a concrete wall. Increases in activations are caused by scattered, low energy neutrons, which activate the blood but contribute little to the neutron dose. Similar curves for the Hydro and Flattop assemblies are shown.

Sodium activation in the body can be detected by placing a G-M instrument in the abdomen. The G-M reading is proportional to the Na activation and the resulting curves (available from the author) are very similar in shape to the curves given in Fig. 2.

Activation of the sulfur in the hair by fast neutrons (>2.9 MeV) produces the beta emitter ^{32}P. If the individual has not been contaminated, the technique preferred, because of accuracy and simplicity, is the direct counting of the hair with a beta counter following the procedure described in detail by Hankins.[3] The counter is calibrated using a ^{90}Sr-^{90}Y source, and the count rate of the hair in counts/min/g is determined and divided by 1.77 to obtain the fast neutron dose in rads (above the 2.9 MeV threshold for sulfur). Unfortunately, the percent of the neutron kerma dose that is from neutrons having energies of 2.9 MeV varies from 2.1 to 48%.

RELATIVE GAMMA AND NEUTRON DOSES

The ratio of gamma-to-neutron doses can be used to determine the total dose if either the gamma or the neutron dose is known. Unfortunately, the estimation of dose based on gamma-to-neutron ratios may not be accurate since this ratio for the various critical assemblies varied from around 0.09 to 2.9. The effect of shielding material is also significant. For example, the HPRR ratios vary from 0.19 for a bare assembly to 1.9 for a polyethylene-shielded assembly.

EVALUATION OF THE DOSE

Following an accident the configuration of the fissile material at the time of the excursion is established, if possible. This configuration can then be compared with those shown in Fig. 1, and if it is similar to one of them, the experimental results (available from the author) obtained with the assembly most closely resembling the excursion can be applied to evaluate the doses.

The neutron leakage spectrum and subsequent activations of hair and blood are also affected by other factors which include: shielding the exposed person; whether the exposed person was indoors or outdoors; where he was with respect to walls or floors; his orientation; the angle of exposure; and his distance from the assembly. Several of the assemblies shown in Fig. 1 have been used to evaluate the effect of many of these factors.

DOSE EVALUATION USING A COMBINATION OF THE BLOOD AND HAIR ACTIVATIONS

The most serious problem following an accident is a lack of information on the configuration of the fissile material which makes it impossible to find in Fig. 1 an appropriate critical assembly. Furthermore, there is often a lack of other information necessary to

accurately assess the dose. The dose can still be evaluated by using a combination of blood and hair activations.

The activation of the blood and hair is determined as described previously and the ratio of the sulfur fluence to the blood sodium activation is calculated. In Fig. 3, we have plotted the ratio of sulfur fluence to blood sodium activation as a function of blood sodium activation and have drawn two curves.

To evaluate the dose using this procedure we first read the blood activation (in pCi $^{24}Na/mg^{23}Na$ per rad of fast neutrons) from the curve at the point corresponding to the measured sulfur-fluence/blood-sodium-activation ratio. Then, we divide the blood sodium activation by that quantity to obtain the neutron dose that the individual received.

This procedure is independent of the neutron spectrum, hydrogenous shielding, victim orientation and distance, and room scatter. Where there were thick metal shields (>10 cm) either associated with the assembly or between the person and the assembly, the curve on the left must be used. Fortunately, thick metal shields are not commonly used. A dose estimate accurate to within ±20-30% should be obtainable using this procedure.

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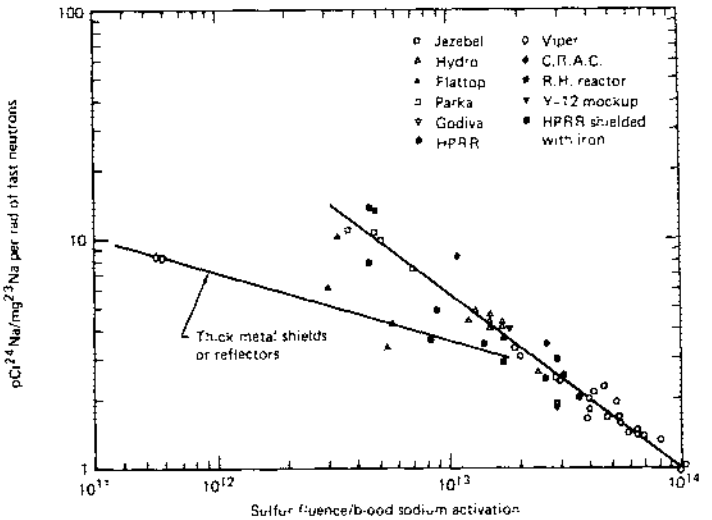

Fig. 3. Curve used to determine the neutron dose using a combination of blood and hair-activation data.

BETA DOSIMETRY WITH SURFACE BARRIER DETECTORS*

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INTRODUCTION

In the vicinity of small, unshielded radioactive substances, the dose rate due to beta radiation may be substantially higher than the dose rate produced by gamma radiation. However, for radiation protection monitoring, a small dosimeter for ß-radiation is still needed, by means of which it will be possible to determine the dose rate on the surface of radioactive substances in an energy-independent manner.

A very small dose rate meter for ß-radiation has been proposed by G. Nentwig (1) (2). He uses scintillators of 10 - 25 mg/cm² thickness and counts the pulses exceeding a suitable discriminator threshold. A low-energy beta particle, passing through the scintillator in the same way as a ß-particle of high energy while giving off only part of its energy, generates a larger pulse, since the linear stopping power dE/dx increases with dropping ß-energy. In connection with a suitable discriminator threshold, a pulse generated by such a particle has a higher probability of surpassing the threshold than a pulse produced by a ß-particle of higher energy. By variation of the scintillator thickness and discriminator threshold, Nentwig obtains a small ß-dosimeter featuring an energy-independent indication. To our knowledge, this interesting proposal made by Nentwig has not been further pursued.

Nentwig's idea has been transferred by us from scintillators to semiconductor detectors (3). In the case of surface barrier detectors, the thickness of the sensitive layer is changed with the aid of the detector voltage. At low voltages, very small detector thicknesses can be obtained.

EXPERIMENTAL DESCRIPTION

By means of a surface barrier detector, the count rate was determined for various detector voltages and discriminator thresholds. As described by Nentwig (1) (2), the pulse rate surpassing a set discriminator threshold was determined integrally. In addition, a single-channel analyzer was used, and the channel width was adjusted optimally as a further parameter.

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Measurements were conducted using a ND-7-S[1] detector with an active surface of 7 mm². The detector was covered additionally with a light-tight plastic foil of 1 mg/cm². Measurements were taken at several distances from different β-emitters. For the individual measuring points, the dose rate had been determined by means of an extrapolation chamber featuring a front electrode thickness of 7 mg/cm².

The β-emitters used were one nuclide of low peak energy (Pm-147), one of medium peak energy (Tl-204), and one of high peak energy (Sr-90/Y-90). In the case of Sr-90/Y-90, the Sr-90 radiation could be absorbed by plexiglass. Additional measurements were carried out on the secondary normal for β-radiation developed by PTB (Physikalisch Technische Bundesanatalt, Braunschweig) (4).

EXPERIMENTAL RESULTS

The sensitivity[2] was determined for various detector voltages, discriminator thresholds and for 2 time constants of the amplifier (0.25 µs and 0.5 µs). In connection with measurements using the single-channel analyzer, the channel width is varied additionally.

When using the discriminator, counting all of the pulses which surpass a set threshold as was done by Nentwig (1) (2), an energy-independent determination of the β-dose rate was only possible with the time constant of 0.25 µs (Fig. 1). For further measurements, the discriminator threshold of 200 mV was generally selected. The sensitivity to β-radiation is then equal to 6.7 cpm/mrad/h within ± 20 %. In the course of earlier investigations (3), we had not succeeded in determining the β-dose rate in an energy-independent manner without the single-channel analyzer.

The use of a single-channel analyzer permits an energy-independent determination of the β-dose rate with both time constants.

Fig. 2 shows the energy dependence of the sensitivity to γ-radiation. For Cs-137-γ-radiation, the sensitivity is equal to that for β-radiation.

The sensitivity as a function of the dose rate is shown in Fig. 3. With Sr-90-β-radiation it was only possible to measure up to 600 rad/h. When measuring with the time constant of 0.25 µs, the sensitivity to γ-radiation only decreased by 20 % at 1400 rad/h as

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[1] Canberra Elektronik GmbH

[2] Sensitivity denotes the ratio of count rate : dose rate below 7 mg/cm² for measurements using the discriminator, and the ratio of count rate in the channel : dose rate below 7 mg/cm² for measurements using the single-channel analyzer.

compared to values at low dose rates. A corresponding decrease in sensitivity already occurs at 300 rad/h in connection with measurements with 0.5 µs and the single-channel analyzer.

Measurements of the dose dependence are shown in Fig. 4 on a second detector. The sensitivity to radiation from 2 nuclides for 2 settings of the electronic system was determined in each case after irradiation with a predefined dose. Up to doses of 50 000 rad, no sensitivity change could be observed.

One disadvantage of the detectors lies in the fact that they become easily defective. For this reason, further measurements are planned using ion-implanted silicon detectors.

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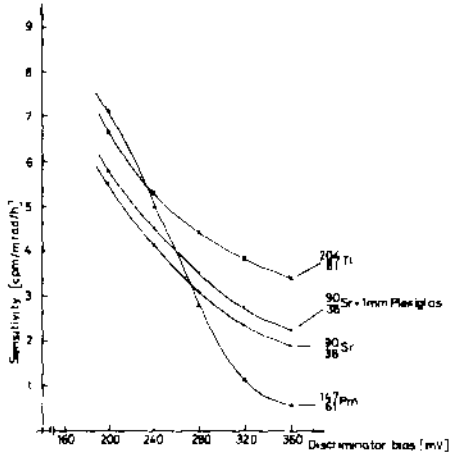

Fig.1 Sensitivity as a function of the discriminator bias for β-radiation of different nuclides
Detector ND-7-S, number 1998
Detector voltage -20 V
Gain 100 x
Shaping time constant 0,25 µs

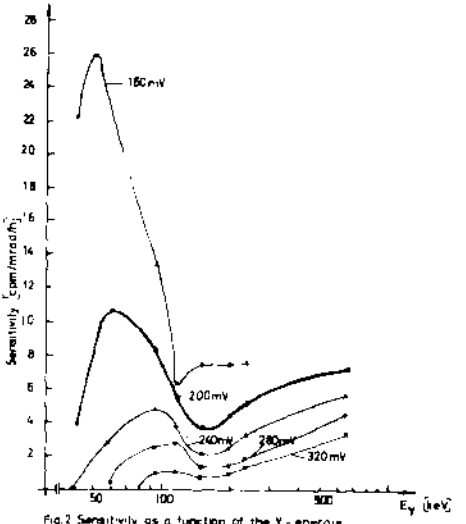

Fig.2 Sensitivity as a function of the γ-energie
Parameters are the discriminator bias
Detector voltage -20 V
Shaping time constant 0,25 µs

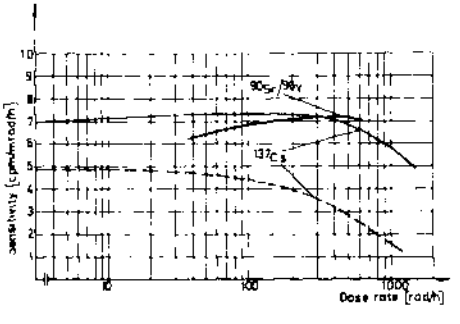

Fig.3 Sensitivity as a function of the dose rate for $^{90}Sr/^{90}Y$ β-rays and ^{137}Cs γ-rays
Detector ND-7-S
Detector voltage -20 V
— Discriminator, bias 200 mV, shaping time constant 0,25 µs
- - Single channel analyser, bias 240 mV, channe, width 130 mV
Shaping time constant 0,5 µs

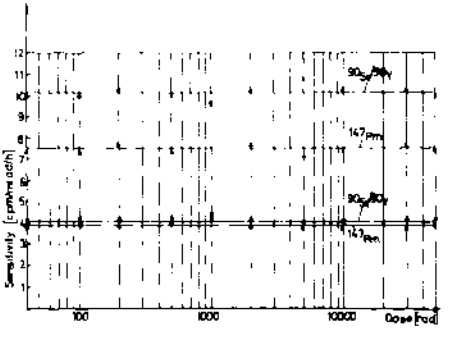

Fig.4 Sensitivity as a function of the dose for $^{90}Sr/^{90}Y$ – and ^{147}Pm – β-radiation
Detector ND-7-S, number 1999
Shaping time constant 0,25 µs
• Discriminator, bias 200 mV
▲ Single channel analyser, bias 230 mV, channel width 100 mV

^{230}TH ASSAY BY EPITHERMAL NEUTRON ACTIVATION ANALYSIS

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^{230}Th (Ionium) is an important member of the ^{238}U decay chain that has recently been shown to be the major contributor to the environmental dose from actinides. (1) Earlier studies (2-4) have shown that following inhalation of ^{238}U + daughters, the uranium is cleared relatively rapidly from the lung via ciliary action and the gastrointestinal tract, while the ^{230}Th daughter is removed much more slowly. This produces a "biological enrichment" of ^{230}Th in later fecal samples and further suggests that ^{230}Th may be of greater import from the standpoint of dose than heretofore considered.

The lack of a simple, inexpensive, rapid, and sensitive assay method for ^{230}Th has limited studies of the metabolism and fate of this nuclide within biological systems. The most widely used available methods involve time-consuming, complex, and often tedious chemical separations followed by alpha counting; accuracy and sensitivity of these methods may be wanting, and interferences from uranium, other actinides or isotopes of thorium may further complicate the assay.

The nuclear properties of ^{230}Th, however, suggest that neutron activation analysis (NAA) may provide a simple, inexpensive, rapid and highly sensitive method of assay. This nuclide has a 1010 barn epicadmium resonance absorption cross-section for activation to 25.52 hour ^{231}Th; ^{231}Th has a complex gamma ray spectrum with the most prominent energies being an 84.4 keV complex with a yield of 6.55% (5-6). Although ^{231}Th emits numerous other photons, yields are significantly (i.e. one or more orders of magnitude) lower, with the exception of a photon at 89.95 keV, which has a yield of 0.95%, or about 1/7 that of the complex line at 84.4 keV.

The suitability of the NAA method in the presence of the ^{238}U parent and other members of the natural uranium decay chain was examined by activating a quantity of various uranium ores with both a reactor (TRIGA MKI) thermal neutron spectrum having a cadmium ratio of approximately 10. The ore was exposed bare and wrapped in 0.5 mm cadmium sheet to eliminate the subcadmium (i.e. thermal) neutrons. The resultant spectra were counted on a 14% coaxial GeLi detector and showed prominent peaks from

uranium activation and fission products, with a "window" where no (or insignificant) peaks were observed between about 75 and 96 keV. This is in part as expected, for natural activity from ^{231}Th (daughter of ^{235}U) is negligible, and would not interfere, and the experiment showed no other potentially interfering peaks. Thus the NAA method appeared feasible as a means of determining ^{230}Th in the presence of natural uranium.

Pure ^{230}Th was obtained and irradiated in the reactor shielded by 0.5 mm of cadmium. The resultant spectrum (Figure 1) showed a pronounced broad peak at 84.6 keV, and secondary peaks at approximately 73.5 and 90 keV. Both the 84 and 90 keV peaks are attributable to ^{231}Th; the other peaks were not identified, but may be attributable to impurities or K x-rays. The presence of ^{231}Th was verified by observing the decay of the peaks; with the exception of the peaks at 68, 73.5 and 145 keV, all peaks decayed with a half-life of approximately 25.5 hours. A combined sample of ^{230}Th and uranium ore was also counted together after irradiation. The results clearly showed ^{231}Th (Figure 2).

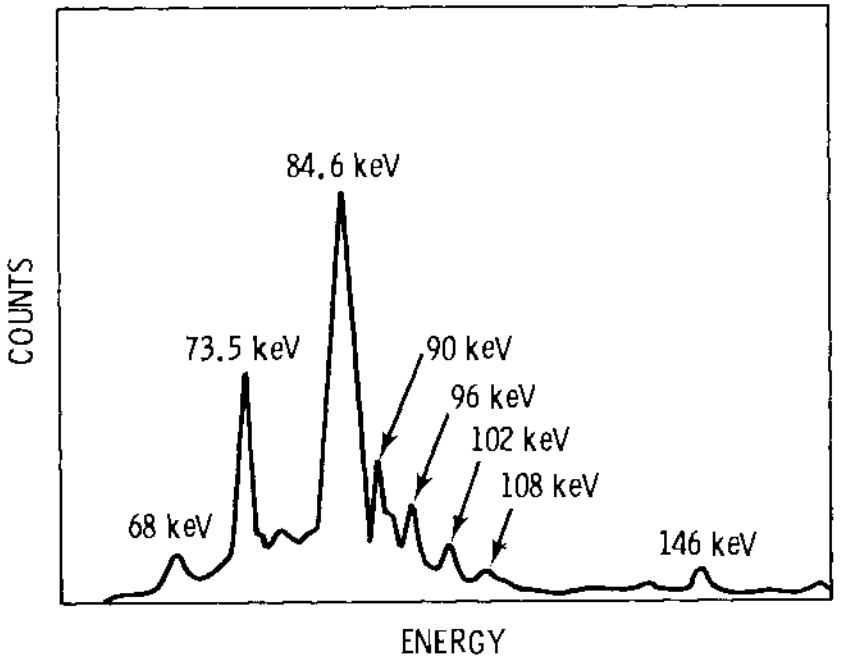

Figure 1. Observed spectrum energy over the range 60 to 150 keV from ^{231}Th obtained by epicadmium neutron activation of ^{230}Th.

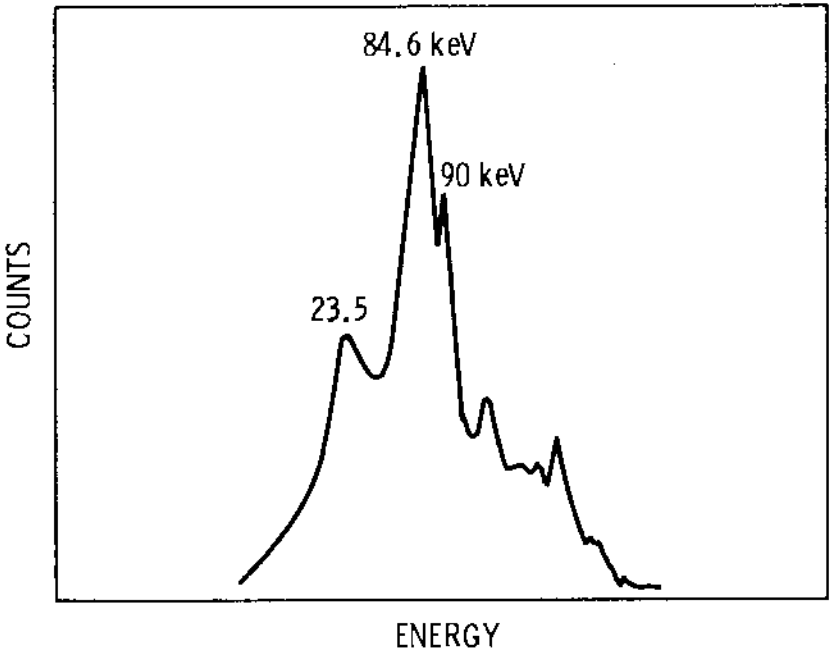

Figure 2. Spectrum in the region of 60-120 keV from mixture of natural uranium and ^{230}Th following epicadmium neutron irradiation. The peaks identified at 84.6 and 90 keV are from ^{231}Th; the 73.5 keV peak is from ^{238}U.

The above preliminary work demonstrates the feasibility of epicadmium NAA as a means of assay of ^{230}Th, and the data suggest a sensitivity of at least 10 ng for a 10 g sample, 100 minute counting time, and epicadmium fluence of 3 x 10^{12}n/cm^2 (5 x 10^{11}n/cm^2-sec for one minute). While interferences from other substances will be minimized through the use of epicadmium neutrons and a brief post-irradiation delay to permit shortlived activation products to decay, sodium, present in large quantities in feces and other biological materials, may produce significant counts in the low energy channels from Compton scattering of the high energy photons associated with the decay of its activation product ^{24}Na. The 15 hour half-life of this nuclide is sufficiently close to that of ^{231}Th to preclude holding the sample for decay. Hence, pre-irradiation sodium removal may be desirable. Further studies along these lines, as well as to improve the method by use of thin intrinsic germanium detectors are now in progress.

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CALIBRATION OF RADIATION PROTECTION INSTRUMENTS AT SSDL LEVEL IN ISRAEL

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INTRODUCTION

The application of ionising radiation in medicine, industry and research in Israel has reached the same level relative to its population as in other industrialized countries. In order to achieve the maximum benefit from the use of ionising radiation, it is essential that dosimeters should be routinely calibrated with respect to the correct dose or dose rate indication. For this purpose, a Secondary Standard Dosimetry Laboratory (SSDL) was established in Israel in 1976. The main tasks of the laboratory have been in the field of therapy level dosimetry, and at present all radiotherapy facilities in Israel have at least one dosemeter calibrated at the laboratory.

In the last years there has been an increased demand for the calibration of protection level instruments. According to unofficial figures, there are in Israel several thousand instruments of this kind. In spite of the fact that the accuracy requirements for these instruments have been relaxed by one order of magnitude compared with therapy level instruments, the need for dose rates ranging from very low to very high values, and other technical problems make calibrations a difficult task.

The present work describes the actions taken at the Israeli SSDL in order to extend its activities down to the low dose region. Furthermore, preliminary results of the calibration of several typical instruments are presented.

FACILITIES

The set-up used for the calibration of protection level dosimeters is esentially the same as that used for therapy level instruments, except for the fact that a distance of 2 meters between the X-ray focus and the ionisation chamber is adopted. In order to allow continuous variation of the high tension and current of the X-ray generator, special modifications have been carried out. After the modifications, the X-ray generator can be operated at currents down to 0.5 mA, the stability of the high

tension being better than 0.5%. In this way, constancy of the output up to ±1% is achieved.

Since the laboratory should be able to calibrate instruments over a wide range of intensities, different series of radiation qualities were adopted (Table 1).

TABLE 1. FILTERED X-RAY RADIATIONS USED AT THE ISRAELI SSDL

Series	Generating Voltage (kVp)	Additional filtration (mm) Pb	Sn	Cu	Al	HVL (mm)
	25			0.2		1.3 Al
Low	50			1.0		5.0 Al
Exposure	75		0.8	0.25	1.0	0.6 Cu
Rate	100		2.5	0.4	1.0	1.4 Cu
(about	125		4.5	0.4		2.1 Cu
50 mR/h)	150	0.9	3.0	2.0	0.8	3.0 Cu
	200	3.0	2.0	0.4	0.8	4.5 Cu
	25	Inherent (about 2 mm Al)				1.0 Al
	40			0.2		2.2 Al
Narrow	60			0.5		5.0 Al
Spectrum	80			2.0	2.0	0.6 Cu
(about	100			5.0	1.6	1.2 Cu
500 mR/h	120		1.0	4.8	0.8	1.8 Cu
	150		2.5	0.4	1.0	2.5 Cu
	200	0.9	3.0	2.0	0.8	4.1 Cu
	250	3.0	2.0	0.4	0.8	5.4 Cu
	60			0.2		3.4 Al
Wide	80			0.5		0.3 Cu
Spectrum	110			2.0	2.0	1.0 Cu
(about	145		0.8	0.25	1.0	1.8 Cu
5000 mR/h)	200		2.5	0.4	1.0	3.3 Cu
	250		4.5	0.4	1.0	4.3 Cu

The qualities adopted are similar to those recommended by ISO (1), except in the 10-50 keV region where filtered radiations are used instead of fluorescent radiations. A detailed discussion about the uncertainties introduced when fluorescent radiations are not used can be found elsewhere (2).

The reference instrument used at the laboratory is a 2550 NPL Protection Level Secondary Standard Dosimeter.

The instrument was originally calibrated at the National Physical Laboratory (UK), but owing to a leak in the chamber it was necessary to replace the rubber ballon, thus invalidating the calibration. The instrument with its new chamber was recalibrated at our laboratory, by comparing it against a 2560 Therapy Level Dosemeter. A high quality 30 c.c. chamber was used as the transfer instrument. It has been estimated that the uncertainties introduced during the transfer process are unlikely to exceed 4% at the 99% confidence level.

Since the use of radioactive sources for checking linearity of the instruments is involved with a considerable cost, it was decided to check this parameter by changing the current of the X-ray machine. For this purpose, a special gold coated transmission monitor chamber was designed by the German factory PTW. This chamber allows the monitoring of beam intensities down to 1 mR/h.

PRELIMINARY RESULTS

The following parameters have been measured during the setting up of the laboratory:

a) HVL
b) Beam homogeneity.
c) Reproducibility of the positioning of the instruments.
d) Reproducibility of the monitor system.
e) Energy dependence of the monitor chambers.
f) Long term reproducibility of the reference instrument.

Parameters of the instrument under test that will be evaluated are as follows:

a) Energy response
b) Linearity
c) Saturation characteristics
d) Directional dependence
e) Temperature dependence
f) Overload characteristics
g) β response
h) Neutron response
i) Transient characteristics

As an example, Figure 1 shows the energy response of some typical instruments used in Israel, while Figure 2 shows the saturation characteristics of a Nuclear Enterprises 30 c.c. chamber. It is evident that owners of radiation protection dosimeters should be aware of these characteristics before choosing the most suitable instrument for a particular application. For instance, the use of an Elscint GSM-1 Geiger Counter might introduce an error of up to 300%, while the use of a Nuclear Enterprises 30 c.c. for the measurement of pulses of radiation (for instance the output of an X-ray machine) will also introduce a significant error.

FUTURE ACTIVITIES OF THE ISSDL

The Soreq branch of the ISSDL is installing a Manganese bath for the absolute calibration of neutron sources. In addition its facilities are scheduled to perform calibrations of radiation protection monitors for neutron and beta radiation. A set of sources produced by Buchler (FRG) and calibrated

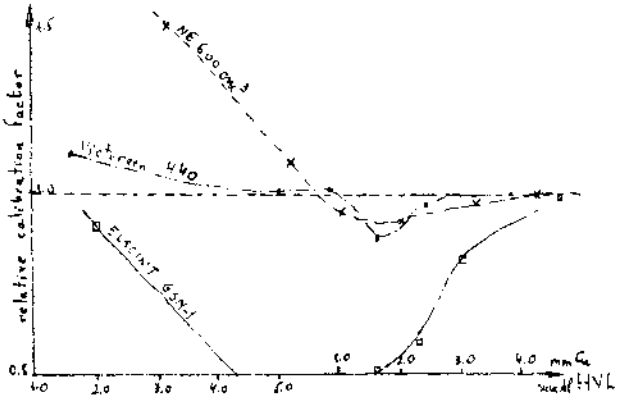

Figure 1. Energy dependence of some typical instruments.

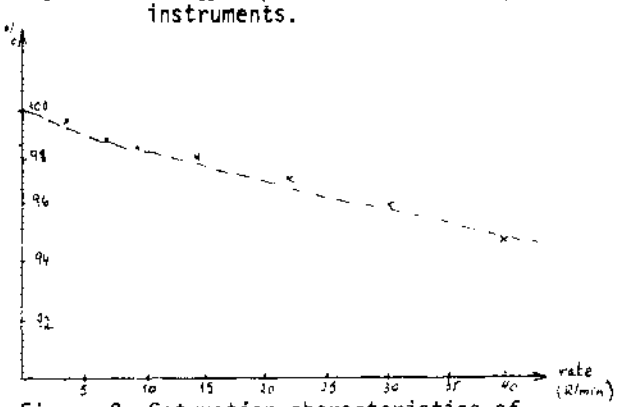

Figure 2. Saturation characteristics of Nuclear Enterprises 30 c.c. chamber (Polarising voltage: 300 V)

at the PTB will serve as the secondary standard for beta radiation.

Absolute calibrations of beta-gamma sources will be performed with a 4π proportional counter and a sodium iodide spectrometer system using beta-gamma coincidence techniques.

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A PASSIVE MONITOR FOR RADON USING ELECTROCHEMICAL TRACK ETCH DETECTOR

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INTRODUCTION

Radon gas and its airbone daughters are a health hazards not only for the workers in uranium mining but also for the population in dwellings. Short-term fluctuations of the radon level outdoor and in houses have been found to be in the order of one magnitude dependent on the emanation rate from the ground, the walls or the floor, the ventilation rate in the room and the meteorological conditions.

In working level meters the decay products of radon are collected by an aerosol filter or an electrostatic field precipitation. For longterm monitoring mainly cellulose nitrate track etch detectors [1] or thermoluminescent detectors [2,3] are applied to registrate α-particles emitted from the filter. In recent studies Makrofol polycarbonate was found to be a promising electrochemical track etch detector (ECED) for the detection of radon daughters [4,5].

For the long-term estimation of the inhalation dose from radon daughters inside buildings a single inexpensive passive radon dosimeter was developed which consists of a Makrofol track etch detector inside a diffusion chamber similar in principle to one described before [6]. It provids a time-integrated indication of the mean dose to the lung with a sensitivity of 130 mrem.

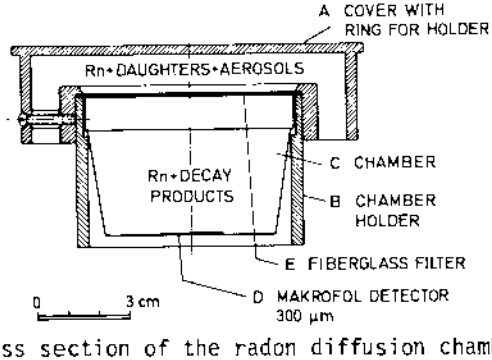

Fig.1 Cross section of the radon diffusion chamber

DOSEMETER DEVICE

As it is shown in Fig.1 the diffusion chamber (C)consists of a plastic cup which is closed at the top by a fiberglass filter (D) through which radon may pass by diffusion while radon daughters and aerosols are retained at the surface of the filter. On the bottom of the cup a Makrofol foil (E) of 300 µm thickness is placed to registrate α-particles reaching the detector with energies between zero

and the maximum value. A special cover (A) placed in front of the filter is an excellent fit for the filter (D) between the chamber holder (B) and the cover (A). The cover avoids any damage on the filter and the deposition of heavy dust particles on its surface.

ETCHING TECHNIQUE

A pre-etching technique immediatly before the ECE is applied which removes a surface layer of the Makrofol foil in order to reduce background tracks and to reveal etchable tracks from high energy α-particles [7]. A solution 4:1 of ethyl alcohol and 6N KOH shows a layer removal rate of 2.34 µm in Makrofol after 1 hour of pre-etching at room temperature. The ECE was performed in a 6N KOH solution containing 20% by volume of alcohol by applying a high voltage at 800 V_{eff} and 2 kHz for 3 hours at room temperature.

After different periods of pre-etching the optimum condition corresponds to a layer removal of 1.17 µm and a background of 37 ± 15 tracks/cm (Fig. 2).

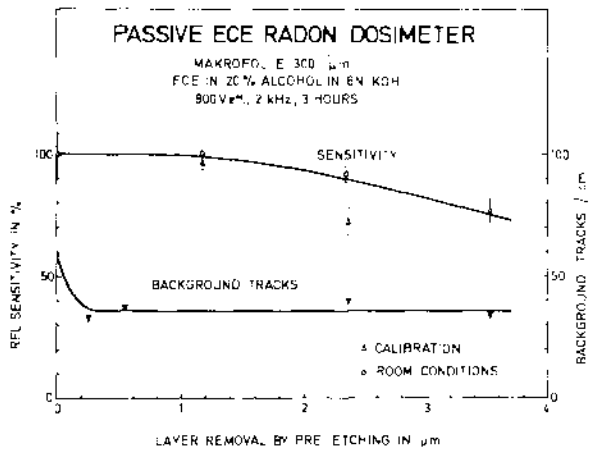

Fig. 2 Rel. sensitivity and background tracks vs. layer removal

ESTIMATION OF RADON EXPOSURE

The exposure of the population in houses is caused by the inhalation of radon daughters as free ions or attached to aerosols in the atmosphere. Inhaled radon daughters result in an inhomogenious irradiation of the various parts of the human respiratory tract primary in the bronchial region which depends on the way of breathing the rate and depth of respiration and the translocation and clearance of the deposited activity.

For the estimation of the radon exposure, the working level WL is defined as the potential alpha energy associated with 100 pCi/l of ^{222}Rn in radioactive equilibrium with its short-lived decay products and can be calculated from the ^{222}Rn concentration C_{Rn} and an equilibrium factor F between Rn and its short-lived daughters. In adequately ventilated rooms a value of F = 0.5 corresponding to a

relative activity concentration ^{222}Rn:RaA:RaB:RaC of 10:9:5:3.5 [8] is a good estimate and agree with short-term experimental indoor results [9]. This value was adopted for our calibration. F varies with the ventilation rate in the room resulting in values of 0.7 for 1 cycle/hour and 0.4 for 2 cycles/hour. For the usual ventilation conditions in uranium mines F values between 0.2 and 0.3 have been found [10].

The passive radon dosimeter has been exposed for 20 hours in a Rn concentration of 4.04 nCi/l measured by means of a surface barrier α-spectrometer [11] resulting in 808 WLh and a corresponding reading of $2.2 \cdot 10^3$ tracks/cm^2 (Fig.3). Additional calibrations have been performed in a closed room with a high radon concentration. The track diameter was found to be in the order of 150 µm.

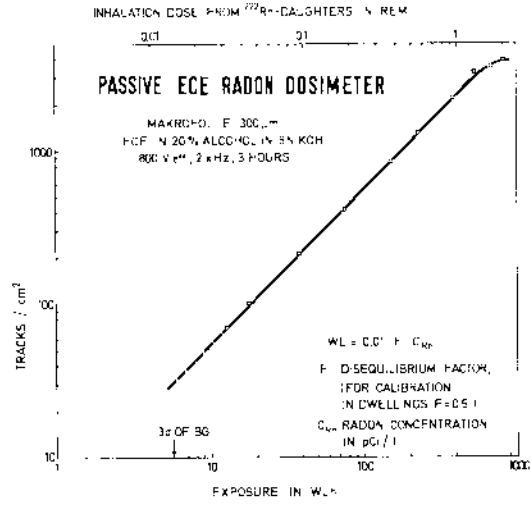

Fig. 3 Track etching detector reading vs. exposure

With regard to the mean dose to the lung UNSCEAR [8] recommand a dose factor 0.2 rad $(WLM)^{-1}$. The working level month (WLM) is defined as he accumulated exposure of 1 WL concentration during a period of 170 hours. Taking into account a quality factor of 20 for α-particles, we applied a conversion factor 4 rem·$(WLM)^{-1}$. This conservativ value agree with new epidemiological data on radiogenic lung cancer resulting in a conversion factor of 6 rem·$(WLM)^{-1}$ with 1.4 rad·$(WLM)^{-1}$ and a rem per rad of about 4 [12].

APPLICATION

The lower detection limit of the radon dosimeter given by the 3σ value of the background tracks was found to be equivalent to 5.6 WLh or a corresponding mean dose in the lung of 130 mrem (Fig. 3). After an exposure period of 3 month (2160 hours) a mean radon concentration of 0.3 pCi/l can be detected.

During storage periods up to 1 month at 50°C no significant fading effect was found for the detection of α-particles [14] which

confirmes earlier fading results for neutron induced recoils.

The passive radon dosimeter discriminates the detection of thoron and its daughters because of the short half-life of thoron compared with the diffusion time necessary to pass the fiberglass filter.

The passive radon dosimeter was applied in a uranium mine as well as for a 3 month monitoring period in houses. Comparisons of the radon dosimeter results with the short-term results of an instant working level meter and a time-dependent decay of radon daughters collected for a period of 3 min on the surface of a filter [13] agree within 20%. In contrast to the radon measurement by means of the passive dosimeter these techniques are based on the measurement of the potential α-energy. Differences in the results are expected from additional thoron contents in the atmosphere and the approximation of the equilibrium factor F. The radon dosimeter described here is applied for a long-term radon exposure study in buildings to estimate the real inhalation dose of the population in Germany.

We wish to thank Dr. Wicke, Justus-Liebig-University Gießen and Mr. Urban for the calibration of our radon dosimeter.

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ATMOSPHERIC DISPERSION STUDY WITH ^{85}Kr AND SF_6 GAS

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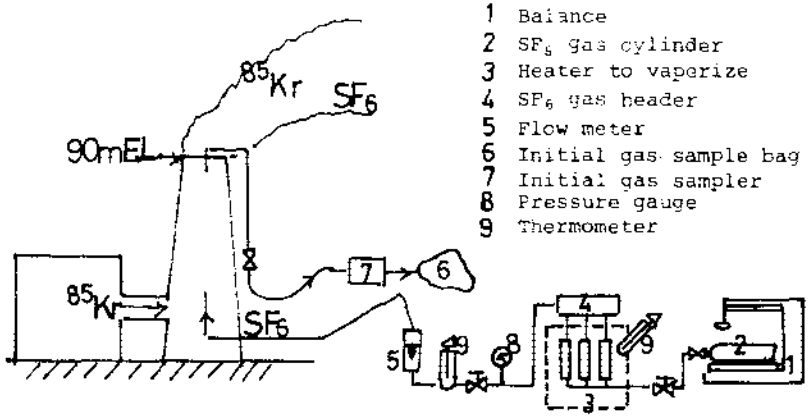
^{85}Kr concentration in air around the reprocessing plant was measured when it was released from the plant stack during the hot-test operation treating PWR spent fuels. Charcoal absorption method was devised for this experiment as a rapid, simple and inexpensive technique for the measurement of ^{85}Kr concentration at about 10^{-5} $\mu Ci/cm^3$. Special air samplers were located at few kilometers distant from the stack and the air was collected for an hour at a flow-rate of 1.5 liters per minute. About thirty air samples were taken and analyzed for ^{85}Kr concentration successfully. The performance of sampling and analytical methods employed was good, although some improvements were found to be necessary. An example is that an air-collection bag should be made of "saran" so that the leakage be minimized. Usually in the atmospheric diffusion study, the sulfer hexafluoride gas (SF_6) is used as an air tracer, and analytical procedure for the gas concentration down to 10^{-3} ppb is well established. In this experiment SF_6 gas was injected into the stack during the discharge of ^{85}Kr and the concentration in sampled air was also determined. The results show a good correlation between ^{85}Kr and SF_6 concentration in sampled air.

METHODS AND RESULTS

PNC reprocessing plant located near the sea-shore of the Pacific-Osean where is about 0.5 km distance from the exhaust point of air tracer. Air tracer gas of SF_6 was released continuously during air sampling duration. Fig. 1 shows air tracer exhaust system. SF_6 concentration was detected by the E. C. D. gaschromatgraph. ^{85}Kr gas was concentrated with cold-charcoal-trap and detected by GM tube detector showing in Fig. 2 and 3. Atmospheric diffusion profile and concentration of air tracer observed on neutral stability condition were good correlation with one calculated by Gaussian plume model equation, but in case of the unstable condition they were not like, as shown in Fig. 4,and relative cross-wind integrated concentration (χcic.U/Q) of air tracer is shown in Fig. 7. As shown in Fig. 5 atmospheric diffusion parameter (σy, σz) based on air tracer concentration are somewhat like on neutral condition pattern in spite of stability class. The relation between dilution factor of ^{85}Kr and SF_6 were good correlation as shown in Fig. 6.

Annual mean concentration with annual weather data was not far different between the value estimated with dilution factor based on this study and calculated with Gaussian plume model equation.



1 Balance
2 SF_6 gas cylinder
3 Heater to vaperize
4 SF_6 gas header
5 Flow meter
6 Initial gas sample bag
7 Initial gas sampler
8 Pressure gauge
9 Thermometer

Fig. 1 Air tracer gas (^{85}Kr and SF_6) exhaust system schematic

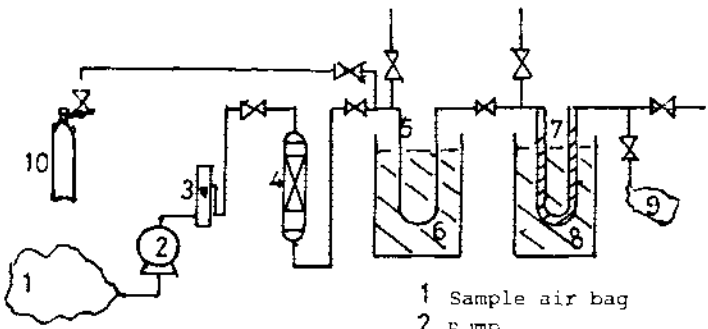
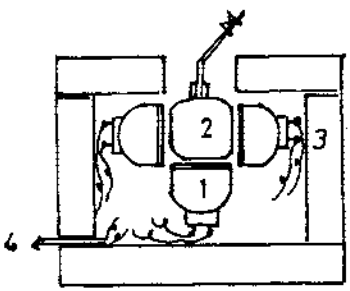


Fig. 2 ^{85}Kr concentration system schematic

1 Sample air bag
2 Pump
3 Flow meter
4 H_2O trap (silica-gel)
5 CO_2 trap
6 Dry ice + C_2H_5OH
7 Krypton trap
8 Liquid nitrogen
9 Concentrate sample bag
10 Helium gas cylinder



1 GM tube (5 detectors)
2 Concentrate sample bag
3 Lead brick shild
4 To counter

Fig. 3 ^{85}Kr detection system schematic

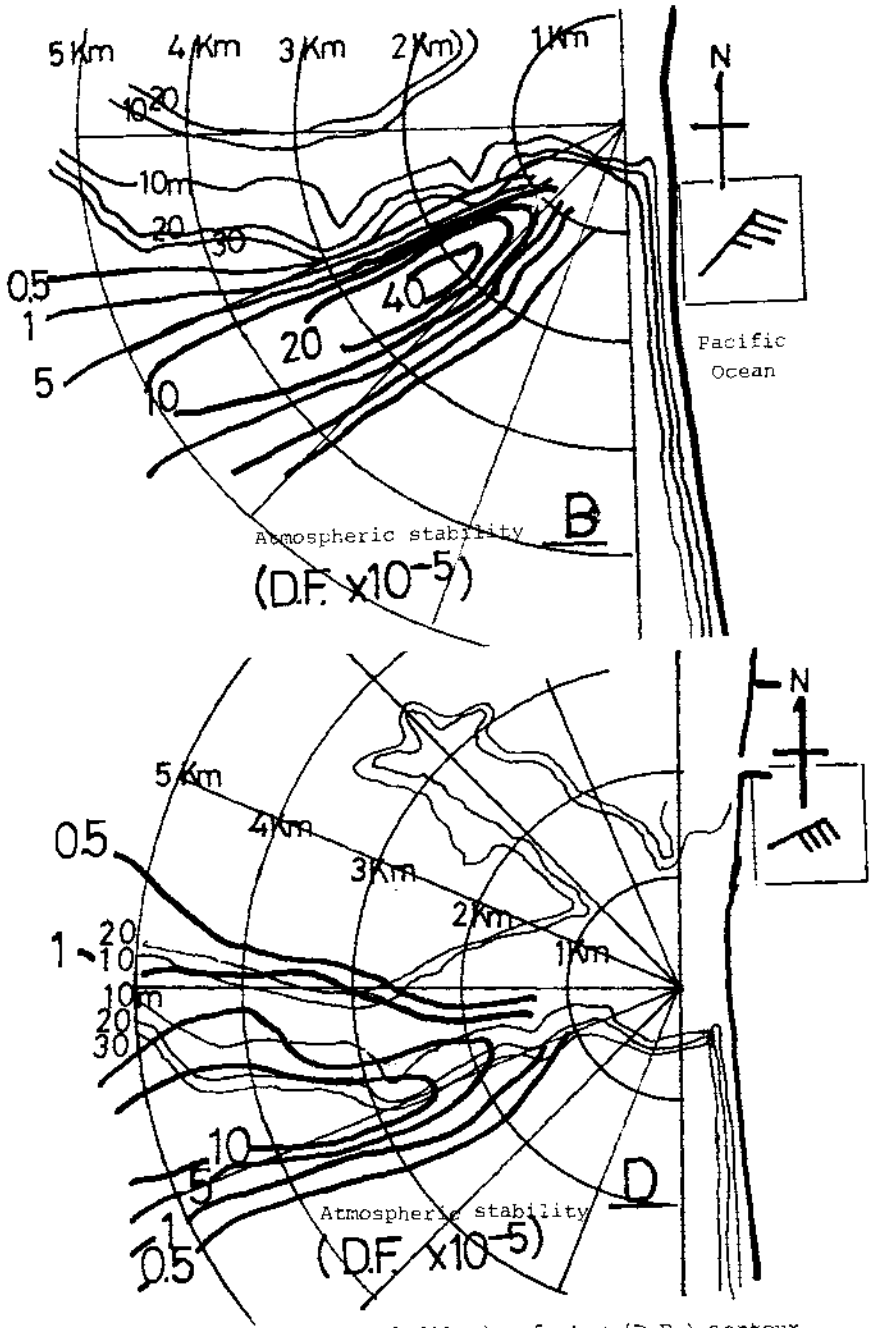


Fig. 4 Horizontal dilution factor (D.F.) contour of the air tracer on the ground surface (at 1m EL.)

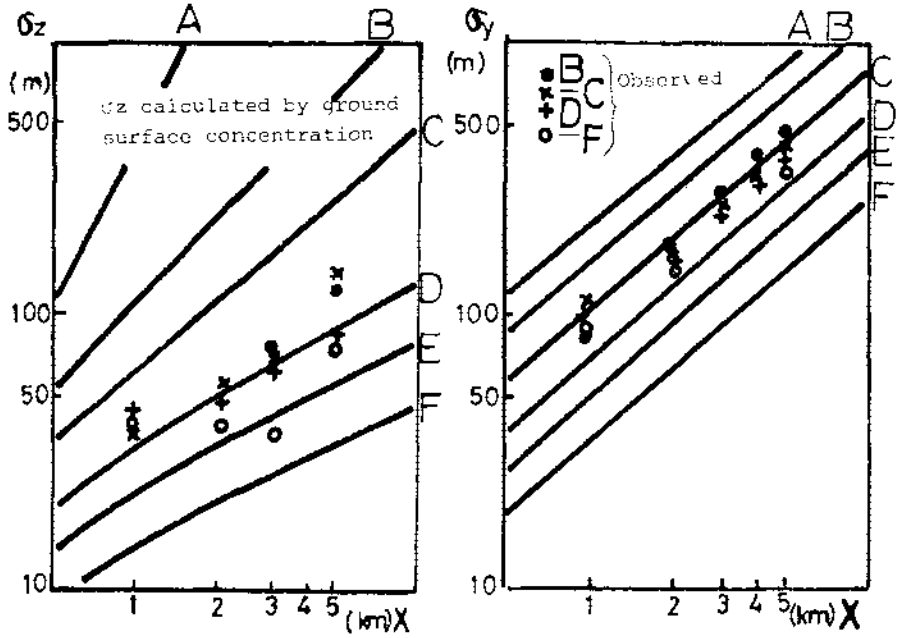


Fig. 5 Atmospheric diffusion parameter (σy, σz) based on observed airtracer concentration profile.

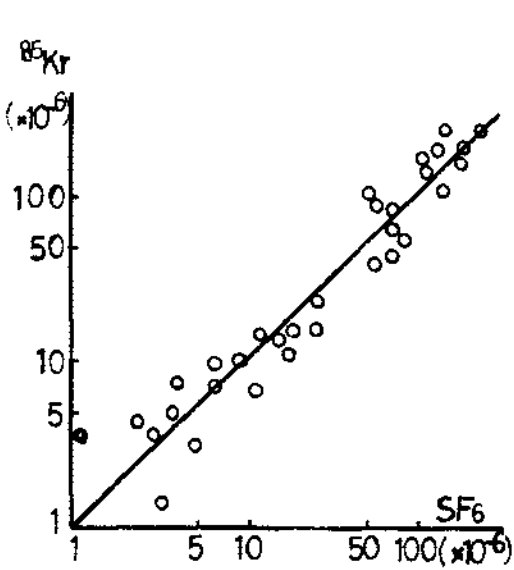


Fig. 6 Relation of dilution factor of ^{85}Kr and SF_6

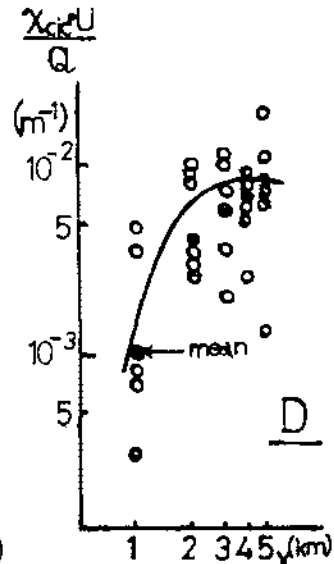


Fig. 7 Crosswind integrated concentration for neutral condition

METHODS OF I-129 ANALYSIS FOR ENVIRONMENTAL MONITORING

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Among the radioiodine isotopes discharged from nuclear facilities I-129 has the longest half-life of 1.7×10^7 years and is accumurated in the environment for a long time period, therefore, it is one of the most important nuclides in the environmental monitoring arround a nuclear fuel reprocessing plant.

Low level contamination of enviornmental samples with I-129 may cause considerably high thyroid dose to the population. For instance, only a tenth pico-curies of I-129 per liter of fresh milk may give one millirem of thyroid dose. Accordingly, in considering the I-129 discharged to the atmosphere or the marine environment, it is important to establish a methodology for evaluating the environmental impacts caused by the long-term accumulation and to develop the measuring techniques of the environmental samples having very low radioactivity.

This paper presents the methods of the analysis of low-level I-129 in the environmental samples such as milk, vegetations, sea weeds and soils.

PROCEDURES

Leafy vegetables, sea weeds and soils are dried with a low-temperature oven and are ground to powders. Milk is pulverized by freeze drying method.

The iodine is separated from the dried or pulverized samples by ignition at high temperature (about 1000°C) in a quartz combustion apparatus with a stream of oxygen. Figure 1 shows the apparatus for ignition used in the experiments. Furnace 1 moves from the edge of the combustion tube to a sample slowly and the sample is heated gradually, finally ignited at 1000°C. The off-gas from the sample is burned completely while passing through Furnace 2. The iodine carried with the off-gases is trapped on a small bed of activated-charcoal. The iodine is then recovered from the charcoal to a dryice cooled quartz tube by heating in vacuum. The cooled end of the quartz tube is sealed off to make a irradiation ampule.

The quartz ampule is irradiated in a reactor for several ten minutes at thermal neutron flux of 10^{13}n/cm^2·sec.

After the irradiation the iodine is purified through the solvent extraction method using carbon tetrachloride. Iodine is finally precipitated as AgI and counted with a Ge(Li) detector. Each activities of I-126, I-128 and I-130 are calculated from the gamma-ray spectra. The chemical yield of this method are calculated by counting I-126 which has been added to the sample prior to the ignition as an yield tracer.

RESULTS

Several samples were analized on I-129 by the method mentioned above and no I-129 concentration higher than detection limits were found. The results of analysis for typical food samples collected near the fuel reprocessing plant of Tokai Works are given in Table 1. The detection limit of I-129 by this method is about 10^{-2} pCi for a 10 g dry sample. Stable iodine I-127 is simultaneously determined and atom ratio of $^{129}I/^{127}I$ are calculated in order to evaluate thyroid dose by the specific activity method and long term environmental impacts by I-129 discharged from nuclear facilities.

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Table 1 Results of Analysis

Sample	I-127 * (μg/g)	I-129 (10^{-3}pCi/g)	$^{129}I/^{127}I$ Atom Ratio (10^{-7})
Seaweed	210 ± 11	< 5.1	< 1.5
Cabbage	4.2 ± 1.3	< 0.8	< 15
Rice	34 ± 1.2	< 2.1	< 3.6

* Determined by counting on I-128

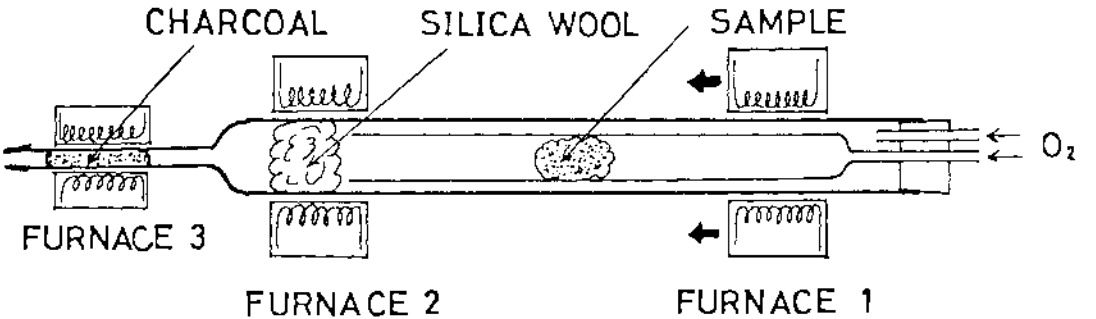

Figure 1 Diagram of Sample Combustion Apparatus

A NEW TECHNIQUE FOR NEUTRON MONITORING IN STRAY RADIATION FIELDS

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INTRODUCTION

At reactors, accelerators and therapy facilities including linear accelerators there is the need to monitor and interprete low level stray radiation fields. The techniques applied in neutron monitoring today is based mainly on the measurement of the dose equivalent by means of rem counters. The response function of the different rem counter types, however, has been recently found to overestimate intermediate neutrons by a factor 6 (Leak type) or factor 8 (Anderson-Braun type) and to underestimate thermal neutrons up to a factor 0.34 (polyethylene sphere 30 cm diam) [4]. Both the Bonner multi spheres and spectrometers, on the other hand, are sophisticated techniques not applicable for routine work.

The new approach in neutron monitoring described here is based on a single sphere technique and passive thermoluminescence detectors which allows to measure

- the total dose equivalent of neutrons and gamma rays,
- the dose equivalent components of thermal neutrons < 0.4 eV, epithermal neutrons between 0,4 eV and 10 keV and fast neutrons above 10 keV,
- the effective neutron energy E_{eff} of the fast neutron component in unidirectional or isotropic stray radiation fields.

MEASUREMENT TECHNIQUE

The single sphere technique applied for the measurement of the dose equivalent and for the interpretation of the neutron spectrum makes use of a passive rem counter (polyethylene sphere of 30 cm diam) and a TLD600/TLD700 detector in the center (see Fig. 1). The rem counter sphere serves also as a phantom for two albedo dosimeters.

The Karlsruhe albedo dosimeter [1] designed as an analyser detector type contains three TLD600/TLD700 detector pairs inside a boron-loaded plastic capsule allowing a separate indication of albedo neutrons (detector i), incident thermal neutrons from the field (detector a) and epithermal neutrons (detector m).

By means of TLD600/TLD700 detector pairs, thermal neutrons are measured via the reaction $^{6}Li(n,\alpha)^{3}H$ and the neutron dose reading is given by the difference of TLD600 and TLD700. The TL detectors are calibrated in a ^{137}Cs gamma field which results in a gamma equivalent neutron dose reading presented here in the unit R.

In stray radiation fields the neutron detection is directionally independent for the rem counter sphere and in a first approximation also for the albedo dosimeter system if the corresponding readings in the opposite position at the phantom surface are summed up.

The response R of the albedo dosimeter i found by calculation [2] and calibration with monoenergetic neutrons [3] is presented in Fig. 1 as a function of neutron energy. The response function of the passive

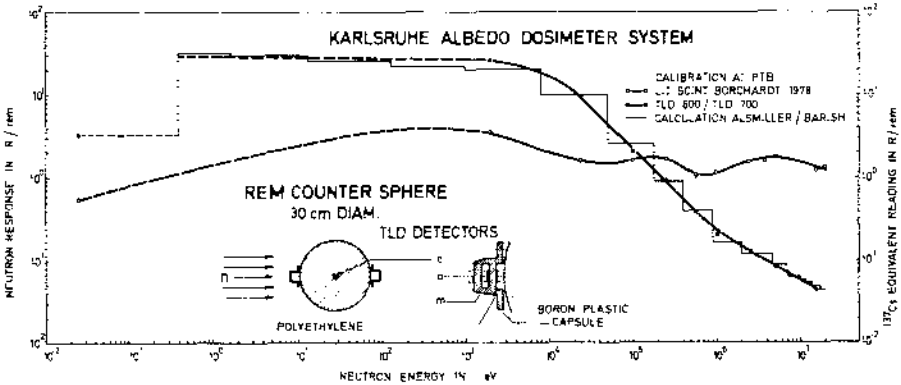

Fig. 1: Response of the Karlsruhe Single Sphere Albedo Dosimeter

TLD rem counter is expected to be equal to that of the rem counter with the LiI scintillation detector [4] and was related to a response of 1.6 R/rem found with Am-Be neutrons. The detector c in the sphere represents the true neutron dose equivalent H_n in a good approximation and the dose reading ratio $\alpha(i)/H_n$ the response R(i) of the albedo dosimeter. The high change in response in the energy range above 10 keV compared to the flat response of the rem counter is the basis for an estimation of an effective neutron energy in stray radiation fields.

INTERPRETATION OF THE STRAY RADIATION FIELD

In practice the effective response $R_{eff}(i)$ of the albedo dosimeter may vary by one order of magnitude around one facility mainly caused by local changes of the thermal neutron fluence and/or the moderation of the fast neutrons.

For the interpretation of the neutron spectrum a computer program is used taking into account the response function of the rem counter (detector c) and of the detectors i, a, m in the Karlsruhe albedo dosimeter. In a neutron stray radiation field the neutron dose reading of the detectors in the albedo dosimeter can be interpreted on the basis of three energy components by the following response matrix

$$\alpha(a) = R_{th}(a) \cdot H_{th} + R_e(a) \cdot H_e + R_f(a) \cdot H_s$$
$$\alpha(m) = R_{th}(m) \cdot H_{th} + R_e(m) \cdot H_e + R_f(m) \cdot H_s$$
$$\alpha(i) = R_{th}(i) \cdot H_{th} + R_e(i) \cdot H_e + R_f(i) \cdot H_s$$

$\alpha(a)$, $\alpha(m)$, $\alpha(i)$ — neutron dose reading in the ^{137}Cs equivalent unit R for the detectors a, m, i measured in the stray radiation field

H_{th}, H_e, H_f — neutron dose equivalent fraction in the unit rem due to thermal, epithermal and fast neutrons with the total neutron dose equivalent $H_n = H_{th} + H_e + H_f$

$R_{th}(k)$, $R_e(k)$, $R_f(k)$ — neutron response in R/rem for thermal, epithermal and fast neutrons for the detector k = i, a or m.

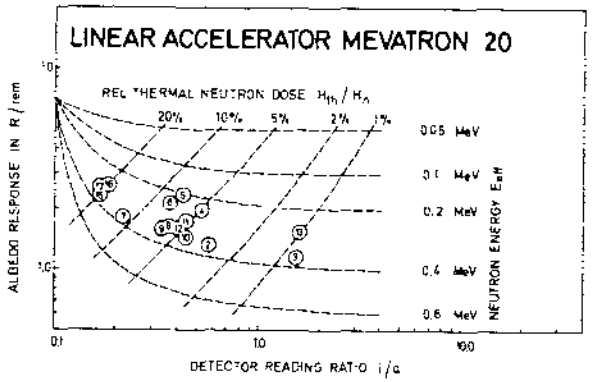

Fig. 2: Albedo dosimeter calibration at the Linac, Karlsruhe Vincentius Hospital

Taking into account the calibration data for $R_{th}(k)$, $R_e(k)$ and the relation $R_f(a) = R_f(m) = \varepsilon \cdot R_f(i)$ the response matrix can be solved resulting in analytical indications of the dose equivalent data H_{th}, H_e, H_f, H_n. For the estimation of the energy dependent response $R_f(i)$ the term H_f is calculated by $H_f = H_n' - A \cdot H_{th} - H_e$ taking into account the underestimation (factor A) of the rem counter reading H_n' for thermal neutrons.

The estimation of E_{eff} is based on the correlation between neutron energy and the albedo response $R_f(i)$ (see Fig. 1). In contrast to the calibration results for monoenergetic neutrons presented in Fig. 1 the applied program is fitted for neutron stray radiation fields assuming a homogeneous energy distribution for fast neutrons (Gaussian normal distribution).

In addition the statistical errors of all computed results are calculated based on the statistical error of each dose reading given by previous data of a reader test [5] (standard deviation vs. exposure) and caused also by the discrimination of the gamma dose reading in the detectors i, a, m and c [6].

APPLICATION IN STRAY RADIATION FIELDS

For personnel monitoring by means of albedo dosimeters an extended field calibration with the single sphere technique is applied at various locations in the stray radiation field of each neutron facility [7-9]. The application in personnel monitoring is based on individual correction factors for energy dependence based on the correlation between the dose reading ratio $\alpha(i)/\alpha(a)$ and the experimental response $R_{eff}(i)$ found by field calibrations before.

In the neutron/gamma stray radiation field of a 20 MV electron linear accelerator, for instance, the albedo response varies between 1 and 2.8 R/rem (Fig. 2). The value for H_{th}/H_n increases from 1% in 1 m source distance up to 20% at the shielding's entrance. The neutron field interpretation results in E_{eff} values between 0.2 and 0.4 MeV in agreement to results found at a similar accelerator by means of activation detectors and a computer unfolding calculation [10].

The single sphere albedo method is applied as a standard technique to interprete low level stray radiation fields [11]. Some experimental results are presented in Table 1. First applications at power reactor

Table 1: Interpretation of neutron stray radiation fields by means of the single sphere albedo technique

Facility		$\dot{H}_n$ mrem/h	H_n/H_γ	Rel. Neutron Dose in % H_{th}/H_n	H_e/H_n	H_f/H_n	E_{eff} keV	$R_{eff}(i)$ R/rem
^{252}Cf in air	2m	306	15.7	[illegible]	0.0	99.1	1300	0.31
20 MV Lin. Electr.	1m	13970	0.99	1.1	1.2	97.7	255	2.1
Accelerator	5m	1496	0.70	8.5	2.3	89.1	260	2.7
Mevatron 20	entrance	48	3.21	30.4	4.7	64.9	260	3.6
Linac St. 75-20	1m	64410	3.40	1.3	0.6	98.1	345	1.7
	entrance	613	4.71	13.5	3.7	82.8	240	3.2
	behind shield	13	1.55	37.3	5.6	56.6	120	5.1
14 MeV KARIN	1m	4360	11.8	1.9	0.5	97.7	1140	0.57
Therapy Fac.	9m	430	5.8	4.5	0.8	94.7	830	1.0
Compact Cyclotron	1m	500	20.2	2.2	0.4	97.4	1220	0.53
Kl. DKFZ	6m	146	13.6	5.0	1.0	94.0	720	1.2
	15m	138	5.1	14.6	2.2	83.3	400	2.7
	15m	16	1.7	43.9	4.9	51.2	410	3.5
Oak Ridge HPRR	Bare	5575	17.1	0.3	0.0	99.7	880	0.53
	Lucite	4715	12.2	4.3	0.4	95.3	650	0.80
	Concrete	4281	10.0	3.1	0.7	95.2	590	1.2
Jülich FRJ-1	Beam	62	7.33	3.0	1.2	95.5	235	2.5
Reactor	3m	8.7	4.08	20.6	5.1	74.3	140	4.6
GKN Power Reactor	in reactor cavity	5423	4.7	14.0	5.9	80.1	205	4.1
	23m	11	0.67	19.2	5.0	75.9	250	4.0
	64m	2.4	0.38	20.7	5.5	73.8	215	4.0
	entrance to	0.06	0.15	70.7	2.8	26.5	200	3.8
	sump	2.6	0.02	52.9	4.3	42.9	240	3.9
Kahl Exper. Reactor	containment	0.5	3.26	36.4	8.1	55.4	174	3.7
	steam heat exchanger	2.4	45.2	13.6	5.3	81.0	216	3.9
	valve in steam pipe	14.6	11.9	14.4	5.0	80.6	173	4.7

sites show that the local neutron spectrum varies only to a small extent for fast and epithermal neutrons but H_{th}/H_n may change from 5% to 70%.

The specific properties of the Karlsruhe albedo dosimeter was found in an effective response which is equal for thermal neutrons and neutrons with E_{eff} between 100 and 200 keV, therefore, at reactor sites only a small change of the dosimeter response $R_{eff}(i)$ is expected.

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MINIDOSIMETRY OF ALPHA-RADIATION FROM 239-Pu IN THE SKELETON

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Plutonium which enters the bloodstream (via one of the main intake routes) deposits primarily in the skeleton. The distribution pattern is largely nonuniform owing to the fact that this radionuclide belongs to the class of the so-called surface seekers which concentrate on all periosteal, endosteal and trabecular bone surfaces (6). Radiation doses derived from assays of whole bones assume a uniform distribution and therefore tend to underestimate local doses to the cell populations at risk. For osteosarcoma induction the endangered cell populations are believed to be the osteogenic cells lying close to bone surfaces (1). To obtain a realistic correlation between the radiation doses delivered at a cellular level and the observed pathological effects the determination of local dose rates on a microscopic scale is essential. Further, this information can be linked to existing quantitative histological data to provide estimates of hit frequencies for cellular targets and of the probability of malignant transformation.

EXPERIMENTAL PROCEDURE

One year old female rats were injected intravenously with 18 kBqkg^{-1} 239-Pu-citrate and killed at various times after injection. Lumbar vertebral bodies from all animals were embedded in Methylacrylate and cut, on a sawing-microtome (Leitz, Germany) with a center-hole diamond blade, into sections of 150 µm thickness. The calcified tissue in these bone sections was stained selectively with Alizarin-red (30 min in 1 % solution).

After staining the sections were mounted in contact with cellulose nitrate foils (LR115, Kodak Pathé, France) which served as radiation detectors. To ensure alignment between detector surface and bone structure, landmarks (100 µm dia.) were drilled into the detector-section sandwich. After about 9 months exposure at -38°C the detectors were removed from the bone samples and etched in 2.5 N NaOH for 1 hr. Then both the detector and the bone sample were scanned with a microscope photometer using 20x20 µm scan fields (MPV II, Leitz, Germany) controlled by a PDP 8/E minicomputer system (DEC, USA). Thereby quantitation of the track density distribution and the bone structure was achieved in the form of light absorption patterns.

Using a series of FORTRAN programs the two files containing the raw scanning data were processed to generate a digital image of the bone structure, to calculate background levels on the autoradiographs, erase artifacts and, finally, to combine the two files into one.These processed files then constituted the input for an evaluation program that calculated mean dose rates in specific anatomical regions, dose rate distributions, the variation of these parameters, the mean burial depth etc.. Fig. 1 shows a computer generated image of a piece of

trabecular bone with its associated light absorption pattern caused by alpha-particle tracks. Images of this kind are derived from the processed files.

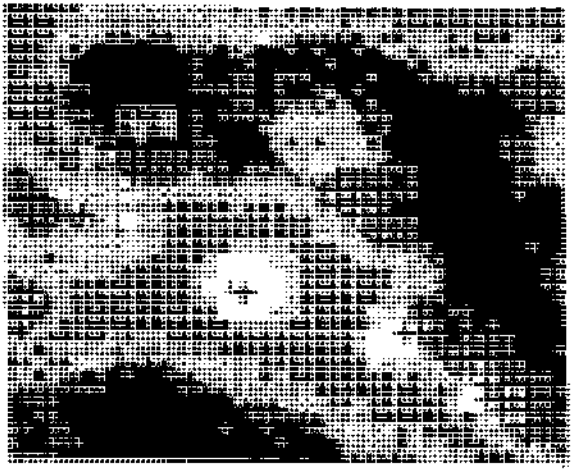

Figure 1. Computer generated display of bone structure (grid) with associated the track distribution expressed as the percentage of light absorbed by the tracks. blank: 0-5 %, x: 5-10 %, ◇: 10-15 %, ◆: 15-20 %, ■: > 20 %.

RESULTS

Radiation dose rates were calculated according to a formula derived by Mays (4) under the assumption that surface deposits are infinitely thin and planar. At sites where no 'hot line' could be detected near a bone surface,the distribution of radioactivity was assumed to be locally uniform (5). Fig. 2 displays the frequency distribution of calculated dose rates in three adjacent bands of 10 μm width each at 28 days after injection. These distribution curves represent the composite variation of activity concentrations and burial depths of 239-Pu deposits and the geometrical locations within the bands.

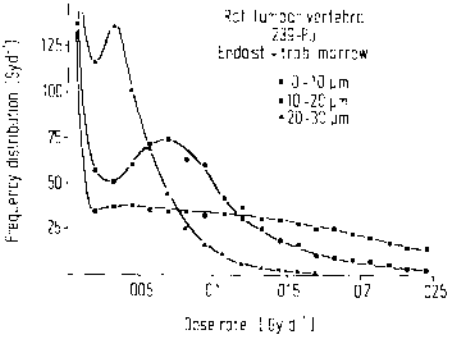

Figure 2. Dose rate distributions in three 10 μm wide bands 28 days after injection of 18 kBqkg^{-1} 239-Pu. The position of the bands relative to the bone surfaces is indicated on the graph.

In the 0-10 μm band 17 % of all sites have a dose rate below 1.25 x 10^{-3} Gy d^{-1}. The corresponding figures for the 10-20 μm and 20-30 μm

bands are 19 % and 30 %, respectively. In a considerable part of the 0-10 μm band dose rates above .025 Gy d^{-1} can be found, whereas in the other two bands such dose rates are very infrequent. Comparison with results from later times after injection reveals that the distributions in Fig. 2 remain fairly constant in shape during the time of observation, with a slight increase from a section average of .0062 Gy d^{-1} (28 days) to .007 Gy d^{-1} (168 days). This indicates a low remodelling activity in the lumbar vertebra, which is expected in adult females. The increase could be attributed to recirculating 239-Pu released from other parts of the skeleton. A spatial dose rate spectrum at 14 different locations is shown in Fig. 3 for early and late times after injection. Initially the mean periosteal dose rates are about 4 times lower than the endosteal and trabecular ones. This relationship however changes in time as the dose rates in periosteal marrow are increasing much more rapidly than those on the endosteal and trabecular surfaces. The enhancement ratio shown in Fig. 3 also indicates that the dose rates in deep bone and deep marrow rise nearly 3-fold and 2-fold, respectively.

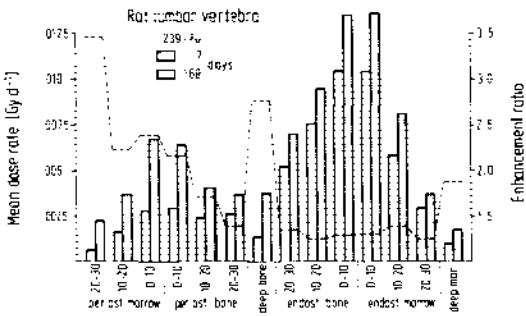

Figure 3. Mean dose rates in three 10 μm wide bands on both sides of periosteal and endosteal + trabecular surfaces and in deep marrow and deep bone. The dashed line represents the enhancement ratio of the values at 168 days after injection relative to 7 days.

The meaning of the dose rate distributions becomes more obvious if they are considered in terms of event frequencies at a cellular level. Specifically, if one considers the cell nucleus of proliferating osteogenic cells close to bone surfaces as the sensitive target and that a certain number of hits by alpha-particles are required to produce harmful effects like cell killing or non-lethal transformation to a malignant state, it can be shown (5) that a conversion from dose rate distribution to hit probabilities can be readily carried out provided the distribution functions are nearly constant in time. If a cell nucleus is approximated by a sphere of 5 μm diameter (2) one has for the fraction of cells f_n receiving n hits up to time t after injection

$$f_n = \frac{(ct)^n}{n!} \int \dot{D}^n \exp(-c\dot{D}t)\delta(\dot{D})d\dot{D}$$

where $\delta(\dot{D})$ is the distribution function of dose rates $\dot{D}$ (5) (c= .882 Gy^{-1}). Curves calculated according to the above expression are shown in Fig. 4 for the 0-10 μm band at periosteal and endosteal + trabecular surfaces. A considerable fraction of the targets receive one and two hits at the interior surfaces, less in the periosteal region.

A large fraction of those nuclei receiving one or more hits are likely to be killed.

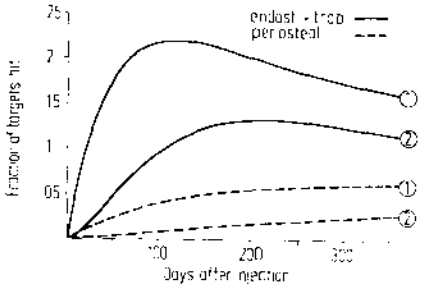

Figure 4. Fraction of spherical targets (5 μm dia.) uniformly distributed throughout a 10 μm band in marrow adjacent to bone surfaces that are hit until time t. Numbers specify the number of hits.

DISCUSSION

The extraction of results from the photometric scan data involves some approximations. For instance the problem of oblique surfaces was not taken into account. In addition to the results shown above information about the remodelling status can be obtained from the distribution of burial depths and its change in time. The linking of minidosimetric results to quantitative histology requires extension of present knowledge about cellular parameters like size and shape of sensitive targets, the position of these targets relative to bone surfaces (3) and the relevant time t in the above expression. With regard to malignant transformation the calculated 2-hit curves may be interpreted as indicating the size of the pool of potentially transformed cells. Moreover the notion of hit frequency perhaps provides a better means of scaling risks down to lower and more realistic skeletal burdens of 239-Pu than the concept of accumulated radiation doses.

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DECONTAMINATION AND MODIFICATION OF LIQUID SCINTILLATORS

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INTRODUCTION

Liquid scintillators (LS) are effectively employed for evaluation of soft beta (H-3, C-14, P-32, I-125) and alpha emitting nuclides. (1) The monitoring of environmental samples at the source of release of tritium (Atomic Power Reactors and nuclear installations) is done very frequently. This involves consumption of large amount of LS and hence generation of active waste. The disposal of liquid waste poses a problem. It would be doubly advantageous to decontaminate the used LS and recycle it.

This paper discusses the new techniques of decontaminating and recycling the used active LS. A modification of hydrophobic scintillator for use with aqueous samples is also described. Both aliphatic and aromatic LS are effectively decontaminated. 1, 4 Dioxane based LS is decontaminated by extraction with NaOH. Single extraction gives a decontamination factor (DF) of about 90% and thus 3-4 extractions decontaminate the LS to background level (2). Aromatic LS Tritol which is a cocktail of 1:2 Triton x-100 and toluene scintillator (3) has also been similarly decontaminated.

EXPERIMENTAL

Aromatic LS (Tritol)

It is spiked with tritiated water and counting efficiency is determined by liquid scintillation spectrometer (LSS 3255). The counted LS is transferred to separating funnel and shaken with solid NaOH for about 20-30 minutes and kept for settling for about two hours. The two phases separate out. The concentration of NaOH should be about 15% in the aqueous phase. The bottom aqueous layer contaminating tritium is collected. The top organic phase which is intact LS is transferred to vails. The orginal amount of distilled water is added (to maintain constant water activity. This process is repeated 3-4 times depending upon initial level of activity.

Hydrophobic LS

The modification of toluene scintillator which does not hold water is attempted for solubilizing aqueous samples. Toluene alcohol solution of varying concentrations from 75% to 30% are prepared prior to addition of Naphthalene 100 gm/l, PPO 7.0 gm/l and POPOP 0.3 gm/l. The counting efficiency and water holding capacity (WHC-maximum percent of water added to LS without phase separation) are evaluated for each ratio of toluene alcohol using LSS 3255. The counted LS is transferred to separating funnel and water to LS ratio ie increased to 1:2 by addition of water. The mixture is shaken vigorously and kept

over-night for settling. The bottom aqueous layer containing tritium is collected. The top organic layer is taken out and made up to original volume with ethyl alcohol as major portion of the alcohol goes along with water. The decontaminated LS is again examined for remaining activity after adding initial amount of water. The counting efficiencies and fluorescence characteristics of recovered and fresh LS are compared.

RESULTS

Atomatic LS

It is observed that about 90% of the spiked activity is transferred with organic phase after first extraction. Figure 1 shows the degree of decontamination of Tritol with number of extractions. Three to four such extractions make the LS compeltely decontaminated.

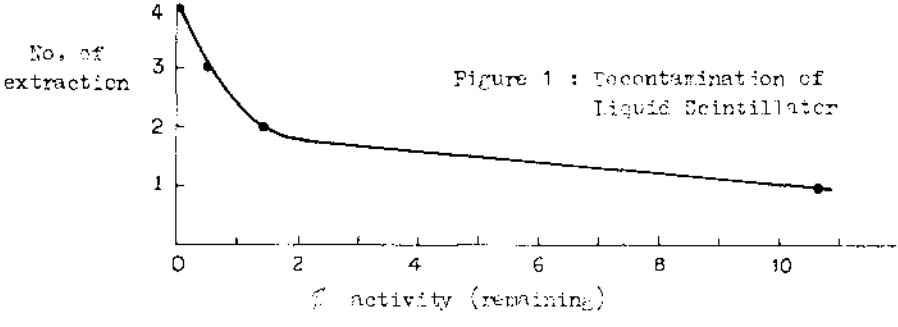


Figure 1 : Decontamination of Liquid Scintillator

The recovered LS is subjected to investigations for reuse. The counting efficiencies of fresh and recovered LS are compared by spiking these with equal amounts of tritiated water. Table 1 shows efficiencies of fresh and recovered LS.

TABLE I Comparison of Scintillators

Fresh Scintillator			Recovered Scintillator		
Bkg. cpm	Activity cpm	Efficiency percent	Bkg. cpm	Activity cpm	Efficiency percent
13.6	2811.0	25.2	21.0	2626.0	23.4
13.9	2847.0	25.5	20.3	2513.0	22.4
16.2	2623.0	23.5	22.2	2201.0	19.6

The wholesomeness of recovered LS and its suitability is examined by fluorescence characteristics also. It is observed that fluorescence characteristics of both fresh and recovered LS are identical except some loss of fluorescence yield in the case of recovered one. This explains the loss in counting efficiency.

Hydrophobic LS

The water holding capacity of modified toluene/alcohol LS decreases with increasing concentration of toluene in LS. Figure 2 shows linear relationship between water holding capacity and toluene concentration. WHC increases from 3% to 13% with decrease in toluene concentration from 75% to 30%.

Fig. 2 Water Holding Capacity

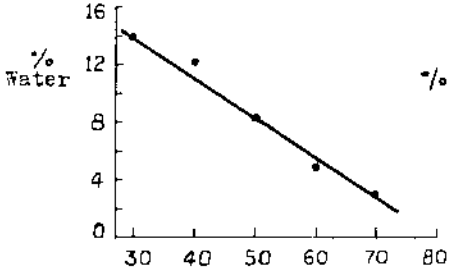
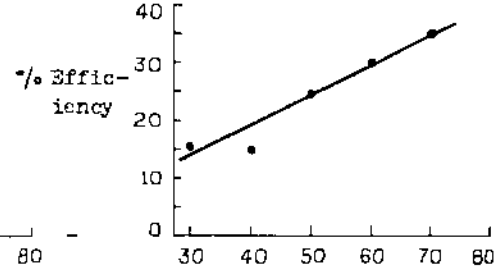


Fig. 3 Efficiency of Scintillator



% Toluene in Scintillator

The counting efficiencies for various toluene/alcohol ratios are determined and shown in figure 3. The efficiency increases with increasing concentration of toluene in scintillator at their maximum WHC. The efficiency increases from 16% to 35% with increase in toluene concentration from 30% to 75%.

The counting efficiency **increases with % toluene in LS while WHC decreases.**. The optimum working range is selected with highest figure of merit (4) which takes into account background, efficiency and volume of the sample. FM is found to be maximum around 50% toluene concentration.

The decontamination of used modified LS is acheived by single washing with excess amount of water. After proper dilution the background is checked prior to spiking. It is observed that the recovered LS has almost similar background as that of fresh one. Table II shows the relative counting efficiencies of 50% toluene/ alcohol LS before and after decontamination. The counting efficiency of decontaminated LS is about 92% of the fresh LS. The quality of recovered modified LS is also compared on the basis of fluorescence characteristics. The spectral characteristics of both the LS are identical except slight loss in fluorescence yield in case of recovered one.

TABLE II Decontamination and Recycling

Fresh Scintillator			Recovered Scintillator		
Bkg. cpm	Activity cpm	Efficiency percent	Bkg. cpm	Activity cpm	Efficiency percent
51.6	1283.0	23.8	59.6	1191.0	21.9
54.7	1265.0	23.4	50.2	1191.0	21.8
50.8	1298.0	24.1	62.0	1209.0	22.2
56.0	1320.0	24.4	47.0	1242.0	23.1

DISCUSSION

It is, therefore, possible to recycle the same LS after decontamination. It will not only save expenditure on LS but solve the disposal problem also, as the activity is contained in aqueous phase with reduced volume. Gaylord (5) has suggested the disposal of used LS vials into sea after crushing them and filling LS in the vessels. This involves further expenditure on disposal besides loosing the liquid scintillation counting waste. Claycamp (6) et al have recently recommended the distillation of used LS to recover toluene for commercial use. They have shown that the energy input for distillation is much less than required to synthesise same amount of toluene from raw materials. However, this will involve the disposal of concentrate active waste and also impart some activity to the recovered toluene.

Our procedure on the other hand requires very little energy input to shake the mixture. The volume of active waste generated ranges from about 12% - 47% depending upon the initial level of activity present in LS. Thus all the three systems, Aliphatic LS, Aromatic LS and Hydrophobic LS can be decontaminated effectively and used repeatedly.

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TRITIUM - IS IT UNDERESTIMATED?

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INTRODUCTION

The Whitlock Tritium Meter was first introduced to the IRPA as a simple and satisfactory method for the direct measurement of Tritium surface contamination, at the 4th International Congress, Paris 1977. It was shown then that though instrumentally the measurements can be made simple, the fundamental characteristics of Tritium air absorption and self-absorption must be clearly understood, if the results of measurements are to be meaningful.

Practical experience of the author in the use of the Whitlock Tritium Meter in various laboratories and industrial establishments throughout the world since its first development has shown that:

a. Measurements by smear/wipe tests can often be in error by three orders of magnitude or more.
b. Sub-visual scratches [8μ deep] are radiologically important.
c. Volatile forms of Tritium exist in 20% to 30% of establishments visited.

The author questions the widespread use of smear/wipe techniques for the assessment of 3H surface contamination based on the assumption that 10% of removable activity is collected by the smear/wipe.

Tritium surface contamination assessed as "Fixed" can contain volatile fractions with a hazard potential which may be considerably greater than the hazard from removable activity at present covered by the Maximum Permissible Level [MPL] recommendations.

The Whitlock Tritium Meter has now been used for radiation protection purposes for six years in all manner of working environments, from Atomic Power Station, to Pharmaceutical laboratories. In addition, the author has demonstrated its use in several hundred establishments for direct measurement of suitable surfaces [1] and the rapid assay of smears.

During this time it has been of concern to observe a discrepancy in measurements which indicate that smear sampling techniques can be in error by a factor of 1000 or more. Of particular significance were surveys of several working surfaces [2] in seperate establishments which hed been shown clean or well below the MPL [Maximum Permissible Level] - notes[1] and [2] by smear surveys carried out by independent authorities a short

time previously.

Direct measurement of these surfaces, using the Whitlock Tritium Meter, revealed a residual activity estimated to be in excess of 1000 times the Maximum Permissible level. A reduction of this activity was observed when surfaces were de-contaminated with detergent and water. Successive cleanings reduced the total activity in a quasi exponential way.

The potential error in the assay of activity on the smears by Liquid Scintillation counting would be very unlikely to be greater than a factor of 2 or 3.

With regard to the smear, at the time when direct measurements were made, the surfaces appeared clean. The activity detected was not in the form of dust particles, and at that moment of time could be considered bound i.e. not easily removable activity. Thus we could conclude that the activity was fairly firmly fixed to the surface and as such the contamination indicated by smear was within the recommended levels.

However, consideration must be given to the following points:

1. With such a high activity present should only 0.1% become unbound by the day or week following the survey the Maximum Permissible Level even in the terms of removable surface contamination would be exceeded.
2. The assumption that the activity was "bound" is not necessarily correct. It was observed that large areas of the working surfaces surveyed by smears were covered with almost sub-visual scratches and activity in these may not be readily collected by a smear.
3. The isotope we are dealing with is Tritium, and the exposure pathways include absorption through the skin [3] as well as inhalation of dust particles.
4. In addition we have potential inhalation of volatile fractions [described later].

The main point of concern relative to these observations is that the present reliance on the smear technique to give reliable information as to the actual contamination hazard is injustified, even if the technique is only applied to smooth working surfaces as defined in the Codes of Practice. They would also suggest that radiological protection consideration of Tritium contamination should be based upon a direct measurement of surfaces contamination, if necessary by the provision of a suitable surface in locations where surfaces suitable for direct measurements [1] would not normally be available. Assumptions for the surrounding work surfaces can then be made relative to factual information.

If our findings are representative of working surfaces everywhere, they illustrate a potentially hazardous situation on a world wide scale, for they apply not only to laboratories where high levels of Tritium are used, but even low level laboratories, because the problem is not due to a "one-time" large spill, but a gradual history

of build-up due to poor house-keeping, as a result of highly inaccurate measurements giving a false sense of security.

The high inaccuracies in measurements also extend to certain types of Tritium sensitive survey instruments, capable of making direct measurements of very small areas of the working surface which, because of the very small sensitive areas [1.6 to 10cm^2] of the detectors, encourage the survey of a surface by a sweeping or scenning technique commonly adopted for isotopes emitting more energetic betas than Tritium, enabling the use of geiger counters. Here the problem of the integration time constant of rate meter instruments is often forgotten, and areas are covered more quickly than the probe area really permits. This is compounded by the very severe absorption by air of Tritium betas (range typically 0.5mm for 5.6 keV betas [1]. Lack of appreciation of the severe limitation imposed by this fundamental characteristic leads to the conduct of surveys where the detector is held outside the range of a significant number of emitted betas which, in turn, produces a potential error of infinity. As is the case with smear measurements, direct measurement by hand held devices can result in the conclusion that it is safe to continue working in the areas surveyed with the erroneous conviction that the contamination is below the Maximum Permissible Level.

To this situation we must add that in 20 - 30% of the establishments in which we have made direct surface measurements, using the Whitlock Tritium Meter, we find the presence of volatile Tritium. At first sight this would appear to be of academic interest, but on further consideration the radiological hazard could be considerable, and most likely specific to one individual among the personnel in the working environment. The Whitlock Tritium Meter detects the presence of volatile Tritium by virtue of the fact that a vacuum of approximately helf of an atmosphere is established in the measuring chamber for each measurement. If volatile fractions do exist, they are "driven off" and can be identified by successive measurements [10 seonds each] of the same area [100 cm^2). It is clear that we do not live in an environment where the atmospheric pressure suddenly changes 50%, so the condition is artifically emphasised, nevertheless, it is quite within the bounds of possibility that working surfaces will be subjected to temperature changes; for instance, sunlight through a window sweeping across the surface, or more likely, in consequence of a hot object, such as a human hand, or a recently heated beaker, being placed on the surface. In such circumstances, the volatile fraction given off will form an active "cloud" local to the person carrying out the work.

As we have seen from the previous discussion the source from which it comes can quite easily be several thousand times greater than the Maximum Permissible Level.

The active cloud would not readily be detected by installed Tritium in air monitoring equipment, due to the dilution volume of the room, or by urine tests, because of subsequest dilution [usually] in the total body water. The hazard seems likely to be the same as inhalation of dust particles from surface contaminetion.

CONCLUSION

Evidence suggests that the question of Tritium surface contamination should be re-considered in its entirety as the actual hazard may widely exceed acceptable levels. In particular the question of calibration of the smear technique, the effect of surface condition and the hazard of volatile fractions need investigation.

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Note 1: The M.P.L. [Maximum Permissible Level] referred to in the text = $10^{-4}\mu Ci/cm^2/100cm^2$

Note 2: There is no derived working limit [DWL] for Tritium

RECENT PAST AND NEAR FUTURE ACTIVITIES OF ICRP COMMITTEE 2*

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The presence of this paper on the program is, I am sure, related to the publication last year of the long-awaited report of ICRP Committee 2 on "Limits for Intakes of Radionuclides by Workers" (1). I will spend most of my time discussing that report, but I think it also appropriate to say something about the committee itself.

RECENT PAST

It has been my privilege to serve as a member of Committee 2 since 1970, and for some years prior to that as a member of its Task Group on Plutonium and Related Elements. Even before 1970, the Committee was at work on the revision of its earlier report on "Permissible Dose for Internal Radiation", which was published as ICRP Publication 2, in 1960 (2). By 1973, the last year of Karl Morgan's tenure as chairman of the committee, the revision was in a draft form covering the major items which eventually appeared last year in Publication 30. The year 1974 marked the beginning of Jack Vennart's tenure as chairman, and was the year the Main Commission adopted its policy of summing risks by means of weighted organ dose, as a replacement for the "critical organ concept." This decision removed the last technical obstacle to completing the report, and we began to actually believe our annual prediction that the report would be published "next year." In fact, publication had to await the issuance of the Commission's basic "recommendations" as included in Publication 26 (3), and the completion of voluminous machine calculations.

I have mentioned the names of Karl Morgan and Jack Vennart, who served so capably as chairmen of Committee 2. I must also acknowledge the major contribution of our late colleague, Geoffrey Dolphin who served as Secretary of the Committee during compilation of the final drafts of Publication 30. The contributions of Norman Adams and Michael Thorne, who assisted the Secretariat in putting the data together, must also be acknowledged. But above all, Walter Snyder must be remembered for the unstinting and patient ministration of his vast knowledge of internal dosimetry to the Committee and to the Committee's publications. I must name one other person--Mary Rose Ford--who assumed the calculational chore after the death of Dr. Snyder in 1977, and averted a catastrophe that everyone feared.

I would not like to leave the impression that Committee 2 was concerned, for the 20 years since 1960, only with the preparation of Publication 30. A supplement to Publication 2 was published in 1964 as a part of ICRP Publication 6 (4). Dosimetry models for the gastrointestinal tract (5,6) and for the respiratory tract (7) were published in 1966 by task groups of Committee 2. A joint task group of Committees 1 and 2 prepared, "A Review of the Radiosensitivity of the

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Tissues in Bone," which was published in 1968 as ICRP Publication 11 (8). Publication 19, "The Metabolism of Compounds of Plutonium and Other Actinides," appeared in 1972 (9). Publication 20, "Alkaline Earth Metabolism in Adult Man," followed in 1973 (10). The monumental "Report of the Task Group on Reference Man," ICRP Publication 23 (11), appeared in 1975, another memorial to the chairman of that task group, Walter Snyder. Other unpublished task group reports have been important in the development of Publication 30.

The current, and recent past membership of Committee 2 is shown in Table 1.

Table 1. Membership of ICRP Committee 2

Current	Recent Past*
J. Vennart, Chmn.	G.C. Butler
R.C. Thompson, Vice Chmn.	B. Chr. Christensen
W.J. Bair	G.W. Dolphin
L.E. Feinendegen	M. Dousset
M.R. Ford**	M. Izawa
A. Kaul**	W. Jacobi
C.W. Mays	J. Lafuma
J.C. Nenot**	J. Liniecki
B. Nosslin**	L.D. Marinelli
P.V. Ramzaev	W.G. Marley
C. Richmond**	K.Z. Morgan
N. Veall**	P.E. Morrow
	J. Müller
	V. Shamov
	W.S. Snyder
	C.G. Stewart

*Since work on Publication 30 began in 1967
**Appointed subsequent to final drafting of Publication 30, Part 1.

NEAR FUTURE

Only Part 1 of Publication 30 has thus far appeared in print, and this includes limits and metabolic data for the radioisotopes of only 21 elements (Table 2). Limits for an additional group of 31 elements (Table 3) are in the final stage of compilation and will be soon published as Part 2 of Publication 30. Michael Thorne, for the Committee, has compiled the metabolic data for an additional 18 elements and is working on 25 more. These will be considered by the Committee at its meeting next week, in Brighton, England, and will eventually be included in a Part 3 of Publication 30. Perhaps a Part 4 will be required to complete the elements.

Table 2. Elements included in ICRP Publication 30, Part 1 (with atomic number)

1	Hydrogen	38	Strontium	53	Iodine	90	Thorium
15	Phosphorus	40	Zirconium	55	Cesium	92	Uranium
25	Manganese	41	Niobium	58	Cerium	94	Plutonium
27	Cobalt	42	Molybdenum	84	Polonium	95	Americium
36	Krypton	52	Tellurium	88	Radium	96	Curium
						98	Californium

Table 3. Elements to be included in ICRP Publication 30, Part 2 (with atomic number)

9	Fluorine	26	Iron	45	Rhodium	76	Osmium
11	Sodium	29	Copper	47	Silver	77	Iridium
16	Sulfur	30	Zinc	48	Cadmium	79	Gold
17	Chlorine	35	Bromine	49	Indium	80	Mercury
18	Argon	37	Rubidium	54	Xenon	82	Lead
19	Potassium	39	Yttrium	56	Barium	83	Bismuth
20	Calcium	43	Technetium	61	Promethium	93	Neptunium
24	Chromium	44	Ruthenium	75	Rhenium		

For each "Part" of Publication 30, there will be issued a "Supplement". These supplements will be reproduced directly from computer printouts, and will tabulate the data employed in arriving at the recommended values of ALI and DAC. These data are necessary for the derivation of limits for exposure to mixtures of radionuclides, and to particles varying from the one micrometer diameter assumed for the tabulated ALI's and DAC's. Finally, there will be published a separate report tabulating the radionuclide decay schemes employed in deriving these limits. Much remains to be done before we have finished with Publication 30, but another year or two should see its completion.

While Committee 2 has, in the past, dealt only with problems of internal exposure, since November of 1977 its official title has been "Committee 2 on Secondary Limits," and it is the intention of the Main Commission that Committee 2 should have the responsibility for derivation of secondary limits for external exposure as well as for internal exposure. Committee 3, which formerly dealt with external exposure, is now Committee 3 on Protection in Medicine. However, the Main Commission has acknowledged that, for the immediate future, Committee 2 will be fully concerned with the preparation of secondary limits for internal irradiation.

Aside from the completion of Publication 30, identified future activities of Committee 2 in the area of internal exposure include the following. A task group of Committee 2, with Dr. Nosslin as chairman, is preparing a report on Dose to Patients from Radiopharmaceuticals. This effort is not concerned with establishing limits, but only with estimating patient dose from unit intake; questions of philosophy and medical ethics lie in the domain of Committee 3. Committee 2 will also be concerned with the improvement of internal dosimetry models, in particular those concerned with bone and lung. The bone model of Publication 30 does not take account of the burial of surface deposited radionuclides, and the lung model is applicable only to inhaled particulates. Finally, the Committee will address

itself to the establishment of radionuclide exposure limits for members of the public. The exact manner of formulating these limits has not been determined; however, it has been agreed that an exhaustive appraisal of the internal dosimetry of all radionuclides at all ages is impracticable, and that the approach will involve the application of a correcting factor to the occupational limit, this correcting factor probably being different for different radionuclides.

ICRP PUBLICATION 30, PART 1

Let me now return to a more detailed consideration of Part 1 of Publication 30 (1). I will concentrate on specific aspects of the Publication, which seem most important to its understanding and application. A most obvious change in Publication 30 is the absence of "Maximum Permissible Concentrations (MPC) or Maximum Permissible Body Burdens (MPBB), to which we had become accustomed in Publication 2. They are replaced by Annual Limits on Intake (ALI) and Derived Air Concentrations (DAC); the DAC being equivalent to the old MPC for air, but renamed to avoid the connotation that it should never be exceeded. The new limits are expressed in SI units, without even parenthetical microcuries.

Previous internal radiation exposure standards, as formulated in ICRP Publication 2, were based on limiting the dose equivalent received by the critical organ after a period of 50 years of continuous exposure, critical organ being defined as "that organ of the body whose damage by the radiation results in the greatest damage to the body" (2). The exposure standards derived in Publication 30, on the other hand, limit the annual effective dose equivalent commitment, thus differing in two respects from Publication 2: (1) the limit is on *annual commitment* rather than on ultimate realization, and (2) the limit is on *effective* dose equivalent, i.e., the sum of weighted organ or tissue dose equivalents. What are the implications of these changes?

The change from a dose rate achieved after 50 years of exposure to an annual dose commitment has no effect at all in a mathematical sense--one arrives at the same exposure limit by either approach. This is best illustrated graphically, as shown in Figure 1, where A_n, B_n, C_n, and D_n represent the dose in successive years resulting from the exposure in year n. For simplification, this illustration assumes that dose contributed beyond the fourth year is insignificant and that a steady-state total dose is therefore achieved after four years of constant exposure. It should be apparent from the graph that the annual dose commitment, $A_1 + B_1 + C_1 + D_1$, is numerically identical to the total dose in the 50th year, $A_{50} + B_{49} + C_{48} + D_{47}$. Concern has been expressed, however, by those who must apply these limits, that controlling to an expressed annual limit on intake may prove more restrictive than the old practice of controlling to a fraction of a permissible body burden. Thus, for a radionuclide tenaciously retained in the body, like plutonium, an accidental exposure to several Annual Limits on Intake might be considered an "overexposure", and as such might limit the work status of the exposed individual in future years, even though the actual radiation dose, received or projected during any year, is never more than a small fraction of that allowed on an annual basis. It must be emphasized that such

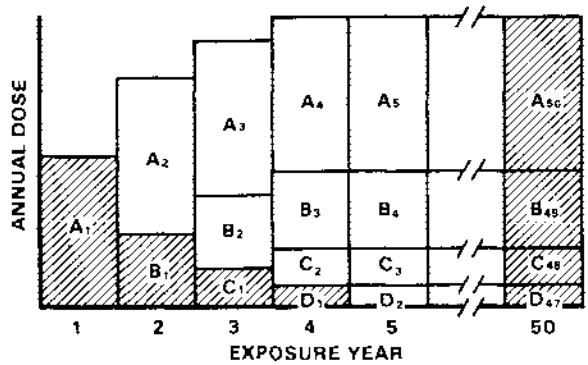

Figure 1. Illustration of dose commitment concept (see text)

an application of the dose commitment concept exceeds the intention of the ICRP, which employs it to calculate ALI's for the control of the environment in which people work, and not to determine the work status of exposed individuals.

The change from the "critical organ concept" to the concept of an effective dose equivalent, has more profound consequences. Instead of applying an assigned dose limit to a single organ or tissue considered critical, the total body limit is applied to the sum of doses received by all significantly exposed organs or tissues, each weighted in accord with its presumed contribution to the total risk of whole body exposure. This is certainly a more logical approach to limit setting and is, in fact, essential to a system based on the limitation of risk. Unfortunately, our knowledge of the biological parameters required to implement such a system is not entirely adequate, and the assumptions required because of that inadequacy result in uncertainties with the new system that are probably as large as the more obvious inaccuracies in the old system. And the calculational complexities introduced are formidable. The new system is, however, a more logically consistent one, and if we lack the information to implement it most effectively, it at least focusses attention on these shortcomings. I would only caution that our numbers are not as good as the refinement of the calculational procedures might suggest.

A vestige of the old "critical organ concept" still remains in Publication 30, in the guise of an overriding "non-stochastic" limit of 0.5 Sv (50 rem) per year, applicable to any organ or tissue except the lens of the eye, where the non-scholastic limit is 0.3 Sv (30 rem) per year. The weighted organ dose system of ICRP-26 would, in certain instances, allow doses to single organs in excess of 0.5 Sv (50 rem) per year, but this is prevented by the non-stochastic limit.

Let me now illustrate some of the previous discussion by considering, as an example, the derivation of the ALI for ingested ^{239}Pu. For this exercise I have gathered in Table 4 information which will appear in the supplement to Publication 30, Part 1. Table 4 lists values of committed dose equivalent per Bq ingested, for the tissues

Table 4. Committed dose equivalent (H_{50T}) and weighted committed dose equivalent ($W_T H_{50T}$) per unit ingestion of soluble ^{239}Pu in units of Sv/Bq, and derivation of annual limits on intake (ALI)

Tissue	H_{50T}	W_T	$W_T H_{50T}$
Ovaries	2.6×10^{-8}	0.25	0.6×10^{-9}
Red Bone Marrow	$17. \times 10^{-9}$	0.12	$20. \times 10^{-9}$
Bone Surfaces	$210. \times 10^{-9}$	0.03	$63. \times 10^{-9}$
Lower Large Intestine	5.4×10^{-8}	0.06	3.2×10^{-9}
Liver	$44. \times 10^{-8}$	0.06	$27. \times 10^{-9}$
			119.8×10^{-9}

Non-stochastic ALI (bone surfaces):

210×10^{-8} Sv/Bq $\approx$ 0.5 Sv/240 kBq

Stochastic ALI:

$\Sigma_T W_T H_{50T} = 119.8 \times 10^{-9}$ Sv/Bq $\approx$ 0.05 Sv/420 kBq

of significance. These values result from calculations based on metabolic models and dosimetric models which we will have more to say about later. The largest committed dose equivalent is calculated for the bone surfaces, which would have been considered the critical organ in the old system. We must still calculate a non-stochastic ALI for bone surfaces, which turns out to be 240 kBq; i.e., 240 kBq will deliver the non-stochastic committed dose equivalent limit of 0.5 Sv.

The committed dose equivalent values in Table 4 are multiplied by the weighting factors, W_T, as defined in ICRP-Publication 26, to give the weighted committed dose equivalents, which are summed to yield the effective committed dose equivalent. This sum is compared with the committed dose equivalent limit of 0.05 Sv for stochastic effects, leading to an ALI of 420 kBq. Since the stochastic limit is higher than the non-stochastic limit, the non-stochastic limit of 240 kBq is taken as the ALI for ingested soluble plutonium. For many radionuclides, non-stochastic limits are controlling and we have, in fact, a limit still based on a single critical organ. We have in the process, however, considered all organs and tissues to insure that a summation of organ risks would not have led to a more restrictive value.

I thought it best to dwell at some length upon the preceding basic changes in philosophy, but it leaves us time to consider only briefly a number of other important changes.

Dose equivalent (H) is now defined as the product of absorbed dose (D), a quality factor (Q), and the product of any other modifying factors (N). Publication 2 employed a conceptually similar "RBE dose", which was the product of absorbed dose, relative biological effectiveness (RBE) and a relative damage factor (n), which applied only to non-radium alpha and beta emitters in bone. Though somewhat differently defined, the new quality factor serves the same function as the old RBE, and the old "n" factor might be considered a specific modifying factor which could be part of N. However, we no longer calculate average dose to bone, so the "n" factor is no longer needed. A significant change has been made in

the numerical value of the quality factor for alpha particles, which is now 20 rather than 10.

Dose equivalent commitment in a given "target" organ or tissue ($H_{50}T$) is now calculated by taking into account the "crossfire" from radiations originating in all other significant "source" organs or tissues, as well as the radiation originating in the target organ itself. This is a complex process, some of the intermediate stages of which will be detailed in the supplements to Publication 30.

The distribution and retention of radionuclides among and within the various source organs is determined by application of suitable metabolic and dosimetric models. These models, for most elements, have become considerably more complex during the interval between Publication 2 and Publication 30. General dosimetric models for the respiratory system, the gastrointestinal tract, for bone, and for submersion in a radioactive cloud, are described in Publication 30, Part 1. The model for the respiratory system is that developed by the Task Group on Lung Dynamics (7), as modified in ICRP Publication 19 (9). The model for the gastrointestinal tract is based on the model developed by Eve and Dolphin (5,6). The bone model estimates dose to red marrow and to bone surfaces, for cortical and trabecular bone; radionuclides being assumed to deposit either uniformly throughout bone or on bone surfaces. For most elements, detailed information on distribution within bone is not available. In the absence of more specific information it is assumed that: (1) alkaline earth radionuclides with radioactive half-lives greater than 15 days are uniformly distributed throughout the volume of bone, (2) radionuclides with radioactive half-lives of less than 15 days are uniformly distributed on bone surfaces, (3) radionuclides on bone surfaces are equally distributed between trabecular and cortical bone, and (4) radionuclides uniformly distributed throughout the volume of bone are present 20% in trabecular bone and 80% in cortical bone.

Specific metabolic models are employed for each radionuclide, and these are not restricted to any particular mathematical form, although most are systems of first order differential equations with constant coefficients. More than one value for the absorption coefficient from the gastrointestinal tract and/or lung may be employed, to represent different compound forms; the respiratory tract model also provides for three different classes of compounds, based upon their assumed clearance rate from the lung. The metabolic models and briefly summarized supporting data are presented separately, for each element in Publication 30, just preceding the tabulated values of ALI and DAC.

Finally, let us look at an example of the actual limit values, as tabulated for plutonium. In Table 5, values are shown for ^{239}Pu; Publication 30 lists values for 12 isotopes of plutonium--from ^{234}Pu to ^{245}Pu. Oral ALI's are listed for compounds exhibiting two absorption fractions, the value of 10^{-5} applying to oxides and hydroxides and the value of 10^{-4} applying to other commonly occurring compounds. It is indicated that the ALI's are based on a non-stochastic limit for irradiation of bone surfaces and that in the absence of such a limit the stochastic limit would have been the value shown in parentheses. Inhalation ALI's are listed for two compound classes, Class Y being applicable to plutonium oxide

Table 5. ALI (Bq) and DAC (Bq/m^3) (40 h/wk) values for ^{239}Pu as listed in ICRP Publication 30

	Oral		Inhalation	
			Class W	Class Y
	$f_1 = 1 \times 10^{-4}$	$f_1 = 1 \times 10^{-5}$	$f_1 = 1 \times 10^{-4}$	$f_1 = 1 \times 10^{-5}$
ALI	2×10^5	2×10^6	2×10^2	5×10^2
	(4×10^5)	(3×10^6)	(4×10^2)	(6×10^2)
	Bone surf.	Bone surf.	Bone surf.	Bone surf.
	$[15 \times 10^5]$*	$[9 \times 10^6]$*	$[2 \times 10^2]$*	$[27 \times 10^2]$*
DAC	---	---	8×10^{-2}	2×10^{-1}
			$[7 \times 10^{-2}]$*	$[15 \times 10^{-1}]$*

*Values derived from ICRP Publication 2.

and Class W to other commonly occuring compounds; the fraction absorbed from the gastrointestinal tract after clearance from the lung is different for the two classes. Again the ALI's are based on a nonstocnastic limit. Values of the DAC are shown for the two compound classes.

As a matter of interest, I have listed in brackets, in Table 5, the values for ALI and DAC which one would obtain from the old limits of Publication 2. The DAC values are derived from the old $(MPC)_a$ values by a simple change of units. The ALI's are derived from the MPC values by multiplying by the assumed annual intake of water or air, and by changing units. It will be seen that, except for inhaled Class W compounds, where there is no significant change, all ^{239}Pu limits have become more restrictive by a factor of about six. Similar comparisons, involving a representative isotope of each element, suggest that about 1/3 of Publication 2 limits have become more restrictive, by as much as a factor of 100; about 1/3 have become less restrictive, by as much as a factor of 25; the remaining 1/3 have changed by less than a factor of 2.

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THE IDENTIFICATION OF CRITICAL GROUPS

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INTRODUCTION

The system of dose limitation recommended by the International Commission on Radiological Protection (ICRP) (1) includes the overriding requirement that dose equivalents received by individuals shall not exceed appropriate recommended limits. Control of radioactive waste disposal to the environment therefore necessitates an identification of those members of the public who are the most highly exposed. The actual doses received due to a given practice will vary because of differences in age, size, metabolism and customs, as well as variations in the environment. The ICRP (1,2) considers that it is usually feasible to take account of these sources of variability by the selection of appropriate critical groups. However, the way in which critical groups should be identified has not been precisely defined: it has been for national authorities to interpret.

In the context of aquatic disposals of liquid radioactive waste it is usual in the U.K. to consider each site individually when assessing doses to members of the public, because of the many variables which require to be taken into account. Standard practice is to investigate each critical pathway by means of a survey of the habits of the (usually) local population. A consistent approach to selection of the critical group from habits survey data is required. In this paper, criteria (mostly based on ICRP recommendations) for identification of critical groups are discussed. Techniques are examined in the light of these criteria and experience at the Fisheries Radiobiological Laboratory (FRL), Lowestoft. Recommendations are made on how critical groups should continue to be identified and characterised.

CRITERIA FOR IDENTIFICATION OF CRITICAL GROUPS

ICRP provides qualitative guidance for identifying critical groups. Two important criteria emerge from paragraph 85 of ICRP 26 (1):

(a) The critical group should be representative of those individuals in the population expected to receive the highest dose equivalent.

(b) The critical group should be small enough to be relatively homogeneous with respect to age, diet and those aspects of behaviour which affect the doses received.

The way in which homogeneity within a sample should be tested is laid down in paragraph 215 of ICRP 26 (1) (quoted): "The dose

equivalent to a specified organ or tissue in a given population group will normally be determined on the basis of a representative sample. The spread of the observed values will be an indication of the homogeneity of the sample, and thus of the group, and will provide a statistical basis for judging whether the group has been suitably defined." It is not of course possible to apply this test in practice. It is, however, possible to define the group such that the test will be largely satisfied. As far as internal irradiation is concerned, actual doses received will also vary with metabolic factors. The test will be essentially satisfied if the range of habits within the group is such that the expected metabolic variations will be predominant in determining the spread of doses. Preston *et al.* (3) have suggested this approach, and have presented a review of relevant metabolic data. It was concluded that a range in habits corresponding to a ratio of not more than 3 between the maximum observed and the mean characterising the group would be sufficient.

With external exposure, there will undoubtedly be some variation between individuals in weighted mean whole body dose equivalent owing to physiological differences, mainly in tissue thicknesses. For penetrating radiation variations are, however, likely to be considerably less than those caused by metabolic factors which affect internal irradiation. For external exposure it is not therefore sufficiently restrictive to accept the same factor for limitation of the critical group.

Other criteria, based on ICRP recommendations (1) for methods of selecting critical groups and guidance from earlier ICRP publications (2,4), may be summarised, with a little comment, as follows:

(c) Results should not depend heavily on the possibly capricious or wishfully-described habits of very few extreme individuals.

(d) The method should not result in the possibility of many individuals exceeding the dose limit for the general public if the critical group were to be exposed near the dose equivalent limit. ICRP accepts that some extreme members of a critical group will in fact receive doses somewhat higher than the mean to which the appropriate dose equivalent limit is applied, but states that doses received will usually be overestimated because of maximising assumptions used.

In addition the further practical considerations have been noted by Shepherd (5):

(e) The method chosen should not allow ambiguous interpretation.

(f) It should not require an unjustifiably large survey effort. The effort will depend on the size of the exposed population, but the method should allow attention to be focussed on the individuals expected to be exposed at the highest rates (cf. (a)).

(g) It should require little mathematical analysis - certainly no more than is justified by the nature of the survey data, which as collected are in many cases subjective and require interpretive skill of the interviewer.

METHODS FOR SELECTION OF CRITICAL GROUPS FROM SURVEY DATA

Preston *et al.* (3) and Shepherd (5) have reviewed methods existing up to 1974 for selecting critical groups. The methods are summarised, and any of the above criteria which are unfulfilled by them are given, in the table below.

Type of method	Method	Criteria unfulfilled
Arbitrary	(i) Selection of exceptional data	(e)
	(ii) Fixed fraction of exposed population	(f)
Statistical	(iii) The deviant subgroup	(a),(f),(g)
	(iv) Statistically defined cut-off	(f),(g)
	(v) Extreme value theory (6)	(b),(f),(g)

Two additional methods may be described:

(vi) Cut-off based on homogeneity

This method derives from the discussion under criterion (b). The lower limit for critical group membership is set such that the ratio between the maximum observation and either the lower limit or, less restrictively, the mean of the chosen group itself is equal to 3. This method, as discussed above, would not be sufficiently restrictive for external exposures. Given this, the only unsatisfied criterion is (c); however, this is not serious since a mean over a number of observations is subject to less sensitivity than the maximum observation itself, and for small populations there will be a greater chance of successfully locating the extreme member.

(vii) The 'top thirty' method due to Shepherd (5)

That reasonable homogeneity in the critical group should be observed, as well as criteria (c) and (d), suggests that it is necessary to choose a critical group whose size is relatively constant compared with that of the exposed population. Shepherd (5) notes that in a number of consumption rate surveys the numbers of individuals exceeding observed levels appears in the upper region to decrease as the power of about 3. This observation has some basis on the assumption of an underlying log-normal distribution. A given homogeneity therefore implies a relatively constant size of critical group, numbering about 3^3 (≈ 30) for a maximum to minimum consumption rate ratio of 3. This size applies to all of those more highly exposed and an estimate of the surveyed fraction is required. The method may also be shown to apply less well to exposed populations of less than a few hundred.

Applying the method to consumption rate surveys in which the exposed population is higher than this gives critical group consumption rates which agree fairly well (5) with those values accepted after applying previous methods and taken as reasonable in the light of experience. So far the method has only been applied to consumption rate data. For occupancies, the exposed populations involved are often small. Also, as already pointed out, the homogeneity criterion is more restrictive for external exposures. The appropriate method is then that of averaging the relevant extreme observations (method (i)). However, for internal exposure due to occupancy, via inhalation as the prime example, the last restriction would not necessarily apply.

RECOMMENDATIONS

Data analysed at present fall mainly into two broad classes: consumption rates in connection with internal irradiation and occupancy data in relation to external exposure. It would appear that both the nature of the data and the way in which the criteria (particularly for homogeneity) apply differ sufficiently strongly between the two classes that separate approaches should be made in identifying critical groups.

For consumption rate data, provided the exposed population is sufficiently large, method (vii) would appear to offer the most objective approach with the ability to satisfy established criteria. The following may clarify the procedure. The consumption rate survey should be carried out with particular emphasis on the higher rate consumers. The proportion of those surveyed should be estimated as objectively as possible. A logarithmic plot of numbers exceeding observed levels of consumption will indicate if the inverse third power law relationship is obeyed. If so the surveyed fraction of the top thirty consumers can be chosen as representative of the critical group. The critical group mean can then be calculated, and this should be the arithmetic mean of the representative observations. If the inverse third power law is not obeyed, as will be the case for small populations of consumers, the homogeneity criterion can still be applied to the observed distribution, as in method (vi). A lower limit to critical group membership should be applied at a consumption rate equal to one-third of the maximum observed.

Regarding external exposures, the exposed populations are often small and the homogeneity criterion is more restrictive than for consumption rates. It is recommended therefore that the simple approach of method (i) be used. In many cases the more highly exposed persons are fairly easy to identify; estimation of occupancies can often be carried out fairly reliably since they may occur by the nature of the persons' work.

Recommendations are also given in relation to internal irradiation following inhalation during occupancy. Distributions of occupancy encountered are described above. These require to be combined with a homogeneity criterion likely to be of similar flexibility to that for consumption rates. This combination is, then, similar to that described for small populations of consumers and method (vi) is appropriate. In some cases, the occupancy distribution will inevitably indicate that as for external exposure the critical group will consist of a few of the more extreme observations.

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ASSESSMENT OF THE RADIATION EXPOSURE FROM THE RADIOACTIVE MATERIAL RELEASED FROM THE STACK OF A 2000 MWe COAL FIRED POWER STATION

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INTRODUCTION

This paper is a summary of work which is in progress at the National Radiological Protection Board to estimate the radiological impact of the discharges to atmosphere from a reference 2000 MWe coal fired power station. Provisional estimates are made of individual committed effective dose equivalents for members of hypothetical critical groups in the population whose habits would result in higher than average radiation exposures, and the collective effective dose equivalent commitment to the population of Great Britain. The reduction in the natural background dose from ^{14}C resulting from releases of stable carbon is considered and finally, areas of further research relating to radiation exposure from the operation of coal fired power stations are identified.

SOURCE TERMS FOR DISCHARGE TO ATMOSPHERE

The activity discharged from the power station is in two phases: a gaseous phase consisting predominantly of radon gas and a solid particulate or ash phase containing ^{232}Th, ^{235}U, ^{238}U, ^{40}K and their chain decay products. The doses from radon discharges are insignificant (less than 0.2% of the total) and are not discussed further in this paper.

To calculate the release rate of radioactivity, the mass discharge rate of ash and the concentration of activity in ash are required. It is assumed that on full load the power station discharges at the Statutory Limit for particulate emission in the UK for modern power stations of 115 mg m^{-3}. With a load factor of 56%, this is equivalent to a discharge of 3.9 10^3 t y^{-1} for a 2000 MWe power station (1). Data on the concentration of radioactivity in fly ash in the UK are limited and until further experimental measurements are reported an interim cautious estimate of 500 Bq kg^{-1} is used for each of the nuclides ^{232}Th, ^{238}U, ^{40}K and their chain decay products assuming equilibrium. It is recognised that this value may overestimate the concentration of non-volatile species in the decay chains and therefore the sensitivity of the results to this parameter is discussed later. Assuming a natural abundance of ^{235}U in uranium of 0.72% by mass, the concentration of ^{235}U in fly ash is taken to be 20 Bq kg^{-1}. A summary of the source terms for discharge is given in Table 1. The power station is assumed to discharge continuously at this rate during an operational life of 30 years.

TABLE 1. SOURCE TERMS FOR DISCHARGE TO ATMOSPHERE

	Discharge rate ($Bq\ y^{-1}$)
^{235}U series	$7.8\ 10^{7}$
^{232}Th series	$2.0\ 10^{9}$
^{238}U series	$2.0\ 10^{9}$
^{40}K	$2.0\ 10^{9}$
^{14}C*	$-9.0\ 10^{11}$

* See text for further explanation

ENVIRONMENTAL TRANSFER AND PATHWAYS ANALYSIS

The environmental transfer models used in this assessment are based on a methodology to evaluate the consequences of routine radioactive releases developed by the National Radiological Protection Board in conjunction with the Comissariat a L'Energie Atomique (2) which was outlined in a previous paper at this Congress (3).

From the source terms, annual average concentrations in air and deposition rates to ground are predicted assuming a uniform wind rose and an effective release height of 300 m in Pasquill dispersion conditions A,B,C and D and taking into account wet and dry deposition. In EF weather, the effective release height is assumed to be at the inversion lid, 200m above ground level. The sensitivity of the results to the choice of stack height has been investigated for release heights of 400 and 600m and the results are found to vary by less than 5%. Three sites in different geographical locations in Great Britain were chosen for investigation (Midlands, Central Southern England and London), and meteorology characteristic of each of these sites was used. There is less than 10% variation in the collective doses from these sites. Intakes of radioactivity by ingestion and inhalation are converted to effective dose equivalents using the dosimetric data given by Adams et al (4).

Five pathways of exposure are considered : (i) External irradiation from the plume of activity - a 'finite cloud' model is used to calculate the dose rate in tissue 1m above ground level from photons originating in the plume. The maximum exposed individual is assumed to remain at the point of maximum average dose rate for a year. No allowance is made for shielding by buildings or clothing. (ii) Inhalation of the plume - the typical breathing rates for an average adult are used to calculate collective doses and those for an individual carrying out heavy work (5) to calculate doses to the maximum exposed individual. This individual is assumed to remain at the point of highest average air concentration for a year. (iii) External irradiation due to surface deposition - it is assumed that material deposited on the ground builds up during the operating life of the plant and that the external dose rate from a given deposit, calculated from a model of a uniformly contaminated plane, falls off exponentially with a 20y half-life. The maximum exposed individual is assumed to remain at the point of highest dose rate (which occurs in the 30th of operation) for a year. Again no allowance is made for possible shielding by buildings.

(iv) Resuspension of deposited activity - the model which is based on a resuspension factor approach, has two components, one of which represents resuspension from recent deposits weathering with a half-life of 55 days, the other from weathered deposits with a removal half-life of 100 years. The maximum exposed individual is assumed to breathe the highest resuspended air concentration. (v) Ingestion of contaminated foodstuffs - a dynamic compartment model is used to predict activity concentrations in plant and animal foodstuffs following transfer in the foodchains. The maximum exposed individual is assumed to ingest contaminated grain, green vegetables, root crops, meat, liver and milk at higher than average rates; these are taken to be 145, 80, 170, 90, 20 kg y^{-1} and 340 l y^{-1} respectively. These figures are consistent with values developed for assessing DL's for radioisotopes in the environment.

RELEASES OF STABLE CARBON

The specific activity of ^{14}C in exchangeable carbon in the biosphere is assumed to be reduced by releases of stable carbon from the power station. The carbon releases are equated with a negative discharge rate of ^{14}C as given in Table 1, and the dose reduction to individuals is estimated from carbon consumption rates of 300 kg y^{-1} and for the population from a per caput consumption rate of 93 kg y^{-1}.

TABLE 2. PROVISIONAL INDIVIDUAL AND COLLECTIVE DOSE ESTIMATES[a]

	Individual dose[b] (μSv)	Collective dose[c] (man-Sv)	Dominant radionuclides
External γ plume	$5.0\ 10^{-5}$	$7.9\ 10^{-3}$	^{214}Bi, ^{214}Pb
External γ ground	2.7	9.1	
Inhalation	1.1	$2.9\ 10^{2}$	^{232}Th, ^{230}Th, ^{228}Th
Resuspension	3.1	6.3	
Ingestion	$2.2\ 10^{2}$	$6.4\ 10^{1}$	^{210}Po, ^{210}Pb, ^{231}Pa
^{14}C	$-1.4\ 10^{-1}$	$-2.2\ 10^{1}$	-
Total	$2.3\ 10^{2}$(d)	$3.4\ 10^{2}$	^{232}Th, ^{210}Po, ^{230}Th, ^{210}Pb, ^{231}Pa, ^{228}Th

(a) Effective release height = 500m in ABCD weather : Site = South Oxfordshire : Site boundary = 400m
(b) Maximum annual committed effective dose equivalent to a hypothetical individual
(c) Collective effective dose equivalent commitment truncated to 500 years to the population of Great Britain from 30 years plant operation
(d) Total based on the cautious assumption that same hypothetical individual is exposed to the appropriate contribution from all these pathways at a distance of 400m from the stack

RESULTS AND DISCUSSION

The provisional individual and collective dose estimates are summarised in Table 2. The reduction in ^{14}C doses is significantly smaller than the dose from releases of fly ash. The annual committed effective dose equivalent to a hypothetical individual with habits characteristic of a critical group is 230 μSv and derives predominantly from ingestion of contaminated foodstuffs. The nuclides ^{210}Pb, ^{231}Pa and ^{210}Po give 43%, 38% and 14% of the maximum individual dose respectively. It should be noted that consistent overestimates of parameters are used in the models to predict the maximum individual dose, for example, in the assumption that the individual derives all components of his diet from the same location which corresponds to the point of maximum deposition . It is likely that, in practice the most exposed individual will receive a dose which is considerably smaller than the above estimate. The collective effective dose equivalent commitment to the population of Great Britain from the operation of the plant in South Oxfordshire for 30 years is 310 man-Sv. The pathways giving the highest collective dose are inhalation of the plume and ingestion of contaminated foodstuffs. The nuclides ^{232}Th, ^{210}Po, ^{230}Th and ^{228}Th give 49%, 10%, 10% and 9% of the collective dose from radioactive releases respectively.

The levels of dose calculated here indicate that further research should be devoted to improving understanding of the radiological impact from coal-fired power stations. Two of the subject areas to which this assessment is most sensitive are the transfer of Pb, Po and Pa into foodstuffs and the concentration of radioactivity in fly ash. For example, there is some evidence that activity concentrations of non volatile elements such as Ra and Th could be a factor of four lower than the values used in this study (6). If this is the case then collective doses would be reduced by up to a factor of three. Research effort should therefore initially be directed towards these subject areas. However this assessment only focusses on one aspect of part of the coal fuel cycle, and to put it into its true perspective the radiation exposure as a result of practices over the whole cycle should also be considered. Even these results would need to be looked at carefully within a wider context.

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EFFECT OF THE FOODCHAIN IN RADIOACTIVITIES RELEASED FROM THERMAL POWER PLANTS

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INTRODUCTION AND SUMMARY

The radiological impact of fossil fuel burning (effect of radioactive impurities) has been investigated since the first paper by Eisenbud and Petrow (1), but no attempt has been made about the effect of the foodchain. Among many radioactivities released from fossil fuel burning ^{210}Pb and ^{210}Po are especially important, because they are very volatile and are released at higher rates than others, and also because they are concentrated very strongly in marine organisms (2,3). This is very important in Japan, because most power plants in Japan are built near the seacoast and the seawater contamination due to these radioactivities is likely. ^{210}Pb and ^{210}Po also contaminate leafy vegetables by fallout. Since Japanese eat vegetables and seafoods very often, this foodchain is important.

The dose due to this foodchain is estimated. It is found to be two to three orders of magnitude higher than the dose calculated previously (4) without considering this foodchain. The average individual dose of Japanese is of the order of several tens of μSv (several mrem)/y. Although much smaller than the natural background, it is comparable to the dose of the *total* fuel cycle of *full scale* nuclear electricity generation (1 kW per person), and so its social implication cannot be overlooked. The same argument applies to other countries where the seafood consumption rate is high.

IMPORTANCE OF ^{210}Pb AND ^{210}Po

Recent coal-fired power plants have excellent dust removal devices, the efficiency of which reaches 99% or higher. However, this removal efficiency is referred to the *weight* of particulates. Small particulates, which can pass through the dust removal device, have a large surface-to-volume ratio, and volatile substances such as ^{210}Pb or ^{210}Po are adsorbed on the surface of these small particulates. Such small particulates are especially dangerous, because they spread to much further distances than larger ones do, are not washed away by rain easily, and when inhaled, reach the deepest part of lungs.

In addition there could be a gaseous emission. The dust removal efficiency is referred to *solid* particulates, and so if the gaseous emission is included, the release rates of volatile substances could be much higher than is usually believed.

One might argue that ^{210}Pb and ^{210}Po produced from ^{222}Rn emanated from the ground is still much larger than those produced from coal burning, but this argument overlooks an important point. Radon daughters are usually adsorbed by atmospheric aerosols, the residence time of which is 10–20 days at most. This is too short for the long-

lived radioactivities such as ^{210}Pb (20 y) or ^{210}Po (138 d) to reach equilibrium with their parents, and hence the atmospheric contents of ^{210}Pb and ^{210}Po are many orders of magnitude lower than ^{222}Rn.

On the contrary, all radioactivities in fossil fuels have completely reached their radioactive equilibrium while the fossil fuels have been underground for hundreds of millions of years. When the fossil fuels are burnt, these radioactivities are released *keeping their equilibrium*. Thus the ratio of ^{210}Pb or ^{210}Po to ^{222}Rn is *one*. Therefore as far as the air contamination due to ^{210}Pb or ^{210}Po is concerned, coal burning is relatively more effective than radon emanation from the ground by several orders of magnitude.

This point is important also when comparison is made with nuclear energy. Radon emanation from uranium tailings is essentially the same as that from the ground, and so the same argument as above can apply. For instance, one 1000 MW coal-fired power plant releases about 1 Ci of ^{222}Rn every year, which in the effect of ^{210}Pb and ^{210}Po is comparable to the release of 7000 Ci from tailings of the Ranger uranium mine in Australia, which can provide 6000 tonnes of U_3O_8 annually.

CONTAMINATION OF LEAFY VEGETABLES AND SEAFOODS

^{210}Pb and ^{210}Po are taken up by plants both through root uptake and direct absorption from the leaves. Although which of the two pathways is more important is somewhat controversial, the argument claiming direct absorption appears more persuasive, especially in the case of leafy vegetables. Therefore in this paper it is assumed that ^{210}Pb and ^{210}Po in leafy vegetables are completely due to direct absorption from fallout.

The ratio of ^{210}Pb or ^{210}Po produced from coal burning to natural ones is denoted as R_a in air and R_w in water. R_a is estimated to be 0.03 ~ 0.1 (5). However, this is the average over Japan. Leafy vegetables are usually grown near cities to be supplied in a fresh form, and R_a in air of urban areas is probably higher than 0.1.

The intake of ^{210}Pb from leafy vegetables is about 3 pCi/d (6). The data on the ^{210}Po intake from leafy vegetables is not available, but can be estimated as follows. The average of ^{210}Po content in leafy vegetables in Japan is 0.35 pCi/100g (7). Since the daily consumption of vegetables by Japanese is about 300 g/d, the ^{210}Po intake is about 1 pCi/d. Therefore contributions from coal burning are 0.3 and 0.1 pCi/d for ^{210}Pb and ^{210}Po, respectively, assuming $R_a = 0.1$.

Contamination of seafoods is much more difficult to estimate, because R_w is very uncertain. However a tentative estimate is $0.001 < R_w < 0.05$ (5). The ^{210}Pb intake from seafoods by Japanese is about 12 pCi/d (6). The data on ^{210}Po is not available, but is estimated to be about 40 pCi/d (5). Then contributions from coal burning are 0.012 ~ 0.6 pCi/d for ^{210}Pb and 0.04 ~ 2 pCi/d for ^{210}Po.

DOSE DUE TO COAL BURNING

The individual dose can be calculated using the model of UNSCEAR 1977 and the data of Parfenov (3). The results are shown in Table 1.

TABLE 1. Individual dose due to contamination of leafy vegetables and seafoods by coal burning in Japan (mrem/y).

Organ	Leafy vegetables	Seafoods[a]	Total[b]
skeleton	3.7	0.15 ~ 7.5	3.9 ~ 11.2
gonads	0.7	0.1 ~ 3.9	0.8 ~ 4.6
breasts	0.2	0.03 ~ 1.3	0.3 ~ 1.5
lungs	0.6	0.07 ~ 3.4	0.7 ~ 4.0
thyroid gland	0.6	0.07 ~ 3.5	0.7 ~ 4.1
liver	1.9	0.2 ~ 10.4	2.1 ~ 12.2
kidney	1.6	0.2 ~ 8.8	1.8 ~ 10.4
lymph nodes	1.2	0.1 ~ 6.7	1.3 ~ 7.9
pancrea	0.4	0.04 ~ 2.2	0.4 ~ 2.6
spleen	0.4	0.04 ~ 2.1	0.4 ~ 2.4
whole body	0.9	0.1 ~ 4.9	1.0 ~ 5.8

a) Upper and lower values correspond to upper and lower values of R_w in the text.

b) For countries other than Japan the values are roughly 1/10.

The population dose is calculated by multiplying the whole body dose by the Japanese population of 116 millions. This corresponds to coal burning of 80 ~ 90 million tonnes/y, which is equivalent to 35-40 1000 MW coal-fired power plant. Thus the above value is divided by 40 to give the collective dose per 1000 MW power plant.

However fossil fuel burning releases CO_2 which lacks ^{14}C, and as a result the atmospheric ^{14}C is diluted (Suess effect). This gives rise to a *decrease* of internal dose due to ^{14}C. The collective dose commitment of ^{14}C integrated over the world population and up to 100 years ia given as 40 man-rem/Ci of ^{14}C (average of three values) (8). Operation of one 1000 MW coal-fired power plant can be regarded as a *negative* emission of ^{14}C of 11.4 Ci/y (5) which corresponds to a *decrease* of the collective dose of about 460 man-rem/y.

Since comparison with nuclear energy is important from the sociological viewpoint, the relevant data of nuclear energy are also shown in Table 2 together with the data of coal.

TABLE 2. Collective dose of coal burning in Japan[a] and its comparison with nuclear energy. (All in man-rem/y except for the Harrisburg accident.)

	Coal	Nuclear energy
	Japanese population dose	
	$1.2 \sim 6.7 \times 10^5$	1.7×10^4[b]
	Per 1000 MW power plant	
(A)	18 ~ 23[c]	13[e]
(B)	$2.9 \times 10^3 \sim 1.7 \times 10^4$[d]	$5.2 \sim 8.2 \times 10^3$[f]
Suess effect	-4.6×10^2	Harrisburg nuclear accident
		3.3×10^3 man-rem

a) In the case of other countries the dose due to coal burning is about 1/10 of Japan.

b) Dose due to all nuclear facilities all over the world (mainly from ^{3}H and ^{85}Kr) in 2000 (9).
c) Values neglecting the foodchain (4).
d) Values including the foodchain.
e) Routine operation only (4).
f) Total fuel cycle (UNSCEAR 1977).

As can be seen from the table, the collective dose of coal burning including this foodchain is 100 to 1000 times larger than that excluding it, clearly showing its importance. The Suess effect is much too small to cancel it. Compared with nuclear energy the dose due to coal burning is comparable to or even greater than that of nuclear energy. Thus its social implication cannot be overlooked.

OIL AND NATURAL GAS

These are usually believed to be much cleaner than coal. As a crude estimate the dose due to oil burning is assumed to be about 1/100 of that of coal, namely about 30 ~ 170 man-rem/y per 1000 MW power plant, which can be cancelled by the Suess effect (-280 man-rem/y for oil). The same is probably true for natural gas also.

However it must be pointed out that natural gas contains a large amount of radon and presumably its daughters including ^{210}Pb and ^{210}Po, which could alter the above estimate significantly. The same can be said for oil as well. This point must await further investigation.

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HAZARDS FROM RADIOACTIVITY OF FLY ASH OF GREEK COAL POWER PLANTS (CPP)

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Abstract. Fly ash and fine dispersion releases by coal combustion in Greek CPPs are radioactive. Concentrations in the fly ash up to 20 pCi/g and 10 pCi/g were measured for ^{238}U and ^{226}Ra respectively (not in secular equilibrium). The radioactivity of fly ash deduces risks in two ways: a) from the escaping fly ash in particulate form or fine dispersion and b) from using fly ash as substitute of cement in concrete.

1. Introduction. Coal burning by CPP is one of the principal sources of Radioactivity of the Atmosphere (1-8). The radioactivity escapes from stacks of CPPs in particulate form (fly ash) or fine dispersion. In both cases a potential hazard in the vicinity of CPPs is constituted.

It has been proposed (9-12) to use fly ash as substitute of cement in concrete. Since fly ash would be radioactive, its use could involve potential hazards, arising, either from direct irradiation from concrete or from radon diffusion from it.

2. Experimental Procedures and Results. The samples of fly ash studied were from the Greek CPP's. Their radioactivity was mainly measured by Ge-Li detectors using the direct γ-spectroscopy and activation analysis by thermal neutrons. Fig. 1 shows a typical gamma spectrum. In the spectrum we clearly see the important gamma transitions of the uranium series from ^{226}Ra onwards. The γ-peaks of the thorium series and of ^{40}K are absent. The results for ^{238}U, ^{226}Ra and ^{232}Th are presented in Table 1a. In 1b, the concentrations of the above nuclides for the lignites of principal Greek Coal Mines, burning in the Greek CPP's, are also presented. It was found that uranium and radium are not in secular equilibrium in lignites and fly ashes, but the physics involved escapes of the present paper.

3a. Radioactivity escaping in particulate form of fly ash. From the stacks of CPP, fly ash escapes in a percentage of 3% or 1% of the total fly ash, in the better cases. The total amount escaping per each Greek CPP reaches up to about 7.5×10^{11}g/y. By differential sieves and weighing, the grain distribution of fly ash was determined. In the Fig.2, the distribution function of fly ash consisted of grains with diameter ranged between δ and $\delta+\Delta\delta$ in presented. We observe that this distribution presents a strong maximum of grain mass with mean diameter ranged between 20 and 70 μ.

The horizontal distance x, in meters, in which the grain of mean diameter δ will fall on the ground surface from the height h of the stack, is given by the equation, derived by Stokes law: $x = 1.36 \times 10^4 . (h/\delta^2)\vec{u}$ (meters), where h the height of the stack, in meters, u the mean wind speed in m/sec and δ the mean grain diameter in μ. As a consequence of the distribution function of Fig.2, the deposition of fly ash shows a maximum which, for the case of u=1m/sec, is at a

region about 400 m upwind of the stack, Fig.3. This maximum is removed to major distances as much as the wind speed is increased. On the site of maximum, ^{226}Ra deposition has the value: $6.2 \times 10^{-9} \mu Ci/cm^2.d$.

3b. Radioactivity escaping from fine dispersion. Taking in consideration the quantity of coal burning per each CPP, which is about $7.3 \times 10^{12} g/y$, the ash content (12%), fly ash release (80% of the total ash content), as well their ^{226}Ra specific activities, it is deduced that 42 Ci/y ^{226}Ra is escaping as fine dispersion (finest particulate form or maybe in gaseous form) from a stack of CPP.

4. Hazards from the Radioactivity escaping from CPP.

a) Hazards from the fly ash. In the site of maximum, for u=1 m/sec, we can find a ^{226}Ra concentration in the air about $2.3 \times 10^{-9} \mu Ci/cm^3$. Also, if we regard all the radio-elements of the uranium series in the grains of fly ash, as well preferable wind direction, we shall have again that the total radioactive concentration, arising from fly ash, will be 20 times lower than the MPC ($\sim 10^{-11} \mu Ci/cm^3$).

b) Hazards from fine dispersion. In Fig.4 typical curves are presented that they give the maximum concentration, x_{max}, in several distances from the stack, horizontally, on its base, in the main direction of the local wind blown, as a function of the height of the stack. Each curve corresponds to one Pasquill condition (13), from A (extremely unstable) to F (extremely stable). For our calculations (14) a wind speed of 1 m/sec was used. In the case of a radioactive release (CPP Kardia-Ptolemais) Q_o=42 Ci/y of ^{226}Ra, it is found that x_{max} is about 2.10^{-11} $\mu Ci/cm^3$ (Pasquill condition "A"). This is the same value as the MPC in the air. The above consideration had as presupposition a stable and singular wind direction. This hypothesis is non-realistic and therefore the maximum concentration of ^{226}Ra will be lower than the one estimated. However, other toxic radionuclides escape from the stack of CPP such as: ^{238}U, ^{210}Pb, etc., which leads to hazard in other (bad) directions.

For the most common Pasquill condition F (extremely stable) the whole body man dose exposure at the distance of 400 m from a stack 120 m in height and u=1 m/sec, was calculated. Using the philosophy of calculations as is given by Coben et al (15), it was found that for 42 Ci/y of ^{226}Ra this is about 0.5 man-rem. This must be compared with 3 man-rem which is the permissible whole body man dose exposure.

5. Hazards from use of fly ash as substitute of cement in concrete.

a) Hazards from wall radioactivity. The cement in concrete is about 30% and the proposed substitution is between 20 to 40% (16). Let us assume 30% fly ashes in the cement. At a distance of one meter across a "doped" wall of "infinite" area and of "infinite" depth, the calculations give an annual dose of the order of 100 mrem. The assumption made in the above estimation was strong, i.e. 24 h permanent living in the active room. So the 100 mrem must be considered as an overestimate.

b. Hazards from diffusion of Radon through concrete.
Radon is diffused across a doped concrete wall. Culot et al (17) studied extensively the problem. Using the philosophy of Culot et al, we found that in a room of dimensions 10x10x4 m^3 the concentration of Radon in the air will be about 10^{-9} $\mu Ci/cm^3$. For the above estimation we used a concrete porosity of 5% and a wall thickness of 20 cm. The estimated concentration of Radon is about two orders of magnitude lower than that of the MPC of Radon in the air, which is about $10^{-9}\mu Ci/cm^3$. However, if we used a 25% porosity, the Radon concentration will be an order of magnitude higher.

6. Conclusions. From the experimental data and discussion we can conclude: a) The major component of the risk is due to the fine structure of release. b) Fly ash would be avoided for use in doped concrete for habitation. It would be used for other concrete constructions "en plain air".

It is expected that the coal use for electric power generation to be increased by a factor almost 10 in the next 10 years. Several estimations, i.e. Klein et al (18), Bertine and Goldberg (3) and Ondov et al (19), are made on the atmospheric releases of various potentially toxic elements from large CPP. In them, the release of radioactive elements must be added. All of them must be taken seriously in consideration in the designing of new large CPP's, since as the present work demonstrates, could have concentrations of radioactivity close,as well greater, to MPC.

The new data indicate, as Eisenbud and Petrow (1), Kolb (20) and Aurand (21) estimate, that the Coal Power Plants discharge relatively larger quantities of radioactive materials into the atmosphere than Nuclear Power Plants, of comparable size, during their "normal" Operation.

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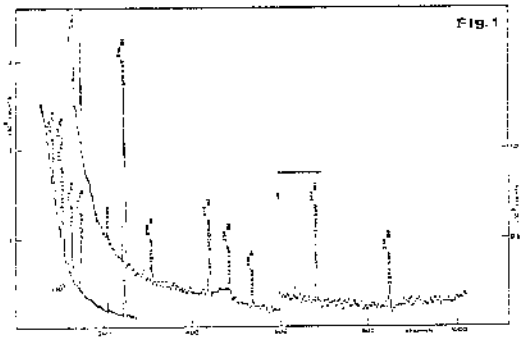
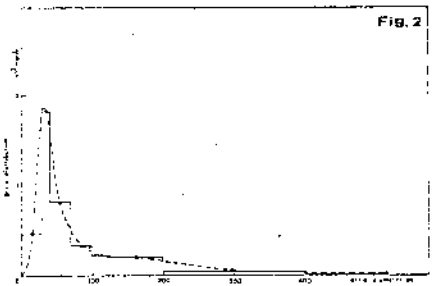
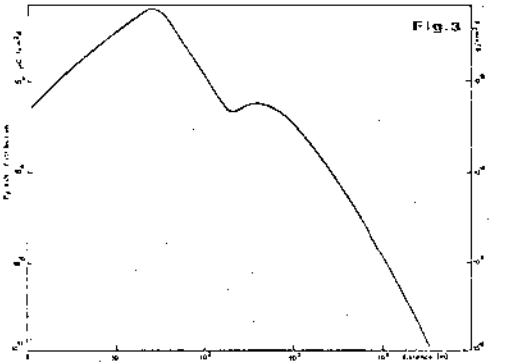


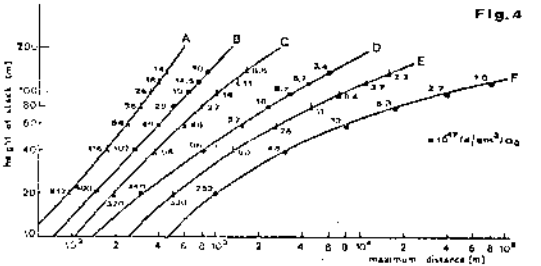
1.Typical "net" gamma spectrum of fly ash from Kardia Ptolemais.The background is subtracted.



2.Histogram of distribution of particle size of fly ash (g/u) as a function of the mean diameter (μ) of grains.



3.Distribution of fly ash and its radioactivity depositing in one direction (wind blown) from the stack released as a function of the distance downwind.



4.Typical curves for various Pasquill conditions A to F. The numbers on the curves show the maximum concentration of Ra-226 (μCi/m³) at several distances downwind from the stack released as a function of the height of the stack. Q_0 is the radioactive release in μCi/d.

FLEXIBILITY IN RADIATION PROTECTION LEGISLATION - THE UK APPROACH

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FLEXIBILITY OR RIGIDITY?

It is usually assumed that flexibility in radiation protection legislation is a desirable objective and that there is no need to argue the case. However, the opposite view, on a number of occasions, has been expressed. For example, managements have suggested that if legislation is clear and unequivocal with no room for opinion then they can get on with their task of management secure in the thought that they cannot be criticised by workers, trade unions or enforcing authorities. In the same way, trade union officials have expressed the view that it is easier for them if the legislation is precisely framed and hence departures from its standards are obvious. There is certainly a view from some parts of the enforcing authority that legislation should be specific and absolute in its requirements. This will, it is claimed, lead to the most effective enforcement using minimal numbers of specialist inspectors.

Flexibility is taken to mean that the objectives of the legislation are clearly spelt out but the means whereby those objectives are to be achieved are left open, or qualified by terms such as "where reasonably practicable" or a variety of options are offered.

The pressure for flexibility comes, in the first instance, from the knowledgeable to whom a variety of options is appealing in its mind stretching potential, to whom a cost benefit analysis is a common technique and to whom the fact that a means to achieve the objective might not be specified is seen as a challenge to their ingenuity. The pressure from this group is echoed, for quite different reasons, by other groups. The enforcing authorities, desirous of simple legislation requiring minimal interpretation, are attracted by the concept of simple regulations specifying objectives and in a field as wide as radiation protection this can lead to a very effective coverage. We see trade unions and managements as less convinced but gradually appreciating the merits of a flexible system and becoming increasingly confident in their ability to operate within it and that flexibility can lead to improved safety standards at marginally extra cost.

PARTICIPATION AND CONSULTATION

In 1974 a substantial new piece of safety legislation was enacted in the UK; it was called the Health and Safety at Work Act (1) and apart from creating a new enforcing authority, the Health and Safety Executive (HSE), it laid the foundations for a new framework of specific safety legislation. The Act allows that Regulations may be approved by the UK Parliament and also allows the preparation of Codes of Practice which, having been approved by the Health and Safety Commission (HSC), becomes official guidance on the means which should be followed in order to meet the objectives specified in the Regulations. Regulations and approved Codes of

Practice may be supported by Guidance Notes which have no legal status but nevertheless give detailed advice on the interpretation and the objectives of codes and regulations.

In order for a Code of Practice to enjoy a special legal status it must first be approved by the HSC. The HSC is a representative body, representing workers' organisations, management organisations and local government authorities. In addition to its role of approving codes it also directs the general policy of the HSE. The Health and Safety at Work Act requires that persons who might be affected by proposed legislation are consulted. New radiological protection legislation is necessary because of the fragmented nature of the existing legislation, the need to conform with the Euratom Directive (3) binding on the UK as a member state of the European Community and because of the new recommendations contained in ICRP 26. Since there is a substantial body of informed opinion we sought to involve that opinion in discussion and debate even before the stage of formal consultation. In the participation phase, some 17 working parties were organised, each dealing with a single aspect of the proposed regulatory package. In this way some 200 experts in varied fields of radiological protection were brought together and participated with the HSE in establishing the general content of the legislative material which was subsequently published as a Consultative Document (2). The extensive participation of experts in the formulation of the Document did not totally inhibit all comment on its contents. However, the comment was quite limited, confined to a few areas where those commenting either had proposals for change or where they found the proposals to lack clarity.

REQUIREMENTS FOR NOTIFICATION OF USE

Like many of the other requirements of the proposed regulations those for notification to the enforcing authority of the "production, processing, handling, use, storage, transport and disposal of natural and artificial radioactive substances and of any other activity which involves a hazard arising from ionising radiations" stems from provisions in the Euratom Directive.

Many exemptions are possible. In particular there may be a small quantities exemption based upon the classification of radionuclides into toxicity groups. These groupings reflecting early IAEA work,(4) needed updating to take account of new data on Annual Limits of Intake, and of the use of SI Units. Other exemptions for particular types of apparatus will undoubtedly be necessary and a procedure for type testing such pieces of equipment will be devised for use in the UK.

It is very tempting to set up a notification of use system in which substantial amounts of information are called for and in which constant updating is mandatory. It needed no consultation for the enforcing authority to realise that in a country such as the UK where there was a widespread use of ionising radiations and substantial turnover of users, that the administrative burden of such a system would be considerable. Analysis of the reasons why an enforcing authority needs to know of use and discussions of what action might be taken by enforcing authorities when they do know about use has led us to the conclusion that a general notification containing only a superficial description of the work to be under-

taken is all that can properly be expected. It has become clear that the inspections of the enforcing authority both in terms of depth and frequency should be based on conditions as found and that notification merely acts as a trigger to cause an inspector to visit the premises. Thus, notification can take a very simple form and can minimise the administrative burden on occupiers of premises.

CRITERIA FOR CONTROLLED AREAS

Controlled areas have always played an important role in UK legislation. The reason is the route by which a Category A Radiation worker is defined is to consider his access to such areas. The objective can be simply stated, that is to define an area which encompasses all those workers liable to exceed 3/10ths of the Annual Dose limit. It was the means whereby to reach that objective that caused debate and discussion. It was felt by many that a simple boundary condition such as 7.5 microsieverts per hour, as read on a dose rate meter, was highly enforceable, easily determined and demonstrably in accord with the objective. However, such a boundary condition can give rise to larger controlled areas than are necessary to meet the objective, particularly obvious if the radiation emission is intermittent. Introduction of a flexible approach clearly requires some expert consideration of the possible doses that might be received by persons working in the controlled areas. This is, of necessity, a time consuming exercise requiring a professionalism which not every employer is prepared to pay for. The solution for the proposed UK legislation is that a Code of Practice will give alternative guidance as to the ways by which the objective of defining the controlled area might be achieved. The first procedure will be based on demarcating areas where the dose rates exceed 7.5 microsieverts per hour averaged over any one minute. The alternative procedure involves the Radiation Protection Adviser (see below) considering the work situation and using his professional judgement to determine the boundaries of the area encompassing all workers likely to exceed 15 millisieverts (mSv) per year. Even then to afford some safeguards it will be necessary to give him some guidelines; the most important of which is that in determining areas he should not take into account occupancy times less than 40 hours per week and he should assume an optimum (but not maximum conceivable) operation of the facility. This accords with ICRP 26 paras 163-6. If all entrants to the area were then to be treated as radiation workers this would lead to a large number of such workers being designated who would, in fact, not exceed 15 mSv per year. Again, flexibility is to be introduced by the Radiation Protection Adviser exercising his professional judgement. It is proposed that he prepare house rules or schemes of work in which he clearly identifies individuals or groups of workers who, albeit that they may need to enter specified controlled areas from time to time, will not, because of the nature of their work and their occupancy time, receive more than 15 mSv per year.

THE RADIATION PROTECTION ADVISER (RPA)

The requirements for the appointment and selection of an RPA in the proposed legislation are that the employer should make an appointment of a suitable person if his operations are such that workers

enter a controlled area - in this way users of essentially harmless sources are exempt. The requirement would go on to state that such appointments must be notified to the enforcing authority, who in turn must "recognise the capacity to act" of the individual (or corporate body) appointed as an adviser. It is not the intention of the enforcing authority to enter into an elaborate scrutiny and certification scheme for advisers for the reason that it is seen as proper that in the first instance employers must consider and exercise their own judgements as to who they should employ. To help them in making such decisions a number of professional bodies in the radiation protection field are setting up certification schemes whereby the professional competence of potential advisers is examined and certified by a committee of the professional society concerned. In this way it seems unlikely that the enforcing authority will ever feel it necessary to reject an application for recognition. The conceptual basis for the use of a RPA as an expert advising users comes from the Euratom Directive. However, the proposed UK Regulations exploit the concept in the interests of flexibility and self inspection. Thus, the RPA will have tasks as varied as definition of areas, advice on selection of workers as radiation workers, checking and calibration of instruments, advice on training and instruction, etc. Particularly he is envisaged as having substantial responsibilities to advise the employer on the effectiveness of the application of the principle of ALARA (as low as reasonably achievable). The application of the principle is seen as a continuous exercise before the work activity begins at any prospective level of individual dose or collective dose and after the event as a retrospective examination of whether or not the procedures did, in fact, result in doses being optimised. Serious consideration is being given at the time of writing to setting specific levels of individual exposures at which the RPA carries out formalised retrospective consideration of the effectiveness of the ALARA principle.

In the UK, workers' involvement in safety matters has found expression by the statutory appointment of workers' safety representatives and the creation of joint Worker/Management safety committees. It is seen as a vital part of the operations of the RPA that his advice to the employer is also transmitted to the safety representative and the safety committee so that they too may express a view on such advice and identify with it.

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OCCUPATIONAL RADIATION PROTECTION LEGISLATION IN ISRAEL

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A committee of experts appointed by the Minister of Labour and Social Affairs has proposed a comprehensive draft regulation, concerning the legal aspects of occupational radiation protection in Israel. The main sections of the proposed regulation are:

1. Control - general
2. Personal monitoring
3. Medical supervision

CONTROL - GENERAL

The first section of the proposed regulation sets forth guidelines for control in facilities where workers handle radioactive materials or radiation equipment. The managers of such places should take the following steps:

a) Nominate a radiation protection officer
b) Advise the inspector of the Ministry of Labour and Social Affairs of all unusual occurrences
c) Procure the equipment necessary for shielding and monitoring of radiation.
d) Restrict access to hazardous areas
e) Ensure compliance with the regulations for the safe operation of the facility
f) Train the radiation workers
g) Advise the officials of radiation exposure in excess of the maximum recommended doses.

Table 1 shows the maximum radiation doses recommended for normal operation.

During special jobs, which cannot be performed under the limitations specified for normal operation, radiation workers may be exposed to twice the annual recommended radiation dose, or - once in a lifetime- to the annual dose multiplied by a factor of five. After such an exposure, further exposure of the worker should be avoided if the integrated dose exceeds (N-18) x the annual dose limit, where N is the age of the radiation worker. The doses given in the table are actual doses, and should not be considered as design base doses for normal operation. Design base doses should be kept as low as reasonably achievable. The annual limits do not include medical exposure. In case of a simultaneous exposure of several tissues, the calculated overall equivalent of the whole body dose should not

exceed the limit indicated for the whole body. The doses should be reduced by a factor of 10 for 16 to 18 year-olds, whose exposure should be allowed only if it is connected to professional training, for which a special permit should be required.

TABLE 1. Maximum recommended doses for radiation workers during normal operation

Tissue	Dose to individual workers (Rem/y)*
Whole body	5
Gonads	20
Breast	25
Thyroid	50
Bone	50
Bone marrow	25
Lungs	25
Eyes	30
Other single organs	50

*1 Rem = 10 millisievert

PERSONAL MONITORING

The second section deals with the monitoring regulations for radiation workers who may be exposed to doses in excess of 500 mRem/y. The regulations stipulate that these workers should:

a) wear radiation badges, to measure external exposure
b) be checked for internal radioactive contamination
c) report on all employment in which they may be exposed to additional radiation.

MEDICAL SUPERVISION

Medical check-ups are required for all applicants for work which involves radiation. Also, radiation workers are required to undergo routine periodical examinations, the type of the examination to be determined by the type of work performed. Radiation workers must also be examined following overexposure or accidents.

A. Routine examinations

Compulsory routine examinations should include the following:

a) Complete clinical check-up
b) General urine analysis
c) Blood count: hemoglobin, white count, differential and thrombocyte count
d) Complete anamnesis, including medical and occupational history.

B. Special examinations

Special examinations of radiation sensitive organs and tissues should be performed according to the type of work and circumstances. These examinations include:

a) Blood analysis, including bleeding and coagulation time, in case of whole body exposure to radiation
b) Skin examination, in case of external exposure
c) Periodical eye examinations,usually once in five years,or once in three years for X-ray machine operators and in special cases, such as exposure to neutrons
d) Chest X-ray, investigation of the performance of the lungs, liver and kidneys, and analysis of internal contamination, in case of internal exposure and contamination.

The results of the medical check-ups should be recorded in a health report, which should be kept for 30 years by the medical authorities. The main results are also recorded on a personal health card. The following information should be recorded on the card:

a) Date and purpose of check-up
b) Any positive finding of the medical check-up or laboratory tests
c) Occupational disease or effects detected
d) Decision on whether the worker is medically fit to work with radiation
e) Date of next medical check-up
f) Name and signature of physician.

ADDITIONAL RECOMMENDATIONS

In addition to the draft regulations, the committee proposed several codes of practice encompassing the principles of radiation protection, compliance, inspection and licensing.

A series of recommendations were also made by the committee, to the Minister of Labour and Social Affairs, indicating the need to:

a) Nominate a national advisory committee for ionizing radiation, safety and hygiene
b) Establish a central medical authority, to deal with emergency cases of high exposure to radiation
c) Publish safety rules for work with ionizing radiation
d) Spell out the training and experience required of persons who install and maintain ionizing radiation machines
e) Establish curricula for the training and instruction of radiation workers, according to the type of work
f) Spell out standards for ionizing radiation machines and radiation measurement instruments.

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In addition to the authors, the following members of the committee contributed to the draft regulation: Dr. J. Caftori, Dr. M. Friedman, Prof. Z. Fuchs, Mr. G. Nativ, Mr. M. Naveh, Dr. M. Ronen, Mr. N. Rosenthal, Mr. K. Schniedor and Mr. J. Talmon. Their contribution is gratefully acknowledged.

LEGAL PROVISIONS CONCERNING THE HANDLING AND DISPOSAL OF RADIOACTIVE WASTE IN INTERNATIONAL AND NATIONAL LAW

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One of the current main problems of the peaceful use of nuclear energy and of radiation protection is the handling and disposal of radioactive waste. The solution of this problem is not only a technical and economic task but also a mission for the legislative bodies, international and national, to provide by legal instruments that damage to the general public and to radiation workers does not occur by the harmful effect of nuclear waste materials and that any danger caused by these substances should be compensated. This paper gives a short survey on the situation of international legislation (I) and of national legislation in countries where nuclear installations are in operation (II) concerning the radioactive waste handling and disposal (1).

I. INTERNATIONAL LAW

Until now there is no special international multilateral convention which governs exclusively the handling and disposal of radioactive waste. Nevertheless we find special rules on the disposal of nuclear waste in a number of conventions on the protection of the marine environment and of the high sea against pollutions (2):

- Convention on the High Sea (Geneva Convention) of April 29, 1958, esp. article 25 (3);
- Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (London Convention) of December 29, 1972 (4);
- Convention on the Protection of the Marine Environment of the Baltic Sea Area (Helsinki Convention) of March 22, 1974 (5);
- Convention for the Prevention of Marine Pollution from Land-based Sources (Paris Convention) of June 4, 1974 (6);
- Convention for the Protection of the Mediterranean Sea Against Pollution (Barcellone Convention) with Protocol for the Prevention of Pollution of the Mediterranean Sea by Dumping from Ships and Aircraft of February 16, 1976 (7).

In addition and in implementation of the mentioned London Convention of 1972 the International Atomic Energy Agency (IAEA) has published in 1974 Provisional definition and recommendations concerning radioactive wastes and other radioactive matter referred to in Annexes I and II to that convention (8) which has been revised in 1978 (9).

Taking in consideration the international conventions on the protection of the marine environment, especially the London Convention of 1972, the Organization for Economic Cooperation and Development (OECD) has set up within its Nuclear Energy Agency (NEA) a multilateral consultation and surveillance mechanism for the sea-dumping of radioactive waste by Decision of the OECD-Council of July 22, 1977 (10). In addition to this decision NEA has published in April 1979 Recommended Operational Procedures for Sea-Dumping of Radioactive Waste and Guidelines for Sea-Dumping Packages of Radioactive Waste.

It should be mentioned that the International Atomic Energy Agency in its Safety Series has enacted recommendations "Radioactive Waste Disposal into the Sea" and "Methods of Surveying and Monitoring Marine Radioactivity" (11).

Concerning the Antarctic Region the disposal of radioactive waste materials is absolutely prohibited by article V of the Antarctic-Treaty of December 1, 1959 (12). In 1975 the parties of that Treaty have recommended again that their governments continue to exert appropriate efforts to the end that no one disposes of nuclear waste in that Antarctic Treaty Area.

Sometimes one may read of proposals to dispose radioactive waste into the outer space. Until now we have no special rules - neither international nor national - on radioactive waste disposal in such a way. The Treaty on Principles Governing the Activities of States in the Exploration and Use of Outer Space Including the Moon and Other Celestial Bodies of January 27, 1967 (13) does not mention that problem. By art. IX of that Treaty in the exploration and use of outer space the State Parties shall be guided by the principle of co-operation and mutual assistance; they are obliged to conduct all their activities in outer space with due regard to the corresponding interests of all other Parties. There is until now no absolute prohibition of the radioactive waste disposal in outer space, but the States are responsible for such activities (article VI of the Outer Space Treaty; and Convention on International Liability for Damage Caused by Space Objects of March 29, 1972).

"Radioactive products and waste" are also subject of the international conventions on third party liability in the field of nuclear energy. By the Paris Convention of 1960 (art. 3) (14) and the Vienna Convention of 1963 (art. II) (15) the operator of a nuclear installation shall be liable in accordance with the provisions of these conventions for nuclear damage upon prove that the damage has been caused by a nuclear incident involving nuclear fuel or radioactive products or waste in his installation or coming from it.

For the nine Member States of the European Atomic Energy Community (EURATOM) the Treaty establishing that Community (16) contains a special provision (art. 37) that

each Member State shall provide the Commission with such general data relating to any plan for the disposal of radioactive waste in whatever form as will make it possible to determine whether the implementation of such plan is liable to result in the radioactive contamination of the water, soil or airspace of another Member State. By art. 3 of the EURATOM-Basic Safety Standards of June 1 1976 (17) each Member State shall make the reporting of the disposal of natural and artificial radioactive substances compulsory; each Member State may decide that such disposal activities shall be subject to prior authorization by the competent authority.

In addition to its recommendations concerning the waste disposal into the sea the IAEA has published some other guidelines for waste disposal (Safe handling of radionuclides, 1973 edition, disposal of radioactive wastes into fresh water; the management of radioactive wastes produced by radioisotope users and technical addendum; radioactive waste disposal into the ground; basic factors for the treatment and disposal of radioactive wastes; management of radioactive wastes at nuclear power plants; disposal of radioactive wastes into rivers, lakes and estuaries; management of wastes from the mining and milling of uranium and thorium ores) (18).

Occasionally waste disposal is subject of bilateral treaties, for instance the Technical Exchange and Co-operation Arrangement between the USAEC in the Federal Ministry for Research and Technology of F.R.G. of December 20, 1974 (19).

II. NATIONAL LEGISLATION

Provisions on the handling and disposal of radioactive waste have been enacted in many countries, particularly during the last years. It is not possible to give here a comprehensive survey, but the following legal provisions should be mentioned:

1. Austria: Radiation Protection Act, June 11, 1969; Radiation Protection Decree, Jan. 12, 1972;
2. Belgium: Radiation Protection Regulations, Febr. 28, 1963 (with amendments) (Sec. 33-37);
3. Denmark: Radiation Protection Regulations, Nov. 20, 1975, (Sec. 8);
4. France: Décret no. 66-450, June 20, 1966; décret no. 63-1228; Dec. 11, 1963, and décret no. 73-405, March 27, 1973; Arrêté, Nov. 7, 1979;
5. Germany, F.R.: Atomic Energy Act 1959/1976 (Sec. 9a); Radiation Protection Ordinance, Oct. 13, 1976 (Sec. 47);
6. Germany, D.R.: Radiation Protection Ordinance and First Executive Order, November 26, 1969; Guidelines on the centralized management of radioactive waste, March 28, 1974;
7. Israel: The Supervision of Supplies and Services (Construction and Operation of Nuclear Reactors) Order,

No. 5735, Sept. 27, 1974;
8. Italy: Act No. 1860 of the Peaceful Use of Nuclear Energy, Dec. 31, 1962; Radiation Protection Regulations No. 185, Febr. 13, 1964 (Sec. 104-107);
9. Luxembourg: Radiation Protection Act, March 25, 1963; Radiation Protection Regulations, Febr. 8, 1967;
10. Netherlands: Decree on nuclear installations, fuel and ores, Sept. 4, 1969; Decree on radioactive substances, Sept. 10, 1969;
11. Switzerland: Atomic Energy Act, Dec. 23, 1959; Revision of the Atomic Energy Act, Oct. 6, 1978; Radiation Protection Ordinance, June 30, 1976; Decree on the compilation and delivery of radioactive wastes, March 18, 1977;
12. Spain: Nuclear Energy Act, April 29, 1964;
13. United Kingdom: Radioactive Substances Act, June 2, 1960 (Sec. 6-10);
14. USA: USNRC Regulations, 10 CFR Part 20 (Sec. 20.301-20.305).

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11. IAEA Safety Series No. 5 (1961) and 11 (1965).
12. UNTS 402, p. 71.
13. UNTS 610, p. 205.
14. Bischof/Goldschmidt/Greulich, Internationale Atomhaftungskonventionen, Textsammlung, Göttingen 1964, p. 131.
15. Bischof/Goldschmidt/Greulich (Fn. 14), p. 13.
16. United Kingdom Treaty Series No. 17 (1979).
17. OJ of the European Communities 1976, No. 187/1.
18. IAEA-Safety Series Nos. 1, 10, 12, 15, 19, 24, 28, 36, 44.
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ENVIRONMENTAL MONITORING AND DEEP OCEAN DISPOSAL OF PACKAGED RADIOACTIVE WASTE

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INTRODUCTION

A basic tenet of the philosophy which underlies radioactive waste disposal control into the marine environment is that of environmental monitoring. Current monitoring principles have recently been discussed by Mitchell (1) with illustration of their application to controlled releases of radioactive effluents to estuarine and coastal waters of the U.K. This paper sets out to discuss environmental monitoring philosophy in the context of dumping of packaged waste in the deep ocean and how far it may be reasonable to apply it in practice.

THE AIMS AND OBJECTIVES OF ENVIRONMENTAL MONITORING: SOME BASIC PRINCIPLES

Both the ICRP (2) and the IAEA (3) have devoted specific publications to the topic of environmental monitoring and U.K. practices are consistent with them. As with any operation associated with the management of radioactive waste, environmental monitoring should be in accord with the fundamental dose limitation principles of the ICRP and the principle of optimization is of particular importance in this respect. The IAEA in its recommendations to the London Dumping Convention (4) recognizes a need for environmental monitoring but adds an important rider 'to the extent feasible and meaningful'. The ICRP lists three objectives for environmental monitoring.

(a) Assessment of the actual or potential exposure of man to radioactive materials or radiation present in his environment, or the estimation of the probable upper limits of such exposure.

(b) Scientific investigation, sometimes related to the assessment of exposures, sometimes to other objectives.

(c) Improved public relations.

To these the IAEA has added:

(i) assessment of the adequacy of controls on the release of radioactive materials to the environment;

(ii) demonstration of compliance with the applicable regulations, environmental standards, and other operational limits; and

(iii) the possible detection of any long-term changes or trends in the environment resulting from the operation.

The estimation of radiation exposure

The fundamental objective of environmental monitoring, as the term is used in the U.K., is to facilitate or otherwise provide for the *direct* estimation of radiation exposure. Reflecting the emphasis of U.K. national waste disposal policy, which is primarily on limitation of radiation exposure to the public as opposed to that of effects on environmental resources, environmental monitoring has come to be associated mainly with measurements needed to assess doses to which human populations are exposed. The justification for this attitude is found in the realization that the potential risk to environmental resources is minor provided that exacting standards for control of public radiation exposure are met and maintained, i.e. those recommended by the ICRP.

Monitoring and research

There is still much confusion between monitoring and research, possibly because they are frequently practised as mutually supporting activities. The objective of estimation of radiation exposure has sometimes been met by utilization of data which have accrued from research programmes, replacing a need which would otherwise have had to be met by mounting monitoring programmes specifically for that purpose. Conversely, environmental monitoring programmes themselves have sometimes generated information of value to research and it is in this kind of situation where there may appear to be some confusion of purpose as between monitoring and research. Whilst in a few marine situations radiation exposure can be estimated from analysis of environmental materials, it is not possible to do this where contamination is below detection limits. In such situations monitoring and research have another interface, for mathematical modelling provides a means of indicating what the levels of radioactivity may be in the environment in relation to given rates of input and hence provides data for the evaluation of levels of dose. Research in support of this kind of modelling activity is especially important in connection with deep sea disposal of packaged radioactive waste.

Monitoring and control measures

As practised in the U.K., environmental monitoring has sometimes provided a means of checking adequacy of control measures, such as treatment plant or filtration systems, relating to the on-site management of radioactive waste. This has certainly been applied in some coastal situations related to the control of liquid radioactive waste discharges but, just as this is not always a practicable proposition there, it would be misleading to consider that such objectives could necessarily be met in regard to deep sea dumping.

Public information

A popular view in some circles is that monitoring should also be done on occasions for what is termed 'public relations purposes'. Whilst the need to provide the public with appropriate information

and data is readily acknowledged and met, it is considered that the conduct of monitoring programmes for which there is no justifiable radiological or scientific need in relation to estimates of human radiation exposure is not a suitable objective. P.R. monitoring must not be allowed to become an end in itself and require the production of data for the sake of being able to state that some monitoring has been done. Such programmes would be misleading, and subject to severe criticism by the scientific community as a conscious effort to mislead the public and allay public concern at a cost out of proportion to the need. Data arising from programmes designed to fulfil the basic aim of providing a sound basis for estimates of radiation exposure should also be sufficient to answer public information needs.

ENVIRONMENTAL MONITORING IN PRACTICE

Compared with disposals of liquid wastes into coastal waters, disposals of packaged waste into the deep sea pose special problems for those involved in environmental monitoring. The capacity of the receiving environment, its remoteness from pathways back to man, coupled with the relative biological and physical unavailability of the radioactivity in the waste means that, curie-for-curie, there is must less chance of being able to measure activity in the critical materials from deep sea disposals. Whilst waste packages are designed to ensure that their active contents are at the very least delivered to the deep ocean bed it is clear that in general they will have a much longer life and in most cases the activity will be released only very slowly into the water.

The major disposals of liquid wastes to coastal waters are characterized by the ease with which environmental monitoring yields positive evidence of the disposal. In such labelled environments discharge rate can be correlated with radioactivity in environmental materials and from this with radiation exposure to man. In contrast, and whilst there is evidence to show that in at least certain circumstances activity from dumped waste may be detected on sediment very close by packages, it cannot be detected in pathways critical to man and monitoring along the lines of coastal disposals would not therefore be meaningful.

Faced with such a situation it is necessary to find an alternative to environmental monitoring, at least to an extent necessary to meet the fundamental objective of monitoring, that of assessing human radiation exposure. Situations where levels of activity attributable to a particular disposal are below limits of detection are not unique to deep sea disposal; it is typical of a majority of the disposals at coastal sites and two options are open to us.

The first option is to make an upper limit estimate based on a judgement of analytical detection limits. As such it provides only a very crude answer and whilst this will be sufficient to show that radiation exposure is within prescribed limits it has little scientific value and serves no other purpose.

The second option is to compute the dose by mathematical modelling. For a system on an oceanic scale this is a considerable undertaking and many of the factors involved are not known with any precision; neither are oceanic processes well understood. Nevertheless it is possible to model the system using pessimistic values of

the necessary parameters, such that upper limit values are produced which are more accurate than those derived from analytical detection limits. The overall oceanographic/radiological model is divided into several parts, viz, release of activity into the water, its dispersion and transport, and uptake into critical pathways. Simplifying assumptions are made, for example that no removal by sediment occurs to reduce the availability of activity to biological pathways, provided that they do not underestimate the dose received. Most of the values of concentration factor needed are reasonably realistic, as would be the consumption rates/occupancy factors used if the pathways were effective now. A fundamental problem to sea dumping assessments is the long delay between dumping and the arrival of dose to man. For the purpose of dose assessment the assumption is made of prompt release after dumping has taken place, a maximizing assumption which leads to exaggerating the resulting dose from the shorter-lived radionuclides. For the present, it is only possible to calculate the dose at equilibrium from continued dumping over very long periods of time and the results are therefore likely to be gross overestimates of the true dose. Nevertheless, work is continuing, in terms both of better mathematical models to predict dose to man and research into oceanographic and biological transfer processes to provide better data and improve the accuracy of the models. Faced with the inability of direct monitoring to provide data by which direct estimates of dose to man can be made, resources can be used to greater effect by devoting them to modelling and oceanic research.

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ABSORPTION OF DOSE RATE IN GREAT BITUMEN BLOCKS - EXPERIMENTAL DETECTION

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GENERAL

Development and widespread application of atomic energy involves a lot of problems, concerning the handling of increasing radioactive waste amounts. For storing and disposing of these big amounts of radioactive trashes several methods have been developed.

Concerning the fact that storing and disposing prices inside an activity group (medium level or low level waste) are nearly independent in a wide range from the activity of the immobilized waste, a volume reduction is desirable and necessary, regarding the principle of cost minimization.

Commonly LLW and MLW are reduced in volume by combustion (for burnable wastes) or by compaction. The residues of these volume reduction methods are conditioned partly by pouring over or mixing with cements or bitumen. Aime of these procedures is the immobilization of the radioactivity against leaching by ground water after a possible water contact. Both methods- cementation and bituminization - are showing advantages and disadvantages. Whereas cements are highly resistent against radiation damage, bitumen shows better properties against water leaching, for example.

Numerical calculations of dose rates occuring inside non artifical waste mixtures often showed great differences to the real values reached, especially regarding beta-doses. Using the bituminization technique as conditioning method, therefore different little "bitumen elements" will suffer a high dose, while others will remain nearly unirradiated. These effects are amplified when the amount of embedded waste in bitumen increases to a very high extent, what is reachable in principle by sedimentation technique (1,2).In this case the specific activity of the immobilised waste increases, the necessary storing place decreases. Presently a specific activity of 1 Ci/l (^{137}Cs + ^{90}Sr) is the limit for bitumen embedding techniques, accepted by many authorities. This value is on the very safe side of the activity limit, above which decomposition of bitumen is initiated, although there is some early work in higher activities enclosed inside bitumen (3). In any case the limits for loading waste into a bitumen matrix are defined by two parameters: chemical stability of the bitumen in a field of radiation, and heat generation and transfer out of the bitumen block.

Facing the goal of as high waste loading as possible inside a bitumen matrix we aimed to elaborate a method for detecting the irradiation affecting the bitumen matrix under actual conditions.

RADIATION SOURCE MATERIAL

To gain short exposition times during the experiments, on the one hand for getting the results after reasonable time, on the other to meet the endeavours in increasing the possible specific activity of the enclosed waste, we decided to use irradiated high temperature coated particle fuel as radiation source. Tc get reproducible conditions coated particles (cp) out of one run were taken. The showed specific activity of 75,46 mCi ^{137}Cs and 15,32 mCi ^{134}Cs and 90mCi ^{90}Sr per gramm particle material. The diameter of these particles was 800 µ in average, their weight 1,25 to 1,38 mg, the density 1,5g/cm^3.The bulk density was estimated with 0,6.

PERFORMANCE OF EXPERIMENTS

The experimental work was performed inside hot cells. In the first experiments we used film dosimeters as detecting device. The dosimeter badges were placed inside the hot cell and afterwards a weighed amount of cp was positioned overneath the dosimeter badge. The cp were fixed in a tub-shaped container on a thin Al-foil in a 2-3 mm layer (about 4 cp one upon another). After few irradiations we recognized that the possible exposure times were too short for reproducable handling operation with the remote controled equipment of a hot cell. The results of a 10 sec irradiation differed for ± 200 %. Therefore we changed over to TLD.

The arrangement of the experiments was the same like that with the film dosimeters. Although the exposition time was now long enough, great difficulties arose from the effect not to reach a sufficient plane layer of cp. All efforts to level the particle layer were successless. Also experiments with an amount of cp for a one-particle-layer only didn't succeded.

So we changed the philosophy of the experiments. We postulated, that not a little grain of waste is surrounded by bitumen, but a "differential element" of bitumen is enclosed by an infinite thick mass of waste.

In that case, the γ-intensity depends on the distance source - bitumen, so that a certain thickness of the particle layers enclosing the bitumen element, the γ-doserate will become constant. The ß-dose belongs mainly to the waste particles, situated next to the bitumen element, while the ß-irradiation of the outer grain-layers will be mainly absorbed by the inner waste layers, surrounding the bitumen element and will not affect the bitumen anymore.

For these experiments we used glass dosimeters, because we had no troubles with light exposition. Also the cleaning of the dosimeter surface before the evaluation was easier as by working with TLD.

To determine the gamma doserate of the used cp mixture we protected the ß-absorber of the glass dosimeters with a thin rubber layer and attached it inside a 250 ml beaker. Afterwards we poured the cp into it. Irradiation time was

10 minutes. Then the dosimeter was pulled out of the particles and brought out of the hot cell. The possible error in timing was about one second or less than 1 %.

Outside the hot cell the rubber was removed and after a conditioning time of one day the glass dosimeters were evaluated. The gamma doserate we found was about 1050 rad in ten minutes, with an accuracy better than ± 5 %.

For the determination of the ß + γ doserate we removed the ß-absorber from the dosimeters and irradiated them in direct contact with the fuel particles. For this purpose we filled half of the particle amount into the beaker, then we dropped the dosimeter into the glass too and immediately afterwards we poured the rest of the cp over the dosimeter. After an irradiation time of 5 min the whole content of the beaker was poured over a sieve, the fuel particles fell through and the dosimeter remained in the sieve. With a pincette we transferred the dosimeters into a little container and brought it out of the hot cell. The possible error in timing was about 3 to 6 seconds or 1-2 %. After a run of 10 irradiations all the dosimeters were decontaminated, dried and evaluated. The results were as accurate as that from the determination of the gamma doserate. The ß + γ dose we found, was about 4400 rad in ten minutes. That means that the ß-dose alone is about 3350 rad per 10 minutes.

NUMERICAL EVALUATION

In a speric shell of a thickness dr the activity is distributed homogeniously. We assume:

$$A_{Ci} = A_o \cdot V_b = A_o \cdot 4\pi r^2 \, dr \qquad (1)$$

A_o=specific activity (Ci/cm^3), V_b=volume of speric shell

The radiation level in the centre of the sphere is equal and independent of the fact, that the activity is distributed in the speric shell or concentrated at a certain point. The doserate in the centre of the sphere can be assumed with

$$D = \Gamma_\gamma \cdot \frac{A}{r^2} \qquad (2)$$

D=Doserate (R/h), γ=dose factor (R.cm^2.h^{-1}.Ci^{-1})
Considering the absorption inside the sphere we get

$$D = \Gamma_\gamma \cdot \frac{A}{r^2} \cdot e^{-\mu r} \cdot B \qquad (3)$$

μ=absorption coefficient, B=build up factor
In this model B=1, because the same amount of gammaquants are scattered out of the considered volume as can become scattered into it. So we can combine (1) and (3) to get

$$D = \Gamma_\gamma \frac{A_o 4\pi r^2 dr}{r^2} \cdot e^{-\mu r} = 4\pi \Gamma_\gamma A_o \cdot e^{-\mu r} \, dr \qquad (4)$$

Integration over the whole sphere leads to the doserate depending on the specific activity:

$$D = 4\pi\Gamma_\gamma A_o \cdot \int_0^r e^{-\mu r} dr \qquad (5)$$or

$$D = 4\pi\Gamma_\gamma A_o \frac{e^{-\mu r}-1}{-\mu} = \frac{4\pi}{\mu}\Gamma_\gamma A_o(1-e^{-\mu r}) \qquad (6)$$

If there are several nuclides present, one has to sum over all activities:

$$D = 4\pi \sum_{i,j} \frac{\Gamma_\gamma(E_{i,j})}{\mu(E_{i,j})} \cdot A_i(1-e^{-\mu(E_{i,j})r}) \qquad (7)$$

When the diameter of this active sphere reaches infinite high values we get for $r \to \infty$ and (6)

$$\lim_{r \to \infty} D = \frac{4\pi\Gamma_\gamma A_o}{\mu} \qquad (8)$$

With respect to the characteristics of the cp used and the numerical values for dose constants and absorption coefficients (4) the calculated value for the centre doserate of an infinitive great sphere is 37,5 kR/h. The doserate inside a sphere with a diameter of 4,2-4,4 cm can be calculated with 8457 to 8810 R/h. From this value one has to substract the volume of the dosimeter itself, corresponding with 2240 R/h.

The result of the numerical calculation with 6217 to 6570 R/h shows excellent agreement with the experimental results. Although it is to mention, that the uncertainties are of great influence on the calculated results, because of the exponential dependence.

FURTHER WORK

In case of embedding these used particles into bitumen in the highest possible volume loading, we will reach an absorbed dose of 10^8 rad in about 160 days. With such a bitumen fuelparticle-mixture we can simulate a 300 years storage in about half a year. To study the leaching behaviour of a bitumen-salt mixture with less active material we will dilute the fuel particles with salt or grafite powder and determine the doserate inside this mixture with the method described.

Finally we can say that this method enables us, to measure the doserate of all mixtures of active wastes we want to embed in several kinds of bitumen, so that we are able to determine the loading capacity of each certain sort of bitumen, according to the radiation damaging we want to reach.

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A REVIEW OF THE DISPOSAL OF MISCELLANEOUS RADIOACTIVE WASTES IN THE UNITED KINGDOM

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In the United Kingdom there are about 25 major nuclear establishments such as reprocessing plants and power reactors, which produce radioactive waste. There are about 1500 other establishments, such as hospitals, universities, research establishments and commercial firms which also produce radioactive waste. The problems of the first of these classes, the enormous research and development programmes being carried out to achieve acceptable disposal routes for the wastes at present stored and the effect of the wastes which are disposed of, are well documented elsewhere. This paper deals with the second of the classes, those which can perhaps conveniently be called "minor users" although some establishments in this class discharge more radioactive waste to the environment than a nuclear power station. The whole of the United Kingdom radioactive waste management policy has recently been reviewed by an expert group [1]; this present paper looks at current practices for waste disposal from the minor users and indicates where the expert group endorses these or recommends changes.

LEGISLATION AND STANDARDS

In the United Kingdom, disposals of radioactive waste are subject to the Radioactive Substances Act, 1960 (RSA 60). Waste can be disposed of only in accordance with the conditions and limitations contained either in an authorisation which is specific to the disposer or, in the case of certain very low level wastes, in one of a series of Exemption Orders made under the Act. The Act covers the disposal of radioactive waste of all forms, solid, liquid or gaseous; there are no lower limits on radioactivity except for the natural radionuclides at levels found in nature; it removes the control of radioactive waste disposal from local authorities and places it in the hands of central government. United Kingdom standards for radiological protection are based on the system of dose limitation recommended by ICRP. In the case of radioactive waste disposal this is achieved by a case by case approach, a practice endorsed by the expert group.

EXISTING DISPOSAL PRACTICE

The following are the practices whereby all the wastes from the minor users are disposed of: where appropriate these practices are also used for wastes from the major nuclear establishments.

(i) SOLID WASTES

Solid waste of activity less than 10^{-5} μCi/g (0.4 Bq/g) is currently regarded as insignificant and it has not been the practice

for this to require authorisation. The expert group agreed with this and recommended it should be formalised in an Exemption Order.

Low Level Wastes in Domestic Refuse

Small amounts of solid radioactive waste are authorised for disposal with ordinary refuse. The limits applied for such "dustbin disposals" are 10 μCi (400 kBq) in 0.1m^3 and 1 μCi (40 kBq) per article. It is also usual to exclude alpha emitters and strontium-90 and to raise the first limit to 100 μCi (4000 kBq) in 0.1m^3 for the weak beta-emitters carbon-14 and tritium which are in common use. The expert group considered all the implications of this method of disposal and concluded that it represents no hazard. They therefore endorsed the practice.

Private Incineration

This method of disposal is useful for wastes which are unpleasant to handle; it also reduces the volume of waste requiring disposal. Separate authorisation is required for the disposal of the ash. The radionuclides and activities permitted for disposal this way are generally authorised on a case by case basis, taking all the local circumstances into account. However, it is usual to permit disposals of up to 100 μCi (4 MBq) a day of the commonly used tritium and carbon-14 without this detailed examination. The expert group find this practice acceptable.

Special Precautions Disposals

Where solid radioactive waste arises which is not suitable for dustbin disposal, disposal at a landfill tip is still permissible provided certain precautions are taken. Authorisations for such disposals specify the tip which is chosen after consideration of its management, its expected life, the probable subsequent use of the land, whether the tip is liable to catch fire, whether there is unauthorised salvage, drainage and any other special features. The two classes of waste suitable for such disposals are:

(a) Packaged Wastes: The limits and conditions normally imposed are:

a. Waste shall be conveyed to the tip in a sealed, plain, unlabelled plastic or multilayer paper sack in a closed metal bin;
b. At the tip, the sack shall be removed from the bin and placed either at the foot of the tipping face or in a hole dug for it and immediately covered with inactive refuse to a depth of 1.5m;
c. No sack shall contain more than 100 μCi (4 MBq) of radionuclides of half-life greater than one year and one mCi (40 MBq) of others, except that in the case of tritium and carbon-14, up to 5 mCi (200 MBq) per sack is permitted.

(b) Bulk Loads: Radioactive waste consisting of relatively lightly contaminated rubble and soil frequently arises as a result of the demolition of premises in which work with radioactive substances has

been performed. Typical examples are luminising works, gas mantle factories and ore-processing factories. Demolition and subsequent site decontamination often produce thousands of tonnes of lightly contaminated waste. When waste of this type is authorised for special precautions disposals, a limit of 10^{-4} μCi/g (4 Bq/g) is placed on it. This corresponds to a surface dose rate about 10 times the background level; when buried to 1.5m it cannot be detected on the surface and if care is taken not to concentrate the waste in one part of the tip the dispersion reduces radiation virtually to background level and subsequent disturbance can create no hazard.

The expert group endorsed all these conditions and practices but recommended in addition that in the case of packaged wastes, radionuclides with half-lives greater than one year (except tritium and carbon-14) should be limited to 10 μCi (400 KBq) per individual article. They also described this method of disposal as a valuable, and with the safeguards described, a radiologically sound method for the disposal of low-activity solid waste. They deplored the uninformed opposition to which these disposals are becoming increasingly subject and recommend that the duty already placed by RSA 60 on waste disposal authorities to accept radioactive waste should be extended to the operators of all landfill tips.

Disposal on Site

Disposal on the site on which the waste arises appears an alternative to special precautions burial on a landfill tip, but the expert group noted objections to this method and only recommended it provided certain conditions, including an assurance of ownership of the site for an appropriate period, could be met.

The National Disposal Service (NDS)

This is available for radioactive wastes not suitable for disposal by the means so far discussed. Generally, if the waste is bulky and within the authorised limits it goes for burial at Drigg, the site in Cumbria operated by British Nuclear Fuels Ltd, and individual items or small quantities go to the United Kingdom Atomic Energy Authority, Harwell where they are drummed for sea disposal. Full details of the operations at Drigg and Harwell are contained in "A Review of Cmnd 884", together with a description of the authorisation under which Drigg works and the international constraints on sea-dumping. In view of the fact that material from the minor users contributes only about 4% of the wastes disposed of by these routes, which have their main use in dealing with wastes arising from the nuclear fuel cycle, they are not considered further here.

(ii) LIQUID WASTES

Liquid radioactive discharges from the major nuclear establishments are, for the purpose of granting authorisations, evaluated individually against Government policy relating to disposals. The principles relating to the exposure of individuals and populations have been rigorously observed and discharge limits have been

assessed quantitatively, often with high precision. But these sophisticated techniques, involving a knowledge of environmental pathways, habit surveys, members of critical groups etc, are rarely necessary for the discharges from minor users.

Drain Disposals

Disposal directly to the drains, without prior collection or storage in hold-up tanks, is the most convenient and radiologically safe method of disposal of relatively small amounts of low activity liquid radioactive waste. Authorisations are usually given in terms of activity per month with limits on individual radionuclides where this is necessary. A few Ci (several GBq) a month, more for tritium, have been authorised for some establishments; others are able to operate with much smaller limits. Hospitals discharging the excreta of patients who have been given therapeutic or diagnostic doses of radioactive substances are amongst the premises having the largest authorisations. The radioactive waste is diluted immediately with other waste waters and in most cases the average concentration in the effluent from the establishment is orders of magnitude below the permissible level for drinking water. There is no formal upper limit for the average concentration of radioactivity in liquid effluents. Each case is considered on its merits: taking account of the toxicity of the radionuclides discharged; the possibility that they may settle out in the sewerage system; their behaviour in the sewage treatment process and ultimately in the effluent, whether discharged to a soakaway, stream, river or the sea. The expert group endorsed these practices with the proviso that the authorising departments should continue to check their assessments by monitoring a few of the most important cases.

(iii) AIRBORNE DISCHARGES

In the case of the minor users, the authorisation contains a specified limit on the activity which may be discharged in a given period. The radionuclides and activities permitted for disposal this way are, as in the case of all authorisations, based on the need of the user to have a particular level of discharge and are granted on a case by case basis taking all the local circumstances into account. The expert group are satisfied that existing controls over emissions to atmosphere are adequate for the time being.

THE EFFECTS OF RADIOACTIVE WASTE DISPOSAL

The effects of all disposals are assessed by the appropriate Inspectorate before authorisations are given, premises are inspected to ensure compliance and any necessary environmental monitoring is carried out. It is therefore possible to say with confidence that the effects of disposals from minor users are insignificant.

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PRESENT STATE OF THE MONITORING FOR INTERNAL CONTAMINATION AT TOKAI RESEARCH ESTABLISHMENT, JAPAN ATOMIC ENERGY RESEARCH INSTITUTE

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The Japan Atomic Energy Research Institue(JAERI) was established in 1956. Activities at Tokai Research Establishment(Tokai R.E.,JAERI) were started in 1958. Research and development of bioassay and whole body counting was started in 1958 and 1959, respectively. As for the internal monitoring, preliminary monitoring by bioassay and whole body counting were started in 1961, and routine monitoring were started in 1964. In 1967, research activities at Oarai Research Establishment(Oarai R.E.,JAERI; not so far from Tokai R.E.) were started, and routine monitoring have been carried out from this year.

At present, Tokai and Oarai R.Es. have about 2,000 and 500 employees, respectively. Among these employees, the number of workers who are registered as a occupational radiation worker are about 2,000(Fig. 1). Main facilities at both Tokai and Oarai R.Es. are; 5 research and testing reactors, 4 accelerators, 3 high level hot laboratories, radioisotope production plant, 2 radioactive wastes treatment plants and many other hot laboratories.

The Bioassay Section,Tokai R.E. is responsible for the monitoring of the internal contamination for both Tokai and Oarai R.Es. The Bioassay Section has three groups as shown below:

Bioassay Sec. Chief,1 Secretary,1 { Bioassay Group; 2 stuffs and 1 technician, External Counting Group; 3 stuffs, R&D Group; 3 stuffs

Main equipments and facilities belong to the Section are; 160 m^2 of chemical laboratory(Bioassay Group), 90 m^2 of laboratory equiped with a heavy shield(20 cm Fe + 5 mm Pb) whole body counter and chair type counter, and 125 m^2 of experimental rooms(R&D Group).

LEGAL LIMIT AND AUTHORIZED DOSE LIMIT OF JAERI

Japanese legal limit and authorized limit of JAERI is presented in Table 1. These limits are applied for both external and internal exposures. Legal limit is based on ICRP Recommendation:1962(Pub. 6). If the exposure dose exceeds the authorized limit(JAERI), then restriction of radiation work and medical care should be taken based on JAERI's manual on radiation safety.

Table 1. Dose limits applied in JAERI

Organ	Legal limit	Authorized limit(JAERI)
Whole body	3rem/3 months	3rem/3 months and 5rem/year
Skin	8rem/3 months	8rem/3 months and 30rem/year
Hand,Foot	20rem/3 months	20rem/3 months and 75rem/year
Bone,Thyroid	———	8rem/3 months and 30rem/year
Other single organ	———	4rem/3 months and 15rem/year

MONITORING OF INTERNAL CONTAMINATION

In JAERI, as to the exposure level, there are no workers who might exceeds 3/10 of the anual dose limit, and also as to the kinds of work, we have no radiation work such as descrived in ICRP Pub.12,Para.105. Therefore, the internal monitoring is not required routinely related to ICRP Recommendation. Internal monitoring is also not absolutely required by Japanese legal regulation, however it is required by JAERI's manual on radiation safety.

Two kinds of monitorings, that is, a Routine Monitoring and a Special Monitoring are carried out. The main purpose of the routine monitoring is to check a presence of significant contamination for the selected workers who are engaging an ordinary(routine) radiation work. The purpose of the special monitoring is to estimate of body burden, and committed dose equivalent, if necessary. Special monitoring is applied in the following cases for all of the workers who are suspicious for internal contamination;(1) a significant contamination is found by the routine monitoring,(2) an accidental contamination is occured, (3) after the work which internal contamination is suspicious.

As for the routine monitoring, methods of the monitoring are shown in Table 2.

TABLE 2. Method of routine monitoring

Method	Subject
External counting	
Whole body counting (Chair type counter)	Workers who are treating γ-emitters ($\geq$100 KeV)
Chest counting (Lung monitor)	Workers who are treating γ(x)emitters ($<$100 KeV)
Bioassay	
Gross α activity (Radiochemically)	Workers who are treating soluble α-emitters
Gross β activity (Radiochemically)	Workers who are treating β-emitters (Except low β-energy nuclides)
Uranium	Workers who are treating uranium
Tritium(Liquid scintillation counting)	Workers who are treating tritium and low energy β-nuclides

As for the special monitoring, various methods are used according to the contaminated nuclides, a heavy shield whole body counter is generally used when the external counting is applied.

FREQUENCY OF ROUTINE MONITORING AND LEVEL OF SIGNIFICANT CONTAMINATION

As a basic principle, we adopt the "investigation level" or "derived investigation level" defined by ICRP . However, regardless of internal contamination level, if the observed value exceeds the detection limit, then the fact is informed to Radiation Control Office. This information will be contribute to improvement of the working conditions. Thus, at present, we are taking the detection limit as the significant contamination level.

TABLE 3. Frequency of routine monitoring and detection limit

Kind of monitoring		Frequency	Detection limit*
External counting	WBC(Chair type)	Every 3 months	~17nCi(^{137}Cs,whole body)
	Chest counter	Once a year	10-15nCi(^{239}Pu,lung)
Bioassay	Gross α activity	Twice a year	~0.2pCi/l (^{239}Pu,urine)
	Gross β activity	Every 3 months	~10pCi/l (^{90}Sr, urine)
	Uranium	Every 3 months	~5μg/l(nat.U, urine)
	Tritium	Every 3 months	~50nCi/l(^{3}H, urine)

*Detection limit: The value of mean+3σ obtained from non radiation workers.

In the routine bioassay, only urine is taken as a sample, but in the special monitoring, both urine and feces are taken as sample.

SELECTION OF SUBJECTS FOR ROUTINE MONITORING

As provided by the JAERI's manual on radiation safety, the subjects for routine monitoring are selected by the chief of each working group(Section or Laboratory), and a request for routine monitoring is presented to Bioassay Section. However, to hold the same views about the selection of subjects is very difficult because there are nearly 100 working group in JAERI. Therefore, the following procedures are taken as a practice.

1. Bioassay Section; Survey the working condition: a questionaire about the working condition is send to each working group.
2. Working Group; Fill up the questionaire for each worker.
3. Bioassay Section; Investigate the results of the survey, discuss with Radiation Control Office, and make an original plan about the number of subjects for each working group.
4. Bioassay Section; Present the original plan about the number of subjects to each working

group, revice the number is made if necessary.

5.Working Group; Select the subjects according to the plan. Request of routine monitoring is send to Bioassay Section.

Main items of the questionaire are; kind of nuclides, amounts, chemical form, degree of air contamination due to the work, ways and means for protection from the air contamination.

RESULTS OF INTERNAL MONITORING

The number of subjects of routine monitoring after 1969 are presented in figure 1. By routine monitoring, the significant internal contamination is sometimes found for the worker who are treating tritium, but it is almost never found for the other workers.

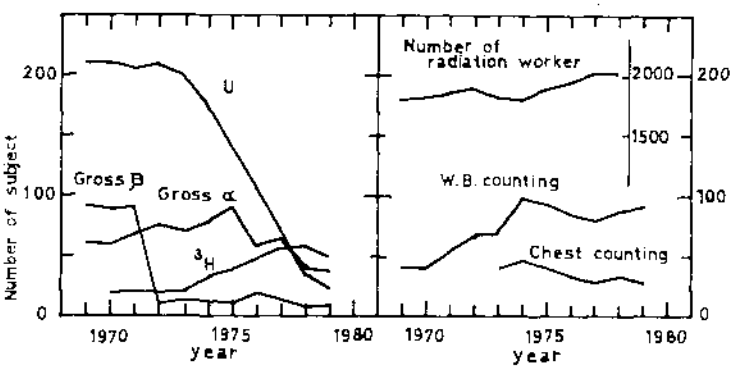

Figure 1. Numbers of subjects of routine monitoring

As for the special monitoring, a number of subjects are largely varied depended on the activities in JAERI. In 1965, the number of subjects who received the special monitoring reached to nearly 300 because a repaire of research reactor was carried out in this year. In the recent few years, the number of the subjects for special monitoring is several tens or so.

As for the special monitoring, the workers who found the significant internal contamination are less than 50%. Moreover, the internal dose(50 years) estimated are mrem order for the most of all subjects. During past 15 years, only a several cases of exposure of rem order were found. The highest dose we had experienced was about 4 rems(^{131}I, thyroid).

RECORDING LEVEL

As regards to a formal recording level, we adopt 10 mrem(usually 50 years dose). When the estimated dose is less than this level, the value is treated as zero.

THE CONTROL OF RADIOACTIVITY IN THE WORKING ENVIRONMENT IN THE FACTORIES FOR PRODUCTION OF PHOSPHATE FERTILIZERS

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It is already known that phosphate minerals of different origin (Togo, Jordan, Marocco) contain certain amounts of uranium. In the production of chemical fertilizers phosphate minerals are used in two different ways: a) as starting raw material for production of phosphoric acid and b) as starting raw material for production of different types of fertilizers (PK,NPK) with different ratios of particular components (N,P_2O_5, K_2O), depending on the kind of fertilizer.

The aim of the present work was to examine, for the purpose of their protection, the extent of contamination of the working and wider living environment of three plants for production of chemical fertilizers.

Dosimetric control of the working environment and control of aerosols radioactivity was performed in the plants which will further be referred to as I, II and III, and samples were then taken for laboratory analysis and for measurement of the total beta radioactivity at those points in the technological process where considerable increase of irradiation dose had been established. On the basis of such measurements and elaboration of the results, conclusions have been drawn concerning the existence, ways of distribution and locations of accumulation of the radioactive component in particular phases of the technological process in the plants for production of both phosphoric acid and mineral fertilizers.

Since the technological procedures are not the same in all the plants an attempt will be made to point out, by comparative analysis of the obtained results, their characteristics from the standpoint of possible contamination of the environment and working premises and to propose protective measures to reduce danger threatening the personnel employed in the plants.

Possibility of utilization of the radioactive component for separation of uranium in these plants is another aspect of dealing with this problem but it was not the subject of our investigation.

EXPERIMENTAL RESULTS

Systematic dosimetric measurements performed in the plant for production of phosphoric acid and chemical fertilizers I have shown at particular places an increase of even up to 1000 times of the radiation exposure dose, as compared with the background which was 6-8 μR/h.

In some parts of the plant where the personnel stays during the whole working day measured radiation doses were in the range from 150-400 μR/h, which is an increase of about 20-50 times with respect to the background. This means that the workers at such places should be classified as professionally exposed to ionizing radiation.

Situation was similar in the plant II where the maximum radiation exposure doses measured were more than 300 times higher than the background, and in the working premises permanently occupied by the personnel for several hours they were up to 10 times higher.

In the plant for production of mineral fertilizers III no more considerable accumulation of the radioactive component has been found by dosimetric measurements owing to some characteristics of the used technological process (which actually represents a closed system). Maximum measured radiation doses did not exceed the background by more than three times.

However, we should point out the fact that the plants I and II were not in operation, while the plant III operated only partly at the time of dosimetric measurements and sampling for laboratory analysis. This also explains the fact that the measured aerosoles' radioactivities in the working premises did not exceed the background values by more than 3-5 times. Under normal operation conditions they would no doubt be considerably higher.

Besides the samples of starting raw materials, final products and waste waters, samples of materials of different origine were also taken for measurement and estimation of the total beta radioactivity at the places in the plants I and II where dosimetric measurements have shown higher radiation doses. In spite of no such indications in the plant III, a certain number of samples were taken from the places where higher concentration of activity could be expected, special attention being paid to waste waters from this plant.

From the results obtained for a large number of analyzed samples, the data for characteristical samples from the three plants are comparatively presented in Table 1, while Table 2 displays the data for the beta radioactivity and other parameters determined for water samples.

CONCLUSION

As the results of the total beta radioactivity measurements of the samples taken in the plants I,II and III

represent the basis for estimation of contamination of the working premises and the environment of the plants it should be pointed out here, with the aim of undertaking preventive measures at critical sites, that more complete and real picture of the existing conditions can be obtained only by the analysis of samples taken when the plants are in full operation. However, certain conclusions can be drawn from the results obtained so far:

- The total beta radioactivity of the starting raw material-the phosphates of different origin-is by two to four times higher than an average radicactivity level of soil materials, except for uranium ores and minerals rich in potassium.
- At the initial stages of H_3PO_4 production processes in the plants I and II radioactive component is accumulated in the liquid phase of the pulp.
- Judging by the radioactivity level of the samples taken in further phases of the technological process in these plants the radioactive component is dissolved in the phosphoric acid and therefore is present in all the phases of the production process, i.e., up to the formation of the final product, concentrated H_3PO_4, the radioactivity of which is the highest except for waste materials, such as the samples of stone deposited in the pipes and the filter cloth of a H_3PO_4 separator, where radioactive material concentrations are even higher.
- Radioactivity of the final products-chemical fertilizers from the three plants-is not increased. Since the percentage of K, as their component, is rather high (even up to 25% of K_2O) it was possible to assume that the greatest part of radioactivity originates from ^{40}K. This has been proved by performed gamma spectrometric measurements.
- Since phosphoric acid, the carrier of the active component, is used as raw material in the three plants for production of mineral fertilizers (whose radioactivity is not considerably increased) special attention should be paid to waste waters which, apparently, convey radioactive and other harmful materials to the rivers on whose banks the plants are located. However, the extent of danger thus caused could not be estimated by the analysis performed.

Measures for water protection could be recommended competently only on the basis of the results of a multidisciplinary approach to this problem. The following general conclusions can be drawn from the above stated facts:
1. Contamination of the working environment is prominently higher in the plants I and II than in the plant III; 2. Adequate protective measures, depending on the degree of contamination, should be undertaken in the working and living environment of the plants I, II and III, including the following:

- Obligatory utilization of protective masks by the personnel working in raw material warehouses, as a preventive measure against contamination by inhalation, introduced after previous medical control of workers when the presence of ^{226}Ra and its daughter nuclides in the phosphate samples

had been proved by spectrometric analysis.

- Classification of waste materials as radioactive wastes
- Special attention to waste water problems
- Introduction of personal dosimeters for workers at particular posts.

TABLE 1. Total beta radioactivities of the samples of starting raw materials, final products and some other characteristical materials from the plants I, II and III.

	Sample	Sampling place	(pCi/gr)	A (pCi/l)
Starting raw material	Phosphate (Togo)	Plant (H_3PO_4) I	86	
		Plant (H_3PO_4) II	98	
		Plant (chem.fert) III	53	
	Phosphate (Jordan)	Plant (chem.fert) III	97	
	Phosphate (Marocco)	Plant (chem.fert) I	86	
	H_3PO_4 28%	Plant (H_3PO_4) II		34.240
	34%	Plant (H_3PO_4) I		40.700
	30%	Plant (H_3PO_4) I		
		Plant (chem.fert) I, III		28.900
	KCl	Plant (chem.fert) (SSSR) II	315	
Final prod.	NPK (chem.fert)	II	161	
	TSP (- " -)	I	71	
	NPK (- " -)	III	140	
Other samples	Filter cloth of a separator	Plant (H_3PO_4) I	92.670	
		II	500	
	Slime from a H_3PO_4 basin (weak acid)	Plant (H_3PO_4) I	16.300	
	Silicohydrofluoric acid	Plant (H_3PO_4) II		38.400
	Scale from a pump for weak acid	- " -	53.800	
	Floor dust	- " -	3.820	
	Filtrate from pulp	- " -		38.600
	Gypsum from pulp	- " -	23	

TABLE 2. Determination of the total beta radioactivity and other parameters of water samples

Sample and sampling place	pH	Elect.cond.	Min.residue	A pCi/l
Wastewater, plant I	was not	determined	719	57
Waste water, plant II	1,9	27,0	660	600
Water from the channel upstream of the drain pipe, plant III	8,6	7,40	321	6
Waste water from the reservoire for neutralization, plant III	7,4	16,45	972	96

EMERGENCY PLAN FOR A URANIUM AND PLUTONIUM HANDLING LABORATORY

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INTRODUCTION

The Austrian "Radioprotection Law" (1)(2) requires for laboratory where open sources like Uranium and Plutonium are handled, that an emergency plan and system must be established, specifying responsibilities and the sequence of actions to be taken in the event of radiation accidents involving individuals or premises.

Radiation accident situations may be confined, i.e. limited in their effects to single individuals or to isolated locations within an establishment, or unconfined and thus having effects upon many individuals and/or many locations within an establishment and its environ.

The purpose of the emergency plan for such a laboratory is thus to establish a system of preparedness, through standing arrangements and specific working instruction by which to mitigate the consequences of any foreseenable unconfined radiation accident situation, which may occur.

SUMMARY

There are three sub-categories of unconfined radiation accident situation in this emergency plan:

a) Emergency Standby
being a situation where no actual radiological hazard exists within the laboratory, but where the potential for the development of such harard is foreseenable.

b) Emergency Alert
being a situation where unacceptable radiological conditions exist or appear to be unavoidable, such that protective measures are to be effected without delay and the controlled evacuation of all personnel from the radiation area of the laboratory is necessary.

c) Fire in the laboratory
whilst not necessarily involving radiological hazard, is unquestionable the most serious foreseen accident condition, which may occur. For this reason a fire detection system was designed and installed to

provide automatic audible and visible annunciation throughout the laboratory in the event of an outbreak of fire. It requires urgent evacuation of all personnel from laboratory.

Detection and Announcement of Emergency Situations

A comprehensive automatic surveillance system was installed throughout the laboratory to detect and annunciate the occurrence of any abnormal condition such as, outbreak of fire, unacceptable concentration of airborne contamination, power supply failure, flooding, ventilation plant failure. The system categorises each individual alarm condition with respect to its significance to emergency situations.

Within the working time it is the responsibility of the head of the laboratory or his deputy to announce any occurance of an emergency situation. Outside normal working hours all alarm situations are detected in the control room of the plant (reactor center Seibersdorf). A standby team who was created in order to ensure that experienced laboratory staff are available at all time, is then informed.

Emergency Teams

Inside the laboratory the responsibilities are given to different emergency teams. The overall responsibility for the supervision has the socalled Emergency Controller (who is normally the head of the laboratory). Others are the health physicist, the fire warden, the lift warden, the first aid officer, the security officer.

The different responsibilities are divided to:

Health Physicist

Provides in the event of radiation accident situation measures necessary to minimize the radiological hazard to personnel and property, conduct the necessary surveillance through which to determine the extent and degree of the radiological hazard and coordinate the necessary decontamination measures. Devise and conduct periodic exercises and training programs to ensure the continuing effectiveness of the emergency plan.

Fire Warden

In the event of fire supervises the initial fire fighting undertaken by the laboratory staff and acts as liaison officer for the plant fire brigade. Devises and conduct also periodic exercises and training programs relating specifically to actions to be taken in the event of an outbreak of fire.

Lift Warden

Supervises the technical measures necessary to rescue any individuals trapped within the lift. Conduct periodic inspections of the emergency equipment installed within the lift.

First Aid Officer

Coordinates the measures necessary to rescue any individuals trapped in the laboratory. Training of the laboratory staff in first aid together with the medical officer.

Security Officer

Maintaines at all times, an accurate list of all personnel within the operational area of the laboratory.

Dependent upon the severity of an emergency situation the emergency controller may call for help from the control room of the plant, where are the complete health physics group, a decontamination group, a medical and first aid group and a fire brigade is available.

Additionally there are arrangements made to transport and to permit the treatment of injured persons at the "Unfallkrankenhaus Meidling", where in particular, basic facilities are available to deal with contamination control.

Another arrangement has been concluded with the civil fire brigades from the nearby towns for support services in the event of a serious outbreak of fire.

Emergency Equipment

In addition to the health physics monitoring equipment, routinely employed in the laboratory, equipment specifically provided for emergency utilisation is located at a special assembly point. Additionally emergency equipment is also available in the plant emergency research.

CONCLUSION

During five year operation of this uranium and plutonium handling laboratory there was no unconfined radiation accident situation.

A few exercises were organized for the training of the different emergency troops and to see if the alarm systems and the standby teams are proper working so that all than can be foreseen for a radiation accident situation is in order.

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RADIATION EXPOSURE OF PERSONNEL IN A REPROCESSING PLANT

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Occupational radiation exposure and the associated collective doses due to the production of nuclear energy are besides the environmental impacts an important issue for its public acceptance. At nuclear power plants experience has already accumulated which encourages authors to develop prediction models, leading for instance to a predicted annual collective dose for a 9oo-MW PWR plant of about 55o man-rem (1). But also in reprocessing plants experience has been gained as to allow a forecast of their contribution to collective doses of the fuel cycle and possibilities of its reduction. The particular experiences in the pilot reprocessing plant Karlsruhe (WAK) are reported and some conclusions drawn.

OCCUPATIONAL EXPOSURE AT WAK

The annual average occupational exposure at WAK was successively decreased during the ten years of plant operation, which is shown in following Table 1.

TABLE 1. Average annual doses due to professional exposure at WAK from 1972 to 1979.

Year	rem	Year	rem
1972	1,33	1976	o,31
1973	1,11	1977	o,44
1974	o,48	1978	o,45
1975	o,18		

To the total professional exposure, expressed as collective doses in man-rem for the purpose of risk comparison, exposures from different operations contribute. In this summary data from reprocessing campaigns and interventions periods are discussed (Fig. 1).

During reprocessing campaigns at WAK the personnel exposure was on the average not less than during intervention periods. Fig. 1 shows the man-rem per year due to reprocessing campaigns.

Of interest is the derivation of a specific quantity for the purpose of comparing the data from plants of different size. A specific collective dose in terms of man-rem/GW_e.y is being used (e.g. 2) and is shown on Fig. 2, together with data from other facilities. It can be concluded, however, that this specific quantity does not allow quantitative comparisons and extrapolations.

During intervention periods there are other parameters influencing the resulting collective doses, which are analysed in (3).

The WAK is a reprocessing plant with direct maintenance. Six extended intervention periods lasting from 2 to 15 month during ten years of active operation have been scheduled for modifications and repair. Personnel exposure during these periods depends largely on the radiation fields, which in turn depend on the decontamination of that part of the installations where work has to be carried out.

It can be demonstrated from the experience so far obtained that personnel exposure can be kept adequately below dose limits even when equipment having been in contact with high active solutions has to be exchanged. An important further result, however, is the fact that the possibility of equipment decontamination must be improved, when its exchange or repair shall be carried out in due time. Of great importance is furthermore the organization and administrative control of personnel exposure.

Data of individual as well as collective doses obtained between 1972 and 1979 during interventions clearly show these dependencies. Collective doses due to interventions contributing to the total annual collective dose are shown in Fig. 1.

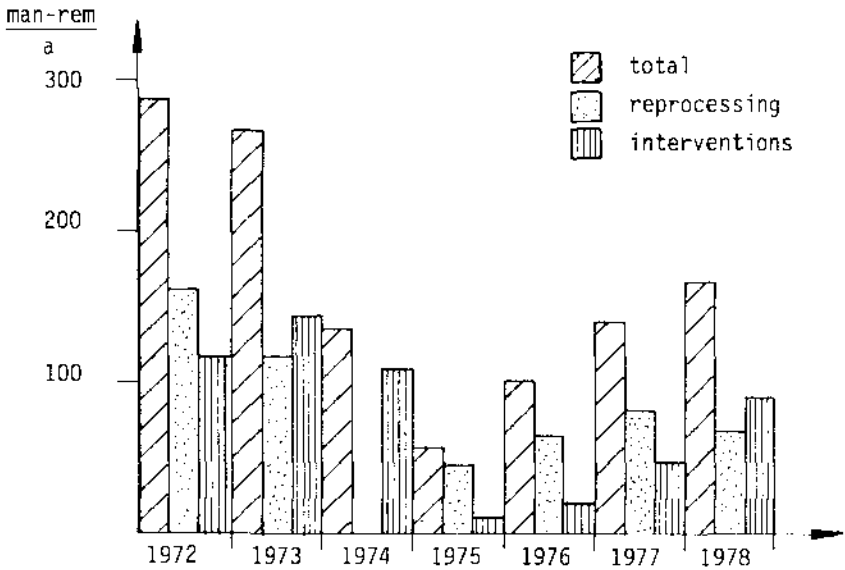

FIGURE 1. Total annual collective doses and due to reprocessing and interventions at WAK, 1972 - 1979

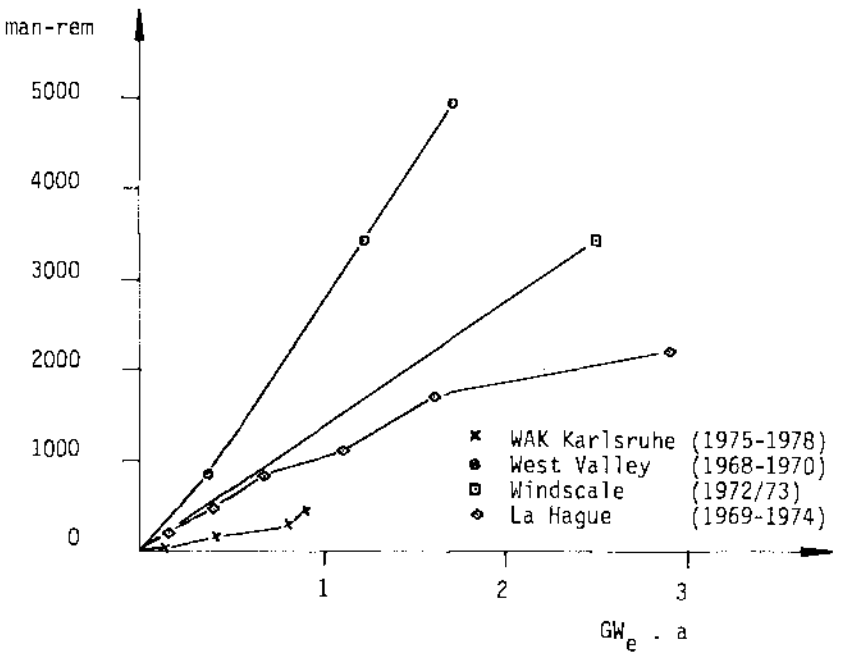

FIGURE 2. Collective doses as function of energy equivalent of reprocessed fuel at different facilities.

DISCUSSION AND CONCLUSIONS

As it is well known, dosimetric data have only a limited accuracy. It can be, however, reasonably assumed that variations are sufficiently smoothed in the statistical collective. Nevertheless the statistical significance of differing data should always be evaluated, if it is argued with them.

A comparison with collective doses from other reprocessing facilities with reference to the energy equivalent of reprocessed fuel shows a rather large variation (Fig. 2). Quantitative extrapolations appear to be not yet possible with this quantity.

Collective doses due to interventions in a pilot plant like WAK can be considered in a first approach as equal to the doses during reprocessing for some ten years of operation. Since in a larger facility the possibility of intervention could be improved, the provisional conclusion can be drawn that personnel exposure due to interventions will be at a maximum of the same size as during reprocessing.

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RADIOLOGICAL PROTECTION ASPECTS OF ^{123}I PRODUCTION

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With the Eindhoven AVF cyclotron ^{123}I for application in nuclear medicine is now routinely produced in quantities of 4 to 20 GBq per batch. Enriched telluriumdioxide on a platinum backing is irradiated with 25 MeV protons. The production reaction is $^{124}Te(p,2n)^{123}I$. Beam currents are used in the order of 20 μA.
The radiochemical separation of iodine from tellurium is carried out by heating the telluriumdioxide just above its melting point for only a few minutes. During this procedure the total amount of ^{123}I is handled as vapour in a quartz tube (1).
During the experimental stage of the project a radiological safety program was developed in close collaboration with the workers. This program was considered to be an integral part of the total production project. The radiological safety program implies normal working conditions, failure analyses and emergency procedures.

SHIELDING AT THE IRRADIATION SITE

The part of the cyclotron hall with the irradiation site is shown in fig. 1. Primal shielding is provided by a concrete wall of 1.5 m thick. The concrete roof is 0.4 m thick. Radiation levels at different locations in the cyclotron hall are permanently measured with a monitoring system for γ-rays and neutrons. The alarm levels of the detectors at the inside of outer (glass) wall correspond to a derived working limit of 2.5 μSv/h for non-supervised areas. Exceeding of the alarm levels will result in automatic interruption of irradiation (with a short delay time for adequate action by the cyclotron operators). Even with additional neutron shielding with 0.4 m paraffin around the production position, the proton beam current was restricted to less than 3 μA. Since neutrons contribute dominantly to the dose rates we paid much attention to neutron measurements. At several places under different shielding conditions neutron energy spectra were measured using a self-developed multisphere technique (2). It was shown that sky shine of neutrons passing through the roof and scattering in the labyrinth caused a major part of the dose rate outside the irradiation facility. In fig. 2 some of the measured neutron spectra are shown. In addition corresponding values of the flux density, mean energy and dose equivalent rate are given in table 1. For neutrons passing through the roof it was found that additional local shielding with 0.2 m paraffin reduced the integral neutron flux density by a factor of about seven. The mean energy of the transmitted neutrons increased with a factor of 2. A minor modification of the labyrinth lay out reduced the contribution from scattered neutrons. At present the improvements in shielding allow for beam currents of 20 μA, which corresponds to a ^{123}I production rate of about 10^{10} Bq. It is noted here that a reasonable high production rate is required not only to shorten the expensive operation time of the cyclotron but also to

Table 1. The neutron fluence rate, the mean neutron energy and the neutron dose equivalent rate at the positions A, B and C, marked in Fig. 1. Notation B_1 denotes without paraffin, B_2 denotes with paraffin. The proton beam current is 20 µA.

Position	Neutron flux density $[m^{-2}s^{-1}]$	Mean energy [MeV]	Dose equivalent rate [rem/h]	[mSv/h]
A	$3.6\ 10^{10}$	0.56	68	680
B_1	$5.6\ 10^{8}$	0.69	1.1	11
B_2	$8.6\ 10^{7}$	1.21	0.21	2.1
C	$8.6\ 10^{7}$	0.08	0.06	0.6

restrict the ^{124}I content to a low level (< 1%).
Radiation exposure during target handling after irradiation is mainly caused by γ-rays from ^{194}Au in the target support. The dose equivalent rate at 0.3 m distance from the target is about 15 mSv/h when 37 GBq of ^{123}I is produced. The irradiation set-up is at one side shielded by 0.05 m lead. The dose rate just behind the lead wall is reduced to 250 µSv/h. At working distance dose equivalent rate is less than 100 µSv/h. Operated from behind the shielding the target is removed from the irradiation position with handling tools and put into a transport container.

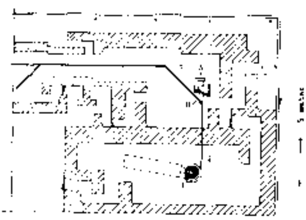


Fig. 1 Part of the cyclotron hall with the ^{123}I production site (to scale). (1) cyclotron (2) beam transport system (3) space for radio-nuclide production, (4) irradiation facility for solid targets e.g. for the production of ^{123}I, ^{77}Br and ^{81}Rb; (A), (B) and (C) denote positions where the neutron spectrum and neutron dose equivalent during ^{123}I production have been determined. Position C is on the roof, 3.85 m above ground level, [illegible] radiation detectors (alarm system).

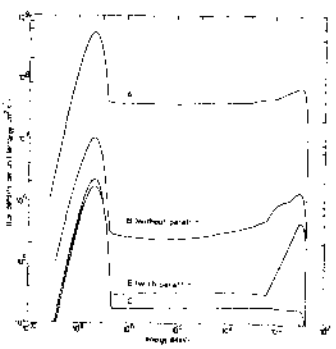
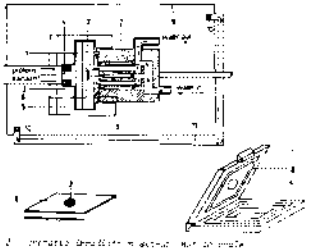


Fig. 2 Neutron energy spectra at the positions A, B and C, marked in Fig. 1. The flux density per unit lethargy as a function of the energy is given (lethargy [illegible] ln E/E_0, where E_0 stands for 10 MeV and E for the neutron energy in MeV). The proton beam current is 1 µA.

IRRADIATION SET-UP

Since the radio-iodine can be liberated from the TeO_2 target material by volatilization at temperatures above 500 °C safety precautions are necessary to minimize the risk for air contamination and consequent internal contamination of personnel. These safety measures basically are two fold: At first target temperatures must be limited and secondly containment must be provided in case radio-iodine escapes. The irradiation set-up is sketched in fig. 3. By defocussing the proton beam, hot spots on the target which may cause problems with regard to temperature control are prevented. The platinum target support is cooled by a forced water flow. The irradiation is automatically inter-

rupted when the flow rate falls below 4.5 liter per minute. The cooling system and target holder were designed for a 5 cm^2 proton beam cross section and beam current density upto 30 $\mu A/cm^2$. To minimize the consequences of accidental volatilization the TeO_2 layer is locked up in an air-tight target holder. The front side of the target holder is a 10 µm thick tantalum foil through which the proton beam passes. A second tantalum foil is placed at the end of the beam pipe to seal its vacuum. It also is a secondary barrier to prevent contamination in the beam pipe. Loss of vacuum is detected and will automatically stop the irradiation process. The entire irradiation set-up is mounted in an air-tight glovebox. The leakage rate of the box for helium gas was measured to be less than 0.1% of the total volume per hour. The air in the box is at ambient pressure and is recirculated continuously over a charcoal filterbed with a trapping efficiency for iodine of at least 80%. The recirculation rate is 1.5 liters per second which corresponds to 1/3 of the total box volume per minute.



THE RADIOCHEMICAL SEPARATION SET-UP

To reduce external radiation exposure the separation process is operated from behind a 0.05 m thick lead shielding. Handling tools were constructed for all types of manipulations including the input of the target from the container into the apparatus, the displacement of ovens, as well as the filling of the-ready-for-transport glass capsules. The dose equivalent rate just behind the lead shielding ranges from 20-50 µSv/h. The separation technique is described in detail elsewhere (1). The apparatus is shown schematically in fig. 4.
During operation the apparatus is kept at reduced pressure which is maintained by the hydrostatic water column (maximum pressure reduction 0.6 m water). Before the start of each separation procedure the entire apparatus is tested for leakage. During separation the iodine liberated from the target due to heating, is forced to flow to the recipient containing glass beads (J). The trapping here is about 75%. The rest of the iodine will be trapped in the charcoal filters (O_1, O_2). The third filter (O_3) is permanently monitored for radio-iodine content. Radioactivity in here has never been detected. The carrier gas is not released but collected in the upper part of the hydrostatic water column. In analogon with the irradiation set-up a perspex glovebox is provided

for reasons of containment in case of iodine escaping out of the separation apparatus. In this box the air is recirculated via filters (O_4, O_5) at a rate of 1.5 liter per second which corresponds to 1/6 of the total box volume per minute. Here also the trapping efficiency is about 80% per passage.
Since heating in three different ovens is essential in the chemical separation process there is a risk for the perspex box to melt, with consequent risk for loss of containment. As safety measure a water cooling system is provided above the ovens. As an additional safety precaution it is possible to turn over to air cooling in case of system failure.

AIR MONITORING

For continuous measurement of the radio-iodine in air outside the glovebox an air monitoring system was developed. Via a maximum number of 6 inlets air is sampled with a flow rate of 2.8 10^{-3} m^3/s from "critical" positions just outside the glovebox. A GM-end-window tube type 18540/01 is used as detector just above the coal filter assembly (trapping efficiency > 95%). The detection efficiency is 3 10^{-2} cps/Bq for ^{123}I. Based on the respective DAC values being 9 10^{6} Bq/m^3 for ^{123}I and 1 10^{3} Bq/m^3 for ^{124}I and considering that the ^{124}I content is about 0.8% at EOB, the weighted value for the effective DAC was taken to be equivalent to 5 10^{4} Bq/m^3 of ^{123}I.
The time derivative of the count rate is simply proportional to the radio-iodine concentration in the sampled air (expressed in units DAC). Since the maximal count rate corresponds to the total accumulated activity it is a measure for the upper level of the potentially inhaled radioactivity. The count rate is recorded during the entire separation process. Adjustment of the air flow rate resulted in a simple relation between the count rate R and the upper level for intake $I = 10^{-8}R \times ALI$. It was shown that concentration at 0.1 DAC level can be detected within 25 seconds.
In three years of production only two times a small air contamination was measured at a level of 2 × DAC. Both cases were due to improper handling of the capsules with the glass beads.

FINAL REMARKS

Just in case of extreme system failure: coincident loss of containment and breakage of separation quartz tube an emergency procedure is developed as a result of which the accidental intake by personnel reasonably can be expected to be less than 0.1 × ALI.

The results of personal dosimetry measurements show that for the group of 5 workers the accumulated collective dose equivalent was less than 5 mSv for a production of 300 GBq in a period of three years.

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EXPERIENCES IN MONITORING AIRBORNE RADIOACTIVE CONTAMINATION IN JAERI

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In Japan Atomic Energy Research Institute(JAERI) there are many facilities and laboratories where the protection from airborne radioactive materials is more important than that from external radiation. In them air monitoring is indispensable to confirm the kinds and amounts of airborne radionuclides and estimate the personnel exposure dose due to these atmospheric contaminants.

This report describes some problems which have been experienced in air monitoring for hot cells for handling highly radioactive materials, glove boxes for handling plutonium and a cell for producing ^{99}Mo.

DISPERSION FACTOR

When irradiated uranium metal fuel was cut in a hot cell, the ratio of activity of airborne dust to that of the whole dust was determined for representative nuclide (1). The dispersion factor $\varepsilon(k)$ for nuclide k was defined as

$$\varepsilon(k) = D(k)/ M(k)$$

D(k) is the activity of nuclide k dispersed in the air in a hot cell when the fuel was cut; it was evaluated from the concentration of airborne nuclide k activity, the volume of the hot cell($144m^3$) and the ventilation rate of the cell(10 volumes/h). M(k) is the total activity of nuclide k contained in the cut dust; it was calculated from the irradiation history of the fuel. Airborne radioactive dust in the hot cell was collected on a cellulose-asbestos filter paper(Toyo HE-40) and a charcoal- loaded filter paper(Toyo CP-20). The activities of the dust were determined using a gamma-spectrum analyzer with a 5"x4"NaI(Tl) scintillation detector.

The results showed that the dispersion factors were the order of 10^{-2} for semi-volatile form of ^{125}Sb and $10^{-3} \sim 10^{-4}$ for particulate forms of ^{137}Cs and ^{144}Ce-^{144}Pr.

SIZE DISTRIBUTION OF RADIOACTIVE AEROSOL

For radioactive aerosols produced during works (mainly decontamination in the hot cells), the size distributions were investigated with a 4-stage cascade centripeter developed by U.K.A.E.A..

Radioactive aerosols were collected by the centripeter operated at flow rate 20 l/min for the work duration(10 min ~ 2 h). The representative size(expressed as aerodynamic diameter) for each stage of the instrument at flow

Table 1 Size distributions of radioactive aerosols produced during various works at the hot laboratory

Group	Radiation work	Height of sampling point (m)	Size distribution	
			AMAD (μm)	σ_g
A	Decontamination of the hot cells	0.7-1.1	2.5-11 (6.0-6.4)*	1.7-3.2 (1.6-1.9)*
	Exchange of the exhaust filters	0.3	7.0	1.7
		1.1	10	2.6
	Overhauling in the hot cells(1)	1.1	6.5	1.8
			8.0	2.0
	Handling of the contaminated container	0.3	8.5	1.8
		0.7	12	2.8
	Handling of irradiated graphite	0.6	15	2.8
	Dismantling of the hot drain pipe	1.1	9.0	1.7
B	Overhauling in the hot cells (include welding) (2)	0.3	0.41	7.1
			1.2	12.5

* Size results of aerosol with alpha activity.

rate 20 l/min was calculated from the data(the effective cut-off sizes for the four stages at flow rate 30 l/min) given by Hounam,R.F. and Sherwood,R.J.(2) and O'Connor,D.T.(3), using the relationship that the aerodynamic diameter of aerosol is inversely proportional to the square root of the flow rate on the basis of impaction theory. Activity median aerodynamic diameter(AMAD) and geometric standard

deviation(σ_g) of the size distribution were estimated graphically on a log-normal probability paper, assuming a linear relationship between cumulative percentage of the activity on each stage and the representative size.

In Table 1 are shown aerosol size results for 24 aerosol samples with beta activity and for 3 samples with alpha activity. AS seen in Table 1, the AMAD and σ_g of the size distributions in usual radiation works(group A) are in the range of 2.5 ∿ 15μm and 1.6 ∿ 3.2, respectively. The most remarkable in the present investigation was group B having AMAD less than 1μm with σ_g of larger than 7. Such fine, respirable aerosols could be produced by high temperature burning in the air of radioactive contaminants within the cell due to arc welding.

RESUSPENSION FACTOR

In a plutonium handling laboratory, surface materials of the floor and equipments were contaminated extensively by particulate plutonium oxide(PuO_2) due to break of the negative pressure in a glove box. Therefore, the suspension factor(the ratio of the concentration of airborne contamination to the level of surface contamination) had to be evaluated to determine the procedure of decontamination and the type of respiratory equipment. Two workers who wore Harwell self-contained air-ventilated blouses, moved around in a PVC sheeting tent to fix the contaminant and seal up the leakage positions on the glove box. Volume and area of the contaminated floor inside the tent were about 6 m^3 and 2 m^2, respectively, and its ventilation rate was 10 volumes/h. Floor surface was almost uniformly contaminated at 4.5 x 10^{-2} $\mu Ci/cm^2$. The contamination was measured with a ZnS scintillation survey meter. Airborne contaminant was sampled with a personal air sampler, and its particle sizes were evaluated by autoradiographic technique. The particle size distribution was in the range of mass median diameter(MMD) 6.4 to 26μm, and σ_g was from 2.3 to 2.7.

The resuspension factor was found to be 4 x 10^{-8} ∿ 2 x $10^{-7}cm^{-1}$ for plutonium oxide particles deposited on the floor surface. The values agreed well with those obtained by Jones,I.S. and Pond,S.F.(4).

COLLECTION EFFICIENCY OF CHARCOAL-LOADED FILTER PAPER FOR AIRBORNE RADIOIODINE

A new charcoal-loaded filter paper(Toyo CP-20T) was made in order to improve collection efficiency of CP-20 filter papers and decrease its air resistance. Specifications of CP-20 and CP-20T filter papers are shown in Table 2. The radioiodine generated in ^{99}Mo production process was used for test, and consisted of 60 % inorganic and 40 % organic iodide. The chemical forms were determined with a sampler modified Maypack(5). The test conditions of air

flow through filter papers were relative humidity 40 ∿ 80 %, face velocity 50 cm/sec and flow time of iodine 4 ∿ 18h.

Table 2 Comparison of specifications of charcoal-loaded filter paper

Specification	Charcoal-loaded filter paper	
	CP-20	CP-20T
Weight (g/m^2)	700	650 ∿ 700
Thickness (mm)	2.0	2.2
Base material	Cellulose-asbestos fiber	Glass fiber
Activated charcoal	Tsurumi coal HC	Tsurumi coal HC
Size of charcoal granule(mesh)	60 ∿ 200	60 ∿ 200
Charcoal content(wt %)	45	75

Following are the results:

(1) Collection efficiencies with and without the triethylenediamine(TEDA)-impregnated charcoal were found to be greater than 95 % and about 80 %, respectively.

(2) Air resistance across the new filter paper decreased to less than 1/2 (0.8mmHg rise/cm face velocity) compared with CP-20.

(3) Iodine retention efficiency of CP-20T during 24h following start of air flow was greater than 95 %, thereafter, there was found no more desorption of iodine.

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STUDIES ON THE RADIATION BURDEN USING ^{131}I FOR THYROID THERAPY *

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The ever increasing use of unsealed ^{131}I sources in the diagnosis and treatment of thyroid disorders (1) at nuclear medicine establishments throughout the world and the growing realisation of the radiation hazards associated with low doses of ionising radiation have led to studies on airborne contamination (2, 4, 7).

The present study aimed at establishing the levels of exposure to personnel from iodine-131 inhaled from unsealed radioactive sources and those to the personnel and non-isotope patients from ^{131}I exhaled by patients treated with doses of radioiodine ranging between 148 and 444 MBq. Variations in ^{131}I concentrations in the air exhaled were also investigated.

MATERIALS AND METHODS

Airborne ^{131}I concentrations were measured in the application room during laboratory and clinical procedures involving between 300 and 1000 MBq, and at 2-3 and 24 hrs after application in the ward room where the patients remained for 3-4 days after having been given ^{131}I for therapeutic reasons.

Airborne ^{131}I concentrations in the application room were found to range between 22 and 1358 Bq/m^3, whereas those in the ward ranged between 17 and 197 and 7 and 17 Bq/m^3 at 2-3 hrs and 23-24 hrs after application, respectively.

^{131}I was found in various form in the application room: that adsorbed on aerosols from 1 to 22%, in elemental state from 3 to 76% and in organic compounds from 18 to 96%, whereas in the ward room its percentage ranged from 1 to 15%, from 14 to 64%, and from 25 to 93%, respectively.

*This work was partly supported by the Alexander von Humboldt Fundation, F.R.G.

The apparatus for detecting airborne ^{131}I equipped with a Hay-pack filter (3) is described in (4) and is shown in Fig. 1. The sensitivity of detection ranged from 0.07 to 2.67 Bq/m^3.

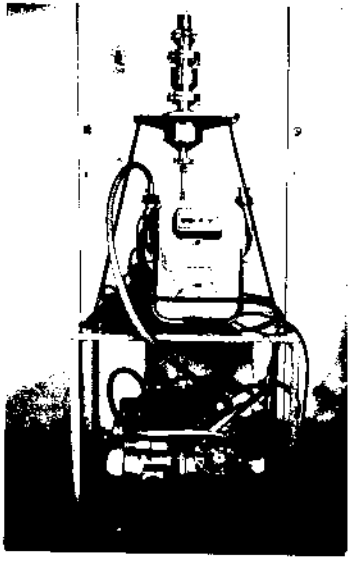

Figure 1. Assembly for sampling airborne ^{131}I in application and adjacent room.

Figure 2. Apparatus for sampling airborne ^{131}I exhaled by patients.

In order to study the concentration of ^{131}I exhaled by 10 patients treated with ^{131}I for hyperthyroidism and by a small group of 6 euthyroids a special assembly was made /Fig. 2/. It consisted of a glass tube with a Petrianov's fiber filter and two 2.5 cm thick NORIT CGI charcoal filters impregnated with 1% of KI. The tube is joined with a flexible tube inserted into a 24 l or 80 l plastic bag.

The ^{131}I concentrations in the air exhaled by patients were measured 2, 3, 5, 24 and 48 hours after oral administration of ^{131}I. Five patients apart from being clinically euthyroid had their 24-hour uptakes below 50%, which is assumed in Poland to represent the mean value for euthyroids and ^{131}I was administered to them to lower their metabolism in coronary insufficiency.

TABLE 1. The exposure per year of personnel and other persons during therapy with ^{131}I

Persons exposed, category of exposure and site of exposure	Ratio of ^{131}I concentration measured to the permissible levels for a given category of exposure	Estimation of the absorbed dose /per year/		
		Real time of exposure per year [h]	Activity inhaled A_i [kBq]	$\frac{A_i}{A_{MPA}}$ **
physicist, A				
application room	1.09	104	81.8	
adjacent room	0.01	1456	7.4	
			89.2	0.10
laboratory technicians, B				
adjacent room during laboratory procedures	0.18	104	2.3	
adjacent room	0.04	1976	10.1	
			12.7	0.05
physician, A				
ward room	0.18	312	22.8	0.03
nurse, A				
ward room	0.18	312	22.8	0.03
non-isotope patient, C				
ward room	5.26	216*	0.52	0.01

* the mean time of patient's stay in the hospital

** A_{MPA} = Maximum Permissible Absorption of ^{131}I through lungs for a given category of exposure

The kinetics of the ^{131}I exhalation in the hyperthyroid patients was found to consist of two widely different exponential processes (5) : one of a short half-life, $T'_{1/2} = 1.43 \pm 0.9$ hrs and the other, $T''_{1/2} = 17.5 \pm 8.6$ hrs.

The kinetics of the ^{131}I exhalation in euthyroids was, however, different, consisting of a sharp rise, with the maximum concentration occuring at 3-4 hours, and a fall faster than that in the hyperthyroid group /$T''_{1/2} = 5.53 \pm 1.03$ hrs/.

The exponential functions calculated from the following shape for hyperthyroid patients:

$$S/t/ = 0.93\ e^{-0.635\ t} + 0.06\ e^{-0.046\ t}$$

DISCUSSION

From the analysis of the exposure of the personnel /Table 1/ the ratio of the ^{131}I concentrations in the air to those permissible for a given category of exposure (3) were found to be exceeded in the case of a physician, and in that of an imaginary non-radioisotope patient placed in a radioisotope therapy room by a factor of 2 and 5, respectively. If, however, the real exposure time is taken into consideration, and the activity of ^{131}I in the body and that taken up by the thyroid is roughly estimated on a yearly basis, the ratio of the activity absorbed through the lungs to that permissible for a given category of exposure is not higher than 10%. The kinetics of the ^{131}I exhaled by hyperthyroid patients seems to be complementary to that of ^{131}I concentration changes in the blood and the curves for hyperthyroid patients can be expressed by two-component exponential equations.

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EVALUATION OF THE PROTECTION FACTOR OF HALF-MASKS WITH RESPIRATOR FITTING TEST APPARATUS

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It is important to evaluate the protection provided by a respiratory protective mask when worn. To select good performance masks and improve them, a knowledge is necessary of the variation of performance of the masks with wearers and their exercises. The authors constructed a respirator fitting test apparatus with some improvement, which essentially duplicates that developed by Hounam (1) and Hyatt (2), and measured the protection provided by half-masks used in a radioactive dust atmosphere. Described in this paper are the modified man-test apparatus and the test results. Emphasis was placed on quantitative evaluation of the effects of exercises on the protection.

MAN-TEST APPARATUS AND TEST PROCEDURE

Man-test apparatus: Sodium chloride aerosol was used as a test aerosol because of its low toxicity and ease of generation. As shown in Fig.1, the test aerosol was generated from a 1 per cent sodium chloride solution using a Wright design nebulizer operated with a pressure of 1.25 atm. The air from the nebulizer was diluted with dry air to produce an aerosol of dry sodium chloride. The aerosol was fed into a test hood at 50_2 l/min_3. Concentration and diameter of the aerosol particles were 10^{-2} μg/cm^3 and 0.47 μm (MMD), respectively. Air from inside the facepiece was continuously sampled at 4 l/min with a diaphragm pump through a lightweight flexible tube, permitting the subject to move his head freely. Filtered air at 6 l/min was added to the sample just behind the pump in order to dry the sample air, minimize deposition loss of the particles and quicken the response. The concentration of sodium chloride in the air was measured with a flame photometer. The minimum detectable leakage rate (MDL) of the apparatus was limited to 0.03 % due to deposition loss and aerosol dilution.

Test procedure: The subject put on a respirator and ensured its fitting by the negative pressure method before entering the hood. The sampling probe on the facepiece was connected to a sampling tube within the hood, leading to the measuring system, as shown in Fig.1. While the subject performed scheduled exercises, described later, an air sample was taken continuously from inside the facepiece to measure the concentration of sodium chloride.

The concentration was first measured without the canisters, and then with the canisters secured to the facepiece. The ratio of the second concentration (C_i) to the first concentration (C_0) was taken as the penetration. The protection factor PF was evaluated from PF = C_0/C_i. The effect of loss of particles by deposition in the respiratory tract was minimized by this test procedure, with comparative

measurements. The exercises were normal breathing, moving head, talking, and smiling. Each exercise was continued during the respective measurement.

The subjects were 44 adult men who had experiences in wearing the respirator. Four types of Japanese mask and two types of U.S.A's were tested. The canisters attached to the masks had filtering efficiencies of over 99.97 % for the sodium chloride aerosol under pulsating flow at a minute volume of 30 litters (3).

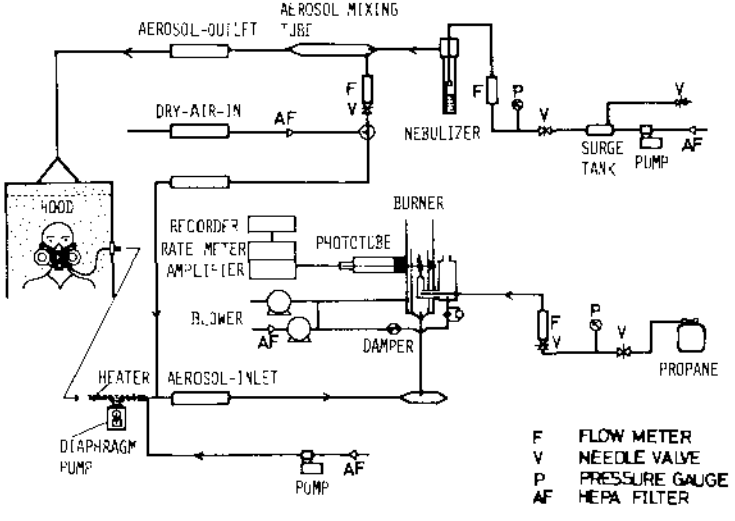

Fig.1 Flow sheet of the respirator fitting test system.

RESULTS AND DISCUSSION

Table 1 shows geometric means, geometric standard deviations, maxima, and minima of the protection factor (PF) for the masks in the exercises by the subjects. When a measured leakage was less than the MDL (0.03 %), the PF was calculated for the MDL of leakage. PFs of the masks for the various exercises are as follows.

Normal breathing: The geometric mean of PFs ranged from 340 to 600 for all masks and particularly 500 to 600 for the masks C,D,E and F. A maximum of PF is 3300 for the masks C,D and E which correspond to the MDL of 0.03 %. This maximum was obtained for 3 % of the subjects. The geometric standard deviation (δg) of PFs ranged from 2.1 to 3.2 in the normal breathing. The largest value of 3.2 for mask B represent large variation of the leakage in the mask with subjects, as compared with other masks. For masks A and B, δgs are 2.3 and 2.4, respectively, and for masks C,E and F the δg is from 2.1 to 2.2. As seen in the table, the performance of the masks A and B is worse because of the small values of geometric mean, maximum and minimum of PF, and in B, large δg of 3.2. On the other hand the masks C,E and F have good performance due to the large values of the geometric mean, maximum and minimum of PF, the minimum values exceeding 100.

Table 1 Summary of the values characterizing the distributions of protection factor(PF).

Mask	Number of subjects	PF,geometric mean				PF,standard deviation			
		NB*	S*	MH*	T*	NB	S	MH	T
A	44	340	120	85	82	2.3	2.9	3.0	2.3
B	44	390	68	120	98	3.2	4.1	3.0	2.4
C	42	530	250	250	240	2.2	3.1	2.1	2.1
D	44	500	97	100	120	2.4	4.1	2.4	2.3
E	42	600	190	300	240	2.1	2.7	2.0	2.0
F	42	470	230	260	230	2.2	2.6	2.2	1.9

Mask	Number of subjects	PF,maximum				PF,minimum			
		NB	S	MH	T	NB	S	MH	T
A	44	2900	900	750	400	17	3	3	3
B	44	1400	720	870	440	8	6	4	5
C	42	3300	1700	1200	1700	100	11	39	34
D	44	3300	3300	830	1000	68	5	6	11
E	42	3300	970	980	1100	120	19	39	64
F	42	2200	1700	1700	1200	100	29	25	35

* NB: normal breathing, S: smiling, MH: moving head, T: talking

Moving head, and talking: As seen in Table 1, these two had similar effects on the variation of PF. The PFs for these exercises were very low, compared with those for normal breathing. The mean of PF markedly decreased to 1/5 those of normal breathing in moving head for mask D. The mean PFs for masks A,B and D decreased to 1/3 to 1/5 . This is in accord with the fact that these masks have rather large σg of PFs even in the normal breathing. Mean PFs of the masks C,E and F were 1/2 to 1/3 those in the normal breathing, and were larger than those of A,B and D. This is consistent with the small σg values of masks C,E and F in the normal breathing. The minima of PFs markedly decreased with these exercises, to less than 10 in masks A,B and D particularly. Nevertheless, the minima in masks C,E and F are relatively high between 25 and 64; the masks are thus superior in this respect.

Smiling: This exercise led to relatively large drops of mean,maximum and minimum of PF compared with the other exercises, and increased the scatter of data (σg). This is probably caused by the leakage through wrinkles of the cheek in smiling. Even in this exercise the masks C,E and F showed good performance, e.g. means 200 and minima more than 10.

Distribution of PFs: The masks C,E and F had similar performance for different exercises. Fig.2 show plots of cumulative percentage of PF for masks C,E and F on log-normal probability paper. As seen in the figure, the values of PF for the exercises are approximately distributed in log-normal form. The data for the normal breathing fit well to a log-normal distribution.

It is seen from Fig.2 that moving head and talking influence the PFs to almost same extents and that smiling raises the scatter of PFs. The expected values of PF for an exercise most liable to leakage, i.e. smiling, is about 30 in reliability of 95 %. Solid lines in the figure are drawn based on statistics of the measured values.

Shape of masks vs. fit: Fig.3 shows external appearance in fitting contour with the face of the tested masks, viewed from the direction of the wearer and from the side. The masks C,E and F with good performance possess the following features. (1) The mask is long in the part in contact with nose bridge. (2) The length of vertical line is about 120 mm. (3) In the side view, the part in contact with the upper nose bridge is curved and the other part is nearly straight. Of the tested masks, mask C and D have the same shape, the D having metalic lining.

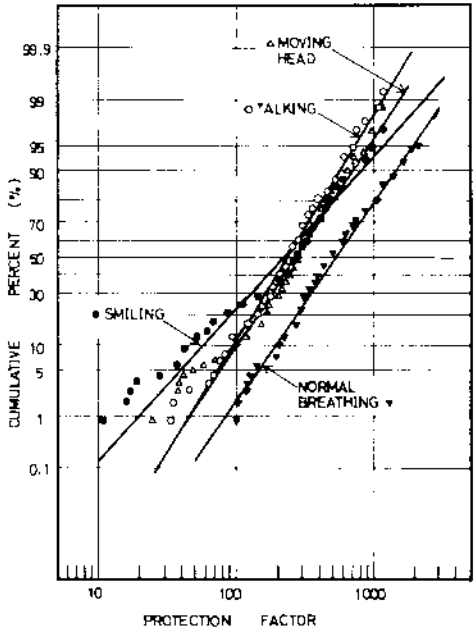

Fig.2 Plots of protection factor of masks C,E and F on the log-normal probability paper.

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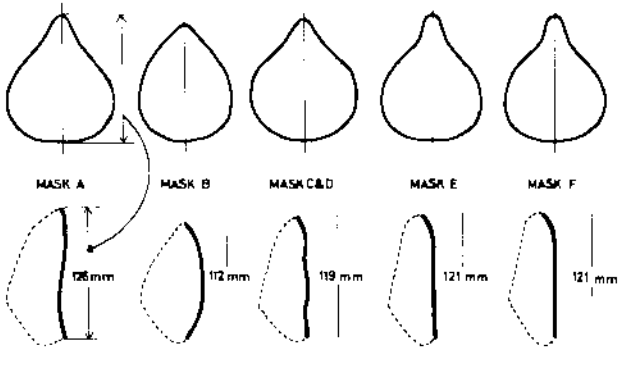

Fig.3 The contours of the half-masks used in the test.

TITRE : Contribution à l'irradiation de l'Am 241 présent dans le Plutonium. Risques radiologiques liés aux manipulations de l'Am 241.

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I - POSITION DU PROBLEME

En fonction du taux de combustion de l'uranium, ainsi que du type de réacteur considéré, le Plutonium extrait, après retraitement du combustible, peut contenir des quantités variables de Plutonium 241. Ce dernier, en raison de sa courte période, 14, 5 ans (1), est à l'origine de quantités croissantes dans le temps, d'Américium 241, de période 433 ans.

L'émission principale de l'Américium 241 est un photon de 60 KeV émis avec un pourcentage de 36 %(Cependant, quand les quantités deviennent importantes, les raies d'énergie supérieure ne doivent pas être négligées dans les calculs des protections).

Le problème consiste à déterminer les limites supérieures des quantités d'Américium, acceptables dans un Plutonium irradié, sans dépasser les doses maximales admissibles dans des installations manipulant du Plutonium irradié en boîtes à gants classiques.

Le tableau 1 donne la contribution des divers isotopes présents dans un Plutonium irradié en fonction de l'âge pour une teneur initiale de 2 % en Pu 241.

TABLEAU I - Contribution de chaque isotope et débit de dose γ total (mrad/h) en fonction de l'âge au contact d'un lingot de Pu irradié.

Age / Isotope	0	1	3	5	10	15	20
Pu 239	1140	1140	1140	1140	1140	1140	1140
Pu 240	880	880	880	880	880	880	880
Pu 242	-	-	-	-	-	-	-
Pu 241	57	54	48	43	35	26	20
U 237	0	163	147	132	103	80	60
Am 241	0	676	1923	3000	4200	7000	8400
TOTAL	2077	2913	4138	5195	6358	9126	10500

Les limites dépendent du vieillissement du Plutonium et du pourcentage initial de Pu 241 ; elles sont également liées aux protections additionnelles pouvant être mises en place sans gêne ni dépenses excessives.

Si ces limites sont dépassées quelles solutions peut-on adopter ? Automatisation, télémanipulation ou éventuellement séparation de l'Américium.

II - METHODES DE PROTECTIONS, SOLUTIONS PROPOSEES

L'expérience et le calcul font ressortir une teneur maximale d'Américium, acceptable dans le Plutonium sans modification des installations classiques. Cette limite se situerait vers 5000 ppm (0.5 %) Ceci correspond à un vieillissement de 10 ans environ pour un Plutonium contenant initialement 1 % de Pu 241.

Si la teneur en Am 241 est inférieure à cette limite, les manipulations peuvent s'effectuer en BàG (boîtes à gants) classiques avec certaines protections supplémentaires plus ou moins légères.
- mise en place de panneaux additionnels en verre au plomb de densité moyenne,
- adoption de gants au plomb lorsque la nature et la précision des opérations le permettent,
- limitation des opérations manuelles,
- nettoyage fréquent des BàG afin d'éviter l'accumulation de dépôts de Pu-Am qui peuvent être très irradiants.

Pour des concentrations supérieures à 5000 ppm, les DMA (doses maximales admissibles) annuelles risquent d'être dépassées, en particulier pour les mains et avant-bras, car les mesures d'irradiation montrent que les temps d'exposition sont 10 fois environ plus limitatifs pour les extrémités que pour l'organisme entier. On peut alors envisager d'autres solutions :
- automatisation des chaînes de fabrication : par exemple par l'adoption de machines à commande numérique, la télécommande des transferts,
- manipulation à distance voire télémanipulation à certains postes de travail.

Ces aménagements se heurtent souvent à certaines difficultés; d'une part, ils ne conviennent pas à tous les postes, d'autre part, les tolérances de précisions ne le permettent pas toujours, enfin, les limitations en masse des unités de travail exigent souvent des transferts entre cellules indépendantes.

Il reste une possibilité: l'extraction de l'Américium afin de rendre le Plutonium réutilisable sur les installations classiques. L'Américium récupéré ne pose pas, à court terme, de problèmes de stockage insolubles; à long terme cependant, des solutions doivent être étudiées.

III - RESULTATS EXPERIMENTAUX

1) Facteurs d'atténuation de quelques matériaux de protection

TABLEAU II

Nature	Pu à 3000 ppm Am	Pu à ~ 4 % Am	Am pur (nitrate)
Plexiglass (ép.= 8 mm)	1,65		1,2
Sandwich verre : 1,4 cm plexi : 0,8 cm plomb : 0,2 cm	~ 50 (3)		
Gant ordinaire au néoprène	~ 1,5		
Gant au plomb	~ 4	1,8	
Plomb (2mm)		200 (1) 180 (2)	400 (1)
Plomb (4mm) (4)			650 (1) 200 (2)

Indices du Tableau
(1) en phase solide
(2) en phase liquide
(3) ce facteur est réduit par les fuites au niveau des ronds de gants
(4) la facteur d'atténuation du plomb varie avec son épaisseur en raison de la largeur du spectre d'énergie

2) Unité expérimentale de séparation

A) Principe : résines échangeuses d'ions avec fixation du Pu et entraînement de l'Américium. L'Américium récupéré est ultérieurement concentré par un évaporateur en continu et stocké définitivement à la concentration désirée. Le cycle porte sur 800 grammes environ de Pu et dure une quinzaine d'heures pour les phases fixation-élution.
B) Protection : La séparation s'effectue en milieu liquide dans des colonnes en pyrex atténuant sensiblement l'irradiation. De plus, une BàG " transfert " permet la commande à distance. Les stockages intermédiaires et définitif sont dans un local protégé par 20 cm de béton avec une protection supplémentaire de 10 cm de plomb autour du stockage définitif d'Américium très concentré. Compte tenu du nombre réduit d'opérations manuelles, les doses reçues par les agents, restent en dessous des DMA, le poste le plus pénalisant étant l'évaporateur lors des reconcentrations finales.
C) RESULTATS
- Débits maximaux aux postes principaux
  . haut de la colonne de résines : ≈260 mrad/h
  . contact de la BàG évaporation : ≈100 mrad/h
  . dose intégrée par agent par cycle : ≈45 mrad

3) Débits de dose calculés au contact d'un container de 500 g d'Am O_2

TABLEAU III

	Source nue	Ecran de plomb (ép. 0,1 cm)	Ecran de plomb (ép. 1 cm)	Ecran de Plomb (ép. 5 cm)
γ (mrem/h)	6.10^5	2640	390	2
$\frac{1}{0}\eta$ (mrem/h	1400	1400	840	200
TOTAL (mrem/h)	$6,014.10^5$	4040	1230	202

IV - CONCLUSIONS

Le Plutonium irradié exige non seulement une surveillance rigoureuse de la contamination, mais aussi une attention accrue au niveau des postes de travail. Selon les procédés de fabrication et de manipulation, la teneur en Américium, les protections en place, les équivalents de dose peuvent être prohibitifs soit au niveau des mains, soit au niveau de l'organisme entier.

Pour des installations à protections légères, la teneur en Américium qui nous paraît être la limite supérieure en routine se situerait aux environs de 5000 ppm.

Seule une étude de poste préalable permettra d'ajuster cette valeur. Au delà d'une certaine limite et en fonction d'une réutilisation du Plutonium, la séparation Pu-Am s'avèrera nécessaire. Les protections autour de l'installation de séparation seront fonction du procédé choisi.

L'Américium obtenu soit en phase solide, soit en phase liquide pose des problèmes pour son stockage en particulier sous forme AmO_2, car dans ce cas l'irradiation neutronique due aux réactions (α , n) devient importante.

V - REFERENCE

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RADIATION PROTECTION PROBLEMS AT COMPACT CYCLOTRONS FOR MEDICAL AND OTHER USE

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INTRODUCTION

Small high yield cyclotrons (compact cyclotrons) will be widely used in the future in medicine, biology and solid state physics. Since they are strong sources of fast neutrons and γ-radiation a number of radiation protection problems has to be solved to achieve a proper protection of workers and environment. The main problems are

a) shield design
b) induced activity
c) exposure to workers

These problems were studied by us in the last five years at the compact cyclotron CV-28 of the Juelich Nuclear Research Centre.

SHIELD DESIGN

The performance data of the cyclotron are given in table 1 (1). The effectiveness of ordinary concrete as shielding material against fast neutrons and γ-radiation from (d,n)-, (p,n)-, (^{3}He,n)-, and (^{4}He,n)-reactions was studied in the experimental shield testing facility at the cyclotron using gas targets and solid targets, such as beryllium, carbon, aluminium, iron, copper and tantalum (1). The results of the first series of experiments on ordinary concrete (1,2) can be summarized as follows:

a) The worst case target is beryllium (see table 2).
b) γ-radiation is sufficiently reduced by the neutron shield (1).
c) The neutron yield of gas targets (14-mev-deuterons on CO_2- or CO-targets) and accordingly the equivalent dose rate behind thick shields only amounts to about 20% of the respective values for the beryllium target.

INDUCED ACTIVITY

A i r

From the point of view of radiation protection the air contamination in the cyclotron target room by induced activity is one of the limiting factors for an effective operation of the compact cyclotron. Usually, this contamination is predicted by mathematical evaluation using sphere or cone models (3). We checked these theoretical models for their validity in practice and found out (4), that they are not suitable to represent the real distribution of induced activity in the air of target rooms. As shown in figure 5, this distribution is

nearly uniformly due to the uniform "neutron climate" in small and well-shielded rooms. The theoretical results are on the safe side for short distances from the target.

Construction material

Target and beam tube

The access to target rooms is limited by the high radiation field originating from induced activity in target and beam tube. In figures 6 and 7 typical decay curves are shown for the dose rate of the target case surface (representative for target handling) and in 1 m from beam tube flange (representative for the time of access to target rooms).

Shield

The cyclotron shield is made of reinforced ordinary concrete, whereas the test slabs used in the shield testing facility were manufactured of ordinary and limonite concrete, respectively. The activation of these shielding materials is small as compared to the activation of materials in the target area.
Table 3 shows typical values found by γ-spectrometric evaluation. The main nuclide found was Mn-56.

EXPOSURE TO WORKERS

Handling of short-lived high-activity radionuclides (up to 1 Ci) such as C-11 (half-time 20 min) possibly results in high whole body and hand doses to the workers concerned. Figures 8 to 10 give the dose record for these workers (about 50 on the average) from 1971 to 1979 (5). The following is evident from these figures:

a) Collective doses increased during the period (1972-1977) of introduction and test of new methods for the production and manufacture of short-lived radionuclides. They could be noticably lowered with increasing experience, stepwise improvements of radiation protection and automation of experimental procedures.
b) Average annual whole body doses never exceeded 10% of permissible dose values.
c) The average specific dose could be lowered to 0.23 mrem · mCi^{-1} (whole body) and 6.6 mrem · mCi^{-1} (hands), respectively.

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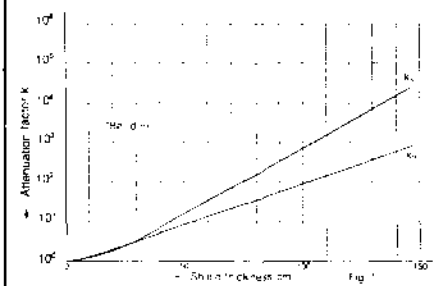
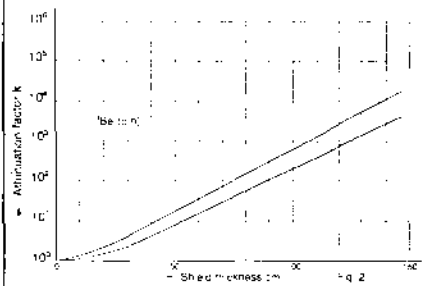
Particle	E MeV	A_{max} μA	
		External	Internal
P	7-24	70	500
D	3-14	100	500
^{3}He	5-36	70	150
^{4}He	6 28	50	100

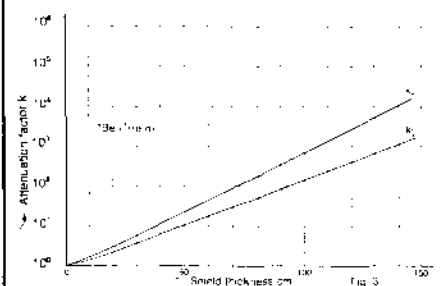
Table 1: Energies and particle current strengths of compact cyclotron CV-28.

Target	Tenth value thickness cm			
	d 14 MeV	p 24 MeV	^{3}He 35 MeV	^{4}He 28 MeV
^{9}Be	30,6	32,5	33,8	39 0
^{12}C	30,4	29,8	33,0	28 5
^{27}A.	28,8	30,3	31,7	33 2
^{56}Fe	28,5	29,6	34 0	33,2
^{63}Cu	29,8	31,1	32,6	36,5
^{181}Ta	28,9	30,2	32 3	38,5

Table 2: Tenth value thickness in cm for different targets and particles.







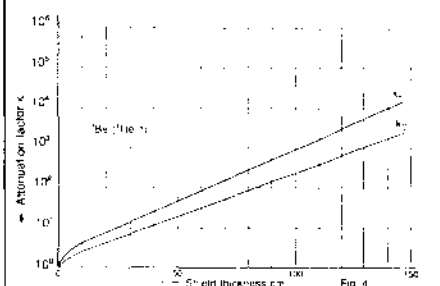


Fig. 1 to 4: Attenuation factor as a function of shield thickness of ordinary concrete for thick beryllium targets

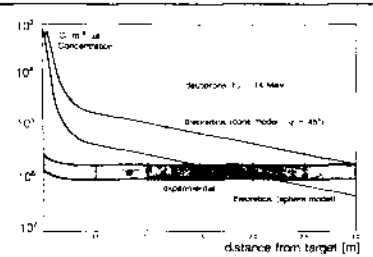


Fig. 5: Air contamination in the target room by induced activity, comparison of theoretical and measured values.

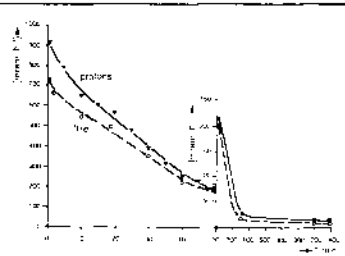


Fig. 6: Dose rate at target case surface as a function of decay time; target current: 1 μa

Limonite concrete			Ordinary concrete		Iron	
Nuclide	Half life	Reaction	Nuclide	Reaction	Nuclide	Reaction
Na-24	15 h	$^{23}Na(n,\gamma)^{24}Na$	Na-24	$^{23}Na(n,\gamma)^{24}Na$		
Mn-52	5.7 d	$^{55}Mn(n,4n)^{52}Mn$			Mn-52	$^{56}Mn(n,4n)^{52}Mn$
Mn-54	297 d	$^{55}Mn(n,2n)^{54}Mn$			Mn-54	$^{56}Mn(n,2n)^{54}Mn$
Mn-56	2.6 d	$^{55}Mn(n,\gamma)^{56}Mn$	Mn-56	$^{55}Mn(n,\gamma)^{56}Mn$	Mn-56	$^{56}Mn(n,\gamma)^{56}Mn$
K-40	$1.4 \cdot 10^9$a	natural	K-40	natural		
Induced activity $\mu Ci \cdot \mu a^{-1} \cdot h^{-1} \cdot g^{-1}$						
0.007			0.002		0.019	

Table 3: Induced activity in shielding material at the compact Cyclotron CV-28; ^3He on Be-target, E = 36 MeV, irradiation time: 3 h.

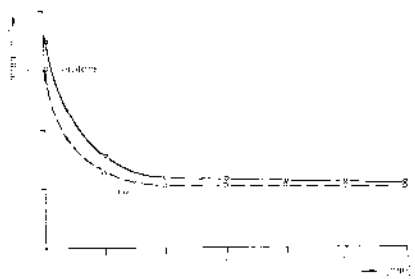


Fig. 7: Dose rate 1m from beam tube flange after dismantling of target as a function of decay time.

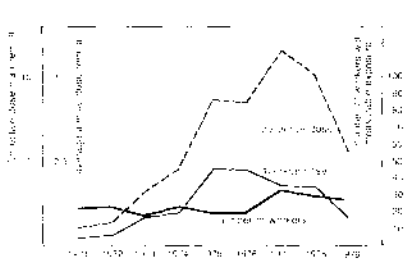


Fig. 8: Collective and average annual whole body dose from 1971 to 1979.

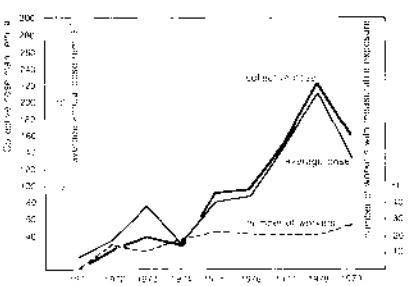


Fig. 9: Collective and average annual hand dose from 1971 to 1979.

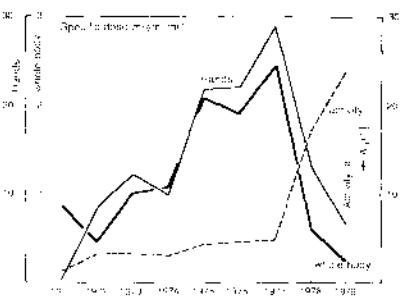


Fig. 10: Specific whole body and hand dose from 1971 to 1979.

DEVELOPMENT OF GUIDELINES FOR INCORPORATION MONITORING PROGRAMS

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1. INTRODUCTION

Federal regulations for radiation protection in Austria (1) require routine checks to be made on occupationally exposed persons if there is a risk of incorporation of radionuclides. No detailed requirements of a practicable routine program of bioassay, however, are given for such a surveillance. It was therefore necessary to develop a special monitoring program for each radionuclide taking into account single, recurrent and continous intakes.

2. PERFORMANCE AND RESULTS OF MONITORING PROGRAM

2.1 Control examinations of workers preparing organic compounds labelled with carbon 14

Various organic substances are synthesized in our center routinely for labelling with carbon 14. The employees are therefore controlled by urinanalysis and the results of a one year's period are shown graphically in fig. 1. Discussing the obtained values in fig. 1, it can be noticed that mainly three higher incorporations must have occurred. An overall estimation based on the mean of incorporated activity gave an amount of $5{,}2.10^{7}$Bq carbon 14 yielding a whole body dose of 10 mSv and a dose commitment of 16 mSv for the bone. If the amount of activity handled is related to the incorporated quantity it gives a ratio about $1{:}10^{-5}$.

2.2 Control examinations of workers preparing organic compounds labelled with tritium

In another laboratory also organic compounds are labelled with tritium. One person processed 1,85 to $3{,}70.10^{1}$ Bq of tritium in the form of T_2 or HTO. The type of handling varied from storage to complex wet-handling techniques involving the danger of spilling liquids and the use of radioactive compounds with high vapor pressure. The results of these examinations are presented in Fig. 2.
If one assumes a constant tritium concentration in urine at the level of the arithmetic mean of all measurements which is 14,4 Bq/ml this would lead to an estimated average dose of 720 µSv for the period mentioned.
If the data presented in fig. 2 are interpreted as results of a recurrent intake, a maximum dose of 4,5 mSv and minimum dose of 400 µSv may have been accumulated over the control period of nearly 2 years.

Evidently, the difference between the extreme values increases with a longer time period between every two samplings. Therefore it is reasonable to use a sampling period not exceeding three times the biological half-life, i.e. one month for tritium.

2.3 Excretion analysis for ^{90}Sr

The handling of radioactive waste - including the gathering sorting and conditioning - involves a certain incorporation risk for the employees.
All people therefore engaged with radioactive waste were routinely surveyed by quarterly urinanalysis (3).
Generally ^{90}Sr-values between 0,2 and 0,02 Bq/l were registered. Higher ^{90}Sr-contents over 0,8 Bq/l ^{90}Sr in urine were only caused by two persons, who incorporated ^{90}Sr during an incident. They were afterwards examined for three month and the observed excretion curves were in good agreement with the biological half-lives cited by ICRP (4).
For the most unfavourable case, a body burden of 592 Bq ^{90}Sr, deposited in the bones, was calculated and a resulting dose commitment of 6 mSv had to be expected.
For comparison, the urine samples of 21 different persons, who were working in inactive areas, were analysed too and the mean value of 32 determinations was 0,031 $\pm$ 0,013 Bq/l
If intakes of the global fallout are included and a variation of the calcium content between 100 - 400 mg Ca per liter urine is taken also into account (5), a possible ^{90}Sr-content of $7{,}8.10^{-3}$ to $7{,}8.10^{-2}$ Bq/l results.
If possible, individual differences in metabolism, feeding habits etc. are considered a ^{90}Sr incorporation above fallout level should be assumed with certainty, only if the ^{90}Sr concentration exceeds 0,2 Bq/l of urine.

2.4 Control examinations of workers in the nuclear fuel section

From technicians engaged in the analysis of fuel elements containing natural uranium regularly an urine sample is taken and the isolated uranium determined by fluorometry (6, 7).
If ^{233}U or enriched uranium can be present, the urine sample must be examined by alphaspectrometry. Datas are given in table 1.

2.5 Excretion analysis for plutonium

The investigation level for ^{239}Pu lies at 1,48 Bq (3), causing a dose commitment of 15 mSv for the bones throughout the following fifty years. If no further incorporation occur a yearly dose of 0,3 mSv will result. If the critical level is set at 3 mSv per year and a steady state is assumed an activity of 3.10^{-2} Bq of ^{239}Pu will

be excreted per day and can be detected easily by monthly urine analysis as shown in table 1.

2.6 Monitoring of ^{99}Mo and ^{99m}Tc

In our center daily several Curies of ^{99m}Tc for medical applications are delivered to hospitals in Vienna. The effective half-life of 1,8 for ^{99}Mo and 0,2 days for ^{99m}Tc make an evaluation of the urine analysis rather difficult but can be best interpreted as a form of continous intake. Results of a survey performed by a body-counter shows a ratio between the handled and incorporated activity of $1:10^{-7}$.

2.7 Monitoring of iodine 131

Various labelling procedures in the $7,8.10^{9}$ Bq range with ^{131}I made a monitoring necessary. The survey was being made with the special incorporation monitor of our institute which has already been described.
Correlation between handled and incorporated activity gives a ratio of $1:10^{-7}$.

2.8 Monitoring of iridium 192

In one section on the isotope production, ^{192}Ir-sources are prepared routinely under strict precautions. In the monitoring interval about $1,5.10^{14}$ Bq of ^{192}Ir were processed and the employees surveyed by whole body counting. The ratio of handled and incorporated activity was $1:10^{-12}$

3. GUIDELINES FOR THE DEVELOPMENT OF SURVEY PROGRAMES

The described cases show clearly that despite of various precautions incorporations of radioactive substances can not be completly avoided.
It was also demonstrated that the risk of incorporation varies with the amount, the chemical form, the toxicity and the way of handling of the used radioactive substances Generally before setting up a survey program two main questions arise:
- above what activity level incorporation monitoring is necessary
- what time intervals must be chosen to insure an effective monitoring.

For the answer of these questions we have defined firstly a so called "handling factor".

$$H_f = \frac{A.I_f.100}{M_1} \qquad (a)$$

A = activtiy /Bq/; H_f = handling factor; I_f = incorporation risk factor; M_1 = monitoring level (= 1 % of the maximum permissible body burden of radionuclides of the tox.class 1 (1) and 10% for all other radionuclides /Bq/).

From the handling ways described in the foregoing chapter, the following in table 2 listed incorporation risk factors could be derived.

TABLE 2: Incorporation risk facotrs (I_f)

Physical state	way of handling: routine	with special precautions	incidents
gaseous and dusty material	10^{-5}	10^{-7}	10^{-3}–10^{-2}
liquids	10^{-7}	10^{-9}	10^{-5}–10^{-3}
solids	10^{-10}	10^{-12}	10^{-8}

If the calculated handling factor is $H_f \geqslant 1$, regular monitoring procedures are necessary.
The corresponding time intervals should be defined in such a manner that a single intake in the magnitude of the fixed monitoring level in the most unfavourable case that is at the beginning of the monitoring interval can be determined safely. For fixing the time intervals the following formula is proposed:

$$M_1 . Y(\Delta t) \leqslant 2\ D_1 \qquad (b)$$

M_1 = monitoring level see formula (a)
Y = value of the excretion function for the time interval Δt
Δt = time interval between two samplings
D_1 = detection limit for the radionuclide

By using the proposed formulas "a" and "b" a special monitoring program for each radionuclide can be set up. Further datas can be found in table 2.

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TABLE 1: GUIDELINES FOR THE PERFORMANCE OF PERSONAL MONITORING

Radio-nucl.	Detection limit (Bq/l)	Monitoring level (Bq)	Dose Commitm. for monitoring level (mSv)	Concentration in Urine (Bq/d) Normal (Bq/l)	after 30days	after 60days	after 90days	after 180days	Recommended interval sampling
^{3}H	74	$3,7.10^{6}$	0,17 Whole B.	185-370	$3,2.10^{4}$	$4,03.10^{3}$	518	-	1 month
^{14}C	74	$1,1.10^{6}$	0,22-0,33 W.B.	-	170	74	-	-	1 month
^{90}Sr	$3,7.10^{-3}$	$7,40.10^{2}$	7,8 Whole Body	$7,8.10^{-2}$	1,0	0,44	0,26	0,11	3 months
^{131}I	$7,4.10^{-3}$	$2,6.10^{3}$	1,16(Thyroid)	-	-	-	-	-	1 month
Unat	0,3(µg/l)	1,5 mg	-	0,3(µg/l) 50 µg permanent					14 days
233/234 Uran	$1,85.10^{-3}$/ 24 h	18,5	0,8	$1,1.10^{-2}$	$4,0.10^{-3}$	$2,82.10^{-3}$			12 months
^{239}Pu	$7,4.10^{-4}$	14,8	2,85 Whole Body	2.10^{-3}	$2,37.10^{-3}$ permanent				1 month

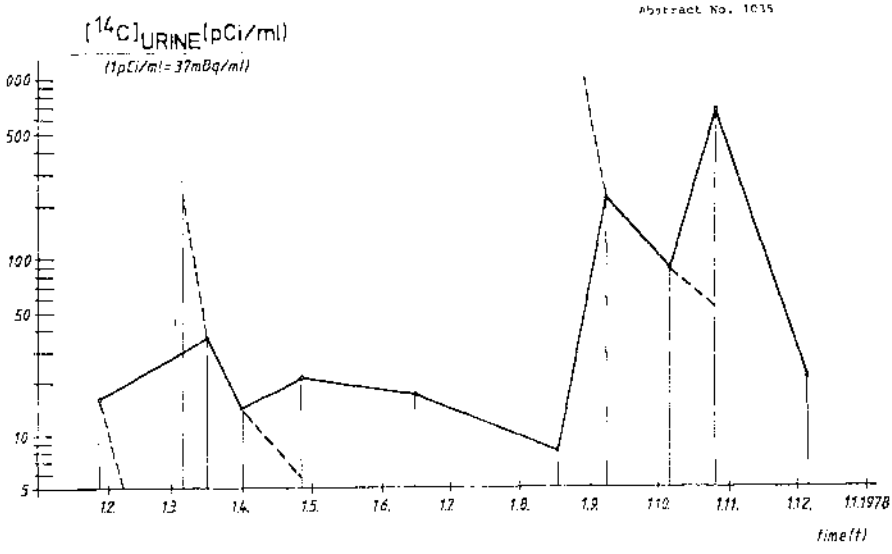

Fig.1 ^{14}C Concentration in urine after labelling work

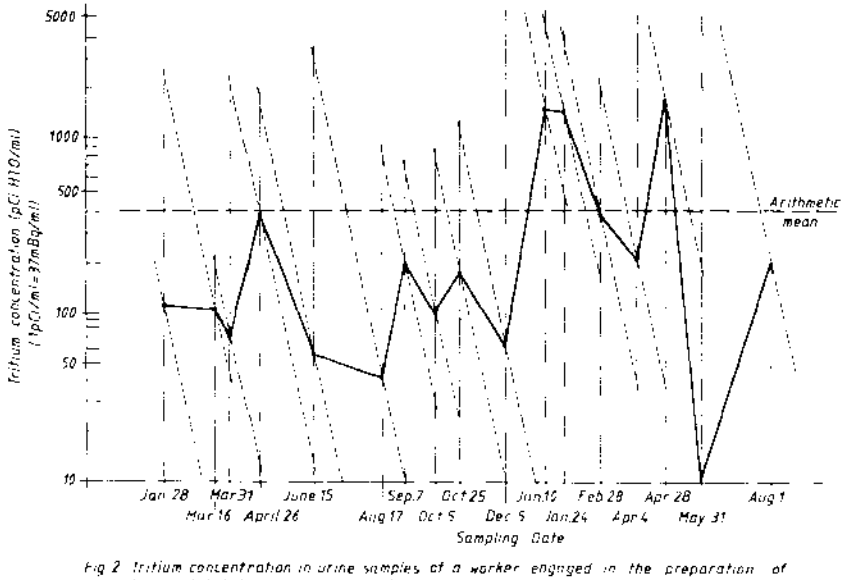

Fig 2 Tritium concentration in urine samples of a worker engaged in the preparation of tritium-labeled organic compounds

CHARACTERIZATION OF WORKING CONDITIONS FOR HANDLING RADIONUCLIDES.

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The handling of materials containing radionuclides, involves the possibility of exposure to ionizing radiation of workers and members of the public. A usual way to restrict these exposures to acceptable levels, is to limit the activity of the radionuclide to the handled.

A wellknown system for determining this limiting amount is described in IAEA Safety Series nr 1 Safe Handling of Radionuclides. This system, that can be found in many codes of practice, gives for each class of radiotoxicity a limiting amount for normal operation. For other types of operation and different types of operating facilities, the amount for normal operation is multiplied or devided by one or more factors of ten.

The assumption this system is based on is that the major hazard that has to be considered is the inhalation hazard. However, this is not always the case. Moreover the inhalation hazard may vary considerably more than is assumed in this system.

In this paper simple formulas are given for estmating the limiting amount for different working conditions. The formulas are based on a rude estimation of the radiation hazard for different modes of exposure. The hazards cosidered are external radiation hazard for workers and inhalation hazard for workers and members of the public.

For radiation workers exposure is considered to be acceptable if the average effective dose-equivalent is limited to 5 m Sv (0,5 rem) per year. For members of the public it is assumed that the average effective dose-equivalent must be limited to 50 µ Sv (5 mrem) per year.

RESTRICTION OF EXTERNAL RADIATION HAZARDS FOR WORKERS.

For external radiation hazard of workers, the working conditions can be characterized by the dose-rate around the unshielded source, the average working distance, the shielding between source and worker and the average exposure time. If we assume that for operations taking place continuously, a dose of 5 m Sv per year will be received if the dose-rate at the working place is limited to approximately 10 µ Sv (1 mrem) per hour, than the maximum amount that can be handled can be estimated using the following formula:

$$A_{me} = A_S . 10^{x+y+z-6} \qquad (1)$$

where A_{me} is the maximum amount in Bq.

A_S is the "standard activity of the radionuclide involved, that is the amount in Bq giving a dose-rate of 1 Sv/h at 1 meter.

x is the parameter for working distance:
x=0 - work close to source, distance about 0,3 meter;
x=1 - working distance about 1 meter;
x=2 - working distance several meters.

y is the parameter for shielding:
y=0 - no shielding;
y=1 - simple shielding facilities;
y=n - shielding facility that reduces dosa-rate by a factor 10^n.

z is the parameter for exposure time:
z=0 - continuous exposure;
z=1 - exposure during 10 % of the working time;
z=2 - exposure during 1 % of the working time(transport, storage).

RESTRICTION OF INHALATION HAZARD FOR WORKERS.

A small fraction of the radionuclides to be handled may become airborne. The size of this fraction depends on the physical form of the radioactive substance and the type of operation involved. If the operation takes place in a containment, only part of the airborne material will be dispersed into the room air. Finally only a fraction of the activity dispersed, will be inhaled by the worker because of general ventilation provisions in the room.

If we want to restrict the activity inhaled in a year to 0,1 ALI, which corresponds with a radiation exposure of 5 m Sv (0,5 rem) in a year(ICRP-publication-26) the maximum amount to be handled can be calculated if certain values are assumed for the different fractions mentioned above. This is expressed in the following formula:

$$A_{mi} = ALI.10^{p+q+r+s-4} \qquad (2)$$

where A_{mi} is the maximum amount in Bq.

ALI is the annual limit of intake by inhalation.

p is the parameter for the physical form:
p=0 - gas, aërosol;
p=1 - dust, liquid with temperature around boiling point;
p=2 - liquid, solution.

q is the parameter for dispersion risk:
q=0 - very high dispersion risk;
q=1 - high dispersion risk;
q=2 - normal dispersion risk;
q=3 - small dispersion risk;
q=4 - very small dispersion risk.

r is the parameter for containment:
r=0 - no extra containment;
r=1 - local exhaust;
r=2 - fume hood;
r=3 - ventilated closed cabinet.

s is the parameter for general ventilation:
s=0 - poor ventilation condition;
s=1 - normal ventilation condition;
s=2 - good ventilation condition.

For the parameters p, r and s the value to be chosen will in most cases be obvious. The choice of q is however, more difficult. It might be useful to know that in setting up formula (2) it is assumed that the total activity that is becoming airborne during a year, is the fraction 10^{-p-q+3} of the amount that is handled. This corresponds with a fraction of about 10^{-p-q-1} for a week or 10^{-p-q} for a day.

RESTRICTION OF INHALATION HAZARD FOR MEMBERS OF THE PUBLIC.

The amount of activity that becomes airborne will be released into the environment by the exhaust system for the ventilation air. Thus the total amount released in a year will be the fraction 10^{-p-q+3} of the amount that is handled. If the exhaust contains a filtersystem, the release will be reduced by a factor taking into account the efficiency of the filtersystem.

Members of the public living in the vicinity of the laboratory will inhale small fractions of the amount that is released. The size of this fraction is determined by atmospheric dilution processes. In most cases this process can be described by assuming a wake off-wind of the building in which the released activity is mixed homogenously. Using this model it can be shown that for a building cross-section of 200 m^2 and a wind velocity of 1 m/s, the fraction inhaled will be about 10^{-6}. Because the amount inhaled must be limited to 10^{-3} ALI, the maximum amount to be handled will be given by the following formula:

$$A_{mI} = ALI.10^{p+q+a} \qquad (3)$$

where A_{mI} is the maximum amount in Bq.

a is the parameter for filterefficiency:
a=0 - no filter present;
a=1 - filterefficiency 90 %;
a=2 - filterefficiency 99 %;
a=3 - filterefficiency 99.9 %.

FINAL REMARKS.

When the formulas presented in this paper are applied in normal laboratory practice, the external radiation hazard will appear to be the limiting factor in many cases. The inhalation hazard for members of the public will seldom restricted the ammount to be handled, especially if the exhaust ventilation contains appropiate filters.

It will be clear that if more radionuclides are handled in one working place, extra limitation might be necessary. In this case, for each of the modes of exposure, the amounts handled must fulfil the following condition:

$$\sum_n \frac{A(n)}{A_m(n)} \leq 1 \qquad (4)$$

where A(n) is the activity of radionuclide n.

$A_n(a)$ is the maximal amount for radionuclide n as calculated by formula (1), (2) or (3).

Finally it must be remarked that the system presented here is only meant to get an idea of the order of magnitude of the activity that can be handled safely under specified working conditions. It must always be kept in mind that radiation and contamination measurements will finally have to answer the question wheather working conditons for handling radionuclides are acceptable.

CHEMICAL PROTECTION AND SENSITIZATION TO IONIZING RADIA - TION:MOLECULAR INVESTIGATIONS

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INTRODUCTION

Chemical radioprotection and radiosensitization are the phenomena induced by the presence of certain chemical compounds, which reduce or enhance respectively the effect of ionizing radiation on living organisms. Such substances are either naturally present or may be artificially introduced in the living cells. When these phenomena occur in complex biological systems, they are a synthesis of many processes of physical, chemical and biological nature. The study of the mecahnisms of chemical radioprotection and radiosensitization could also aim at a better understand - ing of how radiation acts on cells and tissues.

Chemical radioprotectors are interesting for the possible application in health protection of both professionally exposed workers and patients treated by radiation for diagnostic and therapeutic reasons. Although the initial enthusiasm has been not paid by the success of finding the "anti-radiation pill", the problem is so important that such studies are still up to date.

Chemical radicsensitization has boomed in the last years for its potential application in the radiotherapy of tumours since even a modest increase of radiosensitivity of neoplastic cells results in a better therapeutic treatment.

The main classes of radioprotective and radiosensitizing drugs include compounds with respectively reducing and oxidizing properties towards the radiation induced radicals derived from biological molecules. Both processes of radioprotection and radiosensitization occur by means of complicated mechanism, whose the very early stages correspond to very fast reactions. The mechanism of action of such sub - stances can be investigated by means of radiation chemical techniques, i.e. pulse radiolysis (1). Briefly pulse radiolysis uses a short intense pulse of radiation to induce the initial physical-chemical damage and fast recording technique (i.e. absorption kinetic spectrophotometry with oscillographic output) are used to investigate the short-lived

chemical species produced, and to follow their subsequent pathway.

CHEMICAL RADIOPROTECTION.

Chemical radioprotection is still an important field of research in radiobiology at all levels of biological complexity. Different types of substances, including aliphatic alcohols, sulphur or selenium containing compounds etc., offer radioprotection "in vitro" and "in vivo". The most important class of radioprotective substances includes the sulphur-compounds. These compounds can act by different mechanisms: protection by mixed-disulfide formation protection by radical scavenging and protection by the hydrogen transfer mechanism. Of particular interest is the repair model by the hydrogen transfer mechanism, originally proposed by Alexander and Charlesby (2), and further applied to biological systems by Howard Flanders (3). According to this hypothesis, the radioprotection offered by -SH compounds is due to their ability to repair the radical damage of the target molecule by hydrogen donation from the -SH group. This reaction is in competition with the damage fixation produced by oxygen.

Adams and coll. (4,5) have directly observed by pulse radiolysis radical repair reactions such as:

$$. CH_2OH + RSH \longrightarrow CH_3OH + RS^{\cdot}$$

In solution containing excess RSH, the radical anion $RSSR^-$ formed from $RS^{\cdot}$ in the equilibrium reaction:

$$RS^{\cdot} + RS^- \rightleftharpoons RSSR^-$$

absorbs strongly at 410 nm and can be used to monitor the reaction.

A great number of rate constant for such repair reactions have been determined and they are generally lower than rate constants for the reactions of the same radicals with oxygen. If the previous model is correct, one should think that the -SH compounds are present at relatively high concentration in the cell in order to show radioprotection. An important molecule ubiquitously present in the cells at relatively high concentration is the tripeptide glutathione.

Radiation chemical data obtained in this Laboratory (6, 7) show that the -SH group is responsable of all chemical

events occurring in the molecule. Moreover the transfer of hydrogen atom from glutathione to carbon radicals has been demonstrated in model systems and appears to be the most probable mechanism of protection.

Repair of nucleic acid radicals by -SH compounds has not been observed directly by pulse radiolysis and it remains to be demonstrated directly that hydrogen transfer is a mechanism of protection in cellular systems.

CHEMICAL RADIOSENSITIZATION

The effect of oxygen in enhancing radiation damage in most biological systems, has been known for many years and fundamental and clinical work on both animal and human tumours has demonstrated the importance of anoxic regions in tumours as limiting factors in radiotherapy. ANy chemical agent which acts similarly to oxygen on the biological response to radiation is of potential value in radiotherapy. Nowdays some strongly electron-affinic compounds have been shown to be the most interesting radiosensitizers in view of the large number of investigations at fundamental level and for the promising pilot clinical studies still in progress (8).

They include quinones, dicarbonyl compounds, aromatic ketones, nitrofurans and nitroimidazoles. The radiosensitizing ability of these compounds is related to their electron-affinity and to their structure, in which the oxidizing property is due to the stabilizing effect of electron delocalization by resonance. Pulse radiolysis experiments have shown that these compounds are efficient oxidizing agents and can transfer electron rapidly and quantitatively from free radicals derived from different substrated including purines, pyrimidines, nucleic acids and amino acids. These experiments demonstrate that such sensitizers are more electron affinic than target molecules and give support to the electron-trapping model for the sensitization phenomenon proposed by Adams (9). The model suggests that, following direct ionization in the target molecule, thermalised electrons migrate to some electron trapping sites in the molecule. In the presence of electron affinic compounds, electron transfer reaction from the ionized molecule to the radiosensitizers could occur in competition with the internal charge recombination. The final result is an irreversible chemical damage to the critical molecule.

CONCLUSION

The purpose of this communication is to illustrate some significant examples of the application of radiation chemistry to chemical radioprotection and radiosensitization. The results demonstrate the important role played by molecular phenomena for the interpretazion of mechanism of chemical radioprotection and radiosensitization and for the development of more active substances. Much of the information concerning the involvement of fast processes in chemical radioprotection and radiosensitization, derives from studies of simple model chemical and cellular systems carried out with fast radiation chemical techniques. Even though the great complexity of"in vivo" systems excludes a unique explanation of chemical radioprotection and radiosensitization in molecular terms only, the contribution of fundamental radiation chemical studies is still of great importance.

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DYNAMICS OF Cs-137 DISTRIBUTION IN THE MUSCLE TISSUE OF SWINE BY SINGLE AND REPEATED CONTAMINATION

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Having in view that animal production is one of the essential links in the chain: environment-animal production-man, numerous investigations have been carried out on contamination pathways, resorption, distribution and elimination of radionuclides in dependence of contamination duration, biological and physico-chemical properties of radionuclides (1, 2).

Radionuclides enter the animal organism mostly through food. From the digestive tract by resorption through the blood they are deposited into organs and tissues.

Recently, we have undertaken systematic examinations on resorption, distribution and elimination of radioactive isotopes J-131, Sr-89,85 and Cs-137 applied to domestic animals as simulated fission mixtures. In this work is presented the level of deposited radioactive caesium-137 in muscle tissue of pigs and the rate of its elimination from the organism after repeated and single contamination. Muscle tissue was examined in particular owing to its great usability in human nutrition.

MATERIAL AND METHODS

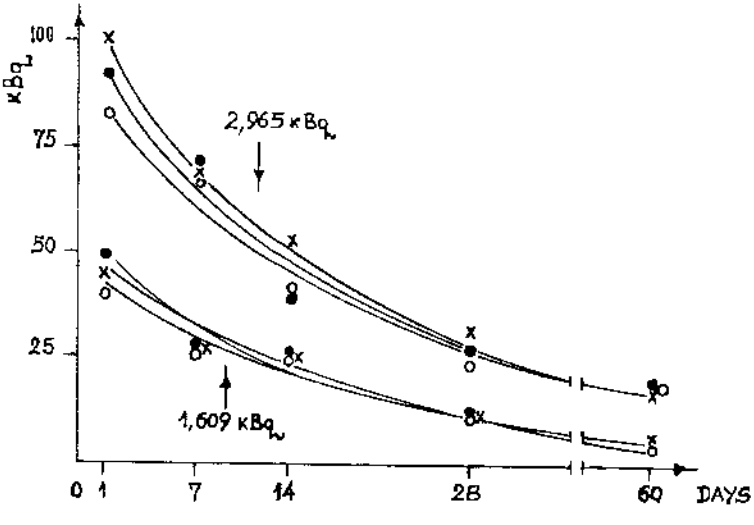
Six month old pigs of "domestic white" breed were contaminated per os by radioactive cesium-137 (CsCl carrier free). Two groups of animals were repeatedly contaminated (for 7 consecutive days) with a total dose of 2,967 kBq (daily dose - 424 kBq) and a dose of 1,609 kBq (daily dose -215.5 kBq per animal).The animals were sacrificed by total bleeding under anesthesia on day 1, 7, 14, 28 and 60 after the contamination. For radioactivity measurings the samples of musculature were taken. Another two groups of animals were singly contaminated with the same isotope and in the same way, and were sacrificed on day 3 after the contamination. Radioactivity was measured in the same samples of the musculature as in previous groups.

The radioactivity was measured in a gamma "INTERTECHNIQUE" counter in the samples of musculus longisimus dorsi, musculus supraspinatus and musculus gluteus superficialis. In order to determine the Cs-137 elimination rate from the organism of the contaminated animals the radioactivity in the samples of urine and faeces was measured every day.

The results were statistically processed and represent the mean value of the six individual samples.

RESULTS

By considering the obtained results it was observed that Cs-137 is almost evenly distribution in the examined muscle tissue of the contaminated animals. The differences among individual muscles during the examined period in both repeatedly and singly contaminated animals are unsignificant. Slight differences among the samples of some muscles are observed only on the first day after the contamination when the highest radioactivity was detected: by the dose of 2,967 kBq: musc. supraspinatus: musc. longisimus dorsi: musc. gluteus -92.87±2.22 : 83.25±18.87 : 101.75±26.27, and by dose of 1,609 kBq by the same order: 50.32±9.32 : 39.59±5.00 : 44.77±5.33. The level of radioactivity by the dose of 2.967 kBq in muscle tissue is nearly twice as high when compared with the dose of 1.609 kBq Graph 1. The radioactivity in all muscles exponentially decreases in the function of time in both applied



Graph. 1.- Concentration of Cs-137 in muscle tissue after the repeated contamination (in kBq/kg of wet sample) ●—● m. long.dorsi; o—o m. super; x—x m. gluteus.

Cs-137 doses. Intensity of decrease of radioactivity in muscle tissue is more expressed during the first 14 days in both applied doses than in the later experimental period, which is particularly expressed in a dose of 2,967 kBq.

The results of examination of the single contamination with two different doses are presented in Table 1. The level of radioactivity in muscle tissue of animals conta-

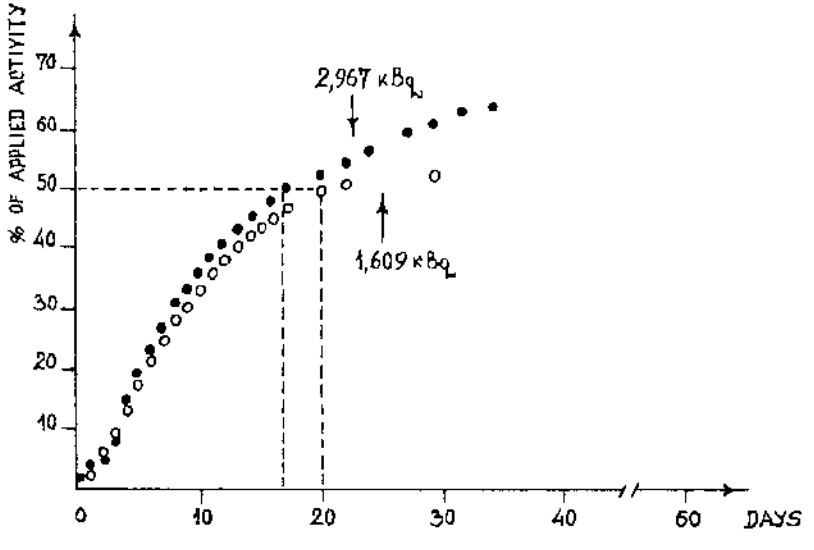
minated with a dose of 11.720 kBq is nearly twice as high when compared with the lower applied dose.

TABLE 1. Concentration of Cs-137 in muscle tissue on day 3 after single contamination (in kBq/kg of wet sample)

Musculus	Dose	
	5,772 kBq	11.729 kBq
m.supraspinatys	116.55±29.97	186.85±21.09
m.longisimus dosi	132.46±29.23	210.16±26.27
m.gluteus superficialis	121.73±5.18	219.78±31.82

Elimination of the radioactive Cs-137 from the organism of the contaminated animals is mostly performed by excretion through urine. Essential difference between the single and repeated contamination is found to be in elimination rate, i.e. in the effective time of half-life of excretion of the contaminant through the excrate (urine and faeces), Graph. 2. It is evident from the Graph that the effective half-life of excretion is between day 17 and 20.

By single contamination 50% of the applied Cs-137 is excreted by day 5 from the beginning of contamination (Tef. 1/2 is 5 days).



Graph.2.- Per cent of cumulatively excreted Cs-137 via urine and faeces by repeated contamination.

DISCUSSION

The analysis of the results presented shows that elimination of the deposited Cs-137 from muscle tissue of pigs, 6 months old, is not even. The elimination of this radionuclide from the muscle tissue is found to be more intensive between day 1 and 14 than later on (up to day 60). By the higher applied dose the constant of elimination rate up to day 14 was 1.50-1.71, and by the lower dose, for the same period, 0.83-1.50. Having in view the dynamics of the process of establishing equilibrium state during radionuclide deposition (1, 3) it could probably be explained the uneveness of elimination rate. The effective time of half-life of elimination of Cs-137 from muscle tissue, which was obtained for this age of pigs, was 14-15 days, while according to the data by Sirotkin (4), Buldakov and Moskalev it was 29.5, but the age of animals was not mentioned.

Burov (5) found that the deposit coefficient in two month old pigs was 7.9, while we found that in pigs of 6 months old by both applied doses was approximately the same deposition coefficient, i.e. 9.5 for the higher and 8.8 for the lower dose. In singly contaminated animals the difference in deposition coefficients was even lower, 11.4 for the higher and 11.2 for the lower dose.

Korneev et al. who established that Cs-137 and I-131 in pigs are excreted mostly via urine in contrast to the ruminants, in which these radionuclides are mostly excreted via faeces, confirm our findings. Approximately the same excretion intensity of Cs-137 from pig's organism (although doubly different doses were applied) during the first 17 days from the beginning of contamination can probably be explained by more rapid elimination of the contaminant immediately after its application. The results obtained by single contamination for excretion rate, as well as more rapid elimination of caesium from muscle tissue during the first 14 days after the contamination also contributed to this findings.

The results obtained as: deposition coefficient, constant of elimination rate from muscles and the effective time of half-life of excretion may be taken for evaluation of contaminant content in meat and its products if the concentration of perorally applied radionuclides is known, meanwhile it should be taken in account the age and species of animals, as well as the dynamics and duration of contamination.

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TUMORIGENIC RESPONSES FROM SINGLE OR REPEATED INHALATION EXPOSURES TO RELATIVELY INSOLUBLE AEROSOLS OF ^{144}Ce

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People may inhale radioactive aerosols in (a) a single exposure from an accidental release, (b) repeated exposures in an occupational setting, or (c) chronically from an environmental exposure. It is normally assumed that the biological behavior and resulting long-term biological effects observed in a single exposure situation can be extrapolated to repeated or chronic exposure situations. The specific objective of this study is to compare the biological effects in Beagle dogs exposed by different sequences of repeated inhalation exposures to a relatively insoluble form of ^{144}Ce at dose levels known to produce tumorigenic responses in dogs exposed once to the same aerosol. After a single inhalation exposure, the dose rate to lung decreases with an effective half-life of about 175 d. For comparison, repeated inhalation exposure sequences were chosen that would (a) increase the dose rate to lung with each exposure, or (b) reestablish a given dose rate.

MATERIALS AND METHODS

Thirty-six Beagle dogs (14 to 18 months old) were given 13 brief (< 60 min.), nose-only inhalation exposures at 8-week intervals. Twenty-seven of them were exposed to ^{144}Ce in fused aluminosilicate particles (AMAD $\sim$ 1.8 um, $\sigma_g \sim 1.6$) and nine controls were exposed to non-radioactive fused aluminosilicate particles. The three exposure sequences to ^{144}Ce (nine dogs/group) were: repeated increase in lung burden of 2.5 μCi/kg body weight, reestablished lung burden of 9.0 μCi/kg body weight, and reestablished lung burden of 4.5 μCi/kg body weight. Post-exposure measurements included whole-body counting, physical examinations, radiography, hematology, clinical chemistry and pulmonary function. At death, necropsies were performed for gross and histopathologic evaluation and measurement of the levels of ^{144}Ce in different tissues.

RESULTS

Typical whole-body retention measurements are shown in Figure 1 for one dog from each exposure group. Each spike represents the increase in body burden due to an inhalation exposure. The difference in body burden immediately before and after each spike represents

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This research was performed under U.S. Department of Energy Contract No. EY-76-C-04-1013 in facilities fully accredited by the American Association for the Accreditation of Laboratory Animals.

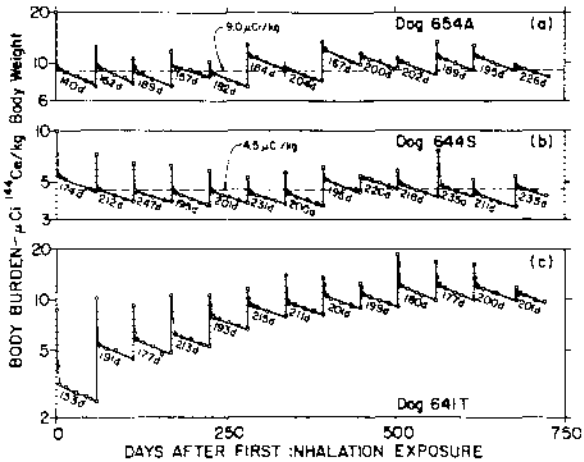

Figure 1. Whole-body counting data for 3 dogs repeatedly exposed at 56-day intervals to ^{144}Ce in fused aluminosilicate particles.

^{144}Ce deposited in the pulmonary region. Calculated pulmonary deposition varied considerably for a given dog. The overall mean lung deposition ± 1 s.d. was 26 ± 7% of the inhaled aerosol. For individual dogs, the values ranged from a low of 12 ± 4.1% to a high of 39 ± 10%.

Effective half-lives for retention in each 56-day interval between exposures are also shown in Figure 1. Group means and standard deviations are 193 ± 12, 205 ± 23, and 186 ± 19 days for the 9.0, 4.5, and 2.5 μCi/kg body weight groups, respectively.

The tissues from dog 655A, who died at 771 days after the initial inhalation exposure, were analyzed for their ^{144}Ce content. The body burden was divided among the lung (76%), liver (9.4%), skeleton (10%), tracheobronchial lymph nodes (1.4%), and other tissues (3.2%).

This study has been in progress for 5.8 years. During the first 2 years, the most pronounced biological effect was a decrease in circulating lymphocytes that occurred first in the 2.5 μCi/kg repeated and 9.0 μCi/kg reestablished groups and later in the 4.5 μCi/kg reestablished group. To date, 11 dogs have died as summarized in Table 1. The primary causes of death were radiation pneumonitis and pulmonary fibrosis (3), neoplasms (2), myelomalacia (1), autoimmune hemolytic anemia (2), bone marrow aplasia (1), parvovirus (1), and an anesthesia accident (1). Six neoplasms were noted, four in lung and one each in the spleen and tracheobronchial lymph nodes (TBLN). The four pulmonary tumors were all noted in dogs that had other primary causes of death.

All survivors in the 2.5 μCi/kg repeated and 9.0 μCi/kg reestablished groups are showing lymphopenia and radiographic signs of radiation penumonitis and pulmonary fibrosis. Alterations of gas exchange and lung mechanics are also becoming more apparent. Similar,

TABLE 1. Biological effects during first 5.8 years after first of 13 inhalation exposures to ^{144}Ce in fused aluminosilicate particles.

Lung Burden:	2.5 μCi/kg			9.0 μCi/kg			4.5 μCi/kg			C	
Dog Number:	644T	664C	645C	648S	648B	654A	665A	649U	655U	646B	648T
Pneumonitis/fibrosis	P				P	P					
Hemangiosarc., lung			X								
Sq. cell carc., lung		X									
Br. alv. carc., lung					X	X					
Hemangiosarc., spleen		P									
Hemangiosarc., TBLN									P		
Other (no tumors)			P	P			P	P		P	P

P = primary cause of death
X = other important observations at death

but less severe, changes of the same types are present in the 4.5 μCi/kg reestablished group.

DISCUSSION

This study provides a means of assessing variability in patterns of both dose and response. The pulmonary deposition data show that there was considerable variability among dogs (approximately 3X) as well as for any given dog. Such variability must be taken into account when assessing inhalation risks for a population (1).

The four pulmonary tumors seen to date in this study are plotted in Figure 2 as a function of the time of death and cumulative dose to lung. In a concurrent single exposure study, 15 pulmonary tumors were observed in 11 dogs during the first 5.8 yr (2). The rectangle drawn in Figure 2 illustrates the bounds of dose and time after initial exposure for these 15 tumors. All eight pulmonary hemangiosarcomas occurred in the dose-time region delineated by the horizontal and vertical arrows. The doses to lung received by dogs in the 2.5 μCi/kg repeated and 9.0 μCi/kg reestablished groups were within the dose range in which the 15 pulmonary tumors occurred in the singly exposed dogs. In spite of this, the first pulmonary tumor in the repeatedly exposed dogs occurred approximately 3 yr later than in the singly exposed dogs. Also in contrast to the results from the singly exposed dogs, the first pulmonary tumors in the repeatedly exposed dogs were not hemangiosarcomas. Another difference is that the only tumor seen to date in the 4.5 μCi/kg reestablished group was not in the lung but in the tracheobronchial lymph nodes.

It appears that the time of tumor occurrence and tumor type may relate to differences in patterns of dose rate to lung. In the single exposure study, the initial dose rates to lung in the 15 dogs with pulmonary tumors ranged from 150 to 320 rads/day. In the 2.5 μCi/kg repeated study, the average dose rate to lung was 19 rads/day

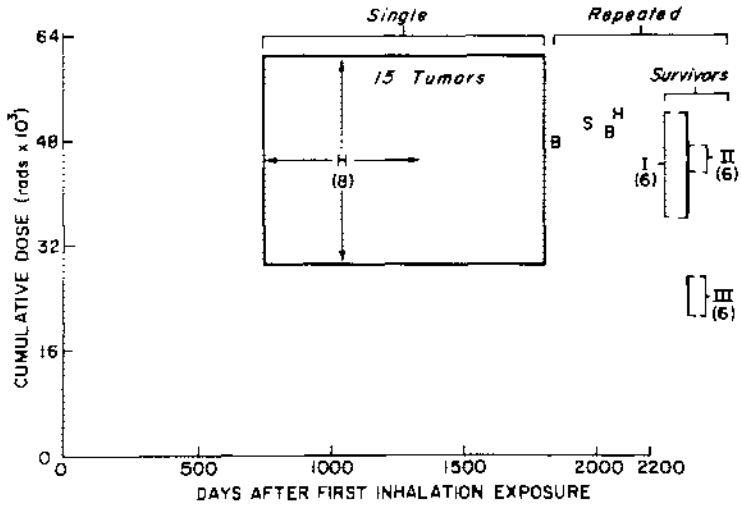

Figure 2. Schematic representation of the occurrence of pulmonary tumors. Tumor types are hemangiosarcoma (H), bronchioloalveolar carcinoma (B), and squamous cell carcinoma (S). Repeated exposure groups are (I) repeated 2.5 μCi/kg, (II) reestablished 9.0 μCi/kg, and (III) reestablished 4.5 μCi/kg.

after the first exposure and increased to 64 rads/day after the 13th exposure. In the 9.0 μCi/kg reestablished group, the dose rate was approximately 60 rads/day after each exposure. In all 3 studies, it has been assumed that the effective half-life of ^{144}Ce in the lung was 175 days after the exposures were completed.

Two major findings stand out in this continuing study. The first is the variability in deposition and retention seen among dogs. The second is the delay in occurrence of pulmonary tumors and the associated trend toward different types of tumors in the repeatedly exposed dogs as compared to singly exposed dogs. Such a comparison yields important information on how dose rate patterns can influence the resulting biological effects.

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RADIOBIOLOGICAL AND RADIOECOLOGICAL STUDIES WITH THE UNICELLULAR MARINE ALGAE *Acetabularia*, *Batophora* and *Dunaliella*.

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Unicellular marine algae are particularly useful for investigating the effects of ionizing radiations on living organisms as well as for studying the radioactive contamination of the aquatic ecosystem (1,2). *Acetabularia* (*A. crenulata*, *A. mediterranea*, *A. peniculus*) and *Batophora* (*B. oerstedii*) are giant unicellular uninucleate green algae, containing several million cytoplasmic organelles (chloroplasts and mitochondria). *Dunaliella* (*D. bioculata*) is a flagellated microalga, belonging to the Volvocales. These algae are being used in our laboratories for biological, radiobiological and radioecological studies. Due to the development of nuclear facilities, a detailed knowledge of the effects of radiations and of the biological behaviour of radioactive substances in the biosphere is urgently needed. This paper deals with the biological and the biochemical effects of X-rays (*Acetabularia*, *Batophora*) and with the incorporation of ^{3}H (*Acetabularia*, *Dunaliella*).

MATERIALS AND METHODS

Most of the methods used have been previously reported (1,2,3). *Acetabularia* and *Batophora* cells were irradiated with increasing doses of X-rays (from 0 to 150 Kr) during their vegetative growth (stage 4) (4). Labeling experiments were performed with *Acetabularia* at stage 4 or with *Dunaliella* being in its stationary phase (about $2x10^{6}$ cells/ml). Tritiated organic molecules, obtained from CIS Association, were added to the culture medium of the algae during various periods of time. Radioactivity was measured in a liquid scintillation spectrometer. The intracellular concentration of radioactivity was calculated on the basis of the cells' fresh weight. *Acetabularia* chloroplasts were observed with Nomarski interference optics. The starch content was determined with the Boehringer hexokinase test after hydrolysis of the storage material with α-amyloglucosidase.

RESULTS

Radiobiological studies.

a) Biological effects of X-rays on *Acetabularia mediterranea*. X-rays interfere with the morphogenesis of *A. mediterranea*. The formation of the reproductive cap and of cysts (gametangia) are strongly reduced only at relatively high doses (> 50 Kr). Several types of morphological anomalies are observed in irradiated cells : 1) loss of the whorls; 2) enlargement of the first order articles of the whorls; 3) development of a new stalk from a hair of the whorls or from a branch of the rhizoid; 4) enlargement of the apex; 5) alteration of the cap symmetry; 6) irregular growth of the caps'

rays; 7) formation of irregularly shaped cysts in the caps' rays; 8) increase of cyst size; 9) cyst degeneration; 10) loss of or impaired compartition of the cytoplasm in the caps' rays prior to cyst formation. Light microscopical observations have shown the presence of elongated chloroplasts having large starch granules. Analyses performed with amyloglucosidase and hexokinase have revealed that starch accumulation in the chloroplasts increases with the radiation dose.

b) Biological effects of X-rays on *Acetabularia peniculus*. The morphogenetic processes of *A. peniculus* are affected by increasing doses of X-rays (Fig. 1A, B). Again, relatively high doses (> 50 Kr) are necessary to inhibit cap and cyst formation. Several types of morphological anomalies were found in irradiated cells : 1) loss of the apical whorl; 2) enlargement of the apical region of the stalk; 3) formation of irregularly shaped caps' rays; 4) reduction of the number and of the size of cap rays (see Fig. 1B); 5) increase of cyst size; 6) cyst degeneration; 7) absence of cyst formation in some caps' rays. Moreover, some irradiated cells turn dark green, suggesting a condensation of the cytoplasm and/or chlorophyll accumulation in the chloroplasts.

c) Biological effects of X-rays on *Batophora oerstedii*. In irradiated *B. oerstedii* cells the formation of the spherical compartments (sporangia), where later on cysts (gametangia) develop, is only delayed by the radiations (Fig. 1C, D). This finding shows that the morphological differentiation of *Batophora* is extremely radioresistant. X-rays, however, induce in *Batophora* several types of morphological anomalies; 1) reduction of the number of sporangia (see Fig. 1D); 2) formation of abnormal sporangia; 3) development of one or more sporangia along the first or second order articles of the whorls instead of at their tip; 4) enlargement of first order articles; 5) enlargement of the apical region of the stalk; 6) cyst formation in the first or second order articles of the whorls; 7) sporangia degeneration.

d) Biochemical effects of X-rays on *Acetabularia mediterranea*. Labeling experiments with thymidine-6-^{3}H, uridine-5-^{3}H and leucine-^{3}H have revealed that X-rays (50 Kr) provoke a strong reduction of DNA, RNA and protein synthesis in the chloroplasts. RNA synthesis was stimulated, however, for doses up to 25 Kr.

Radioecological studies.

a) Experiments with tritiated water (HTO). When *Acetabularia* cells (*A. crenulata, A. mediterranea*) are grown in the presence of HTO (0-5 µCi/ml), a significant amount of ^{3}H is incorporated in the total nucleic acid and protein fraction (2). However, ^{3}H supplied in the form of tritiated water is not accumulated by the algae.

b) Experiments with tritiated organic molecules. Since recent work suggested that ^{3}H may be accumulated when this element is bound to organic molecules (see ref. 3), we have studied the uptake of 10 different tritiated organic molecules by *Acetabularia mediterranea* and by *Dunaliella bioculata* : 1) thymidine-methyl-^{3}H; 2) adenine-2-^{3}H; 3) uridine-5-^{3}H; 4) L-leucine-4-^{3}H; 5) glycine-2-^{3}H; 6) L-arginine-3.4-^{3}H; 7) L-aspartic acid-2.3-^{3}H; 8) L-phenylalanine-

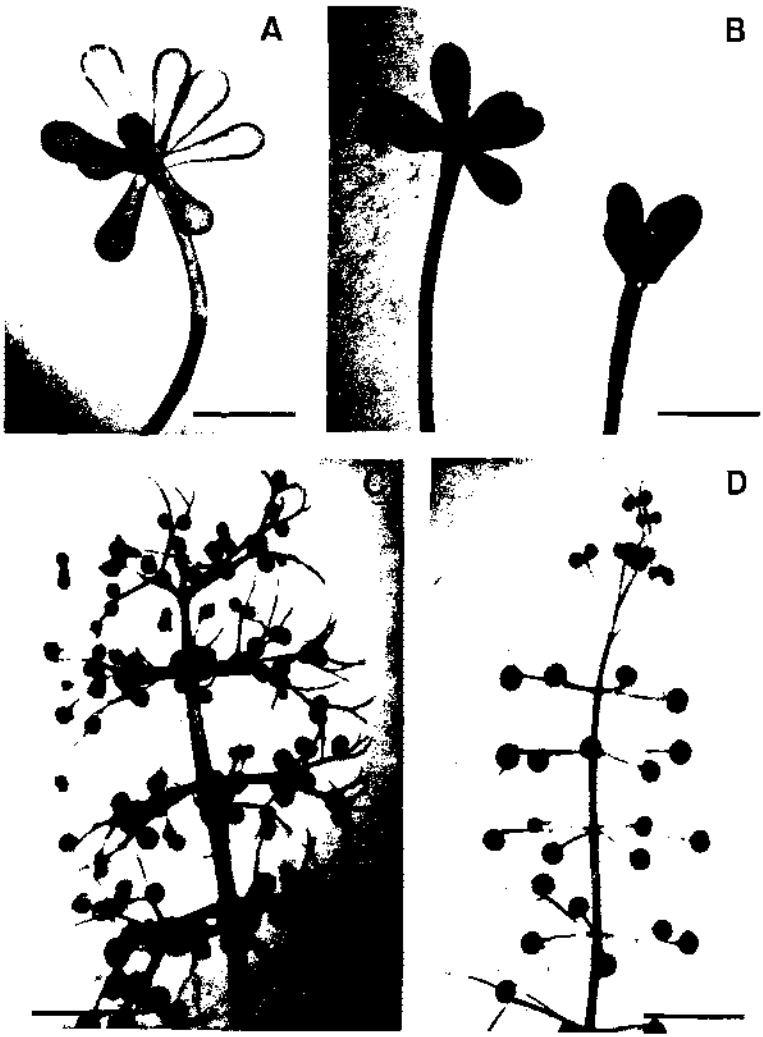

Figure 1. Morphological effects of X-rays on *Acetabularia* and *Batophora*. A : control cells of *A. peniculus* bearing a normal reproductive cap with 9 rays; B : 2 *A. peniculus* cells, 17 days after the irradiation with a dose of 150 Kr, showing a reduced number of caps' rays. C : Control cell of *B. oerstedii* showing the typical spherical sporangia, where later on cysts (gametangia) develop; D : *B. oerstedii* cell, 30 days after the irradiation with a dose of 50 Kr, having a reduced number of sporangia. Scale = 2 mm.

2.3-^{3}H; 9) D-glucose-1-^{3}H; 10) D-glucose-6-^{3}H. After a short incubation (30 min.), the intracellular concentration of the tritiated molecules can reach that of the external medium. However, *Acetabularia* accumulates adenine, arginine and glucose (respective concentration factors : 4.6; 5.1; 5.7), and *Dunaliella* is capable of concentrating adenine and leucine (respective concentration factors : 122.7; 11.4).

DISCUSSION

Our radiobiological studies show that the main morphogenetic processes of *A. mediterranea*, *A. peniculus* and *B. oerstedii* are affected by the radiations. Certainly, the sequence of events during the cells' developmental cycle is only realized under a well co-ordinated co-operation between the nucleus and the organelles. Most probably, X-rays interfere with this intergenomic co-operation, provoking different types of morphological anomalies. The radiations inhibit the syntheses of DNA, RNA and proteins in the chloroplasts of *Acetabularia* cells. Chloroplasts, which transform solar energy for the benefit of the cell, may play an important role for the realization of morphogenesis.

Experiments with tritiated water have revealed that *Acetabularia* cells are unable to concentrate ^{3}H. However, a significant amount of this radionuclide is incorporated into the genetic material of the cells (3). When organically bound ^{3}H is supplied to *Acetabularia* or to *Dunaliella*, a selective accumulation of some substances is observed. Our results contribute to a better understanding of the impact of radiations on living organisms and of the biological behaviour of ^{3}H in the aquatic system.

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DISTRIBUTION OF PLUTONIUM AND AMERICIUM IN HUMAN AND ANIMAL TISSUES AFTER CHRONIC EXPOSURES

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Distribution of plutonium in tissues of a rather homogenous group of fifty southern Finns has been elucidated. The subjects are healthy adults, ages 20 to 60, who died accidentally. Their intake of plutonium has been from fallout via inhalation; no occupational or significant dietary exposure seems possible.

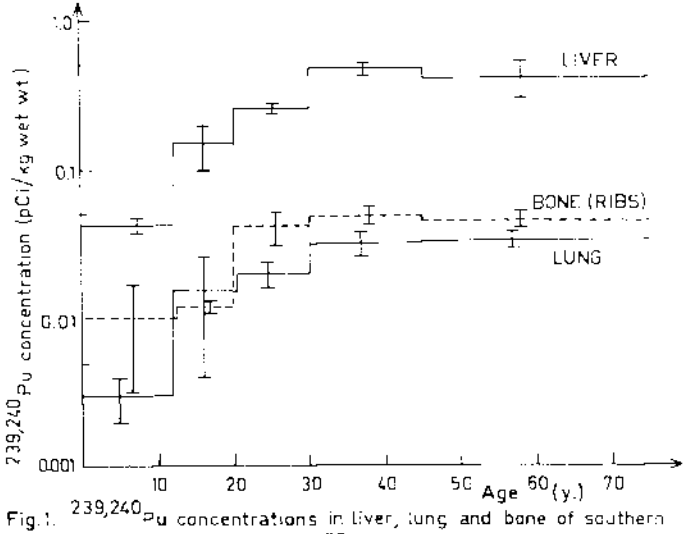

Fig.1. $^{239,240}Pu$ concentrations in liver, lung and bone of southern finns who died in 1976–1977.

Concentration in human liver was approximately 8 times higher than concentration in bones on the basis of wet weight (Fig. 1). Tissue concentrations in subjects over 20 years of age still reflect their high inhalatory intake during the 1960s. Whole liver contains 49, whole skeleton 40, muscles 8 per cent of the estimated total body burden, 1 pCi (1977-78) (Fig. 2).

Plutonium concentration in different bones: vertebrae, long bones, ribs, varied only little (Table 1). As seen in Table 1, Pu concentration in vertebrae is only 1.3 times that in other bones analyzed. Other authors have reported considerably bigger differences (1).

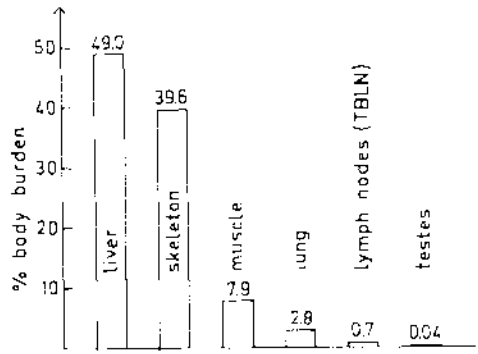

Fig.2. Distribution of plutonium in southern Finns in 1976-1979.

Table 1.
$^{239,240}Pu$ in bone samples of southern Finns

	pCi/kg wet	pCi/kg ash wt	
vertebrae	[illegible]	[illegible]	[illegible]
ribs	[illegible]	[illegible]	[illegible]
femur shaft	[illegible]	[illegible]	[illegible]
femur condyles	0.05 [illegible]	[illegible]	[illegible]

A) Number of samples

Plutonium intake via inhalation during life time was calculated on the basis of analyses of air filters (Fig. 3A). Ratio of the determined liver concentration to the estimated total life time inhalation intake (Fig. 3B) indicated direct correlation (1).

A group of northern Finns, the reindeer-herding Lapps, also obtained the bulk of Pu via inhalation but in addition some plutonium in their diet, rich in reindeer liver. Samples of only a few individuals have been obtained so far. The results on tissue-Pu do not differ significantly from those of southern Finns. Since their dietary intake of plutonium is fairly well known (males 40 pCi/a in 1967, 10 pCi/a in 1977), it can be calculated that on the basis of the absorbability stated by the ICRP ($3 \cdot 10^{-5}$ to 10^{-5}) their dietary plutonium retention is of the order of one per cent of the measured body burden. If the real absorbability would be higher, higher tissue concentrations could be expected, too. Although the Pu results of reindeer herders are not yet statistically conclusive, they provide a positive indication that the ICRP value is correct. Further analyses of this population will be valuable for corroboration of this ICRP value, so far mainly based on animal results.

Reindeer is an exceptional animal since it obtains large amounts of transuranium elements in its natural winter diet, which mainly consists of lichen (Pu-239,240: ca. 100.000 pCi/a in 1967, 15.000 pCi/a in 1976; Am-241 ca. 20.000 pCi/a in 1967, 3.500 pCi/a in 1976). Thus, reindeer provides an opportunity for direct determination with good accuracy of the gastrointestinal absorption of these transuranium nuclides from a natural diet.

Pu-239,240 and Am-241 were also analysed in elk (an animal resembling the American moose) because it is closely related to

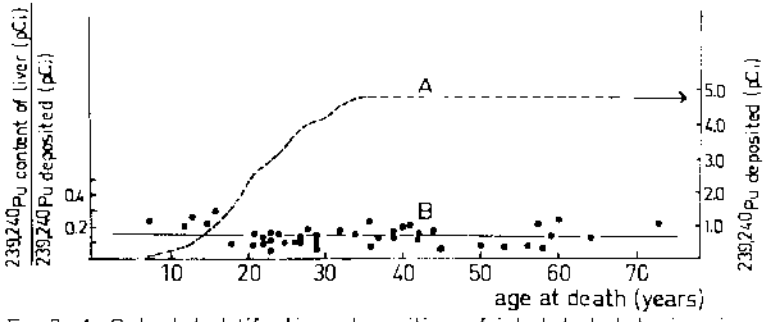

Fig. 3. A: Calculated life-time deposition of inhalated plutonium in total body of subjects who died in 1976–1977. B: The ratio of $^{239,240}Pu$ content in liver to the total lung deposition of plutonium during the life-time.

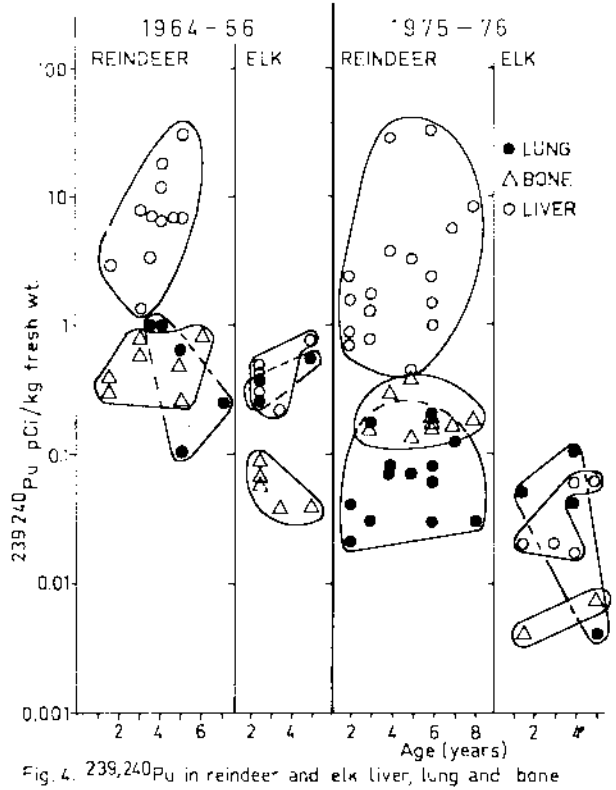

Fig. 4. $^{239,240}Pu$ in reindeer and elk liver, lung and bone during the periods of high (1964–1966) and low fallout (1975–1976).

reindeer but does not feed on lichen. Thus, comparison of reindeer and elk indicates the role of lichen to the tissue concentrations of these transuranium elements in reindeer. In the middle of 1960s when atmospheric Pu-concentrations were high, lung concentrations of Pu in reindeer and elk were similar, but liver concentrations in reindeer one order of magnitude higher than in elk (Fig. 4). Presently, when the atmospheric concentrations are low, the lung concentrations are still approximately equal, but the liver and bone concentrations of reindeer two orders of magnitude higher. Thus, the liver and bone concentrations reflect the dietary intake of plutonium. The liver of reindeer contains 50, skeleton 30 and muscles 10 per cent of the total body burden of plutonium (Fig. 5).

The Am-241 determinations of human tissues are still under study but in reindeer the Am-241 concentrations (on wet weight basis) are ca. 20 to 40 per cent of the Pu-239,240 concentrations in liver and lung, but 80 to 140 per cent in the trabecular bone of old animals (Table 2). Much of the Am-241 in reindeer tissues is due to ingrowth from Pu-241 in the animal. Accumulation of Am-241 in bone is much higher than of Pu-239,240. The purpose of this study is to establish wheather this situation is true also for the human bone.

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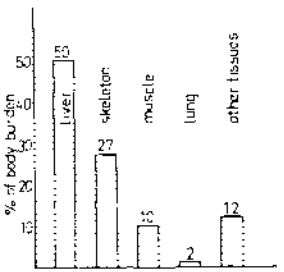

Fig 5 Distribution of $^{239,240}Pu$ in reindeer in Finnish Lapland

TABLE 2. ^{241}Am AND $^{239,240}Pu$ IN REINDEER IN FINNISH LAPLAND DURING 1974-1976. STANDARD DEVIATION OF THE RADIOASSAY (1σ) IS INDICATED.

		^{241}Am mCi/kg WET	$^{239,240}Pu$ mCi/kg WET	$\frac{^{241}Am}{^{239,240}Pu}$ at. %
LIVER				
REINDEER	I (6, 13a)	1.17±0.14	7.10±0.20	0.16±0.02
REINDEER	II (6, 4.5a)	0.19±0.05	0.5 ±0.2	0.39±0.18
REINDEER	III (6, 11a)	0.29±0.16	0.44±0.27	0.20±0.05
TRABECULAR BONE				
REINDEER	I	0.25±0.15	0.63±0.02	0.76±0.46
REINDEER	II	0.16±0.14	0.11±0.04	-
REINDEER	III	0.37	0.27±0.06	1.37±0.46
LUNG				
REINDEER	I	0.08±0.02	0.13±0.05	0.47±0.15
REINDEER	II	0.09±0.05	0.35±0.04	0.26±0.13
REINDEER	III	-	0.20±0.14	-

ETUDE EXPERIMENTALE DES CANCERS INDUITS CHEZ LE RAT PAR DES PARTICULES A TRANSFERT LINEIQUE D'ENERGIE ELEVEE.

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Institut de Protection et de Sûreté Nucléaire - Département de Protection - Service de Radiopathologie et de Toxicologie Expérimentales B.P. n° 6 - 92 260 FONTENAY AUX ROSES - FRANCE -

Malgré l'hétérogénéité de répartition des radioéléments entre les différents organes et à l'intérieur de chaque organe après une inhalation de transuraniens émetteurs alpha chez le rat, on obtenait pour une même dose absorbée calculée (exprimée en nombre de particules émises par gramme d'organe) une fréquence identique de cancers dans le squelette et le poumon (réf.1). Pour une dose moyenne identique délivrée à l'organe, la dose délivrée à la cellule était différente pour l'os et le poumon. Dans le poumon, la dose était répartie d'une façon relativement homogène, alors qu'elle était très hétérogène dans l'os. Il en résultait que la cellule osseuse se trouvait recevoir en réalité une dose beaucoup plus élevée que la cellule pulmonaire. Les radiosensibilités de ces deux organes étant peu différentes, la cellule osseuse pouvait donc être moins sensible que la cellule du poumon à l'irradiation alpha.

Nous avons voulu vérifier ce phénomène pour un autre type d'irradiation, et nous avons soumis des rats mâles de race Sprague Dawley à une irradiation externe chronique par des neutrons, cette irradiation délivrant une dose beaucoup plus homogène à tous les organes de l'animal et le transfert linéique d'énergie était comparable dans les deux cas.

Les rats sont placés dans une cavité du réacteur Néréide situé à Fontenay aux Roses, conçue pour les expériences biologiques (réf.2). Les neutrons émis sont des neutrons de fission d'une énergie moyenne de 1 Mev. Nous avons, en plus, une composante gamma égale à 20% de la dose neutronique. En ce qui concernait les émetteurs alpha, les doses délivrées calculées à l'organe variaient de 3 à 3.000 rad et ne tenaient pas compte de l'hétérogénéité de distribution dans les organes; pour les neutrons, nous avons essayé d'obtenir des doses pouvant se comparer aux précédentes; nous avons choisi une gamme de doses mesurées à la peau variant de 1.5 à 800 rad et délivrées sur une durée variant de 1 jour à 6 semaines.

RESULTATS.

Tableaux 1 et 2.

Comme avec les transuraniens, la durée de vie des rats après irradiation neutronique dépend de la dose totale qui leur a été délivrée.

Pour le moment, nous n'avons de résultats définitifs avec les neutrons que pour 220 animaux irradiés à plus de 100 rad. Pour ces 220 rats, nous obtenons 221 cancers, soit en moyenne 1 cancer par animal.

TABLEAU 1. - Inhalation de transuraniens par des rats

Dose délivrée à l'organe (rad)	CANCERS %		
	Poumons	Squelette	Autres organes
0	1	0	0
3200	85	-	-
1000	54	-	-
750	-	40	-
350	26	22	-
150	10	10	-
110	-	5	-
35	-	2	-
25	-	-	31
3	-	-	10

TABLEAU 2. Irradiation de rats par des neutrons

Dose (rad)	CANCERS %				Durée de vie (jours)
	Poumon	Os	Peau	Autres organes	
0	0	1	1	8	650
800	10	15	10	20	239
460	15	2.5	23	60	365
240	22	8	37	43	455
150	20	6	25	51	485
50 *	4	8	21	38	455
30 *	6	3	6	42	460

* Expériences non terminées.

Les résultats donnés pour les doses de 50 et 30 rad sont des résultats qui ne sont pas définitifs puisque tous les animaux ne sont pas encore morts.

Les organes les plus sensibles à l'action des neutrons sont d'une part la peau et les tissus sous-cutanés et d'autre part les poumons.

Le rapport du nombre de cancers osseux au nombre de cancers pulmonaires obtenus après irradiation neutronique est égal à 0.4; pour des neutrons de 1 Mev le rapport des doses absorbées par l'os et le poumon est égal à 0.65 (réf.3). La radiosensibilité de ces organes est donc comparable à la précision de mesure et de calcul près. Les résultats obtenus avec les émetteurs alpha pouvaient faire croire à une sensibilité cellulaire différente, mais cette interprétation n'est pas compatible avec les résultats obtenus avec les neutrons; les différences de radiosensibilité obtenues ne sont pas assez grandes pour confirmer cette hypothèse.

Un phénomène particulier apparaît dans le cas de ces irradiations neutroniques, c'est la proportion beaucoup plus élevée d'une part de cancers de la peau et des tissus sous-cutanés (28%), d'autre part d'angiosarcomes de tous les tissus (8%). Si on compare ces résultats à ceux obtenus avec les émetteurs alpha où l'on avait 9% de cancers cutanés et 1,9% d'angiosarcomes, on constate que l'irradiation totale et homogène de ces deux systèmes provoque l'apparition d'un fort pourcentage de cancers dans ces tissus.

CONCLUSION.

Lorsque l'irradiation de l'organisme est totale, on voit apparaître une fréquence anormalement élevée de cancers du système vasculaire et du système de revêtement.

Les résulatts des irradiations neutroniques montrent que le fait de calculer la dose à l'organe pour les émetteurs alpha afin d'exprimer la relation dose-effet est une pratique correcte.

L'hypothèse que nous avions émise d'une radiosensibilité différente de la cellule osseuse et de la cellule pulmonaire n'est pas confirmée par ces nouvelles expériences.

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PRESENT STATE OF RADIO-STRONTIUM DECORPORATION RESEARCH WITH CRYPTAND (222) .

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Strontium-90 appears in high percentages in reactor burn-up as well as in nuclear fall-out . In animal experiments it has been demonstrated numerously that this bone-seeking element provokes skeletal malignancy . Therefore it is not only potentially hazardous to workers of nuclear power plants and related industries, but as an environmental contaminant to every body .

Prevention of intestinal uptake , removal during circulation , and removal of bone deposits are the three possibilities of therpeutic onset for decorporating remedies after Sr-90 incorporation .

Cryptating agent (222) is presented here as a potent means to remove radioactive strontium during the circulation of the latter . With (222) the strongest decorporation effects ever reached on the radionuclides Sr-85, Ba-133, and Ra-224 were obtained by us in rats (1,2, 3,4) . Our results were confirmed by Krajl et al.(5) and Batsch et al.(6) . From recent experiments we extrapolate tentatively from rat to man , presenting here a probable treatment scheme, demonstrating the decorporation effect as function of the (222)-dose and the time interval between incorporation and treatment start .

Some toxicological aspects are also discussed.

MATERIALS AND METHODS

The basic parameter in decorporation experiments is the effectiveness quotient (EQ), which permits to judge on the pure decorporation effect of an agent . In our experiments this quotient is defined as

$$EQ = 100 \times \frac{\text{TBR of radionuclide in (222)-treated rats}}{\text{TBR of radionuclide in untreated rats}}$$

(TBR = Total Body Retention) . One obtains this quotient with high significance, using two groups of 5-6 rats in general of about 300 g body weight, which were injected i.v. or i.p. with about 37 KBq Sr-85,(Ba-133 and Ra-224). One of these groups received additionally either together with, or after a time delay of the incorporation of the radionuclide, a constant or varying doses of cryptand (222) either i.v. or i.p. . The TBR was measured with a one-channal gamma-spectrometer immediately after the incorporation and consecutively after 24 hours .

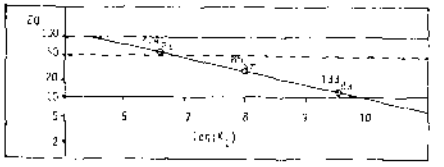
Two of the three questions , which had to be answered experimentally, are demanded by what we call Schubert - Catsch-Heller-Relationship (S-C-H) , valid within the re-

strictions for E and EQ given by Catsch (7) :

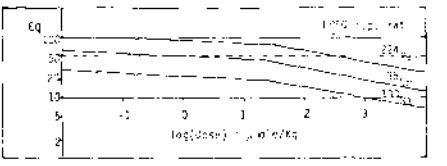
$$\frac{\log EQ_1}{\log EQ_2} = \frac{\log E_1}{\log E_2}; \quad \log E = \log K_{Sr(222)} - \log K_{K(222)} + \log A$$

A comprises the dose of (222). The S-C-H rules decorporation involving the EQ as dependent of two main physico - chemical parameters, 1) the stability constant K_1 of cryptand (222) towards the radionuclide (8), and 2) the concentration e.g. the dose of (222) (9), e.g. $EQ=f(K_1)$ and $EQ=f(Dose)$. Third, the dependency of EQ from the starting point of treatment after incorporation of the nuclide e.g. EQ as a time-fuction $EQ=f(Time)$.

RESULTS

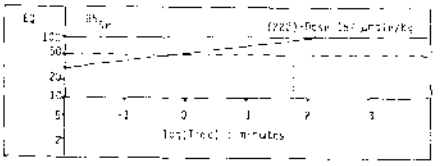


Graph 1 $EQ=f(K_1)$

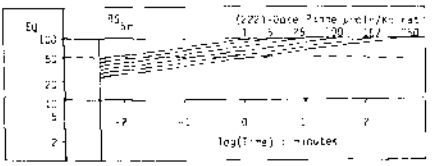


Graph 2 $EQ=f(222\text{-}Dose)$

The decorporation effect rises, e.g. the EQ drops with a rising stability constant K_1 strictly obeying the (S-C-H) (Graph 1) . It rises further with a rising dose, obeying also the S-C-H down to a dose of 25 µmole/Kg; further below it obeys again the S-C-H equations if one introduces instead of A, root of A (Graph 2)(9). The striking fact that EQ as time-function (Graph3)represents also a straight line in a log-log system is explained by the fact that decorporation with (222) follows the availability of strontium in the blood or extracellular space of rats(4). Graph 4, derived from the results of Graph 2 & 3, represents the EQ as function of the (222)-dose and a delayed treatment start (EQ=f(Dose & Time)), which permited to extrapolate from rat to man(Table 1), using further : the Sr-retention in blood of rats (10) and men (11); and the following assumptions : 1) the extracellular space is as well reaction space as distribution space of (222) and Sr-85 ; 2) the Sr-retention in the extracellular space paralells the Sr-retention in the blood; 3) the blood volume of rats and men corresponds to 6% of the body weight; the extracellular space of rat and man corresponds to 16.6% of the body weight;4) the EQ-values obtained from experiments in rats do not essentially differ from those in men .



Graph 3 EQ=f(Time)



Graph 4 EQ=f(Dose & Time)

DISCUSSION

Though, a significant decorporation effect (EQ) in man (Table 1) may obviously be obtained , even after a few days of strontium incorporation , it seems hard to obtain an EQ lower than 50% with a single and relatively high (222)-dose even after a few minutes of treatment start . Further, in man the decorporation of strontium seems to slow much more down with the reduction of the (222)-dose than with an increasing delay of treatment . Consequently high doses of (222) are necessary in order to obtain a sufficient decorporation . High doses of (222) may approach acute toxicity . (LD-50 = 292 µmole/Kg rat i.p.(12)). If several protracted small (222)-doses given consecutively, may avoid toxicity of a single high dose, eventually producing the same or even a better decorporation effect must still be demonstrated. Table 1 permits to conclude finally : With a (222)-treatment one may obtain a decorporation effect of 50% after a very early start of therapy , accopainied by an additional significant percentage of naturally excreted strontium , but it seems inavoidable, that a certain rest of the nuclide will be trapped by bone . Decorporation of radiobarium will be more beneficial because of a higher K_1 (Graph 1 & 2) .

The therapeutic range of (222) deduced from Graph 2, varies from nuclide to nuclide along with its stability constant (8) . For Sr-90-89-85 it equals 100 , for Ba-140-133 even 146000 but for Ra-226-224 only 1.0 in the rat. If these ranges are large enough for man, even in the case of a steep mortality-dose function, cannot be answered at the moment .

If morphological and biochemical,e.g. enzymatical (5) disorders are limited to the lethal dose range of (222) or are reversible respectively not existent in the sub-lethal , therapeutic dose range, is not yet known .

Reversible side effects, such as impairment of protein- and DNA-synthesis (13) and urinary sodium/potassium retention (14) observed after single sub-lethal (222)-doses, may be accepted as those, because (222)- - treatment will always be an acute-treatment but never a chronical one .

Future studies will teach us, if (222) becomes an acceptable agent for the decorporation of radio-strontium barium and perhaps radium during their circulation in the blood and extracellular space of man .

EXTRA-CELL. %								
5	565		-	-	-	-	-	-
10	110	-	-	-	-	-		-
13.8	62	5500	88	-	-	-	-	-
15	48	3500	88	96	-	-		
18.5	30	1050	82	97	-	-		
20	27	1000	82	90	100	-		
25	11	350	77	85	97	-		
30	11	130	73	82	93	-		
35	8.6	80	72	78	90	100		
40	6.6	50	68	76	87	98		
45	5.8	31	68	73	84	94		
45.5	5	25	66	72	83	93		
50	4.3	20	65	72	83	93		
55	4.3	14	63	70	80	90		
60	3.8	9	61	67	78	88		
65	3.4	6	61	67	77	87		
70	3.1	4.5	59	65	76	86		

EQ-VALUES

TABLE 1

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PARA-HYDROXYBENZOIC ACID, A HYPOXIC RADIOSENSITIZER IN BACTERIAL CELLS

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The relative radioresistance of hypoxic cells present in some tumours is a serious limitation in attempts to increase the therapeutic ratio between tumour control and damage to normal tissue during radiotherapy. Hyperbaric oxygen, fast neutrons and π^- mesons are suggested methods of overcoming this problem. The use of chemical radiosensitizers which effectively act upon hypoxic cells is another approach (1). A major group of such hypoxic sensitizers resemble oxygen in their mode of action, and it has now been fairly well established that their efficiency as radiosensitizers, on a molar basis, is directly related to their electron-affinity (2-4). Sensitization in bacterial systems by p-nitroacetophenone (PNAP) (4-6) and its more soluble derivative NDPP (7,8) led to the testing of other nitro-aromatic compounds such as the nitroimidazoles.Two promising drugs to have emerged recently, metronidazole and misonidazole, have been shown to be effective both in vitro and in vivo, and, more recently, the results of tests with these drugs in patients have been encouraging (9,10). Both these compounds are of particular interest because of their clinical use as trichomonicides, with considerable pharmacological, toxicological and pharmacokinetic information available. It seems quite possible that they may prove to be of value in clinical radiotherapy.

The aim of the present investigation is to look for other compounds, whose medicinal use is established and which, by virtue of their electron affinity, might be anticipated to act as radiosensitizers. One such group of compounds are the esters of p-hydroxybenzoic acid which are used in many pharmaceutical formulations as antimicrobial preservatives. In the first instance, the effect of different concentrations of p-hydroxybenzoic acid on the radiation sensitivity of oxic and anoxic buffered suspensions of the bacterium Staphylococcus aureus has been examined.

MATERIALS AND METHODS

p-Hydroxybenzoic acid (PHBA) was supplied by Sigma Chemical Co. The test organism was Staphylococcus aureus Oxford (NCTC 8236) maintained on slopes of nutrient agar at 4°C. Suspensions were grown to log phase in nutrient broth, washed and resuspended in fresh 1/15M phosphate buffer saline (pH 7.0) at a concentration of approximately 10^8 cells per ml prior to irradiation. PHBA was incorporated into the buffer saline at suitable concentrations. Routinely the bacteria were in contact with the additive for at least 45 min prior to irradiation.

γ-irradiation was carried out at ambient temperature using either a M38-3 Gammator 2400Ci ^{137}Cs source or a Gammacell 220 24000 Ci ^{60}Co source. Irradiation vessels were glass vials of od 19.7mm sealed with gas-tight rubber closures. Using the ^{137}Cs source, arran-

gement of these vessels within the irradiation chamber permitted the simultaneous irradiation of 18 suspensions at an average dose rate of 1.75 krads per min. With the ^{60}Co source, 36 suspensions could be simultaneously irradiated at predetermined dose rates ranging from 5.1 to 7.4 krads per min. Deoxygenation was by bubbling oxygen-free N_2 (less than 3 ppm O_2) for 20 min through the suspensions immediately prior to irradiation. For maintenance of suspensions under aerated conditions O_2 was bubbled through the suspensions for 5 min.

Suspensions were irradiated for fixed time intervals such that at the maximum dose level tested at least two decades of inactivation were generally achieved. Following irradiation bacterial suspensions were appropriately diluted and four 0.05ml aliquots of each diluted suspension pipetted onto plates of nutrient agar. After overnight incubation at 37°C, the micro-colonies formed were scored. Each colony was taken as indicative of a single bacterium in the original suspension. Counts performed on unirradiated samples of test suspensions treated identically to those exposed to irradiation were used as control estimates of the number of viable cells in calculations of surviving fractions. Dose-ln survival curves were constructed from five experimental points. Variation in values of slopes of these curves (inactivation constants) with changes in test conditions have been used to demonstrate quantitatively changes in radiation sensitivity.

RESULTS

Typical dose survival curves for Staph. aureus suspensions irradiated in the presence of different concentrations of PHBA in anoxia are depicted in Figure 1. Included in this figure are the responses for suspensions irradiated in air and anoxia in the absence of PHBA. These curves are linear over the dose range tested. All our computed values of inactivation constants together with standard deviations have been plotted in Figure 2 as a function of PHBA concentration. The horizontal lines are the levels of sensitivity observed when cells are irradiated in the presence and absence of O_2 in buffer alone.The corresponding parallel dashed lines represent standard deviation about these values. Increasing concentrations of PHBA from 10^{-6} to 5 x 10^{-5}M in anoxic bacterial suspensions produced no change in the response characteristic of bacteria irradiated in anoxic buffer alone. However further increases in PHBA concentration to 6 x 10^{-3}M caused a marked increase in radiation sensitivity, with the maximal response very close to that for suspensions irradiated in the presence of oxygen. The testing of higher PHBA concentrations was confounded by problems of toxicity.

PHBA was also tested for radiosensitizing activity in the presence of O_2. These tests were carried out at concentrations of PHBA that cause significant sensitizing effect in anoxic suspensions. The responses obtained are very close to that characteristic of bacteria suspended in oxygenated buffer alone, clearly demonstrating that the enhancing effect of PHBA and oxygen are not additive and that the sensitizing action of PHBA operates within the oxygen effect.

Preliminary studies showed that PHBA was somewhat toxic to the bacterium at concentrations of and above 6 x 10^{-3}M. Experiments were done to ensure that PHBA at lower concentrations showed no toxicity

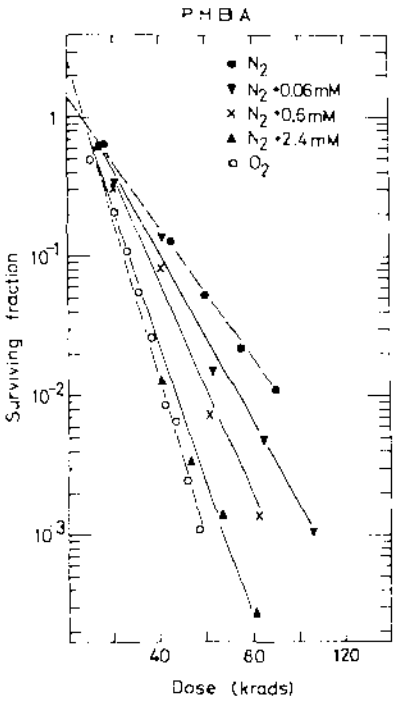

Figure 1. Survival data for hypoxic suspensions of Staph. aureus irradiated in the presence of different concentrations of PHBA.

towards the biological system. These control experiments demonstrated that the unirradiated PHBA had no effect on the viability of the unirradiated cells, that radiolysis products of PHBA at test concentrations had no effect on the viability of irradiated cells, and that these products carried over to the plating medium were without effect on the irradiated bacterium undergoing colony formation.

DISCUSSION

Our major finding is that PHBA acts as an efficient hypoxic radiosensitizer when tested against Staphylococcus aureus. The fact that the degree of sensitization is close to that observed in the presence of oxygen (DMF 2.5), suggests that PHBA may be simulating several of the previously proposed sensitizing actions of oxygen (11, 12). Only one such action may be as a result of its electron-affinity. Furthermore, the lack of sensitizing action in the presence of oxygen is evidence that this agent operates within the 'O_2 effect'.

Although the actual mechanism by which PHBA exerts its radiosensitizing action is unknown, several models have been proposed for the mechanism of sensitization by electron-affinic agents in general (for example, 3,13,14). The testing of PHBA in the presence of specific radical scavengers may help elucidate its mode of operation as a radiosensitizer. Whether the esters of PHBA, which are known to be less toxic than the parent compound, will likewise possess radiosensitizing properties remains to be investigated.

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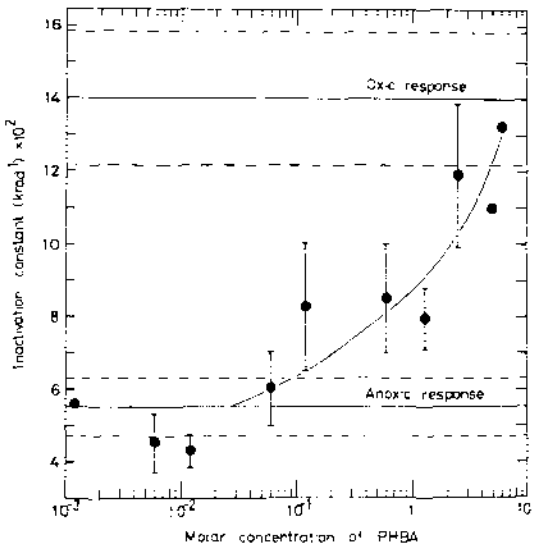

Figure 2. The effect of PHBA concentrations on the anoxic radiation response of Staph. aureus suspensions.

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ERRATUM:

The legend on the abscissa of Figure 2 should read:

"Molar concentration of PHBA (x 10^3)".

SUSTAINED-RELEASE OF RADIOPROTECTIVE AGENTS IN VITRO

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The major functional group of radioprotective agents contains a sulfur atom, either as a thiol (-SH) or in the oxidized state (-S-S-). These compounds protect against radiation as a result of their ability to trap primary free radicals formed via degradation radiolysis of water (1). The practical use of those radioprotectants is limited by their cumulative toxicity and rapid excretion and degradation. Moreover, for provision of adequate protection against irradiation effects, the concentration of the drug requires precise regulation, a difficult goal to achieve due to the rapidity of depletion on the one hand, and the low protective index on the other. Some protective doses and LD_{50} values for a highly effective radioprotective agent, cysteamine, in various mammalian species, are given in the following table (2):

Species	Route	Protective Dose (mg/kg)	LD_{50} (mg/kg)
Mouse	ip	75-150	260
Rat	ip	75-150	140
Dog	iv	75-110	
Rabbit	iv		150
Man	iv	150-400	

The radioprotective drugs chosen for the first phase of this study were cysteamine (β-mercaptoethylamine, H_2N-CH_2-CH_2-SH) and cysteine (3-mercaptoalanine, H_2N-CH(COOH)-CH_2-SH), the most effective agents explored in mammals. The objective of this research is to improve the efficacy of these radioprotectants by development of new pharmaceutical formulations rather than by synthesising of new agents. With the growing use of nuclear enrgy throughout the world, and the risk of accidental irradiation becoming a dangerous hazard, we considered it desirable to develop a sustained-release radioprotec= tive formulation. Prolonged release of the agent in the body at effective concentrations is aimed at avoiding increase in toxicity, and will provide a model for a novel approach to human protection against accidental irradiation. The ultimate goal of this study is then better control of the bioavailabilityof the drug in the serum, with increase in its protective index.

MATERIALS AND METHODS

Cysteamine and cysteine (Sigma) were dried, pulverised and mixed

with stearic acid BP 1973 (sigma) and ethylcellulose (Hercules), in various proportions, in a mortar. Various concentrations of the active material, not exceeding 50%, were prepared in defined ratios of the matrix mixture. The mixture was than compressed into cylindrical tablets of 13.1 mm in diameter and a mean of 4.0cm^2 surface area (13.6 mm height) in a vacuum KBr die, with a laboratory press, at various pressure values (3), as given previosly (4). Release of the drug was measured spectrophotometrically (Unicam, model SP-1805) at 240 nm, using a rotating basket dissolution apparatus as described in USP XIX. The dissolution medium (phosphate beffer pH 7.4) was kept under nitrogen. Exactly one liter of the buffer, previously heated and maintained at 37·0.3°C, was used for each experiment. The basket was immersed in the buffer and rotated at 100 rpm. Experiments were carried out for 8-16 hours, and the concentration of the unoxidized drug was monitored continuously, using a 10 mm flow-cell fed by a peristaltic pump (Buchler, model 2-6100), at flow rate of 60 ml.min^{-1}. The tablets were wrapped in aluminium foil immediately after their compression, and kept at room temperature pending assay. Experimental results obtained spectrophotometrically were computerized, using a programme to fit Higuchi equation treatment described as follows: 10-20 points from each spectrophotometric run were fed into a PDP-15 computer and plotted against the square root of t. The parabolic UV absorption curve, which demonstrates the accumulated released drug when plotted on a linear scale of t, is converted into a linear correlation when plotted against the square root of t, following the Higuchi law. For these calculation the molar extinction coefficients for cysteine (427 M^{-1}.cm^{-1}) and cysteamine (563 M^{-1}.cm^{-1}) were determined experimentally under nitrogen.

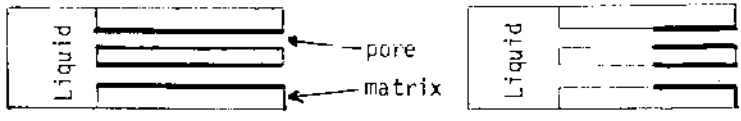
The mechanisms for the sustained-release of drugs dispersed uniformly in hydrophobic matrices were proposed by Higuchi in 1963 (5). He developed equation suitable both for drugs partially soluble in the matrix, and for drugs insoluble in the matrix but soluble in the external medium. In the latter type of system the drug particles are released by leaching action of the penetrating solvent. The Higuchi square-root law, which has been validated in numerous cases (6-8), takes the following form:

$$Q = \sqrt{\frac{D.\varepsilon}{\tau} (2A - \varepsilon . C_s) C_s . t}$$

where Q = the amount of drug released at time t per unit exposed area
ε = the porosity of the matrix
τ = the tortuosity of the matrix
D = the dffusibility of the drug in the dissolution medium
A = the total amount of drug present in the matrix / unit volume
C_s = the solubility of the drug in the dissolution medium

With all other parameters fixed, the release rate is linearily related to the square root of t. Release rate from spherical pellets by this mechanism does not follow a first-order relationship, as the time required to release 50% of the drug from a matrix is expected to be approximately 10% of the time required to dissolve the last trace of the solid drug phase in the center of the pellet. It was shown (9) that dissolution of a soluble drug at high concentration from an insoluble matrix follows the Higuchi square-root equation except during an initial lag phase and a terminal diffusion phase. The main

assumptions made are that the two-dimentional cross-sectional porosity has the same mean as the volumetric porosity, and that the dissolving substance is sufficiently dilute not to affect the porosity. A graphic demonstration of a tablet before (left) and during (right) the drug-dissolution process is shown in the following figure, where the lined areas represent the solvent liquid, and the bold lines - the drug (9):



porosity ε porosity $\varepsilon+A\ (1-\varepsilon)$

RESULTS AND DISCUSSION

A very distinct correlation was obtained in this study between the stearic acid / ethylcellulose ratios and the release rate of 20% cysteine , pressed at 4 tons, from the tablets. The mean release rate at five increasing ratios (expressed as percentage of stearic acid in the matrix mixture) is shown in the following table:

Stearic acid (% in matrix mixture)	Release rate of 20% cysteine
25	4.90
33	4.52
50	3.77
56	3.02
75	2.86

Tablets of 100% stearic acid as matrix liquified during their preparation and could not be manufactured. The significant decrease in rate of cysteine release with increasing the concentration of stearic acid was calculated, and obtained graphically by the computer. In general, the release rate of 20% cysteamine from similar matrix mixtures were much higher than those of cysteine, with 17.20% for 0% stearic acid (i.e. pure ethylcellulose) and 7.10% for 50% stearic acid (i.e. 1:1 ratio). The release rate of cysteamine also decreases with increasing stearic acid concentration in the matrix, and experiments with 90-100% of stearic acid are being carried out currently.

The relation between the concentration of the active drug and its rate of release was also investigated. Preliminary results suggest that the rate increases with increasing percentage of stearic acid. Further studies are needed in order to validate this preliminary finding.

One other factor that has been investigated in this study was the effect on the release rate of the pressure applied to the tablet during its formation. Here, preliminary findings suggest that with 20% cysteamine as the active material, a higher formation pressure results in quicker drug release, while with 20% cysteine no such irregularity was noticed. Although both components of the matrix are hydrophobic, stearic acid is much more hydrophobic than ethylcellulose, due to its waxy character. This character is probably the reason for the decrease in the porosity of the matrix.

Studies of the release of the radioprotectants from their matrices in simulated gastro-intestinal juices are currently being carried out. The results of this work indicate that a correlation might be anticipated between the in vitro profile of release and the in vivo efficacy of the radioprotective agent, due to the sustained-release of the drug.

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INVESTIGATION OF THE SOLUBILITY OF YELLOWCAKE IN THE LUNG OF URANIUM MILL YELLOWCAKE WORKERS BY ASSAY FOR URANIUM IN URINE AND IN VIVO PHOTON MEASUREMENTS OF INTERNALLY DEPOSITED URANIUM COMPOUNDS

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INTRODUCTION

Evaluation of occupational inhalation exposure to uranium is routinely performed using the results of bioassay measurements for uranium in excreta and in vivo scintillation measurements for uranium deposited in the respiratory system of potentially exposed workers (1). Uranium toxicity is similar to other heavy metals, such as arsenic, lead, and mercury, and differs essentially in that uranium is an alpha particle-emitting radioactive material. Therefore, in order to establish an effective worker protection program at a uranium mill, both nephrotoxicity and the radiological hazard to the lung must be considered for the overall assessment of worker exposure.

Recent studies of the solubility of uranium in simulated lung fluid have demonstrated that yellowcake, the generic name of the end-product material from the uranium milling process, does not have a unique solubility classification (2). The solubility of yellowcake has been shown to depend upon the specific chemical separation process employed at the mill (3). That is to say, yellowcake solubility is influenced by the chemical form of the material, the method of chemical extraction of the uranium from the ore, and the temperature at which the yellowcake is calcined or dried. To further exacerbate the lack of a unique solubility classification for yellowcake, each mill will vary its own chemical separation process, depending upon the ore quality, and produce a yellowcake material with slightly different physiochemical characteristics including solubility.

The lack of a specific solubility classification for yellowcake impacts the worker protection program at its most fundamental level. It prohibits development of a simple method for interpreting bioassay and in vivo measurements for uranium which are performed to assess the occupational exposure of workers who come into contact with yellowcake. It is due to the question of the solubility of yellowcake in the human lung that in vivo scintillation measurements for uranium in the respiratory system must be performed as an adjunct to the assay for uranium in excreta.

METHOD

The United States Nuclear Regulatory Commission, Division of Safeguards, Fuelcycle, and Environmental Research is sponsoring a study at the Pacific Northwest Laboratory (PNL) to assess the solubility of yellowcake in the human lung as determined from measurements of workers who were accidentally exposed on the job at a uranium mill. Extensive evaluation of accidental human yellowcake exposures, using

sequential measurements of uranium in excreta and in the respiratory system, should provide data to characterize the solubility of the inhaled material.

Analysis of excreta samples from uranium mill workers routinely indicate the presence of microgram amounts of uranium resulting, presumably, from low-level, chronic exposure at the mill. It is not possible to absolutely identify the source of uranium in these samples as arising from an actual worker inhalation exposure or from external contamination, such as dust in the mill environment.

The first phase of this study is to determine baseline uranium measurements in excreta samples from yellowcake and other mill workers. Intercomparisonsof routine assay procedures at the mills and the methods employed at PNL are performed to relate records of past measurements to the analysis of the current program. Urine and feces are obtained from workers at many different uranium mills. Management personnel from several uranium mills in Wyoming, Washington, Colorado and New Mexico have been very cooperative with this voluntary program and have enthusiastically supported the study. Their assistance in reporting potential overexposure incidents is paramount to the success of the second part of this study, i. e., measuring yellowcake solubility in the lungs of an overexposed worker.

In order to alleviate the potential for external contamination during sample collection and assay volunteer workers are transported to PNL for collection of excreta samples under more controlled circumstances than are available at the mill. The travel arrangements also provide a period of time for recently inhaled soluble airborne particulate to be eliminated prior to initiating sequential sample collections (4). In other words, this phase of the research is an attempt to collect excreta samples for uranium assay in order to characterize any chronic exposure that may exist at the mills.

Measurement of any insoluble uranium material deposited in the respiratory system is performed by simultaneously detecting photon emissions from ^{235}U, ^{234}Th, and x-rays from uranium, protactinium, and thorium (5). Two dual-crystal NaI(Tl)/CsI(Tl) scintillation detectors are placed on the anterior thorax of the worker while he lays prone in a shielded room at the laboratory whole body counting unit. An overabundance of the 216 keV x-rays in an in vivo measurement for uranium in the lung indicates that some fraction of the material is located on the surface of the worker. Evaluation of the internal uranium deposition must be adjusted to eliminate the influence of surface contamination on the worker. Prior in vivo measurements for uranium deposited in the lungs of uranium mill workers have been performed by another researcher without special regard for the presence of external contamination (6). Although, in that study, workers were required to shower and wear coveralls, the presence of uranium bearing dust and imbedded soil on the skin could not have been entirely eliminated by washing.

Once baseline reference values are established for uranium in excreta and in the lungs of uranium mill workers, measurements will be performed with an overexposed yellowcake worker. Uranium excretion from the acutely overexposed yellowcake worker will be compared to baseline values determined from "unexposed" mill workers. The temporal relationship between uranium in urine and feces and the change in the lung burden of the overexposed worker will provide a basis for

describing the solubility of the inhaled material.

It is unlikely that any one of the volunteers who are measured during the initial phase of the program will later become contaminated with yellowcake as a result of an accidental overexposure at the mill. For this reason a representative number of volunteers from several mills must be measured according to the aforementioned protocol. The millworkers will typically spend three to five days at PNL and need only collect excreta and be available for in vivo measurements each day.

The schedule for the overexposed worker will require at least two weeks time at the laboratory for initial sample collections. Immediately following the inhalation incident the worker will be transported by air to PNL so that the metabolism and translocation properties of the rapidly soluble, class D material can be studied. Excreta samples will be collected for several weeks after this initial period. Follow-up in vivo meesurements for uranium in the lung will be performed in order to identify any insoluble uranium material remaining in the lung following the early clearance.

DISCUSSION

A unique application (7) (8) of the dual-crystal detector to the in vivo measurement for uranium in the lung provides a mechanism to distinguish external uranium contamination from that actually deposited in the lung tissue. The thin (3 mm) NaI(Tl) scintillator in the dual crystal detector can identify the presence of uranium on the skin by measuring the ≈ 16 keV x-rays from U, Pa, and Th. Using this same scintillator, the 63 keV and 93 keV ^{234}Th photons indicate the presence of ^{238}U. The thicker (5 cm) CsI(Tl) scintillator simultaneously detects the ^{235}U photons at 186 keV.

In order to take advantage of this technique, a surrogate thorax structure (phantom) is fabricated with a known amount of yellowcake uranium material deposited within the lungs. Two detectors are placed in contact with the surface of the anterior thorax, one detector centered over each lung. A standard ratio of the counts in the x-ray region to the number of counts in the ^{234}Th region is determined from a measure of the phantom. A similar ratio is calculated from an in vivo measurement of a potentially exposed subject. This ratio is first modified for chest wall attenuation and then compared to the surrogate thorax phantom. Detection of an abundance of 16 keV x-rays with the exposed subject indicates that skin contamination is present. The subject in vivo measurement is corrected to account for surface contamination so that the amount of uranium in the lung can be determined.

SUMMARY

Experience has shown that uranium has a low order of chemical toxicity in man and that other heavy metals, such as lead, arsenic, and mercury would produce severe injury at the same levels of exposure (9). However, without a precise knowledge of the solubility of yellowcake in the human lung its hazard to man from an inhalation exposure cannot be adequately determined.

Experiments to measure the solubility of yellowcake in simulated lung fluid indicate that the material exhibits some properties of each classification, i. e., D, W, and Y. This research study attempts

to measure the solubility of yellowcake in the human lung from analysis of the metabolic and translocation characteristics of the material in a worker who has been exposed at a uranium mill. The elimination of uranium from the body will be measured in urine and fecal samples collected from the exposed worker. In vivo scintillation measurements for uranium in the lung will be performed to determine the clearance of the material from the lung.

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EXCRETION OF ORGANIC AND INORGANIC TRITIATED COMPOUNDS IN COW'S MILK AFTER INGESTION OF TRITIUM OXIDE.

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INTRODUCTION

The evaluation of the transfer of tritium in the environment is complicated by the fact that tritium can be incorporated into a variety of organic molecules whose metabolic behavior may differ greatly from that of tritium oxide (2, 3, 6, 7). An important link in the transfer of tritium to man is the secretion of tritiated molecules in milk (4, 5, 8, 10) because this has its impact on infants, the most sensitive part of the population. The secretion in milk of tritium as water and organic molecules (casein and lipids) was, therefore, studied after giving cows tritiated water to drink for a period of 25 days.

MATERIALS AND METHODS

Two lactating cows (weighing about 560 kg) were given drinking water containing 18.9 mCi/l tritium oxide for a period of 25 days (a tritium intake of 871.3 µCi/day, all values given being the means for both cows). The diet consisted of a hay supplemented with a concentrate consumption of food and water (46.6 l/day) as well as production of milk (20.8 l/day), urine (21.9 l/day) and faeces (32 kg/day) were determined daily. Radioactivity in water, dry matter, lactose, lipids and casein of milk was determined during this "loading" period and during a "decay period" of 75 days after application had been terminated. The different constituents were isolated as follows : milk water by scintillation, whole dry matter by lyophilization, lactose by cristallisation in ethanol, casein by precipitation with acid and fats by the Rose-Gottlieb procedure. The radioactivity was determined by liquid scintillation counting after the material had been combusted, except for water and fats which were counted directly. The data were fitted to appropriate functions for the loading and decay period using a nonlinear regression procedure. The calculations were carried out separately for turnover rates and turnover times to obtain standard errors directly. Statistical weighing according to a Poisson distribution was required for the data of the decay period to obtain the second exponential term. No such weighing was needed for the loading period since the range of the activities measured is relatively small. Analysis of variance indicated a good fit for all functions shown. The loading phase is best described by a single term integrated exponential function with a time delay.

$$A = A^{\infty} \left[1 - e^{-\alpha[t - T_d]}\right]$$

where A^{∞} the activity reached after very long times of application, α the turnover rate ($days^{-1}$) and T_d the time delay (days).
The behavior during the decay period can be approached by a two term exponential function :

$$A = A_1 e^{-\alpha_1 t} + A_2 e^{-\alpha_2 t}$$

where A_1 and A_2 the fractions of activity for each turnover component and α_1 and α_2 the respective turnover rates. The formulas for turnover times T (days) are obtained by replacing α with t.

RESULTS

Table 1 presents the parameters and their standard errors for tritium oxide, casein and lipid during the loading period. One recognizes that secretion starts after a short (0.5 day), but significant

Table 1. Parameters ± S.E. for "Loading phase" $\left[A = A^{\infty}\left[1 - e^{-\alpha\left[t - Td\right]}\right]\right.$

Compound	A^{∞} [pCi / g]	α [$days^{-1}$]	T delay [days]	$1/\alpha$ [turnover time] [days]
Water	15 680 ± 49	0.2095 ± 0.00397	0.624 ± 0.057	4.773 ± 0.0904
Lipids	5865 ± 41	0.1478 ± 0.0049	1.08 ± 0.11	6.764 ± 0.225
Casein	2927 ± 26	0.1756 ± 0.0228	0.545 ± 0.105	5.68 ± 0.496

delay; for lipids this delay is about one day. Turnover of all three compounds is short, corresponding to an half life time of about 5 days; it is somewhat longer for lipids than for tritium oxide or casein. No second component can be distinguished during this phase but this is expected since the statistical variability exceeds by far the contribution of such a small fraction. The half life found appears relatively short if compared to that of other nonlactating great mammals (1, 5, 8), but is readily explained by the large water turnover of lactating cows. From the known specific activity of the drinking water and the tritium content of lipid and casein (11% and 7.6% respectively), one can calculate the dilution of the ingested water during metabolism. Thus about 83% of the milk water secreted is found to originate from

drinking water; the rest comes from water in food and from that formed in metabolism. About 38% of the hydrogen in dry matter, 30% of that in lipids, 48% of that in lactose and 22% of that in casein are derived from ingested water or from labeled recycling organic molecules. These values are compatible with observations by others (2, 6, 7).

The parameters of the decay phase presented in table 2 indicate that the principal components of water, casein and lipids have about

Table 2 Parameters ± S.E for "decay phase" $\left[A = A_1 e^{-\alpha_1 t} + A_2 e^{-\alpha_2 t}\right]$

Compound	A_1 [pCi/g]	A_2 [pCi/g]	Time constants α [days^{-1}]	α_2 [days^{-1}]	Turnover times $T_1^{\frac{1}{2}}$ [days]	$T_2^{\frac{1}{2}}$ [days]
Water	16 539 ± 50	7 41 ± 2 15	0 2073 ± 0 00047	0 0172 ± 0 00525	4 822 ± 0 011	58 2 ± 17 8
Lipids	6 845 ± 105	26 6 ± 7 11	0.1834 ± 0.0036	0 00335 ± .00469	5 451 ± 0.071	29 9 ± 41 8
Casein	2994 ± 41	248 ± 47	0 ·889 ± 0 0045	0 0413 ± 0.0056	529 ± 0 ·2	24 7 ± 33

the same turnover rates as during the loading phase. In addition, small fractions (about 0.04% for tritium oxide, 0.4% for lipids and 8% for casein) display much longer half lifes. The half life seems longest for lipids, but its is error marge. It should be noted that the fractions A_1 and A_2 presented in table 2 are those appearing after a loading period of 25 days; the long lived component would be much less important after a single application (table 3).

The data shown allow to calculate how much activity of each compounds and each metabolic component would be excreted after a single and after a continuous application of tritium oxide when integrated over infinite times (table 3). One recognizes that after single application tritium is mainly excreted as tritium water of rapid turnover, organic molecules participate only to about 3.9% and all slow metabolic components represent together about 0.2% of the total activity excreted. It should be noted, however, that when the milk is ingested by man the organic molecules may give rise to a much higher fraction of tritiated molecules with long life span than does tritium water. After continuous application, molecules with long half life, particularly lipids become more important and may represent somewhat more than 4% of the total; the values for lipids remain, however, subject to a rather large error and require further study.

	Fraction (pCi / g or %)		(% of total) Unique application		Continuous application	
	Rapid Component	Slow Component	Rapid Component	Slow Component	Rapid Component	Slow Component
Water	$2.92\ 10^{6}$	11 4				
%	96.7 %	$3\ 97\ 10^{-4}$ %	96	0 0046	92.3	0 052
Lipids	67083	28.9				
%	2.2 %	$9.57.10^{-4}$ %	2.49	0.058	2 71	3.48
Casein	32918	691				
%	1 09 %	$2\ 38.10^{-4}$ %	1.19	0 014	0 85	0 55

Table 3 - Metabolic components after single application and integrated activities after single and continuous application.

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CORRELATION BETWEEN CONCENTRATIONS OF ^{210}Pb IN THE BIOLOGIC SAMPLES FROM MINERS AND INDIVIDUAL LEVELS OF EXPOSURE TO SHORT LIVED RADON-222 DAUGHTER PRODUCTS

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The ^{210}Pb - ^{210}Po skeletal burden may be correlated with specific activities of these elements in biological samples, especially blood and hair levels (1,3). There exists thus the possibility to state some quantitative relations between these parameters and the exposure to inhaled radon daughters, the dose delivered to the bronchial tissue as well as the expected respiratory cancer among miners (4,6).

Compared to radiation exposures evaluated from short lived radon daughters content of mine air, the skeletal levels of lead-210 represents a potentially more accurate measure of miners' exposure. By determining the skeletal reservoir of lead-210 it becomes possible to evaluate the individual cumulated exposures and to make some estimates of prior mine exposures.

MATERIALS AND METHODS

The epidemiological investigation performed by the authors was based on the methodological principle of estimating the miners' exposure both by determining the mine air contamination and the lead-210 blood levels.

As it was expected that cumulated exposure would be estimated from skeletal lead-210, only when steady-state equilibrium was attained between lead-210 bone content in the body fluids, the investigation was carried out only on men who had been exposed over periods of 10 years or more, when miners may be considered to be close to equilibrium (9/10), (2).

The study group included eighty miners and as controls forty adult males, living in the same residence aria as the miners but never being exposed occupationaly to radon daughters.

From the wet-ashed blood samples both ^{210}Bi and ^{210}Po were quantitatively plated on disks. The recovery yield was typically 95 - 8%. The activity of the electroplated elements has been measured by quantitative autoradiography of tracks, using I.F.A.-E.N.1 nuclear emulsion.

The basical hypothesis adopted in the calculation considered that the power function relationship is a statistical property of any biologic sufficiently complex system that has evolved to a state of dynamic equilibrium (7). It is accepted that this kind of mathematical relation characterizes the different biochemical and metabolic mechanisms that involve lead-210 between the actual exposure and the time of sampling (4).

As the power function will be valid only if all lead-210 activity reaching the skeleton is supplied by inhaled radon daughters, the lead-210 body burden originating from other sources was evaluated and suitable corrections had been made (5).

Both WL values calculated from blood ^{210}Pb (4) and data about exposure conditions obtained from occupational histories were utilised in estimating individual cumulated exposures (CWLM).

Simultaneously with the bioassay study, working levels (WL) were determined by dynamic determinations of radon daughter concentrations in the mines, where the investigated group had been working.

For a more accurate quantification of the risk the equilibrium grade between Ra A:Ra B:Ra C, the free atoms fraction and the granulometric distribution of aerosols have been considered.

Rolle's modified method (8) was utilised for routine control of the risk factors in the underground work places with noxious radioactivity.

RESULTS AND DISCUSSION

Pair values of the individual exposures resulted from radon decay products inhalation determined by the two methods are plotted in figure 1.

The good agreement between the WLM values obtained from excess lead-210 in bone and those derived from direct measurements of radon daughters in the mine atmosphere emphasizes that the skeletal reservoir of lead-210 may be an extent of the individual exposure to short lived radon daughter products and that the miner's organism behaves as its own dosemeter.

The WLM values resulting from the biological assay are in most cases higher, revealing more important exposures prior to our making the determinations. It may be due either to increased radon emanations or to worse work conditions.

It must be noted that the two value series are not completely independent of each other, as both resort to the occupational histories of miners.

On the whole it may be concluded that the cumulated exposure estimated from the lead-210 body burden expresses more accurately the actual risk, as it is an individual indicator that takes into account the feature

PAIR VALUES OF THE INDIVIDUAL EXPOSURES

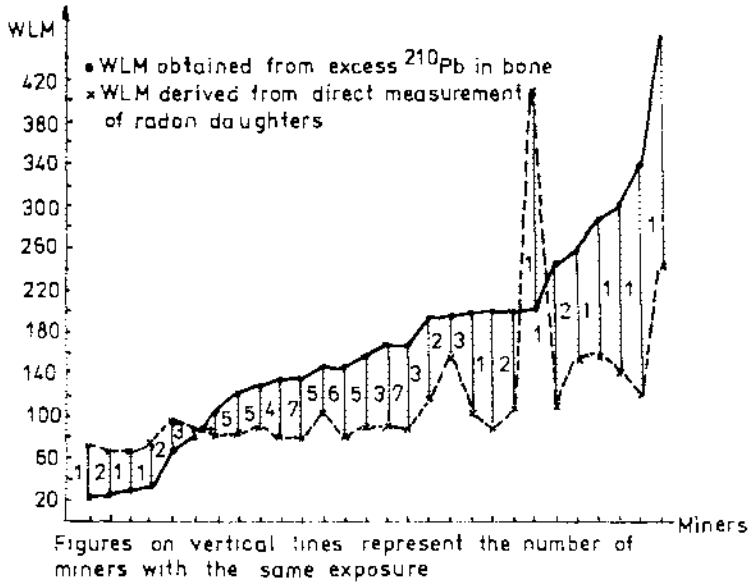

Figure 1

of the organism, the specific of the worker's profession and integrates the aleatory changes in radon daughter concentrations in both their temporal and work place dynamics.

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FATE OF MAJOR RADIONUCLIDES IN THE LIQUID WASTES RELEASED TO COASTAL WATERS*

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In the liquid radwaste released to coastal waters from the nuclear power stations, fission products - radiocesium and radioiodine - and activation product - ^{60}Co are reported as the critical radionuclides (1,2). Behaviour of these nuclides in the sea water depends on the ionic or colloidal state of radionuclide. Part of the discharged radionuclides get absorbed as soon as it comes in contact with suspended silt of sea water and slowly settle to the bottom. Radionuclides are biologically accumulated by marine organisms and also by the algae and ultimately reach human body through consumption of sea foods. In case of sea foods like crabs and prawns significant portion of radionuclides are in the non-edible part which reduces naturally the intake by man. The behaviours of ^{60}Co in sea water has been reviewed by Fukai & Murray (3) in detail.

The study presented here has been carried out in the coastal waters at the Tarapur Atomic Power Station (TAPS) operating from 1969. Controlled release of low level liquid wastes are made to the coastal waters. The water movement studies (1) have shown the oscillating nature of Tarapur near shore water with slow mixing with deeper sea waters. The coastal waters have silt varying from 50 to 200 mg/litre during non-monsoon period, but it goes as high as 1000 mg/litre during monsoon. At TAPS the liquid effluent is injected to the condenser coolant sea water at the outfall where it gets throughly mixed and then this diluted effluent flowsout in open canal along the coast and joins the tidal waters. The activity discharged builds up in the oscillating coastal water body and gets distributed in sea water, silt, algae, fish and other foods. The chemical state of the critical nuclides in sea water, silt absorption and desorption and biological uptake of radionuclides by the marine organisms in the near shore region are described in this paper.

FATE OF THE NUCLIDES AT THE STATION OUTFALL

The liquid waste before release to the condenser coolant is adjusted to pH of 7.5 to 8.5 and the sea waters receiving this effluent has pH of 8.0. Dialysis experiments were carried out on the radwaste sample and also on the sea water mixed radwaste using cellulose tube membrane of 4.8 millimicron pore size. In case of $^{134+137}Cs$ and ^{131}I in radwaste nearly 100% passed through the dialyser membrane but in case of ^{60}Co only 15 to 20% passed through the membrane. Filtered sea water after spiking with radwaste and dialysis showed almost same result for ^{137}Cs and ^{131}I. Only 5 to 10% of ^{60}Co in filtered sea water passed through the dialyser. Thus, I and Cs which are in ionic state in sea water pass through the membrane but Co

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being hydrolysed completely at pH 8, more than 90% is likely to be in hydrous oxide particles of size more than the pore diameter (4.8 millimicron) of the dialyser. In case of cobalt Fukai & Murray (3) have reported negligible formation of cobalt hydroxide in sea water contrary to our observation.

IMMEDIATE SILT ADSORPTION BY SUSPENDED SILT

Liquid radwaste (10 ml) of known composition was mixed with fresh sea water (2 litres) stirred for a minute and filtered through millipore filter paper. The silt with adsorbed radionuclides was retained on the paper which was quantitatively counted by gamma spectrometry. The sea water silt content was 82 mg/litre during the experiments. Observed instantaneous silt adsorptions were ^{137}Cs 1.5 to 5%, ^{131}I 1 to 3% and ^{60}Co 80 to 99.5%.

EXCHANGE OF SILT ADSORBED RADIONUCLIDES

Samples of silt deposited at the bottom of discharge canal and nearby coastal areas were studied for the exchange of activity of the radiocesium and radiocobalt adsorbed on them. Silt at Tarapur coastal area contains about 85% of particles of size 50 microns or smaller. Coarse and fine silt having adsorbed ^{60}Co and ^{137}Cs were leached with 1 M ammonium acetate. The total activity on silt and in ammonium acetate leach were determined by gamma spectrometry. It was observed that only 0.5% of ^{60}Co was leached from fine silt and about 5% was leached from coarse silt by amm.acetate. 30 to 40% of total ^{137}Cs was leached from both coarse and fine silt by amm.acetate. Thus, cobalt adsorbed on silt is not easily available for exchange reaction where as cesium is labile to a significant extent. The exchangeability is seen decreasing with particle size. Organisms get their food through deposited silt and the labile activity in the silt would be easily available for biological uptake.

ACCUMULATION OF RADIONUCLIDES IN MARINE ALGAE

In the coastal environment around the radwaste release point, there is an abundent growth of algae (sea weeds) during the non-monsoon months. From the average radionuclide contents of sea water and algae, the concentration factors calculated for the three varieties of weeds are given in Table 1.

TABLE 1. Radionuclide concentration factors observed for marine algae in the Tarapur environment.

Species	Concentration factor = $\frac{\text{pCi/kg (algae)}}{\text{pCi/litre sea water}}$ Range (average) for the nuclides		
	^{131}I	$^{134+137}Cs$	^{60}Co
1. Sargassum	354 - 2295(1098)	76 - 195 (115)	235 - 920 (567)
2. Ulva Lactuca	77 - 226 (153)	3.6-33.5 (20)	65 - 292 (175)
3. Entromorpha	20 - 436 (192)	51 - 502 (275)	73 - 358 (215)

Fukai & Murray (3) have reported the concentration factor for marine algae as 2000 from the analysis of stable cobalt. The (CF) obtained here from ^{60}Co concentrations is 4 to 10 times less.

ACCUMULATION IN ORGANISMS AND SEA FOODS

Crabs, prawns/shrimps, shell fish, onchidium (gastropod) and small fishes are the main sea food organisms caught in the Tarapur near shore region. The edible soft tissues and non-edible shells and scales were tested separately. Significant amount of radionuclide accumulations on the non-edible portions were observed. The percentage of radionuclide in the edible tissue portion of the organisms are given in Table 2.

TABLE 2. Percentage of radionuclide activity in the edible portion of the coastal sea food.

Sea food variety	Percent of edible tissue to the total wt.	Percentage activity in the edible tissue of sea food		
		^{131}I	^{137}Cs	^{60}Co
1. Shrimps	30.0	12.0	78.0	40.0
2. Prawns	60.0	8.0	47.0	55.0
3. Crabs	81.0	70.0	79.5	71.5
4. Shell fish	11.0	Not analysed	31.0	5.0

Significant amount of activity present in the non-edible portion is a favourable factor decreasing the human intake through sea food.

CONCENTRATION FACTORS FOR SOFT TISSUES OF ORGANISMS

Radionuclide concentration factors (CF) in the sea foods help in radiation exposure evaluation. In the near shore environment of Tarapur Power Station, the CFs for the radionuclides in the coastal organisms were determined under natural conditions and the results obtained are shown in Table 3.

TABLE 3. Concentration factors of radionuclides in marine organisms in the near shore environment.

Organisms	Concentration factor = $\frac{\mu Ci/kg}{\mu Ci/litre}$		
	^{131}I	^{137}Cs	^{60}Co
1. Prawns	11 to 60	6 to 41	10.5
2. Onchidium	13.2	22.5	1.06×10^3
3. Oysters	29.7	26.7	40.0
4. Bombay Duck	11.2	6.5	15.5
5. Crabs	31 to 93	5 to 51	47 to 106

The near shore fishes taking up radiocobalt from coastal water medium have concentration factors ranging from 10 to 30 only, compared to 10^2 to 10^3 reported (3,4) for marine environment from the study of stable nuclides.

CONCLUSION

When ^{131}I, ^{137}Cs and ^{60}Co are released to coastal waters, ^{60}Co gets almost completely and irreversibly adsorbed on the silt, ^{131}I and ^{137}Cs get reversibly adsorbed to a fractional extent.

All the three nuclides are picked up by the crustaceans and benthos to a higher extent than fishes in the near shore region. The radionuclides considered reach the population mainly through sea food items.

The concentration factors observed for these radionuclides in coastal waters are low compared to general CF's reported (4) for sea waters. This may be due to the fact that near shore waters have about 10 to 30 times more inactive trace element content compared to off-shore waters.

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ASSESSMENT OF ^{210}PO EXPOSURE FOR THE ITALIAN POPULATION

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The transfer to man of the normal environmental levels of the naturally occurring ^{210}Pb and of its descendant ^{210}Po is considered to contribute a large fraction of the natural radiation dose to man from internally deposited radionuclides (8). While ^{210}Pb being a beta emitter, contributes a small fraction of the entire dose due to the ^{210}Pb series, the alpha emitter ^{210}Po can be taken as the largest concurrent to the total natural internal dose to the skeleton of members of the general population (8).

^{210}Po has by itself a short effective half-life in the human body (6), which reduces its ability to accumulate in tissue, while the parent nuclide ^{210}Pb due to both its long physical half-life and metabolic behaviour in man, accumulates in bone(6). However, the ^{210}Po formed from ^{210}Pb in bone, it appears to remain in the skeleton in equilibrium with its parent (6). Therefore the knowledge of the pathways from environment to man and of the metabolic properties of ^{210}Pb and ^{210}Po is needed to assess the natural radiation dose due to the ^{210}Po burden in the members of the public. The dietary intake of ^{210}Pb is usually considered the main source of the ^{210}Pb - ^{210}Po internal burden in the general population (4, 7-8).

Increases of ^{210}Pb - ^{210}Po skeletal burden have been reported (1, 3, 6) in subjects exposed in uranium mines and spas to high levels of radon and daughter which are the progenitor of the ^{210}Pb series.

In this paper are summarized the experimental data needed to evaluate the ^{210}Po internal burden and the relative dose given to members of the general Italian population and to those groups of subjects which are exposed in non uranium mines and spas to high radon and daughter air concentration.

MATERIALS AND METHODS

The ^{210}Po content, reported to the sampling time by correcting for decay and ingrowth from ^{210}Pb, has been measured in the following items: - About 40 samples of complete daily diet collected in six different regions of Italy and referred to adult members of the public.

- About 40 samples of urinary excretion collected from adult members of the public both non-smokers and smokers.

- More than 150 samples of urinary excretion collected from subjects working in spas and in non-uranium mines.
- About 20 samples of tooth and bone collected at various ages from members of the public.

The ^{210}Pb content, reported to the sampling time, was measured in most samples. The ^{210}Pb analysis has not been performed in those samples where the long time elapsed between collection and analysis was sufficient to give equilibrium conditions between the components of the ^{210}Pb series at the time of the analysis. The analytical procedure employed to determine ^{210}Pb and ^{210}Po in the various samples considered has been described in detail elsewhere (4). The method is based on a coprecipitation of polonium with manganese dioxide after a wet ashing of the sample. The ^{210}Po plated on silver disk is then measured by solid state detector spectrometry. The chemical yield of any analysis is tested by using ^{208}Po as an internal standard. The ^{210}Pb was evaluated from the ^{210}Po ingrowth in the sample by repeating the ^{210}Po analysis some months after the initial analysis.

RESULTS AND DISCUSSIONS

In table 1 are summarized the data referred to the main components of the metabolic balance of ^{210}Pb - ^{210}Po, evaluated on the basis of the ^{210}Pb - ^{210}Po content measured in the analyzed samples.

TABLE 1. Summary of the components of the ^{210}Pb - ^{210}Po metabolic balance in adult members of the general italian population.

	^{210}Pb	^{210}Po
Daily intake (Bq/day)		
Ingestion (Diet+Water)	0.11	0.11
Inhalation (non smoker)	0.006	0.0007
Daily excretion (Bq/day)		
Feces	0.095	0.095
Urine	0.015	0.015
Total Body Burden (Bq)	26	23
Skeletal Burden (Bq)	20	16

The daily inhalation has been calculated by measuring the ^{210}Pb ^{210}Po concentration in some air samples. The daily fecal excretion has been evaluated by applying the recently reported value (7) of the (daily fecal excretion) / (daily urinary excretion) ratio to the

urinary excretion measured in our subjects. The ^{210}Pb - ^{210}Po concentrations measured in teeth have been found not significantly different from those in bone samples, the mean ^{210}Po concentration value ± S.E. being 3.2 ±0.5 mBq/g. The data of table 1 should be considered entirely indicative of the ^{210}Pb - ^{210}Po intake, excretion and internal burden of the non-smoker adult members of the general italian population, owing to the large individual variability of the ^{210}Pb - ^{210}Po excretion rates at natural levels (5). Furthermore the daily dietary intake of ^{210}Pb - ^{210}Po has been found quite variable as a function of many factors (4). The highest ^{210}Pb - ^{210}Po daily dietary intake (0.3 Bq/day) being measured in diet samples collected in an area with high natural background. The mean ^{210}Pb - ^{210}Po daily intake measured in Italy is in good agreement with the data reported (4,8) for other european countries, while is higher than in some extraeuropean countries (4,8).

A mathematical model has been developed on the basis of the data given in table 1 to describe the metabolic behaviour in man of the systemic ^{210}Po and it should be considered valid for any introduction of ^{210}Po into the systemic compartment of the human body. A detailed description of the model, reporting the retention function in soft tissue and bone and the urinary excretion function for ^{210}Po, is given in (2). These functions have been used to evaluate the ^{210}Pb - ^{210}Po body burden of the subjects exposed to high levels of radon and daughter on the basis of their ^{210}Pb - ^{210}Po daily excretion. The model has been satisfactorily tested on ^{210}Pb - ^{210}Po intake and excretion data reported by other authors (7).

In table 2 are summarized the ^{210}Pb - ^{210}Po internal burden and the resulting dose rate to the skeleton of four groups of subjects selected according to the origin of the ^{210}Pb - ^{210}Po exposure.

TABLE 2. Body burden and yearly dose rate for adult members of the italian population according to the origin of exposure.

Population Group	Body burden (Bq)				Yearly dose rate to the skeleton (mGy/year)
	Skeleton		Soft Tissue		
	^{210}Pb	^{210}Po	^{210}Pb	^{210}Po	
General population non smoker	20	16	6	7	0.07
General population smoker	31	20	9	11	0.10
Non-uranium miners	150	110	30	25	0.60

Spa employees	480	370	110	90	2

The smoker subjects have a ^{210}Pb - ^{210}Po urinary excretion significantly higher than the non smokers, thus confirming the importance of smoking among the various natural sources of ^{210}Pb ^{210}Po contamination in man. A detailed discussion of the data reported in table 2, of their significance and of the dosimetric evaluations is given in (4). The mean dose rate to the human skeleton, due to the ^{210}Po bone content, has been assumed to be equal to the mean ^{210}Po dose rate received by the cells of the cortical bone (4,8). The yearly dose rate reported in table 2 for the adult members (smoker and non smoker) of the general italian population is in good agreement with similar data reported by the Unscear (8). The dose rate referred to spa workers and non-uranium miners should be considered purely indicative of the highest levels of exposure to radon and daughter in those environments. The ^{210}Pb-^{210}Po skeletal burden due to the radon and daughter exposure resulted to have large individual variability as a function of many factors which may seriously affect the lung retention and clearance of the radon daughter which are decaying to ^{210}Pb inside the lung (3,4). Nevertheless the reported data permit to establish the relevance of the ^{210}Po dose to the skeleton of subjects which are currently exposed or have been exposed in the past to high levels of radon and daughter in various environmental conditions.

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REPRESENTATION GEOGRAPHIQUE DE DIVERSES DONNEES DANS UNE GRILLE EUROPEENNE

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RESUME

On décrit un système de représentation uniforme permettant de collationner de manière homogène diverses données : population, données physiques, ressources agricoles, informations sanitaires.. dans les mailles de dimensions appropriées d'une grille européenne. Divers programmes ont été mis au point pour l'utilisation de ces données en vue de l'évaluation des doses collectives.

INTRODUCTION

Pour l'évaluation de l'équivalent de dose collective consécutive à la libération d'effluents radioactifs dans l'environnement, il est nécessaire de connaître la répartition géographique de la population et des productions agricoles dans les régions susceptibles d'être contaminées, afin d'évaluer les quantités de polluants qui, directement ou indirectement, pourraient être inhalées ou ingérées. A l'échelle européenne, on a établi un système défini par des coordonnées géographiques, permettant d'obtenir un ensemble homogène d'informations à partir de données initiales diversement présentées selon les pays - les unes (populations) se rapportant à des régions géographiquement définies, mais dans des systèmes différents de coordonnées - les autres (statistiques agricoles) se référant à des divisions administratives.

EXPOSE ET DISCUSSION

Définition du système (figure 1)

La zone considérée est délimitée environ par les parallèles 37° N et 60° N et par les méridiens IO° W et 20° E de Greenwich, mais le système retenu permet de l'étendre en latitude et en longitude. En effet, la zone est partagée en fuseaux de 1,5°, de part et d'autre du méridien de Greenwich ; le long des méridiens, on a porté des subdivisions variables, de façon à obtenir des mailles de surface équivalente se déformant progressivement du Nord au Sud. L'espacement des parallèles a été calculé de façon que chaque maille, quelle que soit sa position, ait une superficie de IO 000 km2 qui peut être subdivisée par interpolation linéaire, celle-ci étant facilitée par la transformation des coordonnées dans le système centésimal (grades). Ce maillage est directement utilisé dans la transformation des données agricoles disponibles au niveau de divisions administratives (proportionnalité pour les productions et cheptel avec les surfaces) C'est un maillage cent fois plus fin qui est utilisé pour répartir

les effectifs des populations.

Le code de repérage des mailles est simple et très souple. Il utilise un double indiçage, en latitude à partir du parallèle 49° N, en longitude à partir du méridien de Greenwich. Chaque indice peut être positif (vers le Nord ou l'Est) ou négatif (vers le Sud ou l'Ouest) et comporte deux ou trois chiffres selon la finesse de la maille. L'extension du code à une grille plus étendue ou plus fine est donc aisée.

Répartition géographique de la population [1]

Les données proviennent des recensements nationaux, effectués selon les pays entre 1970 et 1975. Elles sont présentées soit par unités administratives (généralement les communes), soit dans une grille à mailles régulières (de 10 km x 10 km ou de 1 km x 1 km selon les cas).

La localisation des centres des communes ou des mailles dans le système de coordonnées géographiques indispensable pour unifier la présentation des données a exigé un gros travail, soit sur le plan cartographique, soit sur le plan informatique.

Plusieurs programmes informatiques ont été mis au point pour associer les données provenant des recensements de population et celles relatives aux coordonnées des centres des agglomérations ou des bornes de maillage. On a notamment classé,par ordre croissant des coordonnées géographiques,les données population des différents pays, ce qui permet, par exemple, de calculer la densité de population dans chaque maille en confondant les nationalités (tableau 1).

TABLEAU 1. Population dans la maille (5, 1) = 2679368

Ab. / Or.	50	51	52	53	54	55	56	57	58	59
I9	8105	4401	7802	7185	108060	17 59	5466	9521	9218	16736
I8	10504	2352	7627	23363	6924	6690	5474	1482	45191	5420
I7	89107	6832	7324	13545	3357	6400	11655	15321	10360	2217
I6	19545	10164	8621	12117	10347	17174	13550	18748	13092	19410
I5	31065	3933	10123	11051	31207	25996	29573	30440	15534	18623
I4	71057	34629	3825	4846	40744	28408	60344	96812	33521	16224
I3	107472	47471	5210	9513	24417	102305	63491	106512	47662	45768
I2	30694	27525	1648	7240	27656	80100	192464	41145	18192	10159
11	40162	149796	4917	3756	35958	49890	27544	44782	6992	3644
10	16341	14317	4838	10726	9302	5656	11380	8831	7658	10730

Acquisition et transformation des données agricoles [2][3]

Les données agricoles de base (moyenne des années 1970 à 1973) sont obtenues à partir d'un fichier donnant des caractéristiques de production végétale (surface, rendement, production des effectifs animaux, et ce pour des divisions administratives de base équivalentes d'un pays à l'autre (quelques milliers de km^2)). Les données manquantes sont estimées en fonction de surfaces cultivées pour les végétaux et d'effectifs animaux (produits laitiers et viande) et de données plus générales disponibles au niveau national pour les pays

à faible étendue et au niveau régional pour les autres. Un exemple de résultats figure au tableau 2.

TABLEAU 2. Production de lait de consommation (milliers de tonnes, années 1970-1973) dans la zone MNPQ (figure 1)

Ab. / Or.	-2	-1	1	2	3	4	5	6	7	8
+4	417,1	227,3	108,6	6,4	90,0	382,3	338,4	166,2	111,4	22,4
+3	605,6	260,0	154,5	89,5	213,7	208,8	169,1	118,7	90,4	1,1
+2	45,9	11,6	14,6	210,0	253,2	241,8	160,3	104,4	107,6	111,3
+1	10,6	37,9	83,1	148,5	133,0	89,1	113,2	86,0	132,7	162,5
-1	57,1	89,0	53,9	30,3	20,2	52,5	106,1	141,7	193,5	295,3

DISCUSSION

. Les dimensions des mailles utilisées paraissent bien adaptées aux possibilités d'obtention des données et à leur utilisation pour les évaluations à l'échelle régionale en ce qui concerne tant les populations que les productions agricoles. Il va de soi que l'étude approfondie de l'environnement d'un site particulier requiert des données plus détaillées .

. La grille de données concernant les productions agricoles sera améliorée dans les mises à jour ultérieures (tous les cinq ans environ) : en s'efforçant d'acquérir plus de données au niveau des divisions administratives de base, en précisant, dans certains cas, les productions destinées à l'homme ou à l'animal (cas des pommes de terre, des céréales), en effectuant des corrections de répartition utilisant le maillage fin avec pour les zones côtières (la présence de population)et pour les zones montagneuses (la prise en compte de l'altitude).

. D'autres programmes permettant de produire les données dans des configurations différentes, telles que cercle ou secteur, selon le mode de présentation des autres données du problème.

CONCLUSION

Ce système homogène d'informations, établi à l'échelle européenne, et encore susceptible d'améliorations, est directement utilisable pour les évaluations d'impact radiologique des installations nucléaires [4] .

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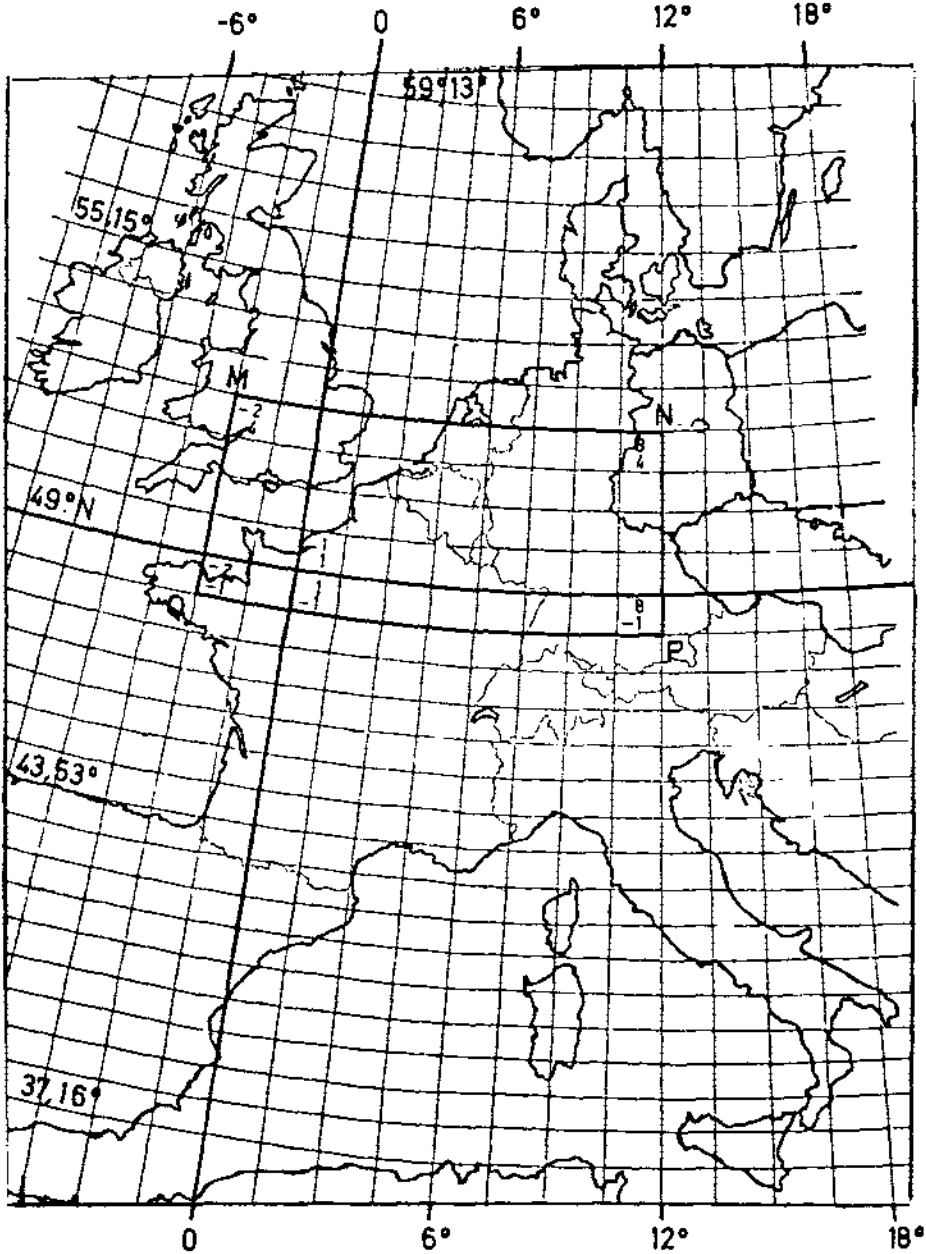

Figure 1 : Définition du système de grille européenne

STUDIES OF CONCENTRATION AND TRANSFER FACTORS OF NATURAL AND ARTIFICIAL ACTINIDE ELEMENTS IN A MARINE ENVIRONMENT.

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Natural (thorium and uranium) and artificial (plutonium and americium) actinides have been studied in a marine environment at the Swedish south west coast.

The actinides have reached the water from fall-out, run off from land, leaching, resuspension or by in situ build up. The partition of these elements between water, sediment and organisms was studied. Special concern was given to plutonium and americium.
Such investigations play an important role for predicting the fate of actinides released into the sea by dumping or wastes from nuclear industry. Their fate will depend on type of source and the physical and chemical parameters.

The brown algae Fucus vesiculosus and Fucus serratus was shown to be excellent bioincicators for actinide elements. Other investigations have frequently used mussles as bioindicator for transuranium elements.

MATERIALS AND METHODS

Fucus vesiculosus and F. serratus have been collectec at the Swedish south west coast occationally during 1967-1976 (1). After 1976 a more regular collection has taken place.

During 1978 and 1979 several watersamples and sediment cores were taken.

The actinides were separated radiochemically by procedures published elsewhere (2). As radiochemical yielddeterminants ^{242}Pu, ^{243}Am, ^{232}Th and ^{229}Th were used. The alpha activity of samples electro deposited onto stainless steeldiscs were measured with surface barrier detectors for 1-20 days.

RESULTS AND DISCUSSION

The area studied is situated in an area around $56^{o}N$, $13^{o}E$. It could briefly be characterized as very shallow with strong currents, high resuspension and high organic particulate in the water.

ACTINIDES IN WATER

In table 1 the results for water samples are given. All values refer to unfiltered water. We find lower plutonium and americium concentrations compared to most other areas such as the Atlantic and the Mediterranean (3, 4). Our results are in agreement with investigations by Murray et. al. (5).

Bottomwater (19 m) shows higher plutonium and americium concentrations than surface water. The uraniumconcentration could be repre-

sentative for this brackish water (salinity 7-10 o/oo at surface). We can not observe any influence of transuranics from european reprocessing facilities although this has been done for ^{134}Cs (6).

We have not yet investigated the partition between particulate and soluble fraction of the actinides. We believe the transuranium elements are predominantly in a particulate form due to the high particulate load in this region.

Table 1. Actinides in water and Fucus

	^{228}Th	^{230}Th	^{232}Th	^{234}U	^{235}U	^{238}U	$^{239+240}Pu$	^{241}Am
Water (μBq/kg)								
Surface	450	270	70	18100	650	15900	14	3
Bottom (19m)							40	7
Fucus (mBq/kg) dry)	5000	400	90	12700	450	11100	190	53
CF (dry)	11000	1300	1300	700	700	700	13500	18000

ACTINIDES IN SEDIMENT

The results for sediment support the theory of rapid sedimentation for the transuranium elements in this area. The concentrations in surface sediments are a factor of 3-4 higher than for other regions contaminated from global fall-out. This in combination with the low concentrations in water gives a dry sediment/water activity concentration ratio for plutonium as high as 250 000.

The results varied much depending on site of collection, but in table 2 are given the average values for 6 representative cores. The integrated area content for these cores was estimated to 72 Bq/m^2 and 20 Bq/m^2 for $^{239+240}Pu$ and ^{241}Am respectively. This is much higher than expected from integrated fall-out at this latitude (48 and 13 Bq/m^2) indicating significant run off from land.

Table 2. Plutonium and americium in sediment cores. (Bq/kg dry)

Depth (cm)	$^{239+240}Pu$	^{241}Am	$^{241}Am/^{239+240}Pu$
0-2	80	18	0.20
2-4	79	17	0.23
4-10	12	4	0.32

Americium and plutonium show different vertical distributions. The activity ratio Am/Pu increase with depth although this ratio agrees with integrated fall-out in the integrated cores. This difference is explained by that americium has mainly been formed *in situ* by the decay of ^{241}Pu($T_{\frac{1}{2}}$=14.2 a). Plutonium deeper down is older and shows then higher Am/Pu ratio. We do not believe that this is an effect

of higher penetration for americium. Of the integrated area content for water and sediment only 0.3 % of plutonium and americium is present in the watercolumn. This is in large contrast to the situation for example in the Mediterranean (7).

ACTINIDES IN ALGAES

The results for actinides in Fucus can be seen in Table 1 and Fig. 1. All values refer to the date of collection. The measured ^{241}Am was corrected to the date of collection by using $^{241}Pu/^{239+240}Pu$ activity ratios measured in global fall-out (8). The $^{239+240}Pu$ concentration in Fucus has decreased during the period. Such a decrease is expected due to that the main fall-out delivery ocurred shortly after 1962.

^{241}Am on the other hand shows an increase during the same period. This increase must be related to the in situ build up of ^{241}Am ($T_{\frac{1}{2}}$= 433 a) from ^{241}Pu($T_{\frac{1}{2}}$=14.2 a) in the environment.

Most americium values fall below the expected ones from Am/Pu activity-ratios in integrated fall-out.

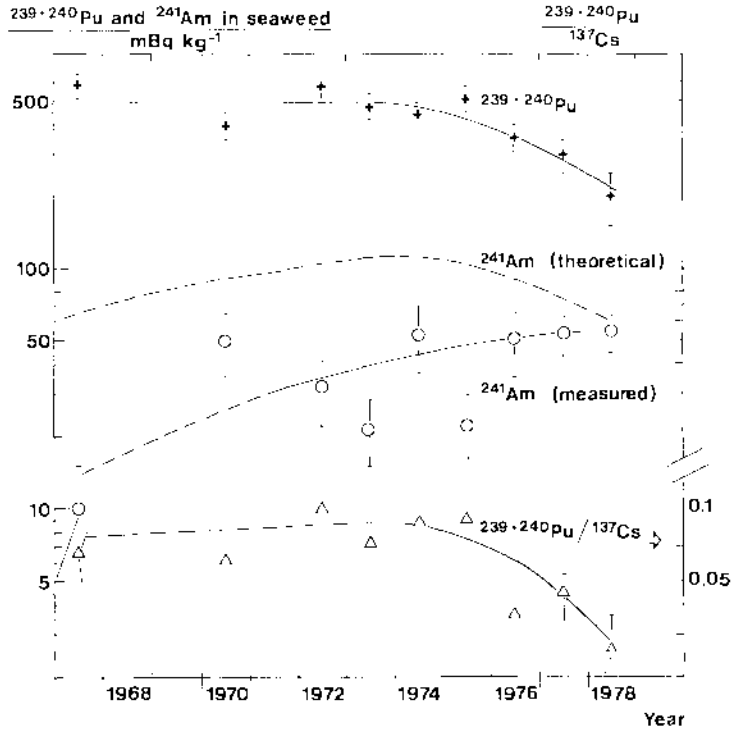

Figure 1. $^{239+240}Pu$ and ^{241}Am activity concentrations in Fucus during 1967-1978. The Pu/Cs activity ratio is also displayed.

Our results indicate that the concentration factor for americium from water to Fucus is higher than for plutonium. The processes in-

volved to reach water and organisms for *in situ* produced americium are different to those for plutonium.

The ^{137}Cs concentration in *Fucus* has been rather constant during 1967-1978 (1), resulting in a decreasing Pu/Cs activity ratio from 0.1 to 0.02 during the period.

By using ^{137}Cs values from Aarkrog et al (9) the $^{239+240}Pu/^{137}Cs$ activity ratio in water in 1978 was about 0.002-0.003. This illustrates the much higher uptake of plutonium than cesium by Fucus.

Recently Hodge *et.al.*(10) showed that uptake of actinides in particulate form by coastal marine organisms was significant. Our $^{234}U/^{238}U$ activity ratio in *Fucus* is about 1.14. The soluble element uranium shows the lowest concentration factor. Plutonium and americium which are expected to be oredominantly in the particulate form, in this area, show the highest concentration factors. Our results indian organic particulate sorbtion of actinides by *Fucus*.

Of the thoriumisotopes ^{228}Th differs remarkably from the others. The lifespan for the algaes is not long enough to explain this high CF for ^{228}Th by *in vivo* build up from ^{228}Ra. This might indicate that ^{228}Th is brought into a more bioavailable form through the decay - chain ^{228}Ra-^{228}Ac.

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^{226}Ra- AND ^{222}Rn-CONTENT OF DRINKING WATER

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^{226}Ra- and ^{222}Rn-concentrations were measured in drinking water in parts of the Federal Republic of Germany. 254 samples were analysed for ^{226}Ra, 57 for ^{222}Rn. The average contents were found to be 4,4 x 10^{-3} Bq/l (0.2 pCi/l) for ^{226}Ra and 7,4 Bq/l (0.2 nCi/l) for ^{222}Rn with maximum values of 0.11 Bq/l (3 pCi/l) and 43 Bq/l (1.2 nCi/l), respectively. For both radionuclides there was a log-normal distribution of concentrations. Statistical analysis revealed no correlation between ^{226}Ra and ^{222}Rn, both for processed drinking and raw water. It is shown that there is a significant influence of the processing method on the final content of radioactivity. The average yearly doses to the critical organs are estimated to be about 0.003 mSv/a for ^{226}Ra (bone lining) and about 70 µSv/a for ^{222}Rn (stomach).

INTRODUCTION

Radium and its daughter products constitute an important part of natural environmental radiation exposure. Since ingestion forms a major pathway - apart from inhalation for Radon - for internal irradiation, the measurement of radioactivity in drinking water is relevant to assess the contribution of these environmental radiation hazards. Although a considerable body of information is already available it is generally felt that more data are desirable (1). We started therefore a survey of ^{226}Ra- and ^{222}Rn-concentrations in a part of central Germany to obtain also a better insight into the distribution of the measured parameters.

MATERIAL AND METHODS

Scope:

Random samples of drinking water (257 for ^{226}Ra and 57 for ^{226}Ra and ^{222}Rn) were taken in Hessia, a state of the Federal Republic of Germany during 1977 - 1979. In some instances the waterpath was followed from the well to

the final consumer.

Measurement

^{226}Ra and initially ^{222}Rn were measured using Lucas szintillation chambers as already described (2). Later ^{222}Rn was determined by means of liquid szintillation in a Beckman LS 8000 according to Pritchard and Gesell (3).

RESULTS

Figure 1 shows the sum distributions of the measured values on a probit scale with a logarithmic abscissa. The approximately straight line indicates a log-normal distribution. The mean values are 0.2 pCi/l for ^{226}Ra and 0.2 nCi/l for ^{222}Rn. In 50 samples both nuclides had been measured. Statistical analysis of these data showed no correlation, the correlation coefficient was 0.12 ($p < 0.9$). Since it is known that water processing strongly influences the radionuclide distribution (see below) the same analysis was carried out with unprocessed well water (22 samples). The correlation coefficient was 0.28, which is again too small to accept a statistical correlation.

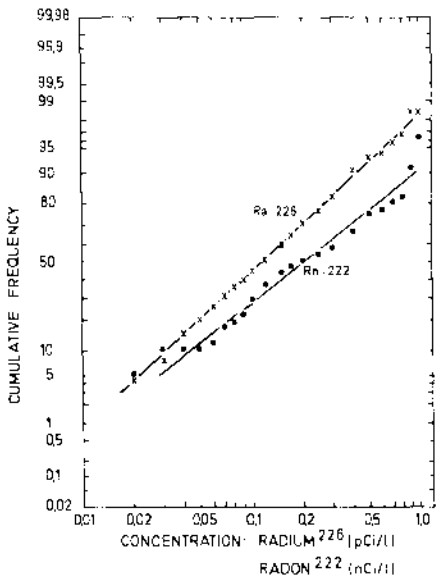

Figure 1.
Sum distribution of ^{226}Ra- and ^{222}Rn-concentrations

In some places it was possible to follow the radionuclide content from the well to the final consumer. Two examples are shown in figure 2A and 2B: In the first one the processing consisted of an aeration stage with subsequent fast filtration through a bed containing oxydizing sub-

stances to precipitate Fe- and Mn-contamination.

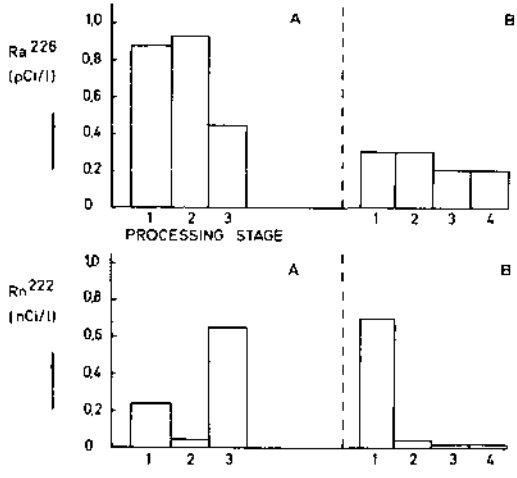

Figure 2. Influence of water processing on radionuclide content: A: Processing using rapid filtration with oxydizing substances; B: slow filtration; stages: 1. Well water; 2: after aeration; 3: after filtration; upper panel: ^{226}Ra; lower panel: ^{222}Rn

It is seen that the ^{226}Ra-content drops to about 50% during filtration, presumably due to Ra-binding in the filter bed. ^{222}Rn is lost drastically by aeration, but its concentration rises to rather high levels after filtration. We suggest that this is caused by Ra-accumulation in the filter bed. If a slow filtration process is used (figure 2) the decrease in ^{226}Ra is smaller and there is no increase in ^{222}Rn after the last stage. It is clear from this comparison that the processing method influences the final concentrations in drinking water.

DISCUSSION

We have measured the ^{226}Ra- and ^{222}Rn-content in drinking water in a part of Germany. The average values found agree reasonably with those reported for other areas (4). If we assume a daily consumption of 1.2 l drinking water the mean dietary intake is 0.14 pCi/l. According to UNSCEAR 77 (1) this would lead to a bone activity of 0.9 pCi/kg^{-1}. From this we estimate a yearly dose equivalent of about 15 µ Sv/a (1.5 mrem/a). For ^{222}Rn only unboiled drinking water has to be taken into account (about 0.15 l/d) which exerts its action mainly on the stomach. The mean somatically significant dose equivalent to this

organ is then estimated to be 70 μ Sv/a (7 mrem/a) (5). Our results demonstrate that the yearly dose-rate expected from ^{226}Ra and ^{222}Rn intake via drinking water is low compared to other radiation sources.

Acknowledgement

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MOBILITY AND RETENTION OF ^{60}Co IN SOILS IN COASTAL AREAS

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The operation of more than 20 commercial nuclear reactors, at coastal sites in Japan during the past several years, has produced a tremendous amount of radioactive wastes which have saturated the waste storage facilities at the reactor sites. Because of present Japanese safety regulations, the wastes cannot be disposed of permanently underground nor on the ocean floor. Two methods of disposal (either into the ocean floor or underground) have been much discussed, but still more information is needed on the movements of radionuclides in the environments. This is definitely so for the movements of radionuclides underground, especially in underground water-flow systems. The lack of information includes data on and the mechanisms of the chemical and/or biological transformation of radioactive materials during the movements under various geochemical conditions.

Among the operating wastes from electricity generating nuclear stations, radioactive cobalt (^{60}Co) is a major radionuclide in quantity and as a radiological hazard. It has a relatively long half life (5.3 years) and emits two gamma rays with high energy (1.17 Mev and 1.33 Mev). Furthermore, cobalt is one of the essential elements to sustain life in animals and plants (for example, as Vitamin B_{12}) and hence is easily accumulated by various organisms, if available to them (1,2).

This paper summarizes the results of our previous investigation and reports some recent findings of ^{60}Co interactions in seawater-sediment systems, especially pertaining to the difference in the mobility of ^{60}Co under aerobic and anaerobic conditions. Furthermore, the difference in the mobility of ^{60}Co in seawater and in freshwater was investigated under the influence of various environmental factors. Details of the investigation will be reported elsewhere (3).

MATERIALS AND METHODS

A series of laboratory experiments was conducted simulating some of the underground environments at coastal areas. The interaction of ^{60}Co with various sediments in both freshwater and seawater systems was observed for a considerable period under either aerobic or anaerobic conditions. The degree of ^{60}Co mobility, after it had interacted with water and sediments, was determined by measuring radioactivity in the water phase and in the sediment phase.

The reversible (and irreversible) nature of ^{60}Co interaction between water and sediments was evaluated by a quick change of the surrounding environments. The ^{60}Co, for example, after having

interacted under anaerobic conditions for the first 30 days, was suddenly introduced into aerobic conditions for the next 30 days. On the other hand, the ^{60}Co, kept under an aerobic condition for the first 30 days, was moved into a condition of anaerobic for a further 30 days. To achieve the new anaerobic and aerobic conditions, nitrogen and oxygen gases were bubbled respectively into the systems.

To evaluate some of the physical and chemical characteristics of the highly mobile ^{60}Co produced in a seawater-sediment system, the ^{60}Co was separated from the rest of the system by means of a dialysis membrane (24 Å pore size), a chemical extraction (dithizone-benzene), and a sorption (various sediments). Details of the physical and chemical characteristic of the sediments used were reported elsewhere (4-6).

RESULTS AND DISCUSSION

The magnitude of ^{60}Co mobility under four different environmental conditions is illustrated in Fig. 1. The mobility in the system of the sand sediments and freshwater under aerobic condition was used as a base line for a comparison. The mobility of ^{60}Co increased at pH values between 6 and 8 in the following order: (1) in freshwater under aerobic conditions, (2) in seawater under aerobic conditions, (3) in freshwater under anaerobic conditions, and (4) in seawater under anaerobic conditions. It is seen that the ^{60}Co in the system of seawater under anaerobic conditions is by far the most mobile.

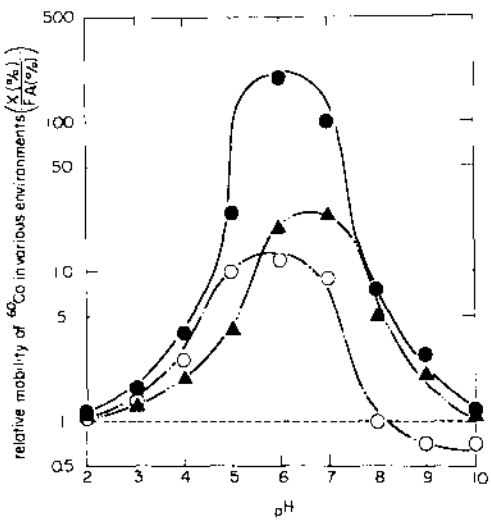

Fig. 1. Magnitude of ^{60}Co mobility in various environments: ▲---▲ X = freshwater under anaerobic, ●—● X = seawater under anaerobic, ○—○ X = seawater under aerobic, - - - - - FA = freshwater under aerobic.)

The irreversible nature of ^{60}Co derived from the changes in the redox environments was observed. In other words, the initial conditions such as aerobic or anaerobic conditions in the sediment and water systems where ^{60}Co was initially discharged seemed to determine the magnitude of ^{60}Co mobility. In Fig. 2, characteristic variation of ^{60}Co mobility was shown when incubating anaerobic conditions were suddenly converted into aerobic, and vice versa. The following results were observed in a series of these experiments; (i) if ^{60}Co initially had a high mobility under anaerobic conditions, it could have the high mobility even if surrounding environments were changed drastically, and (ii) if ^{60}Co initially had a low mobility under aerobic condtions, it would keep the low original mobility even if surrounding conditions were changed.

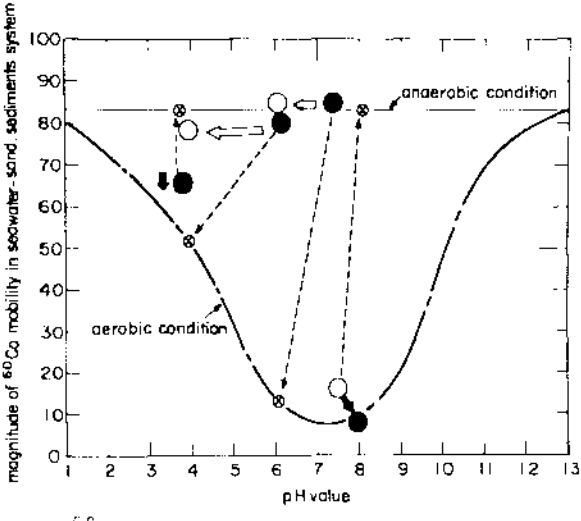

Fig. 2. Mobility of ^{60}Co, in the system of seawater and the sand sediments, with respect to changes in pH. (Incubating condition was changed suddenly from aerobic to anaerobic, or vice versa.) The following five symbols (○, ●, ⊗, ⇨, ➡) indicate respectively: the magnitude of ^{60}Co mobility under aerobic conditions; mobility under an anaerobic condition; mobility of reversible ^{60}Co expected by changes in the incubating condition; direction of change from anerobic to aerobic conditions; and direction of change from aerobic to anaerobic conditions.

A highly mobile form of ^{60}Co produced in the seawater system under anaerobic conditions could pass through a dialysis membrane freely (more than 99%). This result indicated that the mobile form of ^{60}Co consisted of a mixture of ionic cobalt, cobalt compounds with low molecular weight, and cobalt adsorbed (or absorbed) on the surface of very fine particulates whose diameter is less than 24 Å.

A comparison was made between ionic form of cobalt ($CoCl_2$) and the cobalt which was highly mobile and could pass through a dialysis membrane freely. The purpose of this comparison was to define some of

the characteristics of the highly mobile cobalt produced in the system of seawater under anaerobic conditions. Figure 3A shows a result of dithizone-benzene extraction for two forms of the cobalt at various pH values. The highly mobile cobalt was not extracted at all by the chelating agent with an organic solvent, though almost all of the ionic form was extracted at pH values between 4 and 8.

There is a clear difference between two forms of the cobalt in sorption to a fresh sand sediment. Again, the highly mobile cobalt was not reactive to the sediments, though the ionic form was sorbed nearly 100% by the sediments at pH values between 5.5 and 7, Fig. 3B. Unfortunately, the precise nature of the highly mobile cobalt has not yet been determined, but the result of these experiments suggest that it is stable and moves freely with the water in the flow systems of underground environments at the coastal areas.

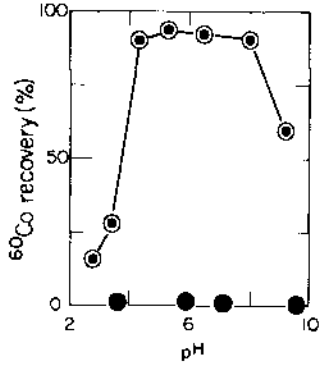

Fig. 3-A. Comparison between recovery of mobile ^{60}Co in seawater under anaerobic conditions by dithizone-benzene and recovery of ionic ^{60}Co.

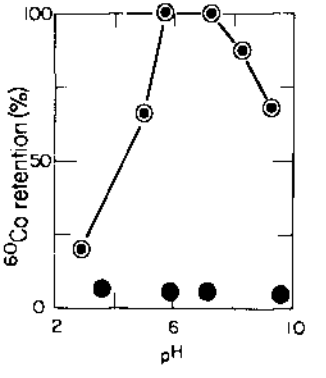

Fig. 3-B. Comparison between sorption of mobile ^{60}Co in seawater under anaerobic conditions on the sand sediments and sorption of ionic ^{60}Co.

(●; ^{60}Co in seawater under anaerobic, ◉; ionic ^{60}Co)

The authors thank the staff members of the National Research Council of Canada for their contribution to this work. Further acknowledgement is due to Dr. D.R. Champ, Chalk River Nuclear Laboratories for his valuable suggestions and to Dr. S. Senshu, Central Research Institute of Electric Power Industries, who made this visiting research possible for Y.M. N.R.C.C. 17873.

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RADIOECOLOGICAL STUDIES OF ACTIVATION PRODUCTS RELEASED FROM A NUCLEAR POWER PLANT INTO THE MARINE ENVIRONMENT

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In order to study the transport of radionuclides released under controlled conditions from the nuclear power industry it is convenient to use bioindicators which concentrate the substances since analytical procedures for seawater are very time-consuming. We have found the brown algae Fucus vesiculosus and F. serratus to be very suitable bioindicators for such studies (3,5). It is also of importance to study the activity concentration in algae because they are used in different kinds of food.

The purpose of this work was to study the distribution along the Swedish westcoast of radionuclides released from the Barsebäck nuclear power plant, to compare the activity concentration in Fucus with that of other algae and crustaceans in the area and to try to study the ^{60}Co-concentration in seawater and sediment profiles.

MATERIAL AND METHODS

Samples of Fucus vesiculosus, F. serratus, Ascophyllum nodosum and Cladophora glomerata, firmly rooted on the bottom, were collected from water depths between 0.5 and 1 m along the coast. Samples of the crustaceans Idothea, mainly I. baltica and I. viridis, as well as Gammarus found on the Fucus plants were sorted out for analysis. A limited number of sediment profiles were also taken. After drying in air at room temperature for 2-3 days the samples were ground and packed in plastic containers of 5 or 180 ml volume. The measurements were carried out using Ge(Li)-detectors of volume 46-100 cm^3 with counting times between 5 and 100 hours. The detector efficiences were determined carefully using samples of different densities containing accurately known activities of ^{152}Eu, ^{57}Co and ^{22}Na. After measurement the samples were dried at 105^0 C for 24 hours to get the reference weight.

Water samples with volumes between 80 and 170 liters were taken at the Fucus fields 2.9 km NNE of the power plant. ^{57}Co was added as a yield determinator and NaOH was then added to pH=12. The precipitate (~ 5 l) was the next day disolved in HCl and coprecipitated with 200 mg Fe as carrier. The pH was adjusted to 12 with ammonia. This precipitation was centrifuged and dried before measurement.

RESULTS AND DISCUSSION

Activity concentration in Fucus

The time variation of the concentration of activation products in Fucus well reflects the increase in the discharged activity during the overhaul periods (3). The local and distant spreads of radionuclides

from the power plant were investigated earlier at 8 stations along the Swedish westcoast (3,5) and showed a distance dependence of the activity concentration in Fucus which could be described by a power function of form:

$$C(z) = \alpha \cdot z^{-\beta}$$

where C(z) is the activity concentration at distance z and α and β are constants. The value of β was found to be around 1.4 for the radionuclides ^{60}Co, ^{58}Co, ^{65}Zn and ^{54}Mn, somewhat lower, 0.8 for ^{110m}Ag and still lower for ^{131}I (3,5). The results of our extended sampling now at 15 stations is given for ^{60}Co in Fig 1 and for ^{60}Co, ^{58}Co, ^{65}Zn, ^{54}Mn, ^{110m}Ag and ^{57}Co in Fig 2. No significant deviation from the above mentioned β-values is recorded. Some of the sampling stations recently used have been chosen in bays. The important parameter regarding the activity concentration in Fucus seems to be the shortest distance in water between the power plant and the sampling station, regardless of the costal configuration. Up to now all our results indicate a very efficient mixture of surface water in the area.

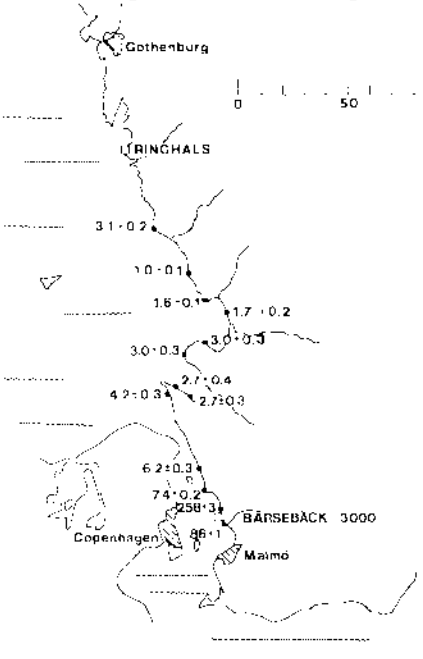

Figure 1.
Map over the Swedish westcoast showing the nuclear power plants BÄRSEBÄCK and RINGHALS and the sampling stations. The ^{60}Co activity concentration (Bq/kg dry weight) of Fucus in August 1979 is also given.

Co-60 in water and sediment

The ^{60}Co activity concentration in unfiltered seawater 2.9 km NNW of the power plant was determined to 5.4 mBq/l (1978-06-15), 4.1 and 5.1 mBq/l (1978-09-22) and 1.3 mBq/l (1979-04-11). Our results, indicate that the concentration factor for Fucus (dry weight) is as high as $(2\pm1)\cdot10^5$ corresponding to a wet weight concentration factor of $(4\pm2)\cdot10^4$. This value is considerably higher than those computed by Fukai and Murray (1) or measured in aquariums by Nakahara et al (4). It is interesting to note that the concentration factors for actinides in this area have also been found to be unusually high (2). Measurable concentrations of ^{60}Co have been found in sediments from sampling stations NW of the power plant 1.7-2.5 km off-shore. The total area content of ^{60}Co was found to be 190 Bq/m^2 at 4.1 km from the plant, 76 Bq/m^2 at 6.4 and 560 Bq/m^2 at 11 km. Around 70% of the

^{60}Co activity was found in the upper 25 mm of the sediment layer and the rest in the next 25 mm.

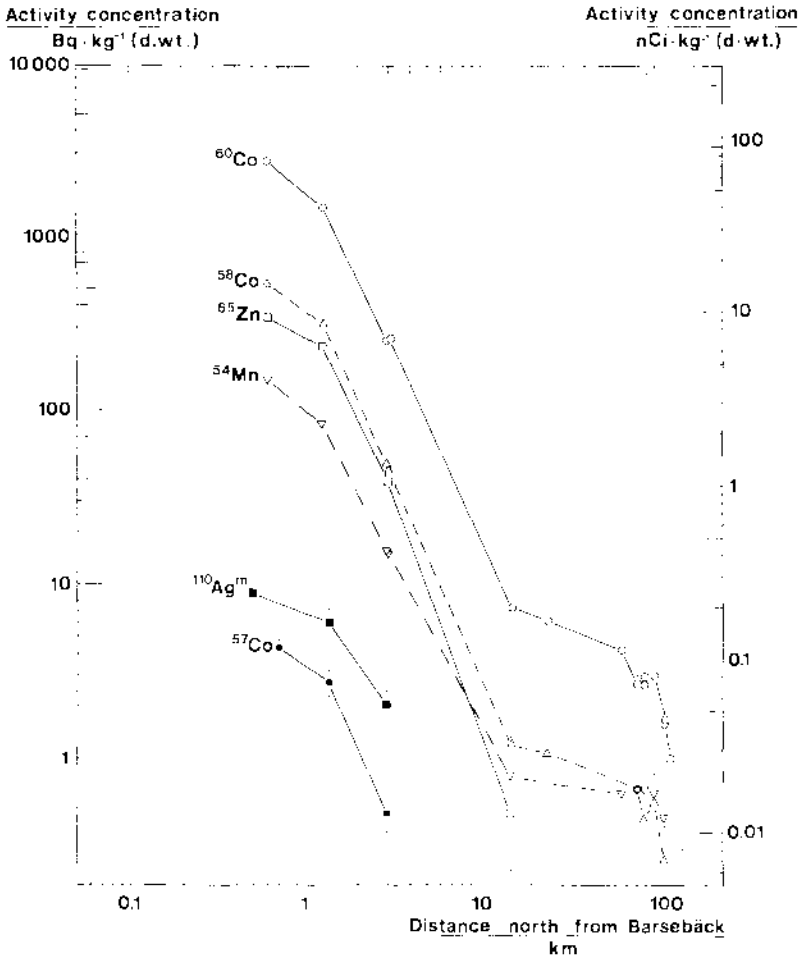

Figure 2. Variation in activity concentration in Fucus by distance northwards from BARSEBÄCK in August 1979.

Comparative studies of activity concentration in different algae and crustaceans.

The activity concentration in some algaes and crustaceans relative to that of Fucus is given in Fig 3. All concentrations are given on dry weight basis. It is interesting to note the differences in activity concentration in Idothea and Gammarus living side by side in Fucus plants. For the cobalt isotopes the concentration in Idothea is a factor of 4 higher than in Gammarus. This may indicate a difference in their feeding-habits. The highest activity concentration

of ^{54}Mn, $^{60,58,57}Co$, ^{65}Zn and ^{131}I are found in Fucus while crustaceans for these nuclides show an activity concentration of 4-40% compared to Fucus. For $^{110}Ag^{m}$, the activity concentration in Idothea and Gammarus is a factor of 9 and 6 respectively higher than in the Fucus plants in which they lives. This finding indicates an active uptake of silver in these crustaceans. Because Idothea and Gammarus are eaten by various fish species, these findings may be of considerable importance for the transport of radioactive substances to man.

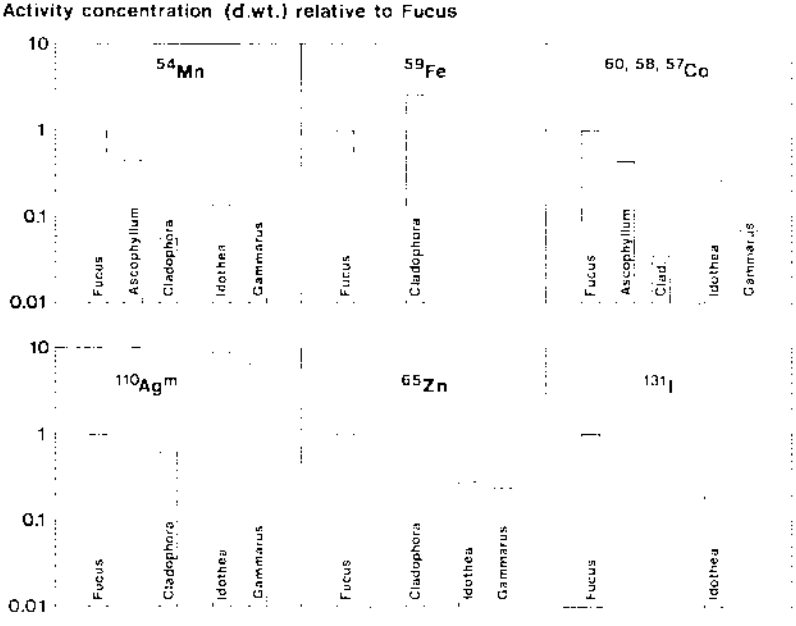

Figure 3. Relative content, per dry weight unit, in different algaes and crustaceans of radionuclides released from BARSEBÄCK in August-November 1979. Dry weight/wet weight ratios: Fucus-0.20, Ascophyllum-0.2, Cladophora-0.05, Idothea-0.19 and Gammarus-0.18.

ACKNOWLEDGEMENT

Thanks are due to Lena Carlsson and Lars-Eric Persson for valuable help with collection and identification of samples.

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MONITORING OF ENVIRONMENTAL RADON-222 IN SELECTED AREAS OF TAIWAN PROVINCE OF THE REPUBLIC OF CHINA

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Radon-222 and its daughters in air may enter human body by inhalation and cause radiation hazard to the epithelial basal cells of the respiratory tract. It may induce lung cancer (1). Therefore, measurement of ^{222}Rn and its daughters in air is an important program in environmental monitoring.

The experimental method used in this work is to measure ^{222}Rn activity concentration in selected areas of Taiwan by sampling air through a filter paper and detecting the gross α counts of the filter paper with an α scintillation counter. The experimental results obtained from sampling in a radioactive waste plant, a storage room for uranium, a natural cave, a room using natural gas, etc., are presented. Continuous sampling of environment at the Tsing-Hua open-pool reactor site for a period of six months is included. The annual dose equivalent of ^{222}Rn at the reactor site is estimated.

EXPERIMENTAL

The experimental setup were simply an air sampler and a ZnS(Ag) scintillation counter. The Whatman No.41 filters were used in air sampler. After a sampling time of 5 min, the filter was α counted with the ZnS(Ag) scintillation counter which was calibrated against an ^{241}Am α standard source. The ratios of ^{222}Rn concentration to its daughters' were assumed to be 10:9:6:4 for RaA, RaB, RaC, respectively (2, 3). A simplified equation can thus be obtained as below:

$$Q = \frac{R}{c\ \eta\ \varepsilon\ v} \qquad (1)$$

where

- Q is the concentration of ^{222}Rn in air (Bq/m^3)
- R is the counts from 2 to 7 min after the end of the sampling
- c is a coefficient determined by the decay constants of ^{222}Rn and daughters (0.975×10^3)
- η is the sampling efficiency ($\eta = 1$ in this experiment) (4)
- ε is the counting efficiency ($\varepsilon = 8.25\%$)

v = is the air sampling rate (0.85 m^3/min).

RESULTS AND DISCUSSION

The ^{222}Rn concentration in air of some selected areas in Taiwan are summarized in Table 1.

Table 1. Rn-222 concentrations in air of some selected area in Taiwan

Location	Concentration (Bq/m^3)	Sampling Date
Radioactive waste plant	3.15 ~ 13.99	1978.7.7
Uranium storage room	3.96 ~ 11.95	1978.9.21
A room 30 m from uranium storage room	3.55 ~ 10.73	1978.9.21
Room using natural gas	3.63 ~ 8.55	1978.10.18
Room not using natural gas	3.81 ~ 8.34	1978.10.18
Natural cave (inside)	13.83 ~ 16.69	1978.11.7
Natural cave (outside)	1.93	1978.11.7

The results of continuous sampling of environment at the Tsing-Hua open-pool reactor site for a period of six months are shown in Fig.1, and Fig.2 shows a typical daily variation in ^{222}Rn concentration.

According to the BEIR Report (5), the mean dose from ^{222}Rn to the epithelial basal cells is 0.01 Gy/WLM. As mentioned before, the concentration ratio of ^{222}Rn to its daughters is 10:9:6:4, then the ^{222}Rn concentration of 1 Bq/m^3 is equal to 1.5×10^{-4} WL. If the quality factor is assumed to be 20, and a multiplier factor of 1.5 is included because of the clean air, then the annual dose equivalent from 1 Bq/m^3 of ^{222}Rn in air to the epithelial basal cells could amount to 2.3 mSv/yr.

From the data shown in Fig.2, the annual dose equivalent of ^{222}Rn can be calculated, and it is equal to 29.6 mSv/yr (indoor) and 13.8 mSv/yr (outdoor).

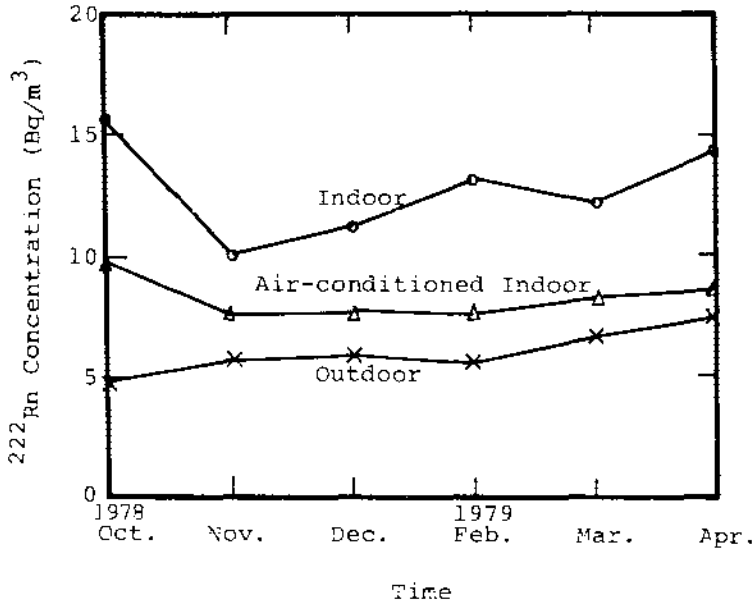

Figure 1

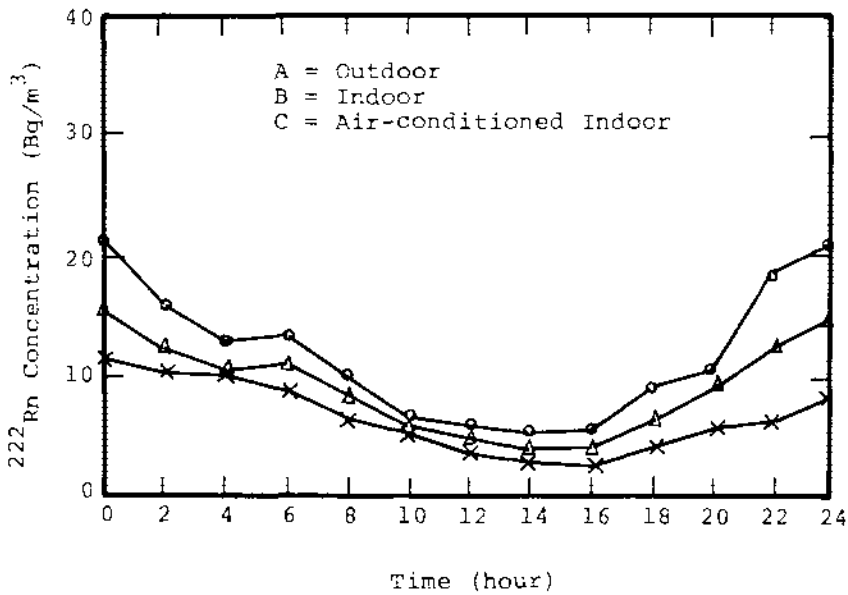

Figure 2

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DISTRIBUTION OF URANIUM, ^{226}Ra, ^{210}Pb AND ^{210}Po IN THE ECOLOGICAL CYCLE IN MOUNTAIN REGIONS OF CENTRAL YUGOSLAVIA

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INTRODUCTION

The distribution of the uranium decay series was investigated in the central parts of Yugoslavia. The characteristic of region are mountains 1000-1500 m high with unexploited and partially unexplored forests, extensive cattle breeding and cheese and milk production. The soil has neve been cultivated and the grass on the pastures grew without human interference.

RESULTS

Four nuclides of the uranium decay series were investigated: ^{210}Pb, ^{210}Po, ^{226}Ra and uranium, either for their longer $T_{1/2}$, or radiotoxicity. Samples of beef (meat and bones), milk, cheese, grass and soil were analyzed to round off a vital part of the ecological cycle.

TABLE 1. CONCENTRATION OF ^{210}Pb, ^{210}Po, ^{226}Ra AND URANIUM IN BEEF (MEAT AND BONES)

Location No	^{210}Pb (mBq/kg)		^{210}Po (mBq/kg	
	meat	bone	meat	bone
1.	18.50	3082,10	37.30	827.32
2.	99.53	777.30	21.46	555.00
3.	15.91	1914.49	12.35	1202.50
4.	227.92	869.50	46.25	1295.00
5.	124.69	1058.20	12.95	1047.10
6.	0	3903.50	18.50	8221.40
7.	134.16	670.44	190.92	1417.10
8.	45.88	3136.86	64.75	111.00

Location No	^{226}Ra(mBq/kg)		URANIUM(µg/kg)	
	meat	bone	meat	bone
1.	167.98	827.32	0.57	5.98
2.	4.43	235.59	0.60	0
3.	16.65	966.44	0.11	2.36
4.	79.92	994.56	0	16.75
5.	12.72	740.00	0.16	24.80
6.	109.15	1733.08	9.25	0.47
7.	122.10	1169.20	0	23.28
8.	79.55	617.90	0	9.22

The contamination of beef bones (Tb.1) with ^{210}Pb, ^{210}Po and ^{226}Ra was on the average 10 to 100 times higher than in meat. While bones were boiled for bouillon only up to 1,0% of ^{210}Pb and 13,8% of ^{210}Po passed over into the solution. The transfer of contamination thus becames negligible. The level of meat contamination with ^{210}Pb and ^{210}Po was of the same order of magnitude as found in literature (1). The ^{226}Ra results for meat were slightly higher (2).

^{226}Ra in dairy products (Tb.2) was higher than in USA where the average value (2) amounts to 10 mBq/kg.

Taking nito consideration a daily intake of soup, meat and dairy products in the whole region, the daily amount of ^{210}Pb, ^{210}Po and ^{226}Ra would only occasionally surpass the values obtained in other countries. The concentration of uranium in meat varies from 0 - 9,25 µg/kg. Supposing that all the beef on this location had the highest uranium concentration the uranium intake in a balanced diet, would be high if 2 ug/kg of total diet is taken as the average intake of uranium for unexposed population. The same applies for uranium milk samples (3).

TABLE 2. CONCENTRATION OF ^{210}Pb, ^{210}Po, ^{226}Ra AND URANIUM IN MILK AND CHEESE

Location No	^{210}Pb (mBq/kg)		^{210}Po (mBq/kg)	
	milk	cheese	milk	cheese
1.	5.32	283.79	18.50	99.90
2.	88.43	173.16	69.19	49.21
3.	18.13	2104.93	18.50	166.50
4.	89.17	0	26.27	55.50
5.	25.53	284.16	52.17	141.34
6.	52.91	227.18	9.25	81.40
7.	23.31	309.69	50.69	122.11
8.	50.32	173.16	9.25	77.70

Location No	^{226}Ra (mBq/kg)		URANIUM (µg/kg)	
	milk	cheese	milk	cheese
1. 1.	22.94	0	0.60	0
2.	15.17	0	0	0
3.	15.17	82.88	0	0.60
4.	51.06	104.71	0.36	0.87
5.	2.22	91.39	0	0.68
6.	28.12	0	0.62	0
7.	87.32	97.31	1.60	0
8.	37.74	55,50	0,67	0.34

TABLE 3. CONCENTRATION OF ^{210}Pb, ^{210}Po, ^{226}Ra AND URANIUM IN GRASS

Location No	^{210}Pb (mBq/kg)	^{210}Po (mBq/kg)	^{226}Ra (mBq/kg	URANIUM (µg/kg)
1.	841.75	555.00	132.83	0.77
2.	3784.36	240.50	480.63	11.88
3.	15932.22	0	721.13	0.97
4.	370.37	222.00	385.91	5.09
5.	11939.30	351.50	392.94	7.96
6.	3241.94	277.50	4329.00	5.46
7.	3592.70	462.50	1951.75	0.48
8.	14936.16	610.50	235.69	8.85

Higher concentrations of ^{210}Pb in soil are followed by higher concentrations in grass on 3 locations. The same is valid for ^{210}Po for locations 5,7 and 8. ^{226}Ra and uranium do not follow this pattern. The amount of ^{210}Pb, ^{210}Po and ^{226}Ra in soil coresponds to podzolic soils found in different parts of the world (1). The concentration of ^{210}Pb and ^{210}Po diminish proportionally to the soil depth. ^{226}Ra and uranium do not show any distinctive differences of concentration between soil layers.

TABLE 4. CONCENTRATION OF ^{210}Pb, ^{210}Po ^{226}Ra AND URANIUM IN PODZOLIC SOILS OF CENTRAL YUGOSLAVIA

Location No	210Pb (Bq/kg)	^{210}Po (Bq/kg)	^{226}Ra (Bq/kg)	URINIUM (ppm)
1.	32.92	43.14	43.73	0.50
2.	36.78	58.09	56.16	0.57
3.	153.18	75.63	61.31	3.64
4.	31.82	64.19	75.74	1.45
5.	122.56	127.28	34.41	1.17
6.	56.39	87.69	76.55	1.72
7.	87.84	108.67	40.55	0.91.
8.	108.78	277.13	55.13	1.46

soil layer: 0 - 5 cm

TABLE 4. (cont.)

Location No	^{210}Pb (Bq/kg)	^{210}Po (Bq/kg)	^{226}Ra (Bq/kg)	URANIUM (ppm)
1.	36.92	39.99	43.73	0.20
2.	36.04	35.48	71.30	0.92
3.	50.69	49.73	62.27	0.80
4.	26.31	52.06	106.56	1.17
5.	83.77	83.62	49.58	1.08
6.	53.42	43.66	71.00	1.64
7.	43.92	272.69	67.30	1.49
8.	272.69	348.54	84.36	0.95
soil layer: 5 - 10 Cm				
L 1.	23.30	34.70	26.89	1.78
2.	31.06	33.74	66.34	0.86
3.	43.58	35.89	16.52	0.92
4.	26.53	43.63	93.98	0.22
5.	78.55	72.52	65.86	1.10
6.	35.75	66.93	70.70	1.60
7.	34.95	76.48	65.86	1.64
8.	66.93	241.98	110.63	0.59
soil layer: 10 - 15 cm				

DISCUSSION

The distribution of uranium, ^{226}Ra, ^{210}Pb and ^{210}Po in an ecologically unpolluted environment is not different from regions in other parts of the world which are under cultivation.

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RA-226 COLLECTIVE DOSIMETRY FOR SURFACE WATERS IN THE URANIUM MINING REGION OF POÇOS DE CALDAS*

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A monitoring survey of the ^{226}Ra concentrations in river waters in the vicinity of the uranium mining, and future milling facilities, in the Poços de Caldas region started in January 1977. Results of this survey have been published elsewhere (1). Dosimetric and environmental models are used in the present work to calculate the Annual Collective Dose Equivalent (ACDE) to the populations potentially exposed. The ACDE for the whole-body, bone, gastro-intestinal tract - lower large intestine (GI tract-LLI), kidneys, and liver, via the pathways of drinking water and ingestion of food, grown in irrigated fields, are presented as a function of the ^{226}Ra concentrations in the surface waters of the Poços de Caldas region.

POPULATION DISTRIBUTION

Most of the population of the Poços de Caldas region are concentrated in cities located near the quasi-circular border of the plateau, which is represented by a dashed line in Figure 1. The central region of the plateau looks uninhabited at a permanent basis, except for the resident farm workers of the rural properties extant in the internal area of the plateau. The economical growth of the Poços de Caldas plateau, allied with the generalized use of motor vehicles in the region, made the local population much more mobile today than in 1970. Besides that, most cities of the plateau are touristic resorts with large seasonal fluctuations in the temporary population.

The Poços de Caldas plateau was divided in areas defined by concentric subregions having the common central point in the mining site of Campo do Cercado, as shown in Figure 1. The subregions are denoted by their radial border limits followed by the distance from the center to the outer ring. The populations, P_i, of the subregions i are shown in Table 1.

The approximate populational distribution of the Poços de Caldas plateau given in Table 1 is based upon the results of the 1970 census, which is the last census available. The city of Poços de Caldas had about 5.8×10^4 inhabitants in 1970, over 95% living within the city limits. The 1980 brazilian census has just started, but new populational data will not be available before two years from now. However, preliminary data collected, unofficially, indicated that the population of the municipality of Poços de Caldas might have exceeded 1.0×10^5 inhabitants sometime along the last ten years. Today the population distribution in the plateau is such that 90% of the entire population

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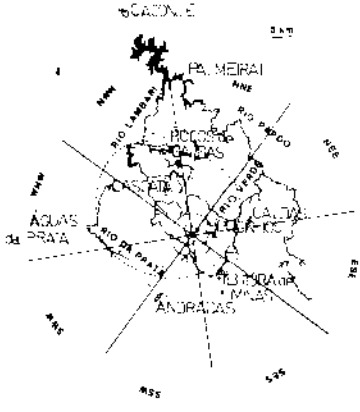

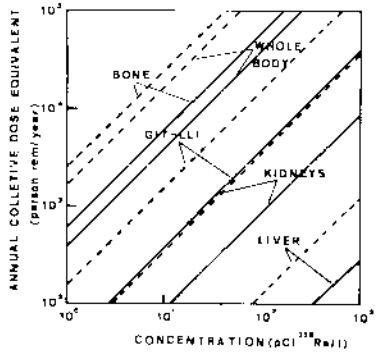

Figure 1. Hydrographic chart and subregions of the P.C. plateau.

Figure 2. ACDEs for the populations of the Poços de Caldas plateau (drinking water) and subregions NWN20 and NWN25 (food ingestion) as a function of ^{226}Ra concentrations in waters.

reside in urban areas, while 10% can be found in rural areas. As a consequence of the facts mentioned above, the populational data shown in Table 1 do not correspond to the actual population of the region at any given time today, but shall only be interpreted as an indication of the percentual distribution that one can expect to find in the region, if the seasonal variations and other fluctuations in the population of the Poços de Caldas plateau are discounted. Collective dosimetry for ^{226}Ra concentrations in the surface waters of the region were undertaken with full awareness of these and other shortcomings.

DOSIMETRIC MODEL

The ACDE, $\dot{S}_j$, for the population of the Poços de Caldas plateau was calculated based upon the formulae recommended by the International Commission on Radiological Protection (ICRP). In particular, the ACDE to an organ j, in person-rem/year, was calculated from the per caput annual dose equivalent, $\dot{H}_j$, in rem/year, for an organ j of a generic individual of a population, assuming that P_i persons in the subgroup i of the total population were submitted to the same dose equivalent (2). However, to evaluate the annual absorbed dose, $\dot{R}_j$, in mrad/year, to an organ j, the dose factors, $\dot{D}_j$, were derived from the exponential models recommended by ICRP (3). The dosimetric data of Tables 1 and/or Figure 2 reflect the following simplifying assumptions: (i) Urban and rural populations are considered together for the calculational purpose of regarding a population, P_i, characteristic of a particular subregion i; (ii) Temporal and spatial variations in the populations P_i of each subregion i were not taken into account; (iii) The maximum ^{226}Ra concentrations as listed in Table 1 for each subregion were considered as indicators for the constant values adopted to allow estimates of the annual absorbed doses, $\dot{R}_j$, to an organ j of a generic individual of

TABLE 1. Populational distribution, ^{226}Ra concentration in water, and ACDEs for the Poços de Caldas plateau

Subregion	City or Village	Population * P_i ($\times 10^3$)	%	Maximum ^{226}Ra concentration pCi/l		Collective annual dose equivalent (person rem/yr)			
					Whole body	Bone	Liver	Kidneys	GI-tract-LLI
NNW 20† NNW 25	Poços de Caldas	58.0	57.5	0.6	1.3×10^{2}	1.9×10^{2}	9.5×10^{-2}	2.7	1.2×10
NNW 25	Águas da Prata e Cascata	4.1	4.1	29	4.3×10^{2}	6.6×10^{2}	3.7×10^{-1}	9.1	4.0×10
SSW 15 SSW 20	Andradas	17.4	17.2	≤ 0.2	≤ 6.4	≤ 9.7	≤ 4.7×10^{-3}	≤ 1.4×10^{-1}	≤ 5.9×10^{-1}
ESE 15	Ibitiúra de Minas	2.7	2.7	≤ 0.2	≤ 2.2	≤ 3.4	≤ 1.6×10^{-3}	≤ 4.7×10^{-2}	≤ 2.0×10^{-1}
ESE 20	Santa Rita de Caldas	7.0	6.9	≤ 0.2	≤ 1.3×10	≤ 1.9×10	≤ 9.4×10^{-3}	≤ 2.7×10^{-1}	≤ 1.2
NNE 10 NNE 15	Caldas e Pocinhos	8.7	8.6	≤ 0.2	≤ 2.0	≤ 3.0	≤ 1.5×10^{-3}	≤ 4.2×10^{-2}	≤ 1.8×10^{-1}
NNE 20	Santana da Caldas	3.0	3.0	≤ 0.2	≤ 5.2	≤ 7.8	≤ 3.8×10^{-3}	≤ 1.1×10^{-1}	≤ 4.8×10^{-1}
				TOTAL	≤ 5.9×10^{2}	≤ 8.5×10^{2}	≤ 4.5×10^{-1}	≤ 1.3×10	≤ 5.3×10

* [illegible]

† [illegible]

the population P_i; (iv) The ^{226}Ra concentrations in food products grown in irrigated fields were estimated based upon local data, from the 1970 census, on agricultural production, and site specific environmental models; (v) The usage factors for drinking water and food consumption, used in the calculations, were essentially those of the Reference Man (4) for Latin American inhabitants. This assumption is made based upon the fact that there are wide variations in usage factors for regions of fast changing parameters and non-uniform distribution of wealth, so the Reference Man values can be considered suitable for the present dosimetric calculations, because of the uncertainties associated to alternative values; (vi) The ACDEs were calculated by assuming no ^{222}Rn loss by gaseous diffusion after ^{226}Ra uptake by the human body.

The above simplifying assumptions are not in complete agreement with the recommendations of ICRP26 (2). However, one should bear in mind when applying the ICRP26 recommendations to a particular region of a developing country, that the demographic distribution as well as food habits may experience significant fluctuations under certain conditions like, for example, fast population growth and local economical development. As a consequence, care must be exercised when using the estimates of collective dose equivalent, from data of a fast developing region, as the basis for decision-making processes as suggested by the ICRP26 (2), since dramatic changes may occur in the calculational parameters within relative short periods of time. Accordingly, ACDEs values of Table 1, like 590 person-rem/year for the population of the Poços de Caldas plateau, are not to be used in decision-making processes.

Figure 2 shows graphs of the annual collective dose equivalents to the whole-body and selected organs as a function of the ^{226}Ra concentrations in water via the pathway of drinking water for the population of the Poços de Caldas plateau, and via the pathway of food ingestion for the populations of the subregions NWN20 and NWN25, by far the most populated subregions. These estimates were made as a contribution for future comparisons, although other parameters used in the dosimetric calculations may also change considerably along the future.

CONCLUDING REMARKS

Paragraph 22 of ICRP26 (2) is very carefully worded to warn about the complexity of the relationship between the distribution of dose equivalent in an exposed population and the assessment of detriment. Furthermore the careful wording of ICRP26, can be found notably also in paragraphs 219, 221 and 232 which are bound to prevent misuses of the concept of collective dose equivalent to make dubious quantitative appraisals of detriment associated with practices, however decisions are likely to be made based upon such appraisals.

The present work intends to be a contribution to the understanding of the shortcomings involved in calculating collective dose equivalent due to the potential enhancement of ^{226}Ra concentrations in the surface waters near the uranium mining region of Poços de Caldas.

General observations and tentative conclusions are listed as follows:

1. The ACDEs for selected populational subgroups as well as for the total population living in the Poços de Caldas plateau have been calculated based upon simplifying assumptions. These ACDEs, rather than the dose-equivalent commitments or the collective dose-equivalent commitments for the predicted time of operation of the uranium mine of the Poços de Caldas region were calculated, because there were several intrinsic uncertainties in the parameters available for the calculations.
2. The fluctuations expected to occur in the data on populational distribution, irrigational practices, agricultural production, and food consumption in developing regions, like the Poços de Caldas plateau, makes the quantitative assessment of the collective dose-equivalent commitments meaningless, unless reliable long range predictions can be made on the varying parameters to enable time integration.
3. Linear models can be used to estimate the ACDE as a function of the ^{226}Ra concentration, based on parameters which may be valid at a particular time, but the actual collective dose equivalent commitment is difficult to predict.
4. Dosimetric models based upon site specific environmental models are helpful to estimate collective dose equivalents to populations from a particular practice, but extreme care should be exercised by competent national authorities when using such estimates in decision-making processes.
5. Paragraphs 22, 219, 221 and 232 of ICRP26 (2) should be taken into full account when estimating collective dose equivalents.

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THE RADIUM CONTAMINATION IN THE SOUTHERN BLACK FOREST

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The high natural radium contamination prevailing in the Southern Black Forest was used to assess the degree of contamination of the environment, the mechanisms of radium transport to man, as well as the radiation impact on the population from natural Ra-226.

The Ra-226 concentration in the environmental air amounted to values between 0.16 and 1.2 fCi/m³. The mean value was 0.5 ± 0.2 fCi/m³. Considering the Ra-226 concentration in the soil, this value corresponds to about 150 µg of dust/m³.

The Ra-226 concentration in the drinking water was measured between March and December 1978. The mean value is 0.31 ± 0.03 pCi/l. The Ra-226 contamination found in the Krunkelbach, was likewise measured continuously during the same period. The mean value of the measured concentration was 0.5 ± 0.3 pCi/l. 23 water samples were taken from the Feldberger Alb brook, the most important tributary of the Krunkelbach brook; the average concentration was 0.19 ± 0.02 pCi of Ra-226/l.

In the Krunkelbach valley the Ra-226 contents were determined of 22 samples taken from several brooks. The concentrations found were < 0.03 to 2.6 pCi/l. In 23 water samples collected in the immediate and more distant vicinity of that valley the Ra-226 contents were determined. The results ranged from 0.03 to 1550 pCi/l. Generally speaking it can be stated that Ra-226 concentrations above 1 pCi are very rare, that the Ra-226 concentrations in a brook or river are maximum at its spring, and that high concentrations are observed when on account of extended dry periods the water stays longer in the soil or when in the winter season the salt concentration in the residual liquid increases by freezing of most of the water.

More than 100 soil samples were examined for Ra-226. In the topmost 20 cm soil layer the mean value was 2.9 pCi/g.

A great number of sediment samples from several brooks and rivers were measured to determine their Ra-226 content. The mean value found in the valley of the Menzenschwander Alb was 3.2 ± 1.0 pCi/g. In other sediments a mean value of 1.1 ± 0.1 pCi Ra-226/g was measured. The Ra-226 concentrations in trouts were measured. The average concentration was 50 pCi Ra-226/kg of fresh weight. The maximum value was 211 pCi Ra-226/kg.

Milk samples were taken regularly from two farms and analyzed for their Ra-226 content. The results measured for one farm are plotted in Fig. 1. The Ra-226 concentrations scatter by more than one order of magnitude and take an average value of 11 pCi Ra-226/l. For the second farm a mean value of 7 pCi/l was measured. Random samples collected at other farms confirmed these results.

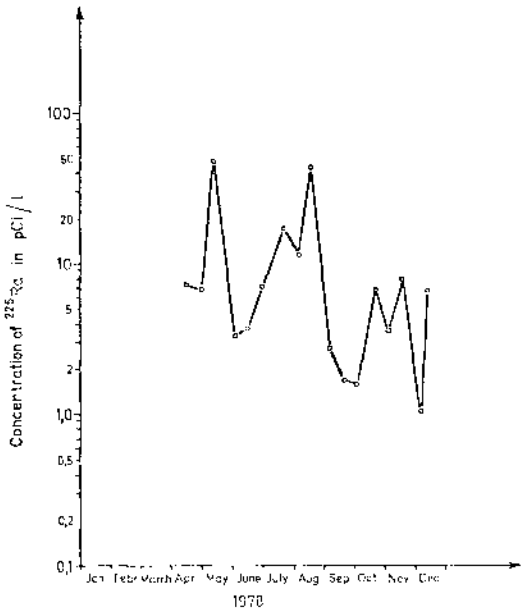

Fig 1: ^{226}Ra concentration in the milk from farm A, Menzenschwand

Foodstuffs produced in the Southern Black Forest were measured for their Ra-226 content. Besides the milk and fish samples already mentioned, wheat, barley, oat, potatoes, salad, cabbage, beans, kohlrabis, blueberries, beef, entrails, venison, eggs, mushrooms and beer were investigated. A summary of values taken from the literature, compared with the radium contents found in the Southern Black Forest, is given in Table 1. The Ra-226 contents of corn, potatoes and milk are surprisingly high as compared with the published data. A more accurate examination is presently performed in order to find out which part of the corn has been contaminated with Ra-226.

Since the Ra-226 transfer from the soil to the grass is very significant for the contamination of milk, a great number of grass and hay samples were examined for Ra-226. Moreover, in search of a bioindicator for Ra-226, quite a number of wild plants were measured to determine their Ra-226 content. The average concentration of the grass and hay samples found was 0.30 ± 0.04 pCi/g of dry substance. For the other wild plants values were found of 0.7 to 22 pCi Ra-226/g of dry substance. A clear bioindicator for Ra-226 was not identified.

Since water and fish samples, grass and milk samples, soil and grass samples as well as soil and plant samples were always taken jointly, it was possible to calculate many transfer factors. The following transfer factors were determined: The transfer factor for fish / water was 28 ± 12. In the guts, on the one part, and in the meat and heads + bones, on the other part, 1/3 each of Ra-226 was contained. The ratio of water to sediment concentration was calculated for a settling tank to be 11 x 10^{-3} and for the brooks and rivers

26×10^{-5}. The grass to milk transfer factor indicates the percentage of Ra-226 contained in one liter of milk consumed daily. A value of 0.3 ± 0.1 % per liter was calculated from the measured grass and milk concentrations. The milk to grass transfer factor is 0.20 ± 0.05. The transfer factor calculated for grass to soil was 0.027 ± 0.005. Slightly higher values were determined for the wild plants. For the milk to soil transfer factor the value 0.005 ± 0.002 can be calculated from the data indicated. The transfer factors calculated for the individual foodstuffs vary from 0.003 for the white of egg up to 0.2 for corn.

Sampling Material	^{226}Ra concentration in pCi/kg	
	Menzenschwand	[1]
Soil:	1200 - 1500	150 - 3100
Water:		
river and lake water	0.03 - 2.5	0.002 - 62
ground and spring water	0.1 - 1549	0.001 - 237800
dringking water	0.11 - 0.57	0.005 - 50
Foodstuffs:		
potatoes	30 - 40	0.8 - 2.8
corn, flour	20 - 240	1.9 - 2.8
meat	2	0.01 - 1.1
milk	0.3 - 48	0.3
vegetables	5 - 170	0.5 - 3.8
fish	1.4 - 211	5.1
eggs	80	3.1 - 6.1
entrails	10 - 200	0.1

Table 1: ^{226}Ra concentration in environmental samples taken in and around Menzenschwand, as compared with values taken from the literature [1].

Taking into account the average habits of consumption in Germany the maximum possible intake per annum of radium was calculated for the population living in the region under investigation. It amounts to 7.1 nCi/a if one assumes that the total demand for foodstuffs is satisfied by local produces. This value is higher by the factor 12 than the annual Ra-226 ingestion value of 580 pCi permitted by the German Radiation Protection Ordinance. The body burden corresponding to this annual intake should amount to 7.4 nCi of Ra-226. To verify this conclusion, 28 members of the local population were measured in a body counter to determine the amount of Ra-226 incorporated. All 28 measuring values were below the detection limit

of about 7.5 nCi of Ra-226. If one takes these 28 measuring values as part of a random sample, the mean value and its error can be calculated for the Ra-226 body burden received by the whole group from the measured values, taking into account the standard deviations according to the counting statistics. The value so determined was 0.3 ± 0.7 nCi of Ra-226. A comparison of the most probable value of 0.3 nCi of Ra-226 with the calculated maximum value of 7.4 nCi of Ra-226 for the body burden yields that either only 4-5 % of the foodstuffs consumed by the group examined can stem from local production or that the radium transport from the gastro-intestinal tract into the blood according to the ICRP model [2] is overestimated.

To assess exactly the body burden of the inhabitants of the Southern Black Forest, the Ra-226 contained in the teeth of people living there is presently examined.

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EVALUATION OF SMALL SCALE LABORATORY AND POT EXPERIMENTS TO DETERMINE REALISTIC TRANSFER FACTORS FOR THE RADIONUCLIDES ^{90}Sr, ^{137}Cs, ^{60}Co and ^{54}Mn

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Much of the information on uptake of radionuclides from soils for determination of transfer factors have been obtained from laboratory experiments with prepared soils or soils contaminated by nuclear weapon tests. These results may not be valid for estimation of transfer factors in the field. On the other hand, it is nearly impossible to conduct field experiments to determine the transfer of radionuclides for every site and condition. We have started lysimeter experiments in a controlled experimental field to study the root uptake of ^{90}Sr, ^{137}Cs, ^{60}Co, and ^{54}Mn under outdoor conditions. (Figure 1).

Since this type of experiments are time-consuming and expensive, and can only be conducted at locations which permit the use of radionuclides under outdoor conditions, we have also set up experiments parallel to those in the outdoor lysimeter using the Kick-Brauckmann experimental pots (3) under greenhouse and using the Neubauer cups under growth chamber conditions. The results obtained from the three types of experiments are compared.

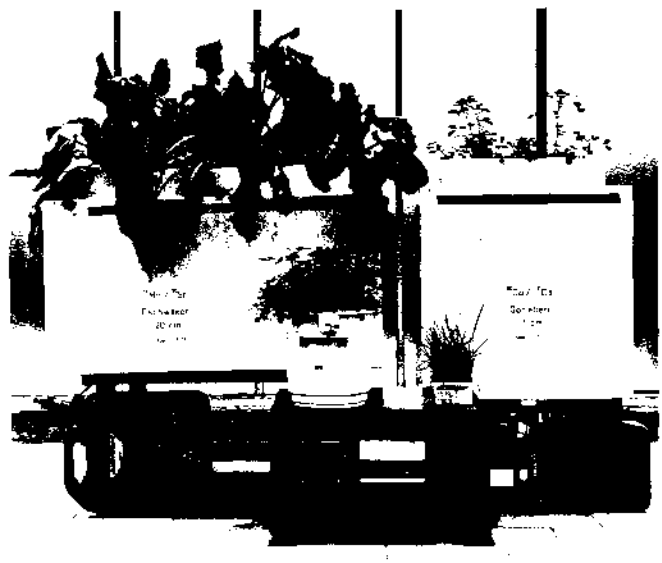

Figure 1. Test units used: 1 m^2 lysimeter filled with topsoil (Ap-horizon), 0,25 m^2 lysimeter with undisturbed soil profile, Kick-Brauckmann pot (8 kg topsoil), and Neubauer cup (400 g of topsoil).

MATERIALS AND METHODS

Two soils were used: a podzolic soil (spodosol) and a degraded loess soil (alfisol). Their properties and details about the lysimeter experiment were presented by Steffens et al. (4). For the pot and cup experiments the soils were sieved (≤ 5 mm and < 2 mm, respectively), fertilized (120 mg N, 120 mg P_2O_5, 200 mg K_2O/kg soil), treated with 10 µCi $^{90}Sr(NO_3)_2$ or $^{54}MnCl_2$ or 5 µCi $^{137}CsCl$ or $^{60}CoCl_2$/kg soil, and thoroughly mixed. The pots in 4 replicates containing 8 kg soil each were watered to 65 % of the water holding capacity of the soil and placed in greenhouse under natural light conditions. The cups in 4 replicates and containing 400 g soil each were placed in the growth chamber maintained at 12-hour day/night photoperiod at 23 °/16 °C and 65/85 % relative humidity. The soils in the lysimeters were equilibrated for 8 months and in the pots and cups for 2 weeks before the initiation of the experiments. Plants were grown to ripeness in the lysimeters and pots, but were harvested from the cups 4 weeks after sowing or planting. The plants were dried at 105 °C. ^{90}Sr was measured according to Cerenkov via the decay product ^{90}Y, ^{137}Cs, ^{60}Co, and ^{54}Mn on a surface Ge(Li) detector. Additional details are listed by Führ (1). The transfer factors are expressed as the quotient

$$TF_{Sp} = \frac{\text{dpm/g plant fresh weight}}{\text{dpm/g dry soil (application)}}$$

The transfer factors from the pot and cup experiments were expressed in relative values based on those from the lysimeter experiments equal to 1.

RESULTS

The results (Table 1) demonstrate that the transfer factors for ^{90}Sr, ^{137}Cs, ^{60}Co, and ^{54}Mn obtained under the specific conditions of the small scale Neubauer cup experiment differed greatly from those obtained from the outdoor lysimeter experiment. These differences may be mainly due to the short equilibration period, daily water addition (repeated desorption), plant density, short vegetation period, and higher transpiration rates in the cup experiment.

In the pot experiment the transfer factors for ^{90}Sr, ^{137}Cs, and ^{54}Mn showed less deviation especially in crops grown on podzolic soil. On the average they are 1.5 to 2 times higher than the lysimeter values, with a tendency to higher deviation in the root crops. In the case of ^{60}Co, transfer factors were found to be similar in potatoe leaves and tuber on podzolic soil, 2 to 3 times higher in sugar beets on loess soil, but up to 23 times higher in barley and salad. Yields and to a smaller extent some of the nutrient contents were generally higher in plants grown in pots than those from field trials (2), probably due to higher ferti-

Table 1: Relative transfer factors of pot and cup experiments compared with transfer factors from outdoor lysimeter experiments.

Plants		Transfer factors[1]	Relative transfer factors[2] pot	cup
Sr-90: podzolic soil				
Barley	straw	2,08	1,6	0,5
	grain	0,17	1,5	-
Potatoes	leaf	0,79	1,3	-
	tuber	0,014	3,7	-
Salad		0,46	2,1	1,7
Sr-90: loess soil				
Barley	straw	1,91	1,2	0,25
	grain	0,09	1,8	-
Sugar beets	leaf	0,21	2,9	3,5
	beet	0,29	2,2	-
Salad		0,34	1,4	3,5
Cs-137: podzolic soil				
Barley	straw	0,081	2,0	5,9
	grain	0,039	1,4	-
Potatoes	leaf	0,057	1,9	-
	tuber	0,046	2,1	-
Salad		0,018	2,2	16,1
Cs-137: loess soil				
Barley	leaf	0,0051	-	5,9
	grain	0,0023	-	-
Sugar beet	leaf	0,0087	5,2	7,9
	beet	0,0037	1,7	-
Salad		0,0036	0,8	0,8
Co-60: podzolic soil				
Barley	straw	0,018	17,8	6,8
	grain	0,013	23,1	-
Potatoes	leaf	0,32	1,0	-
	tuber	0,082	0,9	-
Salad		0,0074	15,0	13,5
Co-60: loess soil				
Barley	straw	0,0088	7,2	0,5
	grain	0,0059	8,1	-
Sugar beet	leaf	0,012	3,1	3,0
	beet	0,0099	2,1	-
Salad		0,0037	6,2	1,7

1) Obtained from lysimeter experiments

2) Based on lysimeter data equal to 1

Continued to next page

Plants		Transfer factors[1]	Relative transfer factors[2] pot	cup
Mn-54: podzolic soil				
Barley	straw	3,40	2,1	0,2
	grain	1,00	1,5	-
Potatoes	leaf	2,00	1,5	-
	tuber	0,14	1,0	-
Salad		0,48	4,2	2,1
Mn-54: loess soil				
Barley	straw	1,30	1,5	0,1
	grain	0,31	1,0	-
Sugar beet	leaf	0,15	8,7	1,9
	beet	0,08	3,9	-
Salad		0,28	2,4	0,8

[1] Obtained from lysimeter experiments

[2] Based on lysimeter data equal to 1

lizer application rates, less competition for light and water, higher root mass per soil unit, and hence higher interception. Therefore transfer factors obtained in pot experiments can only be applicable to a limited extent to field conditions. Factors dominant in influencing the transfer factors in pot experiments may include soil volume, root density and root/shoot ratio, water supply, and fertilizer application rate.

In order to make use of the advantages of pot experiments outlined earlier, these experiments will be continued for two more vegetation periods to reduce the magnitude of variation in transfer factors for some plants on a given soil. The following plants will also be included: wheat, alfalfa, grass, bush beans, carrots, and radish.

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THE TRANSFER OF SR-90, CS-137, CO-60 AND MN-54 FROM SOILS TO PLANTS -RESULTS FORM LYSIMETER EXPERIMENTS.

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Predicting irradiation of man from food intake using computer simulation models (1) the magnitude of transfer of radionuclides from soil to plant known as transfer factor is of importance. This transfer factor is influenced by a number of environmental parameters such as climate, plant species, soil properties, concentration of the stable and radioactive isotope in the soil etc. (2). Therefore,results from laboratory studies or from experiments using radioactively contaminated soils of non-agricultural origin may not be validly applicable from one region to another.

In our investigations outdoor lysimeters containing 2 typical German soils were used to evaluate the transfer of Sr-90, Cs-137, Co-60 and Mn-54 from the soil to a variety of crop plants.

MATERIALS AND METHODS

Soils: A parabrown earth form the Eschweiler region and a podzol (sandy soil) from the Gorleben area being very different in their properties (Tab. 1).

Table 1: Chemical and physical properties of the soils.

Origin	Eschweiler	Gorleben
Soil type	Parabrown earth	Podzol
Horizon	Ap	Ap
pH ($CaCl_2$)	5,9	4,7
Org. C %	1,4	1,1
Total N %	0,1	0,1
Clay %	12,0	2,6
Silt %	28,4	2,9
Fine sand %	58,3	34,3
Coarse sand %	1,4	60,2
T-value	11,2	6,2
S-value	12,8	2,4
Ca (meq/100g)	11,8	2,0
K (meq/100g)	0,8	0,2

Lysimeter:
1) 0,25 m^2 surface undisturbed soil profile
2) 1 m^2 surface, uniformly mixed surface soil(3)

Treatment:
Carrier-free Cs-137 Co-60, Mn-54 in chloride and Sr-90 in nitrate form mixed with
a) 0 - 1 cm soil layer to simulate an accidental contamination, 77 and 80 uCi/kg dry soil
b) 0 - 20 cm soil layer to simulate a 50-years contamination, 5,8 and 6,0 uCi/kg dry soil.

Plants: Grass and alfalfa pasture, sugar beets, winter wheat and summer barley grown in a rotation with

bush beans, carrots, radish, lettuce and clover as intermediate crops. The fertilization corresponded to the amounts usual in practice.

Analysis: Radioactivity measuring was done in a well type Ge(Li)-detector and in a liquid scintillation spectrometer (Sr-90).

RESULTS AND DISCUSSION

The transfer factors for Sr-90 determined in pasture grass, grass hay, and alfalfa grown on the podzol soil contaminated in the 0 - 1 cm layer were similar or up to 3 times higher than those for the parabrown earth soil, but the transfer factors for Cs-137 were 3 - 73 times higher on podzol soil (Table 2). At least in pasture grass, the transfer of both radionuclides was lower than that published previously (3) due to longer time of equilibration of Sr and Cs in the soil. The tendency of the transfer factors in the consecutive cuts was contrary to the expected decrease of transfer.

Table 2: Transfer factors (TFSP) of Sr-90 and Cs-137 for plants grown on parabrown earth and podzol soils after contamination of the 0 - 1 cm soil layer.

Plants		Dry matter %	TFSP x 10^{-1} Sr-90 Parabr. earth	Podzol	Dry matter %	TFSP x 10^{-1} Cs-137 Parabr. earth	Podzol
Grass	1. cut	11,3	4,3	4,0	12,8	0,05	0,60
	2. cut	12,0	2,6	2,9	11,0	0,06	0,53
	3. cut	26,8	7,0	7,7	21,3	0,08	0,51
	4. cut	16,2	4,6	10,7	29,8	0,12	0,99
Grass hay	1. cut	21,0	3,7	5,7	16,5	0,01	0,73
	2. cut	24,3	7,5	12,5	24,4	0,03	0,75
Alfalfa	1. cut	26,2	4,1	13,0	20,4	0,02	0,54
	2. cut	29,7	30,0	41,0	26,0	0,10	0,31

Soil sampling 0 - 10 cm deep, time intervals between the cuts: grass 28 days, grass hay and alfalfa 56 days.

Similary, for winter wheat, summer barley, radish and lettuce grown on soil contaminated in the 0 - 20 cm layer (Table 3 and 4), the transfer factors for Sr-90 were only up to 2 times higher, but for Cs-137 5-59 times, for Co-60 2-27 times, and for Mn-54 1,5-17 times higher for the podzol soil than for the parabrown earth soil. The differences between the 2 soils might be due essentially to the lower sorption capacity and base saturation in the podzol soil.

In all plants and on both soils the transfer factors for Sr-90 and Mn-54 were higher by an order of 1-3 than those for Co-60 and Cs-137 (Table 2-5).

Table 3: Transfer factors (TFSP) of Sr-90 and Mn-54 for plants grown on parabrown earth and podzol soils after contamination of th 0 - 20 cm soil layer.

Plants		Parabrown earth			Podzol		
		D.M. %	TFSP Sr-90	TFSP Mn-54	D.M. %	TFSP Sr-90	TFSP Mn-54
Winter wheat	straw	95,0	1,35	0,53	96,0	2,77	9,16
	grain	91,0	0,05	0,35	90,0	0,14	3,41
Summer barley	straw	92,0	1,91	1,33	92,0	2,09	3,40
	grain	95,0	0,09	0,31	95,0	0,17	1,00
Radish	leaf	8,5	1,20	0,20	8,5	1,77	0,46
	beet	5,8	0,13	0,02	5,7	0,10	0,03
Lettuce		8,6	0,34	0,28	8,6	0,46	0,48

Table 4: Transfer factors (TFSP) of Cs-137 and Co-60 for plants grown on parabrown earth and podzol soils after contamination of the 0 - 20 cm soil layer.

Plants		Parabrown earth			Podzol		
		D.M. %	TFSP x 10^{-2} Cs-137	Co-60	D.M. %	TFSP x 10^{-2} Cs-137	Co-60
Winter wheat	straw	95,0	0,46	1,00	96,0	11,1	26,1
	grain	91,0	0,10	0,74	90,0	5,9	19,8
Summer barley	straw	92,0	0,51	0,88	92,0	8,1	1,8
	grain	95,0	0,23	0,59	95,0	3,9	1,8
Radish	leaf	7,3	0,37	0,56	9,0	2,6	1,4
	beet	5,3	0,04	0,25	5,3	0,4	0,5
Lettuce		7,7	0,36	0,37	5,7	1,8	0,7

Table 5: Transfer factors (TFSP) of Sr-90, Mn-54, Cs-137, and Co-60 for plants grown on parabrown earth and podzol soils after contamination of the 0 - 20 cm soil layer.

Plants		D.M. %	TFSP x 10^{-1} Sr-90	Mn-54	D.M. %	TFSP x 10^{-1} Cs-137	Co-60
Sugar beet	leaf	13,7	2,06	1,49	16,1	0,09	0,12
	beet	26,8	2,62	0,79	26,8	0,04	0,10
Potatoes	leaf	9,2	7,85	19,66	10,7	0,57	3,23
tuber	peel	21,4	0,67	1,32	22,0	0,64	1,05
	flesh	21,2	0,14	1,37	21,7	0,47	0,82

Sugar beets on parabrown earth, potatoes on podzol soil

Because of the higher dry matter content in grain crops, higher transfer factors for all nuclides were found in dicotyledons than in monocotyledons, Lower transfers were found in generative and storage plant parts (grains and tubers) than in straw and leaves except for sugar beets and Sr-90 (Table 3-5).

Comparing with the calculated transfer factors for these radionuclides suggested in the provisional Radioecology Regulatory Guide (1), the transfer factors for plants grown on parabrown earth soil were similar. For plants grown on podzol soil, however, they exceeded the suggested values for Sr-90, Co-60, Mn-54, and Cs-137 by factors up to 20, 30, 65, and 82, respectively.

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THE ASSESSMENT OF RADON AND ITS DAUGHTERS IN NORTH SEA GAS USED IN THE UNITED KINGDOM

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There has been considerable interest in the radon content of natural gas in North America for some years, and activity concentrations of up to 1450 pCi l^{-1} at STP have been measured at wellheads(1). In contrast, reported measurements of radioactivity in North Sea natural gas streams have only appeared relatively recently(2). The National Radiological Protection Board began the project described in the present paper in 1977 at the request of the British Gas Corporation. The objectives were to measure the activity concentrations of radon-222, lead-210 and polonium-210 in those gas streams coming ashore in the UK, and to assess the possible exposure of the public that might result from the use of the gas in the home.

MEASUREMENTS

Natural gas is routinely filtered soon after coming ashore to remove particulate material of greater than 1μm aerodynamic diameter before being distributed on the national grid system. The maximum activity concentrations that could reach domestic appliances were therefore assessed by measurements made by sampling after filtration, but before distribution. The radon-222 content was measured directly using 3 litre chambers designed within NRPB(3). The rate of decay was followed and found to be consistent with the 3.8 day half-life of radon-222. The lead-210 and polonium-210 activity concentrations were determined by passing a known amount of gas through concentrated nitric acid, an effective scrubbing agent for metals. A high pressure bubbling system was employed, which had been developed by the British Gas Corporation. This permitted a representative sample of $1m^3$ of the gas at STP to be scrubbed in a reasonable time. Four samples of gas from each stream were separately scrubbed so that duplicate analyses could be performed for each radionuclide. Lead was selectively stripped from the acid solution by a two-stage solvent extraction, the chemical recovery being estimated gravimetrically. After a suitable ingrowth period, the more energetic beta particles from the bismuth-210 daughter were counted using a coincidence-shielded gas-flow proportional counter; the lead-210 content of the sample was inferred from the results. Polonium-210 was determined by electrodeposition and alpha spectrometry using polonium-208 as a tracer. The results available at the time of writing for all the gas lines currently on stream are shown in Table 1.

TABLE 1. Radon-222, polonium-210 and lead-210 activity concentrations of North Sea Gas streams supplying the U.K.

Stream	Radon-222, Bqm^{-3}	Polonium-210, $mBq\ m^{-3}$	Lead-210, $mBq\ m^{-3}$
A	31	2.5	< 17.4
		2.1	< 15.3
B	35	14.1	21.5
		17.0	< 8.1
C	33	4.1	22.2
		3.3	< 7.0
D	27	< 1.5	21.5
		< 1.9	< 7.4
E	36	< 5.6	< 12.2
		< 2.2	< 8.9
F	39	< 2.6	< 7.8
		< 2.2	< 7.0
G	11	9.3	19.6
		10.0	11.5

Note that since 1Bq corresponds approximately to 27 pCi, an activity concentration of 37 Bq m^{-3} is equivalent to about 1 pCi l^{-1}.

The limits of detection are determined both by the sensitivity of instrumentation and by the chemical recovery. Consequently, the limit of detection is particular to each sample. Typical relative standard deviations for these results are 20% for radon and 25% for polonium and lead, based on counting statistics alone.

ASSESSMENT

If the cautious assumption is made that these activity concentrations persist until the gas is combusted in the home, then the maximum possible exposure of individual members of the public may be calculated using a simple model. As a result of filtration, any particulate matter remaining in the gas stream is well within the so-called respirable range.

If there is a steady input rate of activity A into a room which has a ventilation rate of λ air changes per hour, then the total activity present at time t is given by the equation

$$N_t = \frac{A}{\lambda} (1 - e^{-\lambda t}) \qquad \ldots\ldots(1)$$

As t increases, N_t tends towards an equilibrium value N_e where

$$N_e = \frac{A}{\lambda} \qquad \ldots\ldots(2)$$

A typical room might have a volume of some $30m^3$ and a ventilation rate of 1 air change per hour. A typical gas burner used continuously would consume approximately $0.3m^3$ of gas per hour. Under these

circumstances the equilibrium activity concentration C_e in Bq m^{-3} is given by

$$C_e = \frac{N_e}{30} = \frac{0.3\ a}{30} = a.10^{-2} \qquad \ldots\ldots(3)$$

Here a is the activity concentration in the gas in Bq m^{-3}.

RESULTS

Using the highest measured activity concentrations in the gas, the highest concentrations in air for polonium-210 and lead-210 for the conditions assumed are those shown in Table 2. These may be compared to the Derived Air Concentrations for these nuclides appropriate to members of the public which are taken to be one tenth of those for occupationally exposed workers (4).

TABLE 2. Comparison of maximum possible air concentrations with appropriate derived limits

Nuclide	Air concentration, Bq m^{-3}	Derived air concentration, Bqm^{-3}
Pb-210	$2.2\ 10^{-4}$	3.10^{-1}
Po-210	$1.7\ 10^{-4}$	9.10^{-1}

In the case of exposure to radon-222, its short-lived daughters contribute significantly to the dose and calculations are made in terms of the Working Level (WL). The highest calculated exposure will be obtained if it is assumed that the radon daughters are in full equilibrium in the gas at the time of supply to the house. By definition, an activity concentration of 3.7 kBq m^{-3} (100 pCi l^{-1}) of radon-222 in equilibrium with its daughters corresponds to 1 WL. Applying equation (3), the highest radon concentration measured in the gas then corresponds to $1.05.10^{-4}$ WL in the room. The unit of radon daughter exposure is the Working Level Month (WLM), which corresponds to an exposure to 1 WL for one working month (170h). One year of continuous exposure (8760h) to $1.05.10^{-4}$ WL will therefore correspond to $5.4\ 10^{-3}$ WLM in a year. This can be compared to what might be regarded as the maximum permissible exposure for a member of the public, namely 0.4 WLM in a year, (i.e. one tenth of that for occupationally exposed workers) and to an annual exposure of 0.16 WLM from natural background radiation in the U.K. (5).

In summary, the activity concentrations in the home have been calculated using cautious assumptions. No allowances have been made for dilution with gas from other sources (which is presently not significant) deposition of lead or polonium in pipework and, in the case of radon, the time lapse between coming ashore and combustion. It is also known that ventilation rates are usually increased during cooking perhaps by as much as an order of magnitude. Nevertheless the activity concentrations in the house calculated here are two to three orders of magnitude lower than the appropriate limits for members of the public.

It is therefore concluded that no significant radiation exposure of the public results from the distribution of natural gas from the fields in the North Sea that supply the United Kingdom. Nonetheless the activity concentrations of these radionuclides are being periodically monitored, and measurements will be carried out on any new fields that come on stream.

The co-operation of the British Gas Corporation is acknowledged, for permission to publish these results and for their substantial contributions to the sampling programme.

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AN ANALYTICAL APPROACH TO THE COMPARISON OF CHEMICAL AND RADIATION HAZARDS TO MAN

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INTRODUCTION

The main hazards arising from the exposure to ionizing radiation are the so called late effects; the induction of cancer and of hereditary defects. However, there is an increasing awareness that many other environmental agents such as UVR and chemicals are mutagenic and potentially capable of causing cancer and hereditary defects. Thus, the effects caused by radiation are not unique and consequently it is illogical to place radiation in an exceptional position with respect to the protection of man. A comparison between radiation and other mutagenic agents depends on the availability of comparative analytical and assessment techniques. In this paper we present an analytical model, based on radiation biological concepts at the molecular level which permits an analysis of the effects of other agents and also predicts that a synergistic interaction between two different mutagenic agents can occur at the molecular level.

ANALYSIS

Figure 1 presents schematically the molecular mechanisms assumed to be responsible for biological effects such as cell death, aberrations and mutations. The molecular theory (1,2,3) assumes that radiation induced DNA double strand breaks are the crucial radiobiological lesions. The figure shows that double strand breaks have a linear - quadratic dose relationship, that UVR or a mutagenic chemical causing single strand lesions will have a quadratic exposure relationship and that the combined action of radiation and agent gives an additional contribution of double strand lesions arising from the interaction between a single strand break and a single strand lesion. The total number of lesions is

$$N_T = \quad D + \quad D^2 + \quad XD + \quad X^2 \qquad (1)$$

This predicts that for radiation curves the coefficient remains constant but combined treatment increases the linear coefficient (- + X). For agent curves the coefficient remains constant but combined treatment leads to a linear coefficient(D).

Figure 2 presents a series of radiation survival curves after a UVR pre-dose and a series of UVR survival curves after a radiation pre-dose. The analysis has been made according to equation (1), all radiation curves have the same coefficient, increases with increasing UVR pre-dose, all UVR curves have the same coefficient, the linear coefficient increases with radiation pre-dose.

Figure 3 presents the combination of radiation and BUdR on cell survival and chromosomal aberrations. Again the change in shape of the

1. A. Cebulska-Wasilewska was a visiting Fellow from the Institute of Nuclear Physics, Krakow. Poland and was supported by the Dutch International Agricultural Centre.

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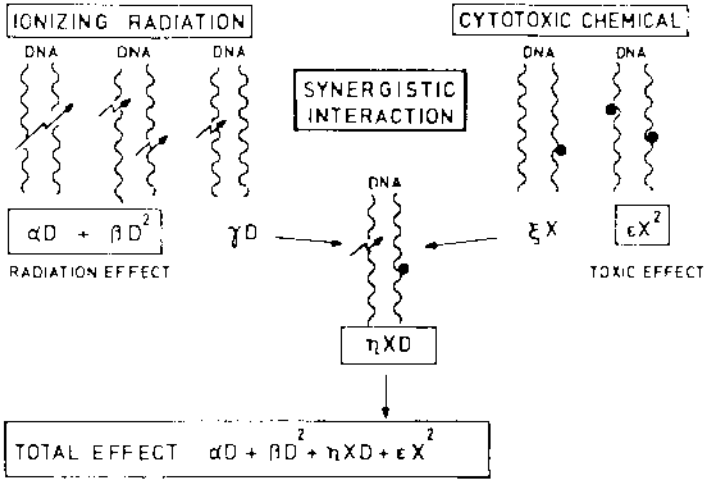

Figure 1. Schematic representation of the action of ionizing radiation, other mutagenic agents and the synergistic interaction between the two agents.

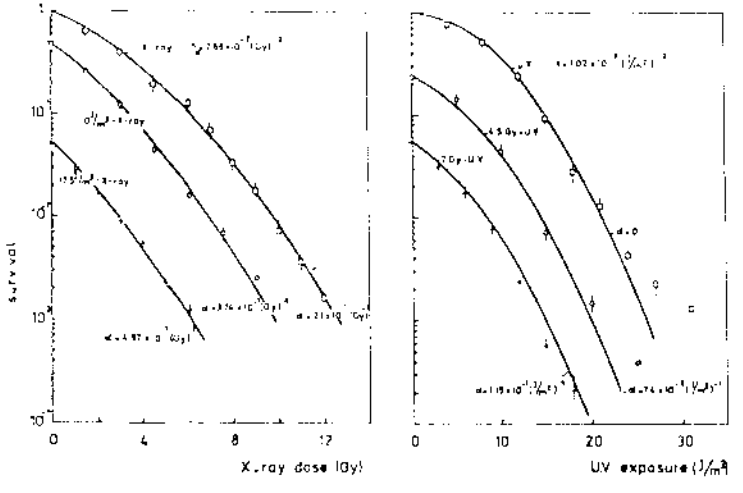

Figure 2. Survival of Chinese hamster cells for combined treatments of X-rays and UVR (4) analysed according to equation (1).

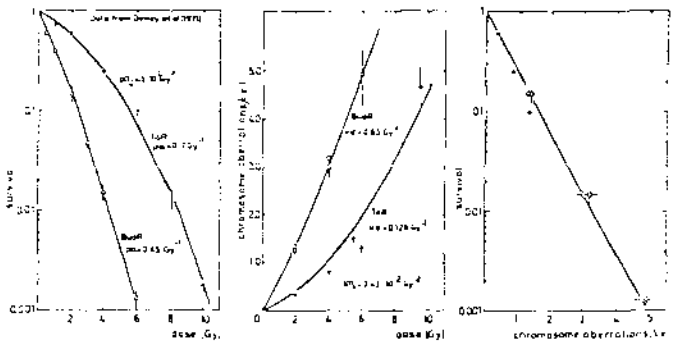

Figure 3. Survival and chromosomal aberrations induced by radiation in synchronised Chinese hamster cells with and without BUdR (5) analysed according to equation (1).

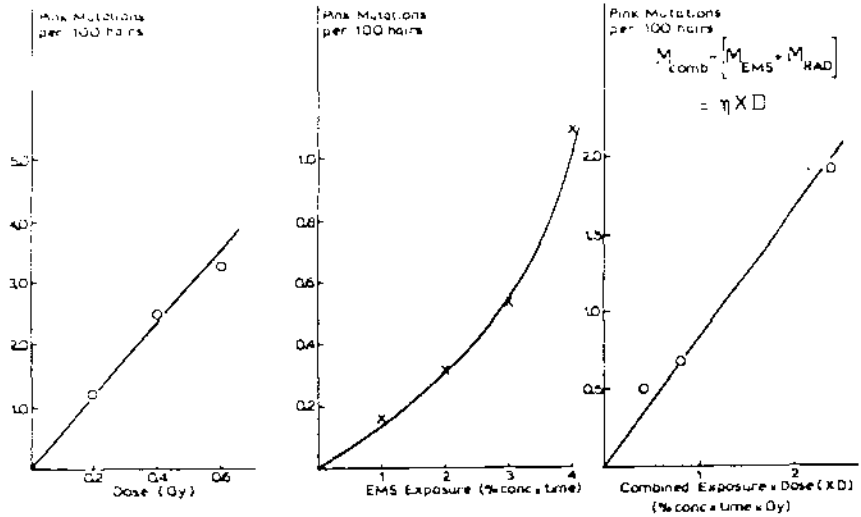

Figure 4. The induction of pink mutations in Tradescantia stamen hairs by X-rays and EMS (ethyl methane sulphonate) and the contribution of mutations arising from the interaction between the EMS and the X-rays as a function of the product of chemical exposure and radiation dose.

curves is reflected in a change in the linear coefficient only. The last part of the figure demonstrates that the change in survival is paralleled by the change in aberrations.

Figure 4 illustrates the effect of EMS and radiation on the induction of pink mutations in the stamen hairs of Tradescantia. The last part of this figure reveals that the difference between the combined treatment and the sum of the separate treatments is proportional with the product of chemical exposure and radiation dose(i.e. XD) as expected from equation (1).

CONCLUSIONS

1. The analysis shows that the synergistic interaction occurs at the molecular level in the DNA.
2. The synergism can be demonstrated in cell survival, chromosomal aberrations and somatic mutations and can therefore be expected forcancer and hereditary defects.
3. The synergistic interaction leads to an increase in the linear component of the radiation effect which is critical at low doses and important for radiological protection.
4. The model implies that a synergistic interaction between two different mutagenic chemicals can also be anticipated.
5. The model provides an analytical vehicle which can be used to compare the effects of radiation and other mutagenic agents at the mechanistic level.
6. If we are concerned to protect man from the increasing mutagenic, and thus carcinogenic load, an integral protection philosophy is essential.

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TECHNOLOGY TRANSFER FROM NUCLEAR AND RADIOLOGICAL TO INDUSTRIAL SAFETY

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The term technology transfer usually means the supply of equipment, information and techniques from a more developed supplier country to a less developed recipient country, together with an educational, human developmental package. In the ideal case, these enable the recipient to use the acquired technology independently and to develop it further.

Technology transfer in a broader sense includes situations in which the transfer occurs between previously not interconnected fields, in which the subject of the transfer is the philosophical, scientific, technical or managerial approach to a set of problems or in which the educational package is replaced by transit of individuals or groups from one activity to another. There are many such examples.

During the last decade another technology transfer process has developed, from the nuclear and radiological to the occupational safety and hygiene fields.

The responsible governmental organs in many developed countries realized that their efforts result in significantly lower levels of safety and health at work than desirable, reasonably achievable and socially acceptable.

The recommendations of governmental review committees on the subject (1) and the legislative actions taken upon their acceptance(2) show the marked influence of approaches well known and widely practised in nuclear and radiological safety, recognizing that the level of occupational safety achieved in these activities - by using specially developed approaches - is significantly better than in comparable non-nuclear ones.

In August 1978 the Minister of Labour and Social Affairs of Israel appointed a committee - under the chairmanship of the author - to review the state of safety at work in Israel. The committee had a broad mandate but rather limited time to present its recommendations. These were submitted to the Minister in March 1979. The final report (3) was published in August 1979 in Hebrew.

The following basic findings of the committee may well have general relevance, as similar conditions existed, exist or may exist in many developed and other countries.

(1) The organs responsible for the control of safety at work are fragmented and scattered between different authorities. The level of interaction between these organs is low and there is no integral approach to safety at any given facility. Coordination between these organs is ineffective and hazards of different nature are handled independently by the different authorities.

(2) The major responsibility for controlling safety at work is that of the work safety inspectorate. It acts on outdated and inadequate legislation, with limited manpower, and is unable to cope with the growing complexity of the industry, with the increasing concentrations of potential energy, hazardous materials at single locations and

mainly with the less obvious and apparent nature of some modern hazards. Actual and potential off-site effects (on populations or on the ecology) are unfortunately outside the jurisdiction of the inspectorate.

(3) The penalties in the law and the actual sentences imposed by the Courts for safety violations and for maintaining unsafe conditions in facilities are not an effective deterrent.

(4) Present legal, taxation and insurance practices enable employers to transfer the burden of economic losses due to unsafe operational conditions including accidents to the State or in general to the public. There is no economic incentive to employers to invest in the safety of their operations.

(5) There is no suitable data base for priority/policy decisions on matters of safety. Courses of action are chosen on intuitive basis. The marginal investment in risk reduction - where it exists - varies by orders of magnitude between different industries and work places both in monetary terms and in inspection effort.

(6) R&D projects are carried out without the establishment of a general plan, as no such plan or priority system could be developed in the absence of a suitable data base. There is no reliable system for the transfer of R&D results into general practice.

(7) There is no systematic and timely approach to hazards of stochastic nature and there is no systematic control of them. The information base on these hazards is insufficient.

(8) Improvements are made on a patchwork basis, usually following severe accidents or the detection of grossly unsafe situations.

(9) Training opportunities are scarce and generally non-specific in quantity and relevance.

(10) In general, there is no emergency planning - not at the off-site level nor at the on-site, intervention level.

(11) The professional level of many engaged in safety related activities (in the work places and in the active governmental organs) needs improvement. This is especially valid with respect to safety personnel at the facility level.

(12) Workers' participation is ineffective due to reasons similar to those already mentioned, little or no influence and deterrent, lack of professional ability and information.

Some of the specific findings of the committee, regarding the Israeli situation were:

(1) the rate of reportable work accidents (absence of three or more days from work) is approximately 80 per year per 1000 employed persons in all types of employment and about twice this rate among the employed in construction and industry.

(2) the rate of fatal accidents is approximately 0.2 per year per 1000 employed (of which ~ 50% are traffic accidents on the way to or from work - considered as work accidents for compensation).

(3) the rate of accidents causing permanent disability above 20% is approximately 0.7 per year per 1000 employed and of those causing temporary disability, about four times higher.

(4) the estimated financial loss to the economy due to accidents at work is in the range of $300-500 million, i.e., ~ 3 - 5% of the GNP. The public expenditure on safety is less than one percent of this sum.

(5) there has been no trend of improvement during the past 5-10 years.

The major recommendations of the committee were:

I. AT THE FACILITY LEVEL

(1) Approach the safety of work places on integrative basis. Aim for safety as a built-in aspect of every activity, not an extra, add-on type one.

For new facilities the consideration of the desirability of an obligatory safety assessment was recommended, as a condition for a licence. For existing facilities the recommendations call for written safety policy statements and improvement programmes based on a thorough review of the existing hazards.

Both types of document should contain a description of the potential hazards of the operation and the appropriate countermeasures employed.

(2) Encouragement of worker-management co-operation in facility safety committees required by law on a more objective and professional basis, with the help of outside specialists if needed. All the potential hazards should be openly discussed - the workers should be properly informed about the risks they are exposed to.

(3) Safety should be part of management's responsibility at all levels. Duties and responsibilities and the authority of the different managerial levels with regard to safety should be clearly defined.

II. AT THE NATIONAL LEVEL

(4) Reorganization of the functions presently dispersed between the different authorities, ministries, etc., into one coherent unit, preferably in a unified national occupational safety, hygiene and health service.

(5) Establishment of research coordination and engineering development units within the unified service. This will also contribute towards the increase of the professional capability of the service as a whole.

(6) Establishment of coordinating functions within the unified service with other, presently not-interacting services fully or marginally relevant to safety at work.

(7) Encouragement of voluntary organizations in the field (professional societies, unions, associations of employers, etc.).

IN SPECIFIC FIELDS

(8) Modernization and updating of the legislation

(a) to reflect the recommended changes;

(b) to re-establish the deterrent force of the law by severely increasing the penalties on safety violations and by speeding up Court procedures; and empowering the safety inspectorate to impose administrative fines.

(c) to widen the coverage of hazardous situations by regulations, including the promulgation of satisfactory, technically acceptable solutions. (Codes of practice).

(9) Establishment of inspection priorities according to the hazard level, safety record and safety arrangements in the inspected facilities. Introduction of obligatory investigation of every serious work accident.

(10) Establishment of a satisfactory data base and of a specialized safety information centre.

(11) Improvements in safety-related R&D management, in occupational hygiene and health control, in training activities at all levels, according to a priority system to be developed.

(12) Encouragement of research and publication on the economic aspects of work safety. Promotion of a cost benefit approach as a tool for priority assessment and investment evaluations.

As said above many of the recommendations, including part of those mentioned above, are based on practices which evolved and are used in the nuclear field. Basically the report calls for the introduction of a similarly organized, systematic approach to the different industrial hazards.

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RADIATION PROTECTION PRINCIPLES APPLIED TO CONVENTIONAL INDUSTRIES PRODUCING DELETERIOUS ENVIRONMENTAL EFFECTS

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Comparison of the radiation protection standards for the population-at-large with the ambient standards for conventional pollutants, reveals differences in the principles upon which the standards are based.

The most important factors considered in establishing the radiation protection standards for populations-at-large (as well as for specific sections of the population) are as follows:

1. Somatic effects, influencing the exposed person himself
2. Genetic effects, influencing the descendants
3. Effects on specific tissues and organs of the human body (leading to the concept of critical organ)
4. Additivity of effects, considering the simultaneous effects on several tissues and/or the whole body, and the total risk to all irradiated tissues
5. Sensitivity of exposed person, e.g. children
6. Stochastic health effects, i.e. the probability of the occurrence of an effect, as a function of dose
7. Non-stochastic health effects, i.e. the severity of an effect, as a function of dose
8. Quantitative acceptable risk (e.g., expressed as deaths per person per year) based on risks experienced or acceptable in other human activities
9. Size of population exposed, expressed as the product of the exposure dose and the number of persons exposed, assuming a linear dose-effect relationship
10. Cost-benefit and ALARA (as low as reasonably achievable) considerations, taking into account technical, economic and social factors in limiting the exposure.

With the exception of the non-stochastic health effects, none of the other factors considered in establishing the radiation protection standards, are taken into account (certainly not explicitly) in spelling out the ambient standards for conventional pollutants, such as SO_2 and NO_x.

These and other differences result in more relaxed ambient standards for conventional pollutants, in comparison with the radiation standards, as illustrated by the ratios between the standards and the natural, medically perceivable and lethal levels (Table 1). The differences are in the range of orders of magnitude. The consequence of the severity of the limitation of exposure to radiation, as compared with conventional pollutants, is the penalization of the nuclear industry due to the increased cost of its safety measures.

TABLE 1. Comparison of different levels of exposure to radiation, SO_2, and NO_2

	SO_2* (ppm)	NO_2* (ppm)	Radiation (mR/day)
Level			
Background	0.0002	0.01	0.35 (130mR/y)
Maximum permissible exposure	0.03	0.05	0.03 (8mR/y)
Medically perceivable effect**	0.02	2	$2x10^4$ (20R)
Lethal**	0.5	500	$5x10^5$ (500R)
Ratios of levels			
Maximum permissible/background	150	5	∿0.1
Medically perceivable effect/ maximum permissible	0.6	40	$\sim 6x10^5$
Lethal/maximum permissible	17	10^4	$\sim 10^7$

* with particulates
**one-day level

The fact that deleterious exposures are not restricted to the same extent in different human activities, appears to cause a misuse of public resources. Considering that there is a limit to the public economic means available, societal expenditures for reducing risks should be spread, as much as possible, over all human activities to get the maximum return from investments. Indeed, the law of diminishing returns indicates that the return on investments to decrease the marginal hazards of an activity is insignificant as compared with the return on initial expenditures which diminish the hazards substantially. The nuclear industry is already at the stage where additional expenditure brings only marginal returns, while many conventional industries are at the initial stage of safety expenditures.

The greater safety cost imposed on nuclear power plants, as compared with conventional power plants, may result in the substitution of a hazard worse than the radiation hazard due to the release of SO_2 and other harmful pollutants from conventional power plants.

It is proposed that, to diminish the hazards to the public uniformly and effectively and also to get an optimum return on the safety investments made by the public, radiation protection principles should be used as prototypes for pollutants having harmful environmental effects. It is also proposed that radiation health physicists should be active in the application of these principles of population protection.

The application of one of the principles of radiation protection, that of limiting the integrated population exposure (expressed as person x rem), is illustrated here for a conventional pollutant, such as SO_2. A study of the atmospheric release of SO_2 under different conditions is analyzed, to emphasize the importance of considering the size of the exposed population.

Assume that the only requirement concerning the release of SO_2 from any installation is to keep the ambient air concentration at the fence of the installation below the half-hour standard of 0.3 ppm, adopted in Israel. Assume also that the density of the population, uniformly distributed around the installation is 400 persons/km^2.

With these assumptions, the ambient SO_2 concentration (ppm) and the total integrated population concentration (person x ppm) were calculated for a distance up to 80 km from the source for two release cases: a) ground-level release (Table 2) and b) release from a height of 200 m (Table 3).

TABLE 2. Integrated SO_2 population concentration for a ground level release. Assumptions: 1) concentration of SO_2 at the fence of the plant (1 km from the source) = 0.3 ppm, 2) deposition velocity of SO_2 = 1 cm/sec, 3) population density = 400 persons/km^2, 4) average atmospheric conditions

Distance (km)	Population	Concentration (ppm)	Integrated concentration (person x ppm)
0-2	4×10^3	1.2×10^{-1}	500
2-3	8×10^3	5×10^{-2}	400
3-5	1.8×10^4	2×10^{-2}	300
5-8	3.3×10^4	7×10^{-3}	200
8-16	3×10^5	5×10^{-3}	1,500
16-30	7×10^5	5×10^{-4}	350
30-50	1.5×10^6	2.5×10^{-4}	400
50-65	2×10^6	2×10^{-4}	400
65-80	2.5×10^6	10^{-4}	200
			~4,000

The classical Gaussian plume formula (1) was used in these calculations, assuming average atmospheric conditions, and the integrated population concentration was calculated by multiplying the number of persons in concentric rings at various distances around the installation by the concentration calculated for the middle of the ring.

Assuming that the ambient concentration at the fence of the installation is at the level of the half-hour ambient standard, it was found, as expected, that the ambient concentration at any distance further from the source is below this concentration, for both the ground-level and 200 m height releases.

However, there is a very significant difference between the integrated population concentrations in the two aforementioned release cases. Assuming a deposition velocity of 1 cm/sec, the integrated population concentration for a ground-level release is about 4,000 person x ppm, while for a release at a height of 200 m, it is about 138,000 person x ppm.

It should be stressed again that in both release cases, the ambient concentrations are below the standard. However, the difference by a factor of up to about 35 in the integrated population concentrations indicates that the integrated population concentration

for conventional pollutants should also be limited as in the case of radiation protection, in addition to the limitation of the ambient concentration.

TABLE 3. Integrated SO_2 population concentration for an elevated release. Assumptions same as in Table 1, except for height of release which is assumed to be 200 m.

Distance (km)	Population	Concentration (ppm)	Integrated concentration (person x ppm)
0-2	4×10^3	0.3	1,200
2-3	8×10^3	0.3	2,400
3-5	1.8×10^4	0.25	4,500
5-8	3.3×10^4	0.25	8,200
8-16	3×10^5	0.07	21,000
16-30	7×10^5	0.04	28,000
30-50	1.5×10^6	0.02	30,000
50-65	2×10^6	0.01	20,000
65-80	2.5×10^6	7×10^{-3}	17,500
			$\sim$132,800

The importance of applying to conventional pollutants, the other principles and factors considered in establishing the radiation protection standards, could be similarly demonstrated.

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A HEALTH AND RESEARCH ORGANIZATION TO MEET COMPLEX NEEDS OF DEVELOPING ENERGY TECHNOLOGIES

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INTRODUCTION

The rapid development of technology, particularly in the area of energy research and development, brings with it increasingly more complex and sophisticated health and safety problems. The complex safety requirements also bring the need for greater interaction between health and safety disciplines that have historically functioned quite independently. The Lawrence Livermore Laboratory employs approximately 7000 people in various areas of energy technology development and nuclear research.

All of the LLL safety functions (except the Medical Department) are assigned to the Hazards Control Department. This department has responsibility for radiation safety, industrial hygiene, industrial safety, fire safety, and explosive safety. The safety program is managed through a system of program oriented field teams. Each team has a leader and a cadre of health and safety technicians. The resources necessary to make the field teams functional are provided from the health and safety discipline areas listed above. For example, the field team responsible for the general area of chemistry would have representatives from industrial hygiene, radiation safety, fire safety, etc. assigned to work together, insuring a multidisciplinary approach to solution of safety problems in that area

We, as many other moderate or large size laboratories, have found that operational safety questions frequently arise that cannot be answered or solved through available data, techniques or instrumentation. These questions can only be resolved through in-house cost effective safety technology development. At LLL, this development is done through a single division within the Hazards Control Department, that responds to health and safety technology problems in radiation science, fire science, industrial hygiene, and general safety.

SPECIAL PROJECTS DIVISION

The Special Projects Division was formed because it was recognized early in the formation of the Hazards Control Department that development of health and safety technology was sufficiently important to require a separate dedicated effort. Special Projects is a 27-person safety technology development organization representing approximately 10% of the Hazards Control Department work force. The primary mission of the Division is to provide health and safety technology development required by the operational safety program at the Laboratory. Division responsibilities include:

- Development of equipment or instrumentation
- Development of techniques for health and safety
- Studies to provide information needed by management or operational safety professionals

- Measurements of safety related parameters of Laboratory facilities, equipment, or materials
- Calculations or calculational studies.

The Special Projects Division is organized into three functional groups - Radiation Science, Safety Science, and Fire Science. These groups share common office and laboratory spaces, and interact with each other on a regular basis. This organization factors multi-disciplinary attention to general safety problems; i.e., a skill generally required for one class of health and safety discipline is available for solutions to problems that occur in others. For example, aerosol physics and filtration are areas that present problems for each of the three of the groups in our Division. Rather than hiring an aerosol physicist for each group, we are able to share the talents of one or two individuals who meet our technological needs. This is also true in areas such as instrumentation, chemistry, computer science, and others. Table 1 shows the current skills inventory in the Special Projects Division. This inventory meets the current needs of our Department and Laboratory. It is intended as an example, and may not apply to other facilities.

Table 1.

SPECIAL PROJECTS SKILLS INVENTORY

Combustion Science	X-Ray Fluorescence Analysis
Analytical Chemistry	Radiation Spectrometry
Risk Analysis	Health Physics Instrumentation
Industrial Hygiene	Industrial Hygiene Instrumenation
Aerosol Physics	Radiation Physics
Health Physics	
Solid State Dosimetry	

The Radiation Science Group (RSG) develops instrumentation and techniques for radiation protection and measurement. The group provides technical support to the Radiation Safety Program at the Laboratory, particularly in the areas of personnel dosimetry development, applied health physics, calibration and standards development, and the Laboratory environmental monitoring program. Expertise developed in radiation safety technology is often valuable to other Laboratory programs. Consequently, this group provides radiation instrumentation/measurement support to particular programmatic efforts at the Laboratory.

The Safety Science Group (SSG) is responsible for solution of problems in the fields of occupational health and general safety. The SSG obtains data and develops specialized equipment to support safety discipline needs in respiratory protection, work place monitoring, improved air cleaning and monitoring techniques, and evaluation of protective clothing. Recently the SSG has undertaken the additional responsibility of technical support to the Laboratory Waste Disposal and Decontamination facility efforts.

The Fire Science Group (FSG) develops solutions to problems faced by the Fire Safety Program at LLL. The FSG also uses its expertise to answer fire safety related questions for the Department of Energy, in general. The technical scope includes: materials testing for small-scale flammability tests to full-scale enclosure fires, development of unique modes and mechanisms for fire extinguishants; tests and analysis of fire detection concepts of hardware, analysis of physical and chemical properties of smoke aerosols; studies of fire retardant application to natural and synthetic materials; and parametric analysis of the interaction between fire management systems and the fire hazards potential of experimental facilities. The FSG maintains a full-scale fire test enclosure that is equipped to simulate fires in laboratory environments with typical fuels that might be found in the laboratory. It also has a fine chemical analytical laboratory including a sophisticated gas chromatograph-mass spectrometer for analysis of fire decomposition products.

PROJECT DEVELOPMENT

A key to the success of an organization such as the Special Projects Division is the need for close, working interaction with the operational health and safety staff of the department. Special Projects personnel are encouraged to interact with their operational counterparts daily in a candid and personal way. Some of the best technology development in the safety field comes from such interaction. A project may be conceived either by a Special Projects Senior Investigator, an operational client or both. Once formulated, projects must be approved by the client and Special Projects Division Leaders. Projects are reviewed formally once each quarter through a system of status reports. The progress of each project is monitored by the Steering Committee. The Steering Committee is composed of the Special Projects Group Leader, the Special Projects Division Leader, and the Division Leader and Group Leader in the counterpart Operational Health and Safety Division in Hazards Control. These steering committees meet regularly to review the status of ongoing projects. It is important for successful completion of a project that the senior investigator and client review the project frequently.

Depending on the complexity and time requirements on any given project, a Senior Investigator, who is generally a safety professional with an advanced degree, will have from one to four projects assigned at a given time. A Senior Investigator may also be called upon occasionally to provide programmatic support to research and develop problems arising from outside the Hazards Control Department, although that is not part of their prime mission.

PROJECT EXAMPLES

A major strength of the Special Projects Division is its multidisciplinary structure - the ability to combine a wide spectrum of skills to achieve successful project completion. Examples of the potential for interdisciplinary accomplishment include testing the fire resistance of neutron shielded storage containers; on-line

measurement of toxic metal concentrations in Laboratory sewage effluent using special X-ray fluorescence and radiation detection techniques; and collaboration between the Safety Science and the Fire Science Groups to develop filtration and sampling techniques for dense smokes.

Other examples such as these could be cited to demonstrate the interaction between the disciplines, but perhaps the greatest benefit to the Laboratory is derived from non-specific daily interactions and discussions both within the Division and the Department. Without the unified structure that the Special Projects Division enjoys, interdisciplinary communications would be much less frequent and productive. An additional advantage to the organization and the Department is the ability to share resources among groups rather than having to duplicate resources. Single high quality laboratories for testing, analysis, dosimetry, etc. provide a maximum flexibility at a minimum cost.

SUMMARY

At the Lawrence Livermore Laboratory, a unique safety technology organization has been established that is especially geared to respond to interdisciplinary health and safety questions in response to rapidly growing energy technology problems. This concept can be adopted by smaller organizations at a more modest cost, and still maintains the efficiency, flexibility, and technical rigor that are needed more and more in support of any industry health and safety problem. The separation of the technology development role from the operational safety organization allows the operational safety specialists to spend more time upgrading the occupational health and safety program but yet provides the opportunity for interchange with health and safety technology development specialists. In fact, a personnel assignment flow between an operational health and safety organization and a special technology development organization provides a mechanism for upgrading the overall safety capability and program provided by a given industrial or major laboratory.

ACKNOWLEDGEMENTS

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DEVELOPMENT AND TRENDS IN RADIOLOGICAL PROTECTION AND THE NEA PROGRAMME IN THIS FIELD

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INTRODUCTION

The Nuclear Energy Agency (NEA) was initially created as a European organisation (ENEA) in 1957, but it assumed its present broader configuration in 1972 with the entrance of Japan among participating countries, successively followed by Australia, Canada, and the United States. A major task of NEA is to encourage harmonization of governments' regulatory policies and practices and to promote the exchange of information and the co-ordination of research and development in the field of radiological health and safety.

The work of the Agency is carried out under the authority of the OECD Council by the Steering Committee for Nuclear Energy which is assisted in its work by a number of specialised committees. In particular, the Committee on Radiation Protection and Public Health, which had originally been set up in 1958 as the Health and Safety Sub-Committee, is responsible for the Agency's activities concerned with radiological protection and related environmental problems. Its activities include the review and discussion of national radiation protection policies and practices, the review of progress of radiation protection philosophy and the interpretation of the ICRP Recommendations, the study of the means of their conversion into practical applications, including the establishment of radiological protection standards as well as the preparation of technical studies and state-of-the-art reviews on specific problems.

THE EARLY DEVELOPMENTS

Since the beginning, radiological protection became a major part of the Agency's programme. The areas of concern for the national and international radiation protection communities changed significantly during the last 25 years, and the Agency programme evolved with them. During the 1950s and the early '60s, the attention of public authorities and radiation protection specialists was primarily focused on two major questions.

One was the need to elaborate radiological protection regulations. Many countries, at that time, had not yet worked out these measures in detail, and still lacked the necessary experience to implement them. Moreover, it was soon realised that these regulations should have been as uniform as possible throughout countries in order to avoid international trade and public opinion repercussions resulting from inadequacy or excessive severity of individual national regulations. This need for an international co-ordination was soon appreciated by the

Steering Committee, and the first task of the newly-created Health and Safety Sub-Committee was the preparation of basic norms for protection against radiation. These norms were adopted in 1959 for use in Member countries as one of the bases for their forthcoming legislations. One specific concern in preparing these norms, in line with the Agency's vocation, was to transfer the ICRP policy and conceptual language into more practical terms, more easily applicable to a regulatory context. The Agency did not limit its effort to the preparation of basic norms, but it continued to keep under constant review the developments and trends of radiation protection policy and the ICRP recommendations. In this context, the basic norms were subjected to revision in 1963 and 1968. A major revision is also currently under way in order to take into account the new principles set down in ICRP Publication 26. This revision presents a significant difference from the previous ones; the latter were in fact carried out independently by NEA, though in co-ordination with other international organisations. In the last few years, however, it was realised that it was in the interest of Member countries to have a single series of international recommendations on radiation protection norms. Therefore, arrangements were made in 1977 with IAEA, WHO and ILO to prepare a joint revision of the relevant norms and to publish a unified set of standards applicable by all international organisations. The publication of the joint revised basic norms is expected by 1981.

The second major concern in the period of the 1950s and early 60s was the public health risks associated with the radioactive fall-out from nuclear explosions, and the methods and techniques for the measurement of low levels of activity in environmental matrices. Also in this field, the Agency played an active role. This was the setting up, in 1959, of a system for the centralised collection, comparison and dissemination of information and data resulting from the network of environmental radioactivity measurement stations located in Europe. This mechanism for an oriented exchange of information was operated for about 10 years; then, with the dramatic decrease in environmental radioactivity levels in the years after 1962, interest in an international exchange of information declined and the system was discontinued. Also, the Agency deployed a limited effort towards standardisation of sampling and measurement methods used in the different countries.

The concern of national authorities at the increasing levels of environmental contamination due to fall-out, and the growing awareness of the risk of nuclear accidents possibly involving bordering countries, induced NEA to set up, in 1961, an international system of supervision and emergency warning in the case of an increase in environmental radioactivity in one Member country. The technical features of this system, the scope of which

was based only on the results from airborne radioactivity measuring stations, were of course not such as to generate a really timely and effective warning mechanism. However, those features were in line with the technical means and knowledge available at that time and, in any event, the NEA system had the undeniable merit of supplying a pre-arranged framework for quick contacts and consultations between national authorities in case of any problems concerning radiological protection of the public. It should also be appreciated that this system, with its obvious limitations, was probably the first international attempt to set up forms of co-operation between countries in the field of nuclear emergencies. The system was operated for several years, and successively, although it was never officially abrogated, it was practically abandoned in the last few years. In any event, the seed of international co-operation had been sown, and since then better and technically more adequate emergency warning and co-operation systems have been set up between bordering countries.

THE MATURITY OF THE AGENCY'S PROGRAMME

The 1960s saw the suspension of significant nuclear tests in the atmosphere, and the parallel impetuous development of nuclear energy and its various applications. In this changing scenario, other issues assumed a growing emphasis. These were the problems and techniques involved in the protection of workers in nuclear establishments, and the methods and instruments for the health physics surveillance and dosimetry. In these fields NEA did not play a leading role, apart from organising a number of seminars and symposia on health physics and dosimetry matters. In the same period, however, NEA showed a particular sensitivity and promptness in tackling some other problem areas which were at that time only beginning to concern public health authorities.

The ICRP Publication 7, in 1965, introduced a significant quality step into the criteria for the environmental monitoring around nuclear facilities. Concepts such as the critical groups of population and critical parameters of the environment were the basis for a more rational and cost-effective approach to monitoring, and paved the way for the development of the analytical models which are now an essential tool for the assessment of the environmental impact of any nuclear facility or waste management operation. Since 1961, the NEA Health and Safety Sub-Committee realised the importance of this subject, and focused its attention on the study of the environmental behaviour of radioactive wastes in the marine environment, and the associated radiological risks. A number of studies were therefore carried out between 1962 and 1964 on the oceanographic and radiological aspects of the presence of radioactive materials and the discharge of radioactive wastes into the North Sea. These preparatory studies culminated with the preparation

of a joint theoretical study of the radiological capacity of the North Sea for the discharge of radioactive wastes. This was probably one of the first examples of a radiological risk assessment applied to a large ecosystem and, we suppose, it was the first to result from a joint international effort. This specialisation of NEA expertise on the problems associated with waste disposal into the sea constituted the basis for the increasing involvement of the Agency in the study and, successively, the implementation of operations involving waste disposal into the ocean under international surveillance. In this field, the Agency had, and still conserves in the framework of the Multilateral Consultation and Surveillance Mechanism for Sea Dumping of Radioactive Waste established in 1977, the role of organising an international co-operation, including radiological risk assessments and radiological protection inspection during the disposal operations, on a practice followed by several countries and which might otherwise be carried out without proper guarantees and information to all countries.

The other problem area in which NEA showed a prompt adaptation to the forthcoming requirements of Member countries was the spreading application of small quantities of radioactive materials in consumer goods of any sort. At that time the concern was focused on the individual risk from radiation exposure and the protection of individuals. Only a few, far-sighted experts were anticipating the far-reaching importance that the collective radiation detriment would take on in the following years and the resulting potential public health problems to be raised by the exposure of large groups of people to small levels of radiation. It is therefore to be ascribed to the merit of NEA the vigorous programme, carried out from 1962 to 1978, of studies and guidelines on the radiation protection problems raised by public exposure to consumer goods containing radioactivity and by natural, but artificially enhanced, radiation exposures. The NEA action in this field was in line with its institutional goal of transferring the philosophical language of the ICRP recommendations and the administrative/regulatory language of the basic radiation protection norms into terms more suitable for direct application to specific situations.

This activity started in 1962 with the preparation of radiation protection standards for radioluminous timepieces (published in 1967) and continued very actively with the publication (between 1970 and 1977) of standards or guidelines concerning the design, construction and use of radioisotopic power generators, gaseous tritium light devices, particle accelerators, cardiac pacemakers, smoke detectors. It is important to note that several of the above guides and standards were developed in collaboration with the other international organisations and some of them were also adopted by IAEA as their own standards.

A particularly interesting case of a guide in this

area was that entitled "Basic approach for safety analysis and control of products containing radionuclides and available to the general public" published by NEA in 1970. While the other guides were intended for use not only by regulatory authorities, but also by designers and utilisers, the latter was exclusively addressed to regulatory authorities. It contains the general principles of the analysis that licensing authorities should carry out before granting authorisations, as well as the technical bases for this analysis and the control procedures to be set up.

At this point, it is worth noting that, besides the basic radiation protection norms, the field of consumer goods is the only one where NEA embarked on an activity of international standards and guides. In the other fields, also in harmony with a general co-operation and co-ordination agreement signed with the IAEA in September 1960, the Agency focused its activity on the study and catalysation of co-operative efforts in advanced areas still presenting open problems.

The activity of NEA in the area of consumer goods and natural radiation was not limited to the publication of standards and guides, but also a number of studies were carried out. It is sufficient to mention the studies on radiation protection problems of lightning conductors, use of depleted uranium as ballast in aircraft, radioactivity of building materials, airborne natural radioactivity, etc. Not all of these studies were widely disseminated through formal publication. A few of them, in fact, remained, for various reasons, at the stage of reports for internal use by the national authorities. But even in this limited distribution, they represented a useful reference contribution to the development of knowledge and radiation protection criteria in Member countries.

THE PRESENT EVOLUTION AND FUTURE TRENDS

In the last ten years nuclear energy, which had previously shown promising development, began to face growing opposition as well as increasing economic and political difficulties. One of the concerns at the basis of this crisis was a deep and sometimes exasperated attention to the environmental impact of nuclear energy. This called for great efforts to be focused on the treatment and retention of radioactive effluents, as well as on the methods for assessment of the environmental behaviour and radiological impact of these effluents and for their monitoring. But the attention of the experts and the public opinion were progressively concentrating on the question of the long-term management of radioactive wastes. This immediately began to raise the problems of the assessment and criteria for acceptance of the associated long-term radiological risks. In this context, a particular international resonance was given to the publication by NEA, in 1977, of a state-of-the-art report,

known as the Polvani report, on the concepts and strategies for the management of radioactive wastes.

During the 1970s, NEA rapidly became sensitive to these new requirements and its programme was progressively adapted by the Committee on Radiation Protection and Public Health to cope with the new challenges. In particular, the Committee reoriented the emphasis of the programme towards the problems of radiation protection and environmental impact of nuclear fuel cycle facilities, with special attention to the front-end (uranium mining and milling) and the back-end (waste management) of the fuel cycle. Its growing involvement in the problems associated with the nuclear fuel cycle obliged the Agency to concentrate its limited resources in those areas where a priority effort appeared warranted. Therefore, while increasing efforts were devoted to sensitive issues in the nuclear fuel cycle, the activities in areas not directly connected with the fuel cycle, such as the consumer goods and other low-level sources of radiation detriment, were progressively reduced to minimal levels.

The turning point of this evolving policy was in 1976. In that year, for the first time, the major part of the Agency's programme was devoted to radiation protection in the nuclear fuel cycle. Here again, the Agency fulfilled its vocation for tackling new problems in the moment of their formation and catalysing international interest and efforts towards their solution. In the last few years, in fact, the areas which called for attention were the transfer into practical terms of the new ICRP recommendations of Publication 26, the protection of the public and future generations against the long-term sources of radiation such as the uranium mill tailings and high level transuranic wastes, and the protection of workers in selected occupations with relatively high risks, such as uranium mining and maintenance work in nuclear power plants. Most of the new lines of activity started in 1976 were in fact focusing on selected issues in the abovementioned areas. In order to put into a proper perspective the radiological impact of the different stages of the nuclear fuel cycle and the problem areas requiring a selective effort, the first action of NEA was to sponsor a study on the relative radiological significance of all potential sources of human exposure to radiations. The report on this study, prepared by Sir Edward Pochin, was published in 1976.

In the field of the long-term impact of the nuclear fuel cycle, the criterion adopted was to select a few radionuclides of particular significance as potential long-term sources of human irradiation, and to study in detail their behaviour and their potential risk for present and future generations. A first study was focused on plutonium and other transuranics: this was of a scientific nature and its objective was to bring together the salient facts about the biological and environmental behaviour of these nuclides in order to assist in the

appreciation of their potential risk. The result of this study is a comprehensive state-of-the-art report, publication of which is planned by mid-1980.

Another case of a specific study in this general area was an analysis of the radiological significance of four long-lived radionuclides arising, as airborne effluents, from the operations of the nuclear fuel cycle, namely ^{3}H, ^{14}C, ^{85}Kr and ^{129}I. The particular interest of this study lies in the fact that it is one of the first examples, perhaps the first from an international group of experts, of an attempt to demonstrate a practical application of the ICRP optimisation principle. The study, in fact, assesses the radiation detriment and costs associated with different retention technologies and combines them into a differential cost-benefit analysis, concluding with recommendations as to the policies to be followed for an optimised management of the above nuclides. In spite of a certain delay due to its industrial policy implications, it is hoped that the final report will be published during 1980.

Another issue of concern, both from the occupational exposure and environmental protection viewpoints, was the expansion in several countries of uranium mining and milling activities. The involvement of NEA in this field began in 1976 with the organisation of exchanges of information on personal dosimetry and area monitoring in uranium mines. The international debate progressed during the last three years with other meetings and seminars, within the framework of NEA, and permitted the clarification of the nature of the problems and identification of key subjects requiring a priority attention or effort. On this basis, the Agency was able to establish, in 1979, a consolidated programme on the long-term radiation protection and waste management aspects of the uranium mill tailings. These were, in fact, identified as a form of radioactive waste of low specific activity but having significant long-term implications due to their enormous quantities and very long half-life of the radionuclides involved. This three-year programme is presently starting and includes items such as the formulation of radiological protection principles and criteria for application to the long-term management of tailings, based on the ICRP system of dose limitation, as well as the study of environmental models for the assessment of the impact of the release of contaminants from the tailings.

Another area where the abovementioned international debate led to the establishment of a stimulating programme was the parallel subject of radon, and its daughters dosimetry and monitoring for workers and for environmental surveillance in connection with uranium mining and with mill tailings management. A sizeable effort is presently going to be devoted to studies on the analytical models for radon dosimetry, the influence of factors and parameters affecting the dose to lung from

radon and its daughters, the principles and methods for measurement and monitoring of radon and daughters.

During the last few years, increasing improvements have been made in the safety of nuclear plants against accidents and the treatment and containment of radioactive wastes. However, these achievements were partly obtained at the cost of increasing radiation exposure to workers. Therefore, the concern of experts and authorities is now focusing on the trends of occupational exposure in nuclear plants, and no doubt this will continue to be a major issue during the next few years until adequate solutions are found and implemented. The achievement of this goal should be pursued applying the ICRP principle of optimisation. For this purpose, an adequate data base should be assembled, from which the criteria for optimising occupational exposure in the design and operation might be derived. In order to contribute to the above data base, NEA, in co-operation with IAEA, launched in 1978 a study on this subject based on an international enquiry aimed at collecting information on the levels and trends of occupational exposure in nuclear facilities and helping to identify critical groups of workers, critical operations and critical equipment in the plants, as a basis for the study of design and operational procedures improvements. This enquiry supplied a large amount of valuable data and the resulting study is very well advanced. Publication is envisaged for 1980.

Only the most important components of the current NEA programme have been briefly described here, but several other activities are also carried out or planned. These include studies and other forms of international co-operation in fields such as the development of practical examples of application of the ICRP optimisation principle, the preparation of state-of-the-art reports on the radiological and environmental protection aspects of the nuclear fuel cycle facilities, the development of radiation protection criteria for the geologic disposal of high level radioactive wastes, the study of radiation protection problems of the decommissioning of nuclear plants, the international review of nuclear emergency planning criteria and reference levels. This last item, which was already the object of a specialist meeting organised by NEA in 1976, has seen a renewed interest since the Three Mile Island accident in 1979 and is presently being considered by the Agency with a view to launching a programme of international co-operation in this area.

We have tried to give an overview of the evolution of the radiation protection problems and concerns in the last 25 years, in connection with the effort of NEA to assist Member countries in finding a timely and efficient solution to their national problems through a harmonised gathering of efforts and contributions at the international level. This effort has frequently been successful in spite of the obvious difficulties of debating certain sensitive matters at the international level, and we are confident that NEA will continue to be a useful contributor to international co-operation for the constant improvement of radiological protection.

EPIDEMIOLOGY, OCCUPATIONAL HYGIENE & HEALTH PHYSICS

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As a practising Occupational Health Physician of over thirty years standing, I have always believed and have said on many occasions, that occupational exposure to ionising radiations should be regarded in the same way as any other occupational hazard. Unfortunately, because the nuclear age was born in a spectacular way at the end of World War II, ionising radiation is imbued emotionally with almost magical and evil qualities. Half a century before the explosions at Hiroshima and Nagasaki the discoveries of Roentgen and Marie Curie were hailed as major advances. These discoveries have universally benefited mankind and I feel that we should keep emphasising this, rather than the association with the bomb. The emotional impact of releases of radioactive material can be totally disproportional to their real effect; this again was well illustrated in the United States when the incident at the Three Mile Island plant caused widespread anxiety, despite the fact that its effect in terms of physical detriment to the environment and the local population was nil. At the same time in Florida 6 people died, 34 were hospitalised and 2,500 were evacuated following a release of chlorine after a freight train accident. This hardly merited mention even though it was the second such accident in the U.S.A. in 48 hours.

I am not advocating a relaxation of standards, but we do need to keep in perspective the relative risk of nuclear power in comparison with other sources of power and other industrial processes. My task this afternoon is to bring together some views and thoughts about occupational hygiene and health physics, or radiation hygiene which, I think, is a better term. Health physics is really a specialized part of occupational hygiene, but because health physics developed behind a powerful security screen in most countries, there was little if any contact or exchange between occupational hygienists and health physicists in the early days.

The major difference between occupational hygiene and health physics is the high level of concern of radiation protection for the general as well as the occupational environment. This is due partly to the all pervading existence of radiation exposure in our daily lives from natural background and also from routine medical investigative procedures. The long term effects of ionising radiation are limited to an increase in the incidence of cancer and the possibility of an increase in

the incidence of hereditary disease.

It is apparent that the principles of protection are identical, viz. the control of exposure to acceptable limits and the monitoring of the exposed individuals to ensure the efficiency of the safety procedures adopted. Whenever possible alternative less hazardous materials are substituted e.g. man-made mineral fibres for asbestos and tritium based luminising paint for radium based luminisers.

For the purpose of the conference I propose to confine my remarks to the working environment, with only passing reference to the general environment since this will probably be dealt with at a number of the other sessions.

Occupational hygiene is concerned with the assessment of the relationship between the health of occupational groups and their exposure to chemical, physical and biological agents; the study of dose response relationships and the promulgation of dose limits referred to as threshold limit values (TLV). TLV's are defined as time weighted average air concentrations which if not exceeded over an 8 hour period can be expected not to cause unacceptable or permanent adverse effects on the health of the majority of persons exposed. Leaving aside acute effects the philosophy adopted for the protection of workers exposed to such chemical and physical agents are the same as for radiation protection. In many instances the relationship between exposure and disease is specific e.g. lead poisoning; cadmium poisoning or pneumoconiosis. The problem of understanding the dose response relationship so far as the induction of cancer by chemical carcinogens is concerned is similar to that of the induction of cancer by ionising radiation.

For the purpose of discussion I propose to consider asbestos and bladder carcinogens with ionising radiations, since a number of points of interest will become evident.

ASBESTOS

Asbestos is a generic term for all fibrous silicates. Exposure to asbestos dust in sufficiently large quantities will give rise to asbestosis. A disease characterised by progressive fibrosis of the lungs. In the U.K. 60-70% of persons diagnosed as suffering from asbestosis die from bronchogenic carcinoma (lung cancer), this high incidence is almost certainly associated with concurrent cigarette smoking.

Asbestos is also associated with a rare tumour of the pleura and peritoneum called mesothelioma; mesotheliomata are highly malignant and invariably fatal. This cancer is associated with exposure to croccidolite (blue asbestos) and to a lesser extent amosite (brown asbestos). Mesothelioma may follow transient exposure to asbestos and there is frequently a latent interval

of 40 years or more.

Mesothelioma of the pleura is rare in the absence of documented asbestos exposure, the association of disease with exposure to a specific substance eases the problem of deciding whether or not an exposure is significant. Unfortunately the extent of the exposure and in particular the 'dose' to the affected tissue cannot even be estimated. Whilst blue asbestos is no longer imported into the U.K. other forms of asbestos continue to be used in some specific situations, e.g. in brake linings for automobile brakes or lift hoists where there is no alternative to asbestos. Discussions of acceptable risk and cost-benefit analysis are therefore just as important as those concerned with radiation risks. These risk assessments are also valid for general public exposures.

CHEMICAL CARCINOGENESIS OF THE BLADDER

A number of chemical substances used in the dyestuffs and rubber industry, α and β naphthylamine and benzidine have the capacity to cause cancer of the bladder even following exposure to low concentrations of the chemicals and with an accompanying long latent interval. The recognition of the disease and its association with exposure to these chemical substances merits further consideration because of similarities to the problem of assessing the significance of low level radiation exposure.

Carcinoma of the bladder has been increasing in frequency in the U.K. over the last thirty years.

In common with many cancers, bladder cancer shows a marked regional variation within the national pattern in the U.K. The figures show that deaths from bladder cancer in U.K. males were increased from 2,000 in 1958 to 2608 in 1968, an annual increase of 30% in 10 years. Within this increasing 'background' of bladder cancer, epidemiological studies in England and in the U.S.A. were able to demonstrate that appreciable numbers of those who died from bladder cancer had an occupational history that warranted further investigation (Veys 1973).

The recognition of the association of bladder tumours with occupational exposure to specific chemicals in a situation when cancer of the bladder was increasing in frequency demonstrate the important role of epidemiological analysis of exposed populations. I believe this emphasizes the need for obtaining basic epidemiological information about occupationally exposed groups of radiation workers. There is no other way of testing the validity of the current dose limits propounded by ICRP.

IONISING RADIATION

The problem of ionising radiation is that of assessing the association of radiation dose with the development of malignant disease and possible harm to the descendants of those persons exposed. The mechanism of cancer production by asbestos (mesothelioma) and bladder cancer from the chemicals mentioned, appear to be similar to that of ionising radiation, i.e. the probability of the development of the disease is associated with the level of exposure. A "stochastic" effect since once the disease process has started it is totally independent of the "dose" of carcinogen.

Unlike the problems associated with the occupational hygiene of the situations discussed above, workers are exposed to radiation the whole time, from natural background, from potassium 40, from medical procedures and from the small additions due to world wide fall-out from bomb testing in the atmosphere.

If we now consider the significance of continued exposure to low level radiation i.e. at doses of about the ICRP recommended limit of 5 rem per annum, and how this limit was arrived at, there are a number of immediate and obvious differences between radiation dose limits and TLV's.

The ICRP dose limit assumes a linear dose response relationship derived from studies of populations exposed to large single doses of radiation at high dose rate. From these data risk factors have been calculated relating the probability of cancer arising in individual tissues with exposure to units of radiation dose in those tissues. This has been possible because reasonably accurate estimates of the acute dose is possible, unlike any retrospective study of asbestos exposure described previously. It is immediately apparent that these risk factors are probably pessimistic since no account can be taken of the modifying effects of dose rate or dose fractionation.

Radiation exposure is also assumed to have no threshold effect, but in these dose response relationships no account is taken of the effect of the level of dose on the latent period between exposure and the appearance of any disease. It could well be that each rem of dose has an effect, but if it takes 200 years to develop the relationship becomes rather academic.

There is one other factor in radiation control which I would like to mention and that is the sensitivity of the measuring instruments and their specificity. Because accurate measurement of a few molecules of radio active substances or photons of energy can be made it is a great temptation to ascribe some effect to these measurable quantities.

DISCUSSION

Since exposure to ionising radiation causes an increase in the incidence of cancer, and the radiation induced cancer is indistinguishable from cancer due to natural or other causes, a number of obvious practical difficulties present themselves.

It is generallybelieved that some 70% of all cancers are environmental in origin, but less than 4% are due to occupational causes. Where a specific disease is caused by a particular substance such as mesothelioma and asbestos exposure then it can be assumed with some confidence that if exposure to asbestos can be established then cause and effect is apparent.

Again, using experience gained from studying the effects of asbestos, when over 60% of people suffering from asbestosis develop lung cancer, if any individual asbestotic develops this disease, it is reasonable to assume a direct relationship. The fact that the individual has contributed to the disease by smoking cigarettes is an additional consideration.

The level of risk which is acceptable will I feel sure be widely discussed at this conference, but what should be done about the radiation worker who develops cancer, but who has not accrued a dose in excess of the dose limits. Is it necessary to accept that all cancers in an occupationally exposed group of radiation workers are related to radiation exposure? Should TCRP risk factors be used to determine the relationship?

It is vital to establish the validity of the TCRP dose limits and it is only by the study of occupationally exposed workers that this can be achieved.

If an excess of cancer is demonstrated in a group of workers whose exposure to radiation is known and is within the TCRP dose limits, then morally any individual suffering from cancer should be accepted as having a radiation induced disease. Care should be taken to ensure that only radiation workers who are exposed to radiation are included in the group. By the same token if no excess cancers are demonstrable in such a group no individual suffering from cancer should be accepted as a case of radiation induced cancer.

Such an approach has the added merit of being the accepted practice in the assessment of the incidence of cancer in fields other than radiation, and also of course for assessing the acceptability of other occupational hazards such as the dust concentration for the control of pneumoconiosis.

It is apparent that radiation protection practices have influenced occupational hygiene by emphasising the need for control of gaseous and liquid effluents to the general environment, by studies on lung aerodynamics which hitherto had always been concerned with mass inhalation of concentrations of dust in the study of pneumoconiosis

rather than the behaviour of dust particles in the lung.

It is appropriate for those concerned with radiation protection to look at the practice of their colleagues in the safety and hygiene fields. The significance of environmental measurements is tested by epidemiological studies of the exposed population. I would therefore suggest that good epidemiological data are required for assessing the relationship of low level radiation exposure and possible effects on the health of those exposed. It is apparent from the studies of bladder cancer in the rubber and chemical industries that an excess of a particular cancer can be recognised even in the presence of an increasing incidence of bladder cancer generally. Attempts are currently under way in the U.K. to assess radiation exposure levels where the large employing organisations are co-operating with the National Radiological Protection Board to establish a national register for radiation workers.

The desirability of such a requirement was discussed at the IAEA Symposium on The Late Biological Effects of Radiation in March 1978 but there is a danger of attempting to collect too much information.

At the end of the day the significance of environmental measurement in relation to the health of an exposed population must be assessed. It is not enough to calculate artificial relationships based upon mathematical extrapolations, it is vital to obtain good epidemiological studies. It is not necessary to wait 20 years or more. If the present dose limits are unacceptably high then this will become evident in a much shorter period of time during the course of a properly designed epidemiological study.

In summary, it is apparent that radiation protection practices have contributed a great deal to the practice of occupational medicine and occupational hygiene. This is particularly the case in the development of philosophies of protection and in stimulating accurate studies in a number of biological systems. It is suggested that it is now time for those of us who have responsibilities for the radiation protection of workers to accept the desirability for accurate epidemiological assessment of persons exposed at or below the recommended dose limits for radiation.

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N.B. This paper represents the personal views of the author.

"A REVIEW FROM THE REGULATORY POSITION OF THE CONTROL OF OCCUPATIONAL EXPOSURE ASSOCIATED WITH THE FIRST 20 YEARS OF THE UNITED KINGDOM COMMERCIAL NUCLEAR POWER PROGRAMME"

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LICENSED NUCLEAR INSTALLATIONS

The Nuclear Installations Inspectorate (NII) was established in 1959 to implement and administer the licensing and inspection of all nuclear installations in the UK, except those operated by the Crown or the United Kingdom Atomic Energy Authority (UKAEA). The UKAEA was already supplying power from the first of the gas cooled, Magnox type power stations at Calder Hall (1956) and Chapelcross (1958) and was also responsible for the manufacture and processing of nuclear fuel. In 1971, responsibility for the operation of these power stations and for the fuel manufacturing, enrichment and reprocessing plants located at Springfields, Capenhurst and Windscale, respectively, was transferred from the UKAEA to a newly formed Company, British Nuclear Fuels Ltd (BNFL) and consequently these plants became subject to licensing.

When the NII commenced work in 1960, the Central Electricity Generating Board (CEGB) and the South of Scotland Electricity Board (SSES), together, had four gas cooled, Magnox type nuclear power stations under construction with five more in various stages of planning, all of which became subject to licensing. The first of these commenced operations in 1962, and currently the Boards have nine twin reactor Magnox type stations and (since 1976) two twin reactor Advanced Gas Cooled (AGR) type stations in operation. Three other AGR stations are approaching completion.

CONTROL OBJECTIVES

Licensed nuclear installations are regulated through a system of conditions attached to the site licence, which cover all aspects of nuclear safety, including radiological protection. The safety principles against which these installations are assessed have been outlined by Gronow and Lewis (ref 1), a fundamental criterion being that an installation should not cause any person to exceed the exposure limits recommended by the International Commission on Radiological Protection (ICRP); currently those incorporated into the Euratom Directive on Radiological Protection (ref 2). The licensee is required to make arrangements covering safety policy and practice and these are assessed, and in some cases formally approved, by the NII. Control extends over the design, construction, operation, maintenance and eventual decommissioning stages of any licensed nuclear installation, and the introduction of any new plant or process on an existing licensed site.

PRINCIPAL SOURCES OF OCCUPATIONAL EXPOSURE - Power Generation

Annual radiological exposure data associated with the operation of nuclear power stations in the UK are summarised in fig 1. The two main occupational dose components are chronic exposure from plant

background, and chronic and acute task oriented exposures. The former depends on the particular plant design and operating power. Earlier steel pressure vessel designs with heat exchangers outside the main biological shield give backgrounds, which although reduced on later designs, are difficult to eliminate. In recent stations, including the AGR's, this background component is virtually eliminated by enclosing the reactor and boilers in a single pre-stressed concrete pressure vessel. The effect of this design development can be seen in Fig 2 which shows the annual whole body dose equivalent distribution, expressed as a percentage, and averaged over the years 1971-78, for typical stations at each development stage.

The main task oriented exposures are usually associated with workers carrying out routine operations and maintenance of "on-load" refuelling plant and irradiated fuel storage facilities. A number of chronic and acute radiation and contamination exposure control problems have arisen which were not foreseen at the design stage (ref 3). In particular, the longer than anticipated pond storage time of irradiated Magnox fuel prior to despatch to the reprocessing plant continues to pose difficulties in controlling exposure for this group of workers. To-date, however, these operations have not resulted in any significant pattern of internal exposure. For those tasks not directly associated with the fuel handling route, including those from the internal inspection of gas ducts and boilers during off-load maintenance, the dose contribution remains relatively small.

In general, the radiological control procedures applied at nuclear power stations have proved effective in limiting individual occupational exposures. Annual (50mSv) and quarterly (30mSv) whole body dose equivalent limits are seldom exceeded, the majority of workers not exceeding an annual dose of 15mSv (5mSv at pre-stressed concrete pressure vessel stations). Fig 1 shows that although the annual collective whole body dose equivalent has remained sensibly constant since 1971 there has been a steady reduction in the annual average whole body dose equivalent, due primarily to a significant increase in the occupationally exposed workforce (5007 to 9132). A better indication of the efficacy of dose management is the annual collective whole body dose per unit of electrical energy supplied (Sv/Gwh). Fig 1 indicates a steady reduction in this value for CEGB and SSEB stations; much of this reduction is probably attributable to the increase in the unit size of new stations.

PRINCIPAL SOURCES OF OCCUPATIONAL EXPOSURE - Fuel Processing

Fuel processing is carried out by BNFL at its factories at Springfields (fuel element production), Capenhurst (uranium enrichment) and Windscale (irradiated fuel reprocessing). These processes and their associated radiological problems have been discussed in detail by Clarke et al (ref 4). When the Company was licensed in 1971 much of its plant had already operated for over a decade. Although complying with the recommendations of ICRP-9, it was not reasonably practicable to limit the annual whole body dose to 50mSv, particularly at the irradiated fuel reprocessing plant, and control was based on quarterly rather than annual limits, provided that the 5(N-18) lifetime dose limit was not exceeded. Progressive improvements of existing plants and in administrative procedures for controlling

occupational exposures were made by the Company so that in 1977, the radiological protection conditions attached to BNFL site licences were made consistent with those for other licensed sites, and in particular, by limiting the annual whole body dose to 50mSv.

Annual average and collective whole body dose equivalents for the three processing sites are shown in fig 1. This indicates a significant decrease in the average dose, particularly at the irradiated fuel reprocessing plant. The annual collective dose is sensibly constant, with the exception of the reprocessing plant where the occupationally exposed workforce has doubled since 1972 to 9000, partly to deal with increased plant throughput but, also, to reduce the number of persons exceeding 50mSv per annum (fig 3). Some of the resultant increase in annual collective dose between 1971 and 1976 was incurred in the improvement of the plant radiological environment, which in turn should help achieve the longer term objective of reducing the collective dose per unit of plant throughput.

The main potential for internal exposure is at the fuel reprocessing plant, either from chronic or acute intakes of fission products or transuranics (mainly plutonium). Whole body monitoring is used to measure γ emitting fission products and lung retained plutonium, systemic plutonium being assessed by urinalysis. To avoid possible over-exposure, it is the licensee's policy to transfer to non-radiation work any person who receives $>50\%$ of the maximum permissible body burden for any of the higher toxicity transuranic radionuclides.

Although particular groups of occupationally exposed workers can be identified as receiving above average external exposures, in contrast to power station operation, the annual whole body dose equivalent received by maintenance and health physics staff at the reprocessing plant tends to be significantly less than that received by the plant operators.

CONCLUSIONS

This review indicates an improving trend in the radiological exposure pattern, despite increasing power generation, fuel burn-up and processing plant throughput. The initial objective of ensuring that individuals do not exceed the statutory dose limits has largely been achieved. There is still, however, a need to ensure that the collective dose for particular working groups within the fuel cycle represents the practical minimum. To achieve this, optimisation procedures, taking into account all the associated detriments and benefits, may have to be employed. A bank of task oriented exposure data will be an essential prerequisite. The NII, therefore, may need to place greater emphasis on more detailed assessment and recording of such task oriented exposure data.

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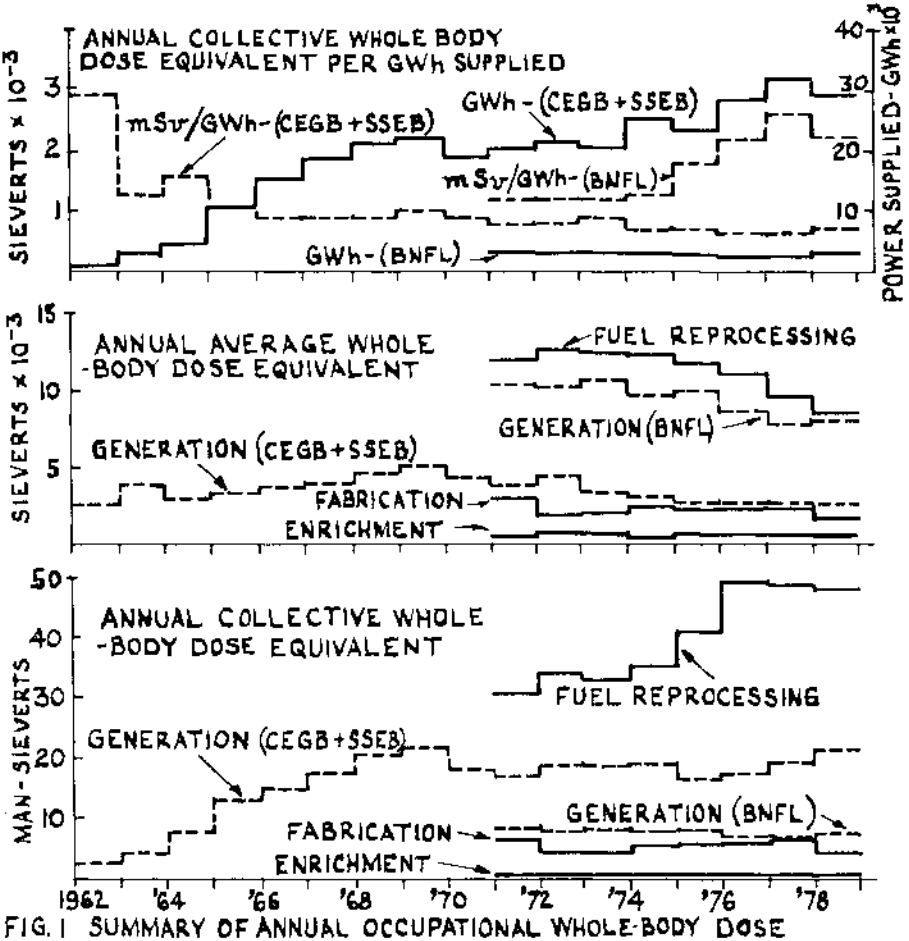

FIG. 1 SUMMARY OF ANNUAL OCCUPATIONAL WHOLE-BODY DOSE FROM EXTERNAL RADIATION.

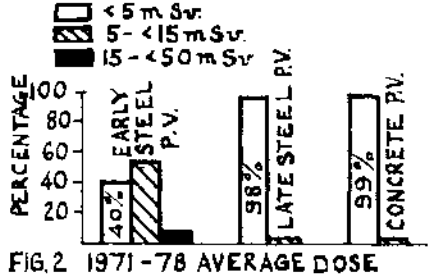

FIG. 2 1971-78 AVERAGE DOSE DISTRIBUTIONS FOR STEEL & CONCRETE PRESSUREVESSELS

FIG. 3 N° OF REPROCESSING PLANT WORKERS > 50 mSv/ANNUM

OCCUPATIONAL DOSE EQUIVALENT LIMITS

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The International Commission on Radiological Protection, in its publication No. 9, adopted by the Commission in 1965, recommended a maximum permissible dose to the critical organs, gonads and red bone-marrow, of 50 mSv in a year. It was acceptable for up to half of this, 30 mSv, to be received in one calendar quarter. The Commission accepted that to provide flexibility it may be necessary to repeat the quarterly exposure each quarter of the year on some occasions but recommended a cumulative limit of 5(N-18) mSv to the age N years. The Commission believed that these cases would occur infrequently. It was also recommended that because any exposure involves a degree of risk, all unnecessary exposures should be eliminated and all doses kept as low as is readily achievable, economic and social considerations being taken into account.

During the last decade new information became available which necessitated a review of the basic recommendations. This review, published as ICRP 26 was adopted by the Commission in January 1977. Certain clarifications were made at the 1978 meeting of the Commission in Stockholm and published within ICRP 28. The system of dose limitation now recommended by ICRP is essentially, that no practices shall be adopted unless their introduction produces a positive net benefit, all exposures shall be kept as low as reasonably achievable, economic and social factors being taken into account, and the dose equivalent to individuals shall not exceed the appropriate limits recommended by the Commission.

In its review of the dose equivalent limit the Commission has found that with the previous annual limit of 50 mSv, the distribution of annual dose equivalents in large occupationally exposed groups has very commonly fitted a log-normal function with an arithmetic mean of about 5 mSv, with very few values approaching the limit. The Commission has also found that the resulting average risk within such occupational groups is comparable with other safe industries and therefore considers that a dose equivalent limit of 50 mSv should be retained. The Commission assumes that workers who are exposed near the dose equivalent limits are unlikely to be so exposed every year, but recognises that some individuals may have a higher than average risk. The Commission recommends too that continued exposure of a considerable proportion of the workers at or near the dose equivalent limits will only be acceptable if a careful cost benefit analysis shows that the resultant risk is justifiable.

The Commission indicated in ICRP 9 that the dose limits were intended for planning design and operation in foreseeable conditions of exposure. In its review in May 1978 the Commission specifically eliminated reference to the use of dose equivalent limits for the purpose of planning.

Legislators and designers must take account of the advice offered by ICRP but nevertheless use it in a practical way. ICRP

offers no advice on the size of the occupational group over which the achievement of an average dose equivalent equal to one tenth of the dose equivalent limit would be acceptable. The development of public and industrial awareness in many countries suggests that perhaps some quantification of acceptable lifetime individual risks is required. In the development of an index of harm (4) ICRP specifically rejects the practicability of controlling the cumulative dose equivalent of workers according to age and sex, regarding the achievement of an adequate standard of protection to be more reliably ensured if a single dose equivalent limit can be used for all workers. The success of this philosophy is entirely dependent on the distribution of dose equivalent within groups of workers remaining unchanged. Furthermore, this philosophy allows individuals a radiation risk comparable with occupational risks in high risk industries such as construction and mining. This radiation risk is additional to conventional risks in the individuals occupation and does not necessarily result in a corresponding benefit to that individual.

This paper considers methods of limiting individual radiation risks by recognising the variation of risk with age at exposure, taking into account both somatic and genetic risks and proposes a simple formula for controlling individual cumulative exposure and hence risk.

VARIATION OF RISK WITH AGE AT EXPOSURE

Two components of radiation risk can be identified, namely genetic and somatic. In both cases dose equivalent received early in life carries a greater risk than that received later. In the case of the genetic component the risk becomes zero after completion of child production. For somatic effects the latent period has the effect of reducing the risk from dose equivalents received later in life. In quantitative terms data on the variation of risk with age is sparse. However, such data as is available has been used by ICRP (4) to produce a curve showing variation of radiation risk with age at exposure, including both genetic and somatic components. The curve, Fig. 4 in (4) may be approximated to a straight line represented as a function of N:

$k(N) = (2.82 - 0.04 25N) \times 10^{-2}$ mSv^{-1} ... where N is age at exposure

A cumulative dose equivalent limit, D(N), expressed as a function of N may be proposed, where D(N) may be a continuous function (e.g. $D(N) = (N-16)^2$ mSv), or have a discontinuity (e.g. $D(N) = (N-16)^2$ mSv with a superimposed over-riding annual limit such as 50 mSv).

If the resulting dose equivalents in years N and N-1 are D_N and D_{N-1} respectively, then the cumulative risk to age N is:

$$10^{-2} \int_n^N (D_N - D_{N-1})(2.82 - 0.0425N)\, dN$$

where n is the age at commencement of occupational exposure.

Figure 1 shows cumulative risks with age for a range of dose limit functions and Figure 2 shows the corresponding cumulative limits of dose equivalent. Table 1 shows the resulting lifetime risks for the same range of cumulative dose equivalent limit functions, expressed as percentages, assuming exposure to age 65.

The risk and cumulative dose equivalent curves are identified as in Table 1. In the case of the cumulative dose equivalent functions 3, 4 and 6 the over-riding limits of 50 mSv, 30 mSv and 20 mSv per year are dominant from ages 44, 54 and 26 respectively. The latter two functions give very little benefit in terms of risk and also lead to very restrictive exposure regimes.

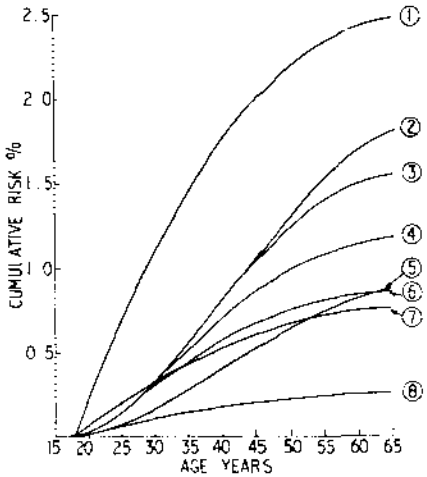

Fig 1 Cumulative risks with age.

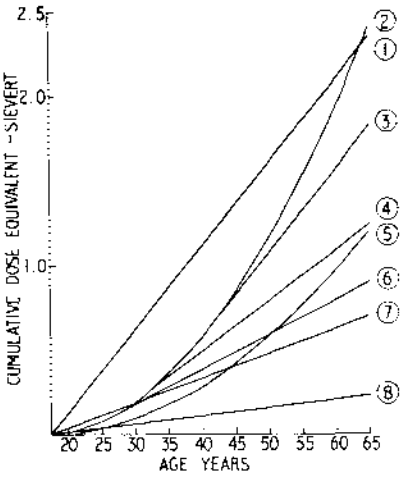

Fig 2 Cumulative dose limits with age.

TABLE 1 Lifetime risks for a range of dose equivalent limit functions.

Dose equivalent limit function	Identity	Lifetime risk (%)
50 mSv per year	①	2.4
$(N-16)^2$ mSv to age N	②	1.7
$(N-16)^2$ mSv with 50 mSv limit	③	1.5
$(N-16)^2$ mSv with 30 mSv limit	④	1.25
$0.5(N-16)^2$ mSv	⑤	0.85
$(N-16)^2$ mSv with 20 mSv limit	⑥	0.8
15 mSv per year	⑦	0.75
5 mSv per year	⑧	0.25

DISCUSSION

It will be seen from Table 1 that the lifetime risk to persons annually exposed to 5 mSv is 0.25%. This represents the average risk within a group of persons if their average exposure is one tenth of the current dose limit of 50 mSv. It is suggested here that it is undesirable for individuals to be exposed to a lifetime risk much greater than five times the average risk. Inspection of Table 1 shows that this can be achieved by limiting individual cumulative dose equivalent to $(N-16)^2$ mSv with a 50 mSv superimposed annual over-

riding limit. If on the other hand the individual risk is limited to three times the average then this is achieved by a cumulative limit of 0.5 $(N-16)^2$ mSv with no individual yearly limit.

The advantages of either scheme are:

i) Retention of the facility to receive dose equivalent of 50 mSv in individual years.
ii) Reduction of individual radiation risks.
iii) Limited dose equivalent from the age of 16 for training purposes.
iv) Higher dose equivalents in the second half of the working life when work skills are fully developed.

It is suggested also that planned special exposures should be limited to the difference between exposures already received and the ceiling given by the cumulative limit without any further need for restriction within that planned exposure.

PRACTICAL APPLICATION

Either scheme may be applied by means of a complete table of permitted cumulative exposure and age. It may in some circumstances be more convenient to limit the average exposure over groups of years. In the latter case the formula $(N-16)^2$ mSv with a 50 mSv annual limit gives almost exactly the same risk as allowing 10 mSv per year from age 16 to 25, 30 mSv per year from age 25 to age 40 and 50 mSv per year above age 40.

CONCLUSIONS

On the basis that the current dose equivalent limit of 50 mSv has been shown to give an average dose equivalent of 5 mSv within a large group of workers, the individual risk within such a group can be limited to no more than three times the average if a cumulative individual limit of 0.5 $(N-16)^2$ mSv is applied or to five times the average if a limit of $(N-16)^2$ mSv with an over-riding annual limit of 50 mSv is applied, where N is the age of the individual.

ACKNOWLEDGEMENTS

The permission from the CEGB to publish this paper is gratefully acknowledged. The views expressed in it are the authors and are not necessarily those of the CEGB. Inspiration and assistance from colleagues is acknowledged, especially Mr. P.F. Heaton and Mr. M.C. Orchard.

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STUDIES OF THE TRANSFER FACTORS OF SR 90 AND CS 137 IN THE FOOD-CHAIN SOIL-PLANT-MILK

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To assess the radiation exposure of the population in the surroundings of nuclear industrial installations with radioactive effluents, calculations of the radionuclide concentrations in the environmental mediums, based on the quantities discharged, must be performed during planning and operation. For these calculations it is necessary to know the transfer factors describing the radionuclide transfer in the foodchains for the ingestion pathway. The plant uptake of a radionuclide from the soil is defined by the transfer factor TF_{SP} as the ratio between the activity concentrations in plant and soil. For the transition plant – milk of the milk pathway the transfer factor TF_{PM} of a radionuclide is the ratio of the activity concentration in milk to the daily intake rate of an animal.

Since the values of the transfer factors recommended for the F.R.G. (1) are based on data from the international literature, they remain to be verified for the ecological conditions prevailing in Germany. This holds especially for Sr 90 and Cs 137 which belong to the relevant radionuclides of the ingestion pathway. Therefore, transfer factors of Sr 90 and Cs 137 were determined by low-level measurements of soil, plant and milk samples. The studies were performed in the surroundings of the site of the planned nuclear waste center at Gorleben.

MATERIALS AND METHODS

In 1978, samples of soil, plant and milk were taken at several locations. The pastures under study had either podsolic or loamy soils. Podsol is a soil type with high contents of sand which is common in the F.R.G. The loamy soils are found in the lowland of the river Elbe. The sampling depth in the soil was 10 cm. The following parameters were determined in soil samples: pH - value (0,1 nKCl), exchangeable Ca and K (Mehlich), glow loss (400 °C ashing), concentration of K and Cs 137 (γ-spectrometry) and Sr 90 concentration after HCl extraction (β-spectrometry of Y 90).

Sampling of plant was performed twice during the vegetat-

ion period on the same pastures, the soil samples had been taken from. Determination of Sr 90 after fusion of the ashes with Na_2CO_3 and K_2CO_3 and of Cs 137 was performed in a similar manner as far the soil samples.

During the outdoor season raw milk samples were collected from 5 farms under study in about monthly intervals. Further, bulk milk of two different collection areas was studied. After drying and ashing of the milk samples Sr 90 and Cs 137 were determined.

RESULTS

Transfer factors TF_{SP} soil-plant were calculated based upon the results obtained from the Sr 90 and Cs 137 determinations in the soil and plant samples. A survey is given in table 1 where the transfer factors are listed according to increasing glow loss of the soil samples. The values of the Sr 90 transfer factor are negatively correlated with the glow loss at the 1 % level. This tendency is due to the positive correlation between the Sr 90 concentration of the soil samples and the glow loss that is significant even at the 0,1 % level.

TABLE 1. Transfer factors TF_{SP} soil-plant for pastures

Posture No.	Soil type	Soil glow-loss (%)	TF_{SP}* Sr 90	Cs 137
1	Podsolic	2,7	0,97	0,086
2	Loamy	6,0	0,46	0,038
3	Podsolic	6,3	0,41	0,108
4	Loamy	7,0	0,40	0,045
5	Podsolic	8,3	0,35	0,094
6	Podsolic	8,3	0,29	0,055
7	Podsolic	12,5	0,27	0,051
8	Loamy	16,2	0,10	0,018

* pCi/kg plant wet weight: pCi/kg dry soil

The glow loss is approximately equal to the organic matter content in the soil. Hence, the cation-exchange capacity and the sorption of Sr 90 at the available exchange sites in soil increases with increasing organic matter content. This may be the cause of the reduction of the Sr 90 uptake by plants. The Sr 90-transfer factors are correlated with the available Ca

content in soil at the 5 % level of significance. No correlations with the other soil parameters were observed.

The values of the Cs 137-transfer factor TF_{SP} were higher in podsol than in loamy soils. Very low values were obtained for posture No. 8 which is situated in the floodplain of the river Elbe. There was a correlation between the transfer factors and the soil parameters only for whole potassium, which was negatively correlated with the values.

Transfer factors TF_{PM} were determined for the transition plant-milk assuming a daily intake for cows of 11 kg dry matter. A survey of the values with the reference to the respective postures is given in table 2.

TABLE 2. Transfer factors TF_{PM} plant-milk

	Associated Pasture No.	$TF_{PM} \cdot 10^{3*}$			
		Sr 90		Cs 137	
		Range	Mean	Range	Mean
Area 1	1;2;8	1,3 - 2,1	1,5	7,0 - 13	9,7
Area 2	2;8	1,1 - 2,4	1,8	6,5 - 11	8,6
Farm 1	3	1,3 - 2,7	1,8	8,5 - 57	32
Farm 2	4	1,4 - 1,7	1,5	2,3 - 5,5	3,8
Farm 3	5	0,7 - 3,2	1,3	6,7 - 11	9,4
Farm 4	6	1,5 - 2,3	1,9	5,3 - 19	11
Farm 5	7	1,4 - 2,0	1,7	5,4 - 30	14

* pCi/l : pCi/daily intake

The mean values of the Sr 90 factors agree well for the different locations. On the other hand, the Cs 137-transfer factors TF_{PM} were found to vary markedly which was due to the great variations of Cs 137 concentrations in the milk. As it has been the case with the transition soil-plant, the highest values were found at farms with sandy soils. Throughout the outdoor season, the lowest factors were determined for farm 2 situated in the lowland of the river Elbe. As shown by sorption studies of different soils using Cs 134 tracer, this may be caused by sorption of Cs 137 on soil particles ingested by the cows together with the feed (2).

DISCUSSION

The determined transfer factors TF_{SP} for the transition soil-plant were found to be dependent on both the soil type and

the soil composition which is in agreement with the literature (3, 4). The values obtained for Sr 90 and Cs 137 were of the same order of magnitude than those determined by lysimeter experiments with podsol of Gorleben (5). Since no discrimination between root uptake and direct contamination of plants by fall out is possible in field studies of Sr 90 and Cs 137, the determined values can be considered the upper limit of the transfer factors TF_{SP}.

The data of the Sr 90-transfer factor TF_{PM} with a mean value of 0,0016 for the transition plant-milk agree well with the value of 0,0014 estimated from the literature (6). Considering the given ecological conditions the recommended factor 0,008 (1) should be doubled.

The determined range of the Cs 137-transfer factor TF_{PM} between 0,0023 and 0,057 is generally higher than the value 0,0071 calculated from the literature (6). The mean value measured at 2 farms during the outdoor season has been found to be higher than the recommended value of 0,012 (1). From the results obtained so far, it is not possible to decide definitely whether the recommended Cs 137-transfer factor TF_{PM} can be verified for the ecological conditions of the surroundings of Gorleben. The studies will be continued for several years in an attempt to support the present data.

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THE RADIATION PROTECTION PROGRAMME ACTIVITIES OF THE WORLD HEALTH ORGANIZATION

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A number of World Health Assembly and the Executive Board resolutions have stressed the World Health Organization (WHO) responsibilities at the international level in respect of protection from radiation hazards, in collaboration with other interested organizations and societies in the radiation field, and in particular the IAEA, UNSCEAR, ICRP, IRPA etc. They have also called "the attention of Member States and Associate Members to the responsibility of their national health authorities in the protection of the population from radiation hazards" and emphasized inter alia the role of WHO in "encouraging and assisting" those authorities "to accept their major role in the public health aspects of radiation from all sources".

RADIATION PROTECTION STANDARDS AND GUIDELINES

The IAEA, WHO, ILO and OECD(NEA) are the four international organizations which have statutory obligations and responsibilities in the field of radiation protection standards. These organizations are now engaged in revising the Basic Safety Standards for Radiation Protection (IAEA Safety Series No. 9, 1967 edition) to implement new ICRP recommendation 26.

A topical seminar on the practical implications of the ICRP recommendations was held in March 1979, by the above-mentioned four organizations, to discuss the practical problems of the implementation of these recommendations. The joint IAEA/WHO Code of Practice on the Basic Requirements for Personnel Monitoring was revised and will be published in 1980. In August 1978 a joint WHO/IAEA/ILO meeting was held to discuss a code of practice on Radiation Protection in Mining and Milling of Radioactive Ores. The resulting document will be published in 1980.

PUBLIC HEALTH ASPECTS OF NUCLEAR POWER

The rapid development of nuclear power in developed countries since the early seventies and, especially, the plans of a number of developing countries to offset the high costs of oil and other fuels through the generation of nuclear energy, will pose new public health problems. Of particular concern is the obvious fact that the operation of nuclear power reactors requires an even higher level of training and competence in health and safety fields than has been

considered adequate in highly developed countries a year ago. The strong diversity of opinions between the advocates of nuclear energy as a necessary alternative to power from fossil fuels and their opponents who wish to avoid nuclear power completely, makes it not only more difficult, but also more important for WHO as the lead international organization in health problems, to present a balanced view of health detriment of nuclear power and its alternatives, and other applications of radiation.

Special attention has been given in the WHO programme to the environmental and health aspects of nuclear energy production. In the joint IAEA/WHO publication Nuclear Power and the Environment, published in 1972, the main principle and philosophy of public health implications of nuclear energy production has been presented, and a statement was made that the nuclear power industry could operate safely for the general public and environment when all technical and control measures are taken and properly executed. To further advise public health authorities on this subject, a report on Health Implications of Nuclear Power Production was published by the WHO European Office in 1977. In the same series of activities, the European Office has held a meeting on the health aspects of transuranium elements, and will organize a meeting on the health implication of the handling of high-level radioactive waste.

The emergence of nuclear power as a significant component of energy systems in developing countries has created the need for extensive training of personnel, particularly of those to hold posts of responsibility in the various aspects of a nuclear programme at all stages of planning and implementation. In these countries, the scarcity of resources and the other pressing priorities of public health will render the task of public health authorities, even though limited to a few key ones, particularly important and at the same time difficult.

To assist these public health authorities, WHO is planning an interregional training course on public health aspects of national nuclear power programmes.

Radiation accidents are of major concern to national and international authorities in relation to nuclear power development and the increasing use of radiation in industry, agriculture and medicine. To cover this aspect, appropriate activities have been developed in close collaboration with the IAEA, ILO and FAO. In 1978, the IAEA/WHO manual "Early Treatment in Radiation Accidents" and a report on the Treatment of Incorporated Transuranium Elements have been published.

WHO has been running an extensive study on biological indicators of radiation injury, including the use of chromosome aberrations. Investigations performed in various countries have indicated the usefulness of this method for biological evaluation of radiation dose, by scoring the chromosome aberrations in lymphocytes of accidentally-irradiated persons.

The question of diagnosis and treatment of internal and external accidental exposure of persons has been dealt with extensively in terms of manuals and scientific meetings. Moreover, WHO is playing

a prominent role in this connection because of its traditional competence and responsibilities in preventive and curative medical domaines.

To extend the service to member states WHO are now planning to establish three WHO collaborating centres on human radiopathology to serve in actual cases of human radiation injuries:

- one in Paris (Institute Curie, Department of Radiation Protection) for Member States in the African, Eastern Mediterranean and European Regions
- one in Oak Ridge University Medical Research Center; for the Americas
- one centre for Western Pacific and South East Asian Regions.

The terms of reference of centres would be:

(i) to serve as focal points for advice and possible medical care in cases of human radiation injuries;
(ii) to facilitate when necessary the establishment of a network of equipment and specialized staff in human radiopathology;
(iii) to assist in the establishment of medical emergency plans for the event of large-scale radiation accidents;
(iv) to develop and carry out coordinated studies on human radiopathology and epidemiological studies that may be appropriate;
(v) to assist in the preparation of relevant documents and guidelines.

In addition, revision of the IAEA/FAO/ILO/UNDRO document on the mutual emergency assistance in radiation accidents is in preparation to provide information on the assistance that Member States might be able to make available at the request of another country.

SURVEILLANCE AND CONTROL OF ENVIRONMENTAL RADIOACTIVITY, ASSESSMENT OF POPULATION EXPOSURE AND HEALTH EFFECTS

Periodical reports prepared by 4 WHO collaborating centres in cooperation with 27 laboratories in 19 countries, are being provided to the Regions and Member States, which include the world data on radioactivity in air, water and food. Intercomparison of measurements of radioactivity in samples of milk and bones were maintained by collaborating centres, to improve the accuracy and comparability of data from various countries. Information on strontium-90 in human bones obtained from tropical area countries were made available to UNSCEAR and participating countries.

A WHO European working group has considered the problem of the acceptable levels of radionuclides in drinking water for the revision of WHO International Standards for Drinking Water. The report of the working group makes few significant changes in comparison with the previously published WHO International Standards. A significant departure from previous practice is the explicit statement that "Where these levels are exceeded, it is recommended that the competent authorities be required to decide what further action, if any, is necessary".

The radiological analysis of fresh water is dealt with in two chapters of a three-volume handbook on "Examination of Water for Pollution Control", sponsored by the WHO European Office and due for publication. Radiation exposure from naturally occuring Radon as one of the pollutants of Indoor Air has been discussed by a working group in Bilthoven in April 1979.

In the assessment and control of health risks from radiation exposure, the Organization also plans to study the exposure of the population due to radiation-emitting consumer products, technologically enhanced natural radiation, building materials etc.

WHO will assist the public health authorities in keeping the public currently informed on the likely health consequences of various uses of radiation. Promotion of education and training of medical personnel in radiation hygiene and protection is an important task of WHO in establishing contacts with populations in order to assure adequate health protection.

NONIONIZING RADIATION

The very high potential for cancer and genetic damage from ionizing radiation in comparison to a high tolerance for less energetic forms of radiation have led most health officials to disregard non-ionizing radiation (NIR). However, NIR, which ranges from ultraviolet through visible light, infrared and microwaves to radiofrequency radiation, as well as electomagnetic fields, is the result of a fast developing technology with a growing potential for affecting public health. To draw the attention of public health to this problem, in 1972 a special chapter on non-ionizing radiation was included in WHO monograph "Health hazards of the human environment"; and an International symposium on biological effects and health hazards of microwave radiation was jointly organised by WHO and the health authorities of the USA and Poland in 1973, with the participation of an IRPA representative. The WHO European Office has organized a series of meetings on various aspects of NIR which will result in the publication of a manual on NIR protection. The WHO/UNEP/IRPA environmental health criteria document on ultraviolet radiation was published in 1979, and the document on microwave and radiofrequency radiation has been presented to the publisher.

New environmental health criteria documents on lasers and ultrasound are at present in preparation, as joint activities with IRPA.

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WHO resources for all radiation protection activities are limited but considerable support has been received from WHO collaborating centres and national authorities, e.g. Belgium's and the Federal Republic of Germany's contributions.

Most of these programmes are being carried out in close cooperation with other international organizations and bodies such as the International Atomic Energy Agency, the International Labour Organisation, the United Nations Scientific Committee on the Effects of Atomic Radiation, the International Commission on Radiological Protection and the International Radiation Protection Association.

ALPHABETICAL INDEX