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Chapter 16

RADIATION PROTECTION: PHILOSOPHY, RECOMMENDATIONS AND PRACTICE*

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Abstract

The philosophy developed by the International Commission on Radiological Protection for the control of human radiation exposure will be described. The application of the ICRP recommendations to the authorization and control of the disposal of radioactive wastes to the sea will be discussed in the context of the practice in the United Kingdom.

16.1. Introduction

Ionizing radiation, in the form of cosmic rays, and α-, β- and γ-rays from cosmogenic and primordial radionuclides, have been an environmental factor on the earth since its formation. Because there is a lack of any significant interactive response from the five human senses, ionizing radiations were not, however, recognised until the development of appropriate detection instruments and media in the latter part of the 19th century. This resulted in the discovery of X-rays by Röntgen (1895) and the natural radioactivity of uranium minerals by Bequerel (1896). There followed a rapid development in the application of X-rays for medical diagnosis and, somewhat later, the use of γ-rays from Ra-226 and its decay products for therapy. In the course of these developments the hazardous side-effects of radiation exposure became apparent, leading to the establishment in 1928 of the International X-ray and Radium Protection Committee. With the subsequent discovery of nuclear fission and the expansion of the applications of radiation and radionuclides outside the medical field, the Committee was reorganized as the International Commission on Radiological Protection (ICRP) in 1950.

The ICRP operates in an independent, advisory capacity and produces considered recommendations that are intended to permit an acceptable balance between the conduct of activities that might result in exposure to ionizing radiations, and the protection of both employees and the general public.

In the early days when most exposure was of occupational origin, it was generally considered that protection objectives had been adequately met if radiation exposures were kept below recommended limits the main purpose of which was to prevent the occurrence of directly observable, non-malignant effects. Over recent decades, as radiation applications have expanded and nuclear energy sources for electricity generation have been developed, there has been a significant shift in emphasis to a more detailed system of radiological control in which the recommended dose limits are the last in a series of defences intended mainly to restrict the incidence of both cancer and hereditary effects that might be induced by increased exposure to radiation.

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16.2. The development of the ICRP philosophy

16.2.1. Interactions of radiation with tissue

The basic interaction of radiation with tissue (or any other medium) is the production of ionization - that is, the separation of orbital electrons from atoms and molecules to form a variety of positive and negative ions. These ions are very unstable and in the cellular environment rapidly become involved in a variety of complex physicochemical reactions, some of which result in damage to biologically-important molecules (e.g. DNA, enzymes, components of intracellular membranes etc). Depending on the degree to which such damage may be correctly repaired the radiation exposure may either:

- kill cells or so damage them that they are prevented from undergoing the next cycle, or at most, a few further cycles of cell division; or
- result in viable but modified cells.

For the first case, most tissues and organs can survive the loss of many cells without any significant loss of function, and, therefore, with no obvious harm. Above some threshold of radiation exposure, however, sufficient cells will be lost or terminally damaged for harm to become apparent, and with increasing dose the probability of its appearance rapidly rises to unity and the severity of harm increases. Although damage to, and death of, individual cells is random, the requirement for a large number of cells to be lost before harm becomes apparent means that the response is of a DETERMINISTIC nature and the dose-response relationship is sigmoid (see schematic example in Fig 1.). Some examples of deterministic responses are radiation burns (skin erythema), opacification of the lens of the eye and temporary or permanent loss of fertility. Responses with a (relatively high) threshold for apparent damage, principally skin erythema, formed the basis of the early recommendations for dose limits.

For the second case, the modified, but viable, irradiated cell might become the parent of a clone of somatic cells which, after a variable latent period, could become malignant. Where such a modified viable cell is produced in gametogenic tissue (spermatogonia or oogonia) the damage may be expressed in subsequent offspring, that is, it is HEREDITARY. The severity of either outcome, that is, cancer or congenital disease, is not dependent on the dose. Because such outcomes arise from a single, randomly modified cell, and because the probability of production of such cells increases with dose without any apparent threshold, this kind of response is termed STOCHASTIC and the incidence is approximately proportional to (accumulated) dose at low doses (see schematic example of a dose-response relationship in Fig 2.). In principal, it is clear that any dose, however small, entails some (very small) degree of risk and it is for this reason that the recommended dose limits have become the last line of defence in a more sophisticated protection system in recent decades.

16.2.2. Dosimetric quantities

The ICRP carries out a continuing review of the available information on the biological effects of radiation in man, the physiological and metabolic behaviour of radionuclides taken into the body by inhalation and ingestion, and models for determining the radiation dose to organs and tissues from incorporated radionuclides. This work is consolidated into comprehensive recommendations for the practice of radiological protection

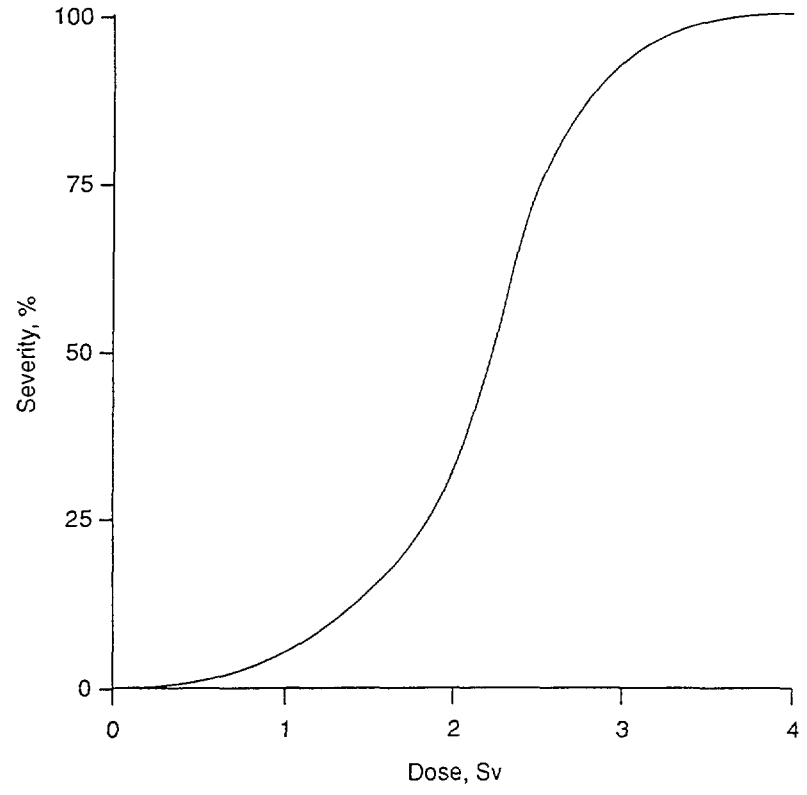

FIG. 1. A schematic sigmoid dose-response relationship for deterministic radiation effects.

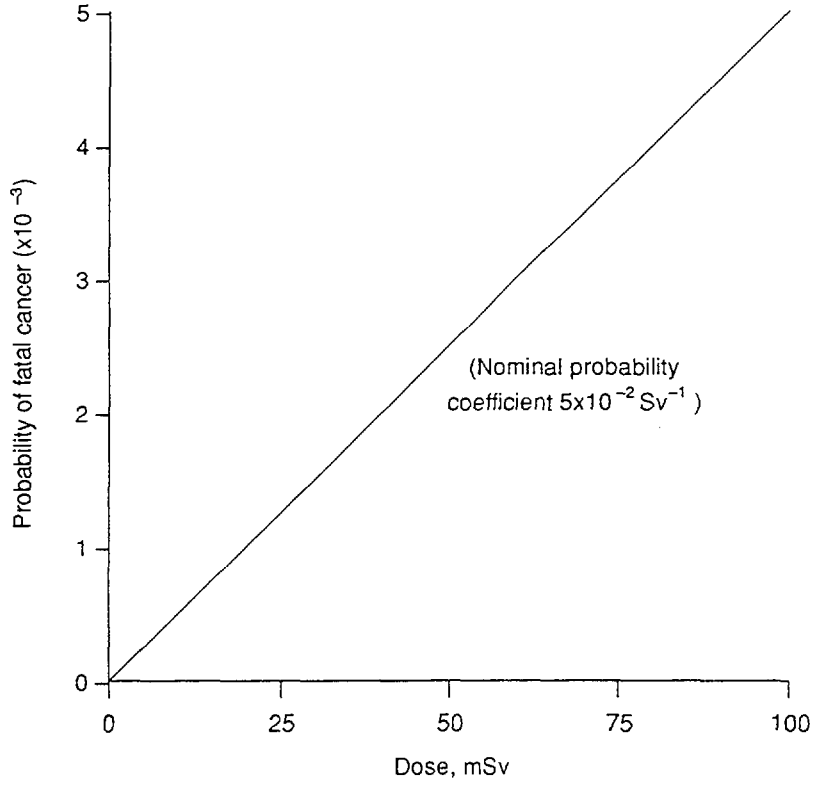

FIG. 2. A schematic linear dose-response relationship for stochastic radiation effects.

both in the work place and for the wider general community. The greater part of the assessment of human radiation exposures following environmental contamination from either authorised radioactive waste disposal or accidental radionuclide releases has been based on the recommendations contained in ICRP Publications 26 and 30 [1-6]. The available information relating to the Sellafield discharges to the northeast Irish Sea and the dumping of packaged low-level solid radioactive waste into the deep Atlantic Ocean falls largely (but not entirely) into this category.

The ICRP have, however, recently published a new set of comprehensive recommendations (ICRP Publication 60) [7] that have led to changes in both the dosimetric terminology and the assessment of the biologically effective radiation dose i.e. the quantity which correlates with the probability of occurrence of stochastic radiation effects. As yet, all the detailed data necessary for the practical application of these new recommendations (particularly the dose per unit intake factors (dose coefficients)) have not been published by the ICRP (due as ICRP Publication 67 in late 1994). Interim advice on dose coefficients, based on ICRP 60, has been provided in the UK by the National Radiological Protection Board [8] and has been applied in parallel with the ICRP 30 data [9-11].

Before discussing the recommended limits on radiation exposure, it is necessary to consider the dosimetric quantities that are employed in radiation protection. Although the basic process of ionization is discontinuous and randomly distributed in the exposed tissue, it is found empirically that the total energy deposited by the radiation in producing ionization correlates reasonably well with the biological response when due account is taken of such factors as radiation type and dose rate.

The fundamental measure of radiation exposure is:

$$\text{ABSORBED DOSE, } D_T = \frac{\text{Energy absorbed in tissue}}{\text{Mass of tissue}};$$

the unit (SI) is the joule per kilogram and it has been given the special name gray (Gy).

Although the formal mathematical definition of absorbed dose allows it to be specified at a point, it is usual, in practice, to use the average value for the whole tissue or organ.

The magnitude of the biological response is found to depend not only on the absorbed dose, but also on the nature and energy of the radiation concerned (the radiation quality). In radiobiological studies the differential response produced by two radiations of different quality (R_1 and R_2) is compared by means of the quantity RELATIVE BIOLOGICAL EFFECTIVENESS (RBE) defined as:

> the inverse ratio of the absorbed doses of the respective radiations required to produce identical biological effects; that is:

$$\text{RBE } (R_1 \text{ compared with } R_2) = \frac{D(R_2)}{D(R_1)}$$

For the purposes of radiological protection it is the probability of a stochastic response (principally tumour induction) that is of importance, and the influence of radiation quality has

TABLE I. RADIATION WEIGHTING FACTORS

Radiation Type	w_R
X-ray and γ-ray photons all energies	1
Recoil electrons and β-particles, all energies	1
α-particles	20

been taken into account through the introduction of an additional dosimetric quantity, the EQUIVALENT DOSE, H_T, that correlates better with this biological response.

The equivalent dose is defined as:

$$H_T = \Sigma w_R . D_{T,R}$$

where $D_{T,R}$ is the absorbed dose averaged over the tissue or organ T due to radiation type R;

w_R is the radiation weighting factor appropriate to the type and energy of the radiation incident on the tissue; and

the summation is over all radiation types.

The recommended values of w_R are given in Table I. H_T is strictly a dose and the unit (SI) remains the joule per kilogram but it is given the special name sievert (Sv). The equivalent doses from different radiations which may be incident on the tissue from external sources (e.g. γ-rays) or emitted by radionuclides incorporated in the tissue (e.g. α- or β-particles) may then be simply summed. The recommended values of w_R relate to tumour induction (stochastic) in humans; for deterministic effects lower values of w_R would be more appropriate.

16.2.3. The influence of differential tissue response

The sensitivity of different tissues to the induction of tumours following a given equivalent dose is also found to be variable. To provide a uniform basis for assessing the probability of stochastic effects either from a combination of different equivalent doses to different organs or from uniform whole body irradiation, TISSUE WEIGHTING FACTORS, w_T, have been introduced which represent the proportion of the stochastic risk resulting from the irradiation of tissue T when the whole body is uniformly exposed. The tissue weighted equivalent dose is called the EFFECTIVE DOSE, E, and is defined as:

$$E = \Sigma w_T . H_T$$

where w_T is the weighting factor for tissue T;

H_T is the equivalent dose in tissue or organ T: and

the summation is over all tissues.

The unit (SI) is the joule per kilogram and it also retains the special name sievert (Sv).

TABLE II. TISSUE WEIGHTING FACTORS

Tissue or organ	w_T	
Gonads	0.20	⎫
Red bone marrow	0.12	⎪
Colon	0.12	⎪
Lung	0.12	⎪ Sum
Stomach	0.12	⎪ for
Bladder	0.05	⎪ the
Breast	0.05	⎬ whole
Liver	0.05	⎪ body
Oesophagus	0.05	⎪ equals
Thyroid	0.05	⎪ 1.00
Skin	0.01	⎪
Bone Surface	0.01	⎪
Remainder	0.05	⎭

Because the values of w_R (radiation weighting factor) have been specifically defined independently of organ or tissue, and because the w_T have been specifically defined to be independent of radiation quality, the effective dose is also given by:

$$E = \Sigma w_T . \Sigma w_R . D_{T,R}$$

where the summation is over all tissues and all radiation types, respectively.

The values of the tissue weighting factors are given in Table II [7]. It can be seen that not all tissues or organs are specifically considered; there is a "remainder" tissue for which $w_T = 0.05$. For the purposes of calculation in a practical application, the "remainder" tissue consists of: adrenals, brain, upper large intestine, small intestine, kidney, muscle, pancreas, spleen, thymus and uterus. In the exceptional case that a single one of the remainder tissues or organs receives an equivalent dose greater than that for any of the tissues for which w_T has been specified, a weighting factor of 0.025 should be applied to that tissue or organ and a weighting factor of 0.025 to the average equivalent dose to the rest of the remainder as defined above. It is also apparent that the normalisation of the total w_T to unity permits the simple addition of the effective doses estimated for any combination of radiation exposures to internal sources in tissues and organs and exposures to either uniform or non-uniform external radiation fields. This, then, provides a single value of effective dose for the assessment of the total probability of a stochastic effect in an exposed person and, more importantly, for comparison with recommended dose limits which are, therefore, effectively limits on the risk of harm.

16.2.4. Subsidiary dosimetric quantities

If a radionuclide is ingested as contaminated food or water or inhaled as an aerosol it may be accumulated in the body, often in a specific tissue or organ, e.g. iodine in the thyroid, strontium in bone etc, and continue to expose the tissue for a period of time. The

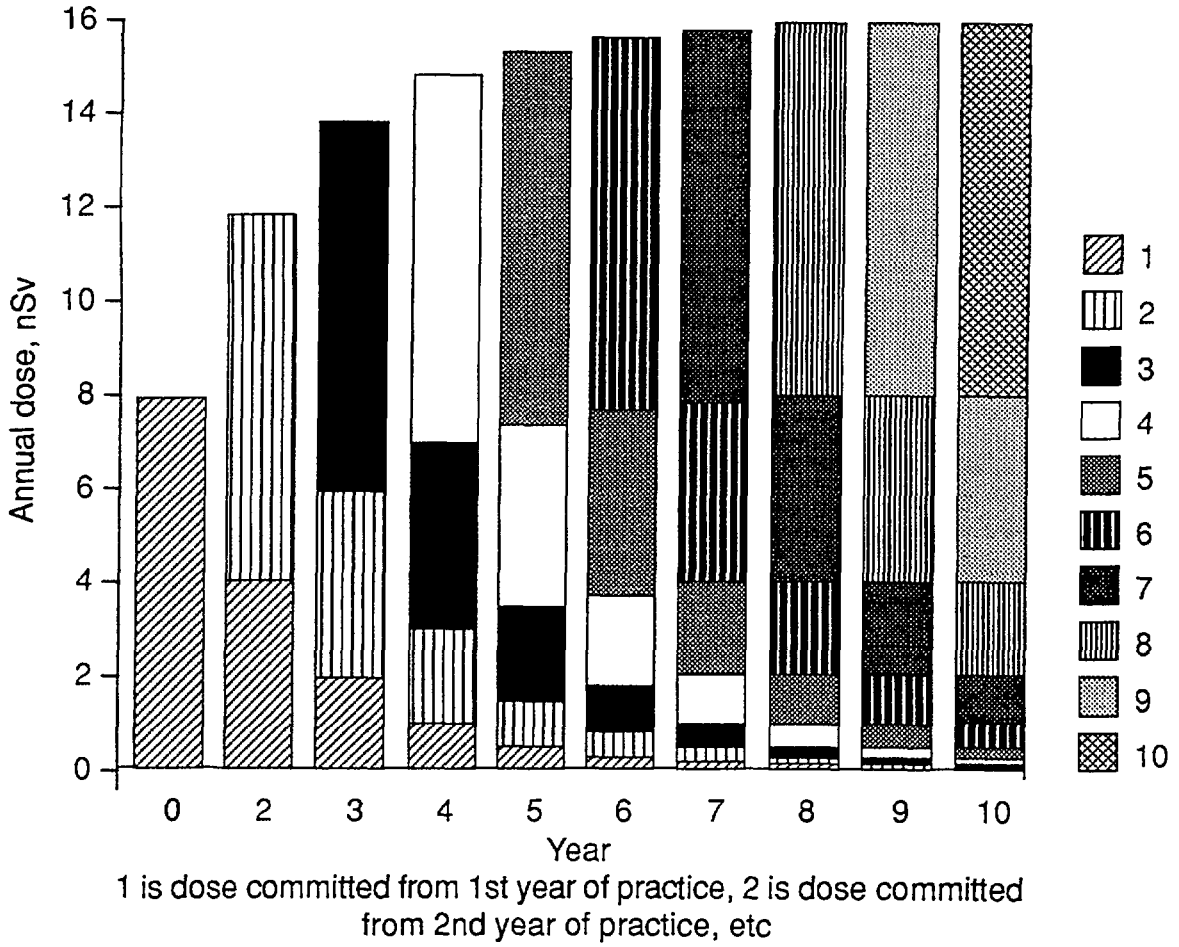

FIG. 3. The annual effective dose from constant annual intakes of a radionuclide; note that, if the intake is both constant and continuous, the equilibrium annual dose rate is equal to the committed effective dose from one year's intake.

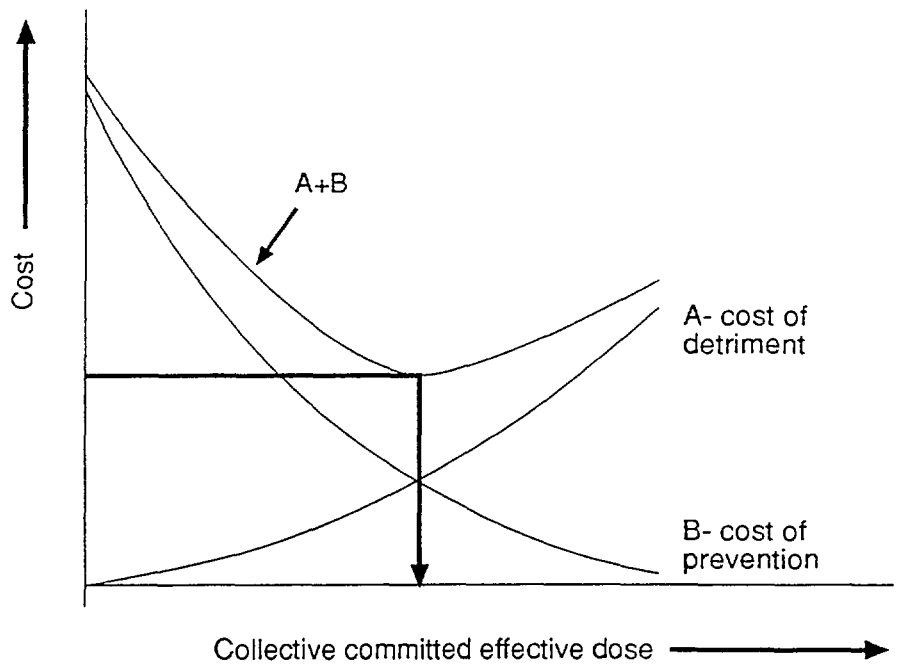

FIG. 4. A schematic example of the principle of optimization through the minimization of the aggregate costs of radiation protection and the residual detriment.

COMMITTED EQUIVALENT DOSE is the time integral of the equivalent dose rate to the tissue following the intake of the radionuclide(s), that is:

$$H_T(t) = \int_t H_T dt$$

where t is the integration time in years following the intake. If t is not specified, it is taken to be 50 years for an adult and the time from intake to 70 years of age for children.

The COMMITTED EFFECTIVE DOSE, E(t) can be defined in an exactly similar manner.

For a single intake, the equivalent dose rate generally declines with time at a rate which depends on the combination of the radioactive half-life and the loss rate of the radionuclide from the tissue. If the intake continues at a constant rate per year it is simple to demonstrate that the eventual equilibrium dose in one year is equal to the committed effective dose for one year's intake [Fig 3].

The ICRP provides compilations of dose coefficients (Sv Bq^{-1}) which permit the estimation of the committed effective dose with due allowance for radioactive half-life and metabolic processes.

All the above dosimetric quantities relate to the exposure of individuals. Additional quantities are required to describe the exposure of groups or populations by taking account of the number of people exposed to a particular source. The sum of the individual equivalent doses (or effective doses) across a group gives the COLLECTIVE EQUIVALENT DOSE, S_T, relating to a specific tissue, or the COLLECTIVE EFFECTIVE DOSE, S; both in units of man-sievert.

16.2.5. The ICRP system of radiological protection

The objective of radiation protection is not to prevent absolutely all exposure to radiation, rather, it is to permit activities which entail radiation exposure while reducing the consequent risks to an acceptable level. The system developed (and still being evolved) by the ICRP has the following three main principles:

(a) JUSTIFICATION OF PRACTICE
No practice shall be adopted unless its introduction produces a positive net benefit.

This means that the activity which entails radiation exposure must produce sufficient tangible benefit to the workers and/or society in general to offset the total harm which would be experienced. The total harm is encapsulated in the concept of DETRIMENT and includes the probability of fatal cancer, the probability of non-fatal cancer weighted for the severity of the illness and distress caused, and hereditary effects over all subsequent generations again weighted for severity. It is implicit within the concept of detriment that it can be extended to include harm which is not related to health e.g. the necessity to restrict access to particular areas, or limit the use of food products normally harvested from wild populations in the natural environment.

(b) OPTIMIZATION OF PROTECTION
All exposures shall be kept as low as reasonably achievable
(**ALARA**), economic and social factors being taken into account.

The individual doses and the number of people exposed should be reduced as far as reasonably possible. Given that the benefits of an activity might not necessarily be received by those experiencing the radiation exposure - particularly in the case of exposure received from an environment contaminated by waste disposal - it may be considered advisable to apply a source-related DOSE CONSTRAINT (within the overall dose limits) on the dose to reduce the extent of the inequity. (The dose constraint is not to be confused with any prescriptive dose limit that might be imposed, by the regulatory authorities, on the exposure resulting from the source.) Optimization of protection is achieved by minimizing the aggregated cost of radiation protection and the residual detriment [see Fig 4]. At low levels of resource commitment to radiation protection (e.g. minimal treatment of a radioactive waste stream discharged to the environment) the total detriment experienced by the exposed population would be relatively high and, assuming that an economic value can be assigned to the detriment, the cost would be high. As increasing resources are devoted to protection (effectively reducing the consequent collective effective dose, S) the detriment is reduced. Eventually, however, this process becomes less effective and higher expenditure achieves very little additional reduction in exposure. At some intermediate value of collective dose the aggregate costs of protection and detriment are at a minimum and it is at this point that protection is said to be optimized.

(c) INDIVIDUAL DOSE LIMITS
The effective dose to individuals shall not exceed
the limits recommended by the Commission

These are aimed at ensuring that no individual is exposed to radiation risks that might be judged to be unacceptable from any combination of controllable sources in normal circumstances. They represent the last line of defence in the system of radiation protection and compliance with dose limits alone does not represent a sufficient degree of control of the radiation exposure.

16.2.6. Dose limits

The choice of a DOSE LIMIT - in practice, a limit on the sum of the separate effective doses which may be experienced by members of a given population from a number of sources that are under regulatory control - is not a simple procedure. For the general public, the ICRP [7] have employed two approaches:

1. The consequences of continued additional exposure at annual effective doses in the range 1-5mSv have been assessed in terms of a number of quantifiable attributes which are indicative of detriment, i.e:

   - the lifetime attributable probability of death (i.e. from radiation-induced cancer);
   - the time lost if the attributable death occurs;
   - the reduction in life expectancy;

- the annual (time-dependent) distribution of the attributable probability of death; and
- the increase in the age specific mortality rate (i.e. the increase in the probability of dying in a year at any age, conditional upon reaching that age).

All these factors relate to mortality. In addition, the Commission allowed for the morbidity, weighted for severity, associated with non-fatal cancer and hereditary disorders in estimating the aggregate detriment. The Commission concluded that the marginal changes in the attributes and, hence, the increase in aggregate detriment incurred by lifetime exposure at 1mSv a^{-1} might be just tolerable.

2. The alternative approach was to consider the magnitude of the natural background. Excluding the very variable contribution due to radon exposure in homes, the background dose varies from about 1mSv a^{-1} to greater than 2mSv a^{-1} depending on altitude (increased cosmic radiation) and the local geological environment (increased exposure to terrestrial gamma-radiation). Although the natural background radiation exposure may not be harmless, the spatial variation can hardly be said to be unacceptable and the Commission again concluded that an additional lifetime exposure at an effective dose rate of 1mSv a^{-1} might be just tolerable.

On the basis of these considerations the Commission recommended that the effective dose limit for a member of the general public for all sources of radiation exposure which are subject to institutional control (i.e. excluding the natural background and exposure in the course of medical attention) should be 1mSv a^{-1} [7]. The requirement that all doses should be as low as reasonably achievable and the empirical observation that in an exposed population the distribution of doses is negatively skewed should mean that the dose limit is rarely approached. It is accepted, however, that variations in the environmental pathways leading to human exposure and in the effectiveness of control measures at the source may result in transient increases in exposure above the limit. In these special circumstances, a higher value of effective dose could be allowed in a single year provided that the average over an inclusive period of five consecutive years does not exceed the 1mSv a^{-1} limit.

To ensure that no particular tissue or organ receives a dose which could lead eventually to the appearance of deterministic effects an over-riding annual equivalent dose limit of 50mSv is recommended. Application of the tissue weighting factors (w_T) and the 1mSv a^{-1} on committed effective dose would generally prevent deterministic effects in the majority of tissues, but additional annual limits of 15mSv for the lens of the eye and 50mSv for the skin (average over 1cm^2 regardless of the area exposed) are recommended [7].

16.3. Practical application of the protection system

16.3.1. Introduction

Human activities may expose members of the general public to a variety of sources of radiation additional to the (unmodified) natural background (assumed to be unavoidable, although remedial action is being taken, where it can be justified, on radon exposure in homes) and exposures during medical treatment (assumed, by definition, to have a large net benefit to the patient). In practice, the most likely additional sources derive from the disposal of radioactive wastes to the environment. Because of its large volume and the natural

processes of dispersion and dilution the world ocean has long been considered a convenient repository for wastes and in recent decades these have included packaged solid low-level radioactive wastes dumped into the deep (and not so deep) sea and discharges of low-level liquid effluents into coastal waters.

To apply the system of radiological protection developed by the ICRP it is necessary to make pre-operational assessments of the potential individual, and collective, effective doses arising in the human population as a consequence of the waste disposal practice.

An assessment of the individual dose is required to ensure that no person is exposed above the appropriate dose limit. It must, therefore, take account of all other pre-existing exposures from human activities as well as the particular practice under consideration and it is the sum total of exposures which must be constrained within the appropriate dose limit. Thus, if the total dose from pre-existing sources amounts to 80% of the relevant limit, it is clear that only 20% is available for allocation to the specific practice being considered (assuming, unreasonably, that there will be no additional interacting sources of exposure in the future).

The assessment of the collective effective dose is used to evaluate the justification of the practice and perform the optimisation of the radiation protection controls to be applied; it is clear that this process is independent of sources of exposure arising from other activities.

In the particular case of liquid radioactive waste disposal into coastal waters the exposure of individuals depends on a very large number of factors. These may be grouped into:

(a) the behaviour of the radionuclides in the marine environment;
(b) the behaviour of the individuals and their use of the environment, and
(c) the biological characteristics of the individuals.

It is clearly not possible to evaluate all these variables for all the people potentially at risk from exposure. It is usually possible, however, to carry out a qualitative assessment which will allow identification of one or more sub-groups of the population whose broadly similar characteristics would combine to cause them to experience the highest radiation exposure. The exposure of these sub-groups, identified as CRITICAL GROUPS, can then be used as an indication of the probable upper limit of the doses likely to be experienced from the proposed practice. Where the practice is expected to continue for a number of years, the potential annual exposure may also increase with time (depending on the environmental behaviour of the radionuclides and their half-lives) and it is the maximum dose which must be compared with the relevant dose limit.

The great complexity involved in assessing the collective effective dose over a whole population (factors (a)-(c) above) means that generalisations, simplifications and approximations are inevitable. This is particularly true when individuals in a large population (millions of persons) may each receive very low doses. These uncertainties must be acknowledged when the data are used for the purpose of justification and optimization.

It is important to note that earlier assessments of disposal practices cannot be judged against current recommendations, in part because past experience has been the basis of current and continuing development.

16.3.2. Practical application in the United Kingdom

In the UK, the general philosophy and objectives of the system of regulation are set out in the Radioactive Substances Act (1993).

AUTHORISATION
No disposal of radioactive waste may be made without a formal authorization, issued by a government department (the Department of the Environment [with the endorsement of the Ministry of Agriculture, Fisheries and Food]), which requires consideration of:

- the ICRP criteria, i.e.:

  JUSTIFICATION in terms of the total practice including the benefits of the processes generating the waste;
  OPTIMIZATION with recognition of the requirement that exposures be as low as reasonably achievable through the application of BEST PRACTICABLE MEANS for waste reduction; and
  the application of the relevant DOSE LIMITS.

- in the case of disposals to the environment that might have transboundary implications, the opinion of the Commission of the European Union (ARTICLE 37).

For disposals to the aquatic environment, the terms of the authorization set out:

- the controls to be applied to reduce the quantities of radionuclides to be discharged: this amounts to an identification of "best practicable means";

- numerical limits on the quantities of different radionuclides that may be discharged; and

- a requirement for specified monitoring of both the effluent and the receiving environment to demonstrate compliance with the numerical discharge limits and the relevant dose limits respectively.

Conformity with the terms of the authorization is overseen by:

- independent INSPECTION of the management of all aspects of the nuclear site, but with particular reference to the waste management function;

- check analyses of effluent samples; and

- independent ENVIRONMENTAL MONITORING AND ASSESSMENT of the consequent exposure of both the critical group(s), for compliance with the

prescribed dose limits, and the population (collective dose), for confirmation that optimization of radiation protection has been achieved.

All of these activities are supported by a continuing programme of RESEARCH to ensure that authorizations of radioactive waste disposal are based upon the best possible scientific understanding of the behaviour of radionuclides in the environment both in the short-term and, for the long-lived isotopes, the long term.

In the UK there is a number of sites on the coast that are authorized to discharge low-level, liquid radioactive wastes into the sea. These include:

- the Sellafield nuclear fuel reprocessing plant;
- the nuclear fuel fabrication plant at Springfields;
- the power stations operated by Nuclear Electric plc and Scottish Nuclear Ltd;
- bases for nuclear powered submarines; and
- the sites operated by Amersham International for the production of radiopharmaceuticals, radioisotope sources, radiochemicals for research etc. [Figure 5].

Of these sites, the Sellafield fuel reprocessing plant is the most significant in terms of the period of operation (1952-1996) and the range and quantities of radionuclides discharged. It provides, therefore, a very informative case study of the application of the radiological control system and this will be discussed in detail; results for other sites are included in the annual monitoring report [10-11].

16.3.3. Pre-operational assessment

To provide the basis for setting an authorization and carrying out the process of optimization of radiation protection, it is necessary to develop a model of the pathways connecting the radionuclides in the discharge with the time, place and extent of human exposure. The concept is outlined schematically in Fig 6. The site operator and the regulatory authorities are aware of the mixture of radionuclides that are likely to be present in the waste effluent and the consequences of a unit discharge (1 TBq d^{-1}) are assessed for each radionuclide to give an estimate of the dose rates to the critical group and the population. Provided that the dose rate to the critical group from the proposed discharge is less than the appropriate dose limit, these data give the basis for optimization, that is, the minimization of the aggregated costs of protection (effluent treatment) and the detriment arising from the incremental radiation exposure. It should be noted that treatment of the effluent to reduce radionuclide concentrations in the environment may result in higher exposures to the workforce in the plant and these latter exposures must also be included in the optimization process.

The characteristics of the effluent - its acidity or alkalinity (pH), ionic strength, the content of non-radioactive elements (e.g. iron), the presence of organic solvents and the chemical speciation of the radionuclides - are all relevant factors influencing the initial behaviour of the radionuclides when they are mixed with coastal seawater. In addition, the nature of the receiving environment in the immediate vicinity of the discharge point - the salinity, the tidal strength and suspended sediment load - also markedly influences the initial behaviour of the nuclides. Information on these factors allows the estimation of the steady-

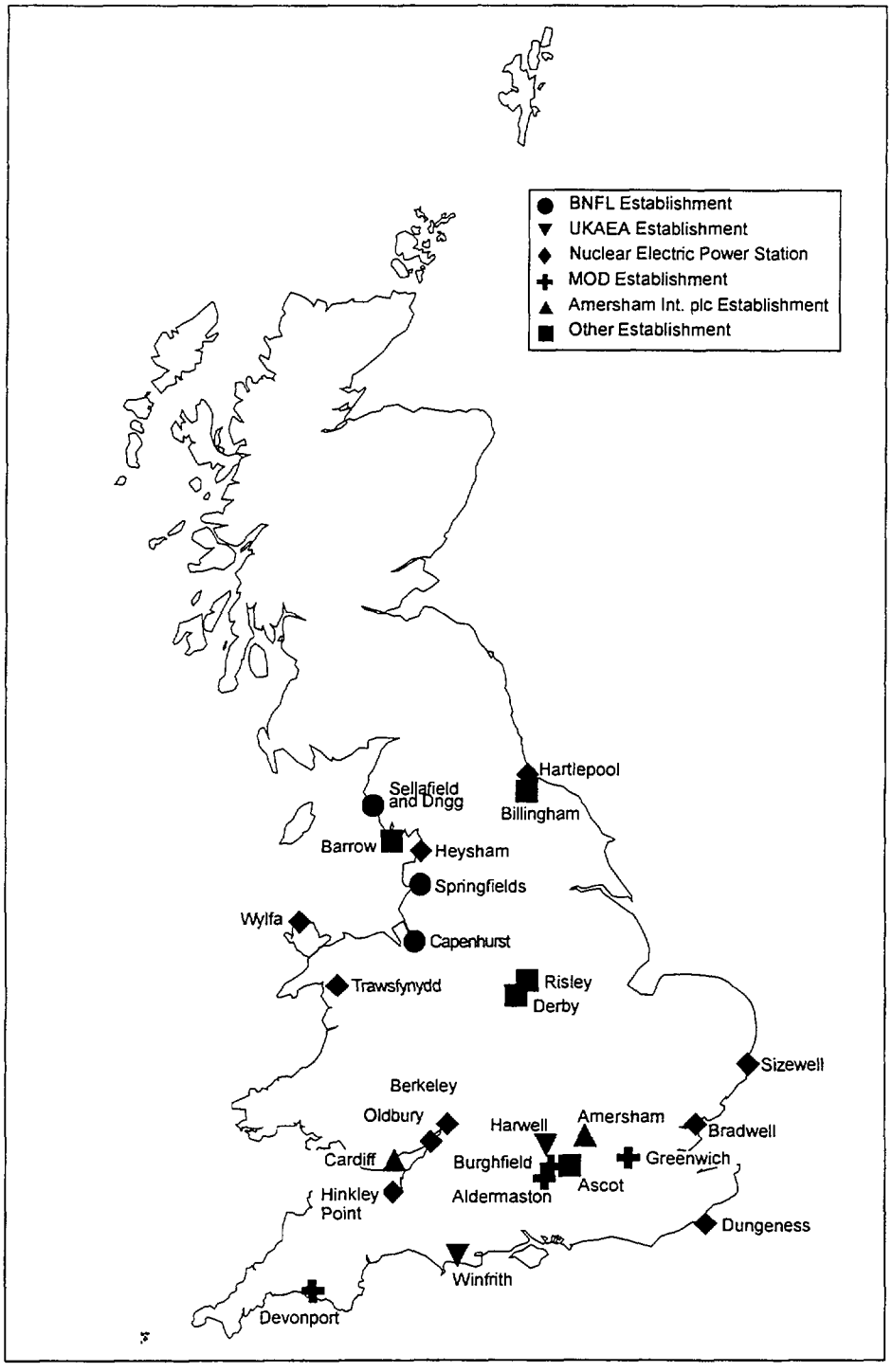

FIG. 5. The principal UK nuclear sites giving rise to discharges of liquid radioactive wastes.

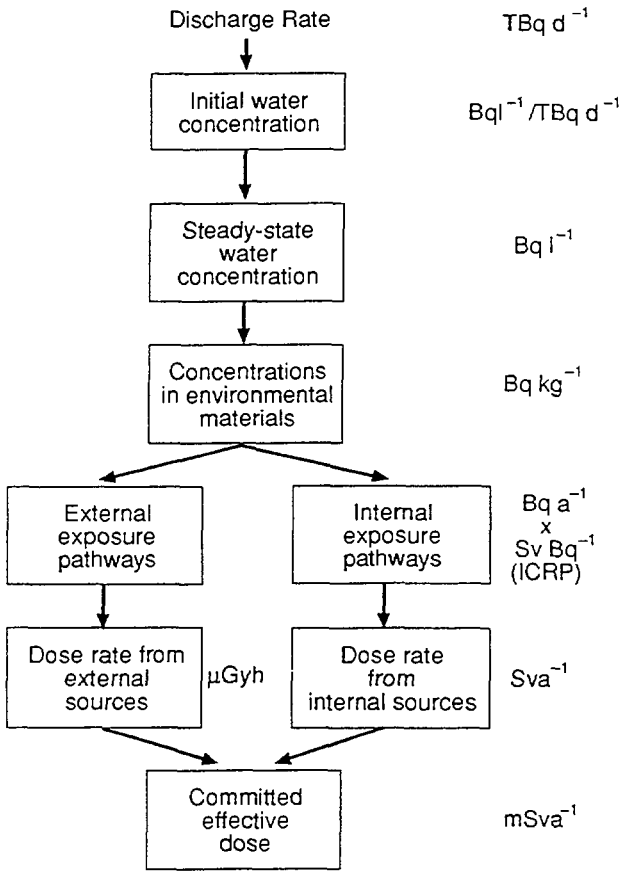

FIG. 6. Conceptual basis for undertaking a pre-operational assessment to develop an authorization for liquid radioactive waste disposal to an aquatic system.

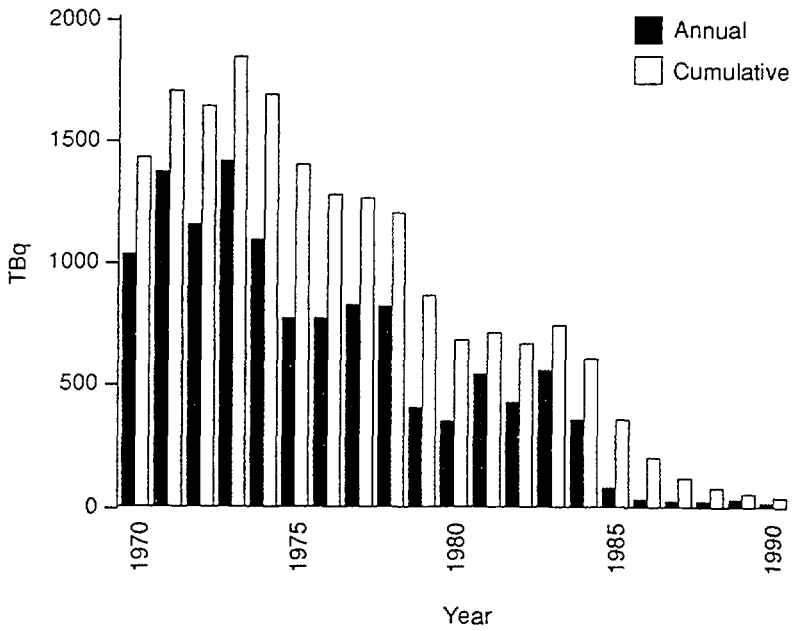

FIG. 7. Sellafield discharges to the NE Irish Sea: Annual input and cumulative decay-corrected environmental inventory of Ru-106.

state water concentration in the local environment. This, together with data on the uptake of the nuclides by sediment (particularly the very fine material less than 62.5 μm in diameter) expressed as:

DISTRIBUTION COEFFICIENT

$$K_D = \frac{\text{nuclide concentration, Bqkg}^{-1}\text{ dry sediment}}{\text{nuclide concentration, Bq l}^{-1}\text{ of seawater}}$$

and their accumulation by marine organisms expressed as:

CONCENTRATION FACTOR

$$CF = \frac{\text{nuclide concentration, Bq kg}^{-1}\text{ fresh tissue}}{\text{nuclide concentration, Bq l}^{-1}\text{ of seawater}}$$

allows the estimation of the equilibrium concentrations in the different environmental compartments.Care must be taken to ensure that any factors likely to result in higher concentrations far from the discharge point, e.g. strong currents, the presence of fine mud in a distant estuary, a distant population of a particular marine species having a large CF for a specific radionuclide etc, are included in the assessment.

The next step is to determine the pathways by which contaminated material in, or from, the marine environment can result in human exposure. For a given individual, this can take either, or both, of two forms:

external exposure from contaminated water (swimming), sediments (working or walking on beaches), fishing gear etc; and

internal exposure following the consumption of seafoods, inhalation of seaspray or fine beach dust etc.

This is essentially a matter of determining the use that is made of the contaminated environment, that is, the working, leisure and eating habits of the potentially exposed population, from which it is possible to identify the critical group(s). The actual radiation exposures are estimated from the time spent on the contaminated foreshore or handling fishing gear, and the combination of the intake (kg a^{-1}) of contaminated seafood (Bq kg^{-1} of the nuclide) and the dose per unit intake (Sv Bq^{-1}) for the radionuclide, as provided by ICRP.

The estimation of the population dose allows the optimization process to be completed and, hence, limits to be placed on the quantities of each nuclide authorized for discharge.

16.3.4. Purpose of monitoring

The practice of monitoring, in the context of radioactive waste disposals to the environment, has a number of objectives:

- analysis of the effluents can provide a check on compliance with the discharge limits for specific radionuclides set in the authorization;

- determination of the concentrations of radionuclides in foodstuffs (wild or cultivated) and the external dose rates in contaminated areas provide a basis for estimating the radiation exposure of the general public;
- it provides an independent check on the results obtained by the operator in fulfilment of the requirements of the authorization; and
- provides surveillance for the detection of accidental releases or unusual behaviour of radionuclides in the environment.

Any practice which is likely to result in effective doses to the general public greater than 1μSv a^{-1} requires the development of an appropriate programme of monitoring and assessment.

16.4. Discharges to sea from Sellafield: a case study

16.4.1. Radionuclide inputs to the sea.

Radioactive wastes have been discharged into the northeast Irish Sea, under authorization, since 1952. Figs 7 to 13 provide data on the annual discharges and decay-corrected environmental inventories of specific radionuclides.

For a short-lived radionuclide like Ru-106 ($T_{½}$ = 1 year), the cumulative environmental inventory is rarely twice the quantity discharged in the previous year except when discharges show a rapid decline (e.g. 1985 onwards) [Fig 7]. This means that if, for any reason, the effective dose rate from Ru-106 (or any other short-lived nuclide) in the marine environment exceeds the appropriate limit, controls applied at source to limit the input rapidly reduce the consequent exposure.

There are two radionuclides of caesium present in the effluent and discharges peaked in the mid-1970's when corrosion of the magnox cans containing the fuel elements in the fuel storage ponds became a problem. Initially, the discharges were reduced by recirculating the pond water through an inorganic ion exchange medium (zeolite) and, more recently, by the commissioning of the site ion exchange plant (SIXEP) to treat the effluent. The cumulative environmental inventory of the shorter-lived Cs-134 ($T_{½}$ = 2.06 years) has rapidly declined, but that of the long-lived Cs-137 ($T_{½}$ = 30 years) remains substantial although falling [Fig 8]. Because the environmental behaviour of the two nuclides is identical, their changing activity ratio in marine samples provides a useful marker to investigate time-dependent processes [the activity ratios in the annual input and the cumulative environmental inventory (decay corrected) are given in Fig 9]. This property has been particularly useful in oceanographic studies of water movement and mixing [12]

Four isotopes of plutonium are present in the effluent from Sellafield: Pu-238 ($T_{½}$ = 87.7 years), Pu-239 ($T_{½}$ = 24,100 years), Pu-240 ($T_{½}$ = 6,550 years) and Pu-241 ($T_{½}$ =14.4 years). The first three isotopes are α-emitters, and the last is a β-emitter and is significant as the parent of the longer-lived α-emitter Am-241. Discharge data are provided for Pu-238 and Pu-239/240 in Figs 10 and 11. The data for Pu-239 and Pu-240 are combined because the two nuclides cannot be determined separately by radiochemical techniques (they emit α-particles of almost identical energy). The inputs of plutonium have now been reduced to the extent that the cumulative environmental inventory of Pu-238 is declining (through decay) while that of Pu-239/240 is increasing only very slowly (at <0.2% per year). The Pu-239+240/Pu-238

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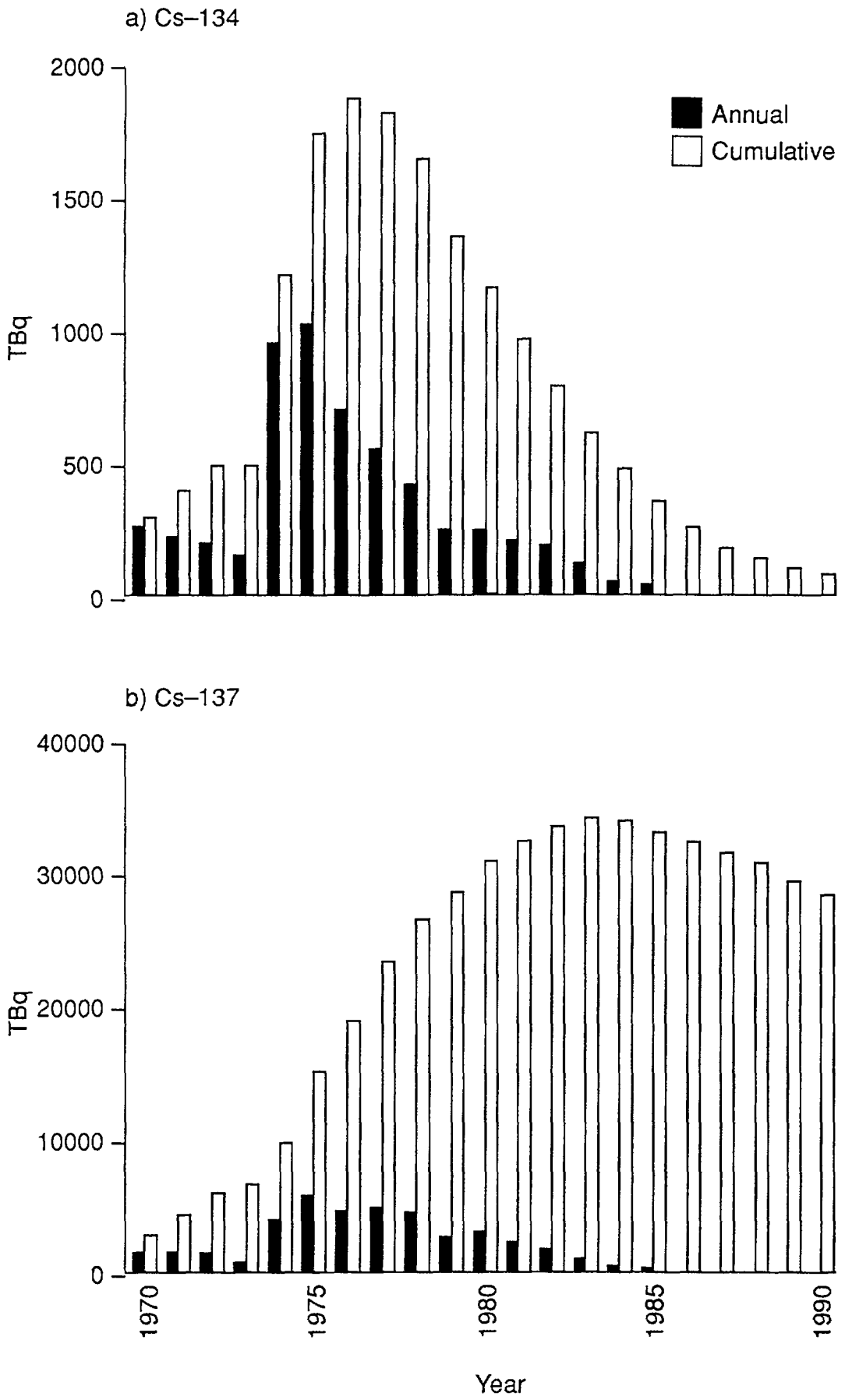

FIG. 8. Sellafield discharges to the NE Irish Sea: Annual inputs and cumulative decay-corrected environmental inventories of Cs-134 and Cs-137.

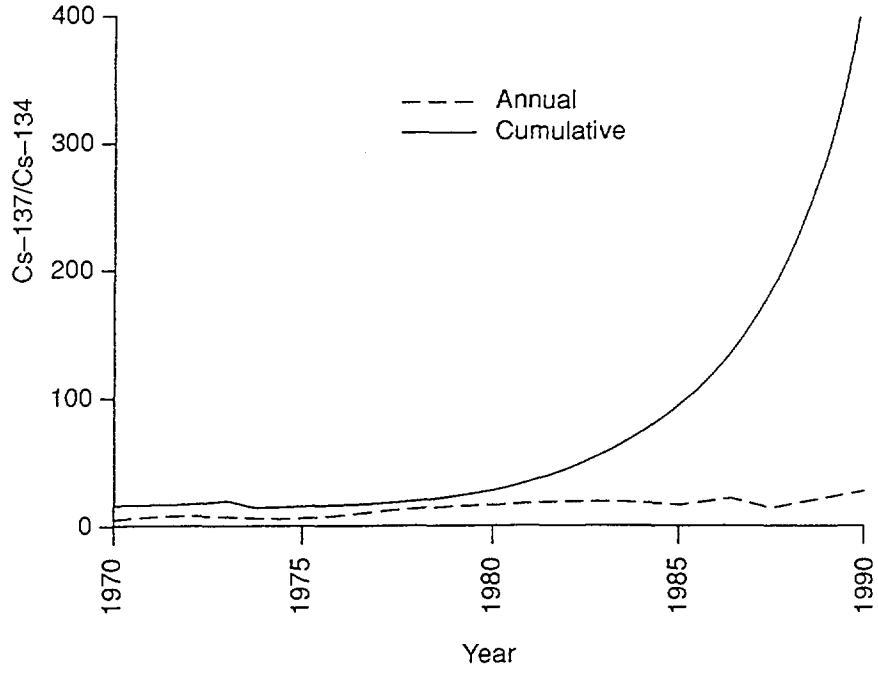

FIG. 9. Sellafield discharges to the NE Irish Sea: Cs-137/Cs-134 isotopic activity ratios for the annual input and the cumulative decay-corrected environmental inventories.

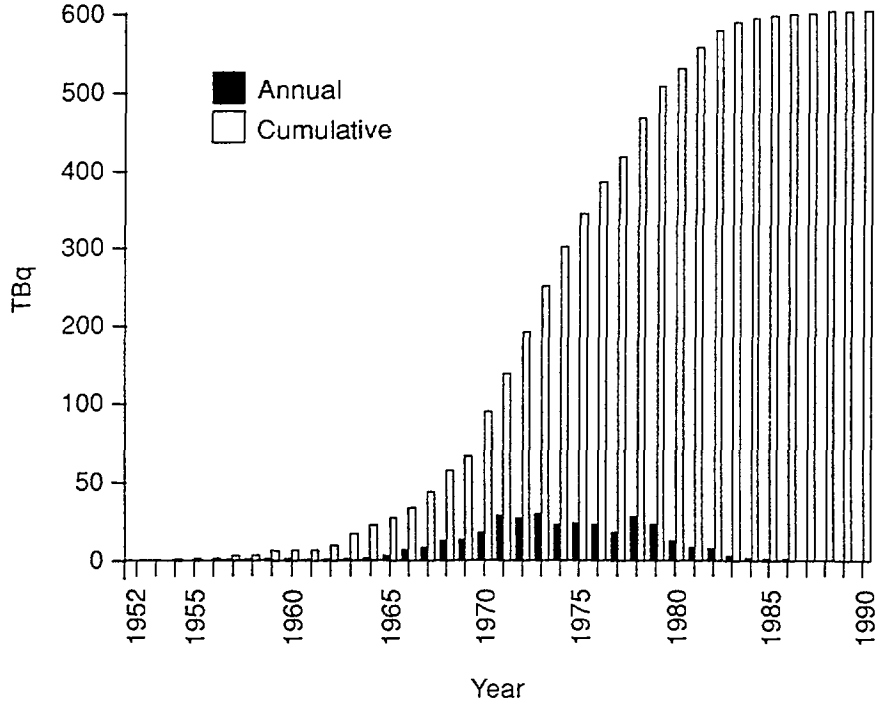

FIG. 10. Sellafield discharges to the NE Irish Sea: Annual input and cumulative decay-corrected environmental inventory of Pu-239+240.

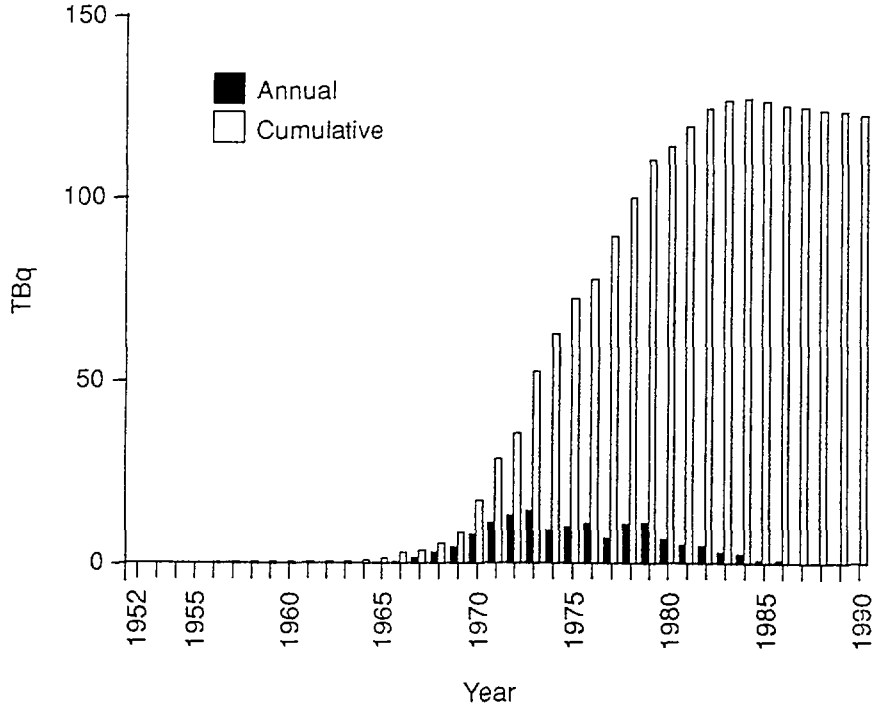

FIG.11. Sellafield discharges to the NE Irish Sea: Annual input and cumulative decay-corrected environmental inventory of Pu-238.

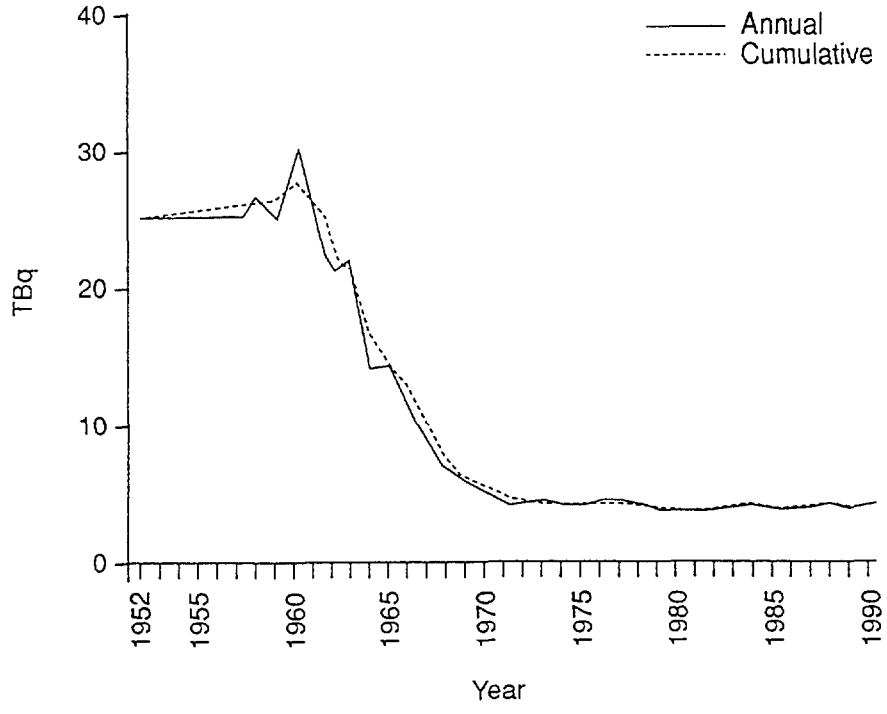

FIG. 12. Sellafield discharges to the NE Irish Sea: Pu-239+240/Pu-238 isotopic activity ratios for the annual input and the cumulative decay-corrected environmental inventories.

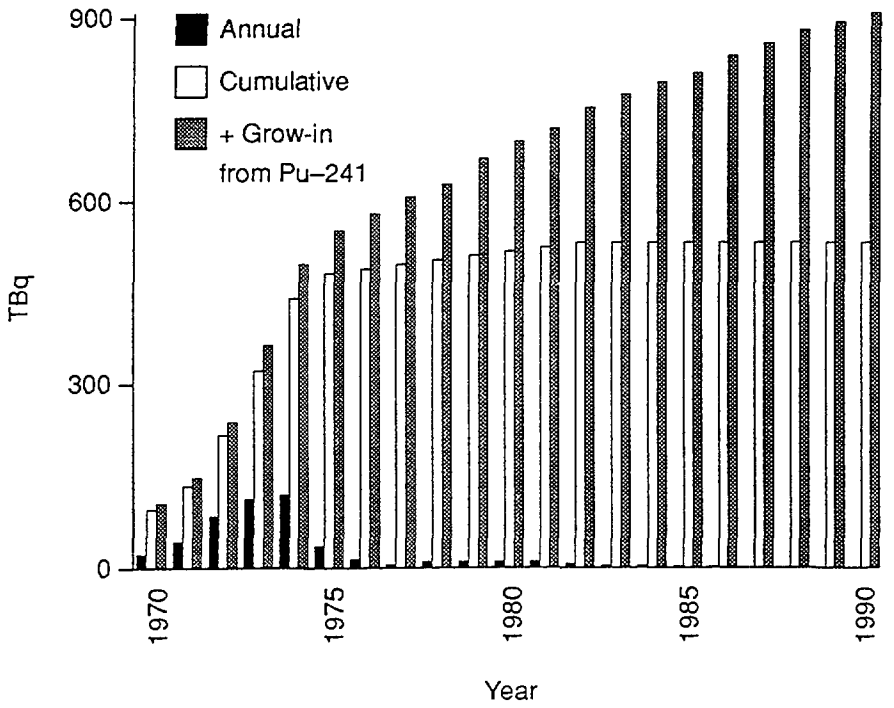

FIG. 13. Sellafield discharges to the NE Irish Sea: Annual input and cumulative decay-corrected environmental inventory of Am-241 from the annual input; and total cumulative decay-corrected environmental inventory of Am-241 including in-growth from the decay of the environmental inventory of Pu-241.

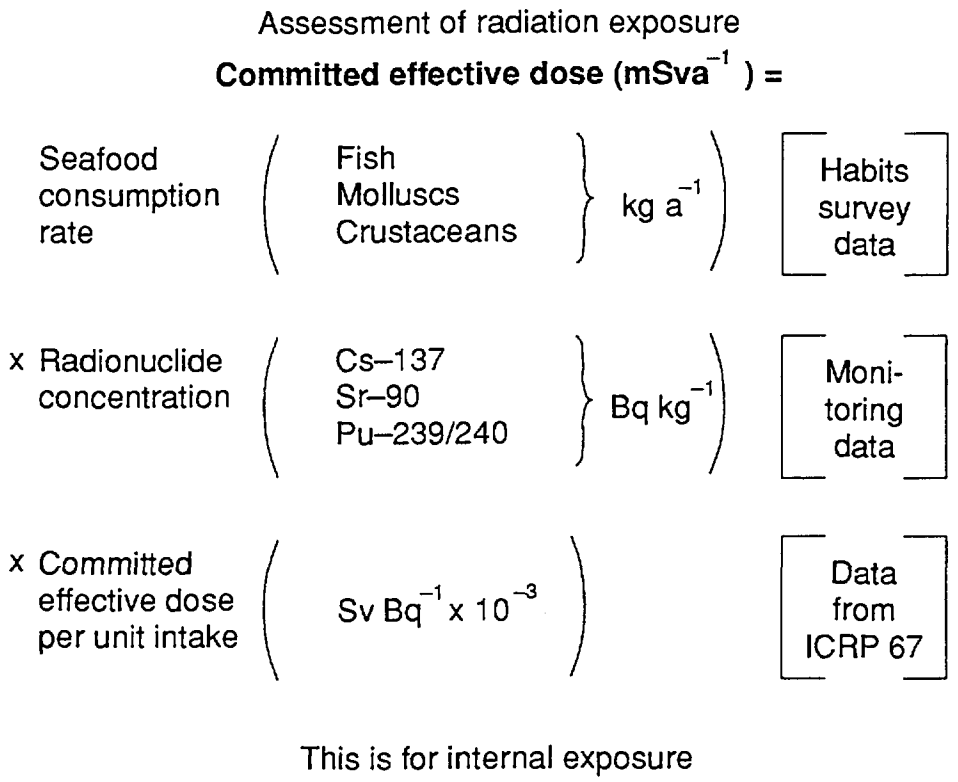

FIG. 14. Procedure for assessing the radiation exposure from internal sources arising from the consumption of contaminated seafood.

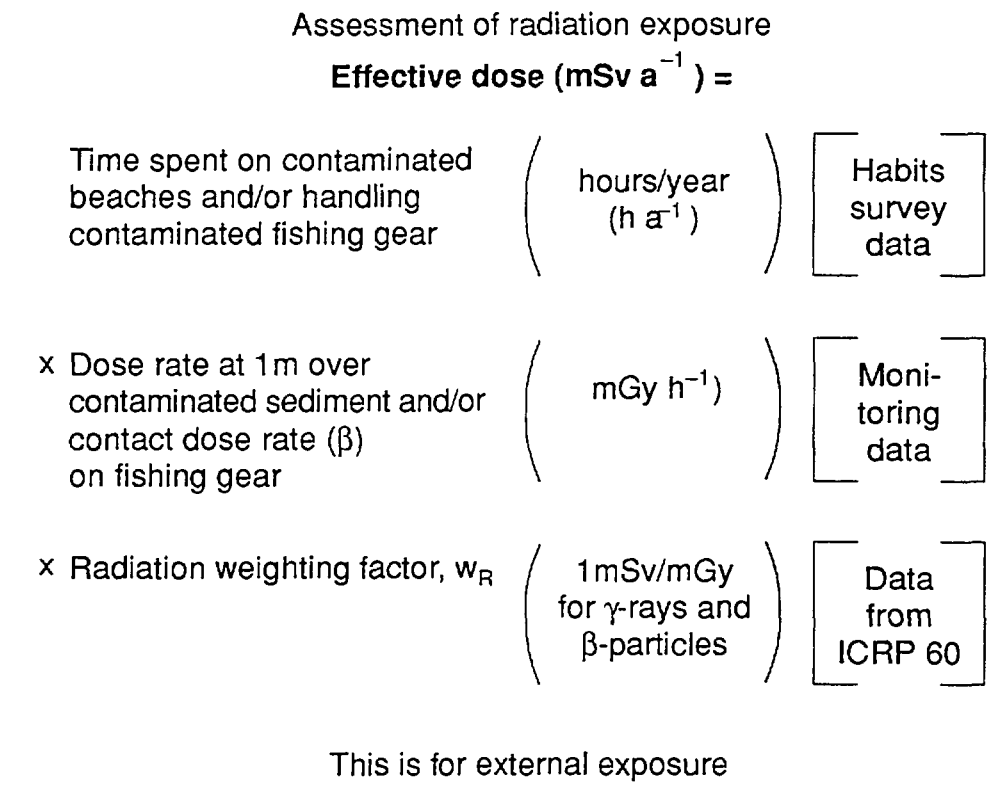

FIG.15. Procedure for assessing the radiation exposure from external sources.

Individual radiation exposures at Sellafield, 1992

Community	Food eaten	Consumption rate, kg a^{-1}	Committed effective dose mSv a^{-1}
Local fisherman	Fish	37	0.12 ^{241}Am, $^{239+240}$Pu, ^{137}Cs
	Crustaceans	6	
	Molluscs	11	
Commercial fisherman (Fleetwood)	Fish	82	0.08 ^{241}Am, $^{239+240}$Pu, ^{137}Cs
	Crustaceans	17	
	Molluscs	23	
Typical member of the general public in Whitehaven or Fleetwood	Fish	15	0.002 ^{137}Cs

Using ICRP-60 methodology

FIG. 16. Estimated individual radiation exposures of three local communities (including the critical group of fishermen) consuming seafood contaminated with radionuclides from the Sellafield discharges; the radionuclides giving the major part of the dose are indicated.

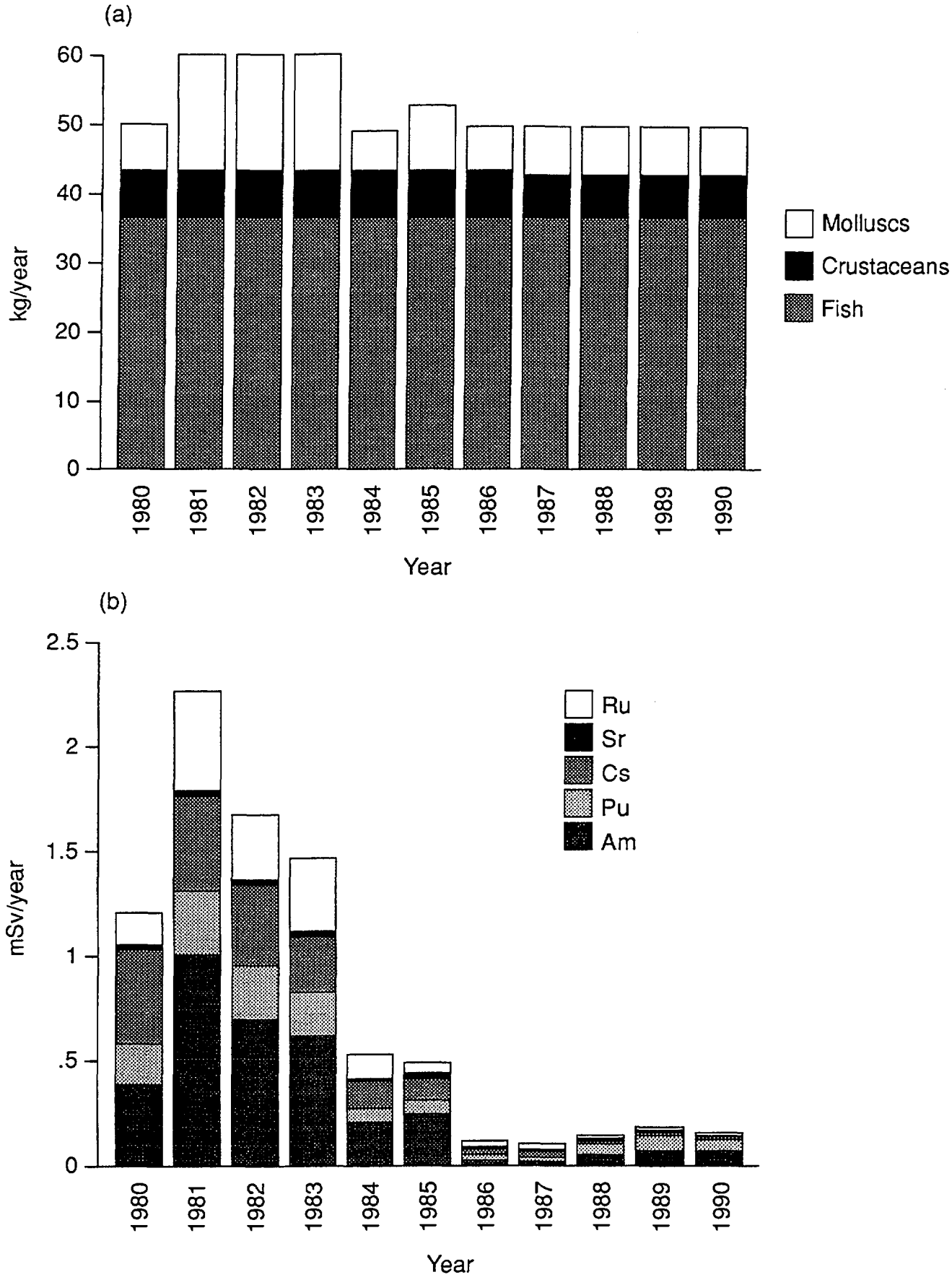

FIG. 17. Variations in the annual seafood consumption rates in the local fishing community (critical group) and the variation in the individual radiation exposures from the major radionuclides in the seafoods.

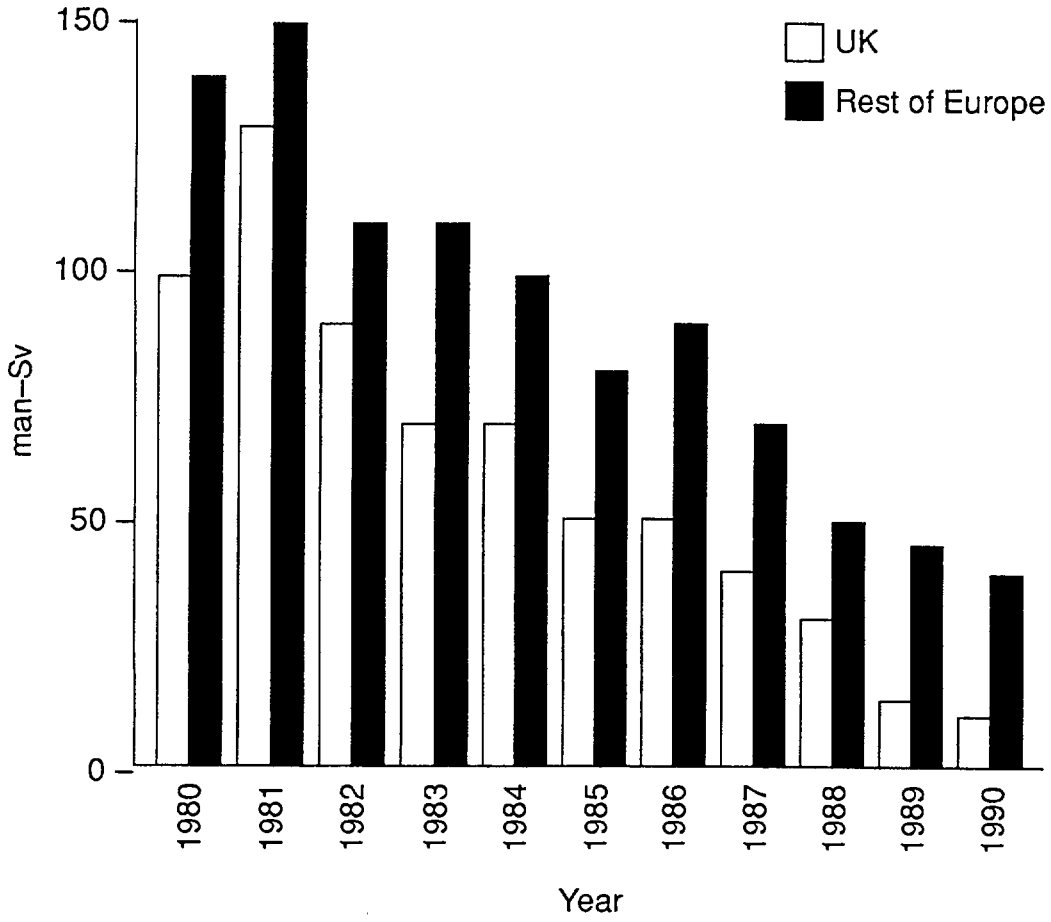

FIG. 18. Estimates of the collective doses to the populations of the UK and the rest of Europe from the Sellafield discharges.

activity ratio in the discharge has varied over time [Fig 12] depending on the degree of "burn-up" in the fuel being processed. Again, this ratio has proved to be a useful marker of both the plutonium of Sellafield origin and the behaviour of plutonium in the marine environment particularly, due to its particle-reactive nature, in seabed sediments [13].

Fig 13 provides data for Am-241 for which the peak discharges occurred in the early 1970s. The cumulative environmental inventory from the direct input in the effluent is essentially at equilibrium but there is another source - the production from the decay of the environmental inventory of Pu-241 - which must also be taken into account. This latter source is generating a substantial increase in the cumulative environmental inventory of Am-241 which, at present discharge rates, will come to an approximate equilibrium value for the environmental inventory in about 40 years.

16.4.2. Assessments of human radiation exposure from the Sellafield discharges.

The critical pathways leading to human radiation exposure from the radionuclides released into the sea from Sellafield have changed over time. This has been a consequence of both the variation in the relative proportions of the radionuclides present in the effluent and changes in human exploitation of the environment. Initially, the principal exposure pathway

arose from the accumulation of Ru-106 by the red seaweed, *Porphyra umbilicalis*, on the Cumbrian foreshore. The seaweed was harvested and transported to south Wales (the Swansea area) where it was processed into "laverbread". The consumption of this foodstuff led to increased radiation exposure of the gastro-intestinal tract from β-radiation. The harvesting of *Porphyra* along the Cumbrian coast declined in the early 1970s and the build-up of contaminated sediment in the Esk estuary led to external exposure of fishermen to γ-radiation (Ru-106, Ce-144 and Zr-95/Nb-95) becoming the critical pathway. The increasing discharges of the Cs radionuclides in the mid-1970s resulted in relatively high concentrations being accumulated by the fish and shellfish harvested from Cumbrian coastal waters. The local fishing community again constituted the critical group. From 1980 onwards, the decline in the Cs discharges and the continuing increase in the environmental inventories of the α-emitting actinide elements - largely retained in the coastal seabed sediments of the northeast Irish Sea - led to a further change in the critical exposure pathway. The local fishing community remains the critical group, but the consumption of shellfish, mainly winkles (*Littorina littorea*), contaminated with Pu-238, Pu-239/240 and Am-241 constitutes the principal source of exposure [9-11]. Notwithstanding these changes in the critical pathways for radiation exposure, the main potential pathways have been monitored continuously [9-11]. The procedures for assessing both internal and external radiation exposure are given in Figs 14 and 15.

The dose coefficients (Sv Bq^{-1} intake) include a factor, f_1, which accounts for the absorption of an element across the gut wall into the systemic circulation. These have changed over time as new information has become available, but this inevitably changes the estimates of committed effective dose for a given intake of radionuclide. The ICRP-recommended values of f_1 are generic, and situation-specific values should be used if they are available. For the northeast Irish Sea, investigations of the local critical group and experiments in which normal, healthy volunteers consumed a single meal of winkles harvested from the Cumbrian foreshore and were subsequently monitored for radionuclide excretion, have provided such site-specific f_1 values for the actinide elements for use in dose assessments.

The individual radiation exposures in three coastal communities (including the critical group of local fishermen) consuming local seafood contaminated with radionuclides from the Sellafield discharge are given in Fig 16 for 1992. Fig 17 shows how the consumption rates and individual exposure rates have changed over time in the local fishing community.

The collective committed effective dose is estimated on an annual basis to provide an indication of the total detriment potentially attributable to the discharges. Over the past decade, the collective dose to the UK population has fallen by a factor of 10; for the remainder of the European population the decline has been less (a factor of 4), mainly due to the time lag in the movement of the effect of the falling Cs discharges through the waters of the European continental shelf [Fig 18]. For 1992 the estimated collective committed effective doses to both the UK population and the population in the rest of Europe had fallen by a further factor of 2 [10].

The individual and collective exposures due to the Sellafield discharges can be given both context and perspective in a number of ways. Fig 19 indicates the number of interactions or radionuclide decays occurring in the human body, per hour, for the sources of radiation making up the natural background; also given are the resulting annual effective dose rates (average and range). Table III shows the contribution to the natural background committed

TABLE III HUMAN EXPOSURE FROM Po-210 AND Ra-226 IN FISH AND CRUSTACEANS

	Po-210		Ra-226
	Fish	Crustaceans	Fish
Concentration Bq kg^{-1}	1.2	3.3	0.11
Consumption Rate kg a^{-1}	82	17	82
Annual intake Bq	100	56	9
Dose Rate: mSv a^{-1}	0.044	0.028	0.003
		TOTAL =	0.075 mSv a^{-1}

TABLE IV. MAXIMUM EXPOSURES OF THE UK POPULATION FROM NUCLEAR POWER

Stage in the Nuclear Fuel Cycle	Pathways	Maximum Exposure mSv a^{-1}
Fuel Fabrication	Atmospheric	0.005
	Aquatic	0.050
Reactor Operation	Atmospheric	0.100
	Aquatic	0.300
Fuel Reprocessing	Atmospheric	0.200
	Aquatic	0.390
Living in West Cornwall	Atmospheric (Rn-222)	2-40

TABLE V. EFFECTIVE RADIATION DOSE TO ADULTS IN THE UK FROM VARIOUS SOURCES

Source	Dose Rate, mSv a^{-1}
Natural Background	1-100 (Average ~2)
Atmospheric Fallout	0.01
Air Travel	0.006
Luminous Watches	0.001
Diagnostic Medicine	0.25
Nuclear Power	0.0015
Coal Burning	0.005

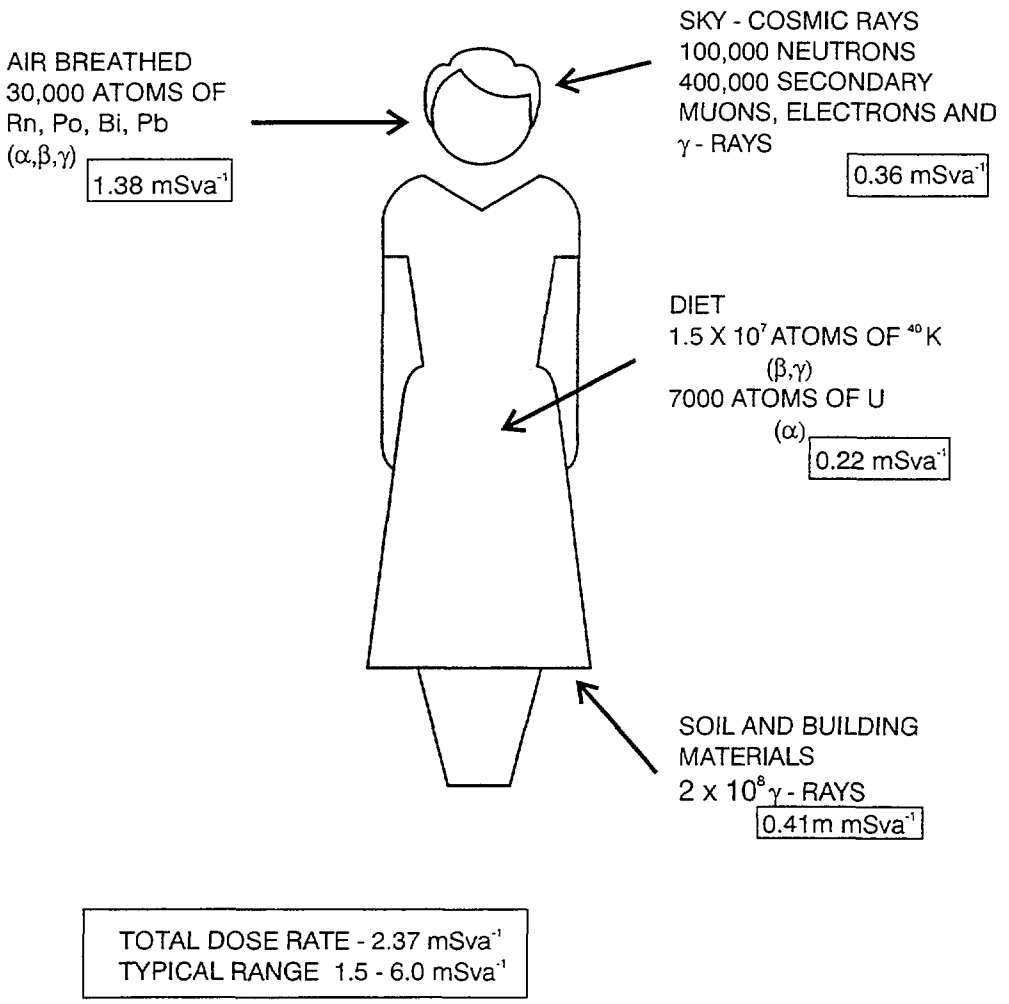

FIG. 19. Estimates of the annual effective dose rates from background radiation sources; the number of interactions per hour for the external cosmic radiation and terrestrial γ-rays, and the number of decays per hour of radionuclides taken into the body with food and inhaled air are also given.

effective dose rate from Po-210 and Ra-226 in fish and crustaceans. The consumption rates are those for commercial fishermen at Fleetwood and the resulting dose rate is of the same order as that from the Sellafield radionuclides. Different stages of the nuclear fuel cycle (fuel fabrication, reactor operation and fuel reprocessing) give different maximum radiation exposures for different discharge routes (to atmosphere and to sea), but these are less than the range of radiation doses arising from exposure to radon in the home in Cornwall [Table IV]. Table V places the average radiation exposure of an individual in the UK population from nuclear power in the context of the radiation exposures from a number of other technological and natural sources. It is of interest to note that the radiation exposure from natural radionuclides discharged to the atmosphere from coal burning (both domestic and power generation) exceeds that from nuclear power operations. The estimated collective effective doses to the UK population from a number of sources of exposure are given in Table VI; of the sources listed, only luminous watches give a lower collective dose than nuclear power.

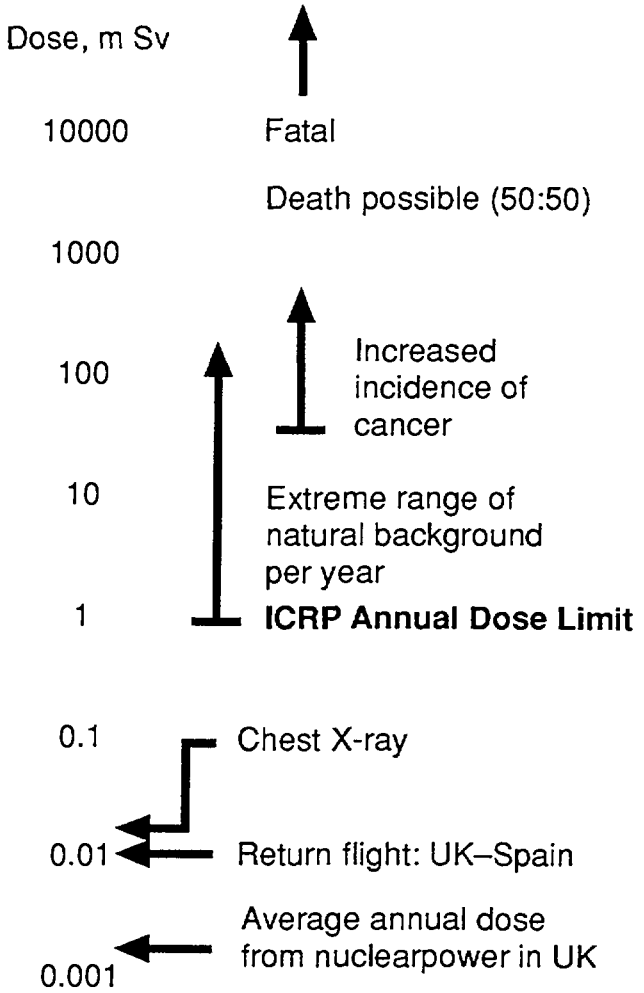

FIG. 20. Effective doses and potential radiation effects in man.

Additional comparisons are provided in Fig 20 together with an indication of the dose thresholds for stochastic (increased cancer incidence detectable in very careful epidemiological studies) and deterministic effects.

Estimates of collective doses are frequently used, together with a risk factor (usually the fatality probability coefficient), to estimate the number of health effects (usually fatal cancers) to be expected in an exposed population. The ICRP have emphasized that caution should be exercised to ensure that this approach is strictly applicable to the doses and dose rates experienced by the exposed population and that the fatality probability coefficient is appropriate to the demographic structure of that population. In practice, this means ensuring that a linear relationship between dose and response is a reasonable and acceptable assumption down to low dose rates and doses, for all age groups and both sexes. The concept of collective dose is more usefully employed as one component of the process of optimisation

TABLE VI. COLLECTIVE DOSES IN THE UK: manSv

Natural Background		105000
Atmospheric Fallout		580
Air Travel		300
Luminous Watches		56
Diagnostic Medicine		13700
Nuclear Power		
Fuel Fabrication	0.2)	
Reactor Operation	4.1)	81.3
Fuel Reprocessing	77)	
Coal burning		240

TABLE VII. "DETRIMENT" IN EUROPE FROM THE CHERNOBYL ACCIDENT.

Country	Estimated Collective Dose manSv	
	1st year	Total
Belgium	480	940
Denmark	650	1100
France	3000	5600
West Germany	14000	30000
Greece	5200	8500
Italy	17000	27000
Holland	720	1200
Spain	45	57
UK	2100	3000
Total: "W Europe	44000	78000
Natural Background	500000 manSv a^{-1}	

of radiological protection. Table VII shows the collective doses estimated for the populations of a number of European countries for the first year after the accident at Chernobyl, and over all time, and indicates the collective dose from the natural background for perspective.

The risk from radiation exposure is placed in the context of other everyday risks in Table VIII and the risks from specific sources of radiation exposure are compared with those from other, non-radioactive, contaminants in seafood in Table IX. It may be concluded that, apart from the essentially unavoidable risk from the natural radiation background, risks from radiation exposure, when the ICRP approach to radiological protection is applied, are not excessive.

TABLE VIII. RISKS TO INDIVIDUALS

Cause	Risk of Death
Accidents:	
On the Road	1 in 9500 a^{-1}
At Home	1 in 26000 a^{-1}
At Work	1 in 43500 a^{-1}
"Natural Causes" at age 40	1 in 850 a^{-1}
Smoking 10 Cigarettes/Day	1 in 200 a^{-1}
Radiation Exposure 1 mSv a^{-1} (Cancer only)	1 in 20000 a^{-1} (ICRP 60, para. 83)

TABLE IX. ESTIMATES OF CANCER INCIDENCE IN THE EEC FROM A VARIETY OF CONTAMINANTS IN SEAFOODS

Source	Fatalities/Year
Natural Radionuclides	170
Total PCBs	180
Dieldrin	150
Radwaste to Sea	17
Chernobyl Fallout	7
HCB	6
Chlordane	5
Weapons Fallout	5
Total DDT	3
Past Sea Dumping	0.005

16.5. Ocean dumping of packaged, low-level waste

The practice of dumping packaged, low-level radioactive wastes into the deep ocean has attracted much political and scientific attention in recent years. The practice has been regulated through the mechanism of the London Dumping Convention which came into force in 1972 (it has recently been renamed the London Convention 1972), with relevant scientific advice being provided by the International Atomic Energy Agency. The dumping of high level waste, e.g. irradiated nuclear fuel, is expressly forbidden and the IAEA have provided a formal definition of waste which is unsuitable for disposal in the deep ocean [Table X]. The Nuclear Energy Agency of the Organisation for Economic Co-operation and Development (NEA-OECD) have provided a practical mechanism by which a dumping operation at an

TABLE X. DEFINITION OF WASTE UNSUITABLE FOR DUMPING

The dumping of high level waste is forbidden: (e.g. irradiated nuclear fuel)

A formal definition of the activity content of other wastes unsuitable for dumping is provided by the IAEA:

α-activity :	$> 5 \times 10^{-5}$ TBq kg^{-1}
β/γ activity with $T_{½} > ½$ year:	$> 2 \times 10^{-2}$ TBq kg^{-1}
β/γ activity with $T_{½} > ½$ year + tritium:	> 3TBq kg^{-1}

The concentrations must be averaged over a gross mass not exceeding 10^3 tonnes.

identified site can be assessed and regulated. The IAEA has developed generic oceanographic and radiological models to determine the definition of radioactive wastes which are unsuitable for dumping. The NEA/OECD, however, has developed site-specific models to assess the potential consequences of dumping at a defined location (site) in the deep ocean with due recognition of the advice, constraints and recommendations provided by the IAEA.

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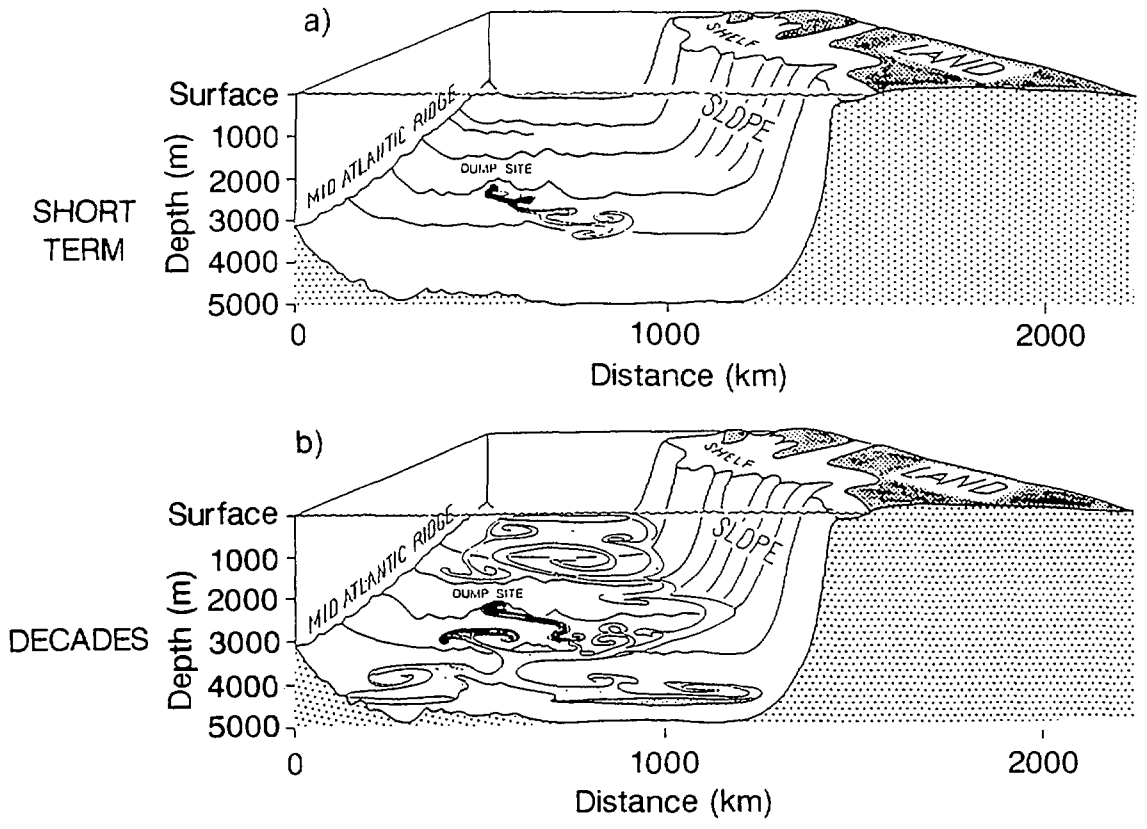

FIG. 21. Schematic diagram of the plume of contaminated seawater developing from the release of radionuclides from packaged radioactive wastes dumped in the deep ocean.

TABLE XI. MODELLING FRAMEWORK USED IN THE RADIOLOGICAL ASSESSMENT

Parts of system included in model	Model	Major processes included in model
Canister and lining. Waste form	Waste Package	Canister corrosion Degradation of package linings and caps. Release of radionuclides from waste forms
	⇓	
	Rates of release of radionuclides into the ocean, as a function of time	
Bottom sediments. Benthic boundary layer (water and particulates). Open ocean (water and suspended particulates)	⇓	Diffusion and advection. Interactions between radionuclides and suspended particulates and bottom sediments.
	Ocean dispersion	
	⇓	
	Radionuclide concentration in water and sediments, as a function of time	
Exposure pathways - sea foods, beaches, atmosphere, salt, water. Marine organisms	⇓	Reconcentration of radionuclides in marine organisms, beach sediments, and aerosols. Radionuclide intake and metabolism by man and organisms
	Dose to Man and Organisms	

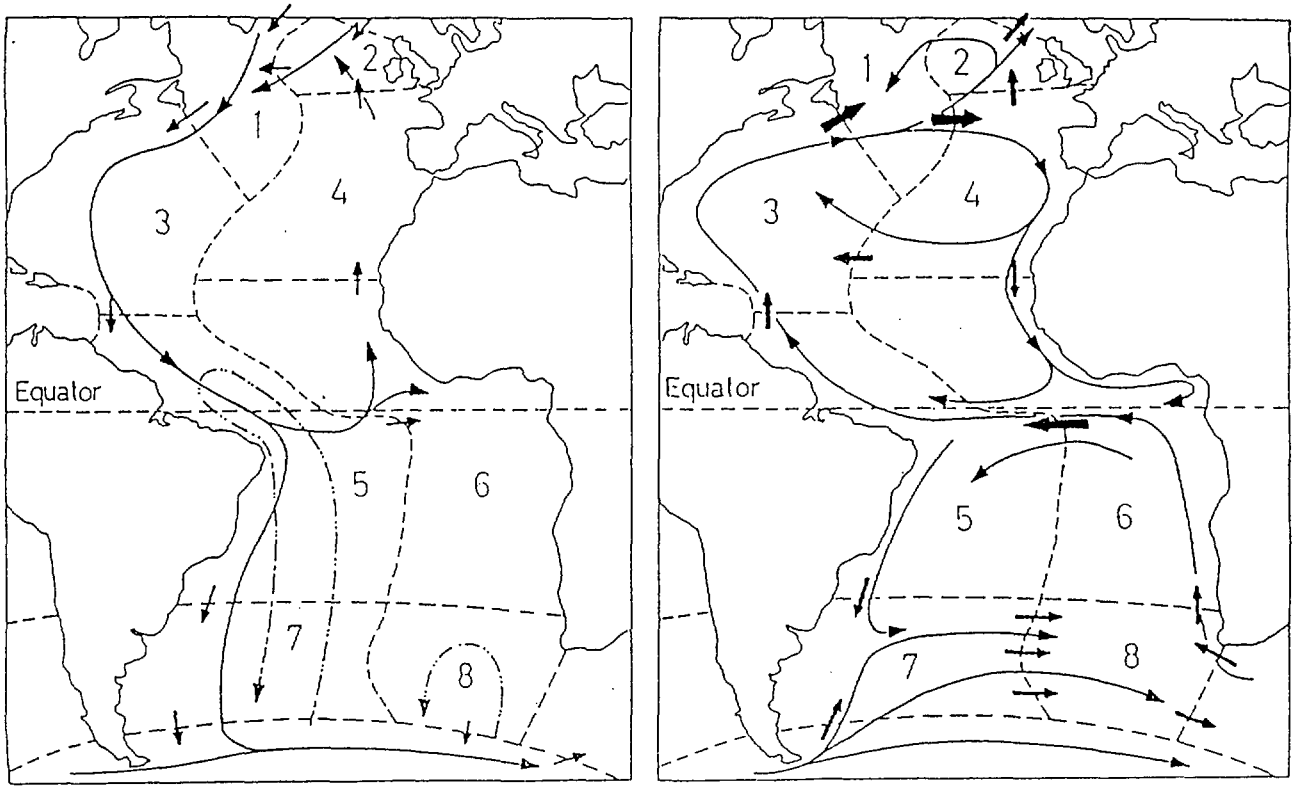

FIG. 22. Seawater flow fields at the surface and at 4000m in the Atlantic Ocean. Reproduced with permission from: Review of the continued suitability of the dumping site for radioactive waste in the North-East Atlantic, OECD/NEA, Paris, 1985.

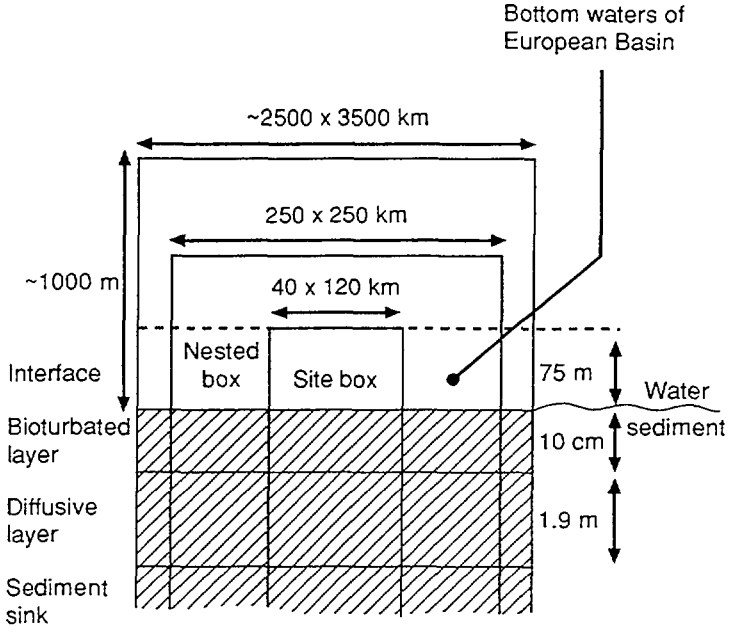

FIG. 23. The nested box structure around the dump site used to model the initial dispersion of radionuclides released from the wastes. Reproduced with permission from: Review of the continued suitability of the dumping site for radioactive waste in the North-East Atlantic, OECD/NEA, Paris, 1985.

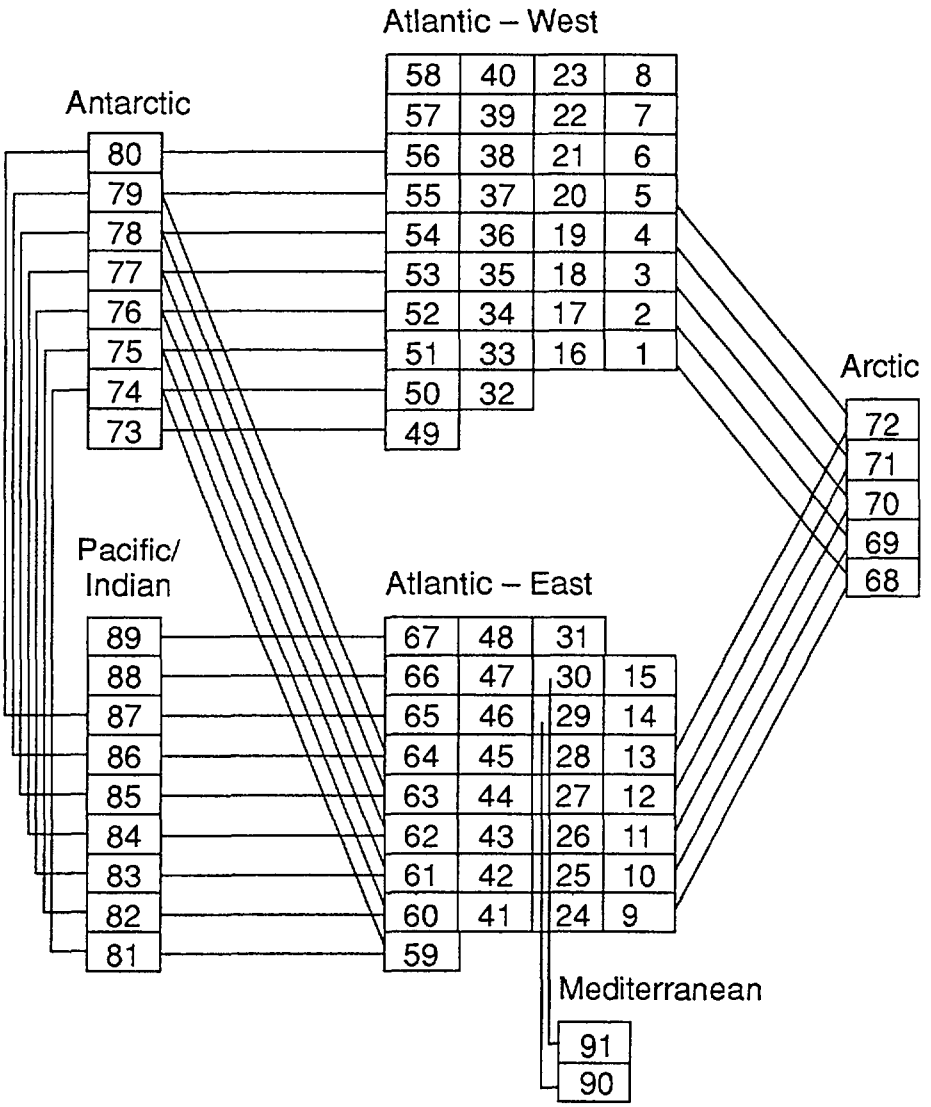

FIG. 24. The box structure used to model the wider dispersion of waste radionuclides in the world ocean. Reproduced with permission from: Review of the continued suitability of the dumping site for radioactive waste in the North-East Atlantic, OECD/NEA, Paris, 1985.

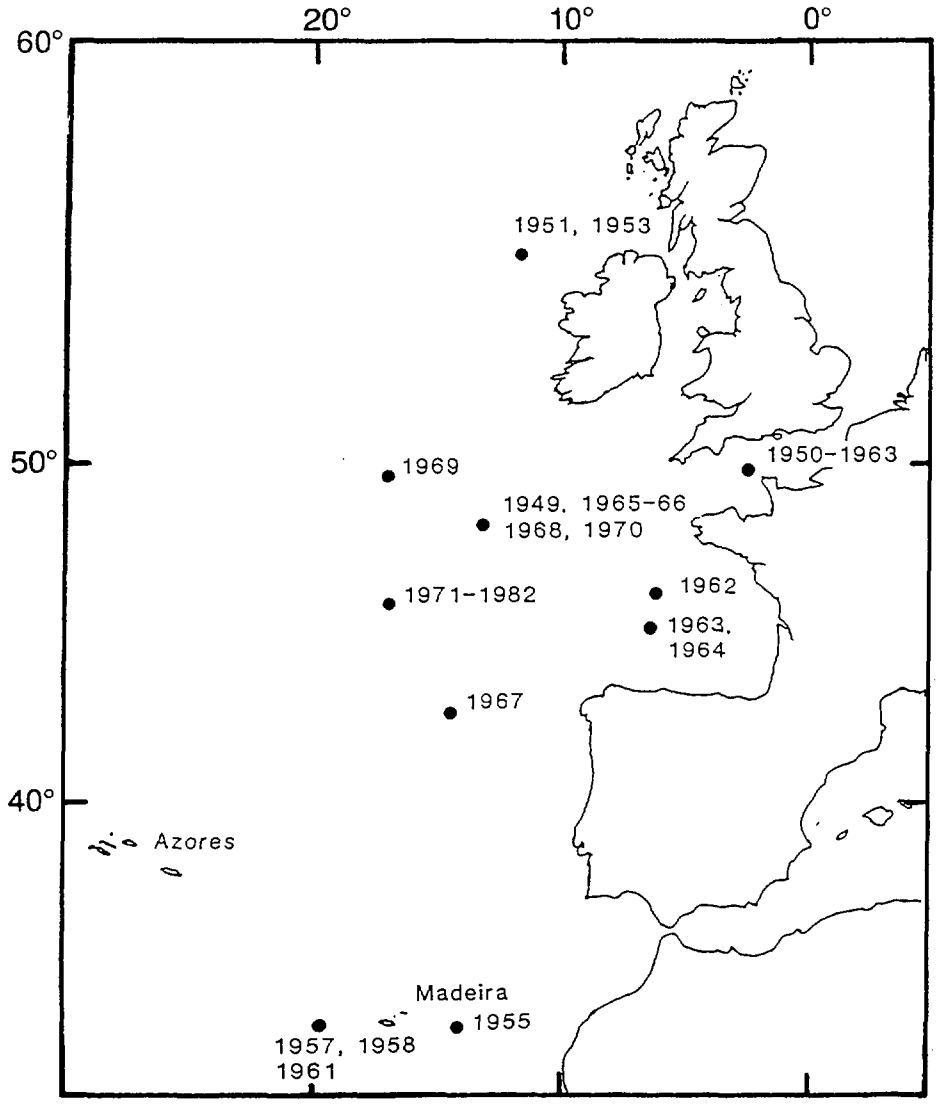

FIG. 25. Sites in the NE Atlantic Ocean used for packaged, low-level radioactive waste dumping in the period 1949-1982. Reproduced with permission from: Review of the continued suitability of the dumping site for radioactive waste in the North-East Atlantic, OECD/NEA, Paris, 1985.

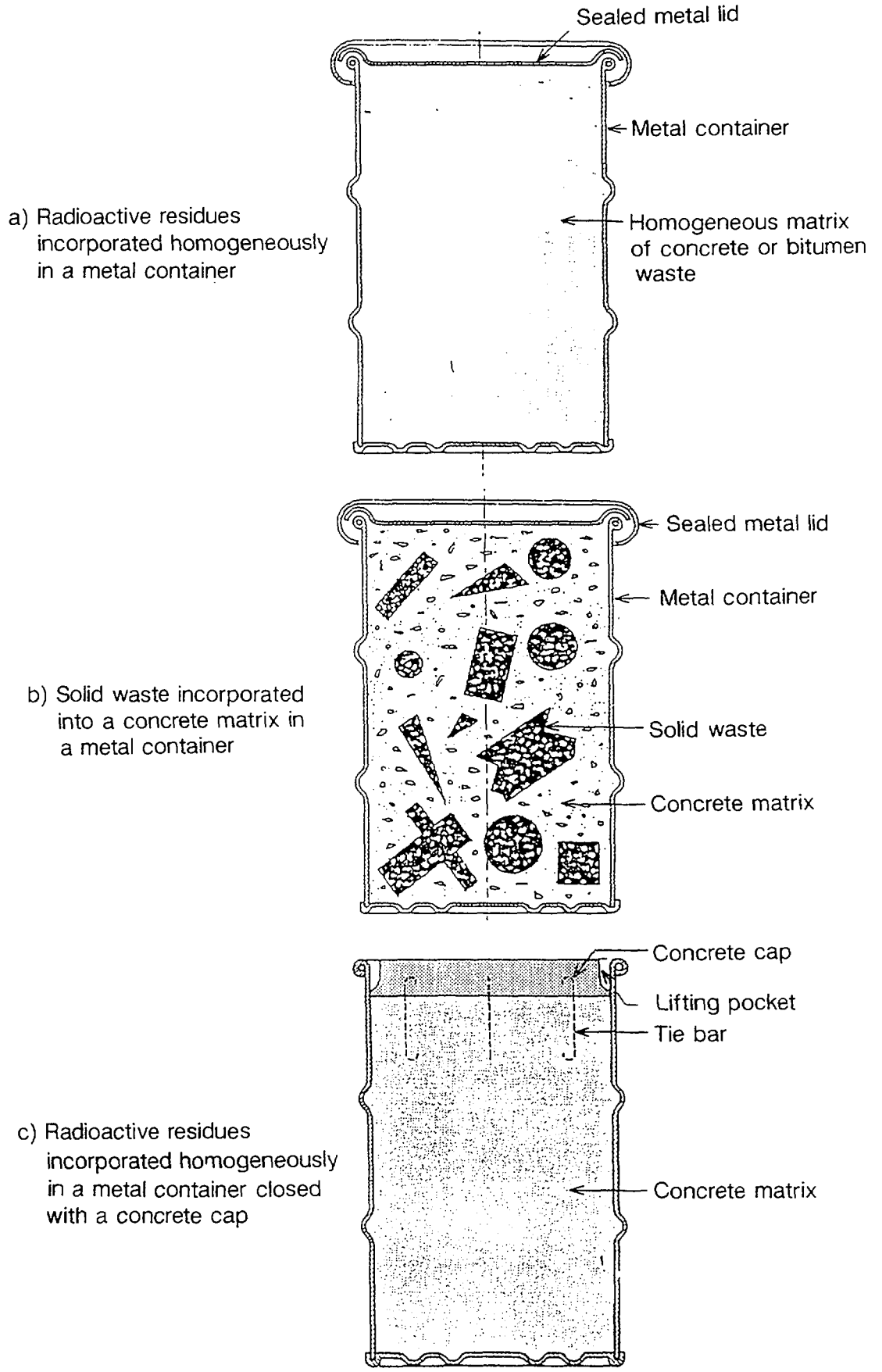

FIG. 26. Approved waste package designs used as a basis for modelling the release of radionuclides to seawater.

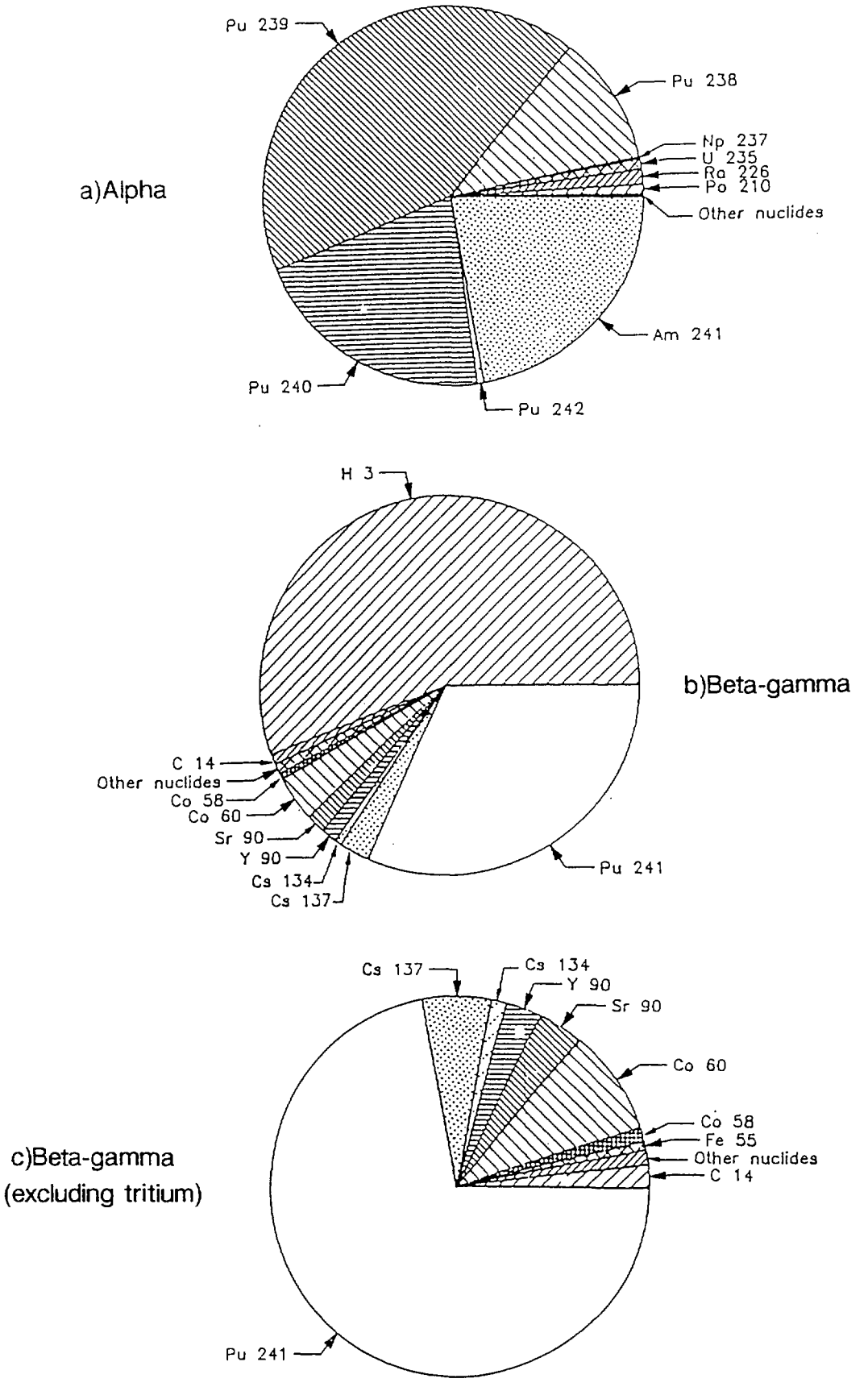

FIG. 27. Estimated quantities of radionuclides included in the waste packages dumped in the NE Atlantic Ocean, 1949-1982. Reproduced with permission from: Review of the continued suitability of the dumping site for radioactive waste in the North-East Atlantic, OECD/NEA, Paris,1985.

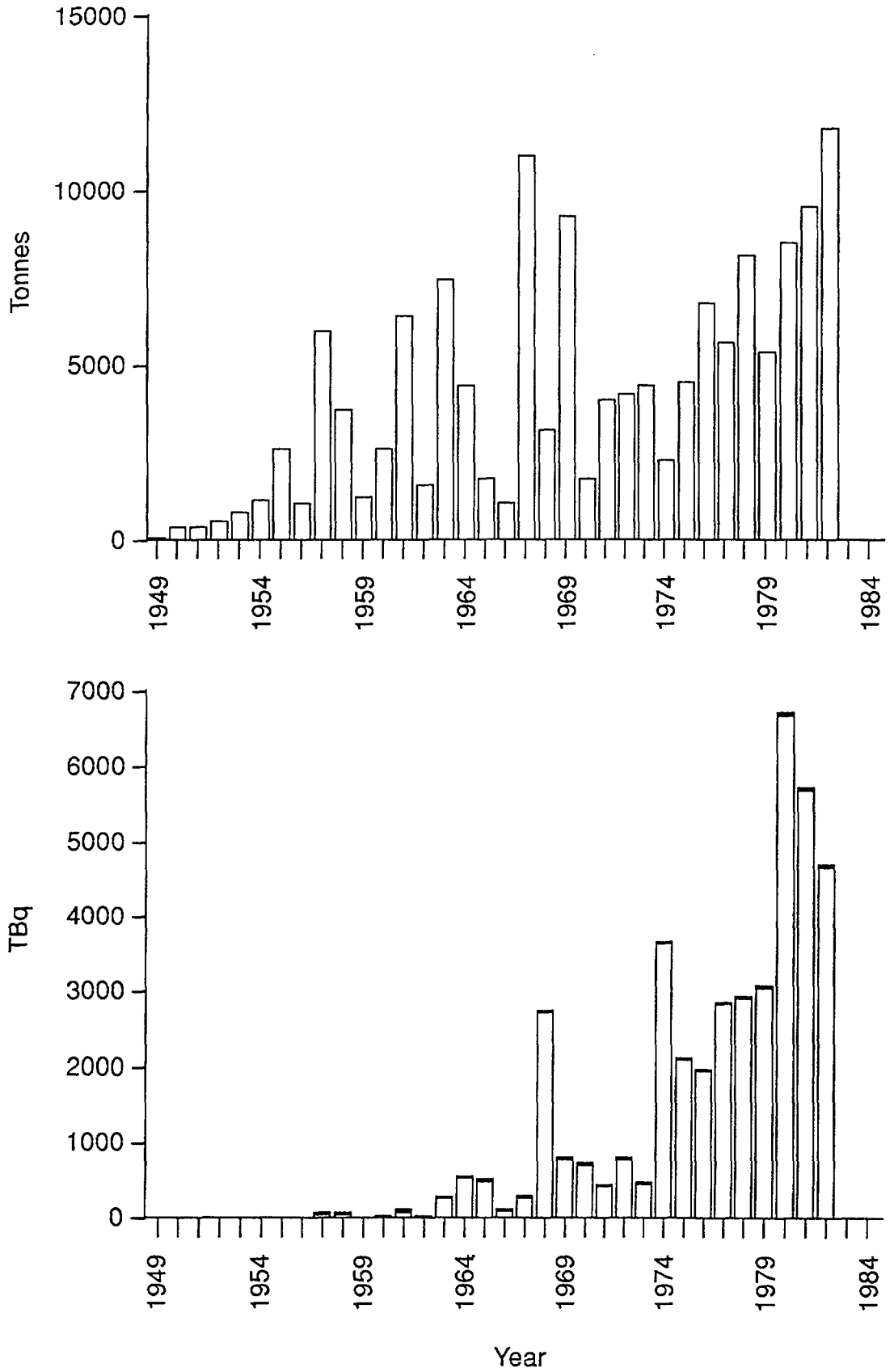

FIG. 28. Estimated total mass of the packages, and the quantities of α-emitting and β/γ-emitting (excluding tritium) radionuclides dumped into the NE Atlantic Ocean each year. Adapted from: Review of the continued suitability of the dumping site for radioactive waste in the North-East Atlantic, OECD/NEA, Paris, 1985.

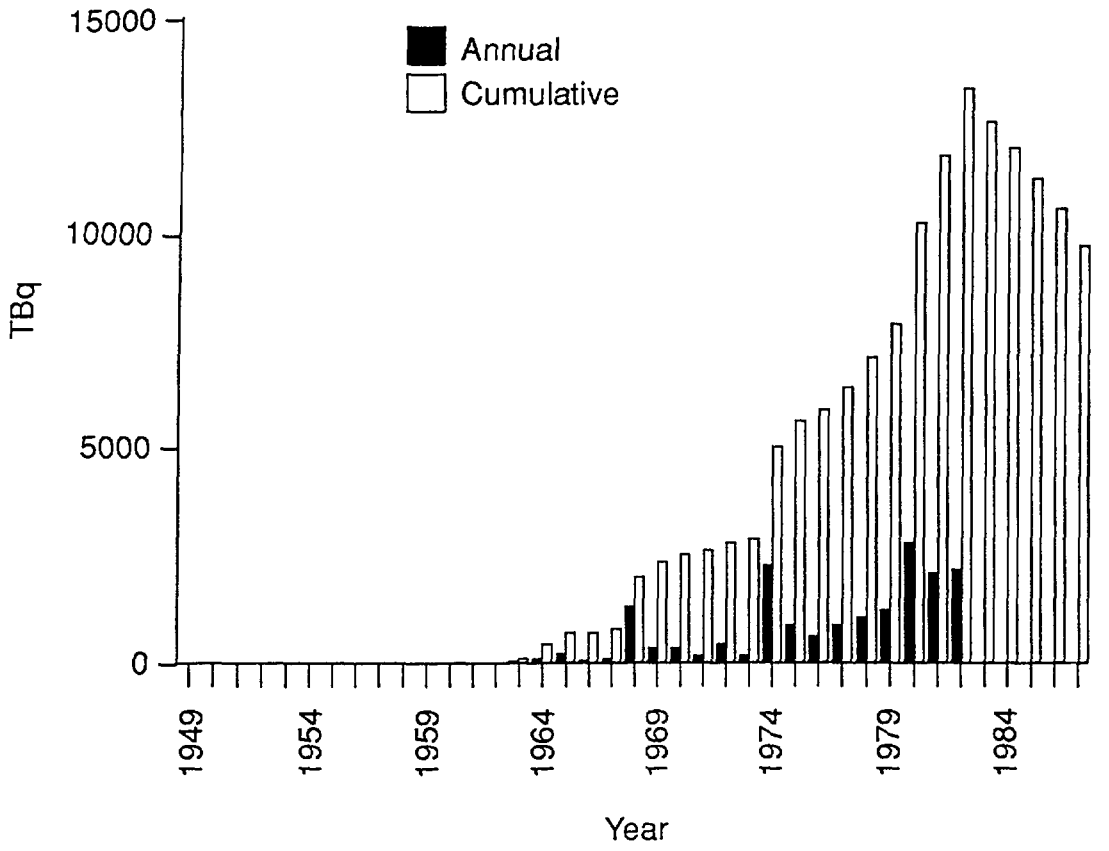

FIG. 29. Estimated annual inputs and decay-corrected cumulative environmental inventory of tritium in the NE Atlantic Ocean from packaged waste dumping. Adapted from: Review of the continued suitability of the dumping site for radioactive waste in the North-East Atlantic, OECD/NEA, Paris, 1985.

An outline of the modelling framework employed in the NEA/OECD site assessment is given in Table XI [14]. It includes components modelling the releases of various radionuclides from different waste matrices into the water; the dispersion of the radionuclides from the release point by diffusion and advection with the inclusion of losses to deep sea sediments by direct absorption and through scavenging from the water column by sedimenting particles; and, finally, the reconcentration of radionuclides into marine organisms or onto beach sediments as proximal sources (the former through ingestion) of radiation exposure. Further details are given in Figs 21-24. Since dumping commenced in 1949, a number of sites have been used in the northeast Atlantic Ocean until the last dumping operation in 1982 [Fig 25]. A number of means of conditioning waste for dumping have been approved [Fig 26]; these have primarily been based on the use of a 50 gallon (approximately 220 litres) metal oil drum as the outer container, but in some instances, monolithic reinforced concrete boxes have been employed for some classes of waste. The quantities of individual radionuclides dumped into the northeast Atlantic Ocean during the period 1949-82 have been estimated [Figs 27-29], and the data used as the input to the oceanographic and radiological models to assess the potential human exposure. Due to the time necessary for diffusion and advection to transport radionuclides from the site at a depth of greater than 4000m to the biologically-productive surface coastal waters, the peak effective dose would be to an hypothetical population in the Antarctic consuming molluscs contaminated with Pu-239/240 some 200 years into the future.

The aggregate dose, over all nuclides and pathways (using the highest likely consumption and occupancy rates) amounts to 0.02m Sv a^{-1} at the peak; this may be compared with an average of about 2mSv a^{-1} from the natural background. The major radionuclide contributing to the collective dose is C-14, and from past dumping this amounts to an aggregate total of 3.8 x 10^4 man Sv distributed through a world population of approximately 10^{10} persons over a period of some 10^4 years. It is interesting to note that, apart from deep geological disposal, the collective dose from C-14 is relatively independent of the means of disposal.

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