MARINE RADIOACTIVITY
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Chapter 7

Radiological assessment of ocean radioactivity

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This chapter begins by setting out the radiological quantities and criteria for protection when assessing ocean radioactivity. The radiological protection standards are those promulgated by the International Commission on Radiological Protection (ICRP). A sophisticated system has evolved to protect man and whilst this has been sufficient in most environments to protect other species, a framework for protection of other species is being considered and this is outlined.

The radiological assessment process is then described using a pathway approach with simple models as examples to describe the consequences of the different marine environmental processes. Ingestion, inhalation and external dose assessments are discussed. For assessment of compliance with dose limits, the selection of an appropriate 'critical group' is a central feature, and this is based on the results of habits surveys. Collective dose is also a consideration in the ICRP methodology and this too is described.

There is then a comparative assessment of sources of ocean radioactivity, looking first at doses due to natural radionuclides, then those due to artificially-enhanced natural radioactivity. Artificial sources due to weapons-test fallout, operations of the nuclear industry, ocean dumping of solid radioactive waste, dumping in the Arctic, and the effect of the Chernobyl accident are all compared in terms of critical group dose and collective dose. Though there are fluctuations near particular sources, generally the highest doses from marine sources derive from natural radionuclides, followed by those from artificially-enhanced natural radionuclides. Weapons-test fallout is the next most significant source of dose in collective terms, but being diffuse, individual doses are very low. Doses via marine pathways due to the nuclear industry, waste dumping operations and the Chernobyl accident have also produced low doses by comparison with natural sources. The differences are illustrated with suitable graphs.

1. Standards for radionuclides in the oceans

1.1. *Introduction*

Other chapters in this book describe inter alia the extent of radioactive labelling of the world oceans. This chapter now examines the methodology for assessment of radiological signifi-

cance of this labelling. In the course of describing the relevant methodology it is necessary to set out the standards on which the assessment methods are based, and this is done first, setting out the quantities and units used and then examining the relevant radiological protection criteria. The application of these criteria to protection of the oceans is then discussed.

1.2. *Quantities, units and perspectives*

The current generally accepted system of radiological units is part of the Système Internationale (SI), in which the basic quantity of radiation exposure, 'absorbed dose', the energy absorbed per unit mass of material, has the units gray (Gy) (1 Gy = 1 J kg^{-1}). The gray is a large unit, equivalent to 100 rads in the older system of units, thus submultiples mGy and μGy are in common use. In human tissue, the radiological effect of an absorbed dose is obtained by weighting the absorbed dose by a quality factor, Q, which depends on the relative biological effectiveness (RBE) of the type of radiation, and other factors. The result is now termed the 'equivalent dose', for which the unit is the sievert (Sv). Again, this is a large unit (=100 rem in the older system) with submultiples mSv, μSv, etc.

To provide some perspective, indeed for the whole of this chapter, the average natural background radiation dose rate to the world population due to cosmic rays, terrestrial gamma rays, inhalation of radon and from foodstuffs is 2.4 mSv y^{-1} (UNSCEAR, 2000). Table 1 shows this and the main man-made contributions to average doses to the world population in 2000. There can be significant variations between individuals due to location, habits, etc.; the natural background itself would generally be expected to be in the range 1–10 mSv y^{-1}.

1.3. *The basis of radiological protection criteria*

The generally accepted standards for radiological protection, and which are adopted by international organisations, are based on the recommendations of the International Commission on Radiological Protection (ICRP). The ICRP was set up with its current name and form in 1950, but its history dates from 1928. The recommendations of the ICRP have developed significantly over the years. Initially, emphasis was given to occupational exposures and to avoiding doses above prescribed thresholds. However, some harm at low doses was suspected and support grew for an assumption of a linear, no-threshold (LNT) dose-response relationship. Recommendations were consequently based on avoiding threshold effects and keeping risks of cancers and hereditary effects (later termed 'stochastic' effects) to an acceptable level,

Table 1
Annual worldwide average *per caput* effective doses from natural and man-made sources in year 2000 (from UNSCEAR, 2000)

	mSv
Natural background	2.4
Diagnostic X-ray procedures	0.4
Atmospheric nuclear testing	0.005
Chernobyl accident	0.002
Nuclear power production	0.0002
Total	~2.8

based on comparisons of risks in safe industries. In the 1960s, dose limits were prescribed for particular organs of the body. The importance of risks at low doses was emphasised by the recommendation that "all doses shall be kept as low as reasonably achievable, economic and social considerations being taken into account" (ICRP, 1966). The LNT assumption allowed the quantity 'collective dose', i.e. summing doses to individuals over space and time, to describe the overall detriment. Tools such as cost benefit analysis were used to judge the extent of optimisation of protection. Application of dose limitation to the public was achieved by setting limits at one-tenth of those for radiation workers, at the equivalent of 5 mSv y^{-1} for doses other than from natural sources and medical exposures. This limit was to apply to the 'critical group', that small group of those whose habits and customs caused them to receive the highest exposures, thereby ensuring that others were also protected.

In 1977, further recommendations (ICRP, 1977) introduced a more formal system of dose limitation with the principles of justification, optimisation and compliance with dose limits. The differences in organ radiosensitivity were dealt with by introducing a quantity 'effective dose equivalent' with appropriate organ weighting factors and an overall limit to effective dose equivalent. This was set at the same level as before for workers (50 mSv y^{-1}) and the public (5 mSv y^{-1}).

Significant developments took place over the next few years, such that further recommendations ('ICRP-60') were published in 1991 (ICRP, 1991) and these are the basis of the standards in place today. One main development in ICRP-60 was the distinction drawn between 'practices' (which add exposures) and 'interventions' (those intended to reduce exposures, e.g. action in an emergency or remediation of contamination already present). For practices, the principles of justification, optimisation and limitation continue; however, the concept of a 'constraint' to optimisation was introduced to cater inter alia for multiple sources. Constraints are to be applied by radiation protection authorities for particular types of practice. A new quantity 'effective dose' was defined with revised weighting factors from the previous effective dose equivalent. The limit on effective dose to members of the public for practices was set at 1 mSv y^{-1}. For interventions, only the justification and optimisation elements of the system of control are relevant.

Radiation risks have been subject to intensive study and form part of the basis for ICRP standards. The most significant source of data has been the study of the Japanese atomic bomb survivors. Current estimates for stochastic effects, which have not changed a great deal from those used in ICRP-60, suggest a nominal fatal cancer risk of 5×10^{-5} per mSv for a population of all ages. The ICRP-recommended dose limit of 1 mSv y^{-1} for members of the public is consistent with a level of risk between 1 in 10^4 and 1 in 10^5 as the maximum tolerable involuntary risk for a member of the public.

The recommendations of ICRP-60 have been widely adopted by international organisations. The International Atomic Energy Agency (IAEA) have incorporated them into their Basic Safety Standards (IAEA, 1996), adopting also as trivial

(a) a level of dose of 10 μSv (equivalent to a fatal cancer risk of less than 1 in 10^6) and
(b) an annual collective dose level of 1 man-Sv for practices.

Since publication, the ICRP-60 recommendations have been subject to interpretation and clarification by ICRP. With relevance to radioactive waste disposal, ICRP Publication 77 (ICRP, 1997) sets out some interpretive policy principles. A constraint level for a single

source of no more than 0.3 mSv y^{-1}, implying a maximum tolerable risk of about 1 in 10^5, is deemed appropriate. Where environmental monitoring is used to assess doses from a combination of sources, it is suggested that control should be based on a dose to the critical group approaching 1 mSv y^{-1}. ICRP Publication 77 also recognises the problems on the use of collective dose, which had been perhaps over-emphasised previously, in summing over long future timescales for which large uncertainties would arise. The suggestion is to divide the assessment into blocks of collective dose received at different rates. It is becoming conventional that meaningful assessments of collective dose would only be available up to 500 years into the future.

ICRP Publications 81 (ICRP, 1998) and 82 (ICRP, 1999) give further guidance on radiation protection as applied to long-lived waste disposal and prolonged exposures, which are relevant to radioactivity in the oceans. The uncertainties in calculating collective doses over long time period are stressed. Earlier recommendations on applying dose limitation to critical groups of members of the public are effectively endorsed.

It is also to be noted that the ICRP-60 recommendations of ICRP are at present under review. Suggested changes have been put forward (Clarke, 1999) which have been under discussion. Revised recommendations are due for promulgation in 2005.

1.4. *Application of standards to radioactivity in the oceans*

Internationally, guidance on application of radiation protection standards to ocean radioactivity has been provided by the IAEA. Major drivers for this role are its relationship to the UN and its designation as the competent authority on radioactivity issues for the Convention of Prevention of Marine Pollution by Dumping of Wastes and Other Matter (The London Convention 1972, previously the London Dumping Convention). The 1961 document "Radioactive Waste Disposal into the Sea" (IAEA, 1961) reviewed the extent of radioactive waste problems and application of then-current radiation protection principles both to sea dumping and pipeline discharges. This document was later updated (IAEA, 1983) to include developments in radiation protection standards. From about that time, and following the cessation of sea dumping of solid radioactive wastes in the 1980s, the needs for international guidance on (a) liquid discharges via pipelines and (b) definitions of 'de minimis' levels of radioactivity in solid wastes developed more separately.

Liquid effluent discharges via pipelines are controlled by appropriate authorisation procedures of each country. Principles for releases of liquid and gaseous effluents and setting of authorisations were reviewed by the IAEA in 1986 (IAEA, 1986) and most recently in 2000 (IAEA, 2000), taking account inter alia of recent ICRP developments, from which were derived the IAEA Basic Safety Standards (IAEA, 1996). Many countries' procedures are subject to regional agreements. One example is the Convention for the Protection of the Marine Environment of the North-east Atlantic (the OSPAR Convention). This Convention, which came into force in 1998 following amalgamation of former Oslo and Paris Conventions, commits contracting parties to take all possible steps to prevent and eliminate pollution of the marine environment of the NE Atlantic by applying the precautionary approach and best available technologies and practices. The Ministerial statement of the contracting parties to OSPAR from the meeting at Sintra, Portugal, in May 1998 included a commitment to progressive and substantial reductions in discharges, emissions and losses of radioactive substances, with

the aim of achieving concentrations near background for naturally occurring radioactive substances and close to zero for artificial radioactive substances.

As regards solid radioactive wastes, dumping of high-level waste has always been prohibited by the London Convention. Since 1993, all radioactive wastes have been prohibited from sea dumping by an amendment to the Convention, and this has now been ratified by most countries. However it was recognised that all materials contain some radioactivity, including natural radioactivity, and contamination from pre-existing activities, thus it was necessary to define so-called 'de minimis' levels. The expression 'de minimis' (literally, 'concerning small things') embraces two legal concepts. The first is that of 'exclusion', where sources are not amenable to control, as in the case of unenhanced natural radioactivity. The second is the concept of 'exemption', such that even though sources would otherwise fall within the ambit of control, they are so low as to be trivial and can safely be disregarded.

The IAEA (IAEA, 1999c) has provided guidance for judgements on whether the concepts of exclusion or exemption can be applied to materials for sea dumping, or whether a specific assessment is needed. The basic radiological criteria given earlier are used, i.e. <10 μSv y^{-1} to members of the public or <1 man-Sv y^{-1} collective effective dose from the dumping practice. This IAEA guidance is currently being developed by the London Convention into a stepwise screening process to enable judgements on whether candidate materials may be further considered for sea dumping.

1.5. *Protection of non-human species*

The established system of radiological protection is based on the protection of humankind. In its recommendations of Publication 26 (ICRP, 1977), ICRP stated its belief that "if man is adequately protected then other living things are also likely to be protected". The more recent recommendations of Publication 60 (ICRP, 1991) included the rider "Occasionally, individual members of non-human species might be harmed, but not to the extent of endangering whole species or creating imbalance between species". Dose limits to humans are set at low levels such that the ICRP belief appears to have largely protected the environment from observable harm, but environments exist where this belief is open to challenge. For example, certain ocean environments (Pentreath & Woodhead, 1988), or contaminated areas from which people have been evacuated (UNSCEAR, 1996), involve remote pathways for exposure of people, and the potential exists for harm to the ecosystem, without the ICRP dose limit of 1 mSv y^{-1} for members of the public being exceeded. Consideration of the environment has in recent years been emphasised through the development of the concept of Sustainable Development and the Rio Declaration (UNCED, 1992). Thus there has emerged the need to develop a more explicit framework for radiological protection of the environment.

Discussions to develop such a framework are taking place at international (e.g. IAEA, 1999b; ICRP, 2003) and regional levels (e.g. the European Framework for the Assessment of Environmental Impact (FASSET, 2003). The following key elements of such a framework have been identified (IAEA, 1999b): specification of endpoints of concern; protection criteria; methods of dosimetry; and how to demonstrate compliance.

Consideration of endpoints involves, inter alia, whether harm to the individual or harm at the population level is of primary importance. Whilst for particular protected species harm at the individual level may be of concern, it is more usual to consider harm at the population

level. Most radiosensitive stages of the life cycle are those associated with reproduction and effects on gametes and embryos. However, genetic damage can be observed in organisms at dose rates well below those at which reproduction is impaired, and this also needs to be taken in account.

Appropriate protection criteria depend upon data on observable harm. Considerable information is available. A review by IAEA (1992) concluded that there is no convincing evidence that chronic radiation dose rates below 1 mGy d^{-1} will harm annual or plant populations. More recently, the detailed review by UNSCEAR (1996) concluded that detrimental effects on the most sensitive populations would not be expected at dose rates below 1–2 mGy d^{-1} for low-LET radiation. This has been confirmed more recently in the FASSET project (FASSET, 2003) in which it is observed that the dose rate threshold for statistically significant effects in most studies is about 10^2 μGy h^{-1}. By comparison, natural background dose rates in normal marine environments are contributed mainly by alpha-emitting ^{210}Po with absorbed dose rates to gonads of up to several μGy h^{-1}, at least an order of magnitude lower.

Dosimetry for protection of humans involves the use of 'equivalent dose' and 'effective dose' (measured in sieverts) which combine a radiation quality factor with the basic 'absorbed dose' (measured in grays). The extent to which absorbed dose should be modified for non-human species could well vary, making a parallel to equivalent dose uncertain in its application. Therefore, the quantity 'absorbed dose' remains the basis for assessing effects on non-human species.

A number of different approaches have been proposed for the purpose of demonstrating compliance. In particular a 'reference flora and fauna' approach is proposed (ICRP, 2003) based on choices of appropriate reference organisms such that other species in the reference group would also be protected; this approach would complement practice elsewhere in radiological protection. Along the principles of earlier work (e.g. Woodhead, 1979; Pentreath & Woodhead, 2001) a hierarchy of reference models is being developed (FASSET, 2003; ICRP, 2003) to assess absorbed dose rates based on radioactivity concentrations derived either from measurement, or calculated from releases to the environment and appropriate models as discussed later in this chapter. In a radiological assessment the absorbed dose rates so estimated for the critical reference species would be compared with the protection criteria derived as described above.

2. Assessment methodology

2.1. *Introduction*

Assessment of doses to man or the environment involves consideration of the potential pathways by which radioactivity can be transmitted through the environment and lead to exposure. The contributions to exposure from each pathway will be additive, but in many cases a particular pathway (the critical pathway) will dominate. So too will exposures to a particular group of people (the critical group) or type of organism. Dose limits and constraints apply to the doses to the critical group, thus this is the most usual type of assessment. This section concentrates on critical group assessments but adaptation can be made to estimate collective doses to larger populations.

Assessments are often classed as 'prospective' (i.e. predictive of doses due to a proposed release scenario and often carried out for the purposes of setting authorised limits) and 'retrospective' (i.e. looking back at the effects of an existing or former scenario, often done to judge compliance with dose limits). Both types of assessment rely on appropriate models, but the retrospective assessment can make use of measured concentrations of radioactivity in the environment as a result of monitoring programmes provided these levels are detectable.

The complexity of the models chosen for an assessment needs to suit its objectives. Initial calculations are often done assuming pessimistic models and conditions for 'screening' purposes, to identify whether a route of exposure warrants further investigation. Many authorisation assessments particularly for the aquatic environment can assume a steady release rate and an equilibrium situation, suitable for the 'concentration factor' method of calculation (see Section 2.3). However, a more complex dynamic model or 'systems analysis' method (ICRP, 1979) may be needed in assessing time-dependent or short-term releases. In recent years and with advances in computing power, 'probabilistic' codes have been developed in which particular input parameters can be sampled against defined distributions and a distribution of results obtained after iteration. These codes are useful for studying the uncertainties in results of model assessment due to uncertainties and variabilities in input parameters.

The main pathways for exposure of humans due to ocean releases of radioactivity are represented in Fig. 1. The assessment procedure is generally broken down into models representing the separate pathways and processes, usually

(a) dilution and dispersion mechanisms and particulate interaction to estimate water concentrations;
(b) uptake by biota and deposition onto sediments;
(c) dose to man by internal exposure;
(d) dose to man from external pathways.

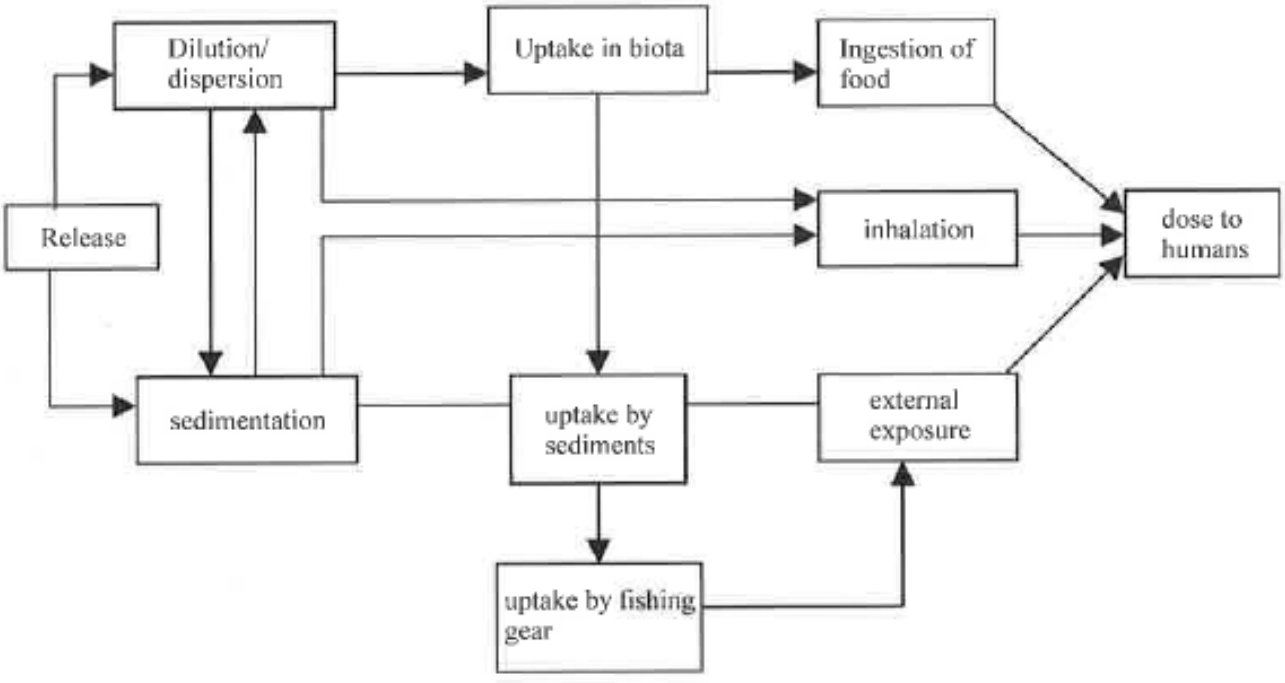

Fig. 1. Pathways for human exposure from ocean releases of radioactivity.

These parts of the overall assessment are described in more detail in the following sections.

2.2. *Dilution, dispersion and particulate interaction*

The objective of this part of the assessment is to derive concentrations of radionuclides in seawater from data on radionuclide releases. Calculations generally rely on mathematical models; there is a range of types and complexities, the choice depending on the objectives of the assessment. The main types are summarised as follows.

Simple screening model
This type is usually used initially with pessimistic parameters, to investigate the need for further study. At its simplest level, sediment interaction may be ignored, and the calculation reduces to:

$$C_i = \frac{\dot{Q}_i}{kV}, \tag{1}$$

where:

C_i is the concentration of radionuclide i and
$\dot{Q}_i$ its release rate into a compartment of volume V with fractional removal rate k (units chosen to be consistent).

k is given by:

$$k = \lambda_i + \frac{\dot{v}}{V}, \tag{2}$$

where:

λ_i is the radioactive decay constant of nuclide i and
$\dot{v}$ is the volume exchange rate of the compartment volume V.

For a coastal site the compartment may be taken to be the immediate tidal area whose dimensions and exchange rate can be obtained from hydrographic charts.

Sediment interaction
Following the most basic screening it is important to consider sediment interaction at an early stage because of the large reductions in concentrations which can be brought about for particle-reactive radionuclides. The interaction is usually described by the distribution coefficient, k_d, which is the ratio between the equilibrium concentration on sediment (usually expressed as dry weight) and the concentration in the water phase. (Units of k_d: $1\,kg^{-1}$) Values of k_d's for pelagic ocean and coastal sediments have been compiled by IAEA (IAEA, 1985, 2004). The factor f_i by which the water concentration of nuclide i is reduced due to sediment adsorption is given by:

$$f_i = \frac{1}{1 + k_{di} S}, \tag{3}$$

where S is the suspended particulate load in kg l^{-1}.

Advection/diffusion models
These models take account of advection (water movement) and diffusion in the marine environment and are generally used to determine concentrations in the vicinity of a release point up to several tidal ranges away, in which the 'critical group' might be exposed. Simple examples are described by IAEA (IAEA, 2001).

Interlinked compartment ("box") models
A simple model near a release point may consist of one compartment with concentrations calculated using some or all of the methodology given above. For calculations at greater distance, for example to investigate collective doses or effects of discharges on other countries, interlinked box models are often used. These models assume well-mixed behaviour within a given box, which is often both convenient and adequate for radiological calculations; transfer parameters are defined between boxes based on hydrographic data. Sedimentation and other loss (or gain) terms are added. One example of such a box model, for the European coastal environment, is described by the European Commission (EC, 1994). The NE Atlantic part of this model has 44 compartments with associated exchange and sediment parameters. A subset of this model was tested as part of the European MARINA study (CEC, 1990) (see Section 3.4) and found to give broadly good agreement with measured concentrations of ^{99}Tc, ^{137}Cs and $^{239+240}$Pu. The MARINA study has since been updated as MARINA II, with 72 compartments (EC, 2002a). A further example of a large box model is that used by OECD (NEA) to assess exposures due to dumping of solid radioactive wastes in the North Atlantic (NEA, 1985). This is described more fully in Section 3.5.

Finite difference models
These models are generally more complex and based on solving appropriate equations on a grid, either in two or three dimensions. The desired range of physical processes can be represented: hydrodynamic and transport equations with tidal forcing; wind driven flows; sediment transport; seabed deposition and erosion interactions. Recent examples of this type of model are described for the Irish Sea (Aldridge, 1998) and the English Channel (Salomon et al., 1993; Perianez & Reguera, 1999). Both models produce results broadly in line with values derived from observations.

2.3. *Uptake by marine biota and sediments*

The next step in the assessment methodology is to derive radionuclide concentrations in marine biota and on intertidal sediments from concentrations in seawater. In many cases the 'concentration factor' approach may be used, based on equilibrium concentration ratios between the material (Bq kg^{-1}) and filtered seawater (Bq l^{-1}) (ICRP, 1979). The 'systems analysis' approach (ICRP, 1979) is used for time-dependent calculations (e.g. following a short-term release); this requires further information such as characteristic response times in environmental media. Concentration factors (CFs) for the marine environment have been compiled by IAEA (IAEA, 2004) as consensus values derived from ranges found in the literature.

Values necessarily encompass a range of species in each of the classes of fish, crustaceans and molluscs into which it has been found reasonable to group similar behaviour and generally similar CFs. However the following pervading principle in radiological protection applies: "if data more appropriate to the specific situation can be justified, it should be used". One well-known example is the CF for ^{99}Tc in crabs (*Cancer pagurus*) and lobsters (*Homarus gammarus*), where values of 16 and 1200 respectively (e.g. Smith et al., 1998) have been observed; the IAEA (2004) value for crustaceans of 10^3 would seem to be pessimistic in the case of crabs.

For the purposes of calculating external exposure of people occupying intertidal areas it is important to derive concentrations of radionuclides on sediments. Using the 'concentration factor' approach, equilibrium concentrations are derived by multiplying the radionuclide concentration in seawater by the appropriate factor or distribution coefficient (k_d). Compilation of these factors and coefficients may be found in (IAEA, 2004). As for biota, they are consensus values derived from ranges found in the literature, and can err on the side of caution. It is important to consider local conditions. Wide differences can occur depending on sediment composition and particle size. For coarse-grained sands to be found on beaches, a reduction in k_d compared with muddy sediments may be justified for a realistic assessment.

2.4. *Assessment of internal exposure*

Following the recommendations of ICRP (e.g. ICRP, 1991) exposures are calculated as 'effective doses' weighted over the organs of the body (see Section 3). For internally deposited radionuclides, which (depending on half life) can deliver a dose throughout a lifetime, assessment is made of the 'committed effective dose' which integrates the effective dose received due to an intake. For adults, 50 years is assumed to be the average remaining lifespan at the time of intake; for children, a longer integrating time is taken depending on age. It is the committed effective dose received due to one year's intake which requires to be compared with the ICRP-recommended dose limit for the public of 1 mSv y^{-1}; the appropriate consumption or inhalation rate for this comparison is that of an average member of the critical group (see Section 1.3). Thus the annual committed effective dose $E_{i,\mathrm{ing}}$ due to radionuclide i is given for ingestion by:

$$E_{i,\mathrm{ing}} = e_{i,\mathrm{ing}} \sum_j C_{ij} R_j, \tag{4}$$

where:

$e_{i,\mathrm{ing}}$ is a nuclide-specific factor called the 'dose coefficient' (earlier term: dose per unit intake);
C_{ij} is the average concentration of radionuclide i in food j; and
R_j is the amount of food j eaten in a year.

Inhalation of radionuclides in air or sea-spray is usually of lower radiological significance

than ingestion, because of concentration mechanisms in biota. However, the annual committed effective dose $E_{i,\text{inh}}$ due to inhalation is similarly given by:

$$E_{i,\text{inh}} = e_{i,\text{inh}} \sum_j a_{ij} B_j, \tag{5}$$

where:

$e_{i,\text{inh}}$ is the dose coefficient for inhalation for radionuclide i in the appropriate form,
a_{ij} is its annual average air concentration in location j, and
B_j the annual volume of air breathed in each location.

Appropriate dose coefficients are calculated on the basis of metabolic models derived by ICRP. The most recent compilation is given in ICRP Publication 72 (ICRP, 1996). Dose coefficients for a wide range of nuclides are given for babies (3 months), infants (1 year), 5, 10 and 15 year-old children and adults. Doses to the embryo and foetus due to intakes by the mother have also recently been calculated (ICRP, 2001) but these are critical for relatively few radionuclides.

Depending on the metabolism of the particular radionuclide in the body, the dose coefficient will be sensitive to assumptions on particular parameters assumed for modelling purposes. One of the potentially most significant parameters is the gut transfer factor, or 'f_1 value'. Many observations of parameters are based on animal studies, and for a particular parameters can vary widely. Much work has been done on the metabolism of plutonium and related elements (ICRP, 1986), and it may be argued that available data do not permit an f_1 value for plutonium to be proposed more precisely than an order of magnitude. In addition, actual parameters are likely to vary significantly between individuals. ICRP practice is to choose parameters to err on the side of caution so as to produce a maximising result in calculating the dose to the critical group.

2.5. *Assessment of external exposure*

There are a number of potential pathways for external exposure; the more radiologically significant pathways usually involve exposure to tide-washed or entrained sediments because of their potential to adsorb radionuclides from seawater. Exposure to the water mass may also be considered, as when swimming or during activities on small boats. Exposure is due to beta particles or gamma rays; alpha particles do not feature because of their small range, being stopped in the upper layers of the skin.

Exposure to gamma rays from sediments

Irradiation due to photons from sediments contaminated by radionuclides is a potential exposure pathway for people who occupy intertidal areas during work or recreational activities. Prospective assessment of this pathway needs to be based on an appropriate model of gamma dose rates in air over sediments, whose radionuclide concentrations can be calculated as described in Section 2.4. Suitable models can be found in the literature (e.g. Beck & de Planque, 1968; Hunt, 1984), and radionuclide tabulations are also given based on photon transport

or Monte Carlo calculations (e.g. Kocher & Sjoreen, 1985; ICRU, 1994). It is important to choose models which correctly describe the depth distribution of radioactivity so as to allow for shielding by overlying sediments. For sandy beaches, where sediments are more uniformly contaminated with depth, a simple model based on energy conservation considerations could be adequate. This gives the dose rate D_i in air (μGy h^{-1}) due to radionuclide i as:

$$D_i = 0.288 C_i E_i, \tag{6}$$

where:

C_i is the concentration of the radionuclide (Bq g^{-1}) in sand and
E_i is the photon energy (MeV) per decay (Hunt, 1984).

In the case of muddy sediments there will generally be variation of radionuclide concentration with depth due to discharge history and radioactive decay. Often, assumption of an exponential distribution is adequate and models and tabulations have been proposed to describe this situation (Hunt, 1984; Kocher & Sjoreen, 1985; ICRU, 1994).

Exposure to beta particles

Beta particles from contaminated sediments present a source of exposure either to skin or to organs of the body near to its surface within the particles' range; the main exposure route in the latter category is of the male gonads. Beta particles of low energy (end-point energy less than about 0.1 MeV) are attenuated in the epidermis and may be neglected. Models and radionuclide tabulations of dose rates from skin contamination can be found in the literature (e.g. Kocher & Eckerman, 1987; Cross et al., 1992). A simple model (Hunt, 1984) is based on energy conservation considerations such that the dose rate D_i (μGy h^{-1}) due to radionuclide i is given by

$$D_i = 0.288 C_i E_i F, \tag{7}$$

where:

C_i is the concentration (Bq g^{-1}) of radionuclide i,
E_i is the mean beta energy for decay (MeV), and
F is a modifying factor to take account of attenuation of the beta particles by the epidermis.

A further allowance can be made for effective dilution of the sediments being entrained on the fishing gear (Hunt, 1984). It should be noted that for skin irradiation the ICRP recommends a deterministic dose limit for the public of 50 mSv y^{-1} (ICRP, 1991); there is also an addition to the effective dose with a weighting factor of 0.01.

Irradiation of the male gonads also requires consideration, as mentioned above, particularly during activities such as angling and wildfowling involving close proximity to sediments. The beta particles are significantly attenuated by clothing and overlying tissues of the gonads. For example, for the relatively hard beta particles from ^{234m}Pa (end-point energy 2.28 MeV) dose rates are reduced to about 2% of their level in air (Hunt, 1992); nevertheless there may still be a significant contribution to overall dose.

Exposure to swimmers and on boats

In this situation similar models to those described above may be used, on the basis that an immersed body will be subject to 4π (i.e. surrounding) rather than 2π (semi-infinite) geometry, and the appropriate concentration to be used is the bulk radionuclide concentration in the water including any mobilised sediment. Shielding by the hull of a boat could be allowed for. Simple models based on these principles have been described by Hunt (1984).

2.6. *Selection of critical groups and the role of habits surveys*

In assessing radiation exposures to members of the public, the concept of the 'critical group' formulated by the ICRP is of basic importance. It is generally not possible, due to a range of variabilities, to determine doses that might be received by members of the public individually (ICRP, 1966). Thus ICRP consider it is feasible to take account of these variabilities by the selection of appropriate critical groups. Such a group should be representative of those individuals expected to receive the highest dose; it is believed reasonable (ICRP, 1966) to apply the dose limit for members of the public to the mean dose to the critical group. The critical group should be small enough to be homogeneous with respect to age, diet and those aspects of behaviour which affect the doses received. Although these criteria were set down in the 1960s, recent recommendations have left them essentially unchanged (ICRP, 1991).

To assess doses due to a particular source it is generally necessary to derive consumption, inhalation and occupancy rates typical of the critical group. The dose to the critical group is the sum of the contributions of dose from all exposure pathways. In some countries, regulations for operational and especially pre-operational assessments specify rates to be employed, usually based on hypothetical, maximising assumptions. Where site-specific data are not available, IAEA have suggested 'default' values which can be used, and examples are given in Table 2. The usual radiological protection principle applies, that if more specific information is available it should be used.

More realistic assessments are based on people's habits which might obtain over the period of the practice, e.g. the discharge authorisation. Surveys of local habits (including consumption and occupancy rates) are the main method of deriving site-specific data. Such site-specific

Table 2
Default values of ingestion, inhalation and occupancy rates for maximally exposed adults given by IAEA (IAEA, 2001)

Food		Consumption rate (kg y^{-1})
Marine fish	(Europe)	50
	(Far East)	60
Shellfish	(Europe)	15
	(Far East)	20
		Breathing rate (m^3 y^{-1})
Inhalation		8400
Activity		Occupancy rate (h y^{-1})
Working/playing over contaminated sediments		1600

information is often important for the marine environment as consumption and occupancy habits tend to be more variable than in the terrestrial situation. Habits surveys are mainly used in the case of retrospective assessments to assess compliance with dose limits. However, they can provide a good guide to consumption and occupancy rates for prospective assessments over timescales of a few years, e.g. the lifetime of a discharge authorisation. Habits surveys also provide another essential function in radiation protection of the public, guiding the formulation of appropriate environmental monitoring programmes. Because the main objective of habits surveys is to identify the critical group, they are essentially different from random, market-research surveys, and concentrate on those people likely to be members of the critical group. Surveys need to be carried out with appropriately framed questions to cover all seasons of the year when, for example, fishing may or may not be carried out. Corrections need to be made for portion sizes and edible fractions of species consumed. Useful validation can be obtained by asking those surveyed to complete a diary of foods eaten at specific times during the year.

The shape of a distribution which might be obtained during a habits survey for consumption rate of a given food (e.g. fish from the vicinity of a nuclear site) is shown schematically in Fig. 2.

Selection of the observations which form the critical group may be based on the ICRP criteria of homogeneity referred to earlier. ICRP have interpreted (ICRP, 1985) that, to satisfy the homogeneity requirement, the range of maximum to minimum observations in the group should be no more than a factor of 3 if the mean effective dose to the group is more than 0.1 times the dose limit. Otherwise a factor of 10 is acceptable. Thus in Fig. 2, in which C_{max} represents the maximum consumption rate observed, the value of C_{min} (which describes the lowest rate for membership of the critical group) would be set to $C_{max}/3$. The value of $\overline{C}$ would then be the mean of all observations greater than C_{min}. The dose corresponding to $\overline{C}$ could be calculated, and following the above principles, if it proved to be less than 0.1 mSv, the calculation could be repeated using $C_{min} = C_{max}/10$.

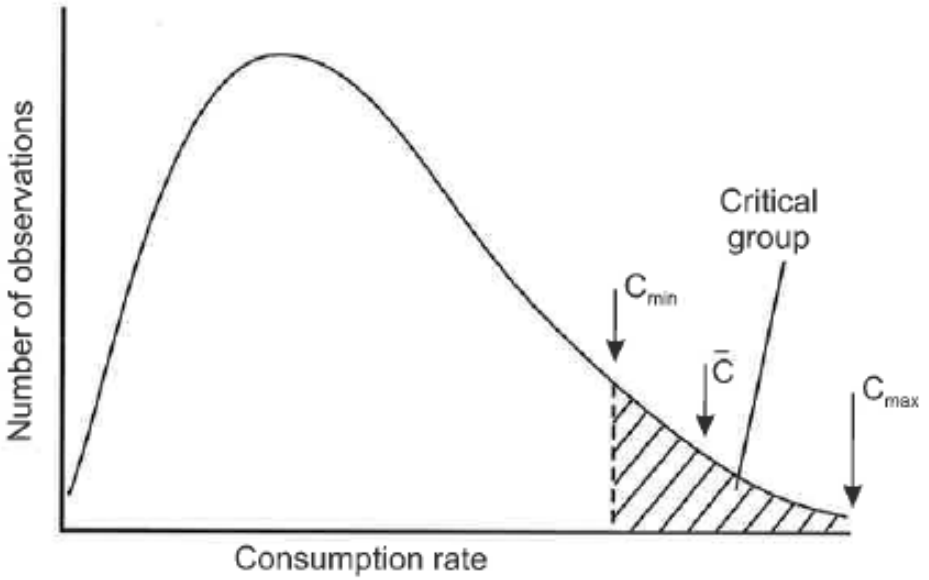

Fig. 2. Schematic distribution of consumption rate data from a habits survey.

The above illustration assumes that doses to the critical group are dominated by one particular pathway. In practice it is usually necessary to consider additive effects due to other contributing pathways, prior to selection of the critical group.

2.7. *Collective dose*

The collective dose (more strictly, post ICRP-60, collective effective dose) S (units: man-Sv) to a population of exposed people can be expressed as:

$$S = \sum_i H_i N_i, \tag{8}$$

where H_i [Sv] is the per caput effective dose to the ith group of people, of size N. The quantity S was recognised to be useful in the 1970s as a consequence of the linear, no-threshold hypothesis; it can express the overall detriment to a group of people, and can be an important input to optimisation as part of the dose limitation system (see Section 1.3). Because the man-Sv corresponds to a detriment which can be calculated from risk factors, it can be ascribed a monetary value. The concept can be used in cost-benefit calculations to assist decision making.

In the marine context, a particular example of use of the concept of collective dose was in identifying levels of interim control of discharges of radioactive waste from Sellafield to the Irish Sea (Handyside et al., 1982). In this example, the detrimental cost associated with collective dose due to discharges of radiocaesium was calculated, along with the cost of installing skips of absorbing zeolite in the fuel storage ponds. By increasing the number of skips, discharges could be reduced, at greater cost. Cost-benefit analysis was used to indicate the optimum number of skips to be installed.

The concept of collective dose remains a useful one but its limitations need to be recognised. First, there is the uncertainty inherent in summing up very small doses, not greatly in excess of natural background, to large numbers of people. Similarly, the summation of doses over timescales in the far future with assumptions on parameters such as the numbers of a population and its habits, will have significant uncertainties. Recent ICRP Publications 77 (ICRP, 1997), 81 (ICRP, 1998) and 82 (ICRP, 1999) have highlighted these uncertainties. In addition, suggested changes to the main recommendations (Clarke, 1999) reflect these developments. The NCRP have published a specific report on principles and application of collective dose (NCRP, 1995).

3. Comparative radiological assessment of sources of radioactivity in the oceans

3.1. *Introduction*

Sources of radioactivity in the oceans may be either natural or anthropogenic; those in the latter category have mostly occurred since the commencement of the nuclear age following the Second World War, and have been both weapons-related and for peaceful purposes. In this section, sources of radioactivity in the oceans are summarised and compared in terms of their

radiological effects. Most of the details of these sources and their distribution are described in greater detail elsewhere in this book.

3.2. *Natural radionuclides*

The main natural radionuclides present in the oceans are listed in Table 3 together with their estimated inventories in the world ocean. The nuclides ^{234}U, ^{226}Ra, ^{210}Pb and ^{210}Po are members of decay chains of the long-lived primordial parent ^{238}U. Thorium is essentially insoluble and rapidly removed to the ocean floor, thus concentrations of ^{232}Th and daughters in seawater are very low. By combining representative concentration data in seafoods with maximising consumption rates and relevant dose coefficients, it is possible to estimate the dose to a high-rate consumer. Estimates using recent dose coefficients and values of consumption rates are presented in Table 4. High-rate seafood consumers could receive up to about 2 mSv y^{-1}, with over 95% contributed by ^{210}Po, mainly due to its concentration in molluscan shellfish.

For collective dose purposes, a detailed study of the ^{210}Po component has been undertaken by the international MARDOS group (IAEA, 1995). This group collated data on measured concentrations of ^{210}Po in seawater, fish, crustaceans and molluscs from the major world fishing areas delineated by the Food and Agriculture Organisation (FAO). There were some variabilities, caused by different species caught in different areas, and a shortage of data for some areas. Dose calculations were carried out assuming appropriate concentration factors (IAEA, 1985) combined with seawater concentrations, and, in addition, doses were calculated using the biota data directly. An average delay time between catch and consumption of 1 half-life of ^{210}Po (138 d) was assumed. The Group calculated the world population collective committed effective dose commitment for fish and shellfish caught using FAO landings statistics for 1990. The results lay in the range 27,000–48,000 man-Sv; this gives an average per caput effective dose (i.e. per head of world population of 5.3×10^9 then used by the Group) in the range 5.1–9.1 μSv. It is noted that this dose is to a world average consumer of fish

Table 3
Main naturally-occurring radionuclides in the oceans (Pentreath, 1988)

Radionuclide	Concentration (Bq l^{-1})	Approximate inventory (TBq)
Primordial:		
^{40}K	12	1.6×10^{10}
^{87}Rb	0.11	1.5×10^{8}
^{234}U	0.046	6.3×10^{7}
^{238}U	0.040	5.5×10^{7}
^{226}Ra	0.0035	4.8×10^{6}
^{210}Pb	0.0030	4.1×10^{6}
^{235}U	0.0022	3.0×10^{6}
^{210}Po	0.0020	2.7×10^{6}
Cosmogenic:		
^{14}C	0.006*	8.0×10^{6}
^{3}H	0.1*	8.5×10^{5}

*Surface water values.

Table 4
Representative concentrations of natural radionuclides in seafoods and effective dose rates to high-rate consumers

Radionuclide	Concentration* (Bq kg^{-1}) (wet)			Dose coefficient† (Sv Bq^{-1})	Effective dose rate to high rate consumers (μSv y^{-1})	
	Fish	Crustaceans	Molluscs		(1)	(2)
^{210}Po	1.5	25	50	1.2×10^{-6}	1008	1980
^{210}Pb	0.04	0.2	3	6.91×10^{-7}	24	47
^{226}Ra	0.1	0.02	0.3	2.8×10^{-7}	2.6	4.6
^{234}U	0.012	0.12	0.3	4.9×10^{-8}	0.2	0.4
^{238}U	0.011	0.11	0.27	4.84×10^{-8}	0.2	0.4
Total					1035	2033

*Concentration data from (Pentreath, 1988) and references therein; it is to be noted that there are significant variations between species and sea areas (e.g. IAEA, 1995).
†(ICRP, 1996).
Note:
(1) Assumed high rates 60 kg y^{-1} fish, 10 kg y^{-1} crustaceans, 10 kg y^{-1} molluscs given for consumers in the far east (IAEA, 2001).
(2) Assumed high rates 100 kg y^{-1} fish, 20 kg y^{-1} crustaceans, 20 kg y^{-1} molluscs (NRPB, 1998).

and shellfish, as compared with the estimate given above for a high-rate consumer. The per caput results compare with an average effective dose due to natural background of 2.4 mSv (Section 1.2). It is to be noted that since these data were calculated, ICRP have increased the value for the dose coefficient for ingestion of ^{210}Po by a factor of 2.8; this would give an annual collective effective dose up to 134,000 man-Sv and annual per caput effective dose up to 25 μSv. In addition there is evidence to suggest that the dose coefficient for ^{210}Po could be higher by a further factor of 1.6 (Hunt & Allington, 1993).

3.3. *Anthropogenically-enhanced natural radionuclides from non-nuclear industries*

Naturally occurring radioactive materials (NORM) contribute to the discharges from many industrial processes, for example phosphate processing, oil and gas production, mining, ore processing and burning of fossil fuels. A review of these sources has been carried out by UNSCEAR (UNSCEAR, 2000) and for European waters in the MARINA II study (EC, 2002a). The main sources of enhanced natural radioactivity to the marine environment from non-nuclear industries are from the operations of the phosphate industry and the discharges from oil and gas production. These are described in more detail below.

In order to produce phosphoric acid, phosphate ores are acidified mainly with sulphuric acid; this produces phosphogypsum as a precipitate which contains small quantities of the calcium analogue, ^{226}Ra. In some cases the waste phosphogypsum is discharged into surface waters and enters the sea. In many developed countries the discharges have reduced or ceased in the last several decades due both to disposals on land and also to economic conditions, as it may be advantageous to import raw phosphoric acid. Exposures due to these operations have been mainly due to the ^{210}Po product of ^{226}Ra decay, and mainly through consumption of molluscan shellfish. Exposures are generally minor and only of local significance, being

dominated at distance by the unenhanced ^{210}Po as described in the previous section. A study of exposures near the phosphate plant at Whitehaven, UK (which ceased operations in 2001 and had used imported phosphoric acid since 1992 following which the discharges reduced significantly) showed that in the period 1962–1992 effective doses above background levels to the critical group of shellfish consumers rose to between 2 mSv y^{-1} and 6 mSv y^{-1}, but have now significantly reduced (Camplin et al., 1996). A survey of non-nuclear discharges to the OSPAR area (OSPAR, 1997) shows that phosphogypsum is released to the mouth of the Rhine by plant in the Netherlands, and to the confluence of the Rivers Odiel and Tinto in Spain. Effective doses above background levels are estimated at between 100–300 μSv y^{-1} to the critical groups, due mainly to ^{210}Po. Releases to the Rhine are estimated to result in a collective effective dose of 170 man-Sv y^{-1} to the Dutch population.

Pumping oil and gas from offshore platforms gives rise to 'produced water' which is contaminated by enhanced levels of natural radionuclides, predominantly ^{226}Ra, ^{228}Ra and ^{210}Pb. Whilst sources of enhanced natural radioactivity are more diffuse than phosphogypsum releases, in some areas they could become more important; in 1999, estimated discharges from oil and gas production accounted for 90% of the discharge of alpha-emitters into the OSPAR region (EC, 2002a). However, collective doses in this area are at present dominated by the effects of mainly past phosphate processing. The collective dose rate received by the EU population in 2000 was 91% due to NORM, of which 61% was due to phosphate processing, the remainder to oil and gas production (EC, 2002b).

3.4. *Weapons-test fallout*

Testing of nuclear weapons took place in the atmosphere from 1945 to 1980; further tests took place underground until 1998. Atmospheric tests have contributed to worldwide radioactivity input to the oceans, and as a greater amount of testing was carried out in the northern hemisphere, deposition amounts were greater there than in the southern hemisphere. Detailed reviews of the deposition have been undertaken by UNSCEAR (e.g. UNSCEAR, 2000), who have also kept worldwide exposures under review. Most of the exposures have been and are due to terrestrial pathways following deposition of radionuclides on land. Exposures are due to external radiation following this deposition and to ingestion of radionuclides taken up into foodstuffs, with a small contribution due to inhalation. Dose estimates by year presented by UNSCEAR (UNSCEAR, 2000) show that world average per caput doses due to weapons-test fallout peaked at about 113 μSv in 1963, reducing to about 6 μSv y^{-1} during the 1990s. Most of this exposure is now contributed by the longer-lived nuclides ^{14}C and ^{90}Sr (via ingestion) and ^{137}Cs (ingestion and external exposure). The contribution from fallout radionuclides in the oceans is relatively small and mainly due to seafood ingestion in the world population diet. It is fallout which has provided the main global source of ^{137}Cs to the world oceans, and an estimate of exposures due to oceanic ^{137}Cs has been given by the MARDOS group (IAEA, 1995). This group, following a similar procedure as for ^{210}Po (Section 3.2) collated data for measured concentrations of ^{137}Cs in seawater, fish and shellfish. Dose calculations were carried out assuming appropriate concentration factors with seawater concentrations and using the biota data directly. Using FAO landings statistics for 1990, the world collective effective dose commitment from ^{137}Cs in fish and shellfish caught that year was about 160 man-Sv using both methods, corresponding to an average per caput dose of 0.032 μSv. This represents

only 0.5% of the total per caput dose due to fallout of 6 μSv given above, due inter alia to the proportion of marine fish to world diet. However, the effective dose to a high-rate seafood consumer would be much greater. For example, concentrations of ^{137}Cs in cod from Icelandic waters (remote from effects of nuclear power production) in 2000 was about 0.2 Bq kg^{-1} (FSA & SEPA, 2001). Based on high-rate consumption of fish and shellfish of 80 kg y^{-1} (IAEA, 2001), the annual dose from fallout ^{137}Cs could be up to 0.21 μSv y^{-1}.

It is important to note that as nuclear tests were carried out in relatively remote areas, exposures of local populations are not likely to contribute significantly to world per caput doses from fallout. People who might now or in the future occupy formerly contaminated areas could receive enhanced exposures, and re-evaluation of these sites is being carried out. Some of these sites are in predominantly marine situations and two examples are summarised here. Surveys have been carried out throughout the Marshall Islands in the Pacific (including Bikini and Eniwetak Atolls) (Simon & Graham, 1996; IAEA, 1996). Effective doses from residual contamination to those who might return to Bikini Atoll were estimated at 4 mSv y^{-1} for a mixed diet and 15 mSv y^{-1} for a diet of solely local origin. An international investigation of the present radiological situation at the former test sites of Mururoa and Fangataufa (IAEA, 1998a) was carried out from 1996–1998. It was concluded that, though it is doubtful if these atolls could sustain a permanent population dependent upon local foods, a hypothetical population with a diet of local produce and seafood would not generally receive a dose attributable to the residual contamination exceeding 10 μSv y^{-1}.

3.5. *Operations of the nuclear power industry*

Nuclear electricity generation began in 1956, followed by rapid expansion until the 1990s, since when the rate of expansion has slowed considerably. Operations of the nuclear industry may be divided into mining and milling of the uranium ore; uranium processing, enrichment and fuel fabrication; reactor operation; fuel reprocessing; and solid waste disposal (liquid and airborne waste discharges being considered as part of the other operations). The mining and milling operations produce very little input of radioactive waste directly into the oceans. Solid radioactive waste dumping into the oceans has been prohibited by an amendment in 1993 to the London Convention and this has now been ratified by almost all countries; the effects of past dumping are discussed in Section 3.6. The effects of uranium processing, enrichment and fuel fabrication, reactor operation and fuel reprocessing are discussed in this section. The effects of reprocessing dominate because the discharges are greater in activity terms, as a result of the process which involves dissolution of irradiated fuel followed by separation of plutonium and uranium from the fission products.

Uranium ore is usually supplied as a concentrate which requires further processing to produce uranium metal, oxide or hexafluoride for enrichment. Separation of the uranium produces waste thorium and daughter products, particularly the ^{234}Th daughter of ^{238}U and its own daughter, ^{234m}Pa. The half-lives of these waste products are short (^{234}Th: 24.1 d; ^{234m}Pa: 1.17 min) thus in most cases storage in lagoons allows radioactive decay, and discharges to the wider environment are low. One exception is the Springfields plant, in the United Kingdom, which discharges to the River Ribble, but even here doses to the public due to discharges from the plant are small. In 2000, the critical group of anglers near the site re-

ceived only 0.02 mSv y^{-1} (Environment Agency, 2001). The process of enrichment involves only very small releases of waste products, especially to the marine environment.

There are a number of different types of nuclear reactor, and the waste discharges from reactor operations have different radionuclide compositions depending on construction materials, type of fuel, cladding, moderator, etc. Discharges tend to be small by comparison with fuel reprocessing, which involves dissolution of the fuel as referred to above. UNSCEAR (2000) have analysed distributions of discharges of radionuclides normalised to energy output for different reactor types and for reprocessing plants, and calculated collective effective doses based on generic assessments. These results include exposures through terrestrial pathways. A large proportion of the world collective dose due to discharges to marine waters is from operations of the Nuclear Industry in Europe. The seas of western Europe receive inputs from some 70 nuclear power stations (including those which discharge to rivers) and three reprocessing plants, namely Sellafield, which discharges to the Irish Sea; Cap de la Hague, which discharges to the English Channel; and Dounreay (where reprocessing has now ceased) which discharges to waters which enter the North Sea. The effects of these plants have been subject to extensive study since the 1960s as part of national and international programmes. A major international study of North European waters was the project MARINA (CEC, 1990). This has recently been reviewed as MARINA II (EC, 2002a), and both studies summarise critical group doses derived mainly from data from national monitoring programmes, using the concentrations of radionuclides in environmental materials combined with data on the habits of the critical groups. A summary of the results from the MARINA studies is shown in Table 5. For comparison are shown data from the 1970–1980s and more recent data, published by the Food Standards Agency and Scottish Environment Protection Agency (2001) and from the extensive study mainly related to the la Hague discharges carried out by the Nord-Cotentin Radioecology Group (2000).

It can be seen that the doses to critical groups have reduced in recent years, and this has been due to reductions in discharges; at Sellafield and la Hague, particularly, these reductions have been due to improved waste management procedures and treatment facilities. For both these sites, reconstruction of the history of doses to the critical groups has been carried out. The la Hague dose reconstruction study was part of the work of the Nord-Cotentin Radioecology Group (2000). Doses due to the Sellafield discharges from the 1950s to the 1990s are shown in Fig. 3 (Hunt, 1997). This reconstruction indicated that average doses to the critical group of fish and shellfish consumers were at a maximum of about 2 mSv y^{-1} in the mid-1970s. The doses were due to the level of releases of radiocaesium and actinides shortly before that time.

Both the original and revised MARINA studies also considered collective doses to the European Union population due to nuclear industry discharges to European waters. The maximum annual collective dose rate was about 280 man-Sv y^{-1} around 1978 due mainly to discharges from Sellafield, which had peaked several years earlier (EC, 2002a). This rate has now reduced to around 14 man-Sv y^{-1}. These data compare with an estimated collective dose rate to the EU population from natural background radiation of about 840,000 man-Sv y^{-1}. The truncated collective dose commitment from the start of nuclear operations up to the year 2500 from discharges to 2000 is calculated to be about 5100 man-Sv (EC, 2002b). The collective dose to the rest of the world due to discharges to EU waters is around an order of magnitude lower than for the EU population (CEC, 1990).

Table 5
Annual individual (critical group) doses from direct inputs to northern European waters

Source	Annual effective dose to critical group (mSv)	
	MARINA studies (data maxima for 1970–1980s)	Recent data
Sellafield	~2	0.15[b]
Dounreay	<0.05	<0.005[b]
La Hague	~0.2	0.026[c]
Other nuclear sites	0.0001–0.3[a]	<0.07[b]

[a]Doses towards the higher end of this range are largely due to the influence of Sellafield.
[b]Data for 2000. Reference: Food Standards Agency and Scottish Environment Protection Agency (2001).
[c]Data for 1996 and apply to the group of Huquets fishermen. Reference: Nord-Cotentin Radioecology Group (2000).

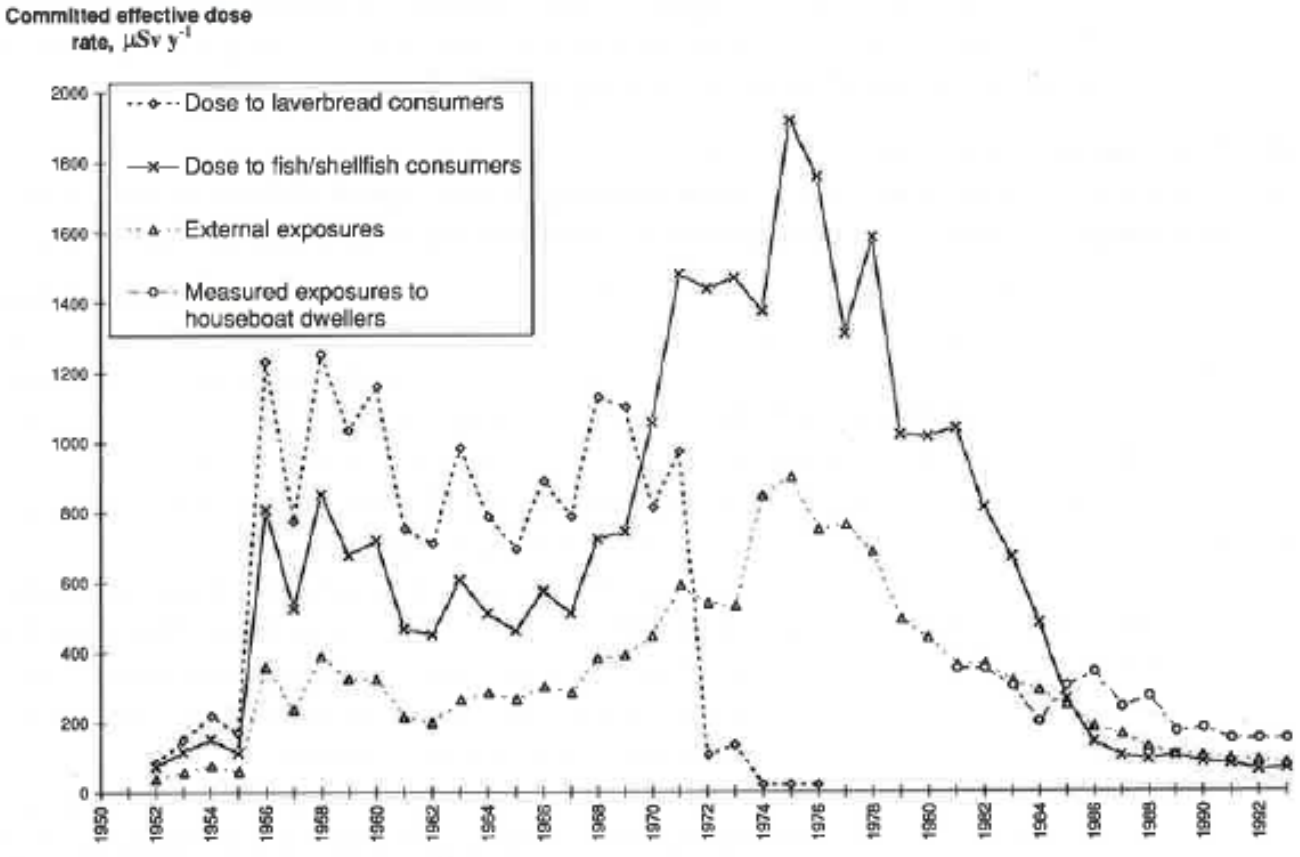

Fig. 3. Doses to critical groups near Sellafield, in 1950–1990 (Hunt, 1997).

3.6. *Ocean dumping of solid radioactive waste*

This practice first took place in 1946 at a site some 80 km off the coast of California. Since then, similar dumping has taken place at a range of sites in the world ocean. Almost all countries have now ratified the 1993 prohibition of this practice by the London Convention. The

history of radioactive waste dumping and the inventory of radioactivity dumped are dealt with in greater detail in Chapter 4, and the data have been published by the IAEA (1999c). Totals of 0.55 PBq were dumped in the NE Pacific, 2.94 PBq in the NW Atlantic, 42.32 PBq in the NE Atlantic, 38.37 PBq in the Arctic, and 0.89 PBq in the W Pacific, in each case at a range of different sites. The greatest amounts were dumped in the NE Atlantic and in the Arctic; the effects of the latter disposals are considered in Section 3.7.

Disposals in the NE Atlantic took place from 1949 to 1982 and consisted of waste from a number of European countries; the majority of the operations were in the later years and were coordinated internationally. From 1977 to 1982 operations were carried out under the Multilateral Consultation and Surveillance Mechanism coordinated by the Nuclear Energy Agency (NEA) of the Organisation for Economic Cooperation and Development (OECD). In 1980 a Coordinated Research and Environmental Surveillance Programme (CRESP) was set up by NEA to provide scientific assessment of continuing dumping operations in the NE Atlantic. The radiological impact of the disposals has not been measurable, such that environmental monitoring does not provide a suitable method of radiological assessment, and the emphasis has been on the use of suitable mathematical models to assess the radiological impact. Such an assessment was carried out as part of a review of the continued suitability of the 1974–1982 dumpsite (NEA, 1985). The radiological assessment was carried out in four parts. First, doses to critical groups were calculated for three dumping scenarios:

(a) past dumping (i.e. to 1982);
(b) past dumping plus a further five years of dumping at rates typical of those of 1978–1982;
(c) past dumping plus five year's dumping at ten times the rate of dumping in 1978–1982.

The last two scenarios were chosen to indicate the possible radiological impact rather than being a prediction of what might be dumped. In the second part of the assessment, doses to marine organisms for these scenarios were calculated and compared with natural background levels and levels at which detrimental effects on marine organisms have been observed. In the third part, the sensitivity of calculated doses to critical groups to variations in key parameters was investigated to study uncertainties and identify priorities for future work. The fourth part produced preliminary estimates of collective doses from dumping at this site.

The assessment was based on a system of linked mathematical models. A release-rate model calculated the releases of radionuclides from different types of waste package. Outputs fed a world ocean dispersion model which was of the box type with 92 water compartments, overlaid with a model to represent sediment interactions. The output from the dispersion model fed a radiological model which calculated doses to man and to organisms.

The results of the first part of the assessment showed that even for scenario (c), peak doses would at most be 0.1 μSv y^{-1} (to notional high-rate mollusc consumers in the Antarctic). For scenario (a), actual past dumping, the peak individual dose was estimated to be 0.02 μSv y^{-1}. Doses to marine organisms were, as expected, highest within the dump site area and whilst for scenario (c) they were greater than the natural background, dose rates were at least two orders of magnitude below the level at which deleterious effects have been observed. The sensitivity analyses showed that exposures (whilst still small) were most sensitive to values chosen for sediment interactions and radionuclide concentration factors. Further, if a marine food pathway existed which provided a 'short circuit' from the deep ocean to man, then individual doses might be higher than the more realistic pathways, but would still be low. The collective

dose assessment showed that for scenario (c) the highest collective dose commitment would be 3×10^5 man-Sv over the world population, and delivered fairly uniformly over 10^4 years. For scenario (a), the commitment would be 4×10^4 man-Sv over the same time, the collective dose rate being ~4 man-Sv y^{-1} achieved some decades after the start of dumping. As for other collective dose calculations, the results are composed of very low doses spread over large populations and timescales, and are subject to considerable uncertainties.

Overall, the conclusion of the study was that the dump site could continue to be used for dumping of packaged radioactive wastes for a further five years. Dumping rates during this period of ten times the average for 1978–1982 were shown to give insignificant doses to the public and benthic species, but the study recommended that if these rates were exceeded, the suitability of the site should be reconsidered. Despite the results of this review, however, for political reasons dumping in the NE Atlantic was not resumed.

3.7. *Radioactive waste dumping in the Arctic seas*

Following the demise of the Former Soviet Union, it became clear in the early 1990s that for over three decades significant amounts of radioactive wastes had been dumped in shallow waters of the Arctic seas. Several international initiatives for studying the effects of radioactivity in the Arctic region including terrestrial effects have been pursued, but to address this dumping specifically, an international study was proposed by the IAEA, and in 1993 the International Arctic Seas Assessment Project (IASAP) was set up. The objectives were: to assess the risks to human health and the environment associated with the wastes dumped in the Kara and Barents Seas, and (principally at the request of contracting parties to the London Convention) to examine possible remedial actions and to advise on whether they were necessary and justified.

Russian estimates of the amount of radioactive waste dumped indicated some 90 PBq at the time of dumping, but an assessment based on reactor operating histories reduced the estimate of the total activity dumped to 37 PBq. Most of the activity was in high level wastes in reactors with and without spent fuel, mainly from submarines but also from the nuclear-powered icebreaker '*Lenin*' and dumped mainly in the Kara Sea, particularly in the shallower fjords of Novaya Zemlya. The depths of the dumping sites were 12–135 m and in the Novaya Zemlya Trough at depths up to 380 m. Details of the dumping are given in the IAEA Inventory of Radioactive Waste Disposals at Sea (IAEA, 1999c). The inventory in 1994 after radioactive decay since dumping was estimated by IASAP to be 4.7 PBq of which 86% were fission products, 12% activation products, and 2% actinides (IAEA, 1998b).

The IASAP examined the current radiological situation in the Arctic based on data from joint Norwegian–Russian cruises and other international monitoring surveys. Measurements of environmental materials suggested that annual individual doses from artificial radionuclides are in the range of only 1–20 μSv. Whilst there were elevated levels of radionuclides detected in some sediments within a few metres of some low level waste containers, indicating leakage, the outer parts of the fjords did not show measurable increases. Thus at present the wastes have a negligible radiological impact.

Potential risks posed by possible future releases focused on the high level sources dumped. Release rates were estimated and radiation doses to humans and biota were assessed using appropriate models. For each of the dumped high-level waste objects the barriers to release

were examined, weak points identified, and corrosion rates taken into account. External events such as collisions and glacial action were also considered. Three release scenarios were considered: a best estimate on the basis of gradual corrosion; a plausible worst case involving catastrophic release of two sources at one dumpsite in the year 2050; and a climate change scenario involving release due to glacial action in the year 3000. A number of different models was used to calculate dispersion of the radionuclides within and from the Arctic Ocean, both of the box model and hydrodynamic type. For estimation of doses to individuals, three population groups were considered:

1. A group whose subsistence is heavily dependent on local Kara Sea fish and other marine produce and activities.
2. A hypothetical group of military personnel patrolling the foreshores of the fjords containing dumped radioactive materials.
3. A group of seafood consumers representative of the population of N Russia on the Kola peninsula eating seafood from the Barents Sea.

The resulting peak doses to these groups under the three postulated scenarios are shown in Table 6 (IAEA, 1998b). The total annual individual doses to seafood consumers (groups 1 and 3) are very small and much less than variations in natural background. Doses to the hypothetical group of military personnel are higher due to proximity to the releases but still comparable to natural background doses.

Collective doses were also estimated for the best estimate scenario. The collective effective dose up to the year 3000 is of the order of 10 man-Sv. Doses to marine organisms were also estimated, the maximum being about 0.1 $\mu Gy\ h^{-1}$, well below levels likely to cause detrimental effects to populations.

Potential remediation was considered as part of the Project. A particular example was chosen, that of dealing with the container of spent fuel from the icebreaker '*Lenin*'. Feasible options after screening out of the less practicable ones involved in situ capping with concrete or other suitable material, or recovery for land treatment and disposal. The costs of the marine operations (i.e. not including any subsequent transport and land disposal) were estimated at US $6–10 million. The radiological protection aspects of the justification for remediation were considered. Individual doses resulting from leaving the wastes in place would be small as described above. Nevertheless, when summed over time and over a population some health effects might be predicted, thus the collective dose was considered further. A simple cost-benefit argument was applied which indicated that remedial measures costing in excess of US $200,000 would not appear to offer sufficient benefit to be warranted. Therefore, it was

Table 6
Maximum total annual doses for postulated Arctic groups and scenarios

Scenario	Annual doses (μSv)	
	Seafood consumers (groups 1 and 3)	Military personnel (group 2)
1. Best estimate	<0.1	700
2. Plausible worst case	<1	4000
3. Climate change	0.3	3000

concluded that, on radiological grounds, remedial steps are not justified. However, it was concluded that institutional controls on occupation of beaches and use of coastal marine resources and amenities in the fjords of Novaya Zemlya must be maintained.

3.8. *Effects of the Chernobyl accident*

Radioactive debris from the unit 4 accident at Chernobyl, Ukraine, on 26 April 1986 caused widespread contamination over much of Europe, and was detectable in more distant parts of the northern hemisphere. A number of European sea areas were affected as a result of changes in wind direction following the accident, and subsequent rainfall during passage of the plumes. The main sea areas affected were the Baltic Sea, the Black Sea and the Irish Sea. Effects were also observed in the Mediterranean (e.g. Papucci et al., 1996), the Aegean (e.g. Polikarpov et al., 1991) and the North Sea (e.g. Mitchell & Steele, 1988) but time-averaged concentrations were generally lower and radiological effects were less than in the aforementioned seas. The major cause of long-term radiation dose through marine pathways, mainly via fish and shellfish consumption and external exposure, was airborne deposition and rainout, of mainly ^{134}Cs and ^{137}Cs on sea and coastal areas. This was even so for the Black Sea, where input of Chernobyl-derived radioactivity from the River Dnieper was relatively small (Kanivets et al., 1999). The fallout from Chernobyl was identifiable in many areas by its characteristic $^{137}Cs/^{134}Cs$ ratio of about 2 : 1 in the early stages. A wide range of data following the Chernobyl accident is published in the literature. Much of the data are on concentrations and inventories rather than radiation exposure. What follows is a brief résumé of reported radiation dose effects, or where these can be derived, from some of the larger studies for these sea areas. It should be pointed out that higher individual doses were potentially received due to consumption of freshwater fish from upland lakes where deposition occurred, and indeed other terrestrial food sources.

The Baltic Sea

Following the accident a wide range of nuclides was detected but these were mainly short-lived and the significant components giving rise to radiation dose were ^{134}Cs and ^{137}Cs. The inventory of ^{137}Cs in the Baltic was estimated at 4300–5000 TBq in 1986, reducing to 2100 TBq in 1990 (HELCOM, 1996). The 'Marina-Balt' study (EC, 2000) assessed critical group and collective doses using model calculations taking account of all the major sources to the Baltic Sea. The maximum critical group doses due to the Chernobyl accident were to consumers of seafood from the Bothnian Sea and Gulf of Finland, at 0.2 mSv y^{-1} in 1986, reducing to 0.02 mSv y^{-1} in 2000. The collective dose commitment to the year 2400 to all countries due to Chernobyl radioactivity deposited in the Baltic was estimated to be 1700 man-Sv.

The Black Sea

A similar overall picture to that of the Baltic was experienced; initially, deposition was uneven but becoming more widely distributed, the significant longer term components giving rise to radiation doses being ^{134}Cs and ^{137}Cs. Most of the literature describes measurements of concentrations/inventories in seawater; the mean concentration of ^{137}Cs in the 0–50 m surface layer is reported as 134 Bq m^{-3} for 1986, the peak year (Egorov et al., 1999). On the

basis of a concentration factor of 100 for radiocaesium in fish (IAEA, 2001), a critical group consumption rate of 50 kg y^{-1} (IAEA, 2001) and ICRP dose coefficients (ICRP, 1996) the dose to the critical group would have been less than 0.02 mSv in 1986.

The Irish Sea

Deposition of fallout from the Chernobyl accident in some coastal areas of the Irish Sea was elevated because of rainfall during passage of the plume. Short-lived radionuclides were observed particularly in shellfish, but these declined such that, as elsewhere, the more lasting effects were due to ^{134}Cs and ^{137}Cs. Doses due to the radioactivity from Chernobyl were assessed after subtraction of other anthropogenic effects (mainly due to Sellafield) and related to steady consumption over one year after the accident (Camplin et al., 1986). High-rate consumers were estimated to have received an effective dose of 0.054 mSv from marine fish and 0.093 mSv from shellfish, mainly molluscs, a total of 0.15 mSv. The collective dose from seafood consumption in 1986 due to input of Chernobyl-derived activity to UK waters was tentatively estimated to be 30 man-Sv.

The North Sea

Radioactivity from Chernobyl was widely detected in the North Sea during summer 1986, largely by means of its distinctive $^{137}Cs/^{134}Cs$ ratio of about 2 : 1. However, concentrations were much lower than in the Baltic Sea, the Black Sea and the Irish Sea. In many areas, including the Northern North Sea, the deposition from Chernobyl overshadowed the effect of BNFL Sellafield, otherwise the main source of radiocaesium (Mitchell & Steele, 1988). However, compared with some other areas, concentrations were low, such that high-rate North Sea fish consumers were estimated to have received only up to 3 μSv y^{-1} in 1986.

Summary of Chernobyl effects on marine waters

The European MARINA studies (CEC, 1990; EC, 2002a) also considered the radiological effects of the Chernobyl accident on marine waters, including collective doses to the EU population. The collective dose to the EU population committed to the year 2500 from marine pathways due to the effects of the Chernobyl accident was estimated to be about 60 man-Sv. This is a small fraction of the total dose commitment due to the Chernobyl accident, estimated to be some 600,000 man-Sv (UNSCEAR, 1993), which is due predominantly to terrestrial pathways. The collective dose rate to the EU population in 2000 from marine pathways is estimated to be about 0.5 man-Sv y^{-1} (EC, 2002a). This would need to be increased by a factor of about 3 to include non-EU populations (CEC, 1990). Thus the world collective dose rate received in 2000 due to marine pathways from the Chernobyl accident would be about 1.5 man-Sv y^{-1}.

3.9. *Conclusions*

A summary of the current radiological effects of the sources of radioactivity in the oceans is presented for comparative purposes in Table 7 and graphically in Fig. 4 (critical group dose) and Fig. 5 (collective dose). All the data have been mentioned earlier in this text and details of their derivation and references may be found there.

Table 7
Summary of current (~Y 2000) annual doses due to major sources of exposure through marine pathways

Source	Dose to critical group (mSv y^{-1})	Collective dose rate to world population (man-Sv y^{-1})
Natural radionuclides	~2	134,000
Enhanced natural radionuclides	0.1–0.3 (to Dutch and Spanish groups)	200
Weapons-test fallout	0.0002	160
Nuclear industry	0.15 (Sellafield) (peak: ~2 in 1975)	20
Ocean dumping of solid radioactive waste	0.00002	4
Radioactive waste dumping in Arctic seas	<0.0001	<0.01
Chernobyl accident	0.02 (Baltic) (peak: ~0.2 in 1986)	1.5

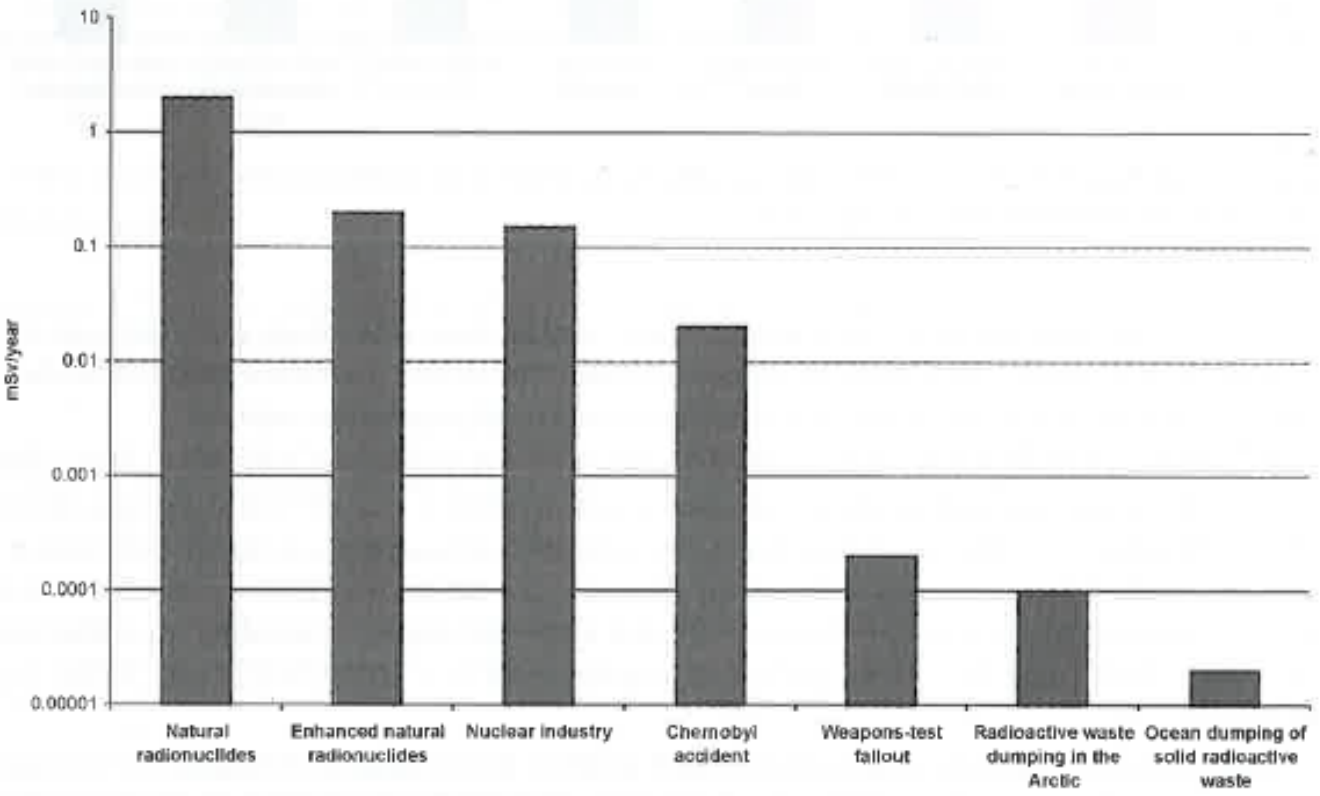

Fig. 4. Comparison of current (~Y 2000) critical group effective dose rates from major sources due to marine pathways (log scale).

The second column of Table 7 and Fig. 4 give the annual effective doses to critical groups currently (i.e. around the year 2000) being received; it is to be noted that the critical groups are not necessarily the same, thus the doses are not necessarily additive. In some cases, for information, the locations of the groups and peak dose rates received in the past are also noted. The third column of Table 7 and Fig. 5 give the current annual collective effective doses to the world population. This is a more effective basis for comparison between sources than integrated commitments into the future which may have also been given in the studies

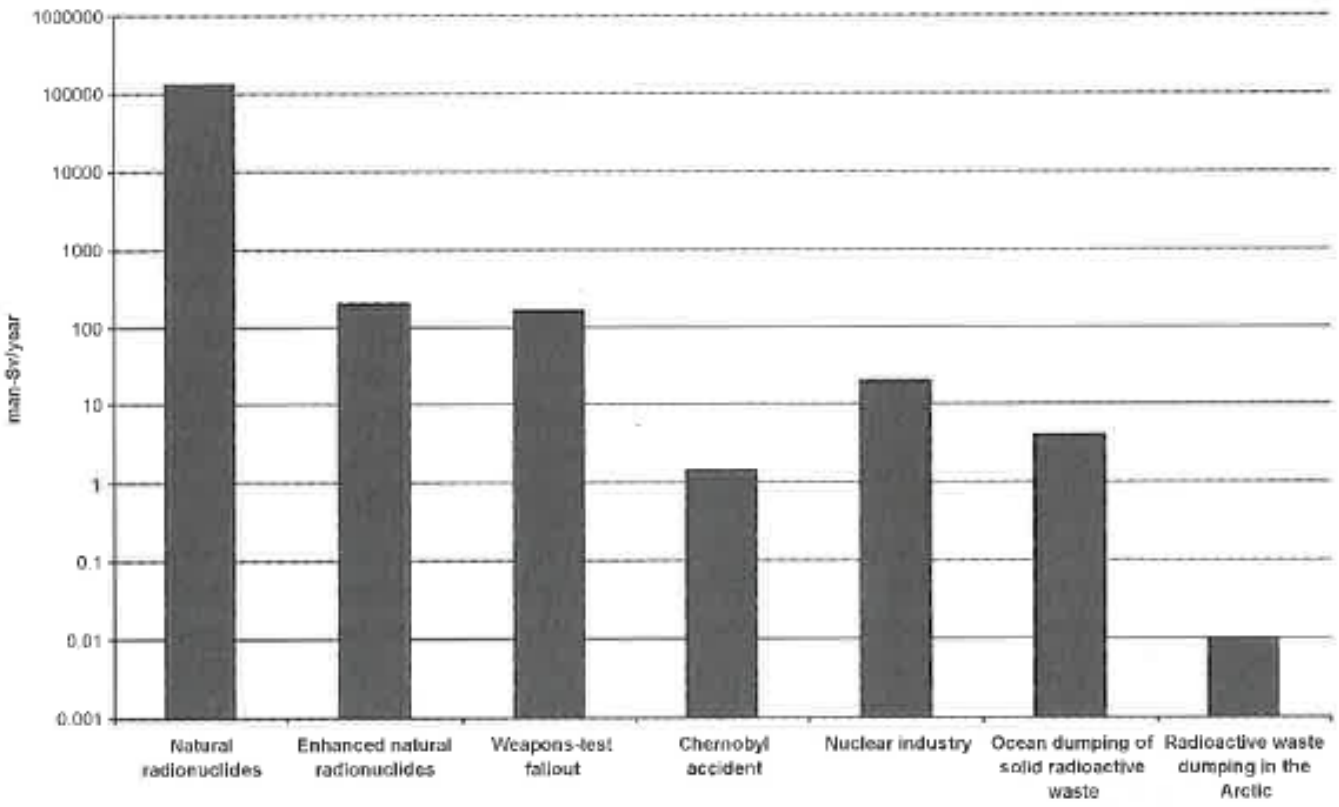

Fig. 5. Comparison of current (~Y 2000) collective effective dose rates to the world population from major sources of exposure through marine pathways (log scale).

referenced, because the timescales used in these studies have often been different, and the results are difficult to extrapolate to a common basis. The reader is referred to the individual studies for more detail, including descriptions of the actual populations exposed.

It is to be noted first that all the exposures are, with one exception, additive to those due to natural background and medical exposures given in Section 1.2, the world average being about 2.8 mSv y^{-1}. The exception is for the collective dose due to average seafood consumption which is considered to be included in the 2.8 mSv y^{-1}. However, the inclusion due to average seafood consumption is small, and indeed derivable by dividing the collective dose of 134,000 man-Sv y^{-1} by the world population of 6×10^9 (UNSCEAR, 2000), i.e. 0.02 mSv y^{-1}.

For individuals the most significant potential addition from marine pathways to the average dose of 2.8 mSv y^{-1} is due to natural radionuclides from high consumption rates of seafood, especially molluscan shellfish; a high-rate consumer could receive an additional ~2 mSv from this source. This dose is overwhelmingly due to ^{210}Po.

Enhanced natural radionuclides due to operations of the phosphate industry are probably the next most significant exposures, with doses to high-rate seafood consumers of 0.1–0.3 mSv y^{-1}. This is small compared with doses due to unenhanced natural radioactivity, potentially to the same individuals. The collective dose is very small compared with that from unenhanced natural radioactivity.

In collective dose rate terms, nuclear weapons-test fallout is the next most significant contribution, due to its wide distribution. However, doses to actual high-rate seafood consumers due to this source are very small.

The highest critical group dose from marine pathways due to operations of the nuclear industry is currently about 0.15 mSv y^{-1}, near Sellafield. This is an order of magnitude lower than from natural radionuclides (potentially to the same critical group). The collective dose consequences of nuclear industry discharges will mostly be in European waters due to the large number of nuclear industry sites and their discharges. Collective doses there from this source, however, are now only of the order of 14 man-Sv y^{-1}, some 4 orders of magnitude lower than from natural radionuclides. An estimate of 20 man-Sv y^{-1} is entered in Table 7 to allow for other world sources.

Ocean dumping has caused concerns about its effects but the potential doses, calculated from modelling studies, are likely to be very small, 4–5 orders of magnitude lower than for natural radionuclides in critical group and collective dose terms.

The radiological implications of radioactive waste dumping in Arctic seas have also been studied by the use of suitable models. Potential critical group doses are of the same order as for the North Atlantic dumping, with collective dose being much less, inter alia because of lower populations and seafood production.

Lastly, the remains of the Chernobyl accident give rise to a critical group dose rate of about 0.02 mSv y^{-1} in the northern Baltic area. Collective dose rates due to marine pathways from Chernobyl deposition are currently of the order of 1.5 man-Sv y^{-1}.

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