ANNEX B

Exposures from nuclear power production

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Introduction

1. The generation of electric energy by nuclear reactors has increased since the Committee’s assessment of doses from radioactive materials released during nuclear fuel cycle operations, as reported in Annex F of the UNSCEAR 1982 Report [U1]. The total world installed nuclear electricity generating capacity at the end of 1987 was 298 GW from 417 units in 26 countries [11]. This represents an approximate doubling of nuclear capacity since the UNSCEAR 1982 Report, as may be seen from Figure I. Nuclear power was responsible for some 16% of the world’s electricity generated in 1987, and currently some 120 reactors are under construction with an electrical capacity of 101 GW [11]. Projections for world nuclear generating capacity for the year 2000 are still somewhat speculative, but the figure seems likely to be in the range of 400-500 GW [12], somewhat less than earlier expectations but still representing a further expansion of 30-60% from currently installed capacity.

2. The number of power reactors operating at the end of 1987, their type and generating capacities for each country of the world is shown in Table 1. The reactor types include the pressurized water moderated and cooled reactor (PWR), the boiling water moderated and cooled reactor (BWR), the gas cooled reactors (GCR) of the Magnox and advanced gas cooled (AGR), graphite moderated type, the light water cooled graphite moderated reactor (LWGR), the heavy water moderated and cooled reactor (HWR), and the fast breeder reactor (FBR). The installed capacity per capita is also given in Table 1; it is highest in Sweden at 1.14 kW per capita and ranges from about 0.1 to over 0.8 kW per capita in other developed countries. The average installed capacity per person at about 0.14 kW represents an increase of 100% over the equivalent figure (0.07 kW) reported in the UNSCEAR 1982 Report. Table 2 shows the amounts and percentage of electricity generated in countries by nuclear power in 1987 [12]. The highest use of nuclear reactors for electricity generation was in France (70%) and Belgium (66%).

3. The nuclear fuel cycle includes the mining and milling of uranium ores, conversion to nuclear fuel material, which usually includes the enrichment of the isotopic content of $^{235}$U and fabrication of fuel elements; the production of energy in the nuclear reactor; the storage of irradiated fuel, or its reprocessing with the recycling of the fissile and fertile materials recovered, and the storage and disposal of radioactive wastes. Almost all of the artificial radionuclides associated with the nuclear fuel cycle are present in the irradiated nuclear fuel, although some neutron activation of structural and cladding materials takes place. The majority of irradiated fuel elements are currently stored; when reprocessing takes place, the highly active liquid wastes containing fission products and transuranium elements are stored in tanks isolated from the environment until they can be solidified. Solid wastes, arising at each stage of the fuel cycle, are mainly stored, although some wastes are disposed of.

![Figure I: The installed nuclear electric energy capacity on 31 December between 1979 and 1987.](image)

[11, 12, 13, 14, 15, 16, U1]
In routine operation of nuclear installations, small quantities of radioactive materials are released in effluents, which disperse in the environment and result in low-level exposures of the public.

4. The interest of the Committee is in assessing the radiation doses to individual members of the public from releases of radioactive materials and also the doses to workers from normal operation of the nuclear fuel cycle. Exposures of the public from high-level wastes, which arise in fuel reprocessing, have not been assessed by the Committee, as these wastes are still in storage. The majority of irradiated fuel is not being reprocessed. Preliminary estimates are made of the exposures in the future resulting from current disposals of radioactive solid wastes. The significant release of radioactive materials and the exposures to workers and the public that resulted from the accident at the Chernobyl nuclear power reactor are discussed in detail in Annex D “Exposures from the Chernobyl accident”, and Annex G “Early effects in man of high doses of radiation”.

5. The quantities of radionuclides in effluents from nuclear facilities are usually reported and available to the Committee, reflecting the operational history of each plant, including periods of abnormal operation and maintenance shut-down. In this Annex the Committee reviews discharge data for the six-year period 1980-1985 and estimates average releases per unit of electric energy generated for each major power reactor type. Because the data for 1985 are incomplete, normalized releases are presented for the quinquennium 1980-1984. These normalized releases do not apply, of course, to any one plant but are deemed to be representative of current nuclear power generation. Future practices may lead to discharge levels considerably different from the normalized values presented here, which include new and old plants; therefore, any extrapolation to the future must be undertaken with caution.

6. Because of the system of controls applied to environmental releases from nuclear power installations, doses to individual members of the public correspond to low levels of individual risk. The doses to the most exposed individuals vary widely from installation to installation and from one location to another, and the level of individual dose generally decreases rapidly with distance from a given source. In this Annex an indication is given of the range of individual doses associated with each type of installation. To evaluate the total impact of radionuclides released at each stage of the fuel cycle, results are presented in terms of the collective effective dose equivalent commitment per unit quantity of electric energy produced, expressed as man Sv per GW a.

7. The collective dose commitment from nuclear power production is considered in four population groups: the occupationally exposed; the local population, being those within about 100 kilometres of the site; the regional population, those within about 1,000 kilometres of the site; and the remaining world population. Each stage of the nuclear fuel cycle is treated separately, and the occupational, local and regional dose commitments are evaluated. The contributions from nuclides that, because of a combination of long radioactive half-lives and rapid dispersal in the environment, become globally dispersed and irradiate the world population are then discussed for the fuel cycle as a whole.

8. Collective dose commitments to local and regional populations must be estimated by environmental modelling, as the activity concentrations resulting from effluents from nuclear fuel cycle operations are very low both in environmental samples and in the general population. Monitoring of activity concentrations due to effluent releases has concentrated on areas immediately surrounding nuclear facilities to ensure compliance with relevant regulations. To estimate collective dose commitments it was decided in the UNSCEAR 1982 Report to establish a model facility at a representative site for each stage of the fuel cycle: mining and milling, fuel fabrication, reactor operation and reprocessing. The environment receiving the normalized releases from each model facility was chosen to represent broad averages containing typical features of existing sites and reflecting the most common environmental pathways. Such generalizations gave dose commitments indicative of the impact of the overall nuclear power programme though not applicable to any one site. In the UNSCEAR 1982 Report, the collective doses were evaluated for reported discharges at the three operating commercial reprocessing plants at Sellafield in the United Kingdom and Cap de la Hague and Marcoule in France.

9. The methods used by the Committee for estimating the dispersion of radionuclides released to the atmosphere or hydrosphere and the resulting doses to individuals were described in Annex A of the UNSCEAR 1982 Report. The Committee considers that, in general, these methods and the model facilities and representative sites used in the UNSCEAR 1982 Report are still valid for assessing the current impact of discharges from the fuel cycle. Therefore, in this Annex, the collective effective dose equivalent commitments are obtained by scaling the dosimetric results from the UNSCEAR 1982 Report, allowing for different releases of the various radionuclides involved. The Committee has decided to treat the reprocessing contribution differently in this Report. The hypothetical model facility is not used, but rather, in order to reflect the actual dose contributions made, the normalized dose commitments from the fraction of fuel reprocessed is added to the contributions from the rest of the fuel cycle.

10. Very long-lived nuclides pose a special problem. One example is $^{129}I$ (half-life: 1.6 $10^9$ a), while another is radon gas, which emanates from mill tailings containing $^{232}$Th (half-life: 8 $10^4$ a) and $^{238}$U (half-life: 4.5 $10^9$ a). Assessments of human exposures over such periods of time are clearly hypothetical and the relevance of the results is doubtful. Dose commitments assessed for the purpose of calculating maximum dose rates in the future involve integration over the period of practice leading to the release of the radioactive material. This approach is taken in this Annex for effluents. For the solid waste disposal assessment, it is
in general only possible to assess the collective effective dose equivalent commitment.

11. There have been a number of attempts to generate rigorous definitions of the waste categories generally referred to as low-, intermediate- and high-level wastes [113]. Although precise definitions have been agreed for particular purposes, the schemes proposed have not been universally satisfactory. None the less, the general characteristics of the three waste types are reasonably well established.

12. High-level wastes (HLW) are primarily the spent fuel elements or the solidified waste products from reprocessing. They have high activity concentrations of both actinides and fission products and are significantly heat-generating. As fuel elements are a significant potential source of fissile material, they will usually be stored in the short-to-medium term rather than disposed of. Occasionally, other waste streams with high activity concentrations are also regarded as HLW, but the quantities of activity in them are relatively small.

13. Intermediate-level wastes (ILW) are defined to some extent by exclusion from the other two categories: they contain either actinides or long-lived beta/gamma emitters in quantities that are not negligible or substantial activity concentrations of short-lived beta/gamma emitters and are not significantly heat-generating.

14. Low-level wastes (LLW) contain primarily reasonably short-lived beta/gamma emitters in low-to-moderate activity concentrations. They may contain actinides or long-lived beta/gamma emitters but only in very small quantities.

15. There will be other categories of materials that are uncontaminated, even though they were generated at a nuclear site or are of such a low level of activity concentration that they can be exempted from the requirements for storage and disposal as radioactive waste. The rationale for such exemption is that the radiological impact of uncontrolled disposal of these materials is insignificant [114, N7]. These wastes are not considered part of this study, as their potential for radiological impact is by definition very low in comparison with that from the other waste categories.

16. In this preliminary assessment of doses from disposed wastes, only LLW and some categories of ILW are considered to be disposed of by shallow land burial. All other wastes are stored under conditions such that the doses to members of the public are essentially zero, and doses to occupational workers are included in those assessed for other operations at the same sites.

17. The Committee presented detailed comprehensive reviews of occupational exposures, including those from the nuclear fuel cycle, in both the UNSCEAR 1977 Report [U2] and the UNSCEAR 1982 [U1] Report. In this Annex the data on occupational exposures throughout the nuclear fuel cycle are brought up to date.

18. With regard to assessing occupational exposures, the relationship between measurements of external irradiation made in radiation fields by film, thermoluminescent or other personal dosimeters and the absorbed doses in the tissues and organs of the body was discussed in the UNSCEAR 1982 Report. The Committee adopted the convention that all numerical results reported by monitoring services represent the average absorbed dose in the whole body, recognizing that these are almost always readings from the dosimeters that are reported, without consideration of the relationships to the absorbed doses in organs and tissues of the body. In this Annex the Committee adopts a similar convention; but to simplify comparisons, and because most exposures are to penetrating gamma-radiation, the numerical result is taken to represent the effective dose equivalent. Exposures of uranium miners to radon and its daughters are also expressed in terms of effective dose equivalent.

19. The characteristics of occupational dose distributions identified by the Committee as of interest were: (a) the annual average effective dose equivalent H_eff., which is related to the average level of individual risk, this average has generally been calculated for all individuals monitored in a given occupational group; (b) the annual collective effective dose equivalent, S_eff., which is related to the impact of the practice; (c) the collective effective dose equivalent distribution ratio, defined as the ratio of the annual collective effective dose equivalent delivered at annual effective dose equivalents exceeding 15 mSv to the total collective effective dose equivalent. This is related to the proportion of workers exposed to higher levels of individual risk. These characteristics may be obtained for any form of the dose distribution, whether or not it exhibits a log-normal or other defined response over any part of the effective dose equivalent range. The collective effective dose equivalent is usually calculated from collated dosimetry results using the definition

\[ S_{\text{eff}} = \sum N_i H_{\text{eff},i} \]

where \( N_i \) is the number of individuals in the effective dose equivalent range \( i \) for which \( H_{\text{eff},i} \) is the mean annual effective dose equivalent. The annual average effective dose equivalent, \( H_{\text{eff}} \), is given by

\[ H_{\text{eff}} = \frac{S_{\text{eff}}}{N} \]

where \( N \) is the total number of workers monitored.

20. The normalized measure of the impact of the various components of the nuclear fuel cycle is the collective effective dose equivalent per unit electric energy generated. This is calculated as an average over a complete power programme or over several years to avoid anomalies such as those connected with the shut-down of reactors for maintenance. The results for doses from occupational exposures and to the local, regional, and global populations exposed as a result of effluent discharges to the environment may be taken to be a relative measure of the health impact of nuclear power production.
I. MINING AND MILLING

21. Uranium mining operations involve the removal from the ground of large quantities of ore containing uranium and its daughter products at concentrations between a tenth and a few per cent U$_{238}$. These concentrations are several thousand times the concentration of these nuclides in the rest of the natural terrestrial environment. Uranium is mainly mined using underground or open-pit techniques, other methods such as heap leaching accounting for only a few per cent of the world production. The quantities produced during the period 1980-1984 are given in Table 3. Milling operations involve the processing of these large quantities of ore to extract the uranium in a partially refined form, often known as yellow-cake. This is further refined, converted and enriched, if necessary, before fabrication into fuel elements. Uranium mills tend to be located near mines to minimize transportation. The number of mills operating is related to uranium demand.

A. EFFLUENTS

22. The predominant gaseous effluent from active uranium mines is $^{222}$Rn in the ventilation air from underground mines or released into the pit from surface mines. In a study covering 27 mines [J2] this accounted for 97% of the radon released. A recent study [N5] has also shown that for some surface mines, especially where a large volume of overburden has to be removed to expose the ore, waste rock piles formed a source of radon of a magnitude comparable to that of the pit. Release rates per unit mass of ore were estimated in the UNSCEAR 1982 Report at about 1 Gbq t$^{-1}$ from underground mines and about 0.1 Gbq t$^{-1}$ from surface mines. In general, however, the ore from underground mines was estimated to have about 10 times the uranium concentration of that from surface mines; the normalized radon emission was thus taken for both types to be 1 Gbq t$^{-1}$ of ore for 1% uranium oxide in the ore. Particulates in airborne dust contain $^{238}$U and its daughters and sometimes $^{222}$Th and its daughters.

23. The results of measurements or estimates of either total radon emission rates or normalized radon emission from a number of mines are given in Table 4. The data for underground mines relate to the ventilation air from the shaft, those for surface mines, to the mine pit. The results support retention of an overall normalized radon emission of 1 Gbq t$^{-1}$ of ore for 1% uranium oxide in the ore.

24. The uranium requirements per unit electric energy generated vary somewhat between current designs of thermal reactors; but the heavy metal requirements are generally in the range of 150-250 t (GW a)$^{-1}$. The grade of ore mined at present is usually between 0.1 and 1% U$_{238}$. Taking a typical value for underground mines from the United States of 0.2% [E1], the normalized radon releases are about 20 Tg (GW a)$^{-1}$. This is the same value that was estimated in the UNSCEAR 1982 Report.

25. The processing of uranium at the mill was described in the UNSCEAR 1982 Report, as were the broad characteristics of the tailings piles, where most of the activity not extracted as usable uranium resides. This activity is predominantly $^{236}$Th and its daughters. There are airborne emissions during operation of a mill, mainly of $^{222}$Rn together with $^{234}$U, $^{228}$Th, $^{226}$Ra and $^{210}$Pb. The ranges of airborne release rates for a typical mill estimated in the UNSCEAR 1982 Report are shown in Table 5.

26. During operation of a mine, there are stockpiles of ore and piles of sub-ore, overburden and waste rock. After closure there will typically be a pile of overburden, possibly covered by sub-ore, in case processing of this becomes economically viable in the future. These also act as sources of airborne emissions, principally of $^{222}$Rn. An estimate of the radon emanation rate from waste rock per 1% ore grade in the United States is 100 Bq m$^{-2}$ s$^{-1}$ [N4]. The number of inactive mines in the United States was estimated to be about 1,250 surface and 2,000 underground in 1980 [H9]. Some useful measurements have been made of radon emanation rates under dry conditions over a wide range of ore grades in the Northern Territory of Australia [L2, M5]. These suggest that a radon exhalation rate of 50 Bq m$^{-2}$ s$^{-1}$ per 1% ore grade is widely applicable; this figure is equivalent to 0.5 Bq m$^{-2}$ s$^{-1}$ per Bq g$^{-1}$.

27. Extraction of uranium during milling is clearly made as complete as possible but cannot reach 100%. Typically, the residual tailings from the mill will contain from 0.001 to 0.01% U$_{238}$, depending on the grade of ore and the extraction process. Tailings are discharged from mills into impoundments, the characteristics of which depend on the local climate and geology [T1]. From the point of view of estimating diffusional effluents, the major differences are whether the tailings pile is wet or dry and whether it has been covered. All tailings piles act as sources of airborne releases, although if they are completely covered by water the rates can be extremely low. Estimates of radon emanation rates for a number of typical tailings areas and impoundments are shown in Table 6. Most of these are taken from an extensive study by the Nuclear Energy Agency (NEA) [N5]. The radon exhalation rate per unit area and specific activity of $^{226}$Ra was estimated in the UNSCEAR 1982 Report at about 1 Bq m$^{-2}$ s$^{-1}$ per Bq g$^{-1}$ of $^{226}$Ra in the tailings, although it was noted that the rate could vary from effectively zero to an order of magnitude higher than the above figure. It has been suggested that a more realistic figure would be 0.2-0.5 Bq m$^{-2}$ s$^{-1}$ per Bq g$^{-1}$ [S12]. For comparison, 0.01% U$_{238}$ ore contains approximately 1 Bq g$^{-1}$ of $^{226}$Ra. Detailed measurements have been carried out on seven tailings dams in South Africa [A7], giving a mean radon exhalation rate of 0.4 Bq m$^{-2}$ s$^{-1}$ per Bq g$^{-1}$ for a radium concentration ranging from 0.2 to 0.7 Bq g$^{-1}$. Measurements on tailings in the Elliot Lake area of Canada [B26] showed a range from 0.2 to 7.6 Bq m$^{-2}$ s$^{-1}$ per Bq g$^{-1}$. Experimental investigations on two types of bare dry tailings in Australia [S13] showed exhalation rates from 0.3 to 0.7 Bq m$^{-2}$ s$^{-1}$ per Bq g$^{-1}$; these were reduced by a factor of 3 for 1 m dry cover and by more than a factor of 10 for 1 m moist cover.
28. In considering the longer-term impact of effluents from tailings piles, it must be assumed that activity concentrations from uranium nuclides remain practically constant indefinitely, due to their long half-lives. The rest of the activity in the tailings is dominated by $^{226}$Th, which has a half-life of 80,000 a. The radionuclides in the decay chain from $^{238}$Th with the greatest radiological significance are $^{228}$Ra, which can be leached out by water access, $^{210}$Pb and $^{222}$Rn, which can escape into the air.

29. At present, tailings have tended to be kept in open, uncontained piles or behind engineered dams or dikes with solid or water cover. It is likely, however, that some further engineering will be carried out to minimize the release of radionuclides from the abandoned piles. Such techniques were analysed in the NEA study [N5] for a number of sites. The radon flux density varied by factors of more than 10$^4$, dependent on the treatment assumed, showing that this is clearly a crucial parameter in the assessment of the impact of tailings piles. The options assumed for one typical site in an arid region and the relative radon flux densities assumed to result are shown in Table 7. Similar reductions in radon emission have been found using covers of various types [H10]. Assuming some reasonably impermeable cover is used, the radon exhalation rate from a typical tailings pile is taken to be $10^{4}$ Bq m$^{-2}$ a$^{-1}$. This is less than the figure assumed for emanation from the unsealed material stockpiled around working mines and comparable with the value expected to be achieved in the United States [E4]. The cover is assumed to provide some protection against erosion, so that the radon exhalation rate remains essentially constant with time. Otherwise, an increase of up to double the initial rate of emanation from a bare pile could have been expected over a period of about 10$^4$ years [N5]. As can be seen from the results of the UNSCEAR 1977 and UNSCEAR 1982 Reports, these are critical assumptions in determining the overall impact of the fuel cycle.

30. Mine and mill sites in dry areas give rise to effectively no liquid effluents. For those in wet climates, however, run-off water will contain radionuclides and may need treatment before release into watercourses. The most important radionuclide in liquid effluents is $^{228}$Ra, and typical releases at wet sites were estimated in the UNSCEAR 1982 Report to be 1 GBq (GW a)$^{-1}$. A review by Kaufmann [K5] suggests values of the order of 0.1 GBq (GW a)$^{-1}$, given normal procedures for water treatment.

B. LOCAL AND REGIONAL COLLECTIVE DOSE COMMITMENTS

31. In the dose estimation procedure used in the UNSCEAR 1982 Report, the typical characteristics of a mine and mill site in terms of population density, rainfall, farming, etc. were first established. The population densities used were 3 km$^{-2}$ for 0-100 km and 25 km$^{-2}$ for 100-2,000 km. A deposition velocity of $10^{-2}$ m s$^{-1}$ was taken for particulate releases. The collective dose for radon release was then calculated using an atmospheric dispersion model with characteristics typical of a semi-arid area and an effective release height of 10 m. The atmospheric dispersion model was described in the UNSCEAR 1982 Report and in the original reference [C1]. The resultant collective effective dose equivalent commitments per unit activity released are shown in Table 8, with the exception of the figure for radon. This has been reduced for the reasons discussed in Annex A which have led to a reduction in the dosimetric coefficient for outdoor air from 17 to 9 nSv h$^{-1}$ per Bq m$^{-3}$. These figures have been used in this Annex to estimate the normalized collective effective dose equivalent commitments from current atmospheric releases which is about 0.3 mSv (GW a)$^{-1}$. The doses from liquid effluents are negligible by comparison.

32. Using the figure estimated for the initial rate of exhalation of radon from a typical tailing pile leads to an annual release of about 1 TBq ha$^{-1}$. The production of a mine generates about 1 ha (GW a)$^{-1}$ of tailings, so the releases during a period of five years, corresponding to the duration taken for the current discharge, would add a normalized collective effective dose equivalent commitment of 0.1 mSv (GW a)$^{-1}$. The rate of release as a function of time is assumed to be constant, and given the very long duration of the source, the normalized collective effective dose equivalent commitment is proportional to the duration considered reasonable for assuming the release. Taking this period to be 10$^4$ years for the sake of illustration, the result is an estimated 150 mSv (GW a)$^{-1}$. An alternative perspective on this component can be obtained by assessing the truncated collective effective dose equivalent commitments up to different times. Some examples of the results of such calculations for the various coverings described in Table 7 are shown in Table 9, taken from the same study [N5].

C. OCCUPATIONAL EXPOSURES

33. The main source of radiation exposure of underground uranium miners is radon and its daughters. The annual average exposure of underground miners was taken to be 1.5 WLM in the UNSCEAR 1982 Report; this was converted to an annual effective dose equivalent of about 13 mSv. Surface miners have a lower exposure to radon and daughters, with annual doses estimated to be about 3-4 mSv, but they and underground miners are exposed through inhalation of dust containing uranium and its daughters. Both underground and surface miners are also exposed to some external gamma radiation. The estimate of annual doses for underground miners was rather broad in the UNSCEAR 1982 Report, 1-10 mSv; that for surface miners was taken to be 1-2 mSv. Where the authors have not carried out their own conversions, use has been made of the conversion coefficients given by the International Commission on Radiological Protection [I12] between committed effective dose equivalent and time integrated equilibrium equivalent radon daughter concentration in air of 17 nSv h$^{-1}$ per Bq m$^{-3}$ or 10 mSv WLM$^{-1}$, where 1 WLM is one working month (170 ha) of exposure to a potential alpha-energy concentration of 2.08 $10^{5}$ J m$^{-3}$ [I10].
34. Exposure of uranium miners to radon and daughters has been monitored by a combination of measurement of levels in air at a variety of places through the mine and estimates of the time spent by miners in those places. In recent years, however, there has been considerable development work on dosimeters suitable for monitoring of radon daughter exposures for individual underground uranium miners. Some recent results for underground uranium miners are shown in Table 10. The United States data for 1980, assumed to be primarily for underground miners, are from a very general summary prepared by the Environmental Protection Agency [E3]; those for 1981 and 1982 relate only to the mines in New Mexico [S8]. The data for Canada [A4] include exposures at the uranium mills associated with the mines. Data can be clearly separated into underground and a surface mine for the Canadian mines, and the results for the surface mine at Key Lake [A4] are shown in Table 11. A comparison between mine company records and exposures based on measurements by inspectors for 1979 and 1980 in the United States showed reasonable agreement [C7]. In this study the annual average effective dose equivalent to underground workers in 61 mines from exposure to radon and daughters was estimated to be in the range of 18-29 mSv, depending on the assumptions made in deducing the personnel exposures from the measurements in working areas. This is somewhat higher than the estimates given in Table 10.

35. Information on gamma exposures to workers in both underground and open pit mines in Canada [A4] shows annual average effective dose equivalents ranging from 0.1 to 3.4 mSv for the years 1981-1983. Some underground mines showed average gamma doses as low as for surface mines, but the major underground mines employing more than 80% of the work-force had an annual average effective dose equivalent of 3 mSv. An estimate of 3 mSv as the annual average effective dose equivalent from inhalation of dust has also been made for the Ranger surface mine in Australia [A8].

36. Taking all the above information into account, the average annual effective dose equivalent to underground uranium miners from both external exposure and radon daughter exposure is 10-12 mSv; that for surface miners is lower, possibly around 5 mSv. Given the predominance of underground miners, an overall annual average of 10 mSv seems a reasonable estimate for the early 1980s. Taking the productivity to be $3 \times 10^{-1}$ of natural uranium per miner and a natural uranium requirement of about 200 t (GW a$^{-1}$), the normalized collective effective dose equivalent would be 0.7 man Sv (GW a$^{-1}$). This is comparable to the estimate in the UNSCEAR 1982 Report of 0.9 man Sv (GW a$^{-1}$), which was rounded up to 1 man Sv (GW a$^{-1}$).

37. Recent data on doses received by 3,000 workers at uranium mills in the United States show an annual average effective dose equivalent of 2.7 mSv [E3]. The external average effective dose equivalent to 131 workers at the Nabarlek mill in Australia during the period 1981-1982 was 1.5 mSv [M9] and at the Ranger mill during the period 1985-1986 as low as 0.9 mSv [A8].

The contribution from workers at mills to the collective effective dose equivalent per unit electric energy generated is so small that in the UNSCEAR 1982 Report it was not included as a separate item. This situation does not appear to have changed.

II. URANIUM FUEL FABRICATION

38. The uranium ore concentrate produced at the mills is further processed and purified and converted to uranium tetrafluoride (UF$_4$), and then to uranium hexafluoride (UF$_6$), if it is to be enriched in the isotope $^{235}$U, before being converted into uranium oxide or metal and fabricated into fuel elements. Natural uranium, containing 0.71% $^{235}$U, is used in graphite or heavy water moderated reactors. Enrichments of 2-5% are required for light water reactors (LWRs) and advanced gas cooled reactors (AGR).

39. To produce natural uranium metal fuel, the uranium tetrafluoride is compressed with shredded magnesium and heated, and the resulting reduced uranium is cast into rods that are machined and inserted into cans. Natural uranium oxide is sintered into pellets and clad in zirconium alloy for HWR fuel pins. For LWR and AGR fuel, the UF$_6$ is converted into the gaseous form UF$_6$. The first type of enrichment plant to be developed commercially employed the gaseous diffusion process. In this, the UF$_6$ diffuses through a porous membrane, the lighter compound containing UF$_6$ and UF$_4$ diffusing more rapidly than the heavier compound containing UF$_6$. Partial separation occurs, but in practice many stages of such membranes are required in series to provide a cascade.

40. The pumping power required to move the UF$_6$ through the cascade requires a large amount of electric energy. The alternative gas centrifuge process consumes only about 5% of the electric energy demanded by the diffusion process. The gas centrifuge process is based on the separation effect on a mixture of UF$_6$, isotopes in a strong centrifugal field in a rotating cylinder, suitably combined with the cascading effect of counter-current circulation. More separation is attained in one centrifuge stage than one diffusion stage but, as the mass flow is less, a series-parallel configuration is required.

41. To fabricate LWR fuel the enriched UF$_6$ is converted to the oxide (UO$_2$) powder, which is granulated, sintered and pressed into pellets. These are inserted into tubes (cladding) that are sealed after being filled with pellets. For LWR fuel cans zirconium alloy is used, while for AGRS stainless steel cans are adopted. After the enrichment process, large quantities of depleted uranium remain, containing about 0.3% or more $^{235}$U. This uranium would become a source of public exposure were it to be disposed of, but currently it is stored for possible use in breeder reactors and for other purposes. The solid wastes arising during operation of the uranium fuel fabrication facilities will contain the same radionuclides as those at uranium mines and mills, but will be trivial in quantity by comparison. It does not, therefore, seem worth while to assess their impact separately.
42. Emissions of radionuclides from the conversion, enrichment and fuel fabrication processes are generally small. Most of the uranium compounds are solid and are easily removed from airborne effluent streams, while settling tanks are used to reduce liquid effluent discharges. Few data published in the United States or Europe give discharge rates of radionuclides from these fuel cycle facilities. The Committee concluded in the UNSCEAR 1982 Report that discharges were small and estimated releases from model facilities producing LWR fuel. In the United Kingdom, reported discharges are given in terms of total alpha, total beta activity and masses of uranium (Table 12), and some isotopic breakdown can be obtained for centrifuge enrichment plant effluents. Most of the beta-discharges are from the short-lived $^{244}$Pa (half-life: 1.17 min) which is separated with $^{210}$Th (half-life: 24.1 d). Canadian data are also available for effluents from a conversion plant [A4, L1] with their isotopic composition [M2]. There are small releases of $^{99}$Tc reported from the British enrichment plant, indicating some recycling of reprocessed uranium, but these releases are atypical and no dose assessment has been made.

43. The data presented in Table 12 have been used to obtain the effluent releases which are applied to the same model facility sited on a river as was used in the UNSCEAR 1982 Report. The normalized releases are based on an LWR cycle uranium requirement of 150 t (GW a)$^{-1}$ and an HWR cycle requirement of 170 t (GW a)$^{-1}$. The results are given in Table 13 for atmospheric and aquatic effluents. The conversion plant figures are based on data from Canada [M2], as are those for fabrication, since these relate to freshwater discharges in contrast to the British figures, which relate to marine discharges. The values quoted in Table 13 are typical figures taken from those calculated for the five Canadian fabrication plants, based on a fuel cycle requirement of 170 t (GW a)$^{-1}$. The discharges of $^{210}$Th are obtained by assuming that this radionuclide is in equilibrium with $^{234}$U.

44. The results in Table 13, which were derived from reported discharges, can be compared with the effluents from the model facilities quoted in the UNSCEAR 1982 Report, which were based mainly on the notional results produced by the Environmental Protection Agency [E2]. The results using present data suggest that for conversion, atmospheric releases are generally about twice those quoted previously for uranium and thorium isotopes, while aquatic releases as reported are about 10% of those assumed previously, and in the case of $^{228}$Ra are only 1% of that in the Environmental Protection Agency model facility assumed previously. Atmospheric releases from enrichment are about one half of those quoted in the UNSCEAR 1982 Report: liquid effluents are only a few per cent of the Committee's previous estimates. For fuel fabrication, based on a weighted average of natural and enriched fuels, the atmospheric and aquatic releases are again about one half the previously assumed values.

45. The Committee concluded in the UNSCEAR 1982 Report that releases to the atmosphere provided the major exposure to the population (over 90%) from fuel conversion, enrichment and fabrication processes. To obtain an order of magnitude assessment of the collective dose commitments, the Committee specified a model facility with a constant population density of 25 km$^{-2}$ out to 2,000 km. This was chosen to be representative of North America and Europe, and collective dose commitments were derived for inhalation from the plume, ingestion of foodstuffs contaminated by activity deposited from the plume and by external irradiation from the activity deposited on the ground. The same results have been used here, but the collective effective dose equivalent commitments have been scaled for the normalized releases derived in Table 13; the resultant doses are given in Table 14. The most significant pathway of exposure continues to be inhalation of particulate activity, with radon daughters contributing about 15% of the dose.

46. In summary, the normalized collective effective dose commitment due to uranium fuel fabrication is estimated to be $2.8 \times 10^{-9}$ man Sv (GW a)$^{-1}$. The main contribution arises from inhalation of the isotopes of uranium. The figure is similar to that derived in the UNSCEAR 1982 Report [$2.0 \times 10^{-9}$ man Sv (GW a)$^{-1}$]. Individual doses in the vicinity of fuel fabrication facilities are estimated to be less than 50 mSv per year for members of the public [B1, B2, B3, B8, B16, B29].

C. OCCUPATIONAL EXPOSURES

47. The annual average effective dose equivalents to workers in fuel fabrication plants were found in the UNSCEAR 1982 Report to be generally low, ranging from 0.3 to 3 mSv. The annual collective effective dose equivalent distribution ratio (see paragraph 19) was also in general small, often approaching zero. Data on the number of workers employed and the corresponding annual average individual and collective effective dose equivalents are given for some countries in Table 15. These are not always complete for a country for any particular year and could include workers not strictly employed in fuel fabrication. For example, the data from the United States [N2] are quoted as corresponding to fabrication and reprocessing, but it has been assumed that the contribution from reprocessing in the years 1980 and 1981 was negligible; the data for the United Kingdom [H8, B12] include exposures during enrichment. Annual average doses to fuel fabrication workers have remained low, in the range of 1-2 mSv. The collective effective dose equivalent distribution ratio for United States workers, which was 0.12 in 1980, decreased to 0.09 in 1981 [N2]; that for British workers was 0 in 1982 and 0.02 in 1983 [B12]; that for Japanese workers was 0 in the period 1981-1984 [T12].

48. Some data on the external doses from the fabrication of plutonium fuel at the PNC works in
Japan have been published [5]. These are shown in Table 16. During the period 1977-1982 the total amount of fuel fabricated was 37.6 t for an advanced HWR and 1.2 t for an FBR. From 1980 to 1982, it was necessary to process reactor grade plutonium recovered from high burn-up fuel, and this led to an increase in both average and collective doses to the work-force.

49. The estimates of normalized collective effective dose equivalent in the UNSCEAR 1982 Report were considerably reduced from previous estimates; the overall figure estimated to be 1 man Sv (GWa)\(^{-1}\). More recent estimates are shown in Table 15. The normalized collective effective dose equivalents for Canada and the United Kingdom were obtained by directly relating the collective dose in a year to the electric energy generated in the year [13, 14, 15, 16], as seems appropriate for nuclear power programmes in an approximately equilibrium situation. For the United States the same assumption is made as in the UNSCEAR 1982 Report; 60% of the fuel fabricated is for United States nuclear power stations. For Japan, figures for 1981-1984 are used [33], and these have been tentatively related to the total electric energy generated in the corresponding years by nuclear power. Giving appropriate weight to more recent data, an overall average of 0.5 man Sv (GWa)\(^{-1}\) now seems more appropriate.

III. REACTOR OPERATION

50. Nearly all the electric energy generated by nuclear power is produced in thermal reactors in which the fast neutrons produced by the fission process are slowed down to thermal energies by use of a moderator. The most common materials still used for moderators are light water, heavy water and graphite. The choice of moderator and coolant, light or heavy water or carbon dioxide gas, greatly affects the design, size and heat removal system of the reactor.

51. The uranium fuel is contained in discrete pins, both to prevent leakage of the radioactive fission products into the coolant circuit and to improve neutron economy by reducing parasitic neutron captures in the resonance neutron energy region of \(^{238}\)U. The heat generated in the fuel pins by the slowing down of the fission fragments is removed by forced convection, the most usual coolants being light or heavy water or carbon dioxide gas. In the case of fast reactors, the neutrons are not moderated and induce fissions with energies close to those at which they are produced. The usual heat removal system is liquid sodium, which is a good heat transfer medium and does not greatly moderate the neutrons.

A. EFFLUENTS

52. During the production of power by a nuclear reactor, radioactive fission products are formed within the fuel, and neutron activation produces radioactive components in structural and cladding materials. Radio nuclides are formed in the coolant circuit because the coolant becomes activated, because of the diffusion of fission product elements with radioactive isotopes from the small fraction of the fuel with defective cladding, and because of the corrosion of structural and cladding materials anywhere in the coolant circuit which leads to particles being carried through the core and becoming activated. All reactors have treatment systems for the removal of radio nuclides from gaseous and liquid wastes, which arise from leakage out of the core or from clean-up of the coolant.

53. The quantities of different radioactive materials discharged from reactors depend on the reactor type, its design and the specific waste treatment plant installed. Radionuclides discharged to atmosphere include fission noble gases (krypton and xenon), activation gases (\(^{14}\)C, \(^{16}\)N, \(^{35}\)S, \(^{41}\)Ar), tritium, iodine and particulates. Radionuclides released into the aquatic environment in liquid effluents usually include tritium, fission products and activated corrosion products. The discharge data for the years 1980-1985 are presented in this section, and the annual normalized releases are evaluated for each reactor type and averaged over all reactors of each type as TBq (GWa)\(^{-1}\). Normalized results are not presented for individual sites because releases in any one year may reflect a need for maintenance or irregular procedures which are the culmination of a number of years of previous operation. The total releases of radionuclides between 1980 and 1984 have been normalized by dividing by the total electric energy generated (GWa) over the same period. These normalized releases are used to assess collective dose commitments because the 1985 data were incomplete. Generally, the normalized releases for 1985 from the partial data lead to lower values than for the previous five years, although the 1980-1985 averages are mostly within 10% of the 1980-1984 averages.

1. Fission noble gases

54. At least nine identified radioactive isotopes of krypton and 11 of xenon are formed during fission. Most have half-lives of minutes or seconds and decay before they migrate significantly in the fuel. A fraction of the noble gas inventory of the fuel pins diffuses to the free space between the fuel and the cladding, leading to a build-up of gas pressure. The presence of noble gases in the coolant circuit is generally an indication of fuel cladding failure.

55. Table 17 lists the reported discharges of noble gases from PWRs. The releases span many orders of magnitude partly because of the design of newer plants and partly because of the need for irregular operations and maintenance. Thus, the normalized releases presented are averaged over all PWR electric energy production from 1980 to 1984. Short-lived noble gases only appear in PWR effluents because of leakages in the primary water pressure circuit. Gaseous wastes can also arise from the condenser exhaust on the steam circuit and from blow-downs or contain-
ment purges. These wastes are usually held under pressure in delay tanks to allow decay of short-lived isotopes before release. The isotopic composition of noble gases released from PWRs in the United States in 1982 is shown in Table 18. Comprehensive data are available for each year from United States reactors; and data available from other countries are similar to those from the United States. The data for the United States for 1982 are therefore assumed to be representative of the isotopic composition of releases between 1980 and 1984 and are used for dose estimation.

56. Data for 1985 are incomplete and the releases are not included in the normalized set. The normalized releases seem to have remained fairly steady over the five-year period 1980-1984, but the average of 218 ± 40 TBq (GW a)^-1 appears to be about half of the value reported previously by the Committee [430 TBq (GW a)^-1]. Xenon-133 accounted for 75% of the discharge and $^{132}$Xe for 12%. In the UNSCEAR 1982 Report the comparable figures were 85% and 5%, respectively. Some of the reduction in discharges is thought to be due to better fuel can performance, which would account for the lower releases to cooling water. The other feature is the inclusion of newer reactors with lower levels of discharge.

57. Reported discharges of noble gases from BWRs are shown in Table 19. The releases vary by six orders of magnitude, although the average releases continue to have been reduced from previous years. The normalized releases are shown in Table 19 for all BWRs from 1980 to 1985. The main source of noble gas release from BWRs is gases in the steam circuit that are continuously removed by the main condenser air-ejector system. The isotopic composition depends on the hold-up time, which is usually less than that for PWRs, thus allowing more short-lived isotopes to be released. Table 20 gives the radionuclide composition of noble gas releases from United States BWRs in 1982, which is similar to that of reactors in other countries. These figures again are taken to be representative of BWR releases in all countries during the period 1980-1984 and are used for dose assessment.

58. For BWRs the average discharge rate for noble gases during the five-year period 1980-1984 is 2,150 ± 520 TBq (GW a)^-1 compared with 8,800 TBq (GW a)^-1 reported in the UNSCEAR 1977 Report for 1975-1979. This reduction seems to have been achieved because of significant reductions in releases from those reactors that previously had the highest discharge rates. The normalized release for 1985 is significantly lower (460 TBq (GW a)^-1), partly because of the missing data but mainly because of very large reductions in discharges from the largest previous sources (Brown's Ferry and Brunswick). The isotopic composition shown in Table 20 reveals that most of the activity consists of $^{85}$Kr (half-life: 2.8 h), $^{129}$Xe (half-life: 5.3 d), $^{131}$Xe (half-life: 9.2 h) and $^{133}$Xe (half-life: 17 min) in almost equal quantities.

59. In GCRs, noble gas releases are insignificant compared with activation gases. Magnox reactors, AGRs, LWGRs and HWRs utilize on-load refuelling and, in the event of fuel element failure, fuel rods can be replaced. Releases of noble gases from HWRs and LWGRs are given in Table 21. Normalized release for HWRs has been 212 ± 48 TBq (GW a)^-1, similar to that for PWRs. The highest figures are for LWGRs at 5,470 ± 1,370 TBq (GW a)^-1, about three times the figures for BWRs. The available discharge data indicate that FBRS have lower releases of noble gases; measurements at BN-350, an FBR in the USSR, indicate 65-130 TBq (GW a)^-1 [P6].

2. Activation gases

60. Although GCRs do not generally release fission noble gases, several gases are formed during gas cooled reactor operation. These are primarily $^{40}$Ar, formed by activation of the stable argon in air, and $^{35}$S produced from sulphur and chlorine impurities in the graphite core. The discharge data for $^{40}$Ar are reported in Table 22. For $^{35}$S, measurements were made in the United Kingdom at Hinkley B, Oldbury and Wylfa [H1, H2], and discharges have consistently averaged 0.2 TBq (GW a)^-1 with only about 20% variation around the mean.

61. The quantity of $^{40}$Ar (half-life: 1.8 h) released depends upon the detailed design of the reactor. For early Magnox reactors having steel pressure vessels, the principal source of $^{40}$Ar is the activation of stable argon in the air used as cooling air around the outside of the pressure vessel. For advanced reactors with prestressed concrete pressure vessels, the principal source of $^{40}$Ar is leakage of the coolant CO$_2$, which contains small amounts of air, to the atmosphere. The normalized releases from AGRs are 5-15% of the values for Magnox reactors. The average normalized release from GCRs is 2,320 ± 220 TBq (GW a)^-1 compared to 3,240 TBq (GW a)^-1 in the UNSCEAR 1982 Report. The reduction is due to the proportion of power now generated by AGRs and addition of French data. For BN-350, the Soviet FBR, normalized $^{40}$Ar releases average 470 TBq (GW a)^-1 [K4, P6].

62. Nitrogen-16 (half-life: 7 s) causes direct external radiation at nuclear power plants. The photons produced in its decay have energies of 6.1 and 7.1 MeV. In BWRs, the $^{16}$N generated in the coolant is transferred in the steam to the turbine buildings. Direct radiation from gas ducts in steel pressure vessel gas cooled reactors produces the major dose to individuals close to those sites.

3. Tritium

63. In LWRs tritium arises from ternary fission in the nuclear fuel and from the neutron activation of lithium and boron isotopes dissolved in, or in contact with, the primary coolant. The Committee assessed in the UNSCEAR 1982 Report the tritium production rate from ternary fission as 0.75 PBq (GW a)^-1. Tritium generation from activation reactions in PWRs seems to result mainly from boron, which is used for reactivity control in the coolant, whereas in BWRs it results mainly from boron in control rods. In GCRs it is the result of lithium impurities in the graphite and
the presence of water vapour in the core. For HWRs it
is principally the result of the activation of the
deuterium moderator and coolant. The activation
production rate only exceeds that from ternary fission
in HWRs, where the activation rate was previously
estimated by the Committee to be 30 times higher at
about 25 Tg (GW a)\(^{-1}\).

64. Table 23 presents the tritium releases to the
atmosphere for 1980-1985 for PWRs, BWRs and
HWRs. For PWRs the normalized release over the
five-year period 1980-1984 is 5.9 ± 2.4 Tg (GW a)\(^{-1}\)
and no particular trend is apparent over this period.
The corresponding figure was 7.8 Tg (GW a)\(^{-1}\) for
1975-1979. The BWR releases normalized for the same
period averaged 3.4 ± 1.6 Tg (GW a)\(^{-1}\), compared
with 3.4 Tg (GW a)\(^{-1}\) for 1975-1979. The decrease in
annual BWR normalized releases from 1980 to 1984
seems primarily attributable to reductions from the
Dresden nuclear plant alone, while the higher figure
for 1985 is due to Hatch 1. These figures indicate that
about 1% of the tritium produced in the fuel of LWRs
finds its way into the coolant and then enters airborne
effluent streams. For HWRs the production of tritium
in the moderator is the most probable source of tritium
releases, which averaged 670 ± 190 Tg (GW a)\(^{-1}\)
for 1980-1984, as compared with 540 Tg (GW a)\(^{-1}\)
for 1975-1979. For some HWRs, however, the coolant
may be the main source of tritium production. There
is little release of tritium to the atmosphere from
Magnox gas cooled reactors mainly because humidifiers
remove water vapour from the gas circuit. There is
some release of tritium to the atmosphere from AGRs,
and the normalized release is 5.4 ± 0.9 Tg (GW a)\(^{-1}\),
similar to LWR releases.

65. From Table 24 it can be seen that releases of
tritium to the hydrosphere from PWRs have been
fairly constant over the past five years, and the 1980-
1984 normalized average is 27 ± 1.8 Tg (GW a)\(^{-1}\),
with the figure for 1985 similar. This compares to the
38 Tg (GW a)\(^{-1}\) obtained for 1975-1979. The comparable
figures for BWRs are 2.1 ± 0.5 Tg (GW a)\(^{-1}\) for
1980-1984, which is 50% higher than the 1.4 Tg
(GW a)\(^{-1}\) for 1975-1979 and no trend is apparent. For
AGRs the normalized release to surface waters is
96 ± 13 Tg (GW a)\(^{-1}\), which contrasts with 25 Tg
(GW a)\(^{-1}\) for 1975-1979. There appears to have been
a significant increase in tritium releases from GCRs
over the past five years. HWR releases in liquid
effluent streams averaged 290 ± 68 Tg (GW a)\(^{-1}\)
for 1980-1984, compared with 350 Tg (GW a)\(^{-1}\) for
1975-1979. LWRGRs have low liquid releases at 1.7 Tg
(GW a)\(^{-1}\).

66. Thus, about 0.3% of BWR tritium production
appears in liquid effluents, with a similar amount
goin the atmosphere. For PWRs about 3% of the
tritium produced is in liquid effluents, about five times
more than that going to the atmosphere. For HWRs
liquid effluents are about one half those discharged to
the atmosphere.

67. For PWRs and LWRGRs in the USSR the
atmospheric releases of tritium are reported to average
7.4 Tg (GW a)\(^{-1}\) and 1.9 Tg (GW a)\(^{-1}\), respectively,
[B6, V4], similar to other PWRs and AGRs. The
liquid discharges amount to about 5 Tg (GW a)\(^{-1}\)
and 1 Tg (GW a)\(^{-1}\) for PWRs and LWRGRs, respectivly
[B6, P6, V4]. Measurements indicated that on
average 90% of the atmospheric releases of tritium
was in oxide form [B6]. Practical experience at the
Novovoronezh APS (PWR) showed that it is possible to
reduce the tritium concentration in the coolant water by
50% [B7].

4. Carbon-14

68. Discharges of \(^{14}\)C are of interest because of its
long half-life (5,730 a) and contribution to collective
dose commitments. Estimates of \(^{14}\)C production in
fuels depend on the nitrogen level in the fuel can,
although some is produced from reactions on oxygen
in oxide fuels. The Committee concluded in the
UNSCEAR 1982 Report that the normalized production
rate within fuel for PWRs, BWRs, GCRs and
HWRs was close to 1 Tg (GW a)\(^{-1}\). Little of this is
released into the reactor coolant circuits, it appears to
be released during reprocessing [B1, B2, B3, B8, B16,
B29]. Carbon-14 is produced in the moderators of all
reactors, production in HWRs being perhaps 100 times
greater than in LWRs or GCRs, because of the \(^{17}\)O
(n, a) \(^{14}\)C reaction in the greater mass of oxygen in the
moderator, and there is a consequential release.

69. The National Council on Radiation Protection
and Measurements (NCRP) [N1] has estimated the
production rate of \(^{14}\)C in PWRs to be between 2 and
3 Tg (GW a)\(^{-1}\), and for BWRs 3-4 Tg (GW a)\(^{-1}\),
arising in both cases mainly in stainless steel and
zirconium alloy. For the estimation of release rates to
the environment, NCRP assumes that the \(^{14}\)C formed
in the hardware remains there, but that the fraction
formed in dissolved nitrogen in the cooling water is
totally released. The NCRP estimate for PWRs is
370 Gtq (GW a)\(^{-1}\) and for BWRs 220 Gtq (GW a)\(^{-1}\).
The NCRP estimate of the release of \(^{14}\)C to the
environment for FBRs is essentially zero at the
reactor.

70. Environmental discharges of \(^{14}\)C are not routinely
reported for all reactors. The data summarized in
Table 25 are from a series of measurements made in
Argentina, the Federal Republic of Germany, Finland,
and USSR. For BWRs it appears that essentially all the
\(^{14}\)C appears as carbon dioxide, and the normalized
release rate for 1980-1984 is 330 ± 110 Gtq (GW a)\(^{-1}\),
significantly less than the Committee's estimate in the
UNSCEAR 1982 Report of 520 Gtq (GW a)\(^{-1}\). For
PWRs in the Federal Republic of Germany [W1],
Finland [B17] and the USSR [R1], the data indicate a
release rate of 345 ± 80 Gtq (GW a)\(^{-1}\), which is
significantly higher than the figure of 220 Gtq (GW a)\(^{-1}\)
given in the UNSCEAR 1982 Report. For PWRs only
about 5-50% of the emission appears as CO\(_2\). It now
appears that normalized \(^{14}\)C releases from PWRs and
BWRs are similar.

71. In recent measurements at three LWRs in the
United States [K2], two PWRs emitted an average of
390 Gtq (GW a)\(^{-1}\). The source of \(^{14}\)C was different at
the two sites: the first had 42% arising from venting of gas decay tanks, 35% from auxiliary building ventilation and 32% from containment venting: the second had emissions resulting primarily from pressure relief venting and purging of the containment air, with only 7% from venting of gas decay tanks. For the BWR, the discharge rate was 460 GBq (GW a)\(^{-1}\) with 97% of the release via the off-gas discharge, which was 95% \(^{14}\)CO\(_2\). For the PWR 94% of the discharge was \(^{14}\)CH\(_4\). The \(^{14}\)C content of liquid and solid wastes was less than 5% of the aerial discharge for all reactors.

72. Measurements at LWGRs in the USSR gave average releases of 1.3 TBq (GW a)\(^{-1}\) [R1]. In the United Kingdom, reported releases were 0.74 TBq (GW a)\(^{-1}\) from Magnox reactors and 1.9 TBq (GW a)\(^{-1}\) from AGRs. Weighted by energy production, the normalized \(^{14}\)C release for GCRs is 1.1 TBq (GW a)\(^{-1}\) [H8]. The main source of \(^{14}\)C releases from GCRs is the leakage of the primary coolant, at a rate typically of a few per cent per day, which contains radionuclides released to the coolant by corrosion of the graphite moderator.

73. For HWRs it has been reported that a significant fraction of the inventory formed in the moderator can be released to atmosphere. Measurements at Atucha 1 [B18, O3], however, for 1983-1985 have indicated that releases are significantly lower than previously calculated for 1980-1982. The five-year normalized release is 6.3 ± 3.3 TBq (GW a)\(^{-1}\), whereas the Committee had estimated 17 TBq (GW a)\(^{-1}\) in the UNSCEAR 1982 Report. The form is again variable, between 40 and 80% being reported as \(^{14}\)CO. In Argentina, regular monitoring of discharges of \(^{14}\)C has continued for several years so that more reliable estimates can be made.

5. Iodine

74. The volatile element iodine is produced by the fission process, the isotopes of radiological interest being \(^{129}\)I (half-life: 1.6 \(\times 10^7\) a), \(^{131}\)I (half-life: 8.04 d), \(^{132}\)I (half-life: 2.3 h), \(^{133}\)I (half-life: 21 h), \(^{134}\)I (half-life: 53 m) and \(^{135}\)I (half-life: 6.6 h). Because, apart from \(^{129}\)I, the iodine isotopes have such short half-lives, equilibrium activity concentrations in the fuel are reached quickly and releases depend on the number of fuel cladding failures and the coolant leakage rate. Iodine has been studied for many years in view of its mobility in the environment and selective thyroid irradiation. Because of its long half-life, \(^{131}\)I is of interest in evaluating collective dose commitments; however, its release from reactors is very small and often not reported. Most of \(^{129}\)I in fuel is released during reprocessing, from which it makes a greater contribution than from reactor operation.

75. Table 26 gives the reported atmospheric discharges of \(^{131}\)I from operating reactors in various countries for 1980-1985. There are considerable differences in the absolute quantities; these appear to be attributable to differences in the ages of the plants and in the waste treatment designs. There does not appear to be any trend in PWR releases, but BWR normalized data show a sharp downward trend.

76. The annual normalized discharges of \(^{131}\)I from PWRs were 1.75 ± 0.33 GBq (GW a)\(^{-1}\) for 1980-1984, not significantly changed when compared with 1.9 GBq (GW a)\(^{-1}\) for 1975-1979. The \(^{131}\)I releases from BWRs for 1980-1984 have averaged 9.3 ± 4.9 GBq (GW a)\(^{-1}\) compared with \(^{131}\)I releases of 40 GBq (GW a)\(^{-1}\) for 1975-1979. This reduction was because the few reactors that had large releases are currently releasing far less. The results for HWRs indicate releases of 0.23 ± 0.08 GBq (GW a)\(^{-1}\). From early GCRs, which utilize on-load refuelling, releases were negligible, and releases from AGRs were 1.4 ± 1.1 GBq (GW a)\(^{-1}\). LWGRs released 80 ± 40 GBq (GW a)\(^{-1}\) [A1], and measurements indicated that 60% of the iodine in the reactor off-gases was in organic form, 40% inorganic and about 1% particulate [B9, D1, S6].

77. The isotopic composition of iodine releases from LWRs in the United States in 1982 is shown in Table 27 [T5]. The isotopic composition was taken to be representative of reactor operations in all countries and was used as the basis for dose calculations. For PWRs about 25% of the discharge is accounted for by \(^{131}\)I and 75% by \(^{131}\)I, compared with the figures reported by the Committee in the UNSCEAR 1982 Report of 30% accounted for by \(^{131}\)I. For BWRs, \(^{131}\)I releases represented about 7% of the discharges, with \(^{131}\)I and \(^{133}\)I contributing 28% and 65%, respectively. This compares with less than 10% previously reported for \(^{131}\)I and 30% and 60% for \(^{131}\)I and \(^{133}\)I, respectively. For LWGRs, 24% is accounted for by \(^{131}\)I, 43% by \(^{131}\)I and 33% by \(^{131}\)I [B21]. It might be concluded that there was little change in the isotopic composition in the periods 1975-1979 and 1980-1984.

6. Particulates in airborne effluents

78. Radionuclides in particulate form can arise directly or as decay products of fission noble gases or may arise from corrosion of materials in the primary coolant circuit. Aerosols are generated because of primary circuit leaks or because of maintenance work on active components removed from the primary circuit. The air in all areas where aerosols might arise is continually purged and the plenum activity is filtered by high efficiency particulate (HEPA) filters. Results of recent measurements on particle size distributions indicate a mean aerodynamic diameter of 1 \(\mu\)m for fission products and 10\(\mu\)m for activation products [B4]. Measurements at LWGRs in the USSR have indicated mean aerodynamic diameters of 0.1-0.4 \(\mu\)m for particulates; for \(^{51}\)Cr and \(^{51}\)Cr, 30-40% were particulates with a mean aerodynamic diameter of less than 0.1 \(\mu\)m [B10, C2].

79. Releases of particulate activity to the atmosphere are summarized in Table 28 for reactors around the world. The quantities are extremely low, and the nuclide composition appears to be unique to each operating plant; it depends on the particular impurities in cladding and structural materials, coolant chemistry and fuel failure modes. The isotopic composition of the release from a plant can vary from year to year, because of different operational and maintenance needs. Consequently, the range of nuclides reported in
atmospheric discharges is extremely large, several tens of nuclides often being reported from one plant. No single nuclide can be identified as contributing the majority of the activity released for any one type of reactor. Radionuclides identified include 7Be, 22Na, 31Cr, 32Mn, 32Fe, 52Co, 60Co, 61Ni, 125Zn, 75As, 85Rb, 88Sr, 90Sr, 91Sr, 93Zr, 95Zr, 99Mo, 99mTe, 103Ru, 106Ru, 108Ru, 109mAg, 110mAg, 113Sn, 115Cd, 112Sb, 112Sb, 113Sb, 112Sm, 112Te, 114Cs, 115Cs, 137Cs, 140Ba, 140La, 144Ce, 144Ce and 137Ba.

80. For PWRs the normalized release was 4.5 ± 2.9 GBq (GW a)\(^{-1}\) for 1980-1984, compared with 2.2 GBq (GW a)\(^{-1}\) for 1974-1979. For BWRs, the average release was 43 ± 24 GBq (GW a)\(^{-1}\), compared with 53 GBq (GW a)\(^{-1}\) for 1974-1979. For HWRs the data yield 0.04 ± 0.016 GBq (GW a)\(^{-1}\), similar to the 0.044 GBq (GW a)\(^{-1}\) normalized release reported previously. While for LWRs the average release appears to have been 15.7 ± 16.2 GBq (GW a)\(^{-1}\). There were no figures previously for LWRs, nor were there any for GCRs, which now average 1.4 ± 0.8 GBq (GW a)\(^{-1}\).

7. Liquid effluents

81. The sources of radionuclides other than tritium in liquid effluents are essentially the same as those described for particulate releases to the atmosphere. The reported levels of discharge are equally variable, the magnitude and isotopic composition depending upon the design and operating practice of the reactor, impurity levels and trace quantities of material in structural and cladding components. Table 29 summarizes reported liquid effluent discharges from reactors around the world. In Table 30 the isotopic composition of liquid discharges from power reactors in the United States in 1982 is presented, and in Table 31, that for GCRs in the United Kingdom is given, also in 1982.

82. The normalized release levels based on the reported discharges for each reactor type using reported figures for electric energy generated between 1980 and 1984 can be summarized from Table 29 and contrasted with the figures given in the UNSCEAR 1982 Report.

PWR: 132 ± 49 GBq (GW a)\(^{-1}\),

compared with 180 GBq (GW a)\(^{-1}\)

BWR: 115 ± 47 GBq (GW a)\(^{-1}\),

compared with 290 GBq (GW a)\(^{-1}\)

GCR: 4.520 ± 1.790 GBq (GW a)\(^{-1}\),

compared with 4.800 GBq (GW a)\(^{-1}\)

HWR: 25.7 ± 8.7 GBq (GW a)\(^{-1}\),

compared with 470 GBq (GW a)\(^{-1}\)

The normalized releases for PWRs between 1980 and 1984 are similar to previous years although there has been an increasing trend, while BWR releases are less. Canadian HWRs were previously reported as giving discharges of about 50 GBq (GW a)\(^{-1}\), while the higher figures for the GCRs reflect the fact that discharges are made, with the exception of Trawsfynydd, to the marine environment. It appears from the above results that aquatic discharges from BWRs have been reduced by a factor of 2.5. In the UNSCEAR 1982 Report, the Committee found that PWR releases had been reduced by a factor of about 2 and BWR releases by a factor of 10 from the figures given in its UNSCEAR 1977 Report. These reductions do not seem to have been attributable to the removal of any single nuclide but are applicable to all nuclides in the release.

83. The isotopic composition of liquid effluents from United States reactors in 1982 is shown in Table 30. About 20% of the normalized PWR discharge is due to \(^{60}Co\) and almost 20% to \(^{134}I\), while \(^{137}Cs\) accounts for about 11%. In the BWRs about 30% of the release is due to \(^{60}Co\) and about 13% to \(^{137}Cs\). The other nuclides with significant contributions being \(^{23}Na\) and \(^{89}Zn\) and \(^{131}I\) contributed about 3%. These figures represent small changes from those in the UNSCEAR 1982 Report, with some reduction in the percentages of caesium.

84. For GCRs, 40% of the discharges to the aquatic environment are due to \(^{137}Cs\) and the ratio of \(^{134}Cs\) to \(^{137}Cs\) is 0.22, compared with 0.6 for PWRs and 0.5 for BWRs, which reflects differences in fuel burn-up. About 16% of GCR releases is due to \(^{35}S\), and \(^{89}Sr\) accounts for about 6%.

85. There is a wide range of activation products and fission products reported in liquid effluent discharges, and the isotopic composition varies even between reactors of the same type. The normalized figures are used, however, to make an estimate of the collective doses due to liquid effluent discharges.

B. LOCAL AND REGIONAL COLLECTIVE DOSE COMMITMENTS

86. National authorities usually require environmental monitoring programmes in the vicinity of a nuclear power plant to be carried out by the operator, another competent agency or both. In general, activity concentrations of radioactive materials from effluent discharges are too low to be measurable except close to the immediate point of discharge. Dose estimates for the population, therefore, rely on modelling the environmental transfer and transport of radioactive materials.

87. In the UNSCEAR 1982 Report, the Committee established a model site that was most representative of areas of northern Europe and north-eastern United States, since those areas contain a large proportion of the power-producing reactors. Agricultural production patterns and population distributions typical of those areas were also established. The cumulative population within 2,000 km of the site is about 2.5 10^6, giving an average population density of 20 km^-2. Within 50 km of the site, the population density was taken to be 400 km^-2 in order to reflect current siting practice. The objective of the Committee remains unchanged to give a representative value of the collective dose commitments per unit of electric energy generated by nuclear power stations and to reflect the levels of dose received by the most exposed individuals. The results do not apply to any one reactor or any one location, and the collective dose
commissions should not be applied to a given reactor with known discharge data to obtain estimates of health detriment.

1. Fission noble gases

88. Using the normalized releases for PWRs from Table 17 for noble gas atmospheric releases and the radionuclide composition from Table 18, the normalized collective effective dose equivalent commitments averaged between 1980 and 1984 were calculated for the model PWR facility and are shown in Table 32. The normalized release term from Table 17 is 218 TBq (GW a)^{-1}, and the radionuclides that contribute significantly to the collective effective dose are ^{133}Xe and ^{135}Xe. The in-growth of daughter products, e.g., ^{88}Rb from ^{88}Kr, has been included in the dose calculations, which are those presented in the UNSCEAR 1982 Report, but scaled for the different normalized release and isotopic composition.

89. The normalized collective effective dose equivalent commitment amounts to 2.6 \times 10^{-2} mSv (GW a)^{-1} compared with the Committee's assessment of 4.2 \times 10^{-2} mSv (GW a)^{-1}, which was given in Annex F of the UNSCEAR 1982 Report. This reflects the reduction in discharges with little difference in the distribution of radionuclide composition. About 64\% of the total collective dose is now due to ^{133}Xe (80\% in 1982) and 28\% to ^{135}Xe (11\% in 1982). As in the UNSCEAR 1982 Report, some 90\% of the collective dose commitment is accumulated within 500 km. There is little contribution from the inhalation of radioactive daughter products, and the dose estimates, as before, include an allowance for the shielding from buildings and the fraction of time spent indoors.

90. For the quinquennium 1980-1984, Table 19 shows the normalized releases from BWRs to be 2,150 TBq (GW a)^{-1}, compared with the value of 8,800 TBq (GW a)^{-1} given in the UNSCEAR 1982 Report. Taking the relative isotopic composition from Table 20, the normalized collective effective dose equivalent commitment is given in Table 33 as 0.56 mSv (GW a)^{-1}, compared with the Committee's estimate in the UNSCEAR 1982 Report of 1.9 mSv (GW a)^{-1}. The main isotope contributing to the collective dose is ^{89}Kr (half-life: 2.8 h) accounting for about 57\%, somewhat more than in 1982. Most of the remainder of the collective dose arises from ^{135}Xe (16\%), ^{134}Xe (9\%) and ^{133}Xe (8\%), in somewhat smaller proportions than in the UNSCEAR 1982 Report.

91. The in-growth of ^{88}Rb (half-life: 15.4 min) from ^{87}Kr and ^{134}Cs (half-life: 32.2 min) from ^{134}Xe decays are included in the dose estimation, and the collective doses include a contribution from the inhalation of the ^{88}Rb and ^{134}Cs radioisotopes. The spatial distribution of the normalized collective effective dose equivalent commitment is biased towards the source, with more than 80\% of the dose accumulated within 50 km and nearly 50\% within 10 km. This behaviour is caused by the dominance of ^{88}Kr, which decays with a half-life corresponding to about 40 km distance travelled.

92. The normalized release of noble gases from HWRs is 212 TBq (GW a)^{-1} (Table 21), and assuming the same relative isotopic composition as PWRs, the normalized collective dose commitment is 0.024 mSv (GW a)^{-1}, while for LWRs a normalized release of 5,470 TBq (GW a)^{-1} (Table 21) and the assumption of an isotopic composition similar to that of BWRs yield a normalized collective effective dose equivalent commitment of 0.72 mSv (GW a)^{-1}.

93. In summary, the normalized collective effective dose equivalent commitment from noble gas releases is 0.20 mSv (GW a)^{-1}, based on the five-year (1980-1984) weighting of electricity generated by PWRs, BWRs, HWRs and LWGRs. The Committee gave a figure of 0.63 mSv (GW a)^{-1} in the UNSCEAR 1982 Report, so that an average reduction of dose from noble gas effluents of about a factor of 3 has been found owing to reductions in reported discharge levels, mainly from BWRs. The annual average effective dose equivalent to the most exposed individuals in hypothetical critical groups has been calculated at 10 \mu Sv for the model BWR and more than 10 times lower for the PWR site, taking an average distance of about 1 km from the site. Many reactors give lower doses, although for some early BWRs, the doses could be about 10 times higher.

2. Activation gases

94. The primary activation product of interest for gaseous releases is ^{41}Ar. Because of its short half-life (1.83 h), it contributes most of its collective dose within a few tens of kilometres of the release point, although the exact result depends on the close-in population density. The normalized release of ^{41}Ar from GCRs (Table 22) between 1980 and 1984 is 2,320 TBq (GW a)^{-1}, and the associated collective effective dose equivalent commitment is 0.78 mSv (GW a)^{-1}, compared with the estimate in the UNSCEAR 1982 Report of 0.95 mSv (GW a)^{-1}. The reduction is due to the fact that newAGR s are producing electricity with much lower ^{41}Ar discharges than GCRs. The weighted collective effective dose equivalent commitment, allowing for the fraction of electricity generated by GCRs, is 0.039 mSv (GW a)^{-1}, significantly lower than the value given in the UNSCEAR 1982 Report of 0.1 mSv (GW a)^{-1}. Because of reporting procedures, ^{41}Ar releases for LWRs are included in noble gas data as shown in Tables 18 and 20.

95. The consequences of the release of ^{90}Sr from GCRs have been studied in detail. The isotope is released in the form of carbonyl sulphide (COS), which has a low deposition velocity and a slow reaction rate in air. The major route of human exposure is via milk, and the Committee estimated 2.2 \times 10^{-4} mSv GBq^{-1} in the UNSCEAR 1982 Report. so that, using the normalized release of 200 GBq (GW a)^{-1}, the normalized collective effective dose equivalent commitment is 0.044 mSv (GW a)^{-1} and the contribution weighted for GCR electricity production is 2.4 \times 10^{-3} mSv (GW a)^{-1}, compared with 3.8 \times 10^{-3} mSv (GW a)^{-1} in the UNSCEAR 1982 Report.
3. Tritium

96. The collective effective dose equivalent commitment to the local and regional population was evaluated in the UNSCEAR 1982 Report on the basis of a specific activity model. For atmospheric releases, the Committee obtained 1.5 \(10^{-3}\) man Sv TBq\(^{-1}\) by inhalation and 9 \(10^{-3}\) man Sv TBq\(^{-1}\) by ingestion to give a total of 0.011 man Sv TBq\(^{-1}\) released.

97. Normalized tritium atmospheric releases for the quinquennium 1980-1984 from PWRs are 5.9 TBq (GWa)\(^{-1}\) (paragraph 64), giving 0.065 man Sv (GWa)\(^{-1}\); BWR releases of 3.4 TBq (GWa)\(^{-1}\) give 0.037 man Sv (GWa)\(^{-1}\); HWR releases of 670 TBq (GWa)\(^{-1}\) give 7.4 man Sv (GWa)\(^{-1}\) for atmospheric releases. Releases from GCRs and LWGRs are comparable with PWRs and give similar dose contributions. In summary, weighted by the amount of electricity generated by reactor type, the normalized collective effective dose equivalent commitment for atmospheric releases of tritium is 0.53 man Sv (GWa)\(^{-1}\), compared with the Committee's estimate given in the UNSCEAR 1982 Report of 0.46 man Sv (GWa)\(^{-1}\). For the model site used by the Committee, the annual individual effective dose equivalent for critical groups would be less than 1 \(\mu\)Sv from LWR atmospheric \(^1\)H releases, while the HWR dose would be 50 \(\mu\)Sv per year.

98. For tritium in liquid effluents, the river model used by the Committee gave a collective effective dose equivalent commitment of 8.1 \(10^{-4}\) man Sv TBq\(^{-1}\), on the assumption that the river is used as a source of drinking water. Using the normalized discharges for 1980-1984 for PWRs, BWRs, HWRs, GCRs and LWGRs from paragraph 65, the normalized collective effective dose equivalent commitments were calculated and are shown in Table 34. The normalized dose weighted by the proportion of electricity generated by each reactor type is 0.033 man Sv (GWa)\(^{-1}\), which compares with the estimate of 0.04 man Sv (GWa)\(^{-1}\) given in the UNSCEAR 1982 Report. The doses from aquatic discharges are therefore about 16 times lower than for atmospheric effluents per unit electric energy generated, similar to the difference of a factor of 10 reported in the UNSCEAR 1982 Report.

4. Carbon-14

99. The local and regional collective doses attributable to \(^14\)C releases from reactors represent only a small proportion of the total dose commitments. The main significance of \(^14\)C stems from its entry into the carbon cycle and the resulting global dispersion, leading to long-term irradiation, which is considered in chapter V. The first pass local and regional collective dose commitment was previously assessed by the Committee using the specific activity approach which was also used for tritium. The Committee also assumed in the UNSCEAR 1982 Report that the form of release of \(^14\)C was CO\(_2\). The normalized local and regional collective effective dose equivalent commitment per unit release previously determined by the Committee was 1.8 man Sv TBq\(^{-1}\) for ingestion and 0.0003 man Sv TBq\(^{-1}\) for inhalation following release to the atmosphere. The normalized doses per unit electric energy generated are shown in Table 35 and are based on the normalized releases taken from Table 25.

100. The normalized collective doses ranged from 0.59 man Sv (GWa)\(^{-1}\) for BWRs to over 11 man Sv (GWa)\(^{-1}\) for HWRs. The weighted average, allowing for the proportion of electricity generated by each reactor type, was 1.6 man Sv (GWa)\(^{-1}\) to the local and regional population. This is about one half of the estimate of 2.8 man Sv (GWa)\(^{-1}\) given in the UNSCEAR 1982 Report, largely because of lower reported HWR releases. For the model site the annual effective dose equivalents to most exposed individuals was 3 \(\mu\)Sv for PWRs and BWRs, 10 \(\mu\)Sv for GCRs, about 70 \(\mu\)Sv for HWRs and about 13 \(\mu\)Sv for LWGRs.

5. Iodine

101. Releases of radioactive iodine from nuclear power plants are small, and there is little contribution to the local and regional collective effective dose equivalent commitment. Because of its long radioactive half-life, \(^131\)I enters the global cycle for iodine and has the potential to irradiate the global population for tens of millions of years. The release of \(^131\)I contributes only to the local and regional effective doses, but its assessment is complicated by the chemical form in which the iodine is released, i.e., elemental, organic or particulate. Elemental iodine readily deposits on vegetation and enters terrestrial food chains. The deposition of organic iodine is usually less than 1% of that for elemental iodine per unit time integrated air concentration. In this Annex, as in the UNSCEAR 1982 Report, the Committee assumes that 75% of the iodine released is in organic form and 25% in elemental form.

102. In the dose evaluation used in the UNSCEAR 1982 Report, the collective effective dose equivalent commitment per unit release of \(^131\)I was 4.0 \(10^{-4}\) man Sv GBq\(^{-1}\). Taking the releases of \(^131\)I from Table 26, the normalized collective doses for \(^131\)I per unit of electric energy generated were calculated and are shown in Table 36. Results for the other iodine isotopes are found by scaling from the results in the UNSCEAR 1982 Report, allowing for the change in isotopic composition. The PWR figures are about 10% lower than in the UNSCEAR 1982 Report, while the BWR results are about 25% of those found previously. For both BWR and LWGR reactors the short-lived \(^129\)I and \(^133\)I make significant additions to the dose. The weighted average, taking into account the proportion of electricity generated by each reactor type, is 3.3 \(10^{-4}\) man Sv (GWa)\(^{-1}\). Representative effective doses to individuals about 1 km from the model site are about 0.5 \(\mu\)Sv per year for PWR releases and 4 \(\mu\)Sv per year for the BWR releases. As in the UNSCEAR 1982 Report, 90% of the collective dose contribution is estimated to come from the milk pathway.

6. Particulates in airborne effluents

103. As noted in paragraph 79, the quantities of radionuclides in particulate releases to the atmosphere may vary greatly, even if releases from reactors of the
same type or those from the same reactor from year to year are compared. Furthermore, there are several tens of radionuclides identified in the releases. The solution previously adopted by the Committee for estimating doses was to assume that the normalized releases are composed of equal amounts of activity concentration from a range of nuclides most frequently reported in atmospheric discharges.

104. Dosimetric calculations allowed for transfer through foodchains to man as well as external irradiation from deposited radionuclides and inhalation from the dispersing plume of activity. Allowance was made for uptake by roots of growing vegetation. The full environmental modelling and resultant doses were described in the UNSCEAR 1982 Report. The nuclides considered were $^{51}$Cr, $^{54}$Mn, $^{59}$Fe, $^{60}$Co, $^{63}$Zn, $^{89}$Sr, $^{90}$Y, $^{99}$Zr, $^{103}$Nb, $^{124}$Sb, $^{133}$Cs, $^{134}$Cs, $^{137}$Cs, $^{140}$Ba, $^{140}$La, $^{142}$Ce and $^{144}$Ce.

105. The collective effective dose equivalent commitments per GBq release of the isotopic mixture is taken from the UNSCEAR 1982 Report to be $5.4 \times 10^{-1}$ man Sv (GBq)$^{-1}$, with nearly two thirds coming from external radiation from deposited activity and one third from ingestion. The collective doses per unit energy generated have been calculated using the normalized releases from Table 28 and are shown in Table 37, from which it can be seen that the most important pathway is the external dose received from activity deposited on the ground, followed by the dose from ingested foodstuffs. The normalized doses cover three orders of magnitude, with HWRs giving the lowest figure of 0.00022 man Sv (GW a)$^{-1}$ and BWRs the highest value of 0.23 man Sv (GW a)$^{-1}$. The doses can be compared with the previous figure of 0.012 man Sv (GW a)$^{-1}$ for PWRs, about one half of the current estimate. For BWRs the previous figure of 0.29 man Sv (GW a)$^{-1}$ was slightly higher than the present value. GCR figures are significantly lower than before (0.007 man Sv (GW a)$^{-1}$ compared with 0.012 man Sv (GW a)$^{-1}$).

106. Some 95% of the collective effective dose equivalent commitment from ground deposits is delivered within 50 years of the deposition and the major nuclides contributing are $^{137}$Cs and $^{60}$Co. For ingestion, $^{89}$Sr, $^{124}$Cs and $^{137}$Cs all contribute equally by three pathways: grain, vegetables and meat. The normalized collective effective dose equivalent commitment, weighted for the proportion of electricity produced by each reactor type, is 0.08 man Sv (GW a)$^{-1}$, essentially the same as the estimate of 0.1 man Sv (GW a)$^{-1}$ in the UNSCEAR 1982 Report. Individual dose equivalent commitments from the normalized releases at the end of a plant’s operating lifetime range from about 0.01 $\mu$Sv at 1 km from the model BWR to 1,000 times less for HWRs.

7. Liquid effluents

107. Aquatic releases are made into freshwater or marine environments. For releases of radionuclides into rivers or lakes, the pathways of exposure were previously taken by the Committee to be drinking water, irrigation leading to transfer to foodstuffs, and external radiation from sediments. For discharges to marine environments it is usually sufficient to consider the ingestion of ocean fish and crustacea. In the UNSCEAR 1982 Report pathways such as swimming in contaminated waters or consumption of unusual food items were considered to contribute little to the collective dose commitment.

108. The Committee has recognized the difficulty in assigning values to parameters in assessing the consequences of liquid effluents, in particular, water utilization and flow rates for rivers, fish production rates and sedimentation rates. The assessments based on the model used in the UNSCEAR 1982 Report, therefore, must be regarded as merely giving a representative value of nuclear power impact and should not be applied to discharges from a specific site to estimate collective doses from that site.

109. The normalized releases for PWRs, BWRs, GCRs, HWRs and LWGRs for 1980-1984 were summarized in paragraph 82, and the isotopic composition for these discharges were assumed to be those of United States reactors shown in Table 30 and those of United Kingdom reactors in Table 31. Collective effective dose equivalent commitments were evaluated assuming the discharges took place to freshwater and to marine environments. The results are shown in Tables 38 and 39.

110. The normalized collective effective dose equivalent commitment for releases from the PWR to fresh water was $1.6 \times 10^{-3}$ man Sv (GW a)$^{-1}$, compared with the finding of $1.0 \times 10^{-4}$ man Sv (GW a)$^{-1}$ in the UNSCEAR 1982 Report. Drinking water accounted for about 80% of the total and $^{131}$I was the major contributing nuclide (70% of the dose). For BWRs the normalized collective effective dose equivalent is $6.6 \times 10^{-4}$ man Sv (GW a)$^{-1}$, compared with the assessment of $2.8 \times 10^{-3}$ man Sv (GW a)$^{-1}$ in the UNSCEAR 1982 Report. The reduction is the result of an overall decrease of discharge, as well as a greater reduction in the more significant radiological nuclides. About 75% of the dose was from drinking water, and $^{60}$Co, $^{131}$I, $^{124}$Cs and $^{137}$Cs contributed almost equally to this dose. No results were provided for GCRs since these were coastal-sited. The collective dose from these radionuclides normalized for the amount of electricity generated were $1.1 \times 10^{-3}$ man Sv (GW a)$^{-1}$ for PWR and BWR releases to the model river.

111. For PWR releases to salt water the normalized results are shown in Table 39. The collective effective dose equivalent commitment was $3.6 \times 10^{-3}$ man Sv (GW a)$^{-1}$, about one half the value found in the UNSCEAR 1982 Report. The fish and mollusc pathways were both equally important, although the most important nuclide was different for each pathway: $^{137}$Cs for fish and $^{60}$Co for molluscs.

112. For BWR releases to the marine environment, the normalized collective effective dose equivalent commitment was $3.8 \times 10^{-3}$ man Sv (GW a)$^{-1}$, compared with the figure of $4.2 \times 10^{-2}$ man Sv (GW a)$^{-1}$ given in the UNSCEAR 1982 Report. The major contribution, was, as before, from $^{60}$Zn, which concentrates in
molluscs, and therefore the marine results differed markedly from those for fresh water. For GCR releases the normalized collective effective dose equivalent commitment was 0.19 man Sv (GW a)\(^{-1}\), essentially the same value found in the UNSCEAR 1982 Report. The majority of the dose arose from discharges of \(^{137}\)Cs. The weighted normalized collective effective dose equivalent commitment, allowing for the respective electricity generation was 0.025 man Sv (GW a)\(^{-1}\).

113. Again, it should be emphasized that the figures given in Tables 38 and 39 are representative of the generation of unit quantity of electric energy and should not be applied to a specific site where particular releases and specific environmental pathways exist that have not been considered here and might lead to significant differences in collective dose contributions. The normalized collective effective dose equivalent commitment due to aquatic discharges has been estimated to be 0.013 man Sv (GW a)\(^{-1}\), assuming that one half the discharges are made to fresh water and one half to the marine environment.

C. OCCUPATIONAL EXPOSURES

114. As was noted in the UNSCEAR 1982 Report, more data on occupational exposure to radiation are reported for reactor operation than for any other area. There are difficulties in normalizing data on occupational exposure to the electric energy generated, particularly for water reactors, as most of the doses are incurred during maintenance when no energy is produced. Normalized results are therefore only usefully derived over several years for a number of reactors. Average annual effective dose equivalents to reactor workers were estimated to be similar in the UNSCEAR 1977 and 1982 Reports and ranged from 3 to 8 mSv. During the same period, however, there was a large increase in the number of workers per reactor in the United States. The trend in the normalized collective effective dose equivalent was downwards, but overall the best estimate for LWRs was taken in the UNSCEAR 1982 Report to be 10 man Sv (GW a)\(^{-1}\), the same as in the UNSCEAR 1977 Report.

115. As can be seen from Table 1, PWRs and BWRs have been installed in many countries, although installed capacity in 1987 was still dominated by the United States. Recent data on occupational exposure and normalized collective effective dose equivalents are given in Table 40 for PWRs and BWRs. For some countries, the data are comprehensive and published regularly by the appropriate authorities. For other countries, data are not available for all years or all the units installed. None the less, the data are sufficient to give a reasonably comprehensive indication of the situation worldwide. As substantial numbers of LWRs enter the middle phase of their predicted operating lifetimes. In general, the data on electricity generated were taken from the summaries produced by the International Atomic Energy Agency (IAEA) [13, 14, 15, 16], if not otherwise given in the references for a particular country.

116. A comprehensive survey of data on LWRs in Western Europe has been carried out [B33]. The data cannot be added to Tables 40 or 41, as the reactors were not identified specifically by country. The normalized collective effective dose equivalent for 23 PWRs dropped from about 6 man Sv (GW a)\(^{-1}\) in 1980-1981 to 4 man Sv (GW a)\(^{-1}\) in 1984. The comparable figures for 17 BWRs were more variable but were in the range of 3-6 man Sv (GW a)\(^{-1}\). A particular study on PWRs has also been carried out by Lochard and Benedittini [LS]. A distinct difference is emerging between PWRs and BWRs in the annual collective dose per reactor and per unit electric energy generated for reactors of similar electrical capacity. This trend, which is illustrated in Table 41 for PWRs and BWRs from the United States and Japan, becomes more apparent as the reactors enter the second decade of their operating life and has been reported in the Federal Republic of Germany [E7], Japan [19, T12], Sweden [P7] and the United States [N3]. The collective western European data, however, do not support the conclusion [B33]. The collective dose can be higher in BWRs than in PWRs by up to a factor of 2, possibly because more maintenance work has to be performed in radiation areas on BWRs, especially around the turbines.

117. It appears that the significant trend towards increasing numbers of workers per reactor, which was noted in the UNSCEAR 1982 Report, especially in the United States, levelled off in the early 1980s. This aspect has been studied in detail by the Nuclear Regulatory Commission [N3], who showed that although the number of workers per reactor doubled from 600 to 1,200 over the period 1975-1980, it remained constant at the higher figure during the period 1980-1983. The collective effective dose equivalent distribution ratio (see paragraph 19) was assessed separately for PWRs and BWRs [N3]. For both types of reactors the average values of the ratios for the years 1981-1983 were in the range of 0.4-0.6. Annual average doses have been quoted for the years 1980-1982 at three PWRs in the USSR [V2, V3]. The values range from 4-8 mSv and provide detailed support for the overall average figure of 5.6 mSv given by Varobyov [V1] and used in Table 40. For the Novovoronezh PWR [V5], however, the normalized collective effective dose equivalent in 1980 of 3.1 man Sv (GW a)\(^{-1}\), based on an annual collective effective dose equivalent of 4.0 man Sv and an electric energy generated of 1.3 GW a [A1] was somewhat lower than the overall figure of 11 man Sv (GW a)\(^{-1}\) used in Table 40 [V1]. Data on doses to personnel at LWRs in the USSR has been reported at some reactors [P4, B15]. For the two reactor units at Kolskaya, the collective dose in 1980 of 2.3 man Sv was typical of earlier years; the normalized collective dose of 1.9 man Sv (GW a)\(^{-1}\) for 1980 was somewhat lower than the average value of 2.5 man Sv (GW a)\(^{-1}\) for 1977-1980. The annual average dose to personnel was about 5 mSv over that period. The data for Japan in Tables 40 and 41 were compiled mainly from detailed statistics supplied by Kumatori [K1] and Terasima [T12].

118. Recent data for HWRs in Canada are shown in Table 42 [A4]. The values include internal doses from
exposure to tritium. The electric energy generated was obtained from IAEA tabulations [13, 14, 15, 16]. The results for Canada show considerably higher collective effective dose equivalent per unit energy generated than would be obtained by considering only the two largest power plants that produce the bulk of the energy. The lifetime collective effective dose equivalent generated by the Atucha 1 HWR in Argentina has been estimated at 27 man Sv (GW a)\(^{-1}\) [P8].

119. Most of the GCRs in the world are in the United Kingdom. Comprehensive data on them [H1, H2, H3, H4, P5, W2] are shown in Table 42. The step change in the number of workers after 1980 is because the figures prior to 1981 did not include workers not directly employed by the Central Electricity Generating Board. The collective effective dose equivalent distribution ratio has been low: less than 0.01 in 1984 [P5]. The GCR in Japan is of a similar design to the early GCRs in the United Kingdom [K1]; that in the United States is a small high temperature gas cooled reactor [B27, N3]. so the doses from it are not directly comparable to the others.

120. Collective doses to personnel at the Dounreay establishment in the United Kingdom concerned with operation of the prototype FBR were 0.15 man Sv in 1984 and 0.29 man Sv in 1985 [U3]. Most of this collective dose resulted from charge machine refurbishing and irradiated fuel cell entries. Although many aspects of the design and operation of FBRs were reviewed at a recent symposium [111], no data on occupational exposures were reported. The Committee would welcome more information on this aspect, especially from prototype and nearly commercial-scale reactors.

121. Despite the lack of data from some countries and for some years, there is enough information from the countries for which each reactor type is installed in large numbers to make a reasonable estimate of the normalized occupation dose for the quinquennium 1980-1984 for the major reactors. These estimates, which are based on the data in Tables 40 and 42, are given in Table 43. Bearing in mind the world-wide predominance of LWRs, the overall estimate must be heavily weighted by the estimate for this reactor type, but a figure of 10 man Sv (GW a)\(^{-1}\) does not appear unreasonable.

D. SOLID WASTE DISPOSAL

1. Solid waste production

122. During operation of a power station, solid wastes are generated in a number of ways. In LWRs the main cause is treatment of the circulating water, giving rise to spent ion exchange resins, filter sludges and evaporator concentrates. Although these are originally wet wastes and may be stored in this form at the site, they are generally solidified before disposal. A similar type of waste arises from purification of the water in spent fuel storage ponds at reactors. Even though fuel elements are eventually removed for long-term storage or reprocessing, provision is made for short-term storage at reactors for initial decay heat removal. In addition, waste may include some structural components from the core or fuel, such as the outer fuel element structures from GCR and AGR fuel elements.

123. The radionuclides present in the above wastes are fission products, activation products and actinides, the particular radionuclides, quantities and relative activities being dependent on the reactor type, the state of the fuel cladding, the levels of corrosion, etc. In most cases these wastes will be the bulk of what is normally classified as intermediate-level wastes (ILW), i.e., wastes containing substantial activity concentrations but not significantly heat generating.

124. The other main cause of solid wastes during operation is the protective material of various kinds used around the station. Much of this is burnable and considerable volume reduction can be achieved by incineration and compaction. The radionuclide composition is even more variable than for the wet wastes and the activity concentrations are small to zero. These wastes are generally classified as low-level wastes (LLW).

125. In order to characterize the wastes for analysis of the impact of disposal, it is necessary to determine the volumes and the activity concentrations with identification of the relative quantities of important radionuclides, especially long-lived radionuclides and any actinides.

126. There were a number of studies in the mid-1970s on the quantities of wastes produced at LWRs [B30, M7, M8]. The results, summarized in Table 44, have been extracted from a review carried out by the Environmental Protection Agency of the United States [E5]. More recent reviews have been carried out in the United States by the Nuclear Regulatory Commission [N8, N9] and by the United States Department of Energy [D4], as well as in other countries using LWRs or planning to do so [N10]. On the basis of these studies, the quantities of conditioned wastes arising from LWRs per unit energy generated are assumed to be as shown in Table 45. These are only approximate; variations of up to an order of magnitude are possible in particular circumstances, depending on the type of treatment or conditioning used.

127. The assumed radionuclide compositions for the wastes in Table 45 are shown in Table 46. These are based primarily on the analyses reported in the United States [N8, N9, E6] and the United Kingdom [P10].

128. The quantities and activity concentrations of operating wastes of HWRs derived from data given for Canadian CANDU reactors [B31] are assumed to be as shown in Table 47. The quantities of wastes from operational GCRs have recently been reviewed [F3], and the results are also summarized in Table 47. The main differences in radionuclide composition from the LWR wastes are the higher alpha activity of the Magnox reactor sludges and the graphite debris containing \({}_{14}^{14}\)C. Although there are significant differences between reported inventories for Magnox
reactors and AGRs [P12] and LWRs, the composition in Table 46 is taken for this preliminary study, given the predominance of LWRs.

2. Solid waste disposal facilities

129. A large proportion of the LLW produced at all facilities during operation can be disposed of by burial at a shallow depth. Burial facilities range from simple trenches or pits containing untreated wastes and capped with soil, to concrete structures containing conditioned wastes and capped with weather-resistant materials. These will be referred to as trenches and engineered disposal facilities.

130. Considerable quantities of LLW have been disposed of in such facilities throughout the past few decades. Many of the earlier disposal sites were not used for disposal of wastes from the generation of nuclear power, except perhaps for some research and development aspects. For example, there are 14 sites in the United States operated by the United States Department of Energy for the disposal of wastes generated from certain defence research activities. Some major closed and currently operating LLW burial sites are shown in Table 48 [C8, C9, N9]. These have accepted wastes from a range of operations [H15, M10].

131. Typical simple trenches are about 10 m deep and 25 m wide and could be from 100 to 200 m in length, depending on the site. They are covered by about 1 m of compacted soil. The waste is not conditioned except to render the material non-combustible where necessary. This is similar to the minimum engineered trench specified by Pinner [P11] and the base case of the Nuclear Regulatory Commission of the United States [N9]. Some major routes by which radionuclides will be released from such a trench will be into rainwater percolating through the trench and into ground water. There will be considerable differences between the behaviour of elements that form easily soluble compounds, such as iodine, and those that do not, such as uranium, as well as a marked dependence on the environmental and hydrological conditions of the site.

132. On the basis of knowledge of the chemical behaviour of the elements of which there are important radionuclides, the release behaviour of the wastes can be classified into three groups [P11]. It is also assumed that, since LLW is usually disposed of in trenches without packaging, radionuclides begin to be released into water as soon as the site is closed. The reference site is assumed to be above the water table in reasonably permeable, weathered material that has an underlying less permeable rock. In a site with these characteristics, water filtering through the waste will tend to move down through the unsaturated zone until it reaches the water table and the impermeable boundary where it moves downslope. It is assumed to reach a stream at a distance of 2,000 m from the site.

133. Some categories of LLW containing radionuclides with longer half-lives or at activity concentrations too great for disposal in simple trenches have been disposed of in engineered shallow disposal facilities. A typical facility is an excavation about 20 m deep and 25 m wide lined with 1 m of concrete. Such facilities are filled with concreted wastes to about one half their depth, the interstices being filled with concrete and finished with layers of concrete and clay to form an impervious cap. The canisters and concrete around the wastes will prevent rain or ground-water access for a considerable time, which is taken to be 100 years. After this time it is assumed that all radionuclides are released into percolating water at a constant fractional rate of $10^{-2}$ a$^{-1}$.

134. As a result of the greater depth of emplacement, it is likely that engineered facilities would be positioned below the water table. It is also sensible to locate the facilities in materials with good sorption properties, so the reference site is assumed to be in clay. Many clay outcrops are associated with harder, more permeable rocks leading to artesian conditions, i.e., rising ground water. The trench would interfere with this locally so that the eventual flow pattern assumed for the reference site is that water infiltrating the trench from above will tend to move downward, then upwards and outwards, eventually entering streams at a distance of about 1,000 m on either side of the site [P10].

135. An alternative method of disposal for packaged solid wastes is to dump them into a sea-bed at considerable depth. Although such disposals were carried out for many years, they ceased in 1982 under a temporary moratorium. The amounts of wastes disposed of to this date in the north-east Atlantic have been summarized by NEA [N6] and are given in Table 49. It is not possible to assign the wastes to a particular power programme, and it is known that some of the major radionuclides, such as $^{40}$C, arose as wastes in the form for sea dumping largely from the preparation of radiopharmaceuticals.

3. Collective dose commitments

136. After closure of a burial facility, there will be a period during which control over the site is maintained. This does not necessarily preclude the transport of radionuclides released from the wastes into percolating ground water but could reasonably be relied upon to prevent major human intrusion into the site, such as for building purposes. Thus, during the controlled period, taken to be 100 years, only release by water contact is considered; other pathways are assessed after this period. The major pathways possibly leading to exposure are shown in Figure 11.

137. The actual transport of radionuclides with ground water, after release from the waste, the container and any surrounding engineered structures, will be very dependent on the hydrogeologic characteristics of the site. Considerable effort is being devoted to the development of calculational techniques capable of handling detailed knowledge of particular sites. For this study a more general approach is appropriate, such as the one adopted in other generic appraisals of shallow land burial [P10, N9, N10].
138. The simplest representation of ground water flow velocity, caused only by the natural hydraulic gradient, is that given by Darcy's law:

\[ V_a = K_p k_i / \varepsilon_k \]

where \( V_a \) is the ground water flow velocity, \( K_p \) is the hydraulic conductivity, \( k_i \) is the hydraulic gradient and \( \varepsilon_k \) is the kinetic porosity. This is the basis for several transport codes, such as FEFLOW, which is used in the Nordic study [N10], GEOS, which is used by the National Radiological Protection Board of the United Kingdom [H12], and that used by the Nuclear Regulatory Commission of the United States [N9]. It is adapted to radionuclide transport through a porous medium by including a retardation factor or distribution coefficient.

139. Generic assessments have also been carried out using a somewhat more realistic two-dimensional model, NAMMU [R4], to calculate the pressure head distributions and, hence, flowpaths and velocities in saturated porous media. This has been applied to migration through undisturbed clay and to movement in the surface soil layer [P10]. Whatever calculational method is used, the general result is that those nuclides with small retardation factors or distribution coefficients, such as tritium, \(^{14}\)C, \(^{99}\)Tc and \(^{129}\)I, move at a velocity close to that of the ground water, whereas nuclides with large retardation factors or distribution coefficients, such as \(^{235}\)U, \(^{239}\)U and \(^{239}\)Pu, move very slowly. The values for some important radionuclides are intermediate. Values adopted in three major studies [H12, N9, N10] are reasonably consistent.

140. The output from the radionuclide transport calculations is the rate of input of activity into either the nearest stream, as described for the generic site, or via ground water into soil that could be used for farming. Water could also be abstracted via a well. In calculating doses it is assumed that the water forms a source of drinking water for humans and animals. It is also assumed that fish from the stream are caught and eaten. The river model is a compartment type, with each compartment representing a homogeneous freshwater body and incorporating adsorption on to and resuspension of sediment particles [I15]. The flow rate of the river is taken to be \( 6 \times 10^6 \) m\(^3\) \( \text{day}^{-1} \). The eventual transfer from rivers via estuaries to the sea is also included. In assessing doses from drinking water, it is assumed that suspended sediments are removed by filtration. Collective doses from streams and wells are assessed on the assumption that 0.2% of the flow rate and 1% of the abstraction rate are actually ingested.

141. If the land is used for farming, this will give rise to a large number of exposure pathways. The contamination can result from transfer directly upwards through the soil from ground water or via streams and rivers through irrigation. The calculation of collective doses requires an estimate of the total quantities of each foodstuff consumed, shown in Table 50, together with average values for activity concentrations obtained from the radionuclide transport models.

142. The collective dose equivalent rates per unit activity as a function of time after release from an engineered facility via all the pathways are shown for a number of important radionuclides in Tables 51, 52
and 53, for three major time periods of interest. In general, farming and water consumption pathways both contribute significantly to the collective dose.

143. The results of applying the specified models are shown in Tables 54 and 55 for the shallow earth trench and the engineered trench, respectively. The results are presented per unit activity in the trench and show the collective effective dose equivalent commitment and the maximum collective effective dose equivalent rate. Also shown is the time at which the specified percentage of the maximum collective effective dose equivalent rate is reached \([S15]\).

144. Using the estimated volume for LLW from Table 45 of 200 m\(^3\) (GW a\(^{-1}\)) at an activity concentration of 1 GBq m\(^{-3}\) as appropriate for PWRs and assuming the radionuclide composition given in Table 46, it can be seen from Table 54 that the normalized collective effective dose equivalent commitment from burial of these relatively short-lived wastes is less than 10\(^{-10}\) man Sv (GW a\(^{-1}\)). Only if long-lived radionuclides were present could there be a collective dose of any significance; if this proportion were taken to be one thousandth of the quantity present in ILW, as shown in Table 46, the normalized collective effective dose equivalent commitment would be about 10\(^{-3}\) man Sv (GW a\(^{-1}\)).

145. Taking the estimated volume for ILW from Table 45 of 50 m\(^3\) (GW a\(^{-1}\)) at an activity concentration of 100 GBq m\(^{-3}\), again as appropriate for PWRs, and combining this with the data from Table 55 for the radionuclides specified to be present in Table 46, the normalized collective effective dose equivalent commitment from disposal of ILW in a model engineered trench is 0.5 man Sv (GW a\(^{-1}\)). The main contribution is from the 0.1% by activity of \(^{14}C\) assumed to be present in the waste. The contribution by radionuclide is shown in Table 56.

### IV. FUEL REPROCESSING

146. At the fuel reprocessing stage of the nuclear fuel cycle, the elements uranium and plutonium in the irradiated nuclear fuel are recovered to be used again in fission reactors. Spent fuel elements are stored under water, which provides both biological shielding and cooling, while waiting to be reprocessed. Fuel elements are usually left until the short-lived \(^{131}\)I has decayed to a low level, normally a minimum of four or five months. Since one reprocessing plant can serve large numbers of nuclear reactors, the quantities of nuclides passing through the plant that are significant from the point of view of health will be high in absolute terms. Careful design limits discharges, however, so that releases per unit of electricity generated by the fuel passing through the plant, i.e., (GW a\(^{-1}\)), may be relatively small.

147. The only commercial operating reprocessing plants are at Sellafield (formerly Windscale) in the United Kingdom and Cap de la Hague and Marcoule in France. The capacity at Sellafield is 2,000 t a\(^{-1}\) (heavy metal) and that of Cap de la Hague is 900 t a\(^{-1}\) oxide fuel, while the plant at Marcoule processes up to 400 t a\(^{-1}\) of GCR metal fuel. The annual throughput of irradiated fuel from civilian power programmes in these three reprocessing plants is currently equivalent to about 8 GW a of electric energy, representing about 5% of the reported annual nuclear electric energy production (189 GW a, Table 2). Thus, the majority of irradiated fuel, which arises from LWRs, is not reprocessed but is stored pending future policy decisions as to whether to dispose or reprocess. A summary of the attitudes of countries with power reactors towards reprocessing is given in Table 57, while in Table 58 national programmes for commercial reprocessing are given \([C3]\).

#### A. EFFLUENTS

148. The design and operation of reprocessing plants to avoid releases of large amounts of radioactive material is complex. The gaseous and volatile fission product elements (iodine, tritium, carbon, krypton, ruthenium, technetium, xenon and caesium) are largely separated from the fuel when it is dissolved in nitric acid. The dissolver off-gas is treated for nitric acid recovery and iodine removal before being mixed with the off-gases from other stages in the process. The vessel off-gases are treated by caustic scrubbing, drying and filtering through high efficiency filtration systems before being discharged to the atmosphere. The aqueous wastes containing almost all the fission products and transuranic elements are concentrated by evaporation and stored in double containment stainless steel tanks before they are treated further.

149. The radionuclides of principal concern in reprocessing effluents are the long-lived nuclides: \(^{40}H\), \(^{14}C\), \(^{85}Kr\), \(^{131}I\), \(^{134}Cs\), \(^{137}Cs\) and isotopes of transuranium elements. Table 59 lists the reported discharges to the atmosphere, and Table 60 those in aquatic releases, from Sellafield, Cap de la Hague and Marcoule for 1980-1985. The amount of activity in the effluents depends upon the specific waste treatment and processing design of the plant, as well as the type of fuel processed, its irradiation history and storage time prior to reprocessing. Table 61 gives the isotopic composition of liquid effluent discharges from the Sellafield and Cap de la Hague plants in greater detail for the years 1980-1985. Atmospheric release data and liquid discharge data for Marcoule are not available beyond 1980.

150. The throughput of fuel at both Sellafield and Cap de la Hague has been calculated on the basis of \(^{85}Kr\) reported discharges and on the assessment of \(^{85}Kr\) generation in different reactor fuel cycles made in the UNSCEAR 1982 Report. In that Report, the Committee has used production rates of 14 PBq (GW a\(^{-1}\)) for GCRs and 11.5 PBq (GW a\(^{-1}\)) for PWRs. These figures make assumptions about fuel burn-up and reactor thermal efficiency that are not likely to have changed significantly since the UNSCEAR 1982 Report. On this basis, the electric energy production of the annual throughput of fuel at Sellafield has varied between 1.7 GW a and 3.7 GW a, while for
Cap de la Hague the range has been 2.4 to 6.1 GW a. For Marcoule, there are little data on atmospheric discharges, although the electric energy of annual fuel throughput for 1980 has been estimated at 1.4 GW a. Previously reported, it seems likely that all the $^{129}$I in the fuel is released with a few per cent going to the atmosphere.

151. For tritium, the Committee used in the UNSCEAR 1982 Report a production rate in LWR fuel of 0.75 PBq (GW a)$^{-1}$ and, assuming this applies to GCR fuel, the inventory passing through Sellafield has varied between 1.3 PBq (1985) and 2.8 PBq (1981). In 1981, atmospheric discharges of $^3$H were 0.46 PBq and liquid discharges 2 PBq, giving 2.46 PBq, compared with the estimated throughput of 2.8 PBq. Thus, it appears that nearly all the tritium in irradiated fuel is released in reprocessing and about 20% is released to the atmosphere. This is identical to the percentage estimated in the UNSCEAR 1982 Report. The remaining tritium may be immobilized in cladding wastes. For Cap de la Hague the normalized releases of tritium have been 0.26 PBq (GW a)$^{-1}$ of which only about 1% is in reported atmospheric releases and thus it seems that only about one third of the throughput is released.

152. The results of routine measurements of $^{14}$C atmospheric discharges from the Sellafield reprocessing plant are given in Table 59. The normalized production rate of $^{14}$C in GCR fuel was estimated in the UNSCEAR 1982 Report at 3.2 TBq (GW a)$^{-1}$. Atmospheric discharges from Sellafield therefore seem to account for essentially the whole of the estimated throughput of $^{14}$C between 1980 and 1985. For the French reprocessing plants, $^{14}$C discharges are not reported. In the UNSCEAR 1982 Report, the Committee estimated the $^{14}$C content of LWR fuel to be 0.66 TBq (GW a)$^{-1}$, of which about 75% was assumed emitted to the atmosphere, but in view of the Sellafield data, all $^{14}$C can be considered released to the atmosphere for the dose assessment.

153. The $^{131}$I content of irradiated nuclear fuel varies, depending upon the cooling time and the final power level of the fuel discharge. The $^{131}$I normalized content of LWR fuel cooled for six months is estimated at 2.8 TBq (GW a)$^{-1}$, falling by a factor of 2,000 for a cooling period of nine months. Since irradiated fuel is generally cooled for at least a year prior to reprocessing, $^{131}$I discharges are very small. For 1980-1985, Sellafield atmospheric releases of $^{131}$I (Table 59) gave normalized values of 19 GBq (GW a)$^{-1}$, the corresponding figure for 1975-1979 was 1.7 GBq (GW a)$^{-1}$, the increase being due to a high release figure in 1981.

154. The quantity of $^{129}$I in fuel depends upon burn-up and is assessed at 37-74 GBq (GW a)$^{-1}$. Atmospheric discharges of $^{129}$I, as well as liquid effluent amounts, have been reported for Sellafield and Cap de la Hague but not for Marcoule. The normalized atmospheric release is 3.7 GBq (GW a)$^{-1}$ for Sellafield and 4.9 GBq (GW a)$^{-1}$ for Cap de la Hague for the period 1980-1985, which is about twice the value given in the UNSCEAR 1982 Report. Liquid effluents averaged about 30 to 60 GBq (GW a)$^{-1}$ at each plant over the same period, compared with 40 GBq (GW a)$^{-1}$.

155. Atmospheric releases of aerosols are summarized in Table 59. The normalized alpha releases from Sellafield are 0.2 GBq (GW a)$^{-1}$, of which more than 75% is due to plutonium isotopes [B1, B2, B3, B8, B16], the remainder being accounted for by $^{241}$Am and $^{242}$Cm. This figure is one half that reported in the UNSCEAR 1982 Report. The alpha-aerosol results are available from France for Cap de la Hague and are some 30 times lower. For atmospheric beta releases, the largest component from Sellafield is $^{137}$Cs, although since 1981 the levels have been reduced. The normalized release is 63 GBq (GW a)$^{-1}$, compared with 88 GBq (GW a)$^{-1}$ for 1975-1979. For Cap de la Hague the normalized release is 0.04 GBq (GW a)$^{-1}$ for beta-aerosols, and no isotopic breakdown is available. The reduction in aerosol releases in recent years from Sellafield is the result of improvements in the Magnox cladding silo stores, including the installation of inert gas blankets and filtration systems.

156. The liquid effluents discharged from Sellafield, Cap de la Hague and Marcoule are given in Table 60 for total alpha, total beta, $^3$H, $^{90}$Sr, $^{106}$Ru and $^{137}$Cs. There is a yearly isotopic breakdown for the French plant at Cap de la Hague but not for Marcoule. The isotopic compositions of Sellafield and Cap de la Hague discharges are given in Table 61 for 1980-1985.

157. The normalized alpha release from Sellafield to the sea is 8.0 ± 5.2 TBq (GW a)$^{-1}$, compared with an average of 25 TBq (GW a)$^{-1}$ between 1975 and 1979. For Marcoule and Cap de la Hague the figures are 0.063 and 0.16 TBq (GW a)$^{-1}$, while for the previous period they were 0.016 and 0.24 TBq (GW a)$^{-1}$. Most of the Sellafield alpha activity was $^{39}$, $^{89}$Pu, and the level of alpha discharge has been reduced by a factor of 6 over the reporting period.

158. For liquid discharges of beta activity the normalized releases from Sellafield, Cap de la Hague and Marcoule are 0.97, 0.24 and 0.027 PBq (GW a)$^{-1}$, respectively, compared with 3.7, 0.52 and 0.04 PBq (GW a)$^{-1}$ for 1975-1979. The isotopic composition of the effluents varies between the sites: 55-70% of the Sellafield discharge is attributable to $^{137}$Cs, whereas 40% of the Cap de la Hague discharge is attributable to $^{106}$Ru. The $^{137}$Cs levels from Sellafield were reduced by a factor of 9 over the review period, although the $^{106}$Ru levels remained constant at about 400 TBq (GW a)$^{-1}$ until 1985. After $^3$H, $^{106}$Ru is the main isotope released from Cap de la Hague; the $^{106}$Ru discharges are comparable to those of the Sellafield plant.

159. Monitoring of the marine environment is undertaken by regulatory authorities to ensure compliance with authorized discharges and to ensure that doses to exposed populations are at the levels predicted. The results of monitoring around the United Kingdom in the vicinity of all operating nuclear plants have been published by Hunt [H5, H6, H7]. The most significant
results arise from discharges of the Sellafield plant. Measurements of activity in fish and shellfish in 1983 are shown in Tables 62 and 63 for various locations around the United Kingdom. In order to interpret these results, consumption data are required to assess intakes of radionuclides.

160. Aarkrog [A2] has summarized bio-indicator studies in Nordic waters to identify levels of radioactive contamination. The marine bio-indicators are the blue mussel (Mytilus edulis) and bladder wrack (Fucus vesiculosus), which are sensitive to contamination from nuclear fallout and from Sellafield discharges and nuclear power plants in Sweden, Finland and the rest of coastal northern Europe. Discharges from Sellafield have been traced from the Irish Sea, along the western Norwegian coast, down along eastern Greenland and then western Greenland. The transit time from the Irish Sea is measured to be four years and the activity concentration is diluted by a factor of 100.

161. The measured concentrations of $^{137}$Cs in sea water decrease from the highest levels of 1,000 Bq kg$^{-1}$ near Sellafield to 8-10 Bq kg$^{-1}$ in the Baltic Sea, 1-2 Bq kg$^{-1}$ near Greenland and less than 1 Bq kg$^{-1}$ near Iceland. Levels of $^{99m}$Tc from Sellafield discharges closely follow those of $^{137}$Cs. Measurements of plutonium show enhanced levels primarily in British and Irish coastal waters, although very low levels have recently been detected in areas further from the coast.

B. LOCAL AND REGIONAL COLLECTIVE DOSE COMMITMENTS

162. The evaluation of the collective dose commitments from reprocessing nuclear fuel requires a study of the local and regional effects and of the global consequences of the releases. Estimates of the local and regional collective dose commitments are given in this section and the global contribution is provided in chapter V. The collective dose commitments are evaluated for the normalized discharges from Sellafield and Cap de la Hague by scaling the normalized results given in the UNSCEAR 1982 Report. As there are only three reprocessing plants operating with significant commercial throughput of fuel, the collective effective dose equivalent commitments per unit of electric energy generated are weighted by the fraction of fuel reprocessed to provide the current contribution from all operating reactors. In the UNSCEAR 1982 Report, the Committee gave typical discharge figures for notional new designs of reprocessing plant. This has not been repeated in this Report since all fuel may not be reprocessed. The weighted average, therefore, reflects actual exposures from the nuclear fuel cycle as currently operated.

1. Krypton-85

163. The averaged $^{85m}$Kr normalized discharge from Sellafield between 1980 and 1985 was 14 PBq (GW a)$^{-1}$ (Table 59), and the collective effective dose equivalent commitment obtained by the Committee in the UNSCEAR 1982 Report was 0.0074 man Sv (PBq)$^{-1}$. Thus, the normalized local and regional collective effective dose equivalent commitment is 0.1 man Sv (GW a)$^{-1}$. The normalized discharge from Cap de la Hague is 11 PBq (GW a)$^{-1}$, giving 0.08 man Sv (GW a)$^{-1}$. The average annual electric energy generated in recent years has been over 160 GW a, and an annual amount of fuel equivalent to 8 GW a was reprocessed. Thus, the normalized collective effective dose equivalent commitment from $^{85m}$Kr is 0.005 man Sv (GW a)$^{-1}$ electric energy generated.

2. Tritium and carbon-14

164. The Committee used in the UNSCEAR 1982 Report specific activity models to estimate collective doses from $^1$H and $^{14}$C discharges. The dose resulting from the release of tritium to the atmosphere was estimated in that Report at 0.0027 man Sv TBq$^{-1}$ for the Sellafield site. Some four times lower than the value for the reactor site, due to differences in site population density and meteorological conditions. Releases from Sellafield to the atmosphere averaged 120 TBq (GW a)$^{-1}$ (Table 59), giving a collective effective dose equivalent commitment of 0.32 man Sv (GW a)$^{-1}$, 85% of which was from ingestion. For Cap de la Hague the normalized release of 3.5 TBq (GW a)$^{-1}$ gives a collective effective dose equivalent commitment of 0.01 man Sv. Releases to the regional marine environment were estimated in 1982 to lead to lower dose commitments. Using the value derived in the UNSCEAR 1982 Report of 1.8 $10^{-3}$ man Sv PBq$^{-1}$ released to oceans and the average release to the sea of 579 TBq (GW a)$^{-1}$ for 1980-1985 from Sellafield (Table 60) leads to a collective dose commitment of 1 $10^{-3}$ man Sv (GW a)$^{-1}$. The total normalized collective effective dose equivalent commitment weighted by the relative energy of fuel reprocessed at Sellafield and Cap de la Hague is 0.15 man Sv (GW a)$^{-1}$, essentially the same figure that was given in the UNSCEAR 1982 Report. Again, allowing for the fraction of fuel reprocessed, the weighted normalized collective effective dose equivalent commitment is 0.007 man Sv (GW a)$^{-1}$.

165. For $^{14}$C releases to the atmosphere, the Committee estimated the collective effective dose equivalent commitment at 0.4 man Sv TBq$^{-1}$, with essentially the same value per TBq released to the aquatic environment. Averaged atmospheric releases from Sellafield are reported to be 3.5 TBq (GW a)$^{-1}$, giving a normalized collective effective dose equivalent commitment of 1.4 man Sv (GW a)$^{-1}$, compared with 0.69 man Sv (GW a)$^{-1}$ quoted in 1982. The difference is accounted for by the reported discharges to the atmosphere being double the values reported in 1975-1979. It would appear that the total throughput of $^{14}$C at Sellafield is now accounted for in atmospheric releases. For Cap de la Hague, the assumed release of 0.66 TBq (GW a)$^{-1}$ gives a normalized collective effective dose equivalent commitment of 0.3 man Sv (GW a)$^{-1}$. Weighted for the fraction of fuel reprocessed, the contribution is 0.04 man Sv (GW a)$^{-1}$.
3. Other atmospheric releases

166. Of the other nuclides released to the atmosphere, ¹³⁷Cs becomes globally dispersed and makes a contribution to the collective dose commitment over a prolonged period, while the remainder contribute only to the local and regional collective dose commitment. A summary is given in Table 64. The total, averaging over Sellafield and Cap de la Hague, amounts to 1.3 man Sv (GW a)⁻¹, compared with the assessment of 3 man Sv (GW a)⁻¹ for atmospheric releases made in the UNSCEAR 1982 Report. Some 65% of the dose is now due to ¹⁴C discharges, which are reported to be twice the previous levels and which counteract reductions in discharges of other nuclides, particularly actinides. Weighted by the proportion of electric energy generated, the normalized collective effective dose equivalent commitment from atmospheric releases during reprocessing is 0.07 man Sv (GW a)⁻¹.

4. Liquid effluents

167. The results are presented in Table 65 for the normalized collective effective dose equivalent commitments for marine discharges from reprocessing at Sellafield and Cap de la Hague. The environmental dosimetric models are appropriate for the specific coastal waters of northern Europe and were fully described in the UNSCEAR 1982 Report. They assume that consumed fish, molluscs and crustacea are the important food pathways to man.

168. For Sellafield, normalized liquid discharges for 1980-1985 were reduced by a factor of 3 since the period 1975-1979. The collective effective dose equivalent commitment per TBq for marine discharges from Sellafield found by the Committee in the UNSCEAR 1982 Report was 0.068 man Sv for ¹³⁷Cs, 0.034 man Sv for ¹⁰⁹Ru and 0.025 man Sv for alpha-emitters. The principal route of exposure is ¹³⁷Cs in consumed fish, as before, and for 1980-1985 the caesium contribution is some 85% of the total collective dose. The normalized collective effective dose equivalent commitment for 1980-1985 from liquid discharge from Sellafield is 44 man Sv (GW a)⁻¹, the estimate made in the UNSCEAR 1982 Report being 124 man Sv (GW a)⁻¹. If data for 1985 alone are taken, the normalized release gives a collective effective dose equivalent commitment of 25 man Sv (GW a)⁻¹, reflecting the lower discharges after the installation of a new plant to remove radioactive substances from effluent streams.

169. In the case of Cap de la Hague, discharges also seem to have been reduced. The Committee’s models used in the UNSCEAR 1982 Report gave dose conversion factors per TBq released from Cap de la Hague of 0.1 man Sv for ¹⁰⁹Ru, 0.09 man Sv for ¹³⁷Cs and 0.4 man Sv for alpha-emitters. The normalized collective effective dose equivalent commitment shown in Table 65 is 11 man Sv (GW a)⁻¹, which compares with the figure of 53 man Sv (GW a)⁻¹ given in the UNSCEAR 1982 Report. The majority of the dose arises from the discharge of ¹⁰⁹Ru.

170. The collective effective dose equivalent commitment weighted for the relative amount of electricity produced by the fuel reprocessed at each plant is thus 25 man Sv (GW a)⁻¹ and, after allowing for the proportion of fuel reprocessed commercially, the normalized contribution is 1.2 man Sv (GW a)⁻¹. Annual committed effective dose equivalents to the critical group of wrinkle eaters close to the Sellafield site were reported to be 0.5 mSv in 1985 [B29]. The doses are reduced as discharge levels fall.

C. OCCUPATIONAL EXPOSURES

171. It was noted in the UNSCEAR 1982 Report that experience of fuel reprocessing is limited to a few countries and that plant design and historical operating conditions may not represent the best current potential for new plants. This view is supported by a recent review of the trends in the annual collective and the maximum individual occupational doses in a number of reprocessing plants [B22]. The review covered not only the large operating reprocessing plants at Cap de la Hague and Sellafield, but also the pilot plants WAK at Karlsruhe, Federal Republic of Germany, the Eurochemie plant at Mol, Belgium, the PNC plant at Tokai Mura in Japan, and the Idaho and Savannah River plants in the United States. Although recognizing the differences in sizes and design age of the various plants and that some of them reprocess fuel for military as well as civilian purposes, a downward trend in average doses was observed starting during the period 1971-1973 and ending during the period 1980-1982. The annual average effective dose equivalent dropped from 4.15 mSv in the early 1970s to 2-4 mSv early in the 1980s. Data since the UNSCEAR 1982 Report are summarized in Table 66 for Japan and the United Kingdom [A5, B12, B23, B28, H8]. The data for Japan refer only to the PNC plant at Tokai Mura. An estimate of 0.5 man Sv has been made of the neutron collective dose equivalent at Sellafield in 1982 [B24]. Data for Cap de la Hague and Marcoule from 1973 to 1985 are given in Table 67, taken mainly from the recent comprehensive review by Henry [H13]. This also shows annual average effective dose equivalents of about 2 mSv in the period 1982-1985 at both establishments.

172. In the UNSCEAR 1982 Report the normalized collective effective dose equivalent commitments for the plants at Windscale (now Sellafield), United Kingdom, and Cap de la Hague, France, were estimated to be 18 and 6 man Sv (GW a)⁻¹, respectively. Some revised estimates for the United Kingdom are given in Table 66, based on ⁴¹Kr discharges related to energy throughput and a fuel content of 14 PBq (GW a)⁻¹. The normalized value for Cap de la Hague is reported to have fallen from 6 man Sv (GW a)⁻¹ in 1975 to 1 man Sv (GW a)⁻¹ in 1985 despite a large increase in reprocessed fuel throughput over this period [B22]. This is in agreement with the data given in the report of a working group [C6] for a period leading up to 1981 and supplemented in a report to the Sizewell B public inquiry in the United Kingdom [Z1] with data for 1982 and 1983. These estimates are in agreement with the detailed results for Cap de la Hague reported by Henry [H13] and given
in Table 67. The Table shows that the normalized collective effective dose equivalent dropped steadily from 2.2 to 0.9 man Sv (GW a)\(^{-1}\) throughout the period 1980-1985. The difference of nearly an order of magnitude between the normalized values for the two major installations makes it difficult to make a clear estimate. It seems, however, that the estimate in the UNSCEAR 1977 Report of the global collective effective dose equivalent per unit electric energy generated, is at 10 man Sv (GW a)\(^{-1}\), too high. Based on the trends reported for Cap de la Hague, the estimate for Marcoule in 1980, the experience in Japan, and taking into account the predictions for the new plant at Sellafield, a better estimate for the whole of the 1980s is about 5 man Sv (GW a)\(^{-1}\). When allowance is made for the proportion of fuel reprocessed commercially, the normalized contribution from occupational exposure is 0.25 man Sv (GW a)\(^{-1}\).

D. SOLID WASTE DISPOSAL

173. The solid wastes that are generated in the handling, processing and disposal of spent fuels are of two broad categories. Most of the activity in the spent fuel is separated during reprocessing and, after a period of storage as a liquid, will be solidified for eventual disposal as high-level waste (HLW), generating significant decay heat. During the reprocessing operations considerable amounts of solid low-level wastes (LLW) and solid intermediate-level wastes (ILW) are produced, some streams of the latter being characterized by an appreciable content of actinides. If the spent fuel is not reprocessed but stored and prepared for disposal, there will be almost no ILW or LLW. But the packaged spent fuel is then treated as HLW; it contains the actinides that would have been separated for re-use by the reprocessing operation. Since neither spent fuel nor vitrified HLW have been disposed of, they are not considered in this assessment of current operations.

174. Production of other solid wastes in reprocessing plants has been highly dependent on the operational characteristics of the particular plant. In particular, much of the waste produced at the Sellafield plant in the United Kingdom is attributable to the degradation of Magnox fuel in underwater storage and should not be taken to be indicative for other plants now or in the future.

175. Production of ILW from the British Magnox reprocessing programme has been estimated [T11] at 47,000 m\(^{3}\) from the reprocessing of 30,000 t uranium metal. The activity is estimated to be about 2 \(10^{10}\) Bq alpha and 2 \(10^{14}\) Bq beta/gamma activity at 1990. Taking an average fuel requirement for Magnox reactors of 200 t (GW a)\(^{-1}\), these correspond to the quantities shown in Table 68. Comparing with the alpha inventory of the fuel throughput, calculated to be 6,700 TBq (GW a)\(^{-1}\) at six months and 3,800 TBq (GW a)\(^{-1}\) at 20 years [G5], the fraction of alpha activity throughput lost to the ILW is about 0.02-0.03. Similarly, taking the beta/gamma inventory of the fuel throughput to be 2.4 \(10^{10}\) TBq (GW a)\(^{-1}\) at six months and 1.7 \(10^{10}\) TBq (GW a)\(^{-1}\) at 20 years [G5], the fraction of beta/gamma throughput lost to the ILW is about 0.005-0.05. These will be very sensitive to reprocessing chemical conditions for some nuclides, especially \(^{239}\)Np. The generation of ILW from the proposed oxide fuel reprocessing plant for AGR fuel was also estimated by Taylor [T11] to be 11,000 m\(^{3}\) from the reprocessing of 600 t uranium metal in the thermal oxide reprocessing plant (THORP). The activity content is 6 \(10^{13}\) Bq alpha and 5 \(10^{15}\) Bq beta/gamma activity, assuming a cooling period of five years. The average fuel requirement of the AGR is taken to be 30 t (GW a)\(^{-1}\) to give the quantities in Table 68. Again, comparing with the alpha and beta/gamma inventories of the fuel throughput at five years, calculated to be 3,100 and 2.5 \(10^{10}\) TBq (GW a)\(^{-1}\), respectively [G5], the fractions of activity throughput lost to the ILW are 0.01 for alpha and 0.1 for beta/gamma activities.

176. The annual rate of waste generation at Marcoule has been reported [B32] to be about 2,000-3,000 drums containing a total of about 4 TBq alpha and 4,000 TBq beta/gamma activity. Assuming the annual fuel throughput of the facility to be 0.8 GW a and the drum capacity to be 0.2 m\(^{3}\) gives the quantities shown in Table 68.

177. An alternative method of estimating the activity content of other solid wastes is to assess it directly as a fraction of the throughput of radionuclides in the fuel. This approach has been used by the United States Department of Energy [D4] to give the results in Table 69. The quantities are comparable with those estimated by Hill et al. [H11] but considerably less than those estimated for an operating plant, as shown in Table 68. The difference is about two orders of magnitude for alpha emitters and nearer to three orders of magnitude for beta/gamma emitters.

178. To give some estimate of the consequences of disposal of such wastes, it is assumed that the alpha wastes are entirely \(^{241}\)Pu and the beta/gamma wastes entirely \(^{137}\)Cs, and a typical normalized production from Table 68 is taken to be 100 TBq (GW a)\(^{-1}\) for alpha wastes and 10,000 TBq (GW a)\(^{-1}\) for beta/gamma wastes. Using the model for a typical ILW engineered disposal trench from section III.D. and the values for collective effective dose equivalent commitment per unit activity disposed of from Table 55, the normalized collective effective dose equivalent commitment would be 1 man Sv (GW a)\(^{-1}\). This is reduced to 0.05 man Sv (GW a)\(^{-1}\) when account is taken of the proportion of fuel reprocessed commercially. Lower losses from throughput to the ILW and LLW waste streams as estimated in paragraph 176 would significantly reduce this estimate; greater losses of the very long-lived radionuclides \(^{14}\)C and \(^{129}\)I would significantly increase it.

V. COLLECTIVE DOSE COMMITMENTS FROM GLOBALLY DISPERSED RADIONUCLIDES

179. The nuclides giving rise to a global collective dose commitment are sufficiently long-lived and migrate
through the environment, thus achieving widespread distribution. Those of interest are $^3$H, $^{14}$C, $^{85}$Kr and $^{129}$I. The environmental transfer of $^3$H, $^{14}$C and $^{85}$Kr is becoming fairly well established, and reliable estimates of collective dose commitments were made by the Committee in the UNSCEAR 1982 Report. Other long-lived nuclides, such as $^{239}$Pu, are far less mobile in the environment and therefore become less dispersed after deposition on to soils or sediments, following release into the local region.

180. The very long-lived nuclides, such as $^{129}$I, pose a special problem because of the uncertainty in predicting population size, dietary habits and environmental pathways over periods of tens of millions of years. Therefore, little use can be made of these collective dose commitments for decision-making purposes. The incomplete collective dose commitment, however, is useful to demonstrate the time distribution of the dose commitment and to estimate the per caput doses arising per year from a finite duration of a practice. In the following paragraphs, complete and incomplete dose commitments are given for the globally dispersed nuclides up to a maximum of $10^8$ a. The collective dose commitments per unit release were taken from the UNSCEAR 1982 Report of the Committee and scaled for the normalized releases derived for 1980-1984 discharges.

181. In the UNSCEAR 1982 Report, a model reprocessing facility was described and all reactor fuel was assumed to be reprocessed. In this Report, the collective doses assessed for reprocessing plant reported discharges are weighted by the fraction of the energy value of the total nuclear fuel that is reprocessed, namely 5% (paragraph 147). The weighted contribution is added to any contribution from reactor operation to reflect the current normalized exposures.

**A. KRYPTON-85**

182. Since krypton is an inert gas, it disperses throughout the atmosphere and achieves a uniform concentration in about two years. The Committee, in the UNSCEAR 1982 Report, estimated the collective effective dose equivalent commitment from $^{85}$Kr to be 0.17 man Sv PBq$^{-1}$, assuming a world population of 4 $10^9$. This must be scaled up to 0.2 man Sv PBq$^{-1}$ for the world population of 4.6 $10^9$ during the period 1980-1985. All the dose commitment is delivered within the first 50 years after release. Paragraph 150 gave normalized production of $^{85}$Kr as 11.5 PBq (GW a)$^{-1}$ for LWRs and 14 PBq (GW a)$^{-1}$ for GCRs, leading to 2.3 man Sv (GW a)$^{-1}$ and 2.8 man Sv (GW a)$^{-1}$ collective effective dose equivalent commitment, respectively. Contributions to the collective effective dose equivalent commitment come almost equally from whole-body gamma-radiation and from beta-irradiation of the skin. Weighting this collective dose by the fraction of fuel currently reprocessed (0.05) leads to 0.12 man Sv (GW a)$^{-1}$. The incomplete collective dose commitments are shown in Table 70, which indicates that half of the dose from $^{85}$Kr is delivered in the first 10 years after discharge.

**B. TRITIUM**

183. The models used by the Committee in the UNSCEAR 1982 Report gave a collective effective dose equivalent commitment of 2.8 $10^{-5}$ man Sv per TBq released. Because of the short half-life of tritium, this applies to the world population at the time of release. For the 1980-1985 world population of 4.6 $10^9$, the dose factor is increased to 3.2 $10^{-5}$ man Sv per TBq. Releases to the atmosphere and hydrosphere were not distinguished, since the exchange of water between the atmosphere and circulating waters of the globe is rapid, and the models assume immediate mixing and exchange with the hydrogen content of the circulating water.

184. The normalized release of tritium to the atmosphere from reactor operations, weighted by electricity production, is 46 TBq (GW a)$^{-1}$, while for liquid discharges the data give 40 TBq (GW a)$^{-1}$. Averaged over Sellafield and Cap de la Hague, aquatic and atmospheric releases from reprocessing add up to about 600 TBq (GW a)$^{-1}$, and since only 5% of the fuel is reprocessed, this adds 30 TBq (GW a)$^{-1}$ to the reactor releases of 86 TBq (GW a)$^{-1}$. The total collective effective dose equivalent commitment amounts to 0.004 man Sv (GW a)$^{-1}$. The incomplete collective dose commitments shown in Table 70 indicate that essentially all of the dose is received in the first few years after discharge. The local and regional contribution from tritium releases from reactor operation and the fractional reprocessing contribution amount to about 0.6 man Sv (GW a)$^{-1}$, which is a factor of over 100 greater than the global contribution.

**C. CARBON-14**

185. The Committee used in the UNSCEAR 1982 Report a relatively complex compartment model to assess the environmental distribution and behaviour of $^{14}$C. This model allows for two hemispheres, each comprising humus, circulating carbon, surface ocean and deep ocean. The circulating carbon represents the carbon in the troposphere and those sectors of the terrestrial biosphere subject to rapid growth and decomposition. Humus represents the carbon content of the terrestrial biosphere which circulates more slowly. Carbon-14 releases are assumed to be instantaneously mixed in the compartment to which release occurs. The results produced by this model are similar to those produced by more complex models; the main area of uncertainty is the rate of transfer of $^{14}$C to the deep ocean, from where it is less available.

186. The resulting collective effective dose equivalent commitment is 67 man Sv TBq$^{-1}$ released, averaged over both aquatic and atmospheric releases and assuming a future global population of $10^{10}$. Normalized releases from reprocessing plants are averaged over the reported figures for Sellafield (Table 59) and calculated throughput for Cap de la Hague (paragraph 152). Measurements appear to show that all the throughput is measured in airborne effluents, and it is assumed that little is discharged to the sea. The normalized release is 3.5 TBq (GW a)$^{-1}$ from Sellafield.
(Table 59); Cap de la Hague is assumed to give rise to releases of 0.66 TBq (GW a)^{-1}. The collective effective dose equivalent commitment for Sellafield is thus 234 man Sv (GW a)^{-1} and 44 man Sv (GW a)^{-1} for Cap de la Hague. Since about 5% of the annual energy equivalent of fuel is reprocessed, the weighted figure averaged over the two sites is 6 man Sv (GW a)^{-1}, the remaining fuel being stored and not giving rise to effluent releases of ^{14}C.

187. HWR releases are about 7.3 TBq (GW a)^{-1} of ^{14}C from reactor operations (Table 25), while those from LWGRs and GCRs are about 1.1 TBq (GW a)^{-1} (paragraph 72). LWR releases at about 0.3 TBq (GW a)^{-1} (paragraph 70) are small in comparison. The normalized collective effective dose equivalent commitment from HWR operation is therefore 490 man Sv (GW a)^{-1}. This is nearly a factor of 3 lower than the estimate given in the UNSCEAR 1982 Report, and is entirely due to lower reported discharge figures. About 6% of total nuclear generated electric energy arises from HWRs and about 10% from LWGRs and GCRs, so the electricity production weighted contribution to collective dose is 32 man Sv (GW a)^{-1} from HWRs and an additional 7.7 man Sv (GW a)^{-1} from GCRs and LWGRs. Although LWR releases are lower, because of their larger electric production, LWGRs add 17 man Sv (GW a)^{-1}. In summary, the present practices of reactor operation and reprocessing lead to a total collective effective dose equivalent commitment of 6 man Sv (reprocessing) plus 57 man Sv (from HWR, LWR, LWGR and GCR operation), i.e., 63 man Sv (GW a)^{-1}. This commitment is received over some 10,000 years, while the temporal distribution is shown in Table 70 to be 3% in 10 years, 10% in 100 years and 19% in 1,000 years.

D. IODINE-129

188. When released to the atmosphere, iodine, because of its environmental mobility, becomes rapidly incorporated into foodstuffs ingested by individuals. The highest concentrations of iodine occur in sea water and, as with ^{14}C, the greatest uncertainties surround the transfer of ^{129}I to deep oceans and any sedimentation that may remove activity from any biological chain.

189. Assuming again a future global population of 10^{10}, the Committee used a collective effective dose equivalent commitment of 1.4 10^4 man Sv TBq^{-1} released [U1]; of this, some 0.003% is delivered within 100 years of release, 0.03% in 10,000 years, 5% in 10^4 years, thus leaving 95% of the collective dose to be delivered from 1 million years after release, most of it coming between 10 million and 40 million years. For this report incomplete dose commitments to 10^8 years are used so that the value of ^{129}I is 700 man Sv TBq^{-1}.

190. The normalized releases from Sellafield and Cap de la Hague from 1980-1985 averaged about 40 GBq (GW a)^{-1} to the sea and 4 GBq (GW a)^{-1} to the atmosphere, giving a total of 44 GBq (GW a)^{-1}. which, when weighted for the fraction of fuel that is reprocessed, gives 2.2 GBq (GW a)^{-1}. The corresponding incomplete collective effective dose commitment to 10,000 years is 1.5 man Sv (GW a)^{-1}. The incomplete value to 10^4 years is 0.0093 man Sv (GW a)^{-1} and for 100 years 0.0008 man Sv (GW a)^{-1}. as shown in Table 70.

VI. TRANSPORT

191. Materials of various types are transported between the installations involved in the entire fuel cycle. The amounts and distances depend on the number of facilities and the degree to which different facilities are located together. An estimate is given in Table 71 of the transport needs in a complete nuclear fuel cycle; this has been adapted from the report of the International Fuel Cycle Evaluation (INFCE) [115]. In general, mills are located together with mines, and tailings are disposed of close by, so that there is no significant requirement for transport of very large quantities of ore or wastes. The other major transport requirements shown in Table 71 can not be eliminated by co-location, as other factors will dominate the site requirements. IAEA has continued to work towards a full assessment of the radiological impact of transport and have recently published the preliminary findings of a technical committee [P13]. The general conclusion was that, although the data available were incomplete, the indications were that exposures resulting from normal transport operations were low both for workers and members of the public.

192. The estimates made during the course of the IAEA study of occupational collective doses from the transport of fuel cycle materials were a recognized cautious estimate of 19 man Sv for the United States as a projection for 1985 [N11] and a more realistic estimate of 0.14 man Sv for the United Kingdom in 1981 [G1]. Estimates of less than 0.01 man Sv were made for selected operations in France, Italy and Sweden, but these could not be normalized to energy production. Using the energy production figures for the appropriate years gives normalized collective effective dose equivalents of 0.5 man Sv (GW a)^{-1} for the United States and 0.04 man Sv (GW a)^{-1} for the United Kingdom. Noting that the United States assessment was pessimistic, that the United Kingdom assessment did not include the transport associated with uranium mining and milling, an overall estimate of 0.2 man Sv (GW a)^{-1} is probably reasonable.

193. Doses to members of the public were also estimated as part of the work of the IAEA committee, based again on submissions from the United States and the United Kingdom. The estimate for the United States was 19 man Sv for 1985 [N11], the same as that for occupational exposure, whereas that for the United Kingdom was several orders of magnitude lower, at 0.001 man Sv for 1981 [G1]. No estimates were available for other countries. Based mainly on the more realistic British assessment, it seems reasonable to conclude that public exposure from transport is less than occupational exposure and to adopt an estimate for the normalized collective effective dose equivalent of 0.1 man Sv (GW a)^{-1}.
VII. SUMMARY

194. In the UNSCEAR 1982 Report the Committee carried out a thorough assessment of the exposures to the public from nuclear power production. In this Report the same basic assumptions and environmental transport models are used to carry out a revised assessment based on discharge data for the quinquennium 1980-1984. Some aspects of waste disposal have been treated here in more detail, especially the long-term impact of uranium mill tailings and the disposal of solid low- and intermediate-level wastes by burial on land. The contribution from reprocessing is based more closely on the results being obtained at operating plants rather than on the notional plant used in the previous report. Occupational exposures from the various stages in the fuel cycle are reviewed in this Annex in association with the other exposures from released radioactive materials.

195. A summary of the local and regional normalized collective effective dose equivalent commitments from the nuclear fuel cycle is shown in Table 72. The total of 4 man Sv (GW a)\(^{-1}\) is essentially the same figure as that derived in the UNSCEAR 1982 Report if the contribution from uranium mine tailings is excluded, although in this Annex reprocessing is added explicitly, whereas a notional plant was used for the UNSCEAR 1982 Report. Contributions other than radon arise mainly from routine atmospheric releases from reactors and the liquid discharges from reprocessing. Effectively, all of these dose commitments are received within one to two years of discharge.

196. The normalized collective effective dose equivalent commitments from the long-term releases from solid waste disposal are shown in Table 73. The dominant contribution, as was recognized in the UNSCEAR 1982 Report, is from mine and mill tailings. The numerical estimate is roughly proportional to the length of time for which release of radon is assumed to occur. The estimate of 150 man Sv (GW a)\(^{-1}\) corresponds to 10000 years for a tailings pile with a reasonable covering. The estimate for disposals of LLW and ILW are for the release from the disposal sites for all time, but a large proportion of the dose is received within about 10\(^4\) years from the date of disposal. This applies also for the globally dispersed radionuclides shown in Table 73, as these are dominated in terms of the normalized contribution by \(^{14}\)C.

197. The contributions of the various stages of the fuel cycle to occupational doses are summarized in Table 74. The dominant contribution is from reactor operation, itself based mainly on recent experience with LWRs in the United States but with considerable data from many other countries.

198. The per caput doses from existing nuclear power production are estimated from the contributions to collective dose commitment in the short term. This collective dose commitment is from local and regional collective doses and from occupational exposure, i.e., 4 and 12 man Sv (GW a)\(^{-1}\), respectively (Tables 72 and 74). Assuming a global population of \(5 \times 10^9\), the per caput dose would be 3 nSv (GW a)\(^{-1}\). The energy production from nuclear power in 1987 is about 190 GW a (Table 2), so that the annual per caput dose is estimated to be 0.6 \(\mu\)Sv.