SOURCES AND EFFECTS OF IONIZING RADIATION

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NOTE

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ANNEX

Effects of radiation on the environment

CONTENTS

INTRODUCTION

1. The effects of radiation on plant and animal communities have not so far been specifically addressed by the Committee. Living organisms have primarily been considered as part of the environment in which radionuclides become dispersed and as resources which, when contaminated, may contribute to human exposures. For example, some plants and animals are elements of food-chains and represent pathways for the transfer of radionuclides to man and an eventual source of internal exposure. Like man, however, other organisms are themselves exposed to internal irradiation from accumulated radionuclides and to external exposure from contamination of both the living and inanimate components of their environment.

2. In the absence of reports of obvious deleterious effects in other organisms from environmental radiation, either natural or from controlled releases of man-made radionuclides, it has generally been accepted that priority should be given to evaluating the potential consequences for man, who is among the most radiosensitive mammalian species, and to providing a sound basis for protecting human health. This position has, however, been questioned recently [D4, T1], and it has been shown that there is at least one situation, namely in deep-sea sediments, an environment very remote from man, where the above accepted priority could be incorrect [13]. It should also be noted that detrimental effects on the environment have been observed in localized areas as a consequence of plants and animals having received short-term, very high radiation doses following major accidental releases of radionuclides, e.g. in areas affected by the southeastern Urals accident of 1957 and the Chernobyl accident of 1986.

3. In response to such concerns, and to demonstrate explicitly that full account can be, and is being, taken of the potential effects of radiation on the environment, reports have been prepared that consider a number of specific situations, e.g. fresh-water and terrestrial environments [14, M5, N1], the deep ocean [13] and coastal waters [W9]. In view of these many activities, it would seem to be an appropriate time for the Committee to make an independent review of the situation and summarize the present state of knowledge of the impact of ionizing radiation on plant and animal populations in natural and contaminated environments. Such a summary would, of necessity, draw upon the previous, more limited reviews and studies, but it should adopt the widest possible scope and, in particular, should utilize the results of the Committee's own earlier summaries of the radiobiological work carried out over the last 50 years. It is not, however, the intention of the Committee to make any judgement on particular environmental situations.

4. The presence of cosmic radiation and natural and artificial radionuclides in the environment implies a

consequential radiation exposure of the indigenous populations of organisms and, as is the case for man, it is to be expected that the probability of adverse effects would be greater where exposures are higher than the range of the natural background radiation dose rate. There is. however, a fundamental difference in the viewpoint adopted for the evaluation of the risk. For man, ethical considerations make the individual the principal object of protection, and in practice this means that the incremental risk to the individual arising from increased radiation exposure must be constrained below some level which society judges to be acceptable; this level of risk. although small. is not zero [18]. Although man displays an enormous range of attitudes towards the other species that share this planet (consider, for example, a population of mosquitoes at one extreme and an individual giant panda at the other), for the vast majority of organisms it is the population that is considered to be important. and protection of each population from increased risk from radiation might be the appropriate objective: exceptions might be populations of small size (rare species) or those reproducing slowly (long generation times and/or low fecundity). for which it might be more appropriate to target protective measures at the level of the individual organism. Thus, depending on whether it is the individual or the population that is of interest. there are likely to be differences in the responses that are significant for the assessment of potential or actual impact. Nevertheless, it is self-evident that there cannot be any effect at the population level (or at the higher levels of community and ecosystem) if there are no effects in the individual organisms constituting the different populations. It is not the case, however, that detectable radiation-induced effects in some members of a population would necessarily have any significant consequences for the population as a whole.

5. In considering impact assessment. it is also necessary to note that natural populations of organisms exist in a state of dynamic equilibrium within their communities and environments and that ionizing radiation is but one of the stresses that may influence this equilibrium. The incremental radiation exposure from human activities cannot, therefore, be considered in isolation from other sources of stress, both natural (e.g. climate, altitude, volcanic activity) and of human origin (e.g. synthetic chemical toxins, oil discharges, exploitation for food or sport, habitat destruction). When, as is not uncommon, ionizing radiation and chemicals, both from human activities, are acting together on a population. there arises the difficult problem of correctly attributing any observed response to a specific cause.

6. The objective of this Annex is to summarize and review information on:

(a) the radiation exposures (actual or potential) received by organisms in their natural habitats from the natural radiation background. from radionuclides released to the environment in a controlled manner from industrial activities and from radionuclides released as a consequence of accidents: and

(b) the responses to acute and chronic irradiation of plants and animals, both as individuals and as populations;

It is not the purpose of this Annex to predict the impact of specific incremental radiation exposures arising from human activities on populations of wild organisms. but with the information base provided, national and international bodies may be able to select appropriate criteria for the radiological protection of natural populations, communities and ecosystems.

I. DOSIMETRY FOR ENVIRONMENTAL EXPOSURES

A. METHODS OF DOSE ESTIMATION

7. Accurate estimation or determination of the dose rate and the integrated dose to organisms is essential for assessing the potential or actual impact of contaminant radionuclides in the environment. This simple statement conceals a multitude of difficulties that prevent the easy achievement of that estimation. In certain very restricted circumstances it has been possible to use thermoluminescent dosimeters to obtain direct estimates of the radiation exposure, but such techniques are of limited general applicability. Direct measurements are especially difficult if the dose rate is low, if the organism is small, if the dose rate to a specific organ or tissue is required or if alpha or beta particles are a major source of exposure. For mobile species of wild organ-

isms there is the additional difficulty of recapturing sufficient dosimeter-marked animals to determine the range of dose rates and doses experienced by the population.

8. Although there is usually little need for high precision in the quantification of the radiation exposure of organisms in the environment, it is nevertheless necessary to distinguish between the various dosimetric quantities and to indicate their interrelationships. In the case of a photon field in air, that is, for laboratory studies employing either x rays or gamma rays from sealed sources and for field studies with large gamma-ray sources. it has been common practice to determine the exposure in free air. The conversion of these data to estimates of the absorbed dose in the relevant targets is

complex because of the assumptions that are necessary concerning the uniformity of the photon field, the presence of full secondary electron equilibrium and the absence of photon scattering.

9. When the now obsolete units roentgen and rad were employed to quantify the irradiation of organisms by photons, it was usual to equate the numerical value of the exposure in free air to the absorbed dose in the organisms (see, for example. [S1]). This approximation may be adequate, provided that the organism is sufficiently small for absorption and multiple scattering to be neglected but large compared with the range of secondary electrons.

10. A more accurate and more general numerical relation can be based on the air kerma, K_a, which is given by

$$K_a = aX \qquad (1)$$

where X is the exposure in C kg^{-1} and a is the conversion factor applicable for most photon energies between 0.1 and 4 MeV. 33.85 Gy per C kg^{-1}. It may be noted that the form of this relation (and those given below) applies equally to the time derivatives.

11. In the case of radiation equilibrium, D_{org}. the absorbed dose in an organism, is numerically equal to K_{org}, the tissue kerma at the same point. Then,

$$D_{org} = K_{org} = s K_a \qquad (2)$$

where s is the ratio of the mass energy absorption coefficients of the tissue of the organism and the air (taken to be 1.10 for photon energies in the range 0.1-4 MeV). Thus,

$$D_{org} = s a X \qquad (3)$$

The product of s and a, which ICRU designates as the factor f [I1], relates the absorbed dose at a specified point in the medium to the exposure at that point. It has the value 37.0 Gy per C kg^{-1} for muscle tissue and 37.5 Gy per C kg^{-1} for water [I1].

12. For organs and tissues at some depth in larger organisms, the effect of geometry is taken into account by introducing the factor g:

$$D_{org} = s g K_a \qquad (4)$$

where K_a is the air kerma in free air. For example, an average value of 0.7 has been assumed for sg to estimate the absorbed dose in the gonads of human beings: this is an average value for males and females to be applied in both outdoor and indoor exposure conditions [U2]. Because the value of s is taken to be 1.10, it may be inferred that g has a value of 0.64. Values of sg have also been given for small mammals

[Z1]. Again, assuming s = 1.10, the values of g are 0.89 for the mouse. 0.87 for the rat, 0.81 for the monkey and 0.70 for the dog in conditions of bilateral exposure with ^{60}Co gamma rays. These values give a qualitative indication of the effect of animal size on the midline absorbed dose in the organism for a constant air kerma but are applicable only for the specific situation described.

13. Even in the relatively straightforward situations considered above, simplifications have been made. e.g. by averaging over photon energy for the parameters a, f and s. In a natural environment, where the assumptions of photon field uniformity, secondary electron equilibrium and no photon scattering are unlikely to be valid and the radionuclide distribution is certain to be inhomogeneous, simple derivation of the absorbed dose from an estimation of the exposure is not possible.

14. In these circumstances it is necessary to employ dosimetric models to estimate the radiation exposure. These can be developed to take account of the radiation type; the specific geometry of the target, e.g. the whole body. the gonads, the developing embryo or the plant meristem: and the source of exposure, e.g. radionuclides accumulated in body tissues, adsorbed onto the body surface or distributed in the underlying soil. Clearly, it is not possible to consider all organisms, and there are limitations on the basic data that can be made available as input for the models, e.g. the spatial and temporal distributions of the radionuclides both within the organism and in the external environment. Additional sources of complexity arise from the behaviour of mobile organisms in modifying the exposure from external sources and, particularly for some aquatic organisms and many insects, from the occupation of different environmental niches at different stages of the life cycle. Thus the models must be simplified and generalized without losing the realism essential for the valid estimation of dose.

15. A dosimetric model is essentially a mathematical construction that allows the energy deposition in a defined target to be estimated from a given radionuclide (source) distribution. The model is often derived using theoretical or empirical functions that describe the distribution of dose about a point source [B11, B12, L5. W7]. The dose at a point in the target is then obtained by integrating the point source dose distribution function over the defined radionuclide source, either internal or external. This procedure is frequently simplified by using simple geometries (spheres. ellipsoids etc.) of appropriate size to represent the target and by assuming that the radionuclide distribution is uniform (over a surface or through a volume) or that it varies in a way that can be described by a simple mathematical expression (e.g. an exponential decline in radionuclide concentration with depth in soil or sediment).

16. Another approach that has been employed for gamma radiation is to use the known cross-sections for photon absorption and scattering in Monte Carlo calculations to determine the absorbed fraction, defined as

$$\phi = \frac{\text{Photon energy absorbed in target}}{\text{Photon energy emitted by the source}} \quad (5)$$

for a variety of source and target geometries [B13, E3]. These data can be used, either directly or with interpolation (or, to a lesser extent, extrapolation), for geometries that can represent targets of environmental concern [N1, O4].

17. In principle, these procedures can be adapted for use with terrestrial and aquatic organisms from both the plant and animal kingdoms and for both internal and external sources of radiation. In practice, the use has been more limited.

18. The presence of an alpha-particle component in the total absorbed dose rate to a tissue in a plant or animal raises the question of how to take account of the probably greater effectiveness of this radiation type (quality of radiation) in producing biological damage. Such considerations have been very critically examined for the purposes of human radiological protection, where each component of the absorbed dose to a tissue or organ is weighted according to the radiation quality [18]. It seems reasonable to apply a similar approach to radiation dosimetry for organisms other than man. In practice, however, there are circumstances that alter the detailed application of the approach, in particular, the choice of the radiation weighting factors and the descriptive terms employed. In the human case, the major concern has been with the induction of stochastic effects (principally cancer) at low doses and dose rates. For alpha radiation, experimental determinations of relative biological effectiveness (RBE) lead to a recommended radiation weighting factor of 20 for the purpose of human radiation protection. In the case of wild organisms, however, it is likely to be deterministic effects that are of greatest significance, and for alpha radiation the experimental data for animals indicate that a lower weighting factor, perhaps 5, would be more appropriate; the weighting factors for beta and gamma radiation would remain unity. It has been assumed that these weighting factors would also apply for effects in plants, although there are no definitive experimental data. It should be pointed out that the various equivalent dose quantities, and the name of the unit, sievert (Sv), are specific to human radiation protection. In the absence of corresponding dosimetric concepts and quantities developed for application to non-human organisms, in this report the absorbed doses from low-linear energy transfer (LET) radiations (beta particles, x rays and gamma rays) and from high-LET radiation (alpha particles) are assessed and specified separately.

The absorbed doses retain the unit joule per kilogramme (J kg^{-1}) and the unit name gray (Gy).

B. DOSE CALCULATIONS

1. Plants

19. In the case of plants, the absorbed dose from photons is generally not equal to air kerma (see paragraph 8), and there is a complex relation between the radiation field and the absorbed dose that depends not only on the relative mass energy absorption coefficients of air and the plant material but also on the relative stopping powers for electrons. For beta particles, electron scattering, as well relative stopping power, is involved. In approximate calculations it may be adequate to assume that the ratios of the coefficients and stopping powers are unity and to neglect scattering. It may also be assumed that there is radiation equilibrium in air, i.e. that the absorbed dose, D, is equal to the kerma, K.

20. With these simplifications Jacobi and Paretzke [J1] considered methods of calculating doses to plants, specifically for leaves of trees and pine needles, from exposure to natural and artificial sources of radiation. The dose rate to the plant material, $\dot{D}_p$, has external and internal components. For the external component, the relationship between the absorbed dose rate to the plant and the air kerma rate is

$$\dot{D}_{p,ext} = s\,g\,\dot{K}_a \quad (6)$$

The factor s is taken to be unity. The geometry factor g equals one for gamma rays and higher energy beta particles but drops to zero for low energy beta particles unable to penetrate the leaf cuticle to the cell growth layer, at a depth of around 0.1 mm. If the radiation exposure results from radionuclides in air, the dose rate may be expressed in terms of the radionuclide concentration in air:

$$\dot{D}_{p,ext} = s\,g\,w\,E\,C_a/\rho_a \quad (7)$$

where w is the disintegration fraction, E is the energy per disintegration (1 MeV = 1.602 10^{-13} J). C_a is the concentration of the radionuclide in air (Bq m^{-3}) and ρ_a is the density of air (may be taken to be 1.3 kg m^{-3}). The cumulative doses to plant components may be estimated from assumed mean lifetimes for leaves of 0.5 years and needles of 7 years [J1].

21. For estimating internal dose rates in plants, the relevant equation is

$$\dot{D}_{p,int} = \phi\,w\,E\,C_p/\rho_p \quad (8)$$

where ϕ is the absorbed fraction of the energy released through disintegrations, C_p is the concentration of the radionuclide in the plant material (Bq m^{-3}) and ρ_p is the density of the plant material (taken to be 800 kg m^{-3}). Because of the small dimensions of leaves and needles, the factor ϕ is small for gamma radiation and higher energy beta particles and increases to unity as the beta energy declines and for alpha particles.

22. There are two points to be noted about this approach: first, the calculation of the dose rate from the external source takes no account of the possibility of self-shielding and is thus likely to be an overestimate; and secondly, the expression for the dose rate from the internal source does not include the contribution from contamination in other parts of the same plant and will thus tend to underestimate the dose rate.

(a) Natural sources

23. For natural tritium and ^{14}C produced in the atmosphere through the interactions of cosmic rays, the doses to terrestrial plants are, owing to the very short ranges of the beta particles, almost entirely from internal sources. The concentrations of the radionuclides in plant tissue may be estimated from environmental data. The natural concentration of tritium in surface waters is about 0.4 Bq kg^{-1} [U4]. For average concentrations of water in leaves and pine needles of 70% and 50%, respectively [J1], and assuming the dry matter is cellulose equivalent (6% hydrogen by weight), the estimated tritium contents are 0.35 Bq kg^{-1} (leaves) and 0.31 Bq kg^{-1} (pine needles). The activity of ^{14}C in terrestrial biological material is 230 Bq kg^{-1} carbon. Again assuming that the plant dry matter is cellulose equivalent (44% carbon by weight), the ^{14}C concentrations are 31 Bq kg^{-1} (leaves) and 51 Bq kg^{-1} (pine needles). These concentrations of cosmogenic radionuclides correspond to absorbed dose rates of about 10^{-6} µGy h^{-1} to both leaves and pine needles from tritium and 0.0009 µGy h^{-1} and 0.0016 µGy h^{-1} to leaves and pine needles, respectively, from ^{14}C. It is fairly straightforward to estimate the dose rates for these natural, global and atmospheric sources of tritium and ^{14}C, because the resultant distributions are, more or less, in dynamic equilibrium throughout the biosphere.

24. There are, however, many data on the concentrations of tritium and ^{14}C in environmental materials that demonstrate, through the presence of different specific concentrations (activity amount per unit mass of the element) in different compartments, time-dependent transport processes. Some of them have been demonstrated, for example, in monitoring and modelling the time-dependence of tritium in air, soil and vegetation at the Savannah River fuel reprocessing plant in the United States [M20]. General features of tritium in plants have been considered in a review by Belot [B16]. The formation and behaviour of organically bound tritium in plants has been determined and compared with that of tritiated water in plant systems, e.g. [D19, 16]. These aspects, which give rise to an overall variation by a factor of up to 10 in specific concentration, complicate the more simplified view of tritium in the environment, which considers that it occurs only in the form of tritiated water and that equilibrium specific concentrations prevail. Similar considerations, although of lesser degree (overall variation in specific concentration by a factor of up to 4 [B31]) arise from more detailed analyses of ^{14}C behaviour. Thus, assessments of the impact of additional sources of these two radionuclides in the environment, often very localized in nature and often involving releases to compartments other than the atmosphere, e.g. surface waters and landfill, need to take account of the time-dependent pathways for radionuclide redistribution, the chemical forms and transformations and the potential influence of source-dependent factors on nuclide accumulations in plants.

25. Potassium-40 is both an internal and an external source of radiation exposure to plants. The external component may be considered using the general approach discussed in paragraphs 8-12. The internal dose depends on the potassium content of the plant, which is 0.1%-0.5% of the fresh weight in leaves and needles. The atom ratio of the potassium isotopes, ^{40}K/^{39}K, is 1.2 10^{-4}, giving a specific concentration of 32 Bq g^{-1} potassium and a ^{40}K concentration in fresh plant tissues in the range 32-160 Bq kg^{-1}. Because of the relatively high beta energy of ^{40}K, the absorbed fraction, ϕ, may be no more than 0.1-0.15 [J1]. In roots, trunks and branches of trees, the value of ϕ would be more nearly equal to one. With these assumptions, the ^{40}K in plant tissue could deliver a dose rate in the range 0.001-0.006 µGy h^{-1}.

26. The concentrations of radon, ^{222}Rn, in surface air are quite variable. Time-averaged concentrations in normal areas may be in the range 2-30 Bq m^{-3}. With less vertical air exchange in forests, the radon concentrations may be expected to be higher there than over open land for the same rate of radon exhalation from the ground. For the noble gases, the radionuclide concentration in plant material is given by

$$C_p = L C_a \qquad (9)$$

where L is the dimensionless solubility coefficient. The value of L depends on temperature and is approximately 0.5 for radon in leaves and needles at 15°C [J1]. In soil in the root zone, the radon concentrations may be higher by a factor of about 1,000 than in open air [J1].

27. The average concentration of ^{222}Rn in groundwater varies widely, depending on the composition of the soil

and the bedrock. For soil with an average ^{226}Ra content of 40 Bq kg^{-1} [U2] the average radon concentration in soil water would be about 60 Bq kg^{-1} [U3]. Much higher values have been measured in deep groundwater in certain areas, for example, 8,000 Bq kg^{-1} in granitic areas of Maine in the United States [H4] and 50,000 Bq kg^{-1} in Finland [J2]. Lewis and McDonell [L6] found that the main route of accumulation of ^{222}Rn into plants was through mass flow with water uptake by the roots, although the subsequent loss was not related to transpiration. The data indicate that the ^{222}Rn is lost faster than the water, resulting in a lower ^{222}Rn concentration in water in the aerial tissues of plants than in groundwater. This is consistent with the data of Kirchmann and Berino [K8], who showed that less than 10% of the ^{222}Rn produced from the *in situ* decay of ^{226}Ra was retained by pea plants. Thus there is likely to be a significant ^{222}Rn concentration gradient in the plant water, from a high value (close to the groundwater concentration) in the roots to a value lower by a factor of as much as 10 in the leaves. Assuming that the radon content in plant water in the leaves is one tenth that in groundwater (i.e. 6 Bq kg^{-1} on average), that the fractional water content of the plant is 0.8 and that the short-lived decay radionuclides are at equilibrium, then the absorbed dose rate from this source is approximately 0.06 µGy h^{-1}, of which 95% is from high-LET alpha radiation. This is within the range 0.005-0.54 µGy h^{-1} suggested by Jacobi and Paretzke [J1]. It is clear that the absorbed dose rate to the root meristem tissue could be substantially greater. Because the data cited above relate to small plants [K8, L6], it is possible that the dose rate to trees, for a given concentration of ^{222}Rn in groundwater, could be greater, owing to the longer residence time of the ^{222}Rn in the plant.

28. The radon decay products ^{210}Pb and ^{210}Po also contribute to the dose to plant components. While ^{210}Pb may be taken up by the plant through roots and leaves, it occurs predominantly as deposited surface contamination [J1]. Average concentrations of ^{210}Pb in leaves and needles are 10 and 5 Bq kg^{-1}, respectively [J1], and a ^{210}Po/^{210}Pb activity ratio of 0.8 may be applied to derive ^{210}Po concentrations.

29. The factors for estimating doses to leaves and needles of trees from natural radionuclides in the plant material, as derived by Jacobi and Paretzke [J1], are presented in Table 1. They provide a basis for estimating the radiation dose rate received by trees from natural background sources. The results are summarized in Table 2. Four additional assumptions are made in deriving these values:

(a) the absorbed dose rate from the ionizing component of cosmic radiation is 0.032 µGy h^{-1} at sea level [U2];

(b) the absorbed dose rate from the neutron component of cosmic radiation is 0.0004 µGy h^{-1} [U2];

(c) the absorbed dose rates for the long-lived ^{238}U and ^{232}Th series radionuclides are as given in [J1];

(d) the average content of ^{222}Rn in forest air is 10 Bq m^{-3}, with concentrations ranging up to 50 Bq m^{-3}. The dose factor (0.00066 µGy h^{-1} per Bq m^{-3} for the beta/gamma component) is taken from [J1]. For ^{222}Rn in air, only the external beta/gamma dose from short-lived decay products is of significance. The remainder of the dose from ^{222}Rn comes from the ^{222}Rn dissolved in groundwater and taken up by the roots into the plant.

30. It is apparent from Table 2 that the greatest part of the dose rate, and its potential variation, arises from the ^{222}Rn dissolved in the groundwater and taken up by the roots. Other sources of variability, such as the range of ^{40}K contents in the plant material and the increase in cosmic radiation exposure with altitude [U2], are of minor consequence. Small herbaceous plants experience an additional exposure from the beta radiation from ^{40}K, ^{238}U series and ^{232}Th series radionuclides in the soil. For the range of concentrations given in [U2] and assuming equilibrium in the two decay chains, the maximum additional exposure from these sources could be 0.1, 0.1 and 0.05 µGy h^{-1}, respectively; the actual exposure is, however, likely to be much less, owing to geometrical factors.

31. Point source dose distribution functions have been used to estimate the absorbed dose rate to phytoplankton in both fresh and marine waters from natural radionuclides [I2, W9]. The results are summarized in Table 3. For both fresh-water and marine phytoplankton, the greatest part of the natural background dose rate is delivered by alpha particles: in the former case by ^{222}Rn and its decay products in the surrounding water and in the latter case by ^{238}U and ^{234}U and by ^{228}Th plus its decay products accumulated in the organisms. As has been noted in another context [H5], low values for the estimated absorbed dose rate to small organisms must be interpreted with care. For example, the low total radionuclide content in any individual phytoplankton cell and the discrete nature of the radiation emission process mean that many cells will pass through a number of divisions without experiencing any dose; conversely, a cell in which a radionuclide decay does occur will experience a relatively higher dose. In these circumstances, macroscopic dosimetry principles, which average the absorbed radiation energy over a relatively large mass (1 g or greater), are very unlikely to be appropriate.

(b) Artificial sources

32. The factors for estimating doses to the leaves and needles of trees from artificial radionuclides of noble gases in air, as derived by Jacobi and Paretzke [J1], are

summarized in Table 4. They can be used to indicate absorbed doses to the leaves and needles of trees using the normalized discharges to the atmosphere from nuclear reactors given in e.g. ([U2]). It is assumed that the average integrated activity concentrations at 1 km from the source for dispersion in all directions can be estimated by a simple approximation [U5], the value being $3 \cdot 10^{-7}$ Bq s m^{-3} per Bq. No allowance has been made for radioactive decay during the transit time from emission to target. The results are given in Table 5. The values of normalized dose may be multiplied by the annual electrical energy generated (e.g. 0.5 or 1 GWa) to obtain estimates of the annual dose for a typical reactor.

33. The potential impacts of releases of tritium and ^{14}C can be assessed in a similar manner, with the additional assumptions [C5] that at 1 km from a nuclear power plant these radionuclides have become uniformly incorporated into the background atmospheric water vapour (3 g m^{-3}) and carbon dioxide (0.046% by weight), respectively, and that the corresponding specific concentrations are also appropriate for tree leaves and needles. Using the assumptions of hydrogen and carbon contents of leaves and needles given in paragraph 23, the normalized absorbed dose rates may be estimated. The resulting values are, however, likely to be overestimates, because the specific concentrations in the terrestrial compartments will probably be less than those in the atmosphere. The extent of the overestimation will depend on the relative sizes of the hydrogen and carbon pools in the different compartments, the net rates of exchange between them and the histories of the discharges; it will, therefore, be site-specific.

34. The absorbed dose rates to fresh-water and marine phytoplankton from contaminant artificial radionuclides have been assessed using point source dose distribution functions [12, W9]; the results are summarized in Table 3. The caveat in paragraph 31 concerning the interpretation of low absorbed dose rates to small organisms is equally applicable here.

35. Absorbed dose rates to aquatic macrophytes have been estimated by a number of authors [K4, M5, N3] using a generally conservative model based on radiation absorbed fractions in a 30 cm diameter sphere. Using derived values for the environmental concentrations arising from discharges in liquid effluents at 16 nuclear power station sites, Kaye [K4] estimated the dose rates from internal sources in the range 0.009-20 µGy h^{-1}, mainly from isotopes of caesium. Myers [M5] estimated the dose rates from a variety of radionuclides for individual concentrations in water used either as drinking water (intake 2 l d^{-1}) or as a source of fish for human consumption (0.1 kg d^{-1}), with either pathway leading to an effective dose equivalent of 1 mSv a^{-1}. In both cases the estimated absorbed dose rates to aquatic plants were as high as 1,000 µGy h^{-1}, with the highest

values being due to the alpha-emitting radionuclides. These figures are theoretical and reflect the limited use of resources (water and fish) by humans rather than potentially realistic doses to aquatic organisms.

36. Neither of the above authors [K4, M5] considered exposure from external sources, and it is likely that the overestimation of the dose rate from gamma radiation within the plants (implicit in the model) would be partially compensated by this absence of an estimate for the contribution from gamma-ray sources in the water and sediment. Although the model probably does not overestimate the absorbed dose rate from alpha radiation from internal sources, for which complete absorption is a reasonable assumption, a number of considerations suggest that the dose-rate estimates should nonetheless be treated with caution:

(a) the large surface area per unit weight of aquatic macrophytes almost certainly means that the observed concentration factors represent adsorbed as well as metabolically incorporated radionuclides;

(b) the dose factors for certain of the radionuclides [N3] include contributions from both short- and long-lived decay products where appropriate, but for the latter the derived concentrations in water are based on an extremely conservative model, in that there are no loss processes operating in the water column over a 30-year period; and

(c) on the basis of the information given concerning the model and its parameters [N3], the dose-rate factors for the alpha-emitting radionuclides appear to be too high by a factor of up to 10.

These considerations lead to the conclusion that the dose rates are likely to be overestimates.

37. A technical report published by IAEA [I4] also provides estimates of the dose rates to terrestrial plants from deposited radionuclides following discharges to the atmosphere. The model PATHWAY [W10], developed to estimate doses to man, was used to derive the equilibrium concentrations of radionuclides in plants (and animals, see below) for the limiting case that man, while living on the land, breathing the air over it and eating food produced from it, would receive an annual effective dose of 1 mSv. To estimate the dose to the plant from internal sources, it was assumed that the energy of alpha and beta particles would be totally absorbed (except ^{32}P, which would be 50% absorbed) and that 10% of the emitted gamma-ray energy would contribute to the dose. An additional degree of conservatism was provided by using estimates of the radionuclide concentrations in plant tissue on a dry weight basis, which are 5-10 times higher than on a wet weight basis, to calculate the absorbed dose rates to living (wet) plant tissue. The results are given in Table 6.

38. Spirin [S23] developed a dosimetric model to estimate radiation doses to agricultural crops from radionuclides in the soil. The air plus the plant material is represented as a homogeneous medium of uniform density (intermediate between the densities of air and of plant material) that attenuates the radiation field, developed by integrating a point source isotropic dose function over the (plane) source in the soil. This provides a dose distribution through the depth of the plant layer. Redistribution of the radionuclides from the surface of the soil into a 20 cm deep layer reduces the estimated beta dose by a factor of 30 and the gamma dose by a factor of 3. It was concluded that the dose from external alpha emitter contamination would be negligible compared with that from radionuclides taken up by the plants. In the latter case, the alpha radiation is assumed to be completely absorbed in the critical tissue (the apical meristem or leading shoot), in which the radionuclide concentration was related to that in the soil by a simple uptake factor. For equal degrees of soil contamination (1 Bq m^{-2} mixed to 20 cm depth) the calculations showed that alpha radiation, e.g. from ^{237}Np, could be a much more significant source of exposure than beta and gamma radiation from ^{90}Sr-^{90}Y or ^{137}Cs [S23].

2. Animals

39. There appear to have been fewer estimates of the potential exposure of truly terrestrial animals than of animals occupying semi- and fully aquatic niches. This is probably a reflection of the greater use that has been made of aquatic systems for waste discharge.

(a) Natural sources

40. Estimation of the natural exposure of terrestrial animals appears to have attracted little attention, probably because it is implicitly assumed that the magnitude (and range) of the exposure will be little different from those of the exposure of man. This is likely to be a reasonable assumption, given that man coexists with animals throughout the terrestrial environment and that even situations leading to enhanced exposure, e.g. arctic food-chains [H3] or the occupation of burrows (enhanced radon exposure) seem to have parallels in the case of man. There can also, however, be significant differences. Animals living above ground in the open are likely to receive a lower alpha radiation dose to the lungs, because the concentration of radon and short-lived progeny will, on average, be lower by a factor of 5 than the concentration experienced by the local human population indoors. Animals living below ground receive, in addition to the increased lung dose from radon and its short-lived progeny, an increased dose from terrestrial gamma radiation; for small animals, the beta component from the external sources would also need to be taken into account.

41. In areas of human habitation with normal soil concentrations of ^{40}K, ^{238}U and ^{232}Th, the mean external gamma-ray dose rate in air has been estimated to be 0.057 µGy h^{-1} [U2]. For animals living most of their life underground in those areas, it may be assumed that the incident gamma radiation field has 3π geometry (approximately), yielding a mean gamma dose rate in air of about 0.09 µGy h^{-1}: if a mean burrow occupancy factor of 0.9 is assumed, the annual dose to small animals is about 0.7 mGy.

42. The alpha dose to rodent species inhabiting burrows from ^{222}Rn and its short-lived progeny arises from two sources: the radon and decay products in the inhaled air and the uptake of radon into soft tissue. For both sources, estimates are required of the radionuclide concentrations in burrow air. The equilibrium radon concentration in soil air, C_{SA}(Rn.z), at a depth of z metres is given by:

$$C_{SA}(Rn,z) = \frac{f \cdot C_S(Ra) \cdot \rho}{\varepsilon} \; 1 - e^{-z\sqrt{\lambda_{Rn}/D_e}} \quad (10)$$

(equation 14, Annex D [U5]), where f is the radon emanation fraction, taken to be 0.2; C_S(Ra) is the mass concentration of ^{226}Ra in the soil, taken to be equal to the representative ^{238}U concentration, 40 Bq kg^{-1}: ρ is the bulk density of soil, taken to be 1,600 kg m^{-3}: ε is the soil porosity, taken to be 0.25: λ_{Rn} is the decay constant of ^{222}Rn, 2.1 10^{-6} s^{-1}; and D$_e$ is the effective diffusion constant for radon in bulk soil, taken to be 5 10^{-7} m^2 s^{-1}. All parameter values have been taken from the UNSCEAR 1993 Report (paragraph 83, Annex A) [U2]. The derived estimate for the ^{222}Rn concentration in burrow air at a depth of 0.2 m is 1.7 10^4 Bq m^{-3}. This will be a maximum value because equation (10) does not take account of the influence of advective flows. Owing to adsorption of the short-lived progeny to soil particles, their concentrations in burrow air will be lower than that of the parent radon by a factor in the range 10-100 [J3].

43. The dose rate from ^{222}Rn taken up into soft tissue can be estimated on the assumption that the mean solubility coefficient for radon is about 0.4 10^{-3} Bq kg^{-1} tissue per Bq m^{-3} in air and that the short-lived decay products are in equilibrium with the ^{222}Rn in tissue. This approach yields a dose factor of about 5 10^{-6} µGy h^{-1} per Bq m^{-3} of ^{222}Rn in air. For the estimated ^{222}Rn concentration in burrow air of 1.7 10^4 Bq m^{-3}, the absorbed dose rate to lung tissue (and probably other soft tissues, including the gonads, in small animals) is about 0.085 µGy h^{-1}. The dose rate to the alveoli of the rodent lung from ^{222}Rn (1.7 10^4 Bq m^{-3}) and its decay products (170-1,700 Bq m^{-3}) in inhaled air can be estimated to be 0.25 µGy h^{-1} using the model developed by Drew and Eisenbud [D11].

44. Rodents inhabiting an area of enhanced natural background, the Morro do Ferro, Minas Gerais, Brazil, were marked with fluoroglass dosimeters and retrapped after about five weeks. The absorbed dose rates from external sources over this period were in the range 1.5-7.6 μGy h^{-1}. The concentrations of ^{228}Th, ^{228}Ra and ^{226}Ra were determined in bone, leading to an estimate of 0.16 μGy h^{-1} for the average absorbed dose rate [D10]. The concentrations of ^{220}Rn and ^{222}Rn in the burrows of the animals were high and very variable. An extensive laboratory study of the deposition of the daughter products in the lungs after inhalation allowed the development of a dosimetric model giving estimates of 180 and 29 μGy h^{-1} for the absorbed dose rates at the surfaces of the trachea and alveoli, respectively; the maximum values could be greater by a factor of 4 [D11]. Studies have been carried out in areas of Russia where the gamma dose rate from external sources reaches 80 μGy h^{-1} [M1]. If the internal dose to soft tissues from ^{222}Rn absorbed through the lungs and from its short-lived decay products varies with the external dose rate in the same way as indicated by the calculations in the preceding paragraph, it may be concluded that the absorbed dose rates to the soft tissues of rodents inhabiting burrows in these high natural background areas of Russia could reach as much as 100 μGy h^{-1}.

45. The transfer of the natural radionuclides ^{210}Pb and ^{210}Po along a lichen-caribou-wolf food-chain has been studied in northern Canada and estimates made of the absorbed dose rates in a variety of tissues. In caribou, there appeared to be no discrimination in uptake between ^{210}Pb and ^{210}Po, and for concentration ratios less than unity, there was no biomagnification from lichen to caribou. The ^{210}Po:^{210}Pb activity ratio was less than unity in bone and greater than unity in all other tissues measured (except the foetus), and with a weighted average value of about 0.9 for the whole body, it was concluded that the greater part of the ^{210}Po body burden was derived from ^{210}Pb decay in bone and translocation of the ^{210}Po to soft tissue. The highest absorbed dose rates, 0.7-1.1 μGy h^{-1}, were estimated for the liver; but perhaps of greater significance in the context of this review, the absorbed dose rate in the testes was estimated to be 0.05 μGy h^{-1}. The wolf appeared to discriminate against ^{210}Pb relative to ^{210}Po and to gain the greater part of its ^{210}Po body burden directly from food intake, although with concentration ratios less than unity, there was no biomagnification from caribou to wolf. Again, the highest absorbed dose rates, 0.4-1.2 μGy h^{-1}, were estimated for the liver, with values in the range 0.02-0.04 μGy h^{-1} in the testes [T25, T26]. The greater part of all the radiation dose was delivered by high-LET alpha particles from ^{210}Po.

46. The methods for estimating the doses to aquatic organisms are more completely developed than those for estimating doses to terrestrial organisms [I2, I3, W7]. This is due more to the growing need in recent years to specifically consider the impact of radioactive waste disposal on natural populations in aquatic environments [I3, N1, P5] than to any intrinsic differences in the problems posed. The models are based on point source dose distribution functions or absorbed fractions, as appropriate, and simplifications are made to the extent dictated by the data available for input to the model. Estimates of radiation doses from natural background are summarized in Table 3 for fish and molluscs. More detailed data for these and other aquatic organisms are given in [I2, I3, P5, W8, W9], but the information in Table 3 is quite representative.

47. Two points may be made about the data presented in Table 3. First, the radiation exposure of the molluscs is generally greater than that of the fish, because the benthic (bottom-living) mollusc is exposed to radiation from the sediments and the pelagic (surface water) fish is not. More importantly, the dose rate from internal sources is greater for the molluscs because most of the available data on radionuclide accumulation (concentration factors) are for filter-feeding detritivores. The fine particles making up the food scavenge radionuclides efficiently from the water, thus leading to higher body burdens of the radionuclides. Secondly, the absorbed dose rates from internal exposures to natural radiation sources are relatively high due to the high-LET contribution from ^{210}Po. Indeed, for certain pelagic oceanic organisms, the absorbed dose rates in particular tissues from this source can be very high, e.g. up to 150 μGy h^{-1} in the hepatopancreas and 4 μGy h^{-1} in the testes of a small mid-water shrimp [C7, H6] and 30 μGy h^{-1} in the intestine and 1 μGy h^{-1} in the gonads of a sardine [C8].

(b) Artificial sources

48. Coughtrey [C5] estimated dose rates to the soil fauna, as exemplified by the earthworm, from the deposition into woodland of iodine, strontium and caesium radionuclides released to the atmosphere from a pressurized water reactor at a distance of 300 m. The radionuclide concentrations in the surface soil were estimated from a dispersion calculation and an assumed deposition rate. The consequent accumulations of the radionuclides by the earthworm were estimated using concentration factors. The dose rates were calculated on the following assumptions: for the external sources, that the dose rate from both beta and gamma radiation was that to the soil, which was effectively infinite in extent; and for the internal source that only the beta component was significant and that it was totally absorbed. A different approach was adopted for ^{3}H and ^{14}C releases, which were assumed to equilibrate with the atmospheric water vapour and carbon dioxide, respectively. The resulting concentrations of ^{3}H per gram hydrogen and

^{14}C per gram carbon in the plume were assumed to apply to the earthworm. This is likely to be a conservative assumption. since it neglects the input of uncontaminated water (as rain from outside the plume) and the diluting effect of the large organic carbon pool in living plants and the soil humus; an estimate of the degree of conservatism would require detailed modelling of the particular situation (see also the next paragraph). The total dose rate to the earthworm for the predicted operating conditions of the power plant was 0.01 µGy h^{-1}, and since the greatest part of this was due to ^{14}C. it is certain to be an overestimate; if the station was operated such that the effective dose rate to man reached 1 mSv a^{-1}, then the estimated dose rate to the earthworm would be 5.7 µGy h^{-1}.

49.   As was the case for plants (paragraph 24), the dose from tritium should consider tritium in body water and organically bound tritium in tissues. The relative amount of the two components depends on the source of the tritium, i.e. either tritiated drinking water or organically bound tritium in food (with food type also having an influence [K21]) [L12, L13, P16, P17, R11, T27]. Although the assumption of a constant concentration of tritium (as Bq kg^{-1} hydrogen) in the biosphere could be questioned, it is difficult to see how more specific data could be translated to the environmental context to generate improved estimates of radiation doses from contaminant tritium. A more detailed model of the behaviour of tritium in the vicinity of the input, including the dynamic processes along food-chains, would be required. Similar considerations probably also apply to ^{14}C.

50.   Soil fauna are also potentially at risk from the release of waste radionuclides from shallow landfill repositories. Coughtrey [C5] considered the case in which radionuclides were accumulated by bank soil from stream water draining a trench disposal site. The total dose rates to the soil organisms were estimated to be 18 µGy h^{-1} from low-LET radiation and 14 µGy h^{-1} from alpha radiation.

51.   The PATHWAY model [W10] has been used to estimate radionuclide concentrations in surface-living animals from aerial deposition [14]. The domestic sheep was used as a model terrestrial animal, and it was assumed that the predicted radionuclide concentrations. except for ^{137}Cs, would be typical of wild animals. For ^{137}Cs, the estimated tissue concentrations generated by the model were increased by a factor of 10 to take account of the known influences of either carnivorous behaviour or the consumption of certain plants on the accumulation of this radionuclide by different wild animals. The predicted concentrations in sheep muscle were assumed to apply to the gonads for all radionuclides except ^{90}Sr, for which the muscle value was increased fourfold to give the concentration in the

gonads. The dosimetry model assumed total absorption of the alpha and beta radiation in the gonads and a 30% absorption of the gamma energy. The results for the absorbed dose rates from these internal sources, given in Table 6, correspond to the limiting case of humans receiving 1 mSv a^{-1} from occupation of the same environment.

52.   Kozmin et al. [K9] developed models to estimate the radiation exposure of farm animals following accidental releases of radionuclides to the atmosphere. The radiation exposure pathways considered were external exposure from the plume. inhalation from the contaminated ground-level air. contaminated pasture (external and internal source). contaminated soil (external and internal source) and contaminated livestock barns (external). In practice, it is the internal exposure following ingestion that is of particular significance. Compartment models were developed to predict the radionuclide distributions within the animals and the consequent radiation exposures. Cumulative dose estimates for sheep, pigs and cattle are presented in Table 7 for acute and chronic exposures to ^{131}I (thyroid). ^{90}Sr-^{90}Y (bone) and ^{137}Cs (muscle).

53.   Models based on point source dose distribution functions or absorbed fractions, as appropriate. have been used to estimate the absorbed dose rates to pelagic fish and benthic molluscs from contaminant radionuclides. The results are summarized in Table 3, where it can be seen that, as was the case for the natural sources and for the same reasons. the exposure of benthic molluscs is greater than that of pelagic fish (see paragraph 47).

54.   The dose rates to fresh-water fish have also been estimated for the situation in which humans receive the limiting effective dose rate of 1 mSv a^{-1} from the combined exposure pathways of drinking water (2 l d^{-1}). fish consumption (0.1 kg d^{-1}) and external exposure from contaminated sediments (occupancy 2.000 h a^{-1}) [N1]. The results for a selection of radionuclides in the liquid effluents are given in Table 6. These data are consistent with those from the controlled waste discharge from the Magnox nuclear power station into Lake Trawsfynydd in northern Wales, where the total absorbed dose rate to fresh-water fish from all sources was estimated to be 0.08-0.3 µGy h^{-1} (0.7-2.6 mGy a^{-1}) [W8] when the effective dose rate to the human critical group was estimated to be no greater than 0.25 mSv a^{-1} [M7].

55.   For a benthic fish (the plaice. *Pleuronectes platessa*) in the northeastern Irish Sea, the estimated dose rates from contaminant radionuclides accumulated in the seabed from the waste discharges from the Windscale (now Sellafield) nuclear fuel reprocessing plant were sufficiently large (up to 50 µGy h^{-1}) to attempt direct measurements. Small thermoluminescent

dosimeters were developed that could be attached to the fish, and 3,580 fish were marked with a combined tag/dosimeter and released in the vicinity of the waste outfall. In a period of 2.5 years, 1,053 fish were re-captured, and the dosimeters on 969 of them yielded useful results. After allowing for the mobility of the fish, the data essentially confirmed (within a factor of 2) the estimates of radiation exposure calculated on the basis of a simple model using the measured distributions of the radionuclides in the sediments [W8, W9]. Small thermoluminescent dosimeters have also been used to obtain an estimate of 4 μGy h^{-1} for the absorbed dose rate to periphyton 24 km downstream from the point of discharge, the Hanford plant, into the Columbia River [L7].

C. COMPARISONS OF ENVIRONMENTAL EXPOSURES

56. At present, data on radiation exposures of wild organisms from both natural background and con-taminant radionuclides are incomplete, the more so in some areas than others. The aquatic environment is probably the most thoroughly studied environment to date [I2, I3, I9, N1, N3, W9], but even for it, substantial generalizations have had to be made, particularly in respect of the range of organisms that can reasonably be considered [I3]. As has been emphasized [I3, I9], the limiting factor is not the development of an appropriate dosimetric model for a particular organism but the acquisition of essential input data on the temporal and spatial distributions of the radionuclides both external to and within the organism. Although dynamic models have been employed to describe the dispersion and dilution of radionuclides in a water body, the accom-panying reconcentration transfers to sediments and biological tissues are almost always modelled as equi-librium processes, i.e. simple distribution coefficients and (whole-body) concentration factors are used. This largely neglects the temporal variation in dose rate due to short-term fluctuations in the discharge rate, to differing stages in the life cycle, to behaviour and to short-term environmental processes (e.g. seasonality) etc. As a consequence, while the estimated absorbed dose rate might be a reasonable indication of the general magnitude of the environmental value, it does not pro-vide a very secure basis for evaluating total doses over time. Such reservations apply with equal validity to the more limited data that have been developed for releases to the atmosphere [C5, 14], for shallow landfill sites [C5, 14] and for a deep repository [M5]. It is, nevertheless, possible to draw some general conclusions.

57. As is the case for man [U2], naturally occurring alpha-emitting radionuclides appear to be the most significant sources of background radiation exposure for the majority of wild organisms. In the terrestrial environment the main source is radon and its short-lived decay products. The estimated range of absorbed dose rate to the leaves and needles of trees from this source is 0.005-0.5 μGy h^{-1}. Given the apparent significance of this source of exposure, it would be useful if the radon content of plant tissues could be determined by direct measurement rather than by inference from concentra-tions in groundwater. Studies of the uptake of ^{40}K into plants from potassium-rich soils would also be worth-while, because plants do not have the same regulatory mechanisms for potassium content as animals. For terrestrial animals in areas where the soil has the normal concentrations of uranium and thorium, the highest radiation exposures will be experienced by burrow dwellers. In this case, the absorbed dose rate from external sources is about 0.09 μGy h^{-1}, that from radon (and from its short-lived decay products) accumulated in soft tissue is about 0.085 μGy h^{-1} and that to the lungs from radon and its decay products in the burrow air is about 0.25 μGy h^{-1}. Exceptionally, in areas of uranium or thorium mineralization, the natural background absorbed dose rates could be higher by a factor of as much as 1,000. In aquatic environments, ^{210}Po is the major contributor, with absorbed dose rates of up to several microgray per hour to the gonads of marine animals. The recognition that alpha particles (short-range, high-LET radiation) are the principal source of background radiation exposure underlines the need for (a) detailed knowledge of the distribution of the radionuclides in relation to potential targets as well as realistic models for the estimation of absorbed dose rates and (b) the selection of a radiation weighting factor appropriate to the biological effect of concern.

58. Discharges of tritium, ^{14}C and noble gas radio-nuclides to the atmosphere during normal operation of a reactor do not appear to be an important additional source of exposure to plants, because the increment is likely to be much less than the variability (and uncertainty) in the estimate of the natural exposure. Discharges to the atmosphere of other radionuclides could be more significant, and in the limiting situation for humans (1 mSv a^{-1}), it appears that some plants and animals in the vicinity of the reactor might experience dose rates up to a few tens of microgray per hour; in practice, the actual discharges during normal operation (and, hence, the dose rates) are likely to be much lower.

59. For discharges to aquatic environments both the estimates of dose rates based on measurements of concentrations of radionuclides in contaminated environments and the estimates inferred for the limiting situation of 1 mSv a^{-1} to man are much greater than the estimates of the natural background exposure (see Table 3). The organisms might receive dose rates up to 60 μGy h^{-1} (500 mGy a^{-1}), and in one case an absorbed dose rate of 5,700 μGy h^{-1} (50 Gy a^{-1}) was estimated for aquatic macrophytes [M5].

60. Accident situations are by nature quite different, particularly in their potential to produce high dose rates and doses in the environment. As the actual exposure regime obviously depends on the quantities of specific radionuclides released, the initial dispersal and deposition patterns and their redistribution over time in the environment, it is difficult to generalize. Following the accident at the Chernobyl nuclear reactor, large quantities of short-lived radionuclides were released, leading to high dose rates in the local area. Total doses up to 100 Gy were delivered to trees (and, by inference, to most other organisms in the locality) over a period of a few days [K3]. This radiation regime may be characterized as "acute", in that the doses were delivered in times that were shorter than, or comparable to, the times taken for severe damage to become apparent. During this initial, acute phase the dose rate declined rapidly as the very short-lived radionuclides decayed. The release following the accident in 1957 in the southeastern Urals was dominated by ^{144}Ce-^{144}Pr (approximately 66%; $T_{1/2}$ = 285 d) and ^{95}Zr-^{95}Nb (approximately 25%; $T_{1/2}$ = 65 d). In that case, the dose rates in the local area were also relatively high ($\geq$4,000 µGy h^{-1}) but declined more slowly, such that high total doses causing severe effects (including mortality) could still be accumulated from essentially chronic exposure. Close to the release point, total doses up to 2.000 Gy were experienced [T17]. In the longer term, the exposure regime for the Chernobyl release has been dominated by ^{137}Cs ($T_{1/2}$ = 30 a) and ^{90}Sr ($T_{1/2}$ = 28.6 a) and that for the southeastern Urals accident area by ^{90}Sr: in both cases, the exposures are chronic and moderately high, with responses other than mortality becoming significant. Further experience derived from the study of exposures and effects from these accidents is presented in Section III.D.

D. SUMMARY

61. Dose rates to plants and animals in the environment from naturally occurring or contaminant radionuclides may be estimated from basic dosimetric relationships. The accuracies of the estimates are generally limited by uncertainties in knowing the concentrations of radionuclides that prevail in the media surrounding the organism and that accumulate within the organism. Based on reasonable assumptions, dose factors have been derived for a number of radionuclides, and these have been applied to estimate dose rates in various environments. It is recognized that the simplifications often implied in these assumptions lead to overestimated dose rates.

62. In areas of normal background, the absorbed dose rates to terrestrial plants are 0.07-0.8 µGy h^{-1} (0.6-7 mGy a^{-1}), mainly from ^{222}Rn taken up from the groundwater. Absorbed dose rates to aquatic plants would normally be at the lower end of this range. Absorbed dose rates to terrestrial animals are highest for those inhabiting underground burrows; in this case the rough estimates are 0.25 µGy h^{-1} in the lungs from ^{222}Rn and its decay products and about 0.09 µGy h^{-1} each from absorbed radon in tissue and external exposure. In areas of uranium or thorium mineralization, the absorbed dose rates could be higher by a factor of as much as 1,000. In aquatic environments, ^{210}Po is a major contributor to dose, with absorbed dose rates to the gonads of some marine animals reaching several microgray per hour.

63. Radioactive waste discharges to the atmosphere, to landfills or to aquatic systems from man-made practices entail increased radiation exposures of wild organisms. The incremental radiation exposures are chronic (i.e. continuing) at absorbed dose rates of generally no more than 100 µGy h^{-1} but very exceptionally several thousand microgray per hour. These additional radiation exposures are greater than the normal range of natural radiation background but are generally within the extreme range of background when the exceptional cases of areas of uranium and thorium mineralization are included. Given that waste discharges are point sources and that releases normally become dispersed and diluted (account is taken of reconcentration processes in developing the database of distributions and concentrations of radionuclides used as input to the dosimetric models), these dose rates are likely to apply to only a small proportion of the individuals in any population; the average dose rate to the population will probably be much lower (see, for example, [W8, W9]).

64. Detailed information on the responses of organisms to radiation exposures at absorbed dose rates up to 1,000 µGy h^{-1} is required to make an assessment of potential environmental impact. Responses to acute irradiation or to short-term exposures at higher dose rates can also provide useful information, particularly about the relative sensitivities of different species and classes of organism and of the different life stages of a species. Information on the sensitivity of individual plants and animals to radiation exposure and the possible effects on populations and communities of organisms is presented in Chapters II and III.

II. EFFECTS OF RADIATION ON PLANTS AND ANIMALS

65. The effects of ionizing radiation on plants and animals began to be studied immediately following the discovery of x rays and radioactivity (see, for example, [A13, T24]). Since 1945, when the first nuclear detonations were conducted, there has been widespread concern about the impact of environmental radiation exposures and interest in the environmental behaviour of radioactive materials, as of result of which many studies, using a wide variety of plant and animal species, have been performed [A1, B1, C3, I5, P3].

66. The responses of organisms to radiation exposure are many and varied and may become manifest at all levels of organization, from the individual biomolecule to the ecosystem. The significance of a given response depends on the criterion of damage adopted, and it should not be concluded that a response at one level of organization will necessarily produce a consequential, detectable response at a higher level. It has been noted (paragraph 4) that for humans it is the risk of harm to the individual that must be minimized but that for the great majority of all other species it is the population that is valued and for whose protection appropriate radiation exposure controls could be implemented.

67. In the widest sense, a population may be defined as all members of a particular species. However, radiation fields arising from radioactive waste discharges generally show large spatial variability, not least because of the discrete nature of the input, and many members of a population may not receive any significant exposure from a particular source. The natural distributions of most species are inhomogeneous ("patchy") in response to the variable occurrence of the ranges of physical, chemical and biological conditions within which, in combination, the individuals of the species are able to survive, i.e. species are geographically restricted. Thus, it is probable that a more limited, and relevant, definition of a population could be developed for the purposes of environmental impact assessment.

68. In a similar context [I4] the following definition, developed for use in population ecology, has been suggested as a useful basis for discussion and progress: "A population is a biological unit for study, with a number of varying statistics (e.g. number, density, birth rate, death rate, sex ratio, age distribution), and which derives a biological meaning from the fact that some direct or indirect interactions among its members are more important than those between its members and members of other populations [B14]." Such a population would be (or could be) a self-sustaining unit, independent of other, geographically separate populations of the same species. Protection of this population would require that increased radiation exposure did not

significantly affect those attributes (i.e. those mentioned in the definition) on which the population depends for maintenance within the normal dynamic range of variation dictated by the interactions of natural physical, chemical and biological factors.

69. These attributes, which can be defined only for populations of organisms and which may be taken to be indicators of their health, are, nevertheless, integrations of properties that relate to individuals (in no sense is this meant to imply simple addition) (see, for example, [M8]). It may be concluded, therefore, that there can be no response to radiation at the population level (or, indeed, at any higher level of organization) without there being some clearly detectable effect in individual organisms (or at lower levels of organization). This clearly implies that the protection of the population (as the ultimate objective) may be achieved by restricting the exposure of individual organisms to the extent that there are no significant radiation effects on those processes that are necessary for the maintenance of the population. It is therefore necessary to consider the available information on the effects of radiation (mainly at chronic low dose rates) on the relevant processes in individual organisms, to consider how these responses might translate to an impact on the population, and to examine the results of studies of population responses to deliberate experimental irradiation or in environments contaminated by controlled or accidental releases.

70. Examination of the population attributes (paragraph 68) indicates that the individual responses to radiation exposure that are likely to be significant at the population level are mortality (affecting age distribution, death rate and density), fertility (birth rate), fecundity (birth rate, age distribution, number, density) and the induction of mutations (birth rate, death rate). The response of these individual functions to radiation exposure can be traced to events at the cellular level in specific tissues or organs. An extended summary discussion of the processes involved was given in the UNSCEAR 1982 Report ([U5], Annex J, "Non-stochastic effects of irradiation"). There is a substantial body of evidence indicating that the most radiosensitive sites are associated with the cell nucleus, specifically on the chromosomes, and, to a lesser extent, implying the additional involvement of damage to intracellular membranes. The end result is that the cells lose their reproductive potential. For most cell types, at moderate doses, death occurs when the cell attempts to divide; death does not, however, always occur at the first post-exposure division: at doses of a few gray, several division cycles may be successfully completed before it eventually occurs. It is also well known that radiosensitivity varies within the cell cycle, with the greatest sensitivities being apparent at mitosis and the

commencement of DNA synthesis [U5]. It follows that the greatest radiosensitivity is likely to be found in cell systems undergoing rapid cell division for either renewal (e.g. spermatogonia) or growth (e.g. plant meristems and the developing embryo); these examples clearly underlie the processes in individual organisms that are important for maintenance of the population.

71.   Fractionation, or protraction, of exposure to low-LET radiation increases the total dose required to produce a given degree of damage. At low dose rates, e.g. those that arise from the controlled discharge of radioactive effluents into the environment, the factors responsible for mitigating the response are the repair of sublethal damage, the repair of potentially lethal damage, the replacement of killed cells through proliferation of survivors. and other slow repair processes not related to cell repopulation [U5]. Although it is clear that repair, in the general sense, is possible, the existence and extent of residual injury is less clear. While such an outcome may be demonstrated for moderate, acute doses, it is not possible to extrapolate these results to predict the likely response to low-level exposures extending over a significant fraction of the lifetime of an organism. However, given that genetic mutations may be passed from generation to generation, it is reasonable to suppose that somatic mutations individually consistent with cell survival can occur and accumulate over time until the combined impact reduces cell viability.

72.   In addition to the intrinsic factors already mentioned as affecting radiosensitivity, account should also be taken of the extrinsic factors in a contaminated environment, such as season (rapid growth of plants in spring, lower metabolic activity in hibernating animals) and temperature (metabolism in poikilotherms). The response of an organism to increased radiation exposure is also likely to be modified if it is under stress from other sources, e.g. if there are non-radioactive contaminants in the environment or if it is at the limit of the normal biogeographical range for the species. It has been shown, for example, that exposure to DDT increases the retention of ^{137}Cs in rats; thus, in addition to the potential stress from pesticide exposure and its possible interaction with radiation, there is also a DDT-dependent increase in the radiation dose rate from internal sources [U13].

A. TERRESTRIAL PLANTS

73.   Radiation injury in plants expresses itself as abnormal shape or appearance, reduced growth or yield, loss of reproductive capacity, wilting and, at high exposures, death [S7]. Acute lethal doses to higher plants range from 10 to about 1.000 Gy (approximate mean absorbed doses averaged over the whole plant).

At the extremes of radiation sensitivity are plants such as mosses, lichens and unicellular species. which are highly resistant, and woody species, which are the most sensitive. In 12 species of woody plants assessed 10-14 months after exposure, the lethal doses were 8-96 Gy [S12]. The pine tree is the most sensitive, experiencing severe mortality following short-term absorbed doses of about 10 Gy [W11]. Growth is severely inhibited at 50%-60% of the lethal dose. Floral inhibition is observed at 40%-50% of the lethal dose and failure to set seed at 25%-35%. Thus, the capacity of the plant population to maintain itself can be damaged at acute doses lower than those required to cause mortality. Below 10% of the lethal dose, effects are not so apparent, and the plant maintains a normal appearance. These general observations from several herbaceous plant species are illustrated in Figure 1 [S8]. Another general relationship is that the dose that reduces survival by 10% (LD_{10}) is roughly equivalent to the dose that reduces the yield by 50% (YD_{50}) [S7].

74.   Because there is interest in the morphological changes produced by tissue damage and repair (rather than by mutation) that may be observed at contaminated sites, it is worthwhile mentioning the changes of this kind that have been observed in the laboratory. Leaves of plants formed during irradiation may show abnormalities such as dwarfing, asymmetrical development and distorted venation as well as changes in texture and thickening, which occurs at dose rates above 2.4 Gy d⁻¹ in *Antirrhinum sp.*, an annual flowering plant (the total dose was not given. but for a late spring-early summer growing season of 150 days, it could be up to 360 Gy) [S9]. Older leaves formed prior to irradiation are little changed by exposures. except in respect of texture: they commonly become dry. stiff. coarse and thickened [S9]. Effects in flowers include additional and abnormal buds and inflorescences at dose rates of 0.20-0.24 Gy d⁻¹ for 56 days (total dose 11-13 Gy). At a higher dose rate of 0.34 Gy d⁻¹ (19 Gy). vegetative rather than floral production was favoured and modified leaves occurred in floral positions [S9]. The needle length of pine trees becomes progressively shorter as acute exposures are increased above 0.5 Gy. Needle length is 10% of the control at 3 Gy in growing plants and at 4.5 Gy in dormant plants [S10].

75.   The sensitivity of plants to external x radiation or gamma radiation varies over a wide range and depends on both the stage of individual development and the phylogenetic category of the particular species. A comparative analysis [S45] of the radiosensitivity of higher plants in terms of the median lethal dose has shown that the range of variation exceeds two orders of magnitude in both the dormant seed phase (from 5 Gy in *Picea abies* to 2.000 Gy in *Raphanus sativus*, of 99 species studied) and the vegetative phase (from 4 Gy in *Pseudotsuga douglassi* to 580 Gy in *Capsella bursa-*

pastoris, of 64 species studied). Within a species, the radiation sensitivity (in terms of LD_{50}) may vary by a factor of more than 20 between different stages of development: the dry seed is most resistant and the gametogenic cells at meiotic prophase (the stage prior to the reduction division, when the duplicated chromosomes become visible in cell preparations) are most sensitive. Overall, the data suggest a close relationship between radioresistance and phylogenetic position [S45]. The more primitive species in the *Gymnospermae* are the most radiosensitive and show a relatively narrow range of LD_{50} (5-63 Gy for dormant seeds and 4.6-16 Gy for the vegetative phase). The more advanced groups show generally greater radioresistance but also greater variability, viz. the Monocotyledon class (10-350 Gy for dormant seeds and 9.6-52 Gy for the vegetative phase) and the evolutionary advanced Dicotyledon class (50-2,000 Gy and 4.8-580 Gy, respectively). It may be noted that a similar trend is apparent within individual families; thus the radiosensitivity of the more primitive *Vicia faba* (50 Gy and 9.8 Gy, respectively) is about 25 times greater than that of the more advanced and specialized *Trifolium repens* (1,250 Gy and 243 Gy, respectively) [S45].

76. Because phylogenetic development in higher plants is accompanied by the elimination of redundant hereditary information and a reduction in genome size [S45], it can be suggested that there should be some relationship between radiosensitivity and cell nucleus parameters. Sparrow and Miksche [S46] investigated the dependence of the tolerance of higher plants to chronic irradiation on cell nuclear volume and DNA content. Linear relationships were developed [S47] that could be used to predict radiosensitivity; Figure II shows the dependence of acute lethal dose on interphase chromosome volume for woody and herbaceous species of angiosperms [S12]. Experimental studies with 28 species demonstrated good agreement between the predicted radiosensitivity and that measured in the field, although it was accepted that significant variability could be introduced by the specific conditions of irradiation and uncontrolled environmental factors [S48].

1. Cultivated and pasture crops

77. Most studies of the effects of radiation on agricultural plants utilize the dose required to reduce the yield by 50% or 100% (YD_{50} or YD_{100}) as the measure of radiation response [S7]. The major cereal crops show the highest radiosensitivity when the ears (seed heads) are developing (YD_{50} = 4-16 Gy); rice is an exception (YD_{50} = 75 Gy). At other stages of development the YD_{50} is 20-60 Gy, with rice again being an exception at 160 Gy [F6]. A specific feature of the response of cereals at early stages of development is an increased production of shoots caused by damage to apical meristems and a loss of apical dominance; by producing

an increased numbers of ears, this may somewhat compensate for the loss of grain per ear [F6]. Legumes have YD_{50}s in the range 2-60 Gy for the vegetative stage and 1-4 Gy for flowering stages. Root crops are less sensitive, with YD_{50}s ranging from 14 Gy for onions to 90 Gy for radishes. Miscellaneous crops, in order of increasing resistance, are lettuce, pineapple, strawberry, squash, spinach, cabbage and tomato, which have YD_{50}s of 45-120 Gy [S7]. Oil seed crops are moderately insensitive, with YD_{50}s of 120-160 Gy [F6]. Pasture and forage crops, where the yield is related to vegetative mass rather than seed production, show the lowest sensitivities (YD_{50} = 150-230 Gy) [S7].

78. A number of factors have been shown to modify the radiation responses of crop plants. Adverse weather conditions (an increase in temperature and a reduction in relative humidity) after acute irradiation of spring wheat reduced the YD_{50} by a factor of 4; this effect was dependent on the cultivar studied. Substantial differences have been found between cultivars of spring wheat (172 varieties) in their growth after seed irradiation. Depending on the age of the plants at irradiation, the YD_{50} can vary ninefold, with the lowest radiosensitivity observed at the time when the multiple apical meristems are produced [F6].

79. The irradiation of seeds with acute doses up to 10 Gy stimulated early emergence in flax and early growth in spring wheat and barley under laboratory conditions. Field trials with irradiated wheat seed showed, however, that any hormetic effect was not consistently translated to an increase in grain yield [S24]. Substantial variations in hormetic responses were found both within and between cultivars and species, and an early stimulation of growth in response to irradiation did not necessarily yield benefits at harvest, mainly because other environmental variables had a much greater influence [S25, S26].

80. As a consequence of the observed relationships between radiosensitivity and nuclear and chromosomal parameters (paragraph 76 [S12, S46, S47, S48]), cultivars of the same species having similar nuclear volumes at interphase would be expected to display similar radiosensitivities. However, a comparative analysis of 172 cultivars of hexaploid wheat [S45] has revealed a 2-5-fold variation in the radiosensitivity of the seeds as indicated by changes in morphometric characters in the seedlings. It was shown that increased radiosensitivity correlated with the degree of biochemical polymorphism, i.e. the differentiation of the genome, in the cultivars [S45].

81. An investigation of the combined effects of irradiation and exposure to lead or cadmium salts on the induction of cytogenetic damage in the leaf meristem of spring barley has demonstrated statistically significant

interactive responses, primarily at low doses of the agents [G11]. Taken together with other data [U5], these results, which indicate non-linear biological responses to combined exposures to contaminants, emphasize the importance of taking account of multiple factors when assessing the potential impact of radiation on the environment.

2. Forest trees

82. Studies have been made of the effects of short-term (16-18 days) acute exposures of pine trees (in a mixed birch-pine forest) in the spring and the autumn. After the autumn exposure the LD$_{50}$ declined from 50 Gy after two years to 30 Gy after five years, when the expression of the acute radiation damage was essentially complete. After the spring exposure the trees showed greater radiosensitivity, with the LD$_{50}$ after two years being 30 Gy (greater by a factor of 1.7) [K5]. It was noted that suppression of normal growth at the main and all lateral shoots was a more sensitive indicator of damage than tree mortality [K5], and an absorbed dose of 15 Gy in the spring has been found to be 100% lethal within one year for cell proliferation in the apical meristem and for the processes of needle formation [S14]. The developing radiation response of pine trees following radiation exposure in the autumn has been covered in more detail by Karaban et al. [K6] (they also give data for birch trees, which show lower radiosensitivity). Although pine trees appear to be more resistant to acute radiation exposure in the autumn, when the period over which radiation mortality is expressed is extended to six years after irradiation, the LD$_{50}$ declines, approaching that shown by trees irradiated in the spring [M10].

83. In the first year following acute irradiation of pine trees in the spring, photosynthetic activity was reduced at all doses greater than 5 Gy in a dose-dependent fashion. Together with the loss of needles and growth points (apical and lateral meristems), this reduction decreased primary productivity. Although the photo-synthetic rate recovered to above control values in the second year for doses below 25 Gy and in the third year for doses below 50 Gy, this was insufficient to counterbalance the continuing loss of needles, and the overall productivity of the pine stands continued to decline. An acute exposure of 12.5 Gy depresses the net rate of photosynthesis in *Pinus taeda* and *Pinus elliottii* saplings in the period 4-20 days after irradiation and also depresses the rate of CO$_2$ evolution by stems [H8]. Birch trees show greater radioresistance [S27]. Using common end-points as indicators of damage, it has been shown that the responses of birch trees to a combination of acute irradiation (0-30 Gy) and sulphur dioxide exposure (7 mg m^{-3}) exceeded by a factor of 1.5 the sum of the responses to each of the factors separately, i.e. there was a synergistic interaction. It was concluded that

a complex mechanism underlay the combined response, because the final outcome depended on the order in which the two factors were applied and was not simply proportional to radiation dose [S16].

84. The maintenance of reproductive viability is necessary for the continued survival of the population and is certainly required for recovery if some radiation mortality occurs. The exposure of pine trees to doses in the range 0.3-22 Gy over 16 days in the autumn, when the early stages of pollen (the male gamete) formation takes place, has been shown to produce damage [T11]. In the spring following exposure, vegetative growth and the production of male cones occurred in trees that had received less than 12 Gy; in the following year all the experimental trees (in which doses had been up to 22 Gy) were productive. At doses greater than 3 Gy there was a substantial reduction in the fertility and viability of the pollen produced; viability recovered in the second year, and in the third year neither pollen fertility nor viability differed from control values at any dose studied (<22 Gy). At doses exceeding 0.7 Gy, morphological changes in the male cones, particularly in terms of size and pollen production rate, persisted into the second year, but by the third year recovery to control values was apparent at doses up to 12 Gy [T11]. From these data it cannot be determined whether radia-tion-induced changes in either male fertility or mortality would be limiting for a pine tree population.

85. In seeds of the ash tree (*Fraxinus americana*, a deciduous tree) water content was shown to signifi-cantly affect sensitivity to acute gamma irradiation. Seeds with a 3.4% water content showed a dose-dependent reduction in germination at doses greater than 100 Gy; storage of the irradiated seed for three years, or irradiation of three-year-old seed, further reduced germination rates but did not materially affect the pattern of dose dependence. The survival of the germinated seed to the production of seedlings was very low at 100 Gy and zero at higher doses. Raising the water content of the seeds to 40% increased the ger-mination rate at all doses up to 400 Gy and substantially increased survival at doses up to 200 Gy. The reason given for this lower radiosensitivity was that the life-time of the radiation-induced free radicals in moistened seeds is shorter than in dry seeds [H7]. More than 50% of one-year-old ash seedlings survived all doses up to 114 Gy when irradiated in the dormant state (late autumn), but exposure in the spring (at bud break) produced over 70% mortality at doses greater than 56 Gy. Over the next few years, the seedlings irradiated in the dormant state grew faster than those irradiated in the spring at all doses greater than 24 Gy, a dose at which the survival was also unaffected [H7].

86. Within a forest, the development and vigour of trees of a given species and age will vary owing to, for

example, light interception (crowding and/or shading by dominant individuals), exposure to wind and soil quality. For disadvantaged pine trees, these factors cause reduced height and poorly developed crowns. The less vigorous trees show greater radiosensitivity over three years following short-term (eight days) acute exposures up to 25 Gy; at 50 Gy there is no difference in the overall cumulative response [S15]. It has also been noted that acute radiation exposure renders pine trees more susceptible to attack by xylophagous (wood-eating) insects [S17]. These results have been confirmed by observations made after the Chernobyl accident, which found that poorly developed and less vigorous pine trees experienced the highest mortality [K22]. Such findings emphasize the importance of regarding radiation exposure as only one of the stresses likely to be experienced by plants.

87. Of course, a pine-birch forest does not consist only of pine and birch trees. In addition to other tree species, there is a herbaceous understorey component of the plant community adapted to the conditions of light intensity, moisture, exposure to wind etc. that prevail under the upper canopy formed by the leafy crowns of the pines and birches. Short-term acute exposure of this community in the autumn did not, in the following year, lead to the loss of any of the 188 herbaceous species present, nor was there any substantial change in species content [S18]. There were, however, delays (in eight species) and advances (in three) in the normal, seasonal developmental progression of some of the plants at absorbed doses in the range 30-50 Gy. In the same dose range, morphological variation from normal was noted in 16 species. These effects had disappeared by the second season after exposure. At doses greater than 100 Gy, and particularly in the second season after exposure, there were increases in the relative biomass and the relative number of species in the experimental plots. This response was ascribed to the increased illumination of the forest floor following the radiation-induced loss of the sensitive upper canopy (the pine trees) and is thus indirect rather than direct [S18]. In the first two years following acute exposure, the light and precipitation penetrating to the forest floor were not much different than in the control areas. By the fourth year there were well-defined, dose-dependent increases in light penetration (up to sixfold) and precipitation (up to twofold) owing to the loss of pine foliage and, at the highest doses, the loss of trees, causing an increase in grass biomass up to 3.5 times that in the control area. In the longer term, the extension of the grass ground cover in the irradiated area is expected to delay the recovery of the coniferous component of the forest by suppressing seed germination and seedling survival [S28].

88. These data on the response of a pine-birch forest community have been incorporated into a model to examine the long-term consequences of acute radiation

exposure [S19]. An exposure of 25 Gy (less than the LD_{50} for pine trees) substantially reduces both the number and productivity of the pines to minimum values at 15 years and 5 years, respectively, with slow recovery thereafter. The birch trees, being more radio-resistant, benefit from the reduced competition, and their number and productivity increase, although not enough to balance the loss of the pines. It was predicted that complete recovery to the pre-irradiation state would require more than 50 years following an acute exposure of 25 Gy [S19]. Such a result is not surprising, since it might be expected that a period longer than the average lifetime of the trees would be necessary to regain the stable climax state.

89. The outcome of these studies of the acute radiation response of the pine-birch forest is broadly comparable with the results of experiments on other plant communities, which were reviewed by [I4, W3].

90. The protraction of radiation exposures increases the total doses required to kill plants: for 11 woody species, the LD_{50} increased by a factor of 4.3 between the first and eighth years of continuous exposure [S13]. Gymnosperms (including the pine species), with a three-year LD_{50} of 46 Gy (5,200 µGy h^{-1}), remained much more radiosensitive than angiosperms, for which the three-year LD_{50} was 330 Gy (38,000 µGy h^{-1}) (see also [I4]). The LD_{50} for *Pinus rigida* increased to 74 Gy after 10 years of exposure (average dose rate of 1,250 µGy h^{-1}) [S29], and that for *Abies balsamea* increased to 110 Gy after 11 years of exposure (average dose rate of 1,500 µGy h^{-1}) and was still increasing [D12]. The numbers of balsam fir saplings in irradiated plots was, however, greater than in control plots at dose rates up to 250 µGy h^{-1} over 11 years [D12]. After 11 years of radiation exposure, the LD_{50} for plantation black ash (*Fraxinus nigra*) specimens planted as two-year-old saplings had increased to more than 900 Gy at a dose rate greater than 10,000 µGy h^{-1}. All naturally growing saplings of black ash were killed within five years at exposures of approximately 10,000 µGy h^{-1}, but two older specimens were surviving after 11 years at dose rates of 11,000 and 4,400 µGy h^{-1}; a green ash (*Fraxinus pennsylvanica*) was also surviving after 11 years at a dose rate of 5,900 µGy h^{-1} [D13]. Changes in canopy cover owing to mortality confirm the relative radiosensitivities of gymnosperms and angiosperms after 10 years of chronic irradiation [A4].

91. In mature trees, needle growth is reduced by exposures over 3-4 years at dose rates greater than 3,000 µGy h^{-1} to *Abies balsamea* [D12] and over 10 years at dose rates greater than 600 µGy h^{-1} to *Pinus rigida* [S29]. For one-year-old saplings of *Pinus sylvestris*, needle length was substantially reduced at 7,000 µGy h^{-1} over a single growing season but was increased marginally relative to the controls at lower

dose rates (<78 μGy h⁻¹); stem length was reduced at all dose rates, and significantly so at 7,000 μGy h⁻¹ [S30]. Two-year-old saplings of *Pinus banksiana* showed reduced stem growth at dose rates greater than 3,700 μGy h⁻¹ over the growing season [A5]. Above 500 μGy h⁻¹, the percentage of dead buds on *Abies balsamea* increased rapidly after five years of chronic exposure [D12]. Trunk growth in mature specimens of *Pinus rigida* can be reduced by dose rates in the range 400-2.000 μGy h⁻¹ over 9 years [W16] and at dose rates above 1.700 μGy h⁻¹ over 10 years [S29]. The radiation response can be intensified by poor growing conditions [S29, W16] or mitigated in trees with large healthy crowns free of competition [W16]. The cambium layer appears not to be terminally damaged by the irradiation [S29]. Net photosynthesis and CO_2 exchange in *Pinus rigida* are poor indicators of damage from chronic irradiation at dose rates less than 12.000 μGy h⁻¹ [B20].

92.   Chronic irradiation over a nine-year period reduced the number of mature seeds in *Pinus rigida* cones to 10% of the control values at a dose rate of about 1.500 μGy h⁻¹; no cones with mature seed were found at nine-year accumulated doses greater than 74 Gy [S29]. White spruce (*Picea glauca*) pollen has been found to have an acute LD_{50} between 60 and 90 Gy for seed yield and quality [R6].

3. Shrub species

93.   Radiation effects in shrub species can be illustrated by the results of two experiments, one short-term and the other longer term. In the short-term experiment, estimated beta doses from simulated fallout (⁹⁰Y-contaminated particles) to the developing flower buds of *Viguiera porteri* (approximate doses of 30 and 60 Gy at two experimental sites) induced a significant (2.5-fold and 3.6-fold, respectively) increase in aberrant flowers (extra petals, fasciated flowers, incomplete flowers) relative to control plants. Flowers with additional petals, which constituted 81% of the aberrant forms, produced 38% fewer seeds than normal controls, with no significant differences between the values for the two exposures. Seeds from flowers receiving 51 Gy showed a delay in the initiation of germination and a 38% reduction in germination rate; the plants from all irradiated seeds were, however, apparently normal [M11].

94.   After seven years of chronic exposure, 31 shrub species growing naturally in a mixed boreal forest showed 50% mortality at dose rates ranging from 800 to over 60,000 μGy h⁻¹; for another 24 species, individuals were surviving at the upper end of the range 10-14,000 μGy h⁻¹ but in only three cases had single individuals died at higher dose rates during the term of irradiation. Where sufficient data were available to estimate it, the 50% lethal dose rate decreased with irradiation time but for four species appeared to have

reached stable long-term survival values of 10.000 μGy h⁻¹ (*Alnus rugosa*) and 15,000 μGy h⁻¹ (*Corylus cornuta, Vaccinium angustifolium* and *Vaccinium myrtilloides*). Differences in radiosensitivity could be correlated to some extent with growth habit, e.g. average mature height, vegetative growth from shielded roots, deciduous or evergreen. Regardless of radiosensitivity, somatic effects were seen in most species, and those that carried aerial buds through the dormant season showed the greatest number of effects on leaves, flowers and branches [D14]. The capacity to regenerate from rhizomes shielded below ground and from the bases of damaged stems at ground level, growth habits that contribute to the relatively high apparent resistance to chronic irradiation in *Gaylussacia baccata* and *Vaccinium vacillans*, appear to derive from adaptations that improve survival in response to a natural stress, namely, forest fires following lightning strikes [B21].

4. Herb communities

95.   The predominant effect of chronic irradiation of herb communities is to reduce species numbers (diversity) as the dose rate increases, caused by the loss of more radiosensitive taxa [F7, F8, W12]. The threshold for change was less than 20,000 μGy h⁻¹ for old field communities [W12], and the dose rate to produce a 50% reduction in the diversity of a plains short grass community declined with extended exposure to about 1,000 μGy h⁻¹ [F8]. Nevertheless, a winter rye-weed community showed little change in composition below 40,000 μGy h⁻¹, although the rye standing crop was reduced at dose rates down to 10.000 μGy h⁻¹ and the production of fertile rye seed was reduced at dose rates above 1,000 μGy h⁻¹ [H9]. Although a number of herb species, one of which was *Erigeron canadensis*, survived at dose rates up to 130.000 μGy h⁻¹, the reproductive potential was damaged above 20,000 μGy h⁻¹ [W17]. In the longer term, chronic irradiation slows the rate of successional change [F7], which may depend, more than usually, on the adventitious arrival of viable seed from outside the irradiated area [W12]. Overall, however, herbaceous plants appear to be somewhat less radiosensitive than shrubs.

5. Lichens

96.   The diversity of lichen communities was reduced by chronic irradiation at a dose rate of 1.1 Gy h⁻¹ over one year. To survive at all at this dose rate (over 9,000 Gy a⁻¹) attests to the radioresistance of the lichen species. An apparent threshold dose rate of 125,000 μGy h⁻¹ was found for effects on the composition of lichen communities, although relative densities were changed at lower dose rates. Crustose forms were more resistant than either fruticose or foliose forms. However, these two less resistant forms grew more vigorously in

areas where the dose-rate range was 26,000-96,000 µGy h⁻¹ than in the control areas, although this may have been an indirect response to irradiation. caused by the collateral loss of the canopy as trees died off at these dose rates. The total density of lichens also declined with increasing dose rate, that is, the more radioresistant species did not colonize the areas vacated by the more radiosensitive species, but this may be a consequence of an observation period that was too short relative to the slow growth rates of these organisms [B22, W17].

6. Fungi

97. The appearance of the fruiting bodies of macrofungi in an irradiated forest appeared to depend more on the availability of an adequately nutritional substrate than on the radiation dose rate. After an eight-year exposure, mushrooms appeared at dose rates up to 100,000 µGy h⁻¹, puffballs at dose rates up to 325,000 µGy h⁻¹ and slime moulds at dose rates up to 770,000 µGy h⁻¹. These values are likely to be overestimates of the actual tolerances, as the fruiting bodies evolve from vegetative organisms, which are shielded to a greater or lesser extent below ground [H10].

7. Indirect effects

98. A number of indirect effects of radiation exposures may become apparent in plant communities. In white oaks subjected to chronic gamma irradiation there is a dose-rate-dependent gradient of damage in the form of delayed bud burst and an extended period of leafing-out, during which process the leaves are softer and of a yellowish hue, making them attractive to aphid parasites. High densities of aphids were observed in some years (1963 and 1969) but not in others; these densities were, moreover, significantly greater on trees at dose rates greater than 4,000 µGy h⁻¹ than on trees at lower dose rates. It was concluded that other factors, primarily meteorological, governed the presence and distribution of overwintering aphids capable of colonizing the weakened host plants [H11]. Insect damage did not, however, contribute to the loss of canopy cover in an irradiated boreal forest [D15].

99. The induction of direct damage in plants by long-term chronic irradiation is likely to have indirect effects on the local environment that may then interact with the radiation stress. Loss of vegetation and canopy cover have been shown to affect evapotranspiration in an irradiated oak-pine forest. Close to the radiation source, where virtually all the vegetation had been destroyed after 1.5 years of exposure at dose rates above 65,000 µGy h⁻¹, the cumulative evapotranspiration was about 85% of that in undamaged forest. The rates of evapotranspiration in the first half of the growing season

increased as the dose rate (and damage) decreased but became independent of irradiation during a drought in the second half of the season. The lowest dose rate at which there was a clear reduction in cumulative evapotranspiration was 4.800 µGy h⁻¹, at which point the tree canopy had been reduced by 50% but the shrub layer had remained almost undamaged [R7]. A chronic irradiation dose-rate gradient has also been shown to have significant effects on litter fall and total organic matter content of the soil. Even 12 years of exposure were not sufficient for these changes to reach a stable equilibrium. At that time, the total annual litter fall declined continuously at dose rates above 1,500 µGy h⁻¹, as did the standing crop of litter. The effect of this reduced organic input was apparent in the humus standing crop, which showed depletion. relative to control areas, at all dose rates above 1.500 µGy h⁻¹; there was, however, a noticeable secondary maximum in the humus standing crop at a dose rate of 10.000 µGy h⁻¹, corresponding to a zone where a relatively radioresistant sedge (*Carex pennsylvanica*) had become established [A6].

100. The devastated area close to the radiation source in the chronically irradiated oak-pine forest would be a site where normal successional colonization would be expected were it not for the continuing radiation exposure. It has been shown that there is a significant input of wind-borne seed of adventive herbs. which are typical colonizers of disturbed areas [W18], and many of these are sufficiently radioresistant to germinate and produce young plants that can grow to maturity in the high-exposure zone. The capacity of these plants to repopulate the disturbed area through reproduction was limited. however. by two factors. First. the radiation exposure severely reduced the normally profligate seed production: secondly, rabbits selectively grazed the plants that were conspicuous in the otherwise barren area. In combination, these factors retarded the rate of recolonization, and the more successful species were present as a few small clumps. These species were generally of normally small stature and thus more likely to receive adventitious shielding from standing and fallen dead trees or they were perennials with the advantage of shielded sub-surface buds. It was this latter characteristic that, together with relative radioresistance, favoured a species normally present at low density in the forest. The sedge (*Carex pennsylvanica*) spread vegetatively and vigorously from the few original individuals and seemed likely to dominate the successional pattern to the exclusion of the hardy herbs [W19].

8. Summary

101. Acute lethal radiation doses in plants have been noted to range from 10 to 1,000 Gy. For some forms of lower plants (mosses, lichen) the upper range of the acute lethal dose may be higher by an order of magni-

tude. In general, larger plants are more radiosensitive than smaller plants, with radiosensitivity decreasing in the order coniferous trees, deciduous trees, shrubs, herbaceous plants, lichen and fungi [W20, W21]. Non-lethal effects are noted at lower doses, for example severe growth inhibition is noted at 40%-50% of the lethal dose and failure to set seed at 25%-35% of the lethal dose. At doses less than 10% of the lethal dose, effects are less apparent, and plants maintain a normal appearance.

102. A range of sensitivities to radiation exposure is exhibited by the components of plants, ranging from dry seed (least sensitive) to apical meristems (most sensitive). Various crop plants show different reductions in yield following radiation exposures, with further modification caused by external factors (temperature, humidity). Young plants and plants disadvantaged by growing conditions may show increased radiosensitivity.

103. Plant species are also variably tolerant to chronic radiation exposures. In the more sensitive pine species, dose rates above 3.000 μGy h^{-1} over 3-4 years reduce needle growth; in one-year-old saplings, needle length was substantially reduced at 7,000 μGy h^{-1} over a single growing season. Trunk growth was reduced in mature pine trees by dose rates in the range 400-2,000 μGy h^{-1} over nine years. Delayed bud burst and an extended period of leafing-out was demonstrated in white oaks exposed to chronic gamma radiation. At dose rates greater than 4.000 μGy h^{-1} the trees were more susceptible to aphid infestation.

104. With effects of chronic irradiation of the most sensitive plants evident at 1.000-3,000 μGy h^{-1} and some specific changes noted at 400-2,000 μGy h^{-1}, it is suggested that chronic dose rates of less than 400 μGy h^{-1} (10 mGy d^{-1}) should have only slight effects in sensitive plants but would be unlikely to produce any significant deleterious effects in the wider range of plants present in natural plant communities.

B. TERRESTRIAL ANIMALS

1. Mammals

(a) Mortality

105. Radiation effects in mammals have been extensively studied in radiobiological experiments using laboratory animals (mice, rats, dogs and monkeys) and domestic livestock (pigs, sheep, goats, burros and cattle) [B8, B9]. Except in the case of exposure at unusually high doses, radiation damage or lethality in mammals is due to disturbances in the haematopoietic system and the gastro-intestinal mucosa. These cell self-renewal systems contain stem cells, differentiating cells and

functional end cells, with the stem cells being the most radiosensitive and thus the predominant influence on the radiation response. Symptoms become apparent when end cells are not replaced.

106. Mammals die from damage to the gastro-intestinal tract within the first 10 days following exposure to relatively high radiation doses (midline doses of 10-50 Gy) ([U3], Annex G, "Early effects in man of high doses of radiation"). The intestinal symptoms (diminished food and water intake, diarrhoea, loss of fluids and electrolytes, decreased intestinal absorption etc.) are a consequence of cell depletion in the intestinal lining due to damage to the survival and proliferative capacity of the crypt stem cells; the normal continuous flow of mature replacement cells from the crypts to the villi is interrupted. The time course of events is almost independent of dose in the range 10-50 Gy but differs widely between species. The differences in radiosensitivity between species result from the complex interactions of the sensitivities of the crypt stem cells and their rates of maturation and progression from the crypts to the villi. Approximate LD$_{50}$s for the gastro-intestinal syndrome are as follows: mouse, 12 Gy; rat, 11 Gy; rhesus monkey, 9 Gy; and dog. 8 Gy [V3]. The values were determined for particular strains of these species, and wide variations are known to occur. However, the approximate species differences (due to the complexities noted above) might arise from variations in intestinal morphology that have to do with adaptations to particular diets, i.e. herbivory, omnivory and carnivory [V3].

107. Mammals die from bone marrow failure (the haematopoietic syndrome) within weeks following whole-body radiation exposures giving midline doses of 1.6-10 Gy. The ultimate cause of death in different species is granulocytopenia, thrombocytopenia or lymphocytopenia [U3]. These conditions are again the result of complex interactions of differential radiosensitivities and cell cycle and maturation kinetics (including stem cell survival) in the various cell lineages. Nevertheless, there is an apparent inverse relationship between the LD$_{50/30}$ and the weight of the animal [U3], and approximate values are 6-10 Gy for small mammals and 1.6-2.5 Gy for larger ones [B8]. For domesticated animals (cattle, sheep, goats, pigs, burros and horses) the LD$_{50}$ lies in the range 1.2-3.9 Gy (midline dose) [K10]. In addition to the variation in LD$_{50/30}$, the rate of change of survival with dose also varies between species (Figure III), with some evidence that at lower sublethal doses there might be less variation between species. There has been considerable effort (e.g. [B10]) to reanalyse experimental results to quantify more exactly species-relevant LD$_{50}$s. There are, however, many differences in response between separate strains of the same species, dosimetric uncertainties and differences in conditions of maintenance

and care of the animals before and after exposures that can affect response. All species are markedly influenced by the presence of pathogenic bacteria in the animal, which reduces survival times and, thus, $LD_{50}s$. The $LD_{50/30}$ for mice housed in a germ-free system has been found to be about 2 Gy higher than that for mice housed in a conventional system [L4].

108. Protraction of a given total exposure generally reduces the extent of injury, as it allows two distinct processes to intervene. First, sublethal damage is reparable at the cellular level, which is particularly important for exposures to low-LET radiation. Secondly, cell proliferation can replace lethally damaged cells and maintain the cell population at a new level, which is determined by the dynamic interaction between the dose rate and the rate of cell death and by the total reserve proliferative capacity.

109. At reduced dose rates (protraction of a given total dose) of low-LET radiation, all species show a gradual increase in LD_{50}: higher total doses are tolerated. This changing response has been attributed to the increasingly effective influence of cellular repair of sublethal damage at the lower dose rates. As the dose rate is further reduced, a sharply increasing trend in the values for median lethal dose is apparent for mice, pigs, dogs, goats and sheep; the approximate threshold dose rates for this change in response correspond to the accumulation of the LD_{50} in periods ranging from 0.2 days (mouse) to 9 days (goat). This rapid change in LD_{50} with dose rate has been interpreted as the consequence of a shifting balance in the dynamic interaction between dose-rate-dependent cell loss and cell proliferation and maturation kinetics in the haemato-poietic system: the latter processes are under homeostatic control, i.e. their rate constants can alter in response to radiation-induced cell loss. The data for the burro (donkey) and the primates do not show any sharp increase in median lethal dose at dose rates down to 8.300 $\mu Gy\ h^{-1}$ (LD_{50} in 18 days) and 5,400 $\mu Gy\ h^{-1}$ (LD_{50} in 60 days), respectively. There do not appear to be any data for $LD_{50}s$ at dose rates less than 4,000 $\mu Gy\ h^{-1}$ or for periods of exposure exceeding 60 days, although studies have been made outside these limits for other purposes.

110. For the survivors of acute doses in the $LD_{50/30}$ range and for organisms exposed to lower acute doses, it has frequently been suggested that there would nevertheless be consequential life-span reduction from non-specific causes, e.g. accelerated ageing or the earlier onset of the pathological conditions responsible for mortality in the unirradiated population. The available data have been reviewed ([U5], Annex K, "Radiation-induced life shortening") and the conclusion reached that there is little evidence that an observed reduction in life-span arises from anything other than an accelerated

onset or increased incidence of neoplastic disease; this conclusion has been confirmed by the Committee's most recent consideration of the available data ([U2], Annex F, "Influence of dose and dose rate on stochastic effects of radiation"). Analysis of the results for mice exposed to x rays or gamma radiation, including those results that show an extension of life-span at doses less than 2 Gy, indicates that the percentage of life-shortening could be most simply expressed by a linear, non-threshold dependence on dose. The slope of the dose-effect relationship yielded a 5% reduction in life-span per gray down to the lowest doses. The much more limited data available for the rat, dog and hamster show that their responses are not very different from that of the mouse at doses below 6 Gy, and in this dose range a factor of 5 would cover the overall intra- and interspecies variability.

111. For chronic lifetime exposure of mice to low-LET radiation at dose rates where haematopoietic damage (bone marrow failure) does not occur, the percentage life-shortening shows an essentially linear response with dose in the range 0-45 Gy (dose rates less than 3,800 $\mu Gy\ h^{-1}$). Thus, chronic exposure is substantially less effective, by a factor of 7, than acute exposure. Again, there is no evidence that anything other than radiation-induced cancer is responsible for the increased loss of life-span relative to the controls [U5]. At a dose rate of 420 $\mu Gy\ h^{-1}$ (a lifetime dose to mice of approximately 7 Gy) the average loss of life-span amounts to 5% (equivalent, for example, to 20% of the population dying from the consequences of irradiation at 75% of the normal mean age). Extrapolation of these specific data on the mouse to the radiation exposure experienced by a population in a contaminated "natural" environment is not straight-forward:

(a)   natural mortality in a wild population of mice subjected to competition (intra- and interspecies) and predation is unlikely to have the same age dependence as observed for control populations in the laboratory; thus a late, incremental incidence of tumours is unlikely to have the same overall effect on life-span;

(b)   extension to other species, especially those with longer average life-spans is unjustified; and

(c)   if the fertility of the older members of the population represents a significant buffer capacity, then earlier than normal mortality due to radiation exposure might have a disproportionate effect on the population at times of stress [F4].

The very limited data available for other species (guinea-pigs, goats and dogs) are insufficient to allow meaningful comparison; they do, however, identify tumour induction as a consequence of radiation exposure [U5].

112. Acute or chronic whole-body exposure to high-LET radiation (fast neutrons) has been shown to be more

effective than the corresponding low-LET exposure regime in producing a reduction in life-span. The relative biological effectiveness (RBE) is greater, as is to be expected. for chronic irradiation than for acute exposure [U5]. These results underline the importance of the potential internal exposure from contaminant radionuclides that decay by alpha emission. There is a body of evidence to show that the injection of radioisotopes of bone-seeking elements (Sr, Pu, Th, Ra and Am) into rats and dogs results in life-shortening due to the induction of bone tumours. with the alpha emitters being more effective per unit of absorbed dose than the beta-emitting ^{90}Sr-^{90}Y (although the precise site of deposition is also a modifying factor). In apparent sharp contrast. ^{239}Pu chronically administered to rats results in very little change in the life-span compared with the controls, although there is an increased incidence of marrow damage, leukaemia and bone cancer [U5]. It is clearly important to identify the appropriate target tissues and organs and to know the exposure from internal or external contaminant radionuclides when attempting to assess the potential impact on the environment.

113. The influence of hibernation on the response of mammals to acute irradiation is complex and depends on the status of the animals before, during and after exposure. For the ground squirrel (*Citellus tridecemlineatus*), hibernation induced by maintenance at 5°C for 2-5 days prior to and during irradiation consistently provided some protective effect in terms of increased mean survival time and percentage survival at 30 days. The degree of protection could be modified by the post-irradiation maintenance conditions, with immediate arousal to an active state improving survival relative to animals maintained in hibernation for an additional 20-30 days [B32, J4, M21]. The conditions employed to induce effective hibernation also modified the degree of protection. Maintenance for 2-5 days before exposure at 13°C, or undisturbed. but closely confined, at 23°C both improved survival relative to animals maintained at 5°C, which, in turn, survived better than animals irradiated while normally active [B32]. Split dose experiments have shown that repair of radiation damage continues during hibernation at 5°C [J4].

114. Some limited data are available indicating that the acute effects of radiation in small mammals are less severe when experiments are carried out under controlled conditions in the laboratory rather than in the natural environment, where additional stresses are present. Although the LD_{50} obtained for female cotton rats was similar for a laboratory study and for irradiated animals released into quasi-natural conditions in large field enclosures, the mortality of rats exposed to doses less than the LD_{50} was greater under field conditions [P7]. Exposure of house mice and field mice at the $LD_{50/30}$ established under laboratory conditions produced the same and lower survivals, respectively,

when the animals were released to field enclosures [G4]. In laboratory studies. the $LD_{50/30}$ for the female house mouse was found to be greater when the treated animals were housed singly than when the females were kept in groups of five [D6]. Although the data were not entirely conclusive. chronic irradiation of pocket mice in an eight-hectare enclosure at approximately 420 µGy h^{-1} appeared to marginally reduce their survival relative to otherwise similar controls. The survival of three field populations (one irradiated and two control) was less than that of an irradiated laboratory group, which in turn was less than that of the laboratory controls. In addition. under laboratory conditions. the irradiated and control male mice survived longer, on average. than the corresponding females [F3].

115. It has to be accepted that the number of mammal species that has been or, indeed, is likely to be studied is an extremely limited, and probably atypical. selection from those extant. Nevertheless the available data, even taking account of substantial interspecies variability, provide very little evidence that dose rates below 400 µGy h^{-1} (approximately 10 mGy d^{-1}) to the most exposed members of the population would seriously affect their mortality (and. thus, the death rate in populations of these species) from either deterministic or stochastic responses.

(b) Reproductive effects

116. The effects of radiation on reproduction have also been much studied. with most of the results suggesting that natality is a more radiosensitive parameter than mortality in species other than man and, therefore, of more relevance in an environmental context. The minimum dose required to depress reproduction rates may be less than 10% of the dose required to produce direct mortality [W5]. Reproductive effects may be apparent at doses that cause no other observable response in the animal. For example. the $LD_{50/30}$ in mice is 6-12 Gy, depending on strain and species, but an acute dose of 0.08 Gy produced a 50% reduction in the number of oocytes at their most sensitive stage in newborn mice [O3]. Because more immature oocytes are normally present than will be utilized for reproduction, the 50% reduction does not translate to a 50% decline in fecundity; it could, however, conceivably lead to a minor shortening of the reproductive span [M5]. In male mice, by contrast, an acute dose of 10 Gy may only temporarily impair fertility in adults, although in young mice (3-5 days of age), it can induce permanent sterility ([U6], Annex H, "Genetic effects of radiation"). The differences in response are a consequence of the quite different processes of gametogenesis in the two sexes [D2]. There are also substantial differences in response to the same dose between species, particularly for oogenesis, with most species showing greater radioresistance than the mouse [C6].

117. Chronic irradiation probably affects oocyte viability by different mechanisms than acute irradiation at the particularly sensitive stages of oogenesis, as discussed in the preceding paragraph. The differences arise from both the timing of the oogonial cell division and the limited size of the stem cell pool. In a female mammal exposed throughout its life, from conception to reproductive maturity, the greater part of the total dose is likely to be accumulated by oocytes in the resting. immature stage rather than in any particularly sensitive stages of oogonial cell proliferation. The effects of chronic exposure are more relevant to environmental concerns. The number of immature oocytes in mice can be reduced by 50% by a dose rate of 130 μGy h^{-1} from tritiated water during the period 19-33 days after conception [D5]. For irradiation from tritiated water from conception to 14 days after birth, the median lethal dose rate for mouse primary oocytes was 180 μGy h^{-1}, and an RBE of about 2 (relative to ^{60}Co gamma rays) was estimated. At lower dose rates (and lesser effects) the RBE increased to 3 [D16, D18]. In 11-week-old female mice chronically exposed to either beta radiation from tritiated water or ^{60}Co gamma rays for 10 days with similar dose-rate profiles. the number of primary oocytes declined exponentially with cumulated dose. A total dose of 0.16 Gy from tritium reduced the number of primary oocytes by 80%, and an RBE of 2 was estimated relative to ^{60}Co gamma rays [D20]. The developing oocytes of the squirrel monkey. exposed to tritium from conception to birth, showed an even higher radiosensitivity. with a median lethal dose rate of 42 μGy h^{-1}; neonates that had been exposed to 420 μGy h^{-1} were found to be sterile [D18]. Similar results have been observed for tritium exposure in monkeys in the last third of pregnancy [C6] and for ^{90}Sr-^{90}Y exposure of mice [R2]. The total accumulated doses were 40-200 mGy. A total dose of 900 mGy from ^{90}Sr-^{90}Y reduced fertility in the mouse [R2], and a dose rate of 3,300 μGy h^{-1} during the period 20-40 days after conception produced complete sterility [R5].

118. The effects of continuous tritium exposure from either tritiated food or tritiated water on primary oocyte production in first-generation (F$_1$) and second-generation (F$_2$) offspring have been studied in rats. The numbers of primary oocytes were determined in 21-day-old F$_1$ and F$_2$ offspring that had effectively been exposed since conception at a dose rate of 77 μGy h^{-1}. Although oocyte numbers were reduced to similar extents under both exposure regimes and in both F$_1$ and F$_2$ offspring, the differences from control values were not statistically significant [P18]. In a second experiment, the reduction in oocyte numbers in 21-day-old F$_1$ offspring was significant at a dose rate of 610 μGy h^{-1} derived from tritium in drinking water and at 200 μGy h^{-1} derived from tritium in food (organically bound tritium) [P14].

119. For mature female rats, dose rates of less than 4,000 μGy h^{-1} from ^{60}Co gamma radiation did not affect reproductive ability [B15]. Female mice exposed at 800 μGy h^{-1} from conception became sterile at 25 weeks of age. and those exposed at 4,000 μGy h^{-1} were sterile at 7 weeks of age [B15]. In eight-week-old female mice exposed to either beta radiation from tritiated water or ^{60}Co gamma rays for 10 days with similar cumulative dose profiles and mated after a further 21 days. pre-implantation losses increased linearly with dose. This loss became significantly greater than for the control at a total dose of 0.31 Gy for tritium exposure, and an RBE of 2.5 was estimated relative to ^{60}Co gamma rays [Z7]. In young mice, ^{210}Po injected as PoCl$_3$ in isotonic saline was found to be preferentially taken up into the ovary and uterus. A dose of 37 mBq g^{-1} body weight was estimated to deliver an average absorbed dose to the ovary of about 100 μGy over a period of four days, at which time there was a significant reduction in primary oocyte survival (79% of control). Comparisons with other published studies of the effects of gamma radiation [O3] provided an estimate of over 370 for the RBE. although lower values were found for higher absorbed doses of alpha radiation and greater effects [S31].

120. The testes are also sensitive to chronic exposure, although the continuous nature of the spermatogenic process allows sperm production to be maintained under irradiation. often at equilibrium levels lower than in control animals. A dose rate of 570 μGy h^{-1} over seven months resulted in the stabilization of the spermatogonial cell population in the mouse at 80% of control values. However, a second study showed that at a dose rate of 750 μGy h^{-1} over 15 weeks, the stem cell population maintained control values, although cell death was occurring [C6. U6]. In male mice a mean absorbed dose rate of 36 μGy h^{-1} to the testes from ^{239}Pu alpha radiation for 5-8 months reduced testes mass and sperm output by approximately 10% and 8%, respectively: an RBE of 10-15 was estimated relative to the effects of ^{60}Co gamma rays [S50]. In the rat, a dose rate of 580 μGy h^{-1} reduced the testes weight by 7% relative to the controls after 75 days of exposure. corresponding to a disruption of the process of spermatogenesis. Cessation of exposure allowed recovery to near-normal values [L8]. At a dose rate of 800 μGy h^{-1} from conception. the fertility of male mice was not affected over the 33-week duration of the experiment [B15]. In beagle dogs, a dose rate of 180 μGy h^{-1} resulted in progressive cell depletion and sterility within a few months, but exposure at 36 μGy h^{-1} over the whole life elicited no response [C6]. No more sensitive terrestrial mammal has been identified [M5. U4], although it is accepted that few of the total number of species have been studied.

121. Continuous irradiation at 500-1,000 μGy h^{-1} for at least 10 generations did not affect the fertility of pairs

of mice from four different strains, as indicated by the average size of the first litter [S11]. This result is not necessarily inconsistent with the data discussed in paragraph 116, because of the lack, noted there, of a direct relationship between oocyte reduction and fecundity. In the environmental context, it is the effect of radiation on the normal distribution of lifetime reproductive capacities that is the relevant factor. Nevertheless, it seems unlikely that a dose rate of 100 μGy h⁻¹ to the most highly exposed individuals in populations of wild mammals and, by inference, a lower mean dose rate to the whole population will have any damaging consequences for the fecundity of populations as a whole.

(c) Effects on the developing embryo

122. As is to be expected from the degree of cell proliferation and differentiation in progress, the developing embryo has been found to be rather sensitive to the effects of radiation. Damage at this stage of the life cycle can reduce fecundity and postnatal survival and, hence, the potential recruitment to the population. The available data, the majority of which relate to the mouse, were thoroughly reviewed in earlier UNSCEAR Reports [U4, U6] and in a report produced by the Commission on Radiological Protection of the Federal Republic of Germany [C22], so only a brief summary will be included here. It is conventional to divide the temporal development of the mammalian conceptus into three main periods corresponding to significant differences in the processes in progress in the embryo: the pre-implantation period, which encompasses the development of the blastocyst; the period of major organogenesis; and the fetal period. These divisions are also convenient for the discussion of the responses to radiation, which differ in nature and degree between the periods, as can be seen in Figures IV and V, which summarize the situation for the mouse [C22].

123. The most conspicuous effect of acute irradiation during the pre-implantation period is the early death of the embryo: post-implantation and postnatal death are less important sequelae of such exposure. Similarly, growth reduction and malformations are a less frequent consequence of pre-implantation exposure [C22, P13]. Even within the pre-implantation period (up to the late blastocyst stage), however, there are variations in response. Exposure of single-cell mouse embryos to 2 Gy produced mainly pre-implantation deaths with many fewer resorptions (post-implantation deaths) and fetal deaths, while similar exposure at the stage of 32-64 cells only produced resorptions [M18]. The mortality of these very early embryos correlates well with the presence of radiation-induced chromosome aberrations in the blastomeres [C22]. Doses as low as 0.2 Gy to mouse embryos in vitro have been reported to produce mortality at the most radiosensitive time, prior to DNA

synthesis in the pronuclear stage. It has been shown that there is a tenfold variation in radiosensitivity through the different stages of the mitotic cycle and between mitotic cycles in the very early development stages. The irradiation also disturbs the normal kinetics of progression through the cell cycle. In vivo studies with rodents have provided qualitative confirmation of these findings. Notwithstanding the general conclusions given above, doses as low as 0.05 Gy to the rat in the pre-implantation period in one experimental series resulted only in fetal mortality, whereas a higher dose (0.1 Gy) produced both embryonic and fetal mortality, with the latter being greater [U4, U6].

124. There appear to have been rather few studies of the effects of chronic irradiation during the pre-implantation period. The impact of chronic exposure from internal radiation (tritium as tritiated water) and external ⁶⁰Co gamma radiation on pre-implantation mouse embryos maintained in culture has been investigated [Y1]. The radiation exposures commenced at the pronuclear, early two-cell and late two-cell stages, and survival to the expanded blastocyst stage was used as the criterion of damage. Exposure from the pronuclear stage showed the greatest radiosensitivity, with 50% survival at an estimated tritium concentration of 4.4 MBq ml⁻¹ (corresponding to an estimated dose rate of 10.000 μGy h⁻¹). The range of tritium concentrations used does not allow assessing the lowest dose rate to produce an effect. The estimated RBE was not significantly different from unity. Other experiments [U4] have shown that tritiated thymidine produces damage at much lower activity concentrations, underlining the importance for the purposes of dosimetry of information on the relative spatial disposition of the radiation source and the putative target for the biological effect being examined (see also [C22]). The available data are insufficient to assess whether protraction of the exposure reduces the effectiveness of a given dose (the sparing effect), but it should be noted that the restricted time available for the accumulation of the dose (pre-implantation periods of 5-17 days [U4]) also limits the potential total dose to this embryonic stage at low dose rates.

125. It has been concluded that the most characteristic (but by no means the only) response to acute irradiation during the period of major organogenesis is the induction of malformations [C22, U4, U6]. These teratogenic effects develop during embryonic and fetal growth and, if not of lethal consequence, persist into the postnatal period. The time of maximum radiosensitivity for the induction of malformation in a particular organ (although there may be growth disturbances in other organs or the whole body as well) coincides with the surge in cell proliferation in the relevant precursor tissue. This is a general finding for the animal species that have been studied (mouse, rat, hamster, gerbil,

rabbit, monkey, dog, sheep, cattle, pig and marmoset), but the response to a given acute dose strongly depends on the species and its developmental stage at the time of exposure. The dose ranges associated with observable malformations in several species are summarized in Table 8.

126.   The dose-response relationships for the induction of malformations are sigmoid and consistent with a deterministic mechanism. Protraction of a given dose reduces the incidence of each class of malformation below that expected as a consequence of the same dose given at the most sensitive stage. Careful experiments with a particular strain of mouse have shown that doses as low as 0.05 Gy at the most sensitive stage of embryonic development (seven days after conception) can triple the incidence of gross skeletal malformations [U4]. Malformations of the central nervous system have been particularly studied, and there is little evidence for any pathological responses at doses below 0.1 Gy (in mice, rats and primates) [U4]. Sterility was induced in 36% of female rats exposed to 0.5 Gy as embryos at nine days after conception [U4]. For the relatively less frequent effect of embryo lethality, rodents show an LD_{50} of less than 1.5 Gy in the early stages of organogenesis, after which there is an increase with development until the fetal stages, which yield values little different from the postnatal LD_{50}.

127.   There appear to be no studies of the effects of chronic irradiation specifically during major organogenesis. A single study of fractionated exposure in rats (four daily doses of 0.01 Gy at 6-9 days after conception) showed a significantly impaired learning capacity at four months of age in behavioural tests in which performance requirements were gradually increased [B19]. Mouse embryos have been exposed to beta radiation arising from a single injection of tritiated water to the mother at 12.5 days after conception, with the male offspring being given behavioural tests from 21 days of age. There was evidence of changes in behaviour, attributed to irradiation, at cumulative doses of 0.1 Gy and greater [W26]. Such a response could lead to reduced viability in the natural environment.

128.   During the fetal period of development the main consequences of acute radiation exposure appear to be mortality, with LD_{50} values similar to those for postnatal exposure, and disturbances of growth. Teratogenic effects become increasingly difficult to detect, although damage to the eye, the central nervous system and the gonads appear to be the main responses [C22, U4, U6]. Acute exposures up to 2 Gy to developing rats between six days after conception and eight days after birth induced no deficit in the immune response up to 12 weeks of age [P8]. Exposure of rats to 1.5 Gy (10-12 days after conception) caused germ cell killing; testis weight was reduced slightly by exposure on day 15 and

became progressively more pronounced at later times, with the peak reduction for exposures between 18 days after conception and 3 days after birth [C15]. The weight deficit was due to the loss of spermatogenic cells in the tubules, but the endocrine function of the testis was unaffected. In another study, 2.5 Gy delivered to gravid rats on day 20 after conception caused sterility in 70% of the female offspring [K7].

129.   Fractionated x-ray exposures (16 exposures of 0.12 Gy, twice weekly for eight weeks) produced severe damage to the ovaries of female bonnet monkeys when the irradiation was given 77-133 days after conception but produced very little response when given 48-104 days after conception [A3]. A single injection of tritiated water to the females on day 13 after conception was used to deliver total estimated doses of 1.3, 0.6 and 0.2 Gy to fetal mice. At two months of age, all male and 40% of female offspring were infertile at the highest dose, and at three months female fertility was declining, while there were signs of some recovery in the males [T16]. It was concluded that doses in excess of 0.5 Gy were required to affect fertility [U4]. A dose of 3 Gy delivered to female rats at 18,000 μGy h^{-1} over the period 13-20 days after conception affected ovarian development (smaller ovaries, absence of follicles and corpora lutea) in offspring, whereas an acute dose of the same magnitude on day 18 after conception produced no response [E4].

130.   Chronic radiation exposure throughout gestation is the more likely possibility in contaminated natural environments. Exposure of pig embryos to ^{60}Co gamma rays at 420 μGy h^{-1} for 108 days during gestation (112 days duration) reduced the weight of the gonads in the offspring, and at 1,300 μGy h^{-1} both female and male offspring were rendered sterile. Histological examination of the gonads showed reduced numbers of germ cells at a dose rate of 210 μGy h^{-1} [E5]. Rat embryos exposed to tritium (as tritiated water) throughout development showed significant reductions in testis weight at an estimated dose rate of 1,300 μGy h^{-1}, and the ovaries were much reduced or absent at 2,500 μGy h^{-1} [C9]. Exposure of female mouse embryos to tritium (as tritiated water) from conception to 14 days of age showed dose-dependent reductions in the numbers of oocytes, with a significant reduction (13%) in primary oocytes per ovary at an estimated dose rate of 10 μGy h^{-1} and a reduction to two thirds of the control value at 100 μGy h^{-1} [D7]. A later series of experiments produced similar results, with the additional finding that the largest part of the response was a consequence of the prenatal exposure and that the tritium beta particles were more effective than ^{60}Co gamma rays in producing the damage (an RBE of 1.9; see also paragraphs 117-119) [D8]. After irradiation at 420 μGy h^{-1} during gestation, germ cell numbers were reduced relative to the controls in the following species

[U4]: pig (1.1 Gy total dose), 1% (male), 5% (female): guinea-pig (0.62 Gy total dose), 41% (male), 71% (female): rat (0.21 Gy total dose), 50% (male), 90% (female): mouse (0.20 Gy total dose), 71% (male), 87% (female). As noted above, these reductions in germ cell numbers would not necessarily produce a corresponding reduction in fecundity.

131. A significant reduction in the birth weight (stunting) of female and male offspring was observed at 2,500 μGy h^{-1} from tritium exposure (as tritiated water) during gestation in rats, and the brain weight was significantly reduced at 1,300 μGy h^{-1} [C9]. A later experimental series [C10] showed significant reduction in brain weight in young born to mothers that had been exposed to 130 μGy h^{-1} from tritium (3.7 10^4 Bq ml^{-1} as tritiated water in drinking water) since their own conception, through 100 days of postnatal life and the 21 days of pregnancy. Simultaneous exposures to graded concentrations of lead in drinking water showed that the interactive effects of these two contaminants were less than additive.

132. The relative effectiveness of tritium beta particles and external ^{137}Cs gamma irradiation in reducing cerebral development in rat embryos has been investigated [U4]. The range of estimated dose rates from tritium was 210-1,600 μGy h^{-1}, and cerebral mass at birth was negatively correlated with the cumulative dose during gestation. The ^{137}Cs gamma radiation was less effective in producing a response (RBE = 2.1 at 500 μGy h^{-1} from ^{3}H). A dose rate of 1,300 μGy h^{-1} (^{60}Co gamma rays) reduced the peri- and postnatal weight of the brain in pigs [U6]. No studies appear to have been made to determine whether such morphological responses have an associated impairment of function that might be significant in the natural environment: the few data noted in paragraph 126 are, however, suggestive of the potential importance of this response.

133. Damage to the developing mammalian embryo appears to be a potentially significant criterion of impact from contaminant radionuclides in the natural environment. Dose rates of 420 μGy h^{-1} throughout gestation produce readily detectable reductions in the populations of germ cells in the developing gonads of a number of mammalian species, and the lowest dose rate at which damage has been seen is 10 μGy h^{-1} from tritium (as tritiated water in drinking water) incorporated in female mouse embryos. In addition, dose rates of the order of 420 μGy h^{-1} induce reductions in neonatal brain weight, although the significance of this deficit in functional or behavioural terms is unknown. The wider significance of these responses at the population level has not been investigated. Even recognizing that only very limited data are available, it may be concluded that maximum dose rates of 100 μGy h^{-1} (2.4 mGy d^{-1}) to gravid members of a mamma-

lian population are unlikely to have any consequences for the population as a whole from induction of damage in the developing embryos.

(d) Somatic and hereditary mutational effects

134. The substantial literature on the induction of mutations (genetic damage) in animals by radiation exposure has been thoroughly reviewed in previous reports of the Committee [U2, U3, U4, U5, U6]. The term mutation encompasses a wide range of damage to the genetic material, and its consequences depend on the nature of this damage and also on the site, i.e. in somatic tissue or in germ cells. In many cases, the damage is severe and incompatible with the survival of the cell as an entity with proliferative potential; the damaged cells die either immediately or after rather few division cycles. This form of damage is largely subsumed in the discussion above of mortality, effects on fertility and effects on the developing embryo. There remain the lesser forms of damage that are not inconsistent with the survival and subsequent long-term proliferation of the cells but which, nevertheless, have a potential to harm the host organism. In somatic cells, the consequence of concern is the induction of cancer [U4]; in germ cells, it is necessary to assess the potential impact on the fitness of offspring in the next and subsequent generations of animals and the potential implications for the population [U3].

135. It has already been noted (paragraph 110) that the chronic exposure of animals throughout their life reduces life-span, mainly by the induction of malignant tumours, although it was concluded that at dose rates less than 420 μGy h^{-1} (10 mGy d^{-1}) the effect would be slight. The data for animals (mainly rats and mice) reviewed in the UNSCEAR 1986 Report [U4] show that there is a wide variation in the sensitivity to tumour induction depending on the tissue and on the species. Nevertheless, there is a consensus that at the dose rates likely to prevail in environments contaminated as a consequence of controlled discharges of radioactive wastes, a linear dose-response relationship would be appropriate for all qualities of radiation. This limited database makes extrapolation to other species, particularly those that are long-lived, uncertain. A review of data on the effects of chronic exposure to ^{90}Sr-^{90}Y beta radiation in beagle dogs showed that cumulative chronic exposures up to about 7 Gy, either from a single injection to young adults or from a body burden accumulated via ingestion from the fetal stage (uptake from the mother) to 540 days of age, did not increase the incidence of cancer in either the bones or the adjacent soft tissues, and there was no significant reduction in life-span compared with controls; this corresponds to a mean dose rate of approximately 60 μGy h^{-1}. There was some evidence from effects at higher total doses that the age at commencement of exposure and the cumulative

dose profile influenced the outcome: for example, the dogs initially exposed *in utero* showed haematopoietic effects and soft tissue cancers that were not apparent in dogs commencing radiation exposure as young adults. Parallel experiments with internal exposure from ^{226}Ra indicated that at the lowest doses causing a radiation-induced increase in bone sarcoma, the RBE (alpha radiation compared with beta radiation) was about 25 [N4]. In addition, the ^{90}Sr body burdens that did not induce cancer also had no effect on the fertility or fecundity of the dogs. As pointed out above, effects on reproduction and on the developing embryo are more sensitive criteria, which leads to the requirement for lower maximal dose rates to individuals in mammalian populations. At lower dose rates the available data indicate that the incremental incidence of radiation-induced tumours is likely to be low and, therefore, of little significance at the population level.

136. Radiation-induced genetic changes in either somatic or germ cells include gross numerical aberrations (e.g. aneuploidy or polyploidy) or structural aberrations (translocations, deletions, duplications etc.) of the chromosomes as well as gene mutations (change in the nucleotide sequence in the DNA constituting the gene). If these occur in germ cells, they may be transmitted to the next generation progeny. In most cases, aneuploidy (at least for certain chromosomes) and polyploidy are incompatible with the production of viable embryos [U4, U5] and would reduce fertility or fecundity. Such effects would be included in the effects described above on reproduction and on the developing embryo. Both aneuploidy and polyploidy occur spontaneously in the oocytes and early embryos of a number of mammalian species, and there is evidence that the incidence increases with maternal age [U5]. Exposure of female mice to acute doses of 0.02-0.16 Gy produces no change in the incidence of either aneuploidy or polyploidy [M9]. The data for male mice indicate a slight possibility that the incidence of aneuploidy in offspring may be increased by the irradiation of sperm and spermatids, but there is no evidence that there is any effect from the irradiation of spermatocytes and spermatogonia [U5]. In the female Chinese hamster an acute exposure of 2 Gy is necessary to produce cytogenetic evidence for a marginal increase in the incidence of aneuploidy. Male northern field voles exhibit increased diploidy in spermatids, which is dependent on dose in the range 0.25-2 Gy [U5]. It has also been noted that any effect that depleted the oocyte population (e.g. irradiation during embryonic development) could advance the maternal age effect for aneuploidy [U4]. Clear evidence has been presented of hyperhaploidy and structural anomalies in metaphase II oocytes from young adult mice that had received gamma-ray doses of 1, 2 and 3 Gy at a dose rate of 6,000 μGy h^{-1} between 4 and 8 weeks of age [G10]. Although the chromosome aberrations reported are

probably not compatible with survival, lesser degrees of cytogenetically undetectable damage may have been present that could result in viable (but, perhaps, less fit) offspring. There appear to have been no studies of the effects of longer-term, lower-dose-rate exposure.

137. Reciprocal translocation (the mutual exchange of DNA between two non-homologous chromosomes) is the structural rearrangement that has received most attention. In the various rodent species that have been investigated, the induction rate in stem cell spermatogonia (assessed in cytological preparations of descendent spermatocytes) is 0.01-0.03 Gy^{-1} at total doses up to 3 Gy of acute low-LET exposure [U3]. Alpha irradiation of male mice for 5-8 months from ^{239}Pu taken up into the testes (36 μGy h^{-1}) increased the incidence of reciprocal translocations (derived from irradiated spermatogonia), acentric fragments (from irradiated spermatocytes) and dominant lethality (determined from post-implantation embryo mortality); an RBE of 22-24 relative to ^{60}Co gamma radiation was estimated [S50]. In non-human primates the induction rate was 0.01-0.078 Gy^{-1} at doses up to 1 Gy [M19]. The magnitude of the potential sparing effect of low-dose-rate exposure appears to be species-dependent, and the incidence could be lower by a factor of 3-10 [U4]. The gametes resulting from a spermatogonial stem cell containing a reciprocal translocation are produced (assuming random segregation through meiosis) in the proportions 1:1:2 of balanced-normal, balanced-translocated and unbalanced, respectively [S21, U6]. The mortality associated with the output of sperm with an unbalanced chromosomal constitution would be included in the reduction of fertility/fecundity noted above. For offspring heterozygous for the translocation, the total production of gametes will have chromosome constitutions in the proportions given above, so these offspring would be expected to be semi-sterile (and are often found to have even lower fertility) [U6]. The likely low incidence of such individuals means that the consequences for the overall fertility of the population would be less than the consequences arising from the direct effects of radiation on gametogenesis. Experiments with female mice have shown that the great majority of chromosomal aberrations induced in pre-ovulatory oocytes were eliminated through pre-implantation and early embryonic mortality. Again, the effects on fertility/fecundity would be included in the responses already discussed.

138. The final class of genetic changes includes gene mutations that are inherited in a simple Mendelian fashion. Detectable by their phenotypic expression in the offspring of irradiated parents or by conducting appropriate breeding experiments, they include recessive visible, recessive lethal, dominant visible, dominant skeletal and dominant cataract mutations. The available data permit estimates for specific types of

mutation. For instance, at doses up to 6 Gy, recessive mutations at seven specific gene loci in the mouse are induced at a rate of about 5 10^{-6} Gy^{-1} locus^{-1} at a dose rate of 5,000 μGy h^{-1}, and dominant mutations causing either skeletal abnormalities or cataracts are induced at rates of 4 10^{-4} Gy^{-1} gamete^{-1} and 2.6 10^{-5} Gy^{-1} gamete^{-1}, respectively, at low dose rates [S21, S43, U7].

139. Acute, sublethal whole-body irradiation of male mice or localized testicular irradiation over a number of generations has been shown to reduce the reproductive fitness of subsequent unirradiated progeny [G5, G6, S32]. This has been attributed to the accumulation of recessive mutations that, even in the heterozygous condition, combine to produce the observed response. The mean survival time under chronic irradiation (50,000 μGy h^{-1}) of mice with accumulated ancestral paternal irradiation was also lower than in controls, and simple breeding tests appeared to indicate that the characteristic was recessive in nature [S32].

140. Classical population genetics assumes that some mutations arising from increased radiation exposure would lead to a reduction in fitness (even for recessives present in the heterozygous state) and therefore have an adverse impact on the population. Selection pressure would tend to eliminate the induced mutations from the population over a number of generations inversely related to the specific reduction in fitness. At some stage (with constant radiation exposure of the population over time) a new equilibrium between mutation and selection would emerge, with a slight change in the population age-dependent mortality rate owing to the genetic changes. Laboratory studies of irradiated populations have not shown unambiguous evidence for reduced fitness from an increase in the incidence of this class of radiation-induced mutations [B17. G9, U7]. While the available data suggest a very limited mutational impact at the levels of radiation to which wild populations in contaminated environments are likely to be exposed, it should be noted that the genetic constitution of these laboratory populations is rarely comparable to that of their wild cousins. In these circumstances, the finding should be extrapolated to the natural environment with caution.

141. There have been a number of investigations of possible genetic damage in the somatic cells of mammals subjected to increased irradiation from Chernobyl fallout (primarily caesium radionuclides). The incidence of micronuclei (indicative of the presence of acentric chromosome fragments) in the erythrocytes of wild house mice was found to be increased, relative to earlier controls, during and after the fallout from the Chernobyl accident at a site north of Rome, Italy, where about 1 kBq m^{-2} of ^{137}Cs was deposited. The frequency of micronuclei was found to be positively correlated with the ^{137}Cs body burden (up to 45 Bq kg^{-1}). Although

no quantitative dosimetric data were given, it was concluded that the dose rate implied by the body burden was insufficient to account for the increased numbers of micronucleate cells [C11]. It could only be suggested that the (unknown) dose rate from external sources might have provided the necessary additional exposure. The external dose rate from ^{137}Cs (even including ^{134}Cs), however, would probably have been within the normal variation in the natural background radiation at this location. Indeed, dose estimates from a subsequent study [C12] confirm that the total dose (external plus internal) from ^{137}Cs and ^{134}Cs would have been only about 5% of the natural background dose rate.

142. A second study by the same researchers was conducted in areas of much higher Chernobyl fallout in Sweden. The bank vole (*Cletherionomys glareolus*) was used as the experimental animal, and estimates were made of the dose rates from internal and external sources of both ^{134}Cs and ^{137}Cs. The beta component of the radiation from internally deposited caesium radionuclides was the major source of exposure (52%-71%), with external gamma irradiation providing most of the balance. The estimated dose rates (in addition to 0.17 μGy h^{-1} assumed for the natural background) at the four sampling sites were 0.01, 0.2, 1 and 1.5 μGy h^{-1}. The body burdens of ^{134}Cs and ^{137}Cs showed substantial variation at each site (coefficients of variation in the range 62%-138% for n = 22-38 animals), and the dose rates received by the individual animals would therefore be expected to show a similar substantial variation (the longer-term accumulated doses might show less variation owing to fluctuations of individual body burdens over time). The incidence of micronucleate erythrocytes, although varying greatly in any given exposure group, correlated well with the mean dose rate [C12]. Ludwików [L11], however, was unable to demonstrate such a clear-cut dose-response relationship for the incidence of micronuclei in larger numbers of peripheral blood cells (rather than bone marrow cells) in a laboratory study employing the same species.

143. The incidence of chromosome aberrations was determined in the peripheral blood of semi-domesticated reindeer living in areas of Norway with differing degrees of Chernobyl fallout contamination. If age is disregarded, the frequency of aberration was not correlated with the incremental dose rates estimated for the different areas. In calves from the most contaminated area, where dose rates were about 8-9 μGy h^{-1} in 1986-1987, falling to 4-5 μGy h^{-1} in 1989-1990, the incidence of two break events (dicentrics, rings) was, however, significantly higher than in calves from other areas, and for this area the reindeer born in 1986 showed a higher incidence than older animals. In a positive control experiment in which 7-8-month-old reindeer calves were fed ^{134}Cs at four different rates for 23 weeks, there was a significant dose-dependent

increase in chromosome aberrations relative to the controls [R8].

144. A range of data has been obtained concerning the genetic damage induced by increased radiation exposure of the germ cells of male wild house mice captured at different sites within the 30-km zone of the Chernobyl nuclear power station. The external gamma-radiation dose rates at the capture sites were in the range 0.2-2,000 μGy h^{-1} (if the data given above for the bank vole are applicable. the total dose rate, inclusive of internal exposures. might be greater by a factor of 2-5). Wild male mice were mated to females of a laboratory strain to determine the incidence of dominant lethality (embryonic mortality). For males captured in 1986 there was no consistent relationship between dose rate at capture (0.5-500 μGy h^{-1}) and observed embryo mortality. In the following year, in addition to the three sites used previously, another site with higher dose rates was sampled. There, at dose rates of 1,700-2,100 μGy h^{-1}, two of the captured males produced no offspring and were concluded to be sterile. and the remainder showed a period of temporary infertility and reduced testes mass, which. however. recovered with time after capture. The only group for which embryo mortality (post-implantation) differed significantly from the rather uniform overall picture consisted of the litters sired by males within two weeks of capture from the high-dose-rate site, i.e. litters resulting from sperm which were at the post-meiotic stages at the time of irradiation [S22]. For all the mice collected at the three or four sites in 1986 and 1987, respectively, there was a dose-rate-dependent increase in the incidence of reciprocal translocations scored in spermatocytes at meiotic metaphase I [S22].

145. The results of these four studies [C11, C12, R8, S22] confirm that incremental exposure from contaminant radionuclides will, as expected, induce genetic changes in both somatic and germ cells in populations of wild mammals. The rather limited nature of the dosimetric data provided makes it difficult to compare these results with the outcome of laboratory studies. In addition, the reports provide no indication of the consequences of the damage for the respective populations. At the two most highly contaminated sites in the Chernobyl 30-km zone, the total dose rates are likely to have been greater than those suggested in paragraphs 126 and 133 as limits for the maximally exposed members of populations in areas subject to controlled contamination from waste disposal practices. It may be concluded that the lower dose rates were completely consistent with the survival of the populations, although it would be desirable to have more data, particularly on population attributes, including such factors as immigration from areas of lower contamination.

2. Birds

146. A review of the effects of ionizing radiation on birds was compiled by Mellinger and Schultz [M2]. Wild birds appear to exhibit LD$_{50}$ values that are in the same general range as small mammals (5-12 Gy). Domestic poultry are reported to exhibit an LD$_{50/60}$ of 9 Gy [B2].

147. Studies of radiation effects on the young of three species of wild birds, tree swallow (*Tachycineta bicolor*), eastern bluebird (*Sialia sialis*) and house wren (*Troglodytes aedon*). showed progressively reduced growth following acute doses to hatchlings of 0.9 to 6 Gy [Z2]. The young birds were irradiated and returned to their nests in the field and observed over the 16-day period until fledging. There was no radiation-induced mortality during the nestling period. No observation could be made of possible life-shortening. Doses of 4 and 6 Gy had statistically significant effects on body mass. with reductions of 10% and 13%, respectively, and also on primary feather length and foot length. A dose of 2.6 Gy had less effect, and 0.9 Gy had little effect on initial growth and development [Z2]. In subsequent similar experiments with tree swallows, the growth data were fitted to the von Bertalanffy growth model. The parameter values from this fit showed that doses of 2.7 and 4.7 Gy to the newly hatched nestlings produced significant growth depression, particularly in body mass: the irradiation also influenced the development time [Z4]. In domestic fowl irradiated at 2 days of age. the growth rate over the subsequent 30 days was reduced at all doses but only significantly so at doses greater than 6.7 Gy [B6].

148. From limited data it has been suggested that the juvenile bird is more resistant to radiation than the adult [Z2]. The lethal dose for the eastern bluebird nestling is given as 30 Gy [W4]. In young chickens exposed at age 3-4 days. however, the LD$_{50/30}$ was determined to be 7-11 Gy (exposure time less than 1 hour) and 12-20 Gy (exposure time of 24 hours) [S3].

149. The effects of acute radiation on the developing avian embryo have been investigated for the black-headed gull (*Larus ridibundus*) and the domestic chicken [P6]. The eggs were irradiated at day 10 of development and incubated artificially. The LD$_{50}$s at hatching for the two species were determined to be 12-13 Gy and 9 Gy, respectively. For the developing tree swallow (*Tachycineta bicolor*) irradiated at days 7-8 and incubated naturally. total doses up to 3.4 Gy did not affect hatching or fledging success, but the time to hatching increased and growth was depressed at doses greater than 1.6 Gy [Z3].

150. No data appear to have been obtained at acute doses less than 0.5 Gy, but for most of the responses

that have been studied, damage is detectable only at higher doses. Studies of effects of chronic exposures are limited. The nesting success of birds was studied in field experiments with chronic irradiation from large gamma-ray sources. Exposure rates of 8,400-42,000 μGy h^{-1} caused embryonic mortality [B7, W14, Z2], while no apparent effects were detected at 30-260 μGy h^{-1} [Z5]. Tree swallows (*Tachycineta bicolor*) occupied nesting boxes independently of radiation exposure in the range from natural background to 6 μGy h^{-1}. The breeding performance over the season was unaffected by the radiation exposure, and the production of fully-fledged young was comparable with controls. These results indicate that the population is unlikely to suffer adversely at annual doses up to 50 mGy [Z6]. Chronic irradiation of developing chicken embryos at dose rates greater than 10,000 μGy h^{-1} until hatching destroyed the process of spermatogenesis and prevented the production of any resting oocytes, effectively sterilizing both sexes [M12]. Thus, while the chronic dose rates causing effects on embryonic development in birds (8,000-10,000 μGy h^{-1}) have been determined, the minimum dose rates that may cause other clearly evident effects have not been.

3. Reptiles and amphibians

151. Studies of reptiles and amphibians suggest that these groups may have a radiosensitivity similar to that of mammals and birds when the observation period is appropriate for registering the incidence of radiation-induced mortality. Frogs, salamanders, turtles, snakes and lizards yield LD$_{50}$ values of 2-22 Gy. The cause of death has usually been identified as damage to the haematopoietic system, with mean survival times after irradiation ranging up to 190 days [C13, C14, D9, T12]. For one species of lizard (*Uta stansburiana*) two separate studies gave LD$_{50/30}$ values of 10-12 Gy [D9] and 17-22 Gy [T12]. Differences in radiation quality, dose rate and maintenance conditions were offered as explanations for the different results [T12]. In addition, since it is not evident from the data presented that the short observation period encompassed all the acute radiation mortality, the lethal dose may have been overestimated. An acute exposure of 50 Gy caused temporary sterility in males of this species of lizard, but recovery to normal proportions of spermatogenic stages was well advanced by day 48 after exposure [D9]. Exposure of adult males and females sampled from a natural population to an acute dose of 4.5 Gy to the gonads (head and thorax shielded) resulted in a substantial reduction after release in the production of offspring in the year of irradiation. This in turn reduced the density of adults in the following year (i.e. there was no compensating increase in survival), but by the second year both natality and population density were recovering to control values [T13].

152. An acute dose of 10 Gy was found to be 100% lethal for four species of amphibians, although only one species showed an LD$_{50/30}$ less than 10 Gy. Using data from the literature for two other amphibian species and estimates of the interphase chromosome volume and assuming a linear relationship between interphase chromosome volume and radiosensitivity, LD$_{50/30}$ values in the range 8-36 Gy were calculated. Extending the assessment period to encompass all the radiation-induced mortality (up to 200 days for one species) at total doses less than 10 Gy yielded lower estimates for LD$_{50}$ values, in the range 0.8-7 Gy. The interphase chromosome volume appears to be a potentially useful indicator of radiosensitivity [S33]. Adult toads, juveniles and tadpoles showed different radio-sensitivities, with estimated LD$_{50/30}$ values of 24 Gy, 10 Gy and 17 Gy, respectively. After extending the observation time, the adult LD$_{50/50}$ declined slightly to 18 Gy, but the sensitivity of the juveniles and tadpoles was much increased, with both showing an LD$_{50/50}$ of 0.1 Gy. In addition, irradiated tadpoles failed to metamorphose [L9]. A toad species exhibiting LD$_{50/30}$ and LD$_{50/60}$ values of 22 and 20 Gy, respectively, showed greater survival after 15 Gy if they were allowed to hibernate than if they were kept active. An acute dose of 15 Gy to a natural population as they emerged from hibernation in the spring had little impact on breeding activity, feeding or preparation for hibernation the following autumn. There was, however, a marked reduction in survival, independent of age and sex, in the population emerging from hibernation one year after irradiation [T18]. A single report [B18] provides evidence for the induction of abnormalities in, and reduced survival of, offspring after paternal exposure of toads to acute doses of 3-20 Gy.

153. There are relatively few data on the effects of chronic irradiation on reptiles and none, apparently, for amphibians. The data for lizards were obtained in field experiments with large gamma-ray sources. A population of *Uta stansburiana* exposed to average tissue doses of 830 μGy h^{-1} showed a sex ratio, a maximal life-span and an age distribution that were not significantly different from those of control populations over a period of five years [T8]. An earlier study of the same system had indicated differences between irradiated and control lizards in growth (length and weight) of young of the year, but this response could not be confidently attributed to the direct effects of radiation exposure; an indirect effect mediated by damage to plants was suggested as one possible cause, but non-radiation environmental factors could also have been involved [T14].

154. Observations have also been made of two other longer-lived species of lizard, *Crotaphytus wislizenii* and *Cnemidophorus tigris*, that received approximate annual exposures of 4-5 Gy and 2-2.5 Gy, respectively.

in the irradiation facility [T15]. After 3.5 years of exposure, demographic data and the failure of mature female *Crotaphytus* to assume reproductive coloration indicated a lack of reproduction in this species. Direct observations after 5.5 years of exposure showed that the ovaries had regressed completely. At this time. one of three irradiated males was also found to be sterile. Control males and females of the same age were reproductively normal. Similar responses became apparent in *Cnemidophorus* some two years later than for *Crotaphytus*; the delay simply reflects the lower average dose rate experienced by the latter species. These results underline the significant degree to which demographic (and behavioural) factors determine the relative responses of populations of animals to a given degree of chronic radiation exposure [F5, T15].

4. Invertebrates

155. A very large database exists for radiation effects on invertebrates, especially on insects. O'Brien and Wolfe [O1] prepared a monograph on the subject and concluded that insects are. in general, far less sensitive to radiation than vertebrates. This conclusion should be tempered by the following considerations concerning differences in physiology and in life cycles. For the vertebrates, there are more or less clearly definable ranges of acute absorbed dose for which the post-irradiation survival time is relatively independent of an increase in dose and that correspond to the induction of damage in specific organs or cell systems (in order of increasing dose and earlier expression time, these are the bone marrow. intestinal and neurological syndromes) ([U3], Annex G). For the mammals, there is virtually no acute (short-term) radiation mortality at doses below those that affect the haematopoietic system, and essentially all the mortality from bone marrow damage is expressed within 30 days; hence the LD$_{50/30}$ is justified and commonly employed as a comparative measure of radiosensitivity. As noted in paragraph 151, longer times (up to 190 days) are necessary to encompass the (low-dose) acute mortality in reptiles. A similar qualification on the use of LD$_{50/30}$ values has been made with respect to aquatic organisms [W15]. The presence of substantial physiological variations between the different groups of organisms means that the LD$_{50/30}$, for all its practical convenience, provides an insecure basis for comparing radiosensitivity across a wide taxonomic range. This is not to say that there are not very real differences in radiosensitivity but that these might not be as great as they appear from the comparisons made so far. For example, the apparent low radiosensitivity of many adult insects has been ascribed to the fact that very little cell division and differentiation is in progress, i.e. there is no direct physiological basis for the equivalents of the bone marrow (haematopoietic) and intestinal syndromes. In these circumstances it may be that the appropriate

comparison should be with the neurological syndrome in mammals. for which a lesser differential radio-sensitivity would be apparent.

156. The life expectancy of 12 species of arthropods (11 insects and 1 isopod) were significantly reduced at doses greater than 80 Gy. and for 6 species a 50% reduction in mean life-span was induced by doses between 20 and 40 Gy [M13]. Acute irradiation of invertebrate populations in soil samples showed that the survival of the majority of the identifiable taxonomic groups was less than 50% of the control values at acute doses less than 100 Gy [E6]. In both studies the qualitative observation was made that the radio-sensitivity of the organisms increased with their activity. Adult earthworms (*Eisenia foetida*) were found to be less sensitive to acute irradiation (LD$_{50/30}$ about 650 Gy) than juveniles. for which growth was completely inhibited at 100 Gy, and developing embryos, for which 20 Gy delivered soon after the ova were deposited in the cocoons (i.e. early in embryogenesis) significantly reduced hatching success. A dose of 20 Gy to mature adults also affected the hatchability of eggs produced subsequently. depending on the time between irradiation and egg deposition [S34]. A second species of earthworm (*Lumbricus terrestris*) also appeared to be unusually radioresistant, with an LD$_{50/30}$ of 680 Gy for acute gamma irradiation at a dose rate of 290 Gy h^{-1} [R3]. The LD$_{50/30}$ for pill bugs (terrestrial isopod crustaceans) was about 300 Gy, but after 60 days, when essentially all of the acute mortality had been expressed, the median lethal dose had fallen to about 50 Gy. The survivors of the latter exposure grew and moulted normally, and one female laid eggs that developed into fertile adults [N5].

157. The reproductive tissues in adult insects are one system in which significant cell division. i.e. gameto-genesis, is in progress, and these tissues can be damaged at much lower acute doses than those leading to mortality. This circumstance forms the basis for the production of radiation-sterilized but otherwise perfectly viable male insects in large quantities for release in pest control programmes. Three examples from a voluminous literature will serve to illustrate the kind of data available. The exposure of mature pupae or adult codling moths to 300-400 Gy completely sterilized the females and reduced male fertility to less than 10% of control values without seriously influencing behaviour except for substantially reducing sperm competitiveness. This reduced male fertility lessened the impact of released irradiated males on population growth unless a high sterile:wild male ratio of 20:1 was maintained. Doses below 250 Gy had a smaller effect on male fertility (15% of control), but the males showed much improved overall mating competitiveness (sperm competitiveness, satisfaction of female mating drive and induction of normal oviposition), with the added advantage that there

was an increased incidence of infertility in otherwise normal F_1 males. The release of irradiated males when the natural population was emerging from overwintering and again in mid-summer provided effective control but did not eliminate the pest [P9]. Similar results were obtained in the case of the spruce budworm [E7]. The red palm weevil was found to be much more radiosensitive, as 15 Gy to one-day-old males induced 90% sterility. The effectiveness of these males in population control was again limited by a decline in mating competitiveness, owing partly to a 10%-20% reduction in adult longevity and partly to reduced sperm production [R9]. In all three cases it is clear that even drastic radiation effects on the reproductive capacity of a proportion of the individuals in the population would have only a limited effect on the total population unless they were being induced extensively in space and time.

158. Like the reproductive tissues in adult insects, juvenile insect forms are much more sensitive to the lethal effects of radiation, as would be expected from the high cell turnover rates in these age classes. These findings reduce, but do not eliminate, the extent of the differential radiosensitivity between vertebrates and insects. Many factors have been shown to modify the response of insects to radiation; it is, however, very unlikely that insect species more sensitive than vertebrates to either the lethal or (more important in the environmental context) the reproductive effects of radiation will be discovered.

159. Some relatively high $LD_{50}s$ in insects have been reported after short observation periods (e.g. at 1 day or within 3-4 days). Latency periods of these lengths may occur before damage is observable [V1]. Dose-rate effects have also been reported. For example, the $LD_{50/20}$ in young adult crickets (*Acheta domesticus*) is about 50 Gy for dose rates above 2 Gy h^{-1}. At 0.3 Gy h^{-1}, however, a 50 Gy dose causes only 10% mortality in 20 days [V2]. Larger, longer-lived insects may be more sensitive than smaller, shorter-lived one.

160. There may be considerable differences in sensitivity in the different stages of insect growth. In general, radiosensitivity decreases as development proceeds from egg to adult stage. Van Hook [V1] reported the $LD_{50/20}$ for eggs of the grasshopper *Melanoplus sanguinipes* to be 4 ± 1 Gy and the first nymphal instar to be one third less sensitive. The $LD_{50/20}s$ for the various stages of growth were 13 Gy (first and third instar), 7 Gy (second instar) and 8 Gy (fourth instar and adult) [V1]. For the soil arthropod *Collembola*, the egg is 10 times more sensitive than the adult. The $LD_{50/30}$ values were reported to be 150-300 Gy for the adult insect, 130-230 Gy for the juvenile stage and 15 Gy for the egg [S6]. An exposure of 30 Gy had little impact on *Collembola* population growth over three generations (16 days), but 70 Gy delayed the commencement of the

exponential growth phase and at 16 days the population size was approximately half that of the controls [A7]. In invertebrates as well as mammals. populations may be more affected by reduced fecundity and egg mortality than by the death of irradiated individuals.

5. Summary

161. Data on radiation sensitivity of terrestrial animals are dominated by data on mammals. the most sensitive class of organisms. Acute lethal doses ($LD_{50/30}$) are 6-10 Gy for small mammals and 1.5-2.5 Gy for larger animals and domesticated livestock. When a total dose of similar magnitude to the $LD_{50/30}$ is delivered over the lifetime, for example 7 Gy to the mouse (420 μGy h^{-1}, or 10 mGy d^{-1}), the average loss of life-span has been estimated to be about 5% and would result from the induction of neoplastic disease [U5]. There is substantial interspecies variability but, in general, little indication that dose rates below about 400 μGy h^{-1} to the most exposed individual would seriously affect mortality in the population.

162. Reproductive capacity is more sensitive to the effects of radiation than is life expectancy (mortality). The reproductive rates of mammals may be depressed at doses that are 10% of those leading to mortality. Some loss of oocytes may occur at 1% of the lethal dose, but because of excess oocyte production, fecundity should be affected to a lesser extent. Exposed from conception to a dose rate of 800 μGy h^{-1}, mice can be made sterile at 25 weeks. In the most sensitive mammal studied, the beagle dog. 180 μGy h^{-1} caused progressive cell depletion and sterility within a few months, but exposure at 36 μGy h^{-1} over the whole life produced no damaging response. It is concluded that a radiation dose rate of less than 40 μGy h^{-1} to the most exposed individual in a population (and most probably, therefore, a lower mean dose rate to individuals in the population as a whole) would be unlikely to have an impact on the overall reproductive capacity of a mammalian population as a consequence of effects on fertility, fecundity or the production of viable offspring.

163. Radiation effects in birds have been shown to be similar to those in small mammals. Reptiles and invertebrates are less radiosensitive, although physiological differences begin to make more direct comparisons with other species less appropriate. The chronic exposure of one short-lived species of lizard in enclosures has shown no evident effects over 5 years at 830 μGy h^{-1}. In two longer-lived species of lizard, some individuals were made sterile after 3.5 years at 630 μGy h^{-1} in one species and after 5.5 years at 210 μGy h^{-1} in another species. Adult invertebrates are seemingly quite insensitive to the effects of radiation exposure in terms of induced mortality, but gametogenesis, developing eggs and juvenile stages are more sensitive.

C. AQUATIC ORGANISMS

164. There have been a number of reviews of the studies of the effects of ionizing radiation on aquatic organisms [A2, B4, C1, E1, I2, I3, N1, N3, P4, T6, W9]. A number of these were prepared specifically to provide a basis for assessing the potential effects of discharges of liquid radioactive effluents on aquatic organisms in their natural environment [I2, I3, N1, N3, W9]. In consequence of these many reviews, it does not seem necessary to repeat the exercise here; there have been no more recent reports of results that would alter the general conclusions. It will be sufficient, therefore, to mention the relevant conclusions, accepting that the supporting evidence has been thoroughly examined.

165. It has been concluded that, among the many non-mammalian phyla represented in aquatic environments, fish are the most sensitive to acute radiation exposure; the developing fish embryo is particularly so. The $LD_{50/30}$ values for 10 marine invertebrates were found to be in the range 2-1,500 Gy; extension of the assessment period to 60 days reduced this range to 2-680 Gy, and the median lethal doses for the apparently most radioresistant species were still declining [E8, W15]. A similar situation was found for six species of marine fish, in which the range of LD_{50}s declined from 11-56 Gy at 30 days to 9-23 Gy at 40-50 days [W15]. The LD_{50}s for developing fish embryos can, however, be much lower. Irradiation of silver salmon embryos at the single-cell stage yielded estimated LD_{50}s of 0.30 Gy at hatching and 0.16 Gy at a larval age of 90 days; the radiosensitivity decreased with embryonic development [B23]. Other fresh-water fish that have been studied appear to be less sensitive [W9]. A single LD_{50} value of 0.09 Gy was determined for a marine flatfish species (the plaice) irradiated at the blastula stage and assessed at metamorphosis [W22]. When account is taken of their poikilothermic nature and generally lower metabolic rate, fish appear to be somewhat more radioresistant than mammals, but there is substantial overlap in sensitivities. While no data are available for the marine mammals, there is no reason to suppose that they would exhibit radiosensitivities substantially different from those of terrestrial mammals.

166. Exposure of adult female loach (family *Cobitidae*, related to the carps) to whole-body exposures of 2.5, 5.0 and 10 Gy produced only very slight effects in mature oocytes at the highest dose. For an exposure of 20 Gy, a clear response was obtained in both developing and mature oocytes; a lesser response was obtained by exposure of the head alone (i.e. the ovary was shielded). and this was attributed to a radiation-induced reduction in gonadotropin secretion from the pituitary gland [E9]. Doses as low as 1 Gy caused a temporary reduction in testes weight in mature medaka (rice fish, *Oryzias latipes*), and slight effects on spermatogonial prolifera-

tion were detected histologically [K11]; the sterility induced by acute doses of 5 and 10 Gy was temporary, and recovery was well established at 60 days after irradiation [H12, M14].

167. In developing medaka embryos, the sensitivity of the primordial germ cells (as indicated by the number present at hatching) decreases with time after fertilization, but 20 Gy at all stages produces a substantial reduction. At the most sensitive early stage the number of germ cells at hatching is exponentially related to dose, and 5 Gy produces a significant reduction. In those embryos that survive a 20 Gy dose at the early stage, the reduced number of germ cells present at hatching proceed to differentiate fairly normally into spermatogonia (through mitotic proliferation) and oogonia and oocytes (through mitosis and meiosis); the greater number of germ cells that survived 20 Gy late in embryonic development were, however, inhibited from progressing into differentiation [I10]. In normal female medaka fry the total number of germ cells increased markedly immediately after hatching, with oogonia generating oocytes in various stages of meiosis. An acute exposure of 10 Gy two days after hatching prevented the normal proliferation of germ cells, and at 10 or 15 days after hatching, the irradiation reduced the germ cell population at all developmental stages. Regardless of the timing of irradiation, there was a substantial reduction (by a factor of about 6) in total numbers of germ cells [H13]. Exposure of embryos to 10 Gy on the last day before hatching had no effect on the numbers of female germ cells at hatching but delayed the commencement of proliferation and maturation; signs of recovery towards control values occurred by day 20 after hatching [H14]. In contrast to these findings, the primordial testes became more sensitive as embryonic development progressed, as indicated by the incidence of sterility and ovo-testes at 70 days after hatching. An exposure of 50 Gy on the day before hatching produced an incidence of approximately 50% equally divided between complete sterility and ovo-testes [E10]. The rainbow trout seems to have a similar radiosensitivity, with exposures of 6 and 8 Gy late in embryonic development producing a greater than 50% incidence of sterility [K12]. Thus, the primordial gonads in the developing fish embryo and the newly hatched fry have been found to be somewhat more sensitive to acute radiation exposure than the adult fish, a situation like that in mammals.

168. There have been few studies of the lethal effects of chronic radiation exposure in aquatic organisms. Exposure of *Physa heterostropha* (a fresh-water snail) at 10,000 µGy h^{-1} over the 24 week life-span under laboratory conditions (total dose 45 Gy) had no significant effect on mortality [C16]. Cumulative doses of 80-88 Gy at 9,000-10,000 µGy h^{-1} had no effect on the survival of juvenile marine scallops and clams

[B24]. A dose rate of 220.000 µGy h⁻¹ from ⁶⁰Co gamma rays, giving a median lifetime dose of approximately 120 Gy, had little effect on the age-specific survival of *Daphnia pulex* populations receiving a fixed per capita food supply, and there was little effect on life expectancy at birth at the higher dose rate of 760,000 µGy h⁻¹ [M15]. However, in food-limited populations of *Daphnia* (that is, populations under additional stress competing for food). the total death rate, including prenatal mortality, was increased at 35,000-45,000 µGy h⁻¹ [M16]. The mortality of blue crabs (*Callinectes sapidus*) was greater than that of the controls after 50 days exposure to ⁶⁰Co gamma rays at a dose rate of 290,000 µGy h⁻¹ (total dose 330 Gy), whereas a dose rate of 32,000 µGy h⁻¹ had no effect [E11].

169. Exposure of medaka to tritiated water or external ¹³⁷Cs gamma radiation during embryonic development (average 9.1 days from three hours after fertilization to hatching) at dose rates between 18,000 and 79,000 µGy h⁻¹ had little effect on hatching rate. Larval survival to one month of age was, however, consistently and significantly decreased at dose rates above 35,000 µGy h⁻¹ for tritium beta particles and at the highest dose rate for gamma rays. The incidence of vertebral anomalies was increased for all dose rates from ¹³⁷Cs (i.e. >18,000 µGy h⁻¹) but only at dose rates above 35,000 µGy h⁻¹ from tritium [H16]. Dose rates in the range 14,000-54,000 µGy h⁻¹ from ⁶⁰Co gamma rays to mosquito fish (*Gambusia affinis*) at either 15 or 25°C for 40 days (total doses 12-50 Gy) induced no increased mortality relative to the controls [C17]. Limited data (small samples) have also been obtained for the guppy (*Poecilia reliculata*). Six pairs of fish exposed to ¹³⁷Cs gamma rays at 13,000 µGy h⁻¹ for 238 days (72 Gy total dose) experienced no mortality. For pairs of fish at the lower dose rates of 4,000 and 1.700 µGy h⁻¹ (771 and 974 days. respectively). the onset and extent of male mortality (470 days and 6 of 9; 230 days and 4 of 10, respectively) differed from the controls (520 days and 3 of 12). For the females. there were no deaths among the controls, 1 from 10 (570 days) at 1,700 µGy h⁻¹ and 2 from 9 (onset at 450 days) at 4,000 µGy h⁻¹. Because of the small numbers involved, these results provide no more than an indication of possible radiation-induced mortality [W9].

170. Small groups of the marine polychaete worm *Ophryotrocha diadema*, which is hermaphroditic but not self-fertile, were exposed to chronic ¹³⁷Cs gamma radiation (1,700, 3,200, 7.700 and 14,000 µGy h⁻¹) over seven generations and the breeding performance recorded. No effects were seen on growth rate or time to first egg laying, but the reproductive performance was affected in all generations. In generation 1, the number of egg sacs, eggs and larvae produced were only reduced at the highest dose rate, but in generations 2 and 3 the reductions were clearly dose-rate-depen-

dent: by generation 7 there was a clear tendency towards recovery at the three lower dose rates (the populations at the highest dose rate went to extinction in generation 3). The lowest dose rate to produce a significant effect. reduction of larvae in generation 2, was 3.200 µGy h⁻¹ [K13]. A similar experiment with the marine polychaete worm *Neanthes arenaceodentata* recorded significant effects on reproduction in generation 1 at a dose rate of 17.000 µGy h⁻¹ but not at 2,100 µGy h⁻¹, and there was little difference in the responses to equal total doses of either acute or chronic exposure [H17]. An exposure to ⁶⁰Co gamma rays at a dose rate of 11,000 µGy h⁻¹ significantly reduced total egg production over 12 weeks (total dose 22 Gy) and 14 weeks (total dose 26 Gy) by *Physa heterostropha* in two replicate experiments: dose rates of 20.000-250.000 µGy h⁻¹ were progressively more damaging. In the first study, the egg production rate recovered towards control values between 12 and 24 weeks at 11,000 µGy h⁻¹ [C18, C19]. In *Daphnia* populations not competing for food and chronically exposed to gamma radiation, reduced fecundity was considered to be the cause of progressive reductions in population birth rate with increasing dose rate above 220,000 µGy h⁻¹ [M15]. In populations receiving a constant total food supply (i.e. in competition), fecundity appeared to increase very slightly. although overall population size declined with increasing dose rate up to 180.000 µGy h⁻¹. This was the maximum dose rate at which populations appeared to be able to maintain themselves indefinitely, and it was concluded that the slight increase in fecundity was due to the increased food available to each individual and that this partly balanced the radiation damage [M16].

171. An earlier review [W9] concluded that minor effects might be expected in gonads of fish irradiated at dose rates greater than 1,000 µGy h⁻¹. More recent studies have provided additional information. Males from mixed-sex populations of *Ameca splendens* irradiated at an average absorbed dose rate of 7.300 µGy h⁻¹ (¹³⁷Cs gamma rays) showed disruption of spermatogenesis at an accumulated dose of 0.95 Gy and were rendered sterile at 9.7 Gy. The secondary spermatogonia were the most radiosensitive stage, while the primary spermatogonia were the most radioresistant, not being completely eliminated until a total accumulated dose of 16 Gy. The testes of males that had received total doses of 3.9 and 7.0 Gy but had been allowed to recover without further exposure for 125, 174 and 236 days showed a multi-lobed appearance at dissection. Histological examination showed that the individual lobes consisted of regenerated testicular tissue (all stages of spermatogenesis present) separated by sterile segments consisting solely of Sertoli cells. Quantification of the recovery was difficult, but in the fish that had received 3.9 Gy it was estimated to be 85%-90% of the controls after both 125 and 174 days of recovery. and for the fish receiving 7.0 Gy it was estimated to be

5%-10% at 125 days and 60%-70% after 236 days of recovery. The development of damage in the ovaries of *Ameca splendens* was less easy to quantify than that in the testes, mainly owing to variation between individuals, but the final result was no less clear cut. Small resting oocytes started to decline in number after accumulating 6.9 Gy and had disappeared completely after 18 Gy. A reduction in maturing oocytes became apparent at 12 Gy, and they declined to zero after 29 Gy. Twenty-five young fish. born on day 74 of the experiment and having accumulated 14 Gy as gametes and developing embryos. were removed from the irradiated aquaria and reared to maturity. Examination of the fish at an age of 400 days, by which time the males were showing normal secondary sexual coloration, demonstrated that there had been no gonad development in either sex [R10].

172. The effects of chronic irradiation from tritium beta particles (as tritiated water) and ^{137}Cs gamma rays on the developing gonads has been investigated in embryos of medaka [E12, H15]. At hatching the male and female gonads are indistinguishable, but the number of germ cells present can be determined. Beta radiation from tritium appeared to be more damaging than ^{137}Cs gamma rays, with accumulated doses of 2.0 Gy (8,000 μGy h^{-1}) and 3.5 Gy (15,000 μGy h^{-1}), respectively, required to reduce germ cell survival to 50% of the control value with no apparent threshold; this corresponds to an RBE of 1.8 for tritium beta radiation relative to ^{137}Cs gamma rays. A combination of a genetic marker and hormone treatment permitted an examination of the sensitivity of the female germ cells alone. The 50% survival doses were 1.4 and 3.1 Gy for beta particles and gamma rays, respectively (RBE = 2.2), and by implication, the female germ cells were more radiosensitive than those of the male [E12]. The effect of these treatments on the fertility and fecundity of the resulting adult fish has also been investigated. In females irradiated as embryos and mated to unirradiated males, the number of ovipositions and the number of fertilized eggs per fish decreased with increasing dose rate, with 50% reductions at approximately 4 Gy (17,000 μGy h^{-1}) and 15 Gy (63,000 μGy h^{-1}) for beta and gamma irradiation, respectively (an implied RBE of approximately 3.8). For irradiated males mated to unirradiated females there was little consistent difference between the radiation qualities in reducing reproductive capacity. and for both radiation types a 50% reduction was apparent at approximately 5 Gy (21,000 μGy h^{-1}) [H15]. None of these results is inconsistent with the expectation of minor effects in gonads of fish irradiated at dose rates greater than 1,000 μGy h^{-1} [W9].

173. The limited information on the mutagenic effects of radiation in aquatic organisms was briefly reviewed [W9], and it was concluded that the sensitivity of these organisms was similar to that of the much more intensively studied mice and fruit flies. Recent investigations of mutation induction at a single specific locus in the pregonial and gonial germ cells of the zebra fish appear to indicate that this species, at least, might be considerably more radiosensitive than the mouse. It was deduced that most of the mutations were due to long deletions and suggested that they occur relatively frequently in pregonial cells because the rapidity of cell division reduces the possibility of repair [C20. W23]. A second specific locus test system has been developed based on the fact that strains of the medaka homozygous recessive at either 1 or 3 autosomal loci have a phenotypic expression (colour. or lack thereof) that can be recognized in the first half of embryonic development through the transparent egg membrane. The incidence of dominant lethals (per gray per gamete) from exposure of medaka sperm appears to be similar to that from exposure of the mouse. but the spermatids appear to be less sensitive. Many of the observed radiation-induced colour mutants were not viable (i.e. they were also carrying dominant lethal mutations). As in the mouse. however, some of the colour mutants were viable, showing that the medaka sperm. spermatids and spermatogonia have a radiation sensitivity for mutation induction similar (i.e. within the uncertainties) to that in the corresponding spermatogenic stages in the mouse [S44]. A study of the production of chromosome (mainly chromatid) aberrations in cultured blood cells from the central mudminnow (*Umbra limi*) by exposure to either tritiated water in the culture medium or chronic ^{137}Cs gamma irradiation showed these cells to be somewhat less radiosensitive than human lymphocytes (by a factor of approximately 5), and a value of 1.9 was estimated for the RBE [S35].

174. *Summary.* Among aquatic organisms. fish are the most sensitive to the effects of radiation, and developing fish embryos are particularly so. The LD$_{50}$ for acute irradiation of marine fish is in the range 10-25 Gy for assessment periods of up to 60 days following exposure. The upper end of the range of LD$_{50}$ for marine invertebrates has been found to be several hundred gray. Embryos. on the other hand. are affected at much lower doses, e.g. LD$_{50/90}$ of 0.16 Gy for salmon embryos [B23].

175. For chronic exposures, dose rates of 10,000-30,000 μGy h^{-1} have had no effect on mortality for snails. marine scallops, clams and blue crabs. Dose rates somewhat above this range had some effects in food-limited populations of *Daphnia pulex*. Short-term (40 days) exposure of mosquito fish at dose rates in the range 14,000-54,000 μGy h^{-1} showed no radiation-induced mortality, but for the closely related guppy, there was some indication that long-term exposure (>470 days) at dose rates above 1.700 μGy h^{-1} reduced the normal life-span, particularly for males.

176. Reproductive effects are a more sensitive indicator of radiation response in aquatic organisms. Chronic dose rates in the range 3,200-17,000 µGy h⁻¹ have been shown to reduce reproductive capacity in the fresh-water snail *Physa heterostropha* and in the marine polychaete worms *Ophriotrocha diadema* and *Neanthes arenaceodentata*. A dose rate of 7,300 µGy h⁻¹ rendered male fresh-water fish (*Ameca splendens*) effectively sterile after 50 days exposure, and a dose rate of 1,700 µGy h⁻¹ over the life-span of pairs of guppies (the fresh-water fish *Poecilia reticulata*) significantly reduced the lifetime production of offspring [W27]. It has been concluded that significant effects in fish gonads from chronic radiation exposure would be unlikely at dose rates less than 1,000 µGy h⁻¹ [I3, W9]. Overall consideration of the data available for the effects of chronic irradiation on aquatic organisms has led to the conclusion that dose rates up to 400 µGy h⁻¹ to a small proportion of the individuals in aquatic populations (and, therefore, lower average dose rates to the whole population) would not have any detrimental effects at the population level [I4, N1].

D. CLASSIFICATION OF RADIOSENSITIVITIES

177. Because radiation effects on organisms trace their origin to interactions and initial damage and its non-repair at the molecular or cellular level, there have been a number of attempts to classify organisms according to their molecular or cellular characteristics. A review of these studies can be found in Sparrow et al. [S1]. More recent studies have included phylogenetic factors in correlations of radiosensitivity with molecular and cellular characteristics [S45].

178. In 8 of 10 viruses, Epstein [E2] found equivalence (within a factor of 2) between the estimated volumes of nucleic acid in individual virus particles and the estimated target volumes obtained from radiation inactivation studies assuming a single-target, single-hit model. The exceptions were two large viruses that were known to have complex internal structures and for which the simple model might have been inappropriate. These data supported the proposition that the radiosensitive component of the virus is the nucleic acid and not the nucleoprotein. Terzi [T2, T3] extended Epstein's general approach to include bacteriophage, bacteria, yeast and cultured mammalian cells in addition to viruses. In this study, a correlation was made between D₃₇ (the dose increment reducing survival to 37% on the exponential segment of the survival curve, i.e. the linear segment on a semi-logarithmic plot) and the nucleic acid content, using the lethal efficiency (the reciprocal of the number of ion pairs produced in the nucleic acid content of an individual organism or cell at D₃₇). It was found that the organisms could be segregated into four

more or less distinct groups on the basis of the calculated lethal (inactivation) efficiency, which appeared to decrease in value as the structural complexity of the genetic material increased from single-stranded RNA and DNA to a haploid chromosome complement. A further increase in radiosensitivity above that of haploid organisms was considered as being due not necessarily to the increased complexity of the genetic material but more likely to a change in the phenogenetic patterns of lethality. Kaplan and Moses [K1] critically re-evaluated the data set analysed by Terzi [T2, T3], rejecting some but also making additions. The outcome was very similar in that essentially the same four groupings were identified: (in order of increasing radiosensitivity) RNA and single-stranded DNA viruses, double-stranded DNA viruses (including bacteriophage), haploid bacteria and yeast, and finally, diploid yeast mammalian cells and avian cells (in culture).

179. Sparrow et al. [S1] showed, for 79 organisms ranging from viruses to mammalian cells, that there is a higher degree of correlation between D₃₇ and the interphase chromosome volume (interphase nucleus volume divided by somatic chromosome number) than between D₃₇ and the nuclear volume. The D₃₇ values of the 79 organisms plotted against their respective chromosome volumes resulted in a series of eight regression lines (corresponding to eight radiotaxa), the slopes of which did not differ significantly from -1 (Figure VI). Thus, the mean chromosome volume appears to be a highly significant index of cellular radiosensitivity. However, when all eight radiotaxa are taken into consideration, it appears that for a given value of D₃₇, there can be a 10⁵-fold range in chromosome volume; conversely, for a constant chromosome volume, D₃₇ can vary by a large factor. The organisms included in those eight radiotaxa did not show any consistent relation with respect to classical taxonomy, genetic complexity or level of ploidy. Organisms without true chromosomes in a morphological sense (prokaryotes) occurred in seven radiotaxa, although viruses were restricted to the first four groups. Organisms with true nuclei or chromosomes (eukaryotes) were restricted to the last four radiotaxa (see Figure VI). Sparrow et al. [S1] concluded that the survival response is partly a function of chromosome size but that other parameters are involved that need to be identified. The results of Sparrow et al. [S1] have been presented schematically by Whicker and Schultz [W5] and are shown in Figure VII. This hierarchy indicates that mammals, including man, are the most sensitive organisms with regard to acute lethal effects. Most resistant to radiation effects are the micro-organisms, bacteria, protozoa and viruses.

180. A cautionary note has, however, been sounded concerning the use of chromosome or cell nucleus characteristics as an absolute basis for predicting

radiosensitivity. The observed effects of chronic gamma irradiation on six species of woody plants in their natural environment were compared with predictions based on chromosome number and cell nucleus volume [W24]. In terms of a 50% reduction in shoot growth relative to controls, the prediction consistently underestimated the radiosensitivity of the plants, often by a large margin. On the basis of the dose rate required to produce greater than 90% mortality, the prediction overestimated the radiosensitivity of three species, although not by a significant margin, and underestimated the sensitivity of the remainder, in two cases by a factor of at least 5. Two factors were put forward as having significantly contributed to the observed general increase in radiosensitivity: (a) the stresses associated with the natural environment and (b) the fact that the prediction was developed from a (broad-based) general correlation based on data for both woody and herbaceous plants, even though it was recognized that, other things being equal. the former are generally more sensitive than the latter [W24].

181. The correlation of radiosensitivity with molecular or cellular characteristics has been extended to mutations, and it has been suggested on the basis of empirical evidence that the forward mutation rate per locus per unit dose of acute radiation might be linearly

dependent (within a factor of 3) on the DNA content of the haploid genome [A8]. The conceptual basis of this relationship has, however, been severely criticized [S42] on the grounds that the studies underlying the apparent correlation employed differing genetic endpoints that were not directly equivalent to specific locus mutations and detection systems of varying sensitivity. In addition, it was pointed out that intra-locus mutation rates are not found to be correlated with the apparent DNA content in specific salivary gland chromosome bands (identified with loci or complementation groups) of *Drosophila melanogaster*. leading to the conclusion that not all the DNA in a band represents a target for radiation-induced mutations of the type detected experimentally. It was concluded that these factors weakened the validity of the hypothesis developed by Abrahamson et al. [A8].

182. In summary, it may be stated that although there is suggestive evidence for correlations between specific characteristics of the nucleus and nuclear components and a variety of indicators of radiation-induced damage, none of these correlations has been developed to the point where it could reliably be used to predict. from the relatively small experimental database. potential radiation effects in the wide variety of organisms likely to be present in a contaminated area.

III. EFFECTS OF RADIATION ON POPULATIONS OF PLANTS AND ANIMALS

A. STUDY LIMITATIONS

183. Literature on radiation effects in an ecological context began to appear in the early 1960s. Although some of these research programmes were terminated by the early to mid-1970s because of changes in funding priorities. investigations continued into the 1980s in a few countries. such as Canada, France and the former Soviet Union [D1, F1, G3. K2. M3, S4].

184. The general approaches to studying radiation effects on populations and communities (assemblages of populations) have included large field irradiation facilities, observations at nuclear test sites and in areas of high natural background radiation or contamination, the release of irradiated individuals to the environment, the experimental application of radioactive particles, and the irradiation or contamination of laboratory systems or microcosms. Each of the above approaches has its advantages. but most suffer from a lack of ecological realism or from dose levels insufficient to produce unequivocal results.

185. The expense and difficulty of doing a meaningful study of radiation effects on plant and animal popu-

lations and communities in their natural environment makes it impossible to provide information on a large number of species and community types. For example, studies have been completed on coniferous and deciduous forests, certain shrublands and grasslands, a tropical rain forest. herbaceous old fields and mosslichen communities [W3]. However, little work appears to have been done on aquatic plant communities or on Arctic or alpine tundra. taiga. savannah or desert communities. Also. very little work has been done on the interaction of radiation with other stresses or agents. This may be a serious omission in light of the multiple forms of pollution that threaten many contemporary populations and ecosystems.

186. Other areas that have received very little attention include the possible long-term effects of chronic. low-level radiation and the patterns of repair and recovery following radiation damage. It may take considerable time for damage from low-level chronic irradiation to be expressed at the ecosystem level, and most studies are short term. Several studies have looked at repair and recovery for a few years, but many systems require a decade or even much longer to complete their recovery from severe damage.

187. Despite these and other shortcomings in the general database. sufficient information is available to predict, within broad limits. the effects of ionizing radiation on at least a reasonable sample of the populations and communities in terrestrial and aquatic environments. However. this prediction must rely on adequate dosimetry, which is often difficult to achieve. For example, the calculation of doses to the critical tissues of higher plants following atmospheric deposition of radionuclides is a complex, difficult task and subject to considerable variability and uncertainty.

188. Ecosystems can be large and complex with many species of organisms, or they can be relatively simple with only a few species of organisms constituting a simple food-chain. The number of organisms in a population may be altered by changes in environmental conditions, or species may be completely replaced.

189. General characteristics of diminished ecosystem functioning under environmental stress have been identified [R4] that may also apply to radiation stress: (a) loose cycling of nutrients. (b) changes in primary production, (c) reduction of species diversity, (d) retrogression, as opposed to natural succession. (e) reduction in the average size of organisms and (f) other distress, such as increases in disease incidence. The stages in ecosystem response to stress are initial effects on sensitive parts, coping mechanisms to counteracting the stress and ecosystem rebalance or breakdown. It is not easy to predict specific effects in diverse ecosystems, which are usually under multiple stress. and further observations in contaminated environments are needed to increase knowledge and experience.

190. In the specific case of continuing exposure to low-level irradiation, Woodwell and Houghton [W25] concluded that the response of a temperate oak-pine forest (and other plant communities) is (a) systematic and predictable. (b) common in nature, (c) cumulative and progressive over an indefinite period measured in decades. (d) measurable in stages and (e) best characterized by the concept of "impoverishment". By "impoverishment" is meant the loss of large, long-lived, slowly reproducing species with replacement by smaller, opportunistic, short-lived species with high reproductive rates. These are readily recognizable as the weeds and pests of common currency and are typical of all disturbed habitats. In ecological terms, impoverishment is the reverse of succession, and it frequently appears to be a more rapid process [W25].

191. One of the problems in evaluating the effects of radiation on populations and ecosystems is to determine which parameters to measure and how radiation influences them. Typically measured attributes at the population level include numbers of individuals, mortality rate, reproduction rate and mean growth rate.

In general, measurable changes in populations and communities require rather severe effects at the cellular and individual organism levels. For example, alteration in the structure of a biotic community requires a change in the component populations. which in turn requires widespread mortality and/or reduced reproduction of individuals [W3]. In the same way, genetic or somatic mutations that can be produced by lower levels of exposure may have little or no impact on population or community performance because of natural selection [B3, M4, N2, P2, T6] and the convergence of genetic information among adjacent populations [R1]. For example. in organisms whose reproductive rates are very high and on which selective pressures are strong, the value of one or even many thousands of individuals to the population may be rather insignificant [I2, T6]. In such populations, normally a small fraction of the individuals will mature and perpetuate the gene pool, even in the absence of radiation or other stresses. In most species. genetic information that is altered by radiation is extremely unlikely to be perpetuated in the population. even though it may not be immediately lethal at the individual level.

192. Radiation effects at the population and community levels are manifest as some combination of direct changes due to radiation damage and indirect responses to the direct changes. This seriously complicates the interpretation of radiation effects on organisms exposed in the natural environment. The wide range of radiosensitivities of the organisms that make up most natural communities creates a situation in which, if doses are such that the sensitive species but not the more resistant ones are affected. the latter may gain a significant competitive advantage and perhaps increase in abundance or vigour [W12]. This could erroneously be interpreted as a hormetic response. Such a response might not be produced if the resistant species alone were irradiated. This is but one of many examples of indirect response to the direct effects of radiation.

193. Individual populations of organisms. and the community as a whole. may be altered by radiation exposure only as a consequence of dependence on a much more radiosensitive species for food or shelter. For example. many plant species in a pine forest would be largely unaffected by an acute dose of 10 Gy. but the pine component would probably experience severe mortality [W11]. This would cause both positive and negative perturbations in the populations of other species not directly damaged by the radiation exposure. Because of such indirect responses. in any situation where all the species in a community have been exposed concurrently, great care will be necessary to identify the species most directly affected by the irradiation.

194. Changes in the composition and structure of vegetation lead to changes in the animal population [C2.

H2, W13]. Such changes could also involve interrelated populations of animals (predator-prey or host-parasite relationships), but these are more difficult to demonstrate. Numerous and extensive changes in the chemical and biological properties of the soil can occur as an indirect effect of radiation damage to vegetation. Recent work in France has demonstrated these kinds of changes clearly [C4, S2, T4, T20] (see also paragraph 215).

195. Relevant information concerning the effects of increased chronic radiation exposure of populations of wild organisms is beginning to become available from two additional sources, the two large-scale accidents that occurred in the former Soviet Union: the one in 1957 at a nuclear waste storage site near Kyshtym in the southeastern Urals and the other in 1986 at the Chernobyl nuclear reactor in Ukraine. Apart from the immediate, effectively acute radiation response observed in plants and animals near the release points for each of these accidents, there have been continuing studies of the effects of the longer-term chronic radiation exposures. The results of these studies are summarized in Section III.D.

B. EFFECTS ON PLANT POPULATIONS AND COMMUNITIES

196. Studies of radiation effects in natural plant communities have been more or less limited to terrestrial systems. The characteristics that have been measured in relation to the stress of ionizing radiation include physiognomy (growth-form), species composition, species diversity and vegetation cover and production [W3]. At the population level, parameters such as density or frequency of occurrence, growth and vigour, mortality, reproduction, phenology and morphology are often measured [G2, O2]. Effects on individuals or components of populations can give some insight into potential changes at the population and ecosystem level.

197. Some specific measures of change in plant communities that have been used to investigate the potential effects of radiation include the coefficient of community, similarity, diversity and biomass. The coefficient of community is a parameter calculated from a list of species present in an irradiated area and a control area, or in one area before and after the irradiation. This parameter is designed to detect changes in species composition resulting from the elimination and possibly replacement of populations. Similarity is like the coefficient of community, except that the relative abundance of individuals within each species is considered as well. Diversity is a measure of the number of species within a community and the evenness of apportionment of individuals among the species. Both the loss of individuals and the loss of species would change the diversity. Biomass is a measure of the mass

of biological tissue per unit area. It may be estimated for individual populations or for entire communities, and owing to sublethal alterations in growth, it can change without a loss of individuals or species. These parameters are defined more precisely in various publications [G2, W3, W6], and more complex indices and means for analysing change are available [G7].

198. An example of radiation effects in plant communities has been seen in a study in a granite-outcrop ecosystem common to the southeastern United States [M6]. The composition of this ecosystem is relatively simple, and there are well-defined boundaries. One such system was treated with simulated fallout particles (^{90}Y), and effects of the beta irradiation were observed over a 60-day period [M6]. Accumulated doses were measured with thermoluminiscent dosimeters at the ground surface and at 40 cm height, the average height of terminal buds of the summer dominant plant, *Viguiera porteri*. The mean doses in two experimental systems were 40 Gy and 70 Gy, with 20% higher doses at the ground surface and 20% lower doses at the 40 cm height.

199. In these irradiated systems, there were no obvious changes in appearance of the experimental plots: rather close observation and measurement were required to detect effects of the radiation exposures. Only one change in the 40 plant species constituting this ecosystem was noted that could be related to the radiation exposure. A sedge, *Bulbostylis capillaries*, decreased by 17% in the 70 Gy community but increased by 12% in the control and 40 Gy communities [M6]. Other studies had shown this plant to be relatively radioresistant, but because its growing apices are close to the ground, it may have retained high densities of fallout particles in close proximity to sensitive meristems. Thus, the geometry of the radiation field is an important consideration when interpreting the observations.

200. In the main species, *Viguiera porteri*, in the 70 Gy community, 46% of all terminal buds died, resulting in a 37% height reduction: however, there was compensatory lateral branch development. In this plot, community biomass was reduced by 16% owing to decreased production of flower and stem biomass, but leaf biomass was similar to control levels [M6]. An additional observation was that beta radiation was twice as effective as gamma radiation over comparable periods and equivalent total doses [M6].

201. The effects of chronic irradiation on the micro-algal population in soil have been examined [F9]. The micro-algae appear to be relatively radioresistant, even though some part of the observed change might have been attributable to the gross effects (mortality, much reduced litter fall etc.) of radiation on the higher plants

in the zone examined, i.e. the change was partly an indirect response. The total number of taxa that could be isolated from the soil samples did not appear to change markedly with dose rate up to the maximum delivered (approximately 1.7 Gy h⁻¹). Above a threshold of approximately 0.3 Gy h⁻¹, however, eukaryotes were replaced by prokaryotes. Estimates of the coefficient of community and percentage similarity (relative to unirradiated controls) were consistent in showing a threshold for decline at approximately 0.4 Gy h⁻¹, 50% reductions at about 0.9 Gy h⁻¹ and extrapolation to zero at 2.5-3.0 Gy h⁻¹ [F9].

202. Chronic irradiation also had a substantial effect on the populations of microfungi at the base of the humus layer in an oak-pine forest. Where higher plants persisted, i.e. in the intact oak-pine forest, and where the dose rate at the sampling depth increased from less than 400 μGy h⁻¹ to about 8,000 μGy h⁻¹, there appeared to be little direct effect of radiation on the microfungi. The fungal species isolated were typical of podzolic soils in mixed conifer-hardwood forests and were dominated by eight morphologically complex taxa (71% of isolates but only 5% of the species). There was also a consistently high concentration of about 10^5 viable spores per gram of dry soil. At the edge of the sedge zone, where the dose rate was about 30.000 μGy h⁻¹, the concentration of viable spores fell to 1.6×10^4 g⁻¹ dry soil, but the species diversity increased in this transitional band between the endemic forest populations and opportunistic pioneer species, and there were few dominant forms at low densities. Closer to the radiation source, where there were no higher plants, the distribution of the microfungi was related to the radiation gradient. The viable spore density, the coefficient of community and the incidence of morphologically complex taxa all declined. At the highest dose rate, about 0.75 Gy h⁻¹, the viable spore density was reduced to 500 g⁻¹ dry soil and the population was dominated by sterile, slow-growing forms of morphologically simple yeasts [G8].

203. Based on the results of several investigators, Whicker and Schultz [W5] summarized the effects of radiation on major plant communities of North America. The results given in Table 9 are the dose levels that give minor, intermediate and severe effects following short-term (8-30 day) exposures. Minor effects are considered to include changes in productivity and reproduction, from which rapid recovery would be expected after the radiation stress has been removed. Intermediate effects include changes in species composition and diversity through selective mortality of more radiosensitive species. Recovery could take place through processes of plant succession requiring one to several generations. Severe effects are those that drastically change the species composition by causing mortality of all or nearly all the higher plants. Recovery

may be slow, requiring decades or even centuries if there has also been extensive leaching of soil nutrients and erosion. To estimate the chronic exposures that would be expected to result in comparable effects, the dose ranges in the Table should be divided by 25-100 to give daily dose ranges.

204. The plant communities included in Table 9 are listed in approximate order of sensitivity, from coniferous forests to moss-lichen communities. The greater radiosensitivity of the coniferous forest is correlated with the large chromosome volumes of pine trees. The other communities listed in Table 9 are much more resistant because they are dominated by more resistant species having generally smaller chromosomes. Lichen-dominated communities are exceptionally resistant [G1]. This resistance may be explained by diffuse centromeres and asexual reproduction as well as by small chromosomes [W3].

205. The general capacity of plant communities to withstand environmental stress (low rainfall, high temperatures) also enables them to withstand low to moderate radiation stress. There may be alterations in community structure (species abundance changes) and morphological changes in individual plants (lateral branch development), depending on the total radiation stress, but the compensations are generally such as to maintain a normal energy balance. While it is possible for radiation stress to induce systemic defence mechanisms in individual plants that could increase their potential to respond to secondary stresses such as parasitic fungi, it should also be pointed out that apparent similarities in plant regrowth and recovery, after either radiation exposure or some other form of stress, may be superficial and conceal the existence of stress-specific responses.

C. EFFECTS ON ANIMAL POPULATIONS AND COMMUNITIES

206. The impact of radiation exposure on an animal population is likely to be a complex combination of effects within individual animals (direct effects) and the positive or negative effects of the responses of the other biological components of the environment with which the population normally interacts (indirect effects). Where there is a close coupling between two species, e.g. plant-herbivore, predator-prey or host-parasite, it is very likely that their responses to chronic low-level radiation (whether similar or different at the individual level) will be modified at the population level by interspecies interaction. The direct effects of irradiation measured in animal populations have included changes in the birth rate, the death rate and, in combination, the intrinsic rate of natural increase [F2]. Because a reduction in reproductive capacity has been considered

to be a more limiting end-point than mortality, in terms of the maintenance and survival of the population, changes in fertility and fecundity have been measured and the response at the population level estimated using an appropriate model of population dynamics. The published studies have rarely, however, taken account of the influence of either intra- or interspecies interactions or factors other than radiation.

1. Terrestrial environment

207. Because animals are mobile, it is difficult to conduct field studies of radiation effects. The use of enclosures to retain animals in exposed areas may unduly restrict their movement, feeding and behaviour, causing effects in addition to those under study. Investigations in areas of high background or in more widely contaminated areas may be less affected by spatial limitations.

208. Notable effects in animal communities can result from changes in vegetative cover, particularly for insects. As vegetation dies in highly irradiated areas, herbivorous insects and their predators disappear and are replaced by species subsisting on dead and decaying material. Aphid increases were noted in irradiated oak-pine forest on Long Island, New York, the insects perhaps having been attracted by the altered quality or appearance of the oak leaves [W13]. Bark beetles invaded an irradiated tropical forest in Puerto Rico when natural defence processes in the trees were weakened [S5]. The presence of moribund pine trees in the high-dose-rate zone of a chronically irradiated oak-pine forest provided an ideal substrate for colonization by bark beetles and other xylophagous insects. Moribund trees were heavily attacked by two species of bark beetle irrespective of the ambient radiation dose rate, but the ensuing degree of developmental success was inversely correlated with dose rate. At above 200.000 µGy h⁻¹ the adults attacked the trees and began excavating galleries but were killed before they could complete the task of preparing cavities for eggs or laying eggs. Between 40,000 and 200,000 µGy h⁻¹, the egg cavities were completed but there was no egg hatch, owing either to infertility or damage to the developing embryos. In the range 30,000-40,000 µGy h⁻¹, where eggs hatched, there was total larval mortality; at 20,000-30,000 µGy h⁻¹, there was low larval mortality but all the pupae died. Between 10.000 and 20,000 µGy h⁻¹ there was some pupal mortality and some adult emergence, and below 10.000 µGy h⁻¹ adult emergence was as high as, or higher than, normal. Within this overall picture there were also interesting observations on the influence of interspecies competition with other xylophagous insects: those species with more protracted life cycles tended, at any given dose rate, to be at a competitive disadvantage. The importance of shielding, in this case between distal and proximal segments of the

trunks, was also noted. Lastly, it was concluded that if pines were debilitated by a short-term but spatially variable radiation exposure, then the less-radiosensitive bark beetles could, owing to the availability of many suitable feeding and breeding sites, reach epizootic proportions and further damage and kill trees that would otherwise have survived [B25]. These findings show how important it is to appreciate the overall context of radiosensitivities and interactions between species when assessing potential environmental damage from chronic, low-level irradiation.

209. Density-dependent responses play an important role in modifying the impact of radiation. Such compensation and adaptation in populations is not apparent in laboratory experiments on individual organisms. Studies of lizards irradiated in a nine-hectare desert enclosure in Nevada in the United States showed impaired fertility and altered population levels [T8, T9]. Species response depended on life-spans, time of sexual maturity and population age distributions. One shorter-lived species of lizard, Uta stansburiana, was able to maintain population numbers in spite of appreciable proportions of sterile older females, apparently because the young animals survived better [T9].

210. Radiation effects in exposed animal populations have not been readily apparent. For example, studies of mammals on the dry bed of the retention pond at White Oak Lake, near Oak Ridge, Tennessee, United States could not ascribe any effects to radiation exposures [D3]. Lifetime doses to the wild rodents were probably 2-3 Gy [T9]. The study of lizards in the irradiated enclosure in Nevada revealed no differences in body weight and no form of tissue pathology, although some changes occurred in ovaries and oviducts of continuously irradiated lizards at annual gamma doses of 2-5 Gy [T9]. In all species of lizard in which female sterility was observed, the amount of body fat was increased.

211. In a region of high natural background radiation in the former Soviet Union, abnormalities, decreased body fat and reduced fertility were observed in mammals living in close contact with soil (Microtus oeconomus, Talpa europaea, Arvicola terrestris, Lutra lutra and Sorex araneus) [M1]. The dose rates were about 80 µGy h⁻¹, or 700 mGy a⁻¹. Since results at such low dose rates have not been obtained in laboratory or other field studies, the validity of the dose calculations has been questioned [T9].

212. In France, definite effects on reproduction in female mice maintained in captivity at a site where the external background was about 80 µGy h⁻¹ were suggested by Leonard et al. [L1, L2, L3]. In the same study, rabbit lymphocytes showed an increased number of unstable chromosome aberrations. It is possible,

however, that internal doses. which were not calculated, contributed substantially to the total dose.

213. Radiation effects in populations of soil invertebrates were reported by Krivolutsky [K2]. Various radionuclides, including ^{90}Sr. ^{137}Cs, ^{106}Ru, ^{95}Zr, ^{239}Pu and ^{226}Ra. were added to the soil in small plots. At various intervals thereafter. the numbers per square metre of several types of soil invertebrates were determined in small ecosystem plots. Dose rates that apparently produced reductions in animal numbers were generally quite high (420-42,000 µGy h^{-1}); however, some effects were reportedly observed at dose rates of around 100 µGy h^{-1}. The most sensitive organism observed was the common earthworm.

214. An apparent response by ants to chronic irradiation has been observed in an oak-pine forest at a dose rate of 80,000 µGy h^{-1}. A thriving nest was identified prior to the commencement of radiation exposure. Over three years of irradiation, the number of ants in the colony declined (in part by emigration to a secondary colony further from the source in the first year), fewer individuals (and eventually, none) were seen on the burned pine stump forming the centre of the nest mound. and less plant debris was collected to maintain the nest mound. At the end of the period the colony appeared to increase in number, and a 1.25 m subsoil surface pathway covered with litter had been formed leading at an angle away from the radiation source: the runway was lost at 29 m from the source (dose rate of around 28,000 µGy h^{-1}). and no secondary colony was found. The ants used this pathway exclusively for travel to and from the nest. Because the vegetation canopy and its dependent populations of insects (ant food) were reduced as early as the first year. at least some of the responses (emigration and foraging further from the source) could be attributed to a decline in the food supply, but the decline in the habit of collecting plant debris to maintain the nest could most easily. albeit speculatively. be attributed to radiation avoidance [B26]. Over an 18-year period of chronic irradiation of a Mediterranean oak forest, the number of ant colonies was reduced in areas where dose rates were greater than 5.000 µGy h^{-1}. Although the number of species was little changed. the species composition was altered. Three species (of which two were not present at the beginning of the irradiation period) were found near the source at dose rates greater than 100,000 µGy h^{-1}. The changes in the ant fauna were mainly attributed to the loss of vegetation canopy, but direct effects of radiation could not be discounted [P10].

215. Two to three years after chronic irradiation of a Mediterranean oak forest there were changes in the populations of insects that could be attributed to both the direct and indirect effects of the exposure. Small insects of the order *Psocoptera* (booklice) were substantially increased in numbers in the dose rate range 14,000-38,000 µGy h^{-1}, and even at approximately 100,000 µGy h^{-1} the increase, although less. was still greater than in the control area. It was suggested that this increase was due to an increase in food supply (unicellular algae) in response to higher light levels following radiation-induced leaf loss in the canopy. In contrast. the populations of springtails (order *Collembola*) declined with increasing radiation exposure and accumulated dose [B27, P11]. In addition. later studies showed decreased overall activity in the soil system following a reduction in the microbial population [T19]. Near the source, where all the trees and shrubs had died and been replaced by small, more radioresistant annual plants. the litter input to the soil was negligible. Unlike in a temperate oak-pine forest [W24] the rate of loss of existing litter declined with increasing dose rate, but in both environments the total organic matter content of the soil decreased with increasing dose rate [P12, T19. T20]. Although the normalized oxygen consumption rate of the highly irradiated soil was also lower than in the control. the specific respiration (CO_2 production normalized to unit soil biomass) was slightly higher than in the control: the NO_3^- content of the irradiated soil was generally higher, owing to a reduced microbial requirement [P12. T19]. While the cumulative consequences of the chronic irradiation, both direct and indirect, are clearly complex. it is nevertheless true that all the observed responses stemmed from the initial damage to the most radiosensitive components of the system. the trees and shrubs; at lower dose rates, where the higher plants were much less affected. there were no consequential indirect responses [P12].

2. Aquatic environment

216. A number of aquatic field studies were conducted at sites of enhanced environmental radiation from anthropogenic sources [W5]. Examples of such sites include the Irish Sea in the vicinity of the Sellafield facilities: the Animas River near Durango, Colorado, United States, where uranium milling waste once entered the river; and a radioactive waste retention pond at the Oak Ridge National Laboratory in the United States. An extensive review of research on the ecological effects of nuclear testing was made by Templeton et al. [T6] for test sites in the Pacific. Several investigations were completed for marine organisms around the atolls [B5, H1. W1, W2]. The effects of the testing programme could not, in general. be ascribed solely to radiation because of concomitant effects of environmental disturbance. The recovery processes following the testing programme were relatively rapid, and deleterious effects on marine and terrestrial populations were not persistent. presumably because of the rapid declines in the intensity of radiation and other impacts and the recolonization of damaged areas by healthy individuals from distant locales.

217. Ponds, ditches and streams on the Hanford reservation in the United States in the past received a variety of radioactive wastes, resulting in radiation dose rates at the sediment-water boundary of up to 44.000 μGy h^{-1} (measured with LiF thermoluminiscent dosimeters). Although diversity was occasionally reduced, there were no consistent differences between the contaminated ponds and streams on the Hanford site and off-site reference locations. Across the sites examined, it was not possible to demonstrate any correlation between the variation in radiation dose rates and the biological parameters investigated, i.e. rates of periphyton and invertebrate colonization (colonization pressure) and a range of diversity indices: individuals per unit area, species richness, community diversity and species evenness [E13].

218. At the Savannah River site in the United States a number of catchment basins have received low-level waste from fuel reprocessing activities, leading to relatively high cumulative inputs of ^{3}H, $^{89/90}$Sr and $^{134/137}$Cs. Natural populations of slider turtles (*Pseudemys scripta*) inhabit the ponds and, because they are known to be long-lived, could have accumulated relatively high total doses. The technique of flow cytometry has been used to investigate the potential genetic effects of accumulated radiation exposure by measuring the coefficient of variation of the distribution of DNA content (at the G$_1$ phase in the cell cycle) of nuclei isolated from red blood cells. The mean coefficient of variation estimated from measurements of 16 exposed animals was significantly greater (and hence indicative of a gain and loss of chromosomal DNA) than that estimated from measurements of 6 control animals. For 4 other exposed animals there was clear evidence of aneuploidy mosaicism in the form of G$_1$ DNA peaks with either a distinct shoulder or an adjacent peak; the finding that 4 of 20 exposed turtles exhibited aneuploidy mosaicism, compared with 0 of 6 controls, although suggestive of an effect, did not reach statistical significance (p > 0.05) owing to the small sample size. Insufficient data were provided to permit dose rates to the animals to be estimated (in a separate study, however, the dose rate to turtles in this environment from internal sources was estimated to be not greater than 240 μGy h^{-1} [N1]); however, the exposed males (n = 10) revealed a significant (p < 0.10) positive correlation between the coefficient of variation in G$_1$ DNA content and plastron length (size). This might be indicative of increasing coefficient of variation with age (and therefore possibly with accumulated dose). A similar analysis for exposed females (n = 6) was non-significant, as was that for control males; there were too few control females to complete the analysis. Again, the presence of other, potentially mutagenic contaminants (chromium and mercury) clouds the picture, and these data represent no more than qualitative circumstantial evidence for a radiation response in a contaminated environment [B28, B29].

219. The sensitivity of developing fish embryos was extensively studied following the experiments of Polikarpov [P4], which had indicated that developing fish eggs were sensitive to minute quantities of radionuclides in water. In their review of the subject, Blaylock and Trabalka [B4] concluded that none of the experiments support Polikarpov's contention. and Woodhead [W9] attributed the conflicting results to the dosimetric problems encountered in this type of experiment.

3. Summary

220. Radiation effects on populations of organisms have been examined in large field irradiation studies. in experimental enclosures and in observations in areas with high natural background, contamination from waste disposal or accidental release of radionuclides. There are inherent limitations in obtaining information on the large number of species and community types making up the biosphere, on the interactions of the many different environmental factors of both natural and anthropogenic origin and on the long-term nature of successional and generational effects. In addition, there are difficulties in providing adequate dosimetric information for complex distributions of sources and movements of organisms within the environments.

221. The exposure to radiation of a natural community, which comprises organisms with a wide range of radiosensitivities, may result in direct damage only to the more sensitive species. Other species may be indirectly affected. for example by loss of habitat or gain of competitive advantage. For any stress to an ecosystem. such as radiation exposure. there may be a complex and long-term sequence of disruption, adjustment and rebalancing.

222. Coniferous trees of the genus *Pinus* have been found to be the most sensitive of the plant species studied following either acute or chronic irradiation. and plant communities including these species are the most radiosensitive of the plant communities for which data are available. In general terms, a forest in which pines are the dominant (or codominant) species would probably suffer minor effects at total short-term doses of 1-5 Gy or at long-term chronic dose rates of 400-4,000 μGy h^{-1}. In this context, minor effects are small changes in productivity and reproduction. from which rapid recovery would be expected following removal of the radiation source. Severe effects in the coniferous forest could occur at acute doses of more than 20 Gy or at long-term chronic dose rates in excess of 40,000 μGy h^{-1}. Severe effects would include mortality of almost all higher plants, and the ecosystem would recover, if at all, only over time periods of decades or centuries. Other types of plant community can withstand doses or dose rates at least one order of

magnitude greater before demonstrating corresponding effects (Table 9).

223. The radiosensitivity of individual organisms is but one factor in determining population effects from radiation exposure, and observations in the laboratory may not apply exactly to the environment, where there are additional stresses from competition, predation and the like. Some compensation for radiation-induced reductions in either survival or reproductive rates may be possible, but there are numerous indirect effects that could be of overriding importance.

224. Changes in vegetative cover can affect animal communities. When plant species die in highly irradiated areas, the food supply of herbivorous animals and insects and their predators is reduced. The animals may disappear and be replaced by species subsisting on dead and decaying material. Aphids were attracted to an irradiated oak-pine forest, and bark beetles invaded trees in a tropical forest when the natural defence processes were weakened. The insects may further damage and kill trees that might have otherwise survived.

225. Because of the compensation and adjustment that are possible in animal species, it is considered unlikely that radiation exposures causing only minor effects in the most exposed individual would have significant effects on the population. Reproductive changes are a more sensitive indicator of radiation effects than mortality, and mammals are the most sensitive animal organisms. On this basis, chronic dose rates of less than 100 μGy h^{-1} to the most highly exposed individuals would be unlikely to have significant effects on most terrestrial animal communities (see paragraphs 121 and 133). It has also been concluded that maximum dose rates of 400 μGy h^{-1} to a small proportion of the individuals in aquatic populations of organisms would not have any detrimental effect at the population level (paragraph 176). These conclusions refer to the effects of low-LET radiation. Where a significant part of the incremental radiation exposure comes from high-LET radiation (alpha particles), it is necessary to apply an appropriate radiation weighting factor to this component of the absorbed dose rate and to use the total weighted absorbed dose rate for the assessment of potential radiation effects.

D. EFFECTS OF ACCIDENTS

226. Two major accidents in the former Soviet Union have provided opportunities to observe radiation-related changes in plant and animal communities. Any accident is likely to be unique in terms of the quantity and composition of the radioactive material released, the time course of the release, the dispersal and deposition patterns governed by local and regional meteorological

or hydrographic conditions and the biogeochemical character of the areas subject to contamination. Where long-lived radionuclides are released, biogeochemical processes will determine the long-term behaviour and redistribution of the radionuclides in the environment. Given the multiplicity of factors, it is to be expected that any accident will yield new radioecological information. It is also true, however, that the primary concern following an accidental release of radionuclides will be to ensure that the radiation risks to human populations are controlled and minimized. An inevitable corollary is that the only environmental information likely to be collected is information immediately necessary to meet this concern, which will therefore be more or less deficient for the purpose of developing a complete radioecological description of the situation. The larger the incident and the greater its potential human impact, the more limited will be the resources available to collect radioecological information in the early phases of assessment and damage limitation.

227. It is particularly the case that the data required to develop estimates of the radiation exposure of wild organisms, that is, the space- and time-dependent variation of the radionuclide concentrations (especially for the short-lived isotopes), both within the organisms and in their external environment immediately following an accident, will not be known. These variations will result in substantial intra- and interspecies inhomogeneities in exposure and will pose considerable difficulties for establishing a clear and reliable relationship between cause (the accumulated radiation dose) and any observed effect. In practice, it is likely that estimates of the dose rates in the early period following the release would be made retrospectively by extrapolation and calculation from the observed distribution of contamination densities of the longer-lived radionuclides, a knowledge of the relative quantities of the radionuclides released and models of radionuclide behaviour in the environment. Such dose-rate estimates are inevitably imprecise and could be subject to significant systematic error.

228. Aside from the radioactive half-lives, other radionuclide-dependent factors significantly affect the potential radiation exposures received by wild organisms. These include the chemical identity of the radionuclide, which influences its environmental behaviour and its radiation characteristics (alpha, beta or gamma emissions), which profoundly affect the spatial dose field from any given source distribution. Additional complicating factors are the highly variable habits and target geometries of the wild organisms. These range, for example, from soil bacteria to single-celled algae and protozoa and include a wide variety of terrestrial and aquatic invertebrates, mammals (shrews to deer) and large deciduous or evergreen trees. If a radioactive aerosol is released, plants provide a very

high surface area to mass ratio (compared with animals) for deposition/adsorption. Because the leaves, flowers and terminal buds of plants are responsible for both energy absorption and growth and reproduction. a coincidence arises between radionuclide accumulation (and hence radiation dose) and potential radio-sensitivity. Other examples of coincidence are the surface litter layer and its populations of invertebrate decomposers. and the surface sediments and benthic organisms in aquatic systems.

229. Depending on the quantities of specific radio-nuclides released in an accident, the radiation exposures might range from low (a few multiples of the natural background) to high (absorbed doses greater than about 1 Gy). More importantly. for the higher total doses. different phases of biological response may be distinguished. Initially, and particularly if short-lived radionuclides make up a significant proportion of the release. there might be an acute phase in which total doses sufficient to produce immediate or relatively early detectable biological responses are accumulated. In the intermediate phase, dose rates become lower owing to decay of the short-lived radionuclides and possibly, but not necessarily, owing to the redistribution of the longer-lived radionuclides by natural processes. In this phase the slower accumulation of radiation dose may still result in total integrated doses sufficient to prevent recovery of organisms damaged in the initial phase or to lead to the appearance of medium-term damage. In the final, long-term phase, post-irradiation recovery (and adaptation) becomes apparent. provided that the initial and medium-term damage have not been large enough to radically alter the population or community structure. It is important to be aware, however. that such descriptions can be no more than qualitative.

1. The accident in the southeastern Urals

230. The accident at the Mayak nuclear materials production complex east of the town of Kyshtym in the southeastern Urals occurred on 29 September 1957. A fault in the cooling system of a concrete tank containing mixed, highly active nitrate-acetate wastes led to a large chemical explosion. Approximately 74 PBq of fission products were released as an aerosol that moved northeast. Approximately 95% of the activity was associated with radionuclides having half-lives of one year or less (^{144}Ce, ^{95}Zr-^{95}Nb and ^{106}Ru), while the remainder consisted almost entirely of ^{90}Sr. More than 15,000 km^2 of territory were contaminated with ^{90}Sr at a density of greater than 3.7 kBq m^{-2} (twice that from global fallout), and this included 1,000 km^2 at over 70 kBq m^{-2} and 120 km^2 at over 3.7 MBq m^{-2} [N6].

231. The greater part of the total dose to wild organisms was delivered within the first year of the accident, during which time the contamination density

(total Bq m^{-2}) decreased through decay by approximately 66% (see Table 10) [T21]. At the time of deposition the decay energy from beta radiation was approximately three times greater than that from gamma radiation, which meant that the dose-rate distribution in space and time in the contaminated area (forest. meadow, swamp, lakes, rivers and agricultural land) closely followed the detailed deposition pattern and the subsequent redistribution of the radionuclides by environmental processes. Estimates of the dose rates to various components of a pine forest (normalized to a ^{90}Sr contamination density of 1 MBq m^{-2}) are given in Table 11 [T17]. Because an area of approximately 120 km^2 was contaminated with ^{90}Sr at a density greater than 3.7 MBq m^{-2}, it is very probable that a wide variety of wild organisms would have received total doses at which acute effects might be expected.

232. The main part of the radiation exposure (and, usually, the highest dose rates) was delivered by the short-lived ($t_{1/2}$ < 1 year) beta/gamma emitters ^{95}Zr-^{95}Nb, ^{106}Ru-^{106}Rh and ^{144}Ce-^{144}Pr and occurred in the autumn and winter immediately following the accident, when the metabolic activity of the wild plants and animals was generally much reduced. This apparently increased radioresistance, in that the exposure had relatively little immediately visible impact. As a corollary. however, repair processes, which are dependent on metabolic activity, were drastically inhibited, and the difference in effectiveness between acute and chronic exposure was largely eliminated. Thus when metabolic activity resumed in the spring of 1958, the effects of the chronic irradiation (effectively equivalent to an acute exposure) became apparent and were more closely related to the total accumulated dose than to the dose rate [T17]. Estimates of the dose rates (normalized) and total doses accumulated during the acute period (autumn 1957 and winter 1957-1958) to a variety of organisms in the contaminated area are presented in Table 12. By the late spring of 1958, the dose rates to the majority of organisms had been reduced by radioactive decay and radionuclide migration, although the latter factor meant that there were also some exceptions. e.g. populations of invertebrates in the leaf litter and at the soil surface and small rodents. From that time on. the radiation exposure was genuinely chronic, and repair processes were largely able to accommodate and mitigate the radiation damage [A14. S49].

233. The major damage to the environment following the southeastern Urals accident was seen in the local forests. This was due to the coincidence of a high capacity for intercepting the active aerosol. a relatively slow clearance of the deposit and a relatively high radiosensitivity, especially in the case of coniferous trees [A9. A10, S20. T22, T23]. Pine trees that had accumulated total absorbed doses estimated to be greater than 30-40 Gy during the autumn and winter of

1957-1958 showed radiation damage, a desiccation of the needles in the crown, the following spring [K23]. The damage appeared initially in the lower and middle parts of the canopy, because the active deposit had been cleared more rapidly from the upper canopy by wind and rain, but by the autumn of 1959 the pine trees had died completely (the absorbed dose to the bud apical meristem was estimated to have been greater than 15-20 Gy). At lower, sublethal doses (greater than 5 Gy). yellowing, desiccation and partial shedding of needles, defects in the development of new needles, inhibition of shoot development and trunk growth and reduced seed and pollen viability were observed in the two seasons following the release. Birch trees showed lower radio-sensitivity, with lethality apparent at estimated absorbed doses greater than 200 Gy and sublethal responses, comparable to those described above for pine trees. being observed at lower total doses (greater than 50 Gy) received in four years following the accident. One characteristic sublethal effect in the trees was a clear shift in the timing of development: delayed sprouting of new leaves in the spring and earlier leaf fall in the autumn. In severely damaged trees (100-200 Gy) the delay in bud burst amounted to an increase in the integrated time-temperature product of 30 °C-days. In 1960, this corresponded to a delay of 7-9 days and in 1961, 4-5 days. Five years after the accident the developmental delay began to decrease. and after seven years the onset of bud-burst had returned to the normal time. The data on the responses of pine and birch trees to irradiation in the contaminated area of the southeastern Urals are summarized in Table 13.

234. Following the acute phase in the first year after the accident, the dose rates dropped to long-term, relatively low levels, and the forest began to recover through both vegetative processes and seed germination. The intensive production of side shoots even restored the crowns of severely damaged pine and birch trees, which had lost up to 95% of their needle or leaf cover and suffered a complete growth check. After 8-10 years. irradiated and unirradiated trees were indistinguishable in superficial appearance and trunk growth had been restored. In areas of ^{90}Sr contamination density above 1,000 MBq m^{-2}, where pine seeds had received radiation doses greater than 20 Gy, the germination potential was lost, but at an intermediate dose of 6 Gy (300 MBq m^{-2}) some pine seeds remained viable and developed. The growth rate of pine seedlings was. however, dependent on the ambient radiation field. and an absorbed dose of 20 Gy or more during the first season severely restricted development. In areas of ^{90}Sr contamination density below 100 MBq m^{-2}, full recovery through seed propagation was possible [T23].

235. Radiation damage was noted in the first year after the accident in all 20 herbaceous plant communities that were recognized in the contaminated

southeastern Urals region. The damage was considerable where ^{90}Sr contamination densities were above 130 MBq m^{-2} but slight where they were 18-26 MBq m^{-2}. The most sensitive herbaceous species were those with dormant buds at or near the soil surface (hemicryptophytes and hameophytes). These species disappeared from plant communities where contamination densities were greater than 18-26 MBq m^{-2} and were replaced by species with dormant buds either below ground or high above the surface and by species with short life cycles [B30, S36].

236. The long-term effects of increased chronic irradiation on the soil alga *Chlorella vulgaris* were investigated in the contaminated zone. Samples of algae were collected in years 5, 6, and 11 after the accident from areas contaminated with ^{90}Sr at between 0.04 and 1.300 MBq m^{-2} (equivalent to absorbed dose rates in the range 4-5,400 μGy h^{-1}) and tested for changes in radiosensitivity in response to an additional acute dose of 300 Gy. At all ^{90}Sr contamination densities an increase in radioresistance was observed. by a factor of up to 1.5-2 at intermediate densities. At the highest environmental dose rates it was concluded that the induced genetic load of mutations had reduced population viability. Over the 11-year study period the changes in radiosensitivity apparently stabilized in the algal populations sampled [S37, S38].

237. Because they were subject to some of the highest long-term dose rates, as the deposited radionuclides accumulated more or less rapidly in this zone, the invertebrate populations inhabiting the litter and underlying surface soils in the birch forests of the contaminated area of the southeastern Urals have been studied in some detail [K15]. Eleven years after the accident, in an area where ^{90}Sr contamination was 165-340 MBq m^{-2}, the total mesofauna densities were less than half those in control plots. The most severely affected group was the saprophages (phytodetritivores: earthworms and millipedes). It was concluded that this response arose from the enhanced radiation exposure attributable to their relatively sedentary lifestyles rather than from any intrinsically greater radiosensitivity: the more mobile predatory species showed a lesser response (Table 14). Although enhanced radiation exposure was considered to be the probable cause of the observed changes, other ecological factors could not be excluded. A second survey was made 30 years after the accident to obtain comparative data. Overall. the total density of organisms and their biomass remained depressed, at about 30% of the control values. The earthworms showed some recovery between 1969 and 1988 (Table 15), but it was notable that juvenile individuals were underrepresented in the irradiated population. The numbers in the other major groups also remained depressed relative to the controls: the phytophages, at 16% of control values, had declined since 1969. while

the predatory beetles, at 61% of control values, showed little change overall. although there was a decline in one of the two taxa studied and a recovery in the other.

238.   At the same density of radionuclide contamination (deposition on the ground), different species of insects and other invertebrates receive different radiation exposures depending on the specific variations in their behaviour and habitat at particular stages of their life cycle, and for a given radiation dose there will be a range of radiation responses as a consequence of interspecies and age-dependent differences in radio-sensitivity. The invertebrate species most likely to be affected by a radionuclide deposit are those whose early (pre-adult) stages of life are spent in the forest litter and surface soil, although length of life cycle is also a significant determinant. It has been concluded that a significant reduction in invertebrate numbers and loss of species from the typical leaf litter communities on the forest floor can be expected at a ^{90}Sr contamination density of 3.7 MBq m^{-2} [G12].

239.   Populations of small rodents (dark field mouse: *Microtus agrestis*; red bank vole: *Clethrionomus rutilis*; and wood mouse: *Apodemus sylvaticus*) also attracted attention in the contaminated zone. Populations were isolated in large open-air enclosures and studied for more than 40 generations. In an enclosure with an initial ^{90}Sr contamination density of 44 MBq m^{-2}, the estimated absorbed dose rate to the rodent skeleton decreased from 340 µGy h^{-1} in 1962 to 20 µGy h^{-1} in 1981. Control enclosures were established at the periphery of the contaminated zone. In the early part of the study, when the dose rates were highest, a number of changes were observed in rodent populations in the con-taminated enclosures: the death rate increased and individual life-spans decreased owing to accelerated ageing of the older animals; the reproductive perform-ance declined owing to increased embryo loss and a shift in the timing of, and a decrease in, the reproductive span; the incidence of ecto- and endo-parasitism increased; the susceptibility to predation increased; there were changes in behaviour, e.g. a decrease in the number of individual habitation sites used by the dark field mice; individual animals appeared to have greater tolerance of stress. e.g. wood mice proved to be more resistant to a physical load such as swimming; individual animals showed a lower rate of oxygen consumption (considered to be an adaptive response); and, very clearly, there was an overall increase in the variability of most population and individual morpho-physiological attributes [113, 114, 115]. In bank voles and wood mice living at sites with ^{90}Sr contamination densities of 19-63 MBq m^{-2}, the incidence of aberrations in bone marrow cells increased over the 25-30 generations after the accident, i.e. with increasing cumulative dose. Individuals from contaminated sites showed greater radioresistance to single injections of

75-185 kBq ^{90}Sr per gram body weight than did the controls [D17]. Despite all these changes, apparently attributable to the increased radiation exposure, the rodent populations in the contaminated enclosures have survived for many generations. and it has been concluded that there has been some homeostatic adjustment or adaptation to the increased irradiation.

240.   At the time of the southeastern Urals accident some farm animals in the contaminated area were still grazing in the fields, although relocation to barns and stables had been in progress. At three sites close to the release point, where the total radionuclide contamina-tion density 20 days after the accident was between 930 and 1,100 MBq m^{-2}, the external dose rate to farm animals was high (Table 16), as was the intake of radionuclides from contaminated grass and subsequent incorporation into tissue (Table 17). Estimates of the absorbed doses over 12 days to different segments of the gastro-intestinal tract (up to 50 Gy) and the skeleton (up to 2 Gy) of cows and sheep are given in Table 18. At these doses the symptoms of acute radiation sickness and the gastro-intestinal syndrome (mucosal bleeding. diarrhoea and leukopaenia) appeared, and the animals began to die 9-12 days after the accident. Where the total radionuclide contamination density was 170 MBq m^{-2}, resulting in an external dose rate from gamma radiation of up to 920 µGy h^{-1} (a total dose of 0.13 Gy in 12 days) and absorbed doses from internal contamination of 4.2 Gy to the rectum and 0.15 Gy to the skeleton, no animals died within six months of the accident. Over a period of 120 days the total radio-nuclide concentration in some of the tissues of the animals had decreased by up to 75%, but those in the liver and skeleton had increased (Table 17). Those animals from this area that were relocated outside the contaminated zone did not differ from the controls in terms of subsequent mortality or reproductive perform-ance, and there was no firm evidence of an increase in teratogenic effects [A11, B30].

241.   Rather less information has been provided concerning the effects of the contamination on aquatic systems. A study of fish populations (pike, perch, roach and golden and silver carp) was made in two lakes 14-15 years after the accident. Although no dose rate estimates are given, the concentration data for ^{90}Sr and ^{137}Cs in water and fish tissues, together with data from previous studies [N1]. suggest that at that time the total absorbed dose rate might have been a few thousand microgray per hour, mainly from ^{137}Cs incorporated into the lake sediments, for which no data are given. No effects. compared with controls, were noted for reproductive performance or morphological character-istics [M17]. The general conclusion has been drawn that there are no observable genetic effects in fish at dose rates below 400-1,200 µGy h^{-1}, although under adverse environmental conditions, the threshold might

be lower by a factor of 10, particularly for developing fish eggs [S37].

2. The Chernobyl accident

242. The accident at the Chernobyl nuclear power plant occurred on 26 April 1986. A brief account of the events leading up to the accident, the time-course of the release of the radionuclides and the initial atmospheric dispersion was given in the UNSCEAR 1988 Report ([U3], Annex D); further details on the cause of the accident are given in [I11]. A wider range of radionuclides than in the southeastern Urals accident was released in this accident because the reactor core contained an operational inventory of fission and activation products at the time. The total release, exclusive of noble gases, was estimated to be 2 ± 1 EBq [I11]. The variable time-dependency of the release rates for the different radionuclides and the changing meteorological conditions over the 10-day release period resulted in very inhomogeneous deposition patterns. Not including the 30-km exclusion zone, an area of approximately 2.4×10^4 km² was contaminated with ^{137}Cs at a deposition density greater than 200 kBq m⁻², with 5,710 km² at greater than 600 kBq m⁻² and 1,360 km² at greater than 1.5 Mbq m⁻² [I12]. Within the exclusion zone the contamination density may have been more than two orders of magnitude greater in limited areas [K14].

243. From the point of view of potential effects on the environment, the main differences between the Chernobyl and southeastern Urals accidents were the time of occurrence and the quantity of activity released. The Chernobyl accident occurred in late April, just as wild plant and animal populations were entering the accelerated growth and reproductive phases of their life cycles, i.e. when they were at their most radiosensitive. Other differences, such as the extended (10-day) period of the release and the radionuclide composition were generally of less significance. As indicated in paragraph 230, three main phases of the radiation impact on the environment were discerned [K3]. In the first, 10-20 days following the accident, essentially acute exposures, in large part from vapour clouds, were delivered to organisms close to the power plant from the large quantities of short-lived radionuclides (^{133}Xe, ^{131}I and ^{99}Mo) released. The second phase extended through the summer and early autumn of 1986, when despite a decline in dose rates at the soil surface to 25%-20% (and sometimes to as low as 10%) of the initial values, damaging total doses were accumulated. In the third and on-going phase of chronic exposure, dose rates are less than 10% of the initial values and are derived mainly from ^{134}Cs and ^{137}Cs contamination. Approximately 80% of the accumulated exposure was delivered within three months of the accident, and over 95% of this was due to beta radiation [S39].

244. Within two weeks of the accident, lethal effects were visible in pine trees close to the damaged reactor. In this zone of 500-600 ha the trees were estimated to have received doses in excess of 80-100 Gy. i.e. doses greater than the acute LD_{50}, mostly from beta radiation. Although the trees were moribund, there was some evidence initially of root survival. The deciduous trees in this zone suffered partial damage. A second zone, of approximately 3,000 ha, received estimated doses above 8-10 Gy, and dieback of new vegetative shoots of pine trees was apparent, needles and buds were damaged and deciduous trees showed morphological changes. In a third zone of 12,000 ha there were moderate effects: coniferous trees received doses estimated to be 3.5-4 Gy and showed various morphological changes, including growth suppression and needle loss, reduced reproductive capacity and genetic damage during 1986 and 1987. There was some evidence of minor abnormalities in growth, morphology and reproduction throughout a large part of the 30-km exclusion zone [K24, K25, S41]. The data on radiation doses in the four zones are summarized in Table 19. It should also be noted, however, that the distribution of absorbed dose in the tissues of trees could be markedly non-uniform owing to the presence of hot particles in the fallout from the reactor. These particles, containing mainly $^{95}Zr-^{95}Nb$, ^{106}Ru, ^{134}Cs, ^{137}Cs and ^{144}Ce, were of irregular shape (2-10 to 30-40 μm) and were found sticking to the waxy or resinous covering of pine needles. It has been estimated that the additional absorbed dose from them may have been an order of magnitude greater than the external gamma-ray dose [K25].

245. The radiosensitivity of spruce trees was observed to be higher than that of pine trees. At absorbed doses as low as 0.7-1 Gy. spruce trees showed disturbances in needle morphology, bud development and shoot growth in 1986. In 1987, the trees developed large apical shoots with needles 35-40 mm long and exhibiting a variety of straight, twisted and curved forms [K25].

246. The effects noted in the pine trees were mainly determined by the initial acute (over a period of days) exposures. During the summer of 1986, as dose rates declined, there was continuing inhibition of growth owing, mainly, to meristem damage and the reduced synthesis of growth hormones; new growth was also evident, dependent on the accumulated dose. By the spring of 1987 stems and leaves were actively growing, although some morphological changes were noted in trees that had received doses greater than 2-2.5 Gy. With an overall decline of dose rates to less than 10% of initial values and a relative stabilization of the exposure from internal sources, growth of trees continued and by 1988-1989 was apparent even in the second zone (Table 19) [K3].

247. The reproductive organs, female (seed) and male (pollen) cones, of the pine trees showed the greatest

radiosensitivity. In 1987, at doses of 0.7-1.1 Gy, the incidence of chromosome aberrations in micro-sporocytes (the premeiotic stage of pollen production) was more than three times the natural level, and the viability of the resultant pollen (tested by *in vitro* germination) decreased by 30%. The female gameto-phyte, a small, multicellular (approximately 2,000 cells) organ that arises from a single haploid cell (the megaspore) following meiosis and produces 2-6 ova, was also found to be very radiosensitive. In 1988, as can be seen from Table 20, there was some evidence for recovery [K25].

248. In the summer following the accident, disturbances were noted in the process of gametogenesis in crop plants, and grain yields were reduced by 50% [S40]. Dose estimates to the affected plants are not available. Within the 30-km zone around the power station, where the initial dose rates to herbaceous plants had ranged from 2 to 8,000 µGy h^{-1}, there was no correlation in 1988 between the contamination density and effects in seeds of different species, as measured by seed mass and aberrations in root cells from germinated seed. In contrast, seeds of *Plantago lanceolata* showed increased sensitivity to additional radiation exposure [T5].

249. In 1986-1987 a marked reduction was observed in the number of species in the litter microarthropod community of the forests of the 30-km zone; the impact was less pronounced for the soil microarthropods and the larger invertebrates. At an estimated total dose of 30 Gy, no changes were noted in adult animals, but the numbers of juvenile stages were seriously depleted. In the succeeding 2-2.5 years the populations recovered [K16]. Immigration of individuals from outside the zone contributed significantly to the recovery of the insects, and larval abundance reached near-normal levels [S41]. Within the 10-km zone changes in the numbers of wood lice (Crustacea, *Isopoda*) persisted until 1988-1989 [K17].

250. Within the 30-km zone it was estimated that the radiation doses to small rodents over the acute phase (up to mid-May) may have reached 880 Gy (20 Gy from gamma rays and 860 Gy from beta radiation), which would have been lethal [T7]. An increase in morphological variability was noted over 10 generations in populations of bank voles in the contaminated region of Mogilev, Belarus. In addition, individual animals showed lower oxygen consumption, increased tolerance to stress and an increase in radioresistance; it was concluded that all these changes were symptomatic of an adaptation to the increased radiation exposure [I7, K18, K19]. The responses here parallel those observed in rodents inhabiting the contaminated area of the southeastern Urals (see paragraph 240). Biochemical studies of rodents captured in the summer-autumn of

1987 in an area where they were receiving a dose rate of 1.500-2,000 µGy h^{-1} from external gamma radiation showed disturbances of membrane lipid oxidation when compared with animals exposed at 2-10 µGy h^{-1} [K26]. The range of radionuclides present in the deposit on pasture meant that the consequent exposure of farm animals was far from uniform; for example, cows grazing open fields for 240 days following the accident were estimated to have received doses to the thyroid, the gastro-intestinal tract mucosa and the whole body in the ratio 230:2.1:1, and it was suggested that the thyroid exposure might well have contributed to the observed morbidity and mortality [A12]. There was no evidence of teratogenic effects in domestic animals outside the 30-km exclusion zone.

251. In populations of brown frogs (*Rana arvalis* Nills) in contaminated areas close to the Chernobyl power plant, a decrease in male fertility (spermato-genesis) was identified as the only effect that could be unambiguously attributed to the increased radiation exposure (dose estimates are unavailable). In the spring of 1987 more than one third of the egg clusters deposited were wholly or partially infertile. In eggs that were fertile, so-called partial division was observed as a consequence of the anomalous behaviour of the male pronucleus and/or the anomalous replication of the chromosomes. In controls, the proportion of clusters with infertile eggs was less than 1.5%. In 1988 the proportion of partially or completely infertile egg clusters remained high (27%), but this declined to a stable incidence of 3% from 1989 onwards. No effects were detected in the processes of either ovogenesis or embryonic development [C21]. An increased incidence, compared with controls, of cells with chromosome aberrations was found in tissues sampled from frogs in the 30-km zone in 1987 [K27]. In frogs inhabiting areas in Belarus with contamination densities of 180-2,200 kBq m^{-2} of ^{137}Cs and 3.7-96 kBq m^{-2} of ^{90}Sr, the chromosome aberration rate in red bone marrow cells was found to be greater by a factor of 2-10 in the period 1986-1989. The estimated absorbed dose rate of 0.2-7.1 µGy h^{-1} from ^{90}Sr-^{90}Y was considered to be the primary cause [E14]. Chromosome aberrations were also observed in *Chironomid* larvae collected from the Chernobyl power station cooling pond and from small ponds near the village of Yanov, within 10 km of the power station [P19].

252. The cooling water reservoir of the Chernobyl nuclear power plant received a substantial input of radionuclides as a consequence of the accident. Estimated inventories for the end of May 1986 are given in Table 21 [K27]. Maximum dose rates to aquatic animals from external sources (water, particulate material trapped on aquatic plants and sediments) were estimated, using established methodologies, to be 4,200-8.400 µGy h^{-1} in 1986 (10^5 times greater than the

natural background dose rate) [12, 19, K28]: the estimated dose rates from radionuclides accumulated in tissues were lower, in the range 80-120 µGy h^{-1} [K20]. In 1987 it was estimated that the external dose rate had fallen to 1,300 µGy h^{-1} [P15], and in 1989 radionuclides within the fish were a more significant source of exposure (20-1,300 µGy h^{-1}) than gamma radiation from external sources (0.4-83 µGy h^{-1}) [L10]. For bottom-living fish of the 1985-1986 year class, the accumulated dose from internal and external sources was estimated to have reached 10 Gy by 1991, with over 50% having been received in the three-year period 1986-1989 [K24, K29, K30]. Radiation-induced damage was observed in the gonads of fish surviving the accident and in subsequent generations (see Table 22) [B33, M22]. Over the period 1989-1992, 5 of the 70 silver carp (*Hypophthalmichthys molitrix*) examined were sterile, and 35% of females and 48% of males showed gonad abnormalities, including degeneration of the gameto-genic cells. The high dose rates experienced by benthic animals from radionuclides accumulated in the reservoir sediments produced reproduction disturbances in bream (*Abramis brama*) and silver bream (*Blicca bjoernka*) and in colonies of molluscs [K24, S41].

3. Summary

253. The effects on ecosystems of radiation exposures caused by the accidents in the southeastern Urals and at Chernobyl in the former Soviet Union have been analysed. Each accident was unique, clearly so with respect to the radionuclides emitted, the time course and season of the release, but also with respect to the local and regional environmental conditions influencing dispersion and deposition. The highest doses resulted from the acute phase following the accident, until many short-lived radionuclides had decayed. The measurements were most limited in this important phase, and the dosimetry was most complex.

254. There are considerable problems in making accurate estimates, particularly retrospectively, of the total absorbed doses received by the wide variety of organisms inhabiting the areas contaminated by the two accidents. There were substantial variations in the dose rates over time, both between the two accidents and between the accidents and controlled experimental studies. Despite these admitted and considerable uncertainties, it must be concluded that the observed environmental consequences of the accidents are generally consistent with the results of experimental work either under controlled laboratory conditions or employing large, sealed gamma-ray sources in the natural environment.

255. General features of species radiosensitivities were noted in these accidents. Coniferous trees were most sensitive, deciduous trees less so. Herbaceous species with dormant buds at or near the contaminated soil surface were most affected and were replaced by species with buds either below or high above the soil surface and by species with short life cycles. Populations of soil invertebrates received high doses in highly contaminated areas, and their numbers were depressed even 30 years after the accident in the southeastern Urals. The observed effects on fecundity and fertility under continuing chronic irradiation conditions are to be expected and, for the particular cases of fish and molluscs in the cooling water reservoir of the Chernobyl power plant, could have been predicted from available information. There has been no report of a local (i.e. isolated) population of a single species having been eliminated as a consequence of the radiation exposure. Populations of small rodents have been isolated in large enclosures and studied for more than 40 generations. Radiation-related effects have been observed, along with some homeostatic adjustment or adaptation to the altered conditions. There is evidence of recovery, in many instances, from the initial acute-phase responses, and in all areas the populations continue to survive under long-term chronic irradiation.

CONCLUSIONS

256. All living organisms exist and survive in environments where they are subject, to a greater or lesser degree, to the natural radiation background and, more recently, to man-made contamination from global fallout following atmospheric nuclear weapons tests. At times, and generally in restricted areas, there are additional increments of radiation exposures either from authorized (controlled) discharges of radioactive wastes to the air, ground or aquatic systems or from accidental releases. In the majority of cases there have been no apparent effects in wild plants and animals from these additional exposures. Following severe accidents,

however, damage has been observed in individual organisms and populations, and long-term effects could develop in communities and ecosystems from the continuing increased chronic irradiation.

257. The available data on the exposure of wild organisms to radiation from the natural background and from contaminant radionuclides are relatively limited. They relate to a very restricted variety of organisms, although for the marine environment they do provide a reasonably representative picture of the range of dose-rate regimes likely to be experienced. Because the

estimates are largely derived either from localized measurements of the concentrations of radionuclides within the organism and in its immediate external environment or from models that assume an equilibrium state, there is very little information on the temporal variation in dose rates to be expected from short-term fluctuations in discharge rates, differing stages in the life cycle, changes in behaviour and short-term environmental factors such as seasonality. It is thus very difficult to estimate from the available data the total doses that are likely to be accumulated over specific stages of the life cycle, e.g. during embryonic development or up to reproductive age.

258. For both terrestrial and aquatic environments, there appears to be a significant contribution to the natural background dose rate from alpha radiation. For the former the main source appears to be ^{222}Rn and its short-lived decay products, and for the latter the main source is ^{210}Po. Owing to the short range of alpha particles, the absorbed dose rates are tissue-specific, and the results underline the crucial need for more detailed information on the distribution of the radionuclides relative to the biological targets that might be considered important (e.g. the developing embryo or the gonads) if accurate estimates of background radiation exposure are to be made. The usual range for the background radiation exposure is up to a few microgray per hour, but in exceptional cases (e.g. the hepato-pancreas of a small pelagic marine shrimp) the absorbed dose rate may be as high as 150 μGy h^{-1}.

259. It is accepted that the release of radioactive wastes to the environment is likely to increase the radiation exposure of wild organisms. For discharges to the atmosphere, to a landfill or to surface waters, the published assessments reviewed in this Annex indicate that the radiation exposures to some (but not all) individuals in endemic wild populations could reach about 100 μGy h^{-1} in general; in exceptional cases, depending on the quantities of specific radionuclides in the wastes, absorbed dose rates might reach several thousand microgray per hour. In a very limited number of instances the dose rates estimated from measured concentrations of radionuclides in the contaminated environment have been broadly confirmed by *in situ* measurements employing dosimeters attached to the animals.

260. The dose rates in the environment following an accidental release clearly depend on the quantities of specific radionuclides involved, the time-scale of the release, the initial dispersal and deposition patterns, and their subsequent redistribution by environmental processes over time. It is equally clear that these accidental releases have the potential to generate much higher dose rates and higher total doses in the environment than do normal operations. Such was the

case following the accidents in the southeastern Urals and at Chernobyl, where numerous studies have indicated that trees (and, by reasonable extension, other organisms) close to the release points could have accumulated doses up to 2,000 Gy and 100 Gy at the two accident sites, respectively, over relatively short periods of time. At both sites, longer-term chronic exposures from the deposit of longer-lived radio-nuclides have continued to be significantly higher than exposures from controlled waste disposal.

261. From these data it may be concluded that it is the responses of plants and animals to chronic radiation exposures up to a maximum absorbed dose rate of 1,000 μGy h^{-1} that are of interest from the viewpoint of providing a basis for assessing the environmental impact of controlled radioactive waste releases: in practice, information at lower dose rates, up to 100 μGy h^{-1}, would probably be sufficient in the great majority of cases. For accident situations, where experience has clearly demonstrated that initial dose rates can be high enough to allow accumulating lethal doses in relatively short periods (days), data are needed to provide the basis for predicting the progress of environmental recovery at generally lower, long-term chronic dose rates, down to the upper end (1,000 μGy h^{-1}) of the range of interest for assessing waste disposal practices.

262. There is a wide range over which organisms are sensitive to the lethal effects of radiation. A general classification has been devised based on the interphase chromosome volume of sensitive cells. These and other results of experimental irradiations show mammals to be most sensitive, followed by birds, fish, reptiles and insects. Plants show a wide range of sensitivity that generally overlaps that of animals. Least sensitive to acute radiation exposures are mosses, lichens, algae and micro-organisms, such as bacteria and viruses.

263. Sensitivity of the organism to radiation depends on the life stage at exposure. Embryos and juvenile forms are more sensitive than adults. Fish embryos, for example, have been shown to be quite sensitive. The various developmental stages of insects are quite remarkable for the range of sensitivities they present. Overall, the available data indicate that the production of viable offspring through gametogenesis and reproduction is a more radiosensitive population attribute than the induction of individual mortality.

264. In the most sensitive plant species, the effects of chronic irradiation were noted at dose rates of 1,000-3,000 μGy h^{-1}. It was suggested that chronic dose rates less than 400 μGy h^{-1} (10 mGy d^{-1}) would have effects, although slight, in sensitive plants but would be unlikely to have significant deleterious effects in the wider range of plants present in natural plant communities.

265. For the most sensitive animal species, mammals. there is little indication that dose rates of 400 μGy h^{-1} to the most exposed individual would seriously affect mortality in the population. For dose rates up to an order of magnitude less (40-100 μGy h^{-1}), the same statement could be made with respect to reproductive effects. For aquatic organisms, the general conclusion was that maximum dose rates of 400 μGy h^{-1} to a small proportion of the individuals and, therefore, a lower average dose rate to the remaining organisms would not have any detrimental effects at the population level. The radiation doses necessary to produce a significant deleterious effect are very difficult to estimate because of long-term recovery (including natural regeneration and the migration of individuals from surrounding less affected areas), compensatory behaviour and the many confounding factors present in natural plant and animal communities in both terrestrial and aquatic environments.

Table 1
Internal dose rates to leaves and needles of trees from natural radionuclides in the plant material
[J1]

Radionuclide	Half-life	Effective energy (MeV)		Dose rate per unit concentration (nGy h⁻¹ per Bq kg⁻¹)	
		α	β, γ	α	β, γ
H-3	12.4 a	-	0.0057	-	0.0033
C-14	5730 a	-	0.048	-	0.028
K-40	1.28 10⁹ a	-	0.068	-	0.039
U-238	4.47 10⁹ a	4.18	-	2.42	-
Th-234 ᵃ	24.1 d	-	0.10	-	0.057
U-234	2.45 10⁵ a	4.76	-	2.76	-
Th-230	7.7 10⁴ a	4.67	-	2.71	-
Ra-226	1600 a	4.76	-	2.76	-
Rn-222 ᵇ	3.82 d	19.2	0.22	11.1	0.13
Pb-210 ᵃ	22.3 a	-	0.12	-	0.068
Po-210	138 d	5.30	-	3.07	-
Th-232	1.41 10¹⁰ a	4.00	-	2.32	-
Ra-228 ᵃ	5.75 a	-	0.22	-	0.13
Th-228 ᵃ	1.91 a	11.1	-	6.42	-
Rn-220 ᵇ	55 s	20.9	0.21	12.10	0.12

a   Including decay product.
b   Including short-lived decay products in equilibrium with the parent radionuclide.

Table 2
Dose rates to leaves and needles of trees from natural background radiation

Source	Absorbed dose rate (μGy h⁻¹)		
	α	β, γ	Total
External radiation			
Cosmic radiation at sea level	0.0004ᵃ	0.032	0.032
Terrestrial gamma radiation		0.01-0.18	0.01-0.18
Internal radiation			
H-3	-	0.000001	0.000001
C-14	-	0.0016	0.0016
K-40	-	0.001-0.006	0.001-0.006
U-238-Ra-226 + Th-232-Ra-224	0.0001-0.001	-	0.0001-0.001
Rn-222 in air	-	0.007-0.035	0.007-0.035
Rn-222 in groundwater	0.005-0.54	-	0.005-0.54
Pb-210–Po-210	0.013-0.025	-	0.013-0.025
Total	0.02-0.57	0.05-0.26	0.07-0.8

a   From cosmic-ray neutrons.

Table 3
Estimated maximum total absorbed dose rates to aquatic organisms from various sources [a]

	Dose rate (μGy h⁻¹)							
	Fresh water			Coastal seas			Deep ocean (> 4,000 m)	
Source	Phyto-plankton	Pelagic fish	Benthic molluscs	Phyto-plankton	Pelagic fish	Benthic molluscs	Bathypelagic fish	Benthic molluscs
Natural background								
Cosmic radiation [b]	0.027	0.022	0.022	0.027	0.022	0.022	-	-
Internal radionuclides	-	0.049 (0.013)	-	0.073 (0.072)	0.047 (0.022)	0.15 (0.12)	0.06 (0.03)	0.19 (0.16)
Radionuclides in water	0.062	0.007	0.0035	0.0043 (0.0002)	0.001	0.0005	-	-
Radionuclides in sediment [c]	-	-	0.16	-	-	0.16	-	-
Total	0.089	0.078	0.19	0.1	0.070	0.33	0.06	0.19
Global weapons fallout								
Internal radionuclides	-	0.26	0.0015	0.25 (0.003)	0.018	0.080	-	-
Radionuclides in water	0.0053	0.0026	0.0013	0.00016 (0.00001)	0.00007	0.00003	-	-
Radionuclides in sediment [c]	-	-	0.058	-	-	-	-	-
Total	0.0053	0.26	0.061	0.25	0.018	0.080	-	-
Waste disposal								
Columbia River								
Internal radionuclides	2.0	21	240					
Radionuclides in water	0.026	0.023	0.011					
Radionuclides in sediment [c]	-	-	8.6					
Northeast Irish Sea								
Internal radionuclides				21	0.015	0.59		
Radionuclides in water				0.033	0.024	0.012		
Radionuclides in sediment [c]				-	-	33		
Northeast Atlantic dump site								
Internal radionuclides							0.0068 (0.0024)	0.40 (0.39)
Radionuclides in water							-	-
Radionuclides in sediment [c]							-	0.025
Total	2.0	21	250	21	0.039	34	0.0068	0.42

[a] Values in parentheses are the contribution to the absorbed dose rates from high-LET radiation.
[b] At 1 m depth for phytoplankton and 2 m depth for fish and molluscs.
[c] Gamma radiation only.

Table 4
Dose rates to leaves and needles of trees from noble gas radionuclides in air
[J1]

Radionuclide	Half-life	Decay mode	Effective energy (MeV)		Dose rate per unit concentration in air (10^{-12} Gy h^{-1} per Bq m^{-3})	
			External	Internal	External	Internal
Ar-41	1.83 h	β	0.25	0.083	110	0.004
		γ	1.3	-	580	-
Kr-85m	4.48 h	β	0.080	0.086	36	0.008
		γ	0.16	-	72	-
Kr-85	10.7 a	β	0.08	0.10	40	0.01
Kr-87	76.3 min	β	1.2	0.040	540	0.004
		γ	0.80	-	360	-
Kr-88 ^{a}	2.86 h	β	0.57	0.26	250	0.025
		γ	2.1	-	950	-
Xe-131m	11.9 d	γ	0.04	0.16	18	0.04
Xe-133m	2.2 d	γ	0.06	0.20	30	0.05
Xe-133	5.25 d	β	0.004	0.12	2	0.03
		γ	0.046	-	21	-
Xe-135m	15.6 min	γ	0.43	0.10	190	0.025
Xe-135	9.1 h	β	0.12	0.10	54	0.025
		γ	0.25	-	110	-
Xe-138 ^{a}	14.2 min	β	0.8	0.13	360	0.03
		γ	1.8	-	810	-

a Including decay products.

Table 5
Estimated doses to leaves and needles of trees from normalized discharge to the atmosphere of noble gases from nuclear reactors

Radionuclide	Normalized discharge [a] [TBq (GW a)$^{-1}$]	Integrated concentration in air at 1 km [b] [Bq h m^{-3} (GW a)$^{-1}$]	Dose factor [c] [nGy h^{-1} per Bq m^{-3}]	Absorbed dose per unit energy generated [µGy (GW a)$^{-1}$]
**Pressurized water reactors**			
Ar-41	0.87	72	0.69	0.050
Kr-85m	0.24	20	0.11	0.002
Kr-85	3.5	290	0.04	0.012
Kr-87	0.04	33	0.90	0.003
Kr-88	0.16	13	1.2	0.016
Xe-131m	0.69	57	0.018	0.001
Xe-133m	0.53	44	0.03	0.001
Xe-133	82	6830	0.023	0.16
Xe-135m	0.062	5.2	0.19	0.001
Xe-135	3.5	290	0.16	0.047
Xe-138	0.17	14	1.2	0.017
Total				0.3
**Boiling water reactors**			
Ar-41	0.15	12	0.69	0.009
Kr-85m	4.0	330	0.11	0.037
Kr-85	1.3	110	0.04	0.004
Kr-87	4.0	330	0.90	0.30
Kr-88	6.5	540	1.2	0.65
Xe-131m	0.22	18	0.018	0.0003
Xe-133m	0.28	23	0.03	0.0007
Xe-133	26	2170	0.023	0.050
Xe-135m	3.8	320	0.19	0.060
Xe-135	12	1000	0.16	0.16
Xe-138	13	1080	1.2	1.3
Total				2.6
**Gas-cooled reactors**			
Ar-41	2150	180000	0.69	120

a   Data from UNSCEAR 1993 Report [U2].
b   Assumes dispersion factor of 3 10^{-7} Bq s m^{-3} per Bq [U4].
c   From Table 4 [J1].

Table 6
Estimated dose rates to organisms from controlled discharges of radionuclides that would each result
in a dose rate of 1 mSv a⁻¹ to man residing in the same environment
[I4, N1]

Radionuclide	Dose rate ($\mu Gy \ h^{-1}$)		
	Plants [a]	Animals [a,b]	Fish [c]
H-3	5.8	5.8	0.59
C-14	18	11	
P-32	32	28	4.8
Co-60			0.53
Sr-90	2.0	0.042	67
Zr-95	38	2.0	
Tc-99			3.8
I-131	1.2	0.058	
Cs-137	5.4	3.1	0.72
Ra-226 [d]			3.6
U-235 [d]			2.6
U-238 [d]			4.7
Pu-239 [d]	0.023	0.00055	0.49
Am-241 [d]			0.71

a   Discharges to atmosphere.
b   Domestic sheep.
c   Discharges to water (lakes).
d   High-LET radiation.

Table 7
Cumulative absorbed doses in agricultural animals following single intakes of radionuclides and for similar periods of chronic intakes of radionuclides
[K9]

Animal species	Type of exposure [a]	Cumulative absorbed dose (μGy)						
		After 10 days	After 20 days	After 30 days	After 60 days	After 90 days	After 180 days	After 365 days
Iodine-131 [b]								
Cattle	Single	200	300	340	370	370		
	Chronic	900	2,000	3,000	4,200	4,300		
Sheep	Single	1,100	1,600	1,800	2,000	2,000		
	Chronic	4,900	11,800	16,600	22,200	23,000		
Swine	Single	420	600	700	740	740		
	Chronic	1,800	4,400	6,200	8,300	8,300		
Strontium-90 [c]								
Cattle	Single	0.6	1	1.2		2.4	4	6
	Chronic	2	10	20		128	420	1,360
Sheep	Single	4	8	10		20	32	52
	Chronic	22	82	166		1,080	3,420	11,320
Swine	Single	8	14	20		40	60	100
	Chronic	40	140	300		1,920	6,180	20,400
Caesium-137 [d]								
Cattle	Single	0.07	0.12	0.17		0.37	0.50	0.56
	Chronic	0.32	1.25	2.7		19.7	60	160
Sheep	Single	0.5	1	1.4		3.1	4.2	4.7
	Chronic	2.8	10.5	22.6		165	500	1,330
Swine	Single	0.32	0.6	0.9		1.9	2.5	2.8
	Chronic	1.6	6.3	13.5		98.5	300	800

a   Intake of 1 kBq on the first day for both single and chronic exposures; for chronic exposure the intakes declined according to the radioactive decay rates; radionuclides administered orally in experimental setting.
b   Absorbed dose in thyroid.
c   Absorbed dose in bone.
d   Absorbed dose in muscle.

Table 8
Absorbed dose and period in organogenesis for radiation-induced, externally detectable malformations
[C22]

Radiosensitive organ	Animal species	Main induction period (days after conception)	Lowest reported dose to cause effect	Main dose range to cause effect [a]
Central nervous system	Mouse (strain-dependent)	8-13	0.25 Gy	1.0-2.0 Gy
	Rat	9-14	0.50 Gy[b]	1.0-2.0 Gy
Eye	Mouse	7-8	0.25-0.50 Gy	1.0-2.0 Gy
	Rat	9-10	0.25 Gy	1.0-2.0 Gy
	Hamster	8-9	>1.0 Gy	
	Rabbit	-10	>1.0 Gy	
Skull	Mouse	7-10	0.5 Gy	1.0-2.0 Gy
	Rat	9-12	1.0 Gy	1.5-2.5 Gy
	Hamster	7-8		>2.0 Gy
	Rabbit	9-11		>4.0 Gy
	Monkeys	8-12		>2.5 Gy
Trunk	Mouse	6-13		>0.25 Gy
	Sheep	-23		>1.0 Gy
	Cattle	-32		>1.0 Gy
Extremities	Mouse	10-3		>1.5 Gy
	Rat	>9		>2.0 Gy
	Hamster	>9		>2.0 Gy
	Dog	25-28		>1.3 Gy

a   Single x-ray exposure.
b   Histological findings from 0.1 to 0.4 Gy.

Table 9
Dose levels from short-term irradiation producing damage to plant communities [a]
[W5]

Plant community	Dose range to produce effects (Gy)		
	Minor effects	Intermediate effects	Severe effects
Coniferous forest	1-10	10-20	>20
Deciduous forest	10-100	50-350	>100
Shrubland	10-50	50-200	>200
Tropical rain forest	40-100	100-400	>400
Rock outcrop	80-100	100-400	>400
Old field	30-100	100-1,000	>1,000
Herbaceous forest understorey	200-400	400-600	>600
Grassland	80-100	100-1,000	>1,000
Herbaceous invaders	400-600	600-1,000	>1,600
Moss lichen	100-1,000	500-5,000	>2,000

a   Exposures of 8-30 days. Dose range may be 2-4 times less for more acute exposures. Equivalent daily dose ranges for chronic effects are 1%-4% of the listed values.

Table 10
Primary composition of radionuclides deposited in a contaminated area following the southeastern Urals accident
[T21]

Radionuclides	Half-life	Fraction of total activity (%)						
		October 1957	March 1958	July 1958	July 1959	July 1960	July 1961	July 1962
^{144}Ce - ^{144}Pr	265 d	66.3	78.2	80.0	68.6	49.6	29.6	15.0
^{90}Sr - ^{90}Y	28 a	5.15	8.4	12.3	26.0	45.6	66.6	82.7
^{106}Ru - ^{106}Rh	1 a	3.45	4.6	4.9	5.3	4.8	3.8	2.3
^{95}Zr - ^{95}Nb	65 d	25.1	9.0	2.8	0.1	0	0	0

Table 11
Vertical distribution of dose rates from beta and gamma radiation to components of a pine forest in an area
of contamination from the southeastern Urals accident normalized to deposition density of ^{90}Sr of 1 MBq m^{-2}
[T17]

Months after deposition	Absorbed dose rate ($\mu Gy \cdot h^{-1}$)				
	At soil surface	At litter surface (2 cm)	At bottom part of crowns (3 m)	At middle part of crowns (6-7 m)	At top part of crowns (9-10 m)
0	67	270	2,200	3,000	1,700
1	1,700	9,400	3,300	1,700	1,200
7	2,300	12,000	1,300	1,500	830
21	2,300	10,000	830	670	330
33	6,300	4,000	250	83	50
45	9,400	2,000	130	50	25

Table 12
Normalized radiation dose rates to organisms during the acute period following the accident in the southeastern Urals [a]
[T17]

Organism	Component, location or specific type of organism	Maximum normalized absorbed dose rates ($\mu Gy\ h^{-1}$ per $MBq\ m^{-2}\ {}^{90}Sr$)	Absorbed doses during the acute period	
			Normalized doses (Gy per $MBq\ m^{-2}\ {}^{90}Sr$)	Maximum doses near release point (Gy)
Pine trees	Bud meristem	1,300-2,100	3-5	100-800
	Seeds in the canopy	830-1,300	2-3	50-400
	Seeds on the soil	420-830	0.5-1	20-200
Birch trees	Bud meristem	830-1,300	0.5-2	20-200
	Seeds in the canopy	420-830	0.3-0.7	10-100
Herbaceous plants	Dormant buds	0-4,200	0-10	0-2,000
	Seeds on the soil	830-4,200	2-10	70-2,000
Soil invertebrates	In the leaf litter	130-830	0.5-5	200-800
	At 1 cm depth in the soil	83	3	10-40
Mammals	Large herbivores (GI tract)	1,300	3	100-400
	Small rodents (whole body)	130-210	1-2	10-100
	Carnivores (GI tract)	420	3	30-100
Birds	Small, overwintering	830-1,300	2-3	50-400
	Carnivorous, overwintering (GI tract)	420	1	30-100

a   The dose rates have been normalized to a strontium-90 contamination density of 1 MBq m^{-2}, because there was a relatively constant relationship between all the radionuclides throughout the deposition.

Table 13
Radiation damage to trees in the contaminated area of the southeastern Urals

${}^{90}Sr$ contamination density ($MBq\ m^{-2}$)	Average absorbed dose (Gy)		Radiobiological effect
	Needles	Bud meristem	
1.5-1.8	5-10	2-4	Pine: desiccation of needles in the lower part of the crown, non-viability of pollen and seeds, reduction in growth increment
3.7-4.4	10-20	5-10	Pine: desiccation of 95% of the crown, growth retardation
6.3-7.4	20-40	10-20	Pine: complete death (LD_{100})
37-59	-	40-60	Birch: desiccation of the upper storey in 1% of trees, up to 30% reduction of young growth, low germination capacity of seeds, reduction in growth increment
92-140	-	100-150	Birch: desiccation of the upper storey in 30% of trees, up to 75% reduction of young growth (LD_{50})

Table 14

Numbers of the main groups of soil mesofauna in ^{90}Sr-contaminated and control sites in the southeastern Urals
[K15]

Type of invertebrate	Species	Absolute number per m²		Proportion of control
		At contaminated site	Control	
Saprophage	Earthworms	0.1	9.4	0.01
	Millipedes		6.0	
	Total	0.1	15.4	0.01
Phytophage	Snails/slugs	0.7	3.0	0.23
	Herbivorous insect larvae	9.0	14.5	0.62
	Total	9.7	17.5	0.55
Predators	Spiders	2.0	7.0	0.29
	Insect larvae	5.0	6.5	0.75
	Beetles	12.5	8.5	1.47
	Centipedes	1.4	9.0	0.16
	Total	20.9	30.0	0.70

Table 15

Changes in numbers of the main groups of soil mesofauna in ^{90}Sr-contaminated sites in the southeastern Urals
[K15]

Type of invertebrate	Species	Absolute number per m² in 1988		Proportion of control	
		At contaminated site	Controls	In 1969	In 1988
Saprophage	Earthworms	3.7 ± 0.9	8.8 ± 3.7	0.08	0.42
Phytophage	Click beetles	4.3 ± 1.8	25.5 ± 10.7	0.59	0.17
	Weevils	1.8 ±0.7	12.3 ± 5.1	0.44	0.15
Predators	Carabid beetles	5.8 ± 0.6	7.8 ± 2.5	0.48	0.77
	Staphylinid beetles	1.7 ± 0.6	4.3 ± 1.3	0.83	0.40
Other	Lithobiidae	0.3 ± 0.1	2.0 ± 0.8	0.10	0.15
	Diptera	2.0 ± 0.6	2.0 ± 0.6	0.24	No difference

Table 16

Degree of contamination and external dose rates at sites where farm animals were grazing following the accident in the southeastern Urals
[A11]

Site	Contamination density on day 20 (MBq m²)	Concentration of beta emitters in grass (MBq kg⁻¹)	External dose rate at 1 m (µGy h⁻¹)	Total absorbed dose over 12 days (Gy)
1	1,100	360	15,000	2.9
2	1,100	340	11,000	2.0
3	930	28	6,300	1.4
4	170	-	920	0.13

Table 17
Total concentration for all radionuclides in animal tissues near sites of the accident in the southeastern Urals
[A11]

Animal	Mass (kg)	Number of animals	Concentration (kBq kg⁻¹)						
			Muscle	Lung	Liver	Kidney	Skeleton	Large intestine	Wool, feathers
Site 1 [a]									
Cow	300	1	19		150		370	200	
Cow fetus	30	1	20	14	37	39	1,200	21	
Goat	30	1	24	240	98	230	1,100	410	160,000
Sheep	30	1	58	56	170	510	190	190	46,000
Geese	3.2	3	1.1	10	3.3	5.6	59	3.3	1,700
Hens	1.2	3	3.0	20	28	37	250		12,000
Site 4 [a]									
Cow									
Day 11	300	1		14	7.0	8.1	24		
Day 120	380	1	0.33	8.1	24	2.9	63	0.33	
Sheep									
Day 11	30	1	6.7	34	1.5		74	19	2,100
Day 120	48	1	1.9	5.2	5.6	4.1	310	59	9,300
Geese									
Day 11	4	3	1.2	41	1.4	2.8	41	170	1,800
Day 120	5.1	2	0.44	19	31	1.1	240	19	520
Hens									
Day 11	1.2	3	0.96	5.9	0.93	2.1	81	27	220
Day 120	1.4	5	0.22	2.1	0.26	0.26	85	5.6	85

a Contaminated sites defined in Table 16.

Table 18
Estimated absorbed dose over 12 days to different segments of the gastro-intestinal tract and to the skeleton of sheep and cows at three sites contaminated by the accident in the southeastern Urals
[A11]

Organ	Absorbed dose (Gy)					
	Sheep			Cows		
	Site 1	Site 2	Site 3	Site 1	Site 2	Site 3
Segment of GI tract						
Rumen (1)	9.1	9.0	2.5	9.3	8.9	2.5
Rumen (2)	7.5	7.1	2.0	6.2	5.9	1.6
Omasum	11	10	2.5	16	15	4.2
Abomasum	4.5	4.3	1.2	3.8	3.7	1.0
Duodenum	3.8	3.6	1.0	2.2	2.2	0.61
Jejenum	4.9	4.7	1.3	2.6	2.5	0.69
Ileum	7.6	7.3	2.0	5.6	5.4	1.5
Large colon	30	29	8.0	14	13	3.7
Small colon	34	33	9.2	17	17	4.6
Rectum	54	52	15	23	22	6.0
Skeleton	1.9	1.9	0.53	1.9	1.8	0.48

Table 19
Distribution of radiation damage in the forest around Chernobyl nuclear power plant
[K24, K25]

Damage zone/area	Type of damage	Absorbed dose from external gamma radiation (Gy)	Absorbed dose rate on 1 October 1986 (μGy h⁻¹)	Absorbed dose in needles (Gy)
Lethal zone, 4 km²	Complete death of pine trees Partial damage of deciduous trees	>80-100	>5,000	>100
Sublethal zone, 38 km²	Death of most growth points, partial dieback of coniferous trees Morphological changes in deciduous trees	10-20	2,000-5,000	50-100
Zone of medium damage, 120 km²	Suppressed reproductive ability, desiccated needles, morphological changes	4-5	500-2,000	20-50
Zone of minor damage	Disturbances in growth and reproduction, morphological disturbances in coniferous trees	0.5-1.2	<200	<10

Table 20
Chromosomal aberrations at meiosis in pine microsporocytes in an area close to
the Chernobyl nuclear power reactor
[K25]

Estimated absorbed dose (Gy)	Number of cells analysed		Chromosomal aberrations (%)	
	1987	1988	1987	1988
0.7-1.1	4,200	1,800	22.0	14.4
1.7-2.3	6,300	2,200	30.2	9.6
Control	3,000	1,000	5.7	5.8

Table 21
Estimates of quantities of radionuclides in the Chernobyl nuclear power plant cooling pond on 20 May 1986
[K27]

Radionuclide	Activity (10¹² Bq)	
	Sediments	Water
Cs-137	110 ± 50	60 ± 30
Cs-134	60 ± 40	30 ± 15
Ce-144	860 ± 400	30 ± 20
Ce-141	640 ± 280	50 ± 30
Ru-106	220 ± 100	20 ± 10
Ru-103	700 ± 360	40 ± 15
Zr-95	1200 ± 500	50 ± 30
Nb-95	1100 ± 400	70 ± 40
Ba-140	400 ± 140	120 ± 70
La-140	280 ± 120	80 ± 40
I-131	30 ± 10	250 ± 60
Sr-90	50 ± 20	6 ± 4

Table 22
Effects of radiation on silver carp fish surviving in the Chernobyl nuclear power plant cooling pond after the accident
[B33]

Year of sampling	Number of fish analysed				Proportion of fish with abnormalities in generative cells (%)		
	Females	Males	Sterile	Total	Females	Males	Total
Caged silver carps [a]							
1989	17	8	2	27	0	25	-
1990	11	6	3	20	55	33	47
1991	9	7	0	16	78	57	69
1992	3	4	0	7	33	100	71
Total	40	25	5	70	35	48	44
Uncaged silver carps [b]							
1992	9	10	2 [c]	19	89	90	90

a   Confined in aquaculture pens.
b   At large in the cooling pond.
c   Partial sterility.

Mortality in 50% of plants ——●——

Severe growth inhibition ——●——

——●—— Floral inhibition

——●—— Pollen sterility

——●—— Mortality in 10% of plants

——●—— Yield reduced by 50%

——●—— Failure to set seed

——●— Yield reduced by 10%

●————— Normal appearance

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|       |       |       |       |       |       |       |       |
0      10      20      30      40      50      60      70      80
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PERCENTAGE OF LETHAL DOSE
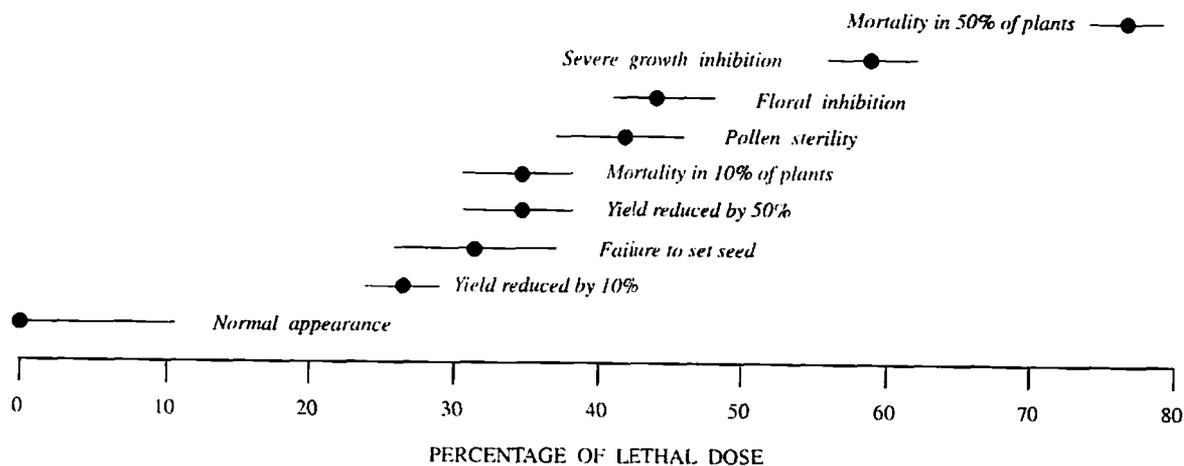
Figure I.
General range of radiation response in herbaceous plants in comparison to the lethal dose (LD_{100}).
[S8]
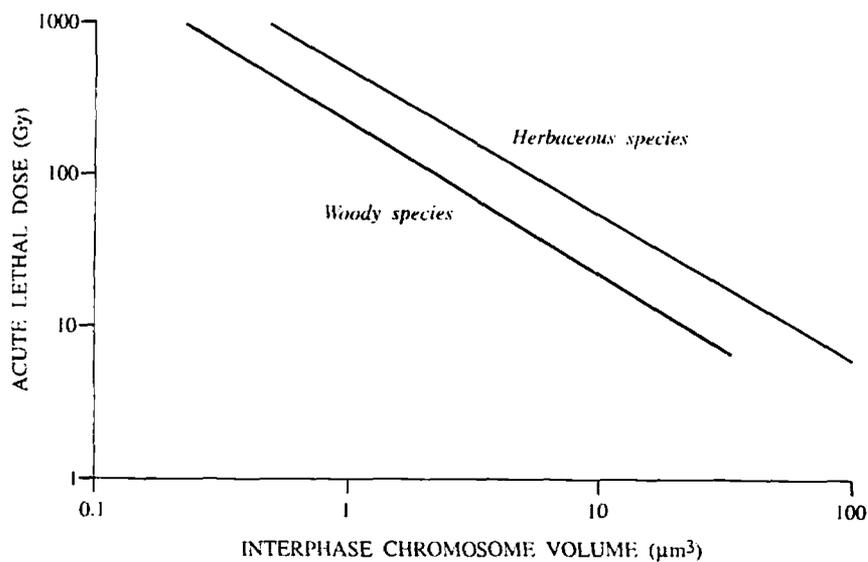
Figure II.
The acute lethal exposure to high-dose-rate, low-LET radiation in relation
to the volume of interphase (non-dividing) chromosomes of angiosperms.
[S12]
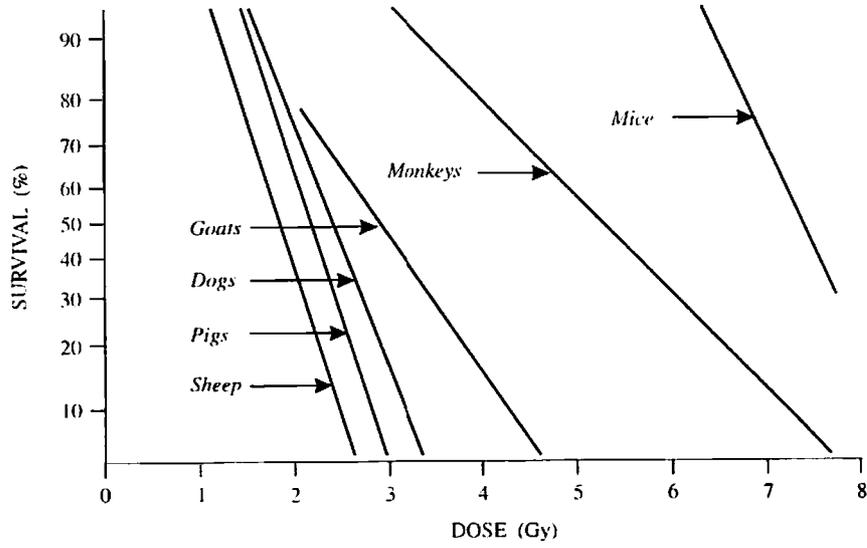
Figure III.
Midline dose-survival curves for various species of animals irradiated bilaterally.
[T10]

TIME AFTER CONCEPTION (days)
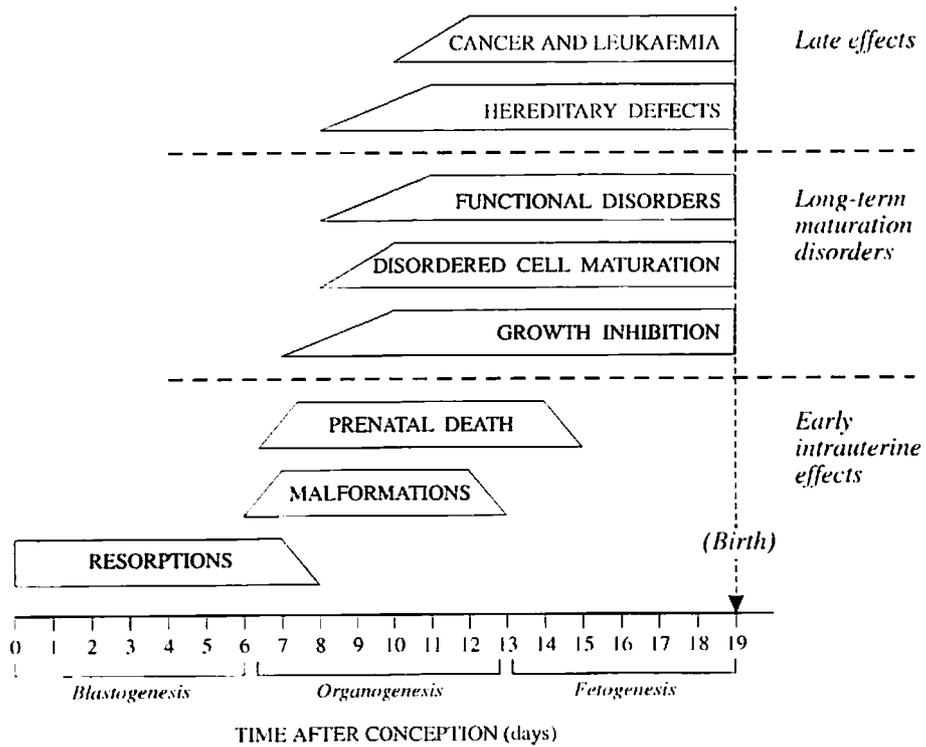
Figure IV.
Prenatal phases for induction of radiation effects in the mouse
[C22]
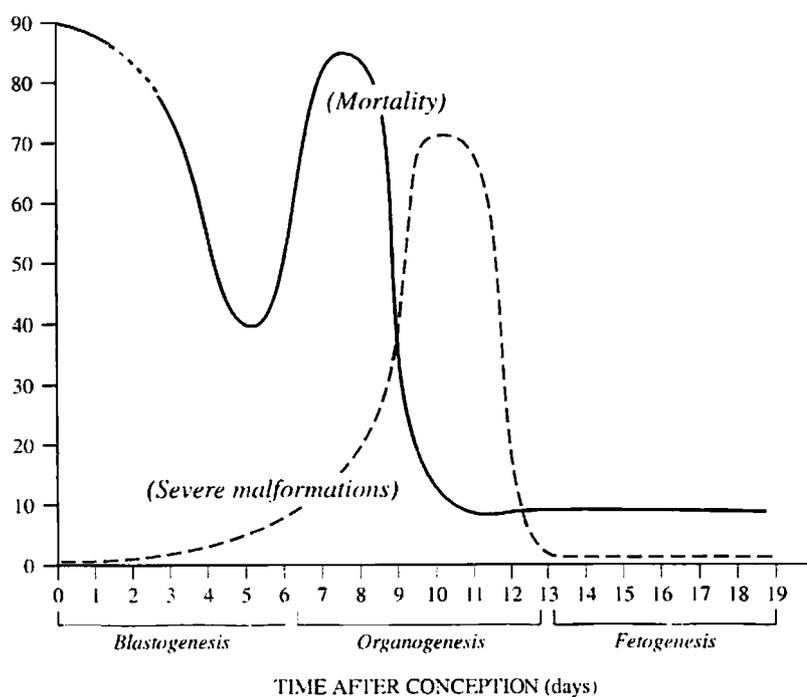
Figure V.
Relative occurrence of intrauterine injuries in the mouse after acute x irradiation with 1.9 Gy. For irradiation following
day 14 after conception, there is no radiation-induced injury, and the spontaneous rates are observed.
[C22]
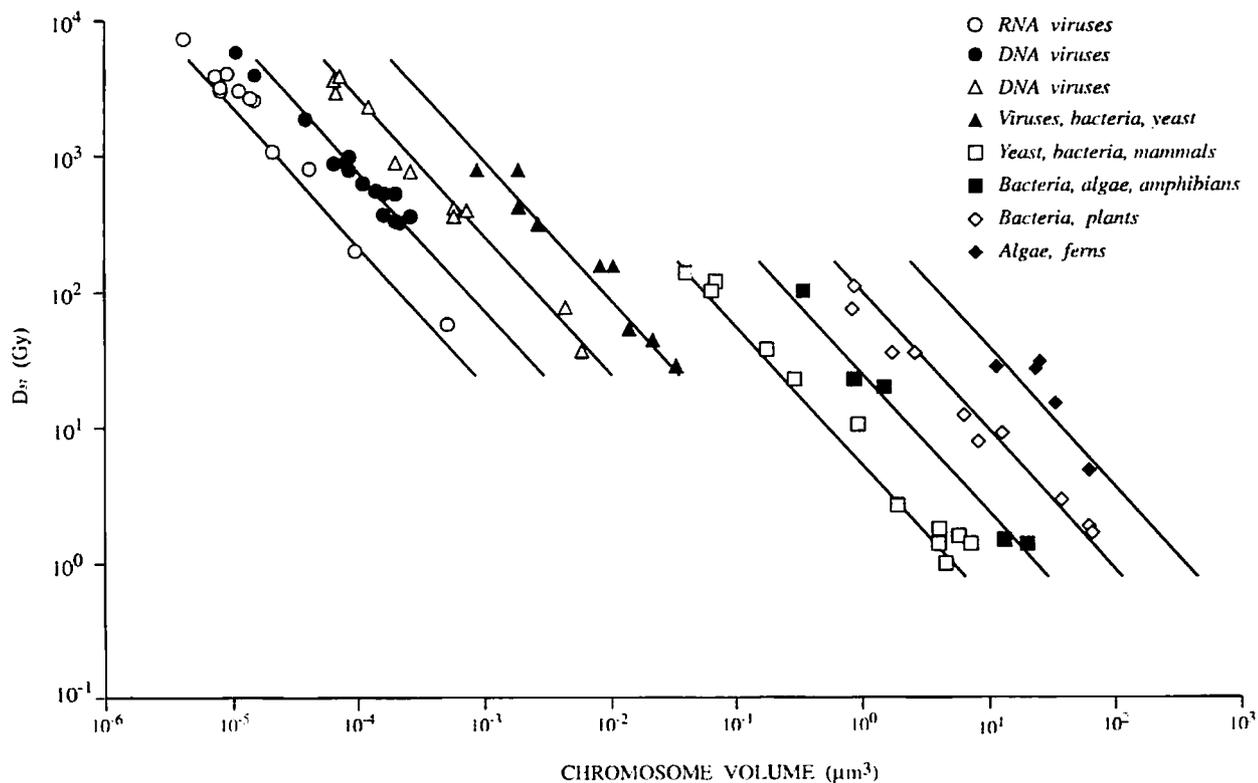
Figure VI.
Cell radiosensitivity based on correlation between D_{37} (37% survival dose) and chromosome volume.
[S1]
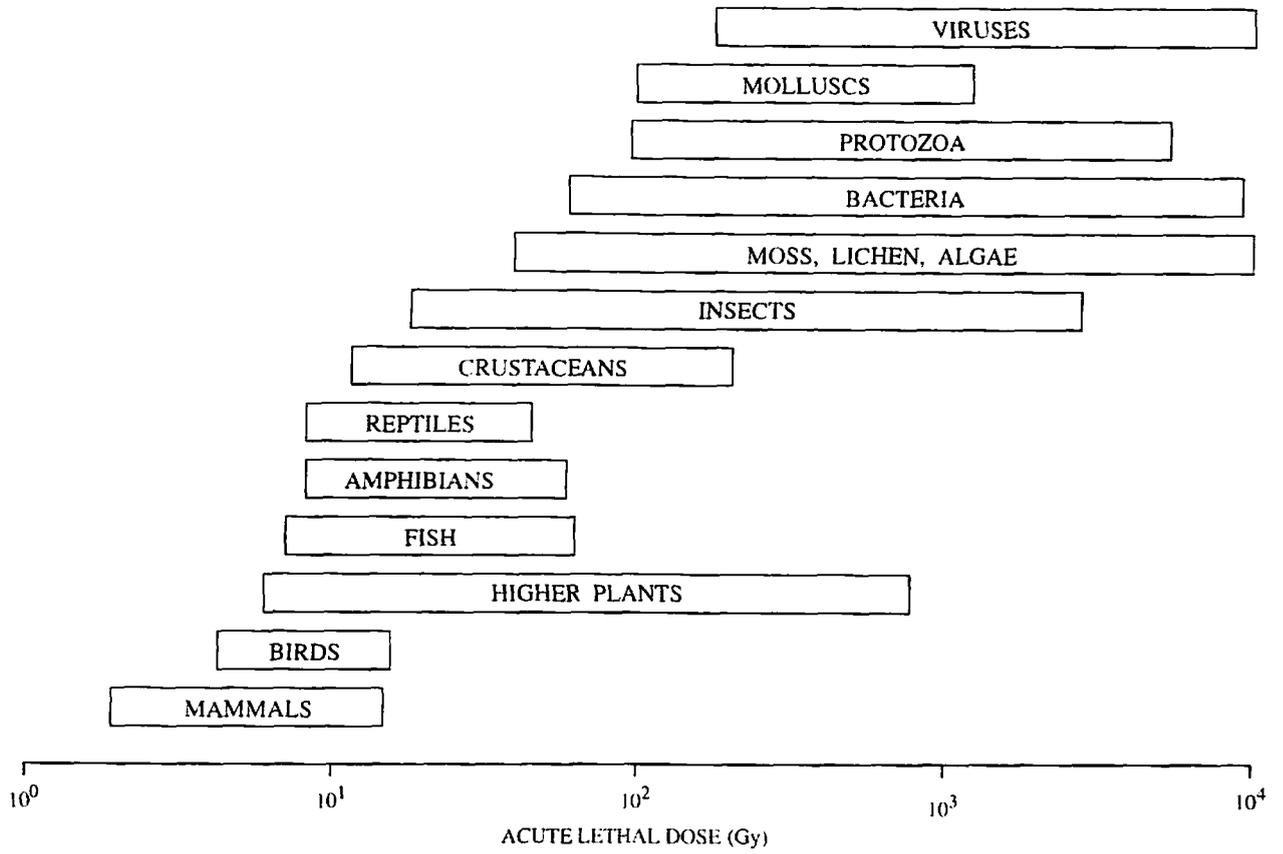
Figure VII.
Approximate acute lethal dose ranges for various taxonomic groups.
|S1. W5|

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