Effects of Ionizing Radiation on Plants and Animals at Levels Implied by Current Radiation Protection Standards
EFFECTS OF IONIZING RADIATION ON PLANTS AND ANIMALS AT LEVELS IMPLIED BY CURRENT RADIATION PROTECTION STANDARDS
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EFFECTS OF IONIZING RADIATION ON PLANTS AND ANIMALS AT LEVELS IMPLIED BY CURRENT RADIATION PROTECTION STANDARDS
FOREWORD

Releases of waste gases and liquids into the environment from facilities using radioactive materials are controlled so that people living in the vicinity are adequately protected from exposure to ionizing radiations. Principles for release control are described in IAEA Safety Series No. 77, Principles for Limiting Releases of Radioactive Effluents into the Environment (1986). It is implicitly assumed that protecting humans will also protect the environment in which they live. Although this assumption has never been formally defended, neither has it been seriously challenged. Nevertheless it is clearly desirable for the assumption to be supported by scientific argument and evidence to the extent possible.

This report examines the validity of the assumption for the case of radioactive releases to the terrestrial and freshwater environments and also for solid waste disposal underground. A similar study has been carried out in the context of the marine disposal of radioactive wastes (Assessing the Impact of Deep Sea Disposal of Low Level Radioactive Waste on Living Marine Resources, IAEA Technical Reports Series No. 288 (1988)).

The present study began in 1986 and involved several consultants meetings, an Advisory Group Meeting and a Technical Committee Meeting. The IAEA wishes to acknowledge the work of the experts who took part in the project (a full listing is given at the end of the report), especially W. Whicker (Colorado State University, United States of America), G. Blaylock (Oak Ridge National Laboratory, United States of America) and D. Woodhead (Ministry of Agriculture, Fisheries and Food, United Kingdom), who were largely responsible for the initial drafting and the revisions to the text after the comments of the Advisory Group and the Technical Committee. G.S. Linsley and H. Köhler, of the Division of Nuclear Fuel Cycle and Waste Management, were the responsible officers at the IAEA.
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1. INTRODUCTION

1.1. BACKGROUND

Radiation protection standards have been expressly developed for the purpose of protecting the health of human beings. The 1977 Recommendations of the International Commission on Radiological Protection (ICRP) contain the following statement:

"Although the principal objective of radiation protection is the achievement and maintenance of appropriately safe conditions for activities involving human exposure, the level of safety required for the protection of all human individuals is thought likely to be adequate to protect other species, although not necessarily individual members of those species. The Commission therefore believes that if man is adequately protected then other living things are also likely to be sufficiently protected." [1]

This assumption has been generally accepted and adopted by those involved with radiation protection standards, even though 'sufficient protection' has never been quantified nor the assumption proven. The ICRP clearly regards the assumption to be qualified rather than absolute. It is a prevailing viewpoint[1] which has often been expressed (e.g. [3, 4]) but not seriously challenged (except for a recent paper [5]) or formally defended. However, the assumption has been shown to be tenable at specific sites. Kaye [6] examined the environmental impact statements prepared for 16 nuclear power station developments in the United States of America for which the potential hazards to aquatic organisms in lakes, rivers and estuaries were assessed. It was concluded that, even though conservative assumptions had been employed, the estimated incremental radiation exposures from waste disposal would be less than the dose rates at which any significant biological effects would become apparent. A similar conclusion has been reached for the discharges to the northeast Irish Sea from the fuel reprocessing plant at Sellafield [7, 8].

There is little doubt that radionuclides in the environment can produce doses to certain organisms similar to or even substantially higher than doses to people living in and deriving sustenance from the same environment. Therefore, the risk of effects for natural biota (discounting variations in radiosensitivity, life span, etc.)

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1 More recently the ICRP has modified its statement on the subject as follows:

"The Commission believes that the standard of environmental control needed to protect man to the degree currently thought desirable will ensure that other species are not put at risk. Occasionally, individual members of non-human species might be harmed, but not to the extent of endangering whole species." [2]
would appear to be as high as, or higher than, for humans. However, there is a basic
difference in the manner in which we view the risk to people as compared with that
to other species. For people, our values are strongly focused upon the individual,
as individuals are considered to have great value and importance. In contrast, most
other species are viewed and valued more as populations than as identifiable
individuals. In certain cases, however, emphasis may be placed on individual orga-
nisms rather than on populations. For example, individuals of rare or endangered
species would warrant special consideration, and economic and other factors may
make individual domestic animals of particular value, although it is clear that the
criteria are different in each case. Generally though, a particular radiation dose may
produce occasional harmful effects to a few individuals in an irradiated population
without causing any noticeable deleterious effect on the population as a whole. This
dichotomy in the human value system must be recognized because it is perhaps the
basic explanation for the current philosophy on protection standards for species other
than man.

1.2. OBJECTIVE

The purpose of this report is to determine whether the statements of the ICRP
about the protection of non-human organisms and populations are consistent with
current knowledge [1, 2]. A clear resolution of this question would provide the basis
for determining whether or not radiation protection standards for aquatic and terres-
trial biota are warranted.

1.3. SCOPE

This report deals primarily with the potential effects on plant and animal popu-
lations in their natural environment in the case of routine, chronic releases of radio-
nuclides which are controlled to keep human exposures below specified limits. The
cases of accidental releases or possible releases to uncontrolled areas are not specifi-
cally considered. It is assumed in this report that releases to the environment are con-
trolled by reference to a group of humans who by virtue of their location and habits
are the most highly exposed in the population (the critical group). The biota of the
natural environment are assumed to share the same environment as the critical group.

This report does not deal with the marine environment, which was the subject
of a recent IAEA report in the context of radioactive waste disposal at sea [9].
1.4. APPROACH

The basic approach adopted is as follows: (a) the available information on the effects of ionizing radiation on natural organisms is reviewed (Section 2); (b) the radiation doses and/or dose rates above which there are deleterious effects on populations of different types of plants and animals are determined (Section 2); (c) the radiation doses and/or dose rates to plants and animals which result when releases of radionuclides are controlled on the basis of the standards for the protection of humans are estimated (Section 3); and (d) the radiation doses and dose rates in (b) and (c) are compared to establish whether or not plant and animal populations are afforded adequate protection under radiation protection standards for humans (Section 4).

1.5. HISTORICAL PERSPECTIVE

The effects of ionizing radiation from man-made sources on plants and animals have been of widespread concern since 1945, when the first nuclear detonations occurred. Because of this interest and the relative ease of conducting laboratory studies, much work on various species was completed in the ensuing decades [10–15]. Much of this work was directed at acute exposures of the individual organism where pathological responses were observed. The acute response was often quantified in terms of the dose which caused death in 50% of the exposed animals within a specified period. For mammals this interval is the 30 days following exposure, during which by far the greater part of the mortality is expressed, thus giving the LD$_{50/30}$. Beyond this period most of the exposed animals which have survived have essentially the same mortality pattern as the unexposed controls. In the late 1950s increased effort was focused on more ecologically relevant research, i.e. experiments with longer term exposures at much lower dose rates, with much greater attention being given to responses other than mortality. By the early to mid-1970s this effort was in decline, with many of the programmes being substantially reduced or terminated. Thus there was only about a decade and a half of sizeable effort in the USA, but fortunately research relevant to the ecological consequences of increased radiation exposure continued in Canada, France, Japan, the Union of Soviet Socialist Republics and the United Kingdom [16–20].

The general approaches employed to study radiation effects on populations and communities have included large field irradiation facilities, observations at nuclear test sites and areas of high natural background radiation or contamination, the release of irradiated individuals to the environment, experimental application of radioactive particles, and irradiation or contamination of laboratory microcosms. Each of these approaches has certain advantages, but most suffer from a lack either of ecological
realism or of a sufficient range of dose rates to produce unequivocal data suitable for extrapolation to actual situations of low level environmental contamination.

Large field irradiators have been successfully used to study population and community response to total dose or dose rate [21]. This approach has produced information on the response of certain terrestrial plant communities to irradiation over various time periods [22]. However, exposures from large field sources are confined to those from external γ radiation. Dose distributions within the organism from radioactive contaminants would be significantly different. Observations on contaminated or high natural background areas have not generally been as instructive, owing in some cases to insufficiently high dose rates and in others to the influence of confounding variables. Such confounding variables may include, for example, the presence of toxic metals, differences in microclimate or soil fertility and competition from other species. A notable exception is the radioactively contaminated White Oak Lake (a radioactive waste retention pond) at Oak Ridge National Laboratory (Tennessee, USA), where many useful data on aquatic organisms have been collected [23]. However, most work on effects on aquatic organisms and populations has been carried out in laboratory systems and microcosms.

The expense and difficulty of doing meaningful work on radiation effects on natural populations and communities in their natural environments have limited the ability to produce information on a large range 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 moss-lichen communities [22]. However, little work appears to have been done on aquatic plant communities, arctic or alpine tundra, taiga, savannahs, or desert communities.

Also, very little information is available in the area of interactive effects of natural or man-made radiation with other natural or man-made stresses or agents. This may be a serious omission in the light of the multiple forms of pollution which threaten many contemporary populations and ecosystems. This fact should be remembered in making assessments and in the development of protection standards for the environment. However, it should also be borne in mind that most field experiments with ionizing radiation have been performed in the presence of other pollutants and also that experiments with chemical pollutants were necessarily performed in the presence of natural background radiation.

Other areas which 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 might take considerable periods of time to express damage at the population and community levels, under chronic low level irradiation, and consequently most studies have probably been terminated prematurely. Several studies have investigated repair and recovery for a few years, but many systems require decades or more for complete recovery from severe damage.

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

1.6. PROTECTING SPECIES OTHER THAN MAN

The effects of ionizing radiation can be viewed at all levels of biological organization, ranging from the molecular level to ecosystems [24]. Effects at all higher levels of biological organization can be traced to molecular and cellular responses. However, molecular and cellular responses do not necessarily lead to observable effects at the population or ecosystem level. This is because the measurable attributes of the levels differ, despite the fact that all levels are interrelated [25]. In general, measurable changes in populations and communities (population assemblies) require rather severe effects at the cellular level for many individual organisms. For the structure of a biotic community to be altered requires a change in component populations, which in turn requires widespread mortality and/or reduced reproduction of individuals [22]. On the other hand, genetic or somatic mutations which can be produced by lower levels of exposure may have little or no impact on population or community performance, because of natural selection [4, 26-29] and convergence of genetic information among adjacent populations [30].

A basic premise of the discussions to follow is that the main concern for non-human species is focused at the population level of organization. This permits the possibility of deleterious effects at the individual level which are not sufficiently frequent to become manifest in population level responses. 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 [27, 31]. In such populations, normally only a small fraction of the individuals will mature and perpetuate the gene pool, even in the absence of enhanced radiation or other stresses. Genetic information that is altered by radiation is extremely unlikely to be perpetuated in the population, because it is unlikely to confer a selective advantage. Typically, measured attributes at the population level include numbers of individuals, mortality rate, reproduction rate, mean growth rate, etc.

Radiation effects at the population and community levels are manifest as some combination of direct changes from radiation damage and consequential responses to the direct changes. This seriously complicates the interpretation of radiation effects on organisms exposed to radiation in the natural environment. The wide range of radiosensitivities of organisms comprising most natural communities creates a condition where, if doses are such that sensitive species are affected but not the more
resistant ones, the latter may gain a significant competitive advantage and perhaps increase in abundance or vigour [32]. This could erroneously be interpreted as a hormetic (or stimulatory) 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.

Because of indirect ecological responses to radiation, if several species in a biotic community receive concurrent doses, it is necessary to look for the species most likely to exhibit direct effects. Populations and the community as a whole may be altered by radiation only because of the dependence on a much more sensitive species. For example, most species within a pine forest may be unaffected by a dose rate of 0.5 Gy·d⁻¹, but the pine component would experience severe mortality [33]. This would cause both positive and negative perturbations in species populations not directly damaged by irradiation. The phenomenon of indirect response leads to the concept of a 'critical species' in the context of the natural environment.

In addition to indirect responses to radiation, indirect effects produced by the physical interaction of ionizing radiation with the environment might be considered (e.g. the production of radicals in water and indirect results of altering the constituents of the atmosphere). However, at dose levels which are produced by routine radioactive releases, it is extremely improbable that these phenomena will have a sufficient degree of severity to result in measurable impacts on organisms and ecosystems [34].

It is recognized by many ecologists that the appropriate definition of a population of organisms is dependent entirely on the problem which is under consideration [35] and will be determined, among other things, by the scale of interest, in both space and time [36]. The following definition seems useful:

“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.” [35]

It is implicit within this definition that the population would be a self-sustaining unit of a particular species. In the present context, the "direct or indirect interactions" within the population would be broadened to include the responses to the incremental radiation exposures arising from the presence of increased quantities of radionuclides in the environment. Thus the increased radiation exposure might be experienced by either all of the population, e.g. most individuals of aquatic species inhabiting ponds and lakes, or only a part of the population for many terrestrial and marine species.
2. SUMMARY OF THE AVAILABLE INFORMATION ON THE EFFECTS OF IONIZING RADIATION ON POPULATIONS

2.1. METHODOLOGY

2.1.1. Methods for data acquisition

There are a number of potential sources of, and means of acquiring, information concerning the possible effects of radiation on populations of wild organisms. Each has certain advantages and limitations from the viewpoint of providing the basis for the protection of populations in environments contaminated with radionuclides.

The exposure of natural populations of terrestrial and freshwater organisms to controlled irradiation provides useful information, but it also presents many problems. It is seldom possible to provide a dose rate which is uniform or accurately known over a reasonably large area. This creates at least two problems. One is that the sample size becomes restricted, and this may reduce the power of statistical tests. Another is that ecological responses to stress over small areas may differ from responses to the same intensity of stress applied to larger areas. Many of the studies of radiation effects in natural terrestrial systems have employed single, large, sealed γ ray sources. These present the problem of a rapid decrease in intensity with distance according to the inverse square law and due to absorption and scattering [37-39]. For example, the exposure rate may decrease from several thousand röntgen per day a few metres from the source to \( \sim 1-2 \, \text{R} \cdot \text{d}^{-1} \) at 125 m [40]. Near the source, the trunks of trees and other natural objects can produce significant shielding; for example, a 10 cm diameter tree trunk may reduce the exposure by one half [41]. In these studies the detailed ecological data have been related to the average radiation exposure at any distance [33]. Essentially, the area being irradiated was divided into concentric rings of diminishing average exposure, and the organisms occupying these rings were the experimental populations from which the effects of irradiation were inferred. This approach has been considered successful with plants, but it is less satisfactory for animals, owing to their mobility and the relatively short distance between the area of acute mortality and that where no observable effects occur.

In an attempt to minimize these problems, shielding has been included in the design of a large source to provide a more uniform dose rate to plant and animal populations [42]. Although the shield design was not ideal, the variation in the dose rate was reduced from a factor of 120 to a factor of 10 over an annular area with radii of 15 m and 168 m.
Large sources have not been used to irradiate natural aquatic systems, owing to the greater problem arising from the large variations in dose rate due to the shielding by the water.

Useful information can be derived from studies of natural populations occupying areas which have become contaminated by radionuclides. The assessment of the radiation exposure experienced by the organisms presents a major problem, with the possibility of large temporal and spatial variations in the distributions of $\alpha$, $\beta$ and $\gamma$ emitting radionuclides both within the animals and in their external environment (see Section 2.1.2). Studies of this type have encompassed both terrestrial [24] and aquatic [27, 31] environments. The contamination may have resulted from controlled waste disposal (e.g. [23]), weapons testing (e.g. [43]), deliberate dispersion of radionuclides over experimental areas (e.g. [19]) or accidental releases (e.g. [44]).

A very similar source of information arises from areas of enhanced natural radioactivity although here, in addition to the problems posed by the assessment of the radiation exposure, the interpretation of the data is made more difficult by the potential influence of other environmental variables, such as the presence of toxic heavy metals or different soil properties. Such studies have been made primarily in terrestrial environments.

There is some information that can be gained from laboratory irradiation studies. These studies usually lack ecological realism and in the majority of cases the data obtained refer to individuals and means must be developed for translating these responses into an integrated effect at the population level. The advantages of laboratory studies are that variables can be better controlled, with the dosimetry for sealed $\gamma$ ray sources being relatively straightforward, and the different components of the radiation response can be studied in detail.

In certain cases it has been possible to combine the controlled radiation exposure of individuals with an otherwise essentially natural existence. For example, Donaldson and Bonham [45, 46] exposed eggs and alevins of Chinook salmon to radiation and then allowed the fingerlings to migrate to sea. The number, size and fecundity of returning irradiated fish were then compared with controls. Another example is the release of irradiated mammals into the areas from which they had been trapped so that their survival in the natural environment could be compared with controls [47].

2.1.2. Dosimetry

The estimation, by direct measurement or by calculation, of the dose rates experienced by organisms is a necessary requirement for an assessment of the potential impact of increased concentrations of radionuclides in any environment.

A number of different methods have been used to measure dose rates. These have been summarized for terrestrial ecosystems [22] and include: film [48],
phosphate glass rods [39], proportional counters and sulphur pellets [49], ion chambers and lithium or calcium fluoride thermoluminescent dosimeters [50–53]. Techniques for measuring the dose rates to aquatic organisms have been reviewed in Refs [23, 31, 54–56].

Such techniques are appropriate for external sources of γ radiation and large organisms but do not provide information concerning the dose rate from internal sources of α and β emitting radionuclides or the exposure of particular small targets, e.g. the gonads, the developing embryo or plant meristems. In these cases calculations employing suitable models have to be used.

Turner [21] has discussed the methods pioneered by Kaye and Dunaway [57] and employed by Hatch et al. [58] to estimate the dose rate from radionuclides accumulated in specific tissues of small mammals inhabiting areas contaminated with radionuclides. Aquatic organisms inhabiting a contaminated environment will receive external exposure from radionuclides in the water and accumulated by sediment and internal exposure from radionuclides assimilated from food and absorbed from the water. Models have been developed to calculate the dose rate to aquatic organisms by using the concentration of radionuclides in the water as an input quantity (e.g. [9, 59]). The methods for calculating the dose rates to aquatic organisms are summarized in a number of publications [9, 31, 54–56].

When estimates are made of the radiation exposures of animals from radionuclides incorporated within the body, under either natural or experimental conditions, a problem arises when there are contributions to the absorbed dose from α, β and γ radiations, since the biological effectiveness per unit absorbed dose of these radiations is not the same. For the purpose of this report the biological effectiveness of radiations in organisms other than humans is assessed using the quality factors recommended for humans, although it must be recognized that this procedure is open to debate. For α particles the quality factor may be taken to have a value of 20, and for X rays, γ rays and electrons a value of 1.

This report cites values of radiation 'exposure' or 'dose' from many different literature sources and a systematic means must be adopted for making the results from various studies comparable. Every cited figure not reported in gray has been converted and its gray equivalent is given in brackets after the original value. The approximations 100 rad = 1 Gy and 100 R = 1 Gy have been used.

2.1.3. Relevant biological responses

Before coming to a discussion of biological responses, it is helpful to consider the circumstances of the radiation exposures of relevance in the present context, i.e. to attach an operational meaning to the terms 'acute' and 'chronic' as descriptors of
the exposure. The following appear to be useful operational descriptors for the purpose of this report:

An *acute* exposure is one which is delivered in a time period which is short compared with the time over which any obvious biological response develops.

A *chronic* exposure is one which could continue over a large fraction of the natural life of the organism.

These descriptors as a function of time lead naturally to qualifiers of total dose:

A *high* exposure is one which leads to an acute response, usually a severe (and obvious) pathological reaction, with the primary one being mortality.

A *low* exposure would have only marginal and late effects on the normal mortality/time relationship of the organism, but it may produce detectable effects in the normal biological processes of the organism without necessarily producing any obvious harm to the individual.

Naturally, these descriptors are not totally clear-cut; they are contingent to some extent upon the organism under consideration. For example, a chronic exposure for a bacterium, or more realistically an aphid, might be an acute exposure for the host plant. Thus the life cycle of the organism and its radiosensitivity are factors which need to be taken into account.

Notwithstanding these conceptual problems, it is clear that an accident releasing radionuclides to the environment might, depending on its severity and situation, produce every combination of radiation exposure outlined above. On the other hand, it is also clear that discharges of radionuclides to the environment which are controlled in accordance with the ICRP system of (human) dose limitation will generally result in chronic low dose rate exposure of populations of wild organisms. It is the latter circumstance which is the main concern of this report. Although it is perhaps obvious, it needs to be stated that there can be no effects at the population level if there are no detectable effects in the individuals making up that population. Similarly, there can be no effects in a community if there are no effects in a component population, and so on, up through the hierarchy of levels of biological organization. If it is accepted that it is the population of an organism that is the object of protection then there is no need to be concerned about any responses at the community and ecosystem levels provided that protection is given to the most sensitive component population.

Thus it appears necessary to consider those attributes of individuals which might be affected by long term, low level irradiation and which have implications for the maintenance of the population. Such attributes clearly include mortality, fertility, fecundity (all age dependent), growth rate, vigour and mutation rate; and it must be stressed that all of these attributes can be influenced by environmental factors other than radiation.
It has been implied above that studies of the effects of acute and high dose irradiation have rather little relevance to the main concerns of this report, but such studies do provide insights and information which are not readily obtainable by other means. The available data can be used to show:

(a) Whether acute lethality could be a problem in the context of planned controlled releases, providing a basis for assessing the potential impact of accidental releases.

(b) The relative radiosensitivities of different species, provided that the responses have been determined on a strictly comparable basis. For LD$_{50}$ data, this means that the period of observation should have been sufficiently long for all the acute mortality to be manifest. For example, in the case of poikilothermal fish a 60 or 90 day period is more appropriate than the 30 day period normally employed for mammals.

(c) Whether there are effects which have implications for populations, e.g. effects on reproductive capacity, at lower acute doses than those which produce mortality.

In addition, this may often be the only experimental approach which can be used to investigate conveniently the interactions between irradiation and other environmental stresses, including other contaminants.

Under natural conditions, the population of a species will vary in numbers and age distribution in a complex way in response to both the ever changing pattern of environmental variables and the intra- and interspecific pressures within the community. To ensure protection at the population level, it will be necessary to demonstrate that any radiation-induced changes in the population attributes of individuals will be unlikely to influence the normal, but variable, structure of the population.

When populations are exposed to radiation or radionuclides in their natural environments, the simultaneous exposure of other species is unavoidable. Populations that are members of the same community have evolved interactive relationships including, for example, competition, predation and mutualism [24]. Because of these relationships, and the fact that species differ in their sensitivity to radiation and in their tendency to accumulate radionuclides, the response of any one population to environmental radiation exposure may be affected not only by the direct effects of the exposure but also by the radiation responses of other populations. Thus the phenomena of direct and indirect effects need to be recognized [21]. The effects of radiation on attributes such as natality and mortality are often interpreted as direct effects [60], but in the community context these parameters may actually reflect both direct and indirect effects.

A specific example of an indirect radiation effect is the alteration of the composition and structure of the vegetation that leads to changes in the animal populations [61–63]. Such changes could also involve interacting 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. Perhaps the work done in France has demonstrated these kinds of change most clearly [64–67].

2.2. EFFECTS ON TERRESTRIAL POPULATIONS AND COMMUNITIES

Terrestrial (land dwelling) populations considered in this summary include plants, mammals, birds, reptiles, amphibians and invertebrates. The intent is to identify acute doses and chronic dose rates below which the likelihood of observing population level effects is remote. It is not necessary to present here all the original relevant research because extensive reviews are available [21, 22, 24, 68, 69]. This summary is divided into experiments involving acute or short term and chronic irradiation, and observations in areas of high natural background radiation or significant anthropogenic contamination.

2.2.1. Effects of acute irradiation

2.2.1.1. Plants

Studies of the effects of irradiation on natural plant populations and communities are difficult to place within the conventional categories of either acute or chronic exposure. There are severe practical difficulties in delivering total doses amounting to tens of grays over large areas in minutes or, at most, a few hours. The studies which have been made have involved continuing exposures at a wide range of dose rates. This has resulted in a few plants near to the source receiving sufficiently high total doses in sufficiently short periods of time for the doses to be described as acute in the sense defined in Section 2.1.3. In the same experiments, however, plants at a distance from the source experience genuinely chronic radiation exposure. Thus, care is required in interpreting the various data derived from these experiments in the context of the main requirements of this report.

The experiments have shown that pine trees (genus *Pinus*), for example, are among the most sensitive to irradiation. All specimens of *P. elliottii* which received $\geq 300 \text{ R (} \geq 3 \text{ Gy)}$ in 200 h died within a few months of exposure. All specimens of *P. palustris* less than five years old receiving doses of $\geq 800 \text{ R (} \geq 8 \text{ Gy)}$ also died; older trees of this species died after exposures of $> 2800 \text{ R (} > 28 \text{ Gy)}$ [70]. In this experiment the exposures at the highest dose rates, $\sim 0.7 \text{ Gy} \cdot \text{h}^{-1}$ to trunks and $\sim 0.22 \text{ Gy} \cdot \text{h}^{-1}$ at canopy height, may perhaps be classified as acute, but the lower dose rates represented short term chronic exposure (down to 0.01 Gy·h⁻¹). A second study [33], again with dose rates ranging from several thousand röntgen per
day (~1 Gy·h⁻¹), delivering acute exposures within a few metres of the source through to genuinely chronic exposure, confirmed the relatively higher radiosensitivity of the pines (P. rigida). The deciduous trees (the oaks Quercus alba and Q. coc-cinea), the shrubs (particularly Vaccinium vacillans) and the herbs (particularly Carex pensylvanica) showed progressively less radiosensitivity, although it should be noted that within each broad plant grouping there were substantial variations in sensitivity. In this study the irradiation was extended over four years and the chronic dose rates causing mortality decreased with time such that the 50% lethal dose rates for whole trees after 32 months of exposure were ~7.4, 35 and 55 R·d⁻¹ (0.07, 0.35 and 0.55 Gy·d⁻¹) for P. rigida, Q. alba and Q. coccinea, respectively. After four years of exposure, the dose rates causing a 50% reduction in tree crown condition (sprouting buds) had declined to ~3.9, 7.8 and 6.2 R·d⁻¹ (0.04, 0.08 and 0.06 Gy·d⁻¹) for the three species, although such damage does not necessarily imply mortality for the trees. Few data were obtained concerning either the production of pollen or egg cells or the viability of the seeds, although it was noted that flowering and seed setting were common for C. pensylvanica in the dose rate range 17-160 R·d⁻¹ (0.17-1.6 Gy·d⁻¹).

In addition to providing data on the radiation responses of individual plants, these and other studies yielded information concerning the responses at the population and/or community level. The indices determined included:

1. The coefficient of community (CC), which measures qualitative changes in species composition (i.e. the presence or absence of species):

   \[ CC = \frac{c}{a + b - c} \]

   where
   
   - \( a \) is the number of species in the plot before irradiation,
   - \( b \) is the number of species in the plot after irradiation,
   - \( c \) is the number of species in the plot both before and after irradiation.

2. The similarity index (S), which considers changes in the abundance of individuals within species, as well as in the number of species:

   \[ S = 1.0 - 0.5 \sum_{i=1}^{n} (a_i - b_i) \]

   where
   
   - \( a_i \) is the density of individuals of species \( i \) in the plot before irradiation,
   - \( b_i \) is the density of individuals of species \( i \) in the plot after irradiation,
   - \( i \) is the index for the species,
   - \( n \) is the total number of species in the plot.
(3) The diversity index \( (H) \), which measures the balance among individuals of different species, as well as the number of species:

\[
H = -k \sum_{i=1}^{n} P_i \ln P_i
\]

where

\( P_i \) is the probability of sampling the \( i \)th species,
\( n \) is the total number of species in the plot,
\( k \) is a constant [71].

(4) The biomass index \( (B) \) is measured as the dry above ground mass of biological tissue per unit ground area.

(5) The leaf fall index \( (L) \) is measured as the dry mass deposited per unit ground area.

These quantities are explained in detail in Ref. [22].

Most of the available data on terrestrial plant communities are summarized in Table I. Among the communities studied, the pine forest appears to be the most sensitive, with a threshold total dose of \( \sim 300 \) R (\( \sim 3 \) Gy) causing changes in CC [70]. It should be noted that in the oak–pine forest, where the pine was again the most sensitive species, the value of CC was unaffected at exposure rates less than 50 R·d\(^{-1}\) (<0.5 Gy·d\(^{-1}\)) over a period of 18 months [33].

The radiosensitivity of \( \text{Pinus} \) and other coniferous trees has been correlated with the large chromosomes of these plants [78]. The other communities listed in Table I are much more resistant, possibly because they are dominated by species having generally smaller chromosomes. Lichen dominated communities are exceptionally resistant, and this resistance may be explained by the presence of diffuse centromeres and an asexual mode of reproduction in addition to small chromosomes [22].

The radiation sensitivities of cultivated plants, such as vegetables, grains and fruit trees, are in general similar to those of closely related species that occur in the wild. Such radiation sensitivities are predictable to within a factor of perhaps two from cellular characteristics, particularly the interphase chromosome volume [79].

Lettuce (\( \text{Lactuca sativa} \)) has been found to have an \( \text{LD}_{50} \) of \( \sim 5 \) kR (\( \sim 50 \) Gy), while barley (\( \text{Hordeum vulgare} \)) and wheat (\( \text{Triticum aestivum} \)) have \( \text{LD}_{50} \) values of \( \sim 2 \) and 3 kR (\( \sim 20 \) and 30 Gy), respectively [80]. For peach and apple tree buds and seedlings, \( \text{LD}_{50} \) values ranged from 3.2 to 15 kR (32–150 Gy), depending on the stage of leaf development [81]. Since cultivated plants are highly manipulated by man, the primary concern is with effects on small populations and perhaps individuals in some cases. Therefore, most work on cultivated plants has stressed effects on mortality and growth, rather than on more ecologically relevant parameters, such as reproductive capacity or community level changes.
TABLE I. MINIMUM \( \gamma \) RAY EXPOSURES AND EXPOSURE RATES OBSERVED TO PRODUCE DETECTABLE EFFECTS IN TERRESTRIAL PLANT COMMUNITIES

<table>
<thead>
<tr>
<th>Community type</th>
<th>Exposure period (d)</th>
<th>Attribute measured</th>
<th>Minimum exposure rate ( (R \cdot d^{-1}) )</th>
<th>Minimum total exposure ( (kR) )</th>
<th>Ref.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pine forest</td>
<td>8</td>
<td>CC</td>
<td>375</td>
<td>0.3</td>
<td>[70]</td>
</tr>
<tr>
<td>Oak–pine forest</td>
<td>540</td>
<td>CC</td>
<td>50</td>
<td>27</td>
<td>[33]</td>
</tr>
<tr>
<td></td>
<td>900</td>
<td>( H )</td>
<td>50</td>
<td>45</td>
<td>[33]</td>
</tr>
<tr>
<td></td>
<td>1440</td>
<td>( L )</td>
<td>2</td>
<td>2.9</td>
<td>[22]</td>
</tr>
<tr>
<td>Deciduous forest</td>
<td>165</td>
<td>( B )</td>
<td>24</td>
<td>4</td>
<td>[72]</td>
</tr>
<tr>
<td>Tropical forest</td>
<td>34</td>
<td>( B )</td>
<td>118</td>
<td>4</td>
<td>[73]</td>
</tr>
<tr>
<td>Old fields</td>
<td>17</td>
<td>( S, H )</td>
<td>59</td>
<td>1</td>
<td>[74]</td>
</tr>
<tr>
<td>(abandoned cropland)</td>
<td>29</td>
<td>CC</td>
<td>1200</td>
<td>35</td>
<td>[75]</td>
</tr>
<tr>
<td></td>
<td>29</td>
<td>( B, S, H )</td>
<td>586</td>
<td>17</td>
<td>[75]</td>
</tr>
<tr>
<td></td>
<td>365</td>
<td>CC</td>
<td>50</td>
<td>18</td>
<td>[32]</td>
</tr>
<tr>
<td></td>
<td>365</td>
<td>( H )</td>
<td>100</td>
<td>36</td>
<td>[32]</td>
</tr>
<tr>
<td>Meadow vegetation</td>
<td>11</td>
<td>CC</td>
<td>227</td>
<td>2.5</td>
<td>[20]</td>
</tr>
<tr>
<td>Shortgrass plains</td>
<td>30</td>
<td>CC</td>
<td>467</td>
<td>14</td>
<td>[51]</td>
</tr>
<tr>
<td></td>
<td>30</td>
<td>( H, B )</td>
<td>300</td>
<td>9</td>
<td>[51]</td>
</tr>
<tr>
<td></td>
<td>420</td>
<td>CC</td>
<td>120</td>
<td>50</td>
<td>[51]</td>
</tr>
<tr>
<td></td>
<td>420</td>
<td>( H )</td>
<td>40</td>
<td>95</td>
<td>[51]</td>
</tr>
<tr>
<td></td>
<td>510</td>
<td>( B )</td>
<td>170</td>
<td>87</td>
<td>[51]</td>
</tr>
<tr>
<td>Lichen</td>
<td>92</td>
<td>( S, B )</td>
<td>2200</td>
<td>200</td>
<td>[76]</td>
</tr>
<tr>
<td></td>
<td>780</td>
<td>( CC, H )</td>
<td>300</td>
<td>234</td>
<td>[77]</td>
</tr>
</tbody>
</table>

In summary, it appears that in the natural environment the most sensitive plants display acute radiation sensitivities which are similar in magnitude to those found for mammals, but that the majority of data relate to radiation exposures which are not acute for the plant species investigated but are more correctly described as short term or chronic.
2.2.1.2. Mammals

In the case of mammals, most work relevant to populations has involved studies of lethality. Sacher and Staffeldt [82] summarized data for numerous small mammal species. The $LD_{50/30}$ values ranged from ~500 to 1100 R (~5–11 Gy). Direct mortality has been observed in individuals at acute whole body doses down to ~200 rad (~2 Gy). Considerable work has also been done on reproduction, and the majority of results suggest that natality is a more radiosensitive parameter than mortality [83]. Minimum acute doses required to depress reproduction rates may be less than 10% of the doses required to produce direct mortality [24]. Various factors such as competition, hibernation, degree of confinement and temperature can modify the mammalian response to acute radiation, but such modifications appear insufficient to cause significant effects on mortality at acute whole body doses below ~100 rad (~1 Gy).

The basic radiosensitivity of domestic mammals, in terms of lethality, appears similar to that of the wild mammals which have been studied. Considerable work on the lethal effects of acute radiation exposure of domestic livestock was completed by Bell et al. [84]. These investigators reported whole body $LD_{50/60}$ values for $\gamma$ irradiation in the range 4–7 Gy for sheep, cattle, pigs and horses. One of numerous factors affecting $LD_{50}$ values is dose rate. For example, reported $LD_{50/60}$ values for sheep range from >1 kR (>10 Gy) delivered at <1 R·h$^{-1}$ (<10 mGy·h$^{-1}$) to 250 R (2.5 Gy) delivered at 650 R·h$^{-1}$ (6.5 Gy·h$^{-1}$) [85].

Acute doses of 50 R (0.5 Gy) produced slight changes in bull semen [86], but recovery was virtually complete by 30 weeks after irradiation. Sheep irradiated with 25 and 10 R (0.25 and 0.1 Gy) every 28 days for 13 months appeared capable of normal reproduction, and all lambs born to irradiated parents were healthy [87]. Species vary greatly in the radiosensitivity of the ovary, but mice are among the most sensitive. In studies with mice, Gowen and Stadler [88] found that reproduction was impaired by doses down to 20 R (0.2 Gy) for females. Male mice were less sensitive, requiring doses of over 320 R (3.2 Gy) to impair reproduction. Permanent sterility in female mice was produced by 100 R (1 Gy) [89].

2.2.1.3. Birds

The literature on the effects of ionizing radiation on birds has been summarized by Mellinger and Schultz [90]. In terms of mortality, birds (including domesticated varieties) appear to exhibit $LD_{50/30}$ values in the range 4.6–30 Gy [91]. Bell et al. [84] also report an $LD_{50/60}$ value of 900 R (9 Gy) for domestic fowl. Brisbin [92] exposed 2-day-old broiler chicks (domestic fowl) to $^{60}$Co $\gamma$ rays (8 R·min$^{-1}$) and measured the subsequent average growth rate over 30 d. It was decreased at all doses, but significantly so only at doses above ~700 R (~7 Gy). Irradiation of tree swallow ($Tachycineta bicolor$) and house wren ($Troglodytes aedon$) nestlings
immediately after hatching has shown that growth through the nestling stage is
unaffected by a total dose of 0.9 Gy but may be slightly depressed at doses greater
than 2.6 Gy [93, 94].

There have been few studies of the effects of acute irradiation on the reproduc-
tive capacity of birds. In male weaver finches (*Quelea quelea*), exposures of ~50
and 210 R (~0.5 and 2 Gy) produced no testicular damage, but ~420 R (~4 Gy)
induced apparent abnormalities [95]. The effects on egg production of 60Co γ
irradiation have been studied in white leghorn chickens, Japanese quail and bobwhite
quail [96]. The total dose, rather than the dose rate, was the main determinant of
the changes in subsequent egg production. In white leghorn chickens, a dose of
400 R (4 Gy) reduced egg production for 10 days after exposure. At higher doses
the effects were greater and longer lasting. The two species of quail appeared to be
more sensitive. Similar results have been reported for the Japanese quail [91] and
the white leghorn chicken [97], although the latter study also showed that a reduction
in the dose rate to 1 R·min⁻¹ (10 mGy·min⁻¹) eliminated any effects of a total
exposure of 800 R (8 Gy) on subsequent egg production. This total exposure had no
effect on the hatchability of the eggs produced for 14 d after irradiation until the
exposure rate exceeded 25 R·min⁻¹ (0.25 Gy·min⁻¹). The reproductive perfor-
mane of the progeny was not affected [97]. The LD₅₀ values at hatching have been
determined for artificially incubated chicken and black-headed gull (*Larus
ridibundus*) embryos irradiated at day 10 of development, giving values of 12–13 Gy
and 9 Gy, respectively [98]. A similar study, but with natural incubation, has been
conducted with swallow (*Tachycineta bicolor*) embryos irradiated at 7–8 days of
development. Total doses up to 3.2 Gy did not affect hatching or fledgling success,
but incubation times were slightly increased and growth was slightly depressed after
total doses of > 1.6 Gy [99].

It should be noted that there are no data at total acute doses below 0.5 Gy,
although most of the effects studied show thresholds at higher doses.

2.2.1.4. Reptiles and amphibians

The literature on reptiles and amphibians suggests that these groups are some-
what less sensitive to acute radiation in regard to lethality than birds and mammals,
although there is substantial overlap in sensitivity [79]. A similar comparison for
reproduction has not been made, but it is likely that the response for reproductive
effects is roughly similar to that of mammals [100].

2.2.1.5. Invertebrates

A very large database exists for radiation effects on invertebrates and espe-
cially on insects. O’Brien and Wolfe [68] prepared a monograph on the subject and
concluded that insects are, in general, far less sensitive to radiation than vertebrates.
To produce lethality in adult insects usually requires about 100 times the dose required for vertebrates. This difference has generally been ascribed to the fact that there is very little cell division and differentiation in progress in adult insects. The gonadal cells of adult insects do divide, however, and it is found that reproduction can be impaired at much lower doses. Juvenile insect forms are much more sensitive to the lethal effects of radiation, as would be predicted from the high cell turnover rates in these age classes. Many factors have been shown to modify the response of insects to radiation; however, it is very unlikely that species more sensitive than vertebrates to either the lethal or the reproductive effects of radiation will be observed.

2.2.1.6. Summary

The most relevant points with regard to the acute effects of ionizing radiation on terrestrial organisms can be summarized as follows:

(a) Reproduction (encompassing the processes from gametogenesis through to embryonic development) is likely to be the most limiting end point in terms of survival of the population.

(b) Lethal doses vary widely among different populations, with birds, mammals and a few tree species being the most sensitive among those considered here.

(c) Acute doses of 0.1 Gy or less are very unlikely to produce persistent, measurable deleterious changes in populations or communities of terrestrial plants or animals.

2.2.2. Effects of chronic irradiation

2.2.2.1. Plants

It has already been noted (Section 2.2.1.1) that the majority of data obtained from radiation studies of natural plant communities relate to chronic exposure regimes, and these have been summarized in Table I. It can be seen that there is substantial variation among the plant communities in their sensitivities to chronic radiation exposure. Within the constraints of the attributes measured, the oak–pine forest appeared to be the most sensitive, with exposure rates down to ~2 R·d\(^{-1}\) (~20 mGy·d\(^{-1}\)) producing measurable changes in litter production and leaf fall [22]. Again, this effect was mainly caused by the responses of the radiosensitive pine trees. The lichen community shows by far the greatest radiation resistance.

While a reduction in productivity and leaf fall may be a sensitive indicator of stress in the oak–pine community, there may be little impact on the community structure in the long term. This would be particularly true if only a small part of the community experienced dose rates sufficiently high (~20 mGy·d\(^{-1}\)) to induce this
response. However, it must also be remembered that a significant change in leaf fall (and litter production) could have implications for the ground dwelling invertebrate populations which, while relatively insensitive to the direct effects of radiation, could respond indirectly to the exposure through the change in food supply.

The data in the literature also indicate another aspect of the responses of organisms to radiation: the total accumulated dose at which a given response was observed increased as the dose rate declined (and the exposure became more protracted). This is more clearly illustrated in Fig. 1, where the LD50 value and corresponding dose rates for various species of pines are plotted as a function of the total exposure time. Overall it would appear that there are unlikely to be any detrimental long term effects on plant communities in which the maximum dose rate is on the order of 10 mGy·d⁻¹ or less.

The sensitivities to chronic irradiation of cultivated crops and other plants manipulated by man are similar to those of related species which grow wild. The general principles of cellular radiobiology as discussed in Ref. [79] can be used to predict, within a factor of 2–3, the exposure rates required to cause deleterious changes in cultivated species.
2.2.2.2. Mammals

With regard to the effects of chronic radiation exposure on animal populations, the literature that appeared prior to 1974 was carefully summarized by Turner [21]. The experimental approaches have been varied, ranging from observations of populations occupying radioactive areas to those in fields irradiated with large point sources. After a detailed and critical consideration of the available data, it was concluded that reproduction was the population attribute most sensitive to damage from chronic irradiation and also the attribute of greatest significance in the ecological context. A number of data indicated that the long lived species in which reproductive activity was spread over a number of years would be the most sensitive to radiation stress. The studies reviewed (e.g. [105]) indicated that a dose rate of 1 rad·d$^{-1}$ (10 mGy·d$^{-1}$) represented an approximate threshold at which effects became apparent in the irradiated populations, and it was very tentatively concluded that annual doses of several grays would be required to bring about the extinction of the most sensitive populations.

A later study [106] of the mammals in an irradiated hardwood forest detected no effects on survival at dose rates up to a maximum of 2 R·d$^{-1}$ (20 mGy·d$^{-1}$). In an irradiated boreal forest, the expected effects of chronic irradiation at average dose rates up to 15 mGy·h$^{-1}$ on redback voles (*Clethrionomys gapperi*) appeared to be totally masked by immigration from the surrounding unirradiated area [17]. This result underlines the importance of understanding the dose rate (and total dose) distribution throughout the population when attempting an assessment of the potential effects of the exposures on the population.

Garner and Barber [69] reviewed the literature concerning the effects of chronic radiation exposure in dogs and farm animals. At 10 R·d$^{-1}$ (0.1 Gy·d$^{-1}$), pigs and donkeys showed some deterioration in a few weeks and died after a few months of continuous exposure. At an exposure rate of 0.1 R·d$^{-1}$ (1 mGy·d$^{-1}$) no effects were observed. However, in other experiments chronic exposure of 3 rad·week$^{-1}$ (~4 mGy·d$^{-1}$) produced measurable declines in the number, motility and viability of sperm in the dog [4], while exposure rates of <0.12 R·d$^{-1}$ (<1.2 mGy·d$^{-1}$) failed to produce sperm count changes in dogs [107].

Experimental chronic whole body γ radiation was applied by De Boer [108] to mice at dose rates of 2–20 R·d$^{-1}$ (20–200 mGy·d$^{-1}$), and many parameters indicating damage to the reproductive system were measured. Relative to controls, nearly all parameters were affected by all dose rates, including 20 mGy·d$^{-1}$. In contrast to the results of work on acute exposure, males were more adversely affected than females by chronic irradiation. Continuous irradiation of ten generations of mice at 1.3–2.6 R·d$^{-1}$ (13–26 mGy·d$^{-1}$) did not produce changes in litter sizes or sex ratios of the progeny as compared with control mice [109]. Exposure rates of 5 R·d$^{-1}$ (50 mGy·d$^{-1}$) for four generations of albino rats failed to affect reproductive capacity [110].
Overall it may be concluded that a dose rate of \( \sim 10 \text{ mGy} \cdot \text{d}^{-1} \) represents the threshold at which slight effects of radiation become apparent in those attributes, e.g. reproductive capacity, which are of importance for the maintenance of the population. The laboratory studies tend to indicate a slightly higher threshold, but this may be due to other stresses being fewer or less severe than those experienced by natural populations.

2.2.2.3. Birds

Studies of chronic irradiation of bird populations are inherently more difficult because of birds' mobility; hence relatively little work has been done in this area. A few investigators (e.g. [111, 112]) have studied the nesting success of passerine birds in irradiated ecosystems. In these studies, exposure rates of 21 R \cdot \text{d}^{-1} (0.2 \text{ Gy} \cdot \text{d}^{-1}) caused embryonic mortality. In contrast, the breeding success of swallows and wrens exposed to 18–160 \( \mu \text{C} \cdot \text{kg}^{-1} \cdot \text{d}^{-1} \) (\( \sim 0.7–6 \text{ mGy} \cdot \text{d}^{-1} \)) appeared essentially normal [113]. However, large dose rates (1 \text{ Gy} \cdot \text{d}^{-1}) reduced hatching success [99]. Longevity was not investigated in these studies. The minimum chronic exposure level at which effects on reproduction or mortality would become manifest does not seem to be well established.

2.2.2.4. Reptiles

Turner et al. [114] studied populations of lizards in irradiated and control enclosures in a desert ecosystem. A chronic radiation exposure of \( \sim 2 \text{ R} \cdot \text{d}^{-1} \) (\( \sim 20 \text{ mGy} \cdot \text{d}^{-1} \)) was maintained in the enclosure. After five years of chronic irradiation, no significant differences in sex ratios, age distributions or life spans were found between irradiated and control iguanid lizards. After one or two years, however, females of two other lizard species occupying the same enclosures became sterile, reproduction was blocked and the populations later drifted towards extinction. The varied responses of the species examined may be related to life history differences. The iguanid lizards usually mature much earlier, produce more egg clutches and die earlier than the two longer lived species which became extinct. Perhaps the fact that the ovaries of the two affected species would accumulate a greater total dose before sexual maturation is at least a partial explanation of the results.

2.2.2.5. Invertebrates

In the case of invertebrates, some work has been done with respect to chronic exposures in natural environments (e.g. [62, 63, 115–118]). One general conclusion is that invertebrates appear to be more affected by indirect than direct effects (see
Ref. [21] for a review of these studies). Exposure rates that significantly alter vegetation structure or character may not have a direct impact on the invertebrates. However, these animals exhibit clear responses, both negative and positive, to the vegetation changes. This is not surprising in view of the radioresistance and general resilience of invertebrates. An example of an indirect chain of effects is exhibited in the work of Tabone and Poinsot-Balaguer [66], where radiation killing of trees and shrubs led to reduced litter production. Reduced litter decomposition led to depletion of carbon and nitrogen. Such reductions in resources would be manifest in alterations of the entire ecosystem, including invertebrates.

As noted for mammals and reptiles, the direct effects of radiation on insect populations are mostly likely to be limited by responses in fertility rather than mortality. This was clearly demonstrated for adult collembolans (Folsomia sp.) by Styron and Dodson [119]. Teresi and Newcombe [120] reviewed literature on the effects of fallout β radiation on insects and reached the same general conclusion. Genetic effects on insect populations from chronic irradiation are not likely to be more important than effects on fertility. Even severe genetic damage, such as that noted for Drosophila populations maintained in the Marshall Islands during and after nuclear testing, was reparable in succeeding generations (Stone and Wilson [121] as cited by Green [122]).

2.2.2.6. Summary

Some relevant generalizations on the effects of chronic radiation on terrestrial organisms can be summarized as follows:

(a) Reproduction (including the processes from gametogenesis through to embryonic development) is likely to be the most limiting end point in terms of population maintenance.

(b) Sensitivity to chronic radiation varies markedly among different taxa; certain mammals, birds, reptiles and a few tree species appear to be most sensitive.

(c) Indirect responses to radiation-induced changes in vegetation appear more critical than direct effects in the case of invertebrates.

(d) Irradiation at chronic dose rates of 1 mGy·d⁻¹ or less does not appear likely to cause observable changes in terrestrial animal populations.

(e) Irradiation at chronic dose rates of 10 mGy·d⁻¹ or less does not appear likely to cause observable changes in terrestrial plant populations.

(f) Reproductive effects in long lived species with low reproductive capacity may require further consideration.
2.3. EFFECTS ON AQUATIC ORGANISMS

2.3.1. Introduction

It is not necessary here to give a detailed discussion of all the published data on the effects of radiation on aquatic organisms because a number of more or less comprehensive reviews have already been compiled from a variety of viewpoints, and their conclusions provide a sufficient basis for the present summary.

The earliest review was by Donaldson and Foster [123] and it was set in the context of a broad ecological consideration of the behaviour of radionuclides present in the ocean as a consequence of radioactive waste disposal. Although the volume of data available was relatively small, and was mainly derived from experiments employing acute exposures, some generalizations and well defined conclusions were drawn. The general pattern of increased radiosensitivity (as measured by induced mortality) in groups of advanced aquatic organisms relative to more primitive phylogenetic forms was established. It was also noted that there were substantial variations in sensitivity with life stage within species: in general, the gametes and the developing embryo were less resistant than the adult organism. Of greater relevance in the context of the present work were the findings that non-lethal exposures of developing embryos reduced the numbers of primordial germ-cells present in larval fish, and that the fecundity of adult fish was reduced by non-lethal doses. All of these results were expanded in later studies. It was concluded that damage to aquatic organisms in the vicinity of radioactive waste discharges would be unlikely in the presence of adequate controls limiting human exposure. However, it was demonstrated that such controls would need to be extended beyond consideration of lakes, rivers and streams solely as sources of drinking water to take account of the actual or potential use of fish or other aquatic organisms as sources of food for human populations. Protection of the aquatic environment from the effects of radioactive waste disposal through control of human exposure should consider all aspects of human utilization of that environment.

Bacq and Alexander [10] placed the radiosensitivity of aquatic organisms in the context of a wide ranging summary of the biological effects of radiation. It was concluded that there were no qualitative differences in the responses of aquatic organisms and the phylogenetic dependence of radiosensitivity was confirmed.

Polikarpov [124] considered the effects of radiation on aquatic organisms in an extensive review of the behaviour of radionuclides in the aquatic environment. The notable conclusion drawn in this review concerned the apparent radiosensitivity of developing fish eggs exposed to the radiations from very low concentrations of radionuclides in the water. However, it is to be noted that no dosimetric basis for this conclusion was given.

Templeton et al. [27] and Chipman [125] discussed the data on the effects of radiation on marine organisms and pointed out the apparently low radiosensitivity of
fish compared with other vertebrates, particularly mammals. However, the 30 day assessment period for mortality was inappropriate for poikilothermic aquatic organisms, and an increased assessment period reduces the difference between the responses of fish and mammals [126].

A report by the IAEA [31] not only provided a comprehensive review of the literature on the effects of radiation on aquatic organisms but also made an attempt to provide sound estimates of the radiation exposures of a variety of representative aquatic organisms from both natural and contaminant radionuclides. These studies provided the basis for an assessment of the potential effects of waste disposal on natural populations of aquatic organisms. It was concluded that maximum dose rates of \( <1 \text{ rad}\cdot\text{d}^{-1} (\text{<10 mGy}\cdot\text{d}^{-1}) \) would be very unlikely to result in any significant deleterious effects on aquatic populations.

Blaylock and Trabalka [23] reviewed the available studies on the genetic effects of radiation in aquatic organisms. It was concluded that the radiation-induced mutation rates were similar to those which had been determined for terrestrial animals and that the pressures of natural selection would eliminate such mutations before any effect could become apparent at the population level. The earlier data concerning the effects on developing fish embryos of irradiation from radionuclides present in the water were also carefully reviewed and it was concluded that none of the experiments supported the contention [124] that this developmental stage showed exceptional radiosensitivity. Woodhead [56] also considered this subject and concluded that, for the majority of experiments, comparisons were not possible because there were insufficient data to make valid estimates of either the radiation dose rate or the total dose experienced by the developing embryo. In the few cases where valid estimates of the radiation exposure could be made, the experiments did not indicate an exceptional radiosensitivity for the developing fish embryo.

Methods for estimating the dose rates to aquatic organisms have been given by both the IAEA [54] and a study group convened by the National Research Council of Canada [55]. Such estimates are a necessary prerequisite for an assessment of the potential effects of radiation in contaminated aquatic environments.

Woodhead [56] considered the behaviour and distribution of radionuclides at four sites receiving controlled discharges of liquid radioactive wastes. The resultant radiation exposures of a wide variety of aquatic organisms were estimated using dosimetry models. On the basis of these results, combined with a critical review of data concerning the effects of radiation on aquatic organisms, it was concluded that, while there might be minor effects in the few individual organisms experiencing the highest dose rates, there would be no response at the population level. Again it is apparent that maximum dose rates of \(<10 \text{ mGy}\cdot\text{d}^{-1}\) would be very unlikely to result in significant damage to aquatic populations.

Anderson and Harrison [127] summarized the available data from the viewpoint of determining whether there were responses to radiation exposure in aquatic organisms which could be used to monitor effects in contaminated environments.
The review indicated that the dose rate range 0.5–10 rad·d⁻¹ (5–100 mGy·d⁻¹) would encompass the level at which a variety of low level effects on reproduction, development and genetic integrity are detectable in sensitive tissues and organisms.

In a recent report by the IAEA [9] an assessment was made of the consequences for the deep sea environment of the dumping of packaged low level radioactive wastes for 1000 years. It was concluded that increased mortality might be expected at sustained dose rates of >240 mGy·d⁻¹, while reduced reproductive success would be likely at dose rates in the range 24–240 mGy·d⁻¹. It was concluded that at lower dose rates there would be minor effects which could be accommodated within the reproductive capacity of the population or eliminated by the process of natural selection.

A Scientific Committee of the National Council on Radiation Protection and Measurements (USA) [128] has provided a summary of the available information and suggested a guideline of 10 mGy·d⁻¹, which would provide protection for populations of freshwater organisms. It was stated, however, that detailed site specific studies would be required if this guideline were to be approached for a substantial proportion of an aquatic population.

2.3.2. Effects on populations

Notwithstanding the conclusions of the many reviews cited above, it is useful to consider here the results of those few studies which have examined the effects of radiation on population attributes either in the field or in the laboratory.

Populations of the midge (Chironomus tentans) and the snail (Physa heterostropha) which inhabit White Oak Lake have been the subject of several investigations. In 1960, an increased frequency of chromosome aberrations was found in the salivary gland chromosome of Chironomus larvae that inhabited White Oak Lake, where they received a dose of 230 rad·a⁻¹ (~2 Gy·a⁻¹), approximately 1000 times normal background level [28, 129, 130]. However, ten years later, when the dose rate had decreased to 11 rad·a⁻¹ (~0.1 Gy·a⁻¹), the frequency of chromosome aberrations was not significantly different from control populations. This decrease in frequency of chromosome aberrations supported the previous conclusion that chronic irradiation of 230 rad·a⁻¹ (~2 Gy·a⁻¹) increased the frequency of chromosome aberrations in the Chironomus population, although there were no apparent additional consequences for the population [130].

Cooley and Nelson [131], Cooley and Miller [132], and Cooley [133] investigated the fecundity of the snail population in White Oak Lake, which received an estimated dose rate of 0.65 rad·d⁻¹ (~6 mGy·d⁻¹). The frequency of egg capsule production in the irradiated population was reduced. However, egg production by the irradiated and the non-irradiated populations was similar, because the irradiated population produced an increased number of eggs per capsule.
The number of laboratory studies on invertebrates involving chronic irradiation is limited. Blaylock and Trabalka [23] and Anderson and Harrison [127] have summarized these studies. For example, Cooley and Miller [132] investigated the effects of chronic radiation from a $^{60}$Co source on the survival, size and reproduction of laboratory populations of the snail (*Physa heterostropha*). A dose rate of 1 rad·h$^{-1}$ (10 mGy·h$^{-1}$) to snails throughout their life spans resulted in no significant effect on reproduction, mortality or size of the snail. A dose rate of 10 rad·h$^{-1}$ (0.1 Gy·h$^{-1}$) significantly changed all of these factors. Blue crabs (*Callinectes sapidus*) maintained under a chronic irradiation regime, receiving dose rates of 3.2, 7.3 or 29 rad·h$^{-1}$ (32, 73 or 290 mGy·h$^{-1}$), showed significant reductions in growth rate and survival only at the highest rate of exposure [134].

Marshall [135-138] exposed laboratory populations of *Daphnia pulex* to chronic $\gamma$ radiation from a $^{60}$Co source. For this species the maximum dose rates that were compatible with the survival of the populations were in the range 436-1330 R·d$^{-1}$ (4-13 Gy·d$^{-1}$) and depended upon the extent of competition for food and predation (produced by periodic culling) [23].

It was estimated [139] that the *Gambusia affinis* population at White Oak Lake had been exposed to dose rates initially as high as 0.4 rad·d$^{-1}$ (4 mGy·d$^{-1}$) in 1965, declining to 0.2 rad·d$^{-1}$ (2 mGy·d$^{-1}$) in 1971 and 0.06 rad·d$^{-1}$ (0.6 mGy·d$^{-1}$) in 1975. The frequencies of dead and abnormal embryos were compared with those in control populations [139, 140]. Although significantly more dead and abnormal embryos were observed in the irradiated population than in the control populations, a significantly larger brood size was found in the irradiated population. To test further the fitness of the White Oak Lake *Gambusia*, the males of the F$_1$ generation were subjected to critical thermal maxima tests [139]. The means and variance of the critical temperature recorded for the irradiated fish were significantly different from those of the control populations. These data were considered as evidence that the White Oak Lake population had an increased frequency of deleterious genes in its gene pool. Again, there was no evidence that these responses had had any effect on the viability of the populations.

Peshkov et al. [141] reported detrimental effects on the fecundity of the roach (*Rutilus rutilus lacustris*) which had received an effective dose rate of 0.19–0.7 rad·d$^{-1}$ (~2–7 mGy·d$^{-1}$) from internal emitters and 0.32–1.05 rad·d$^{-1}$ (~3–10 mGy·d$^{-1}$) from bottom sediments.

The plaice (*Pleuronectes platessa*) stock in the northeast Irish Sea has been exposed to low level irradiation for many years as a consequence of the waste discharges from the fuel reprocessing plant at Sellafield [7]. Estimates of the radiation exposure based on the measured concentrations of radionuclides in this environment indicated that the maximum dose rate at the sediment surface could be as high as 120 mrad·d$^{-1}$ in the vicinity of the discharge point, although it declined rapidly with distance. In situ measurements with lithium fluoride dosimeters attached to plaice confirmed that a few fish received dose rates of 60 mrad·d$^{-1}$ (0.6 mGy·d$^{-1}$),
but that the average dose rate to the population (based on a sample of 969 recaptured fish) was substantially less at 8.4 mrad·d⁻¹ (≈0.08 mGy·d⁻¹). Examination of catch statistics for this stock did not indicate any apparent radiation effects at the population level [31].

2.3.3. Summary

From previous reviews it may be concluded that:

(a) Aquatic organisms are no more sensitive than other organisms; however, because they are poikilothermic animals, temperature can control the time of expression of radiation effects.

(b) The radiosensitivity of aquatic organisms increases with increasing complexity, i.e., as organisms occupy successively higher positions on the phylogenetic scale.

(c) The radiosensitivity of many aquatic organisms changes with age or, in the case of unhatched eggs, with the stage of development.

(d) Embryo development in fish and the process of gametogenesis appear to be the most radiosensitive stages of all aquatic organisms tested.

(e) The radiation-induced mutation rate for aquatic organisms appears to be between that of Drosophila and the mouse.

The conclusion of the first IAEA review [31] of this subject that appreciable effects in aquatic populations would not be expected at dose rates lower than 10 mGy·d⁻¹ has not been challenged by subsequent studies or reviews. Thus it appears that limitation of the dose rate to the maximally exposed individuals in the population to <10 mGy·d⁻¹ would provide adequate protection for the populations.

2.4. OBSERVATIONS IN AREAS OF ELEVATED RADIOACTIVITY

Numerous local areas exist where radiation exposures are substantially above normal levels. This may be a consequence of higher than normal concentrations of naturally occurring radionuclides in rock, soil, water or air, or of the area's having become radioactively contaminated as a result of human activities. Increased background radiation due to cosmic rays can also be expected at high elevations and latitudes [24]. Observations of plant and animal populations in such areas have been conducted in an effort to determine whether various biological or ecological attributes vary between adjacent areas that differ in radiation exposure level. Such studies could be useful in demonstrating the presence of ecological differences between radiation and control areas. However, in cases where ecological differences have been observed, it is seldom possible to positively ascribe the differences to radiation, because other factors also vary between the irradiated and the control areas.
2.4.1. Areas of high natural radioactivity

Studies in a number of areas of high natural radioactivity have been reviewed by Turner [21] and Whicker and Schultz [24]. For example, attributes of a population of the black rat (*Rattus rattus*) were studied in a natural radioactive area in southern India by Grünenberg et al. [142]. The average $\gamma$ radiation level in this area was about seven times that of the control area. Styron [143] studied a population of invertebrates (the isopods *Liceus fontinalus*) on the granite outcrops of Mount Arabia in Georgia (USA), where the background radiation level was elevated above that of the surrounding areas. Lizards were studied by Tanner [144] on the Colorado Plateau (USA), where background exposure rates were 2–5 times normal. In all these studies, comparisons of the various parameters failed to indicate any significant differences between the populations receiving elevated exposure rates and those experiencing the normal radiation background.

More recent work, however, suggests definite effects on reproduction in female mice maintained in captivity at a site in France where the natural background dose rate from external sources was measured to be 8 mrad·h$^{-1}$ ($\sim$2 mGy·d$^{-1}$) [145], $\sim$1000 times the normal background exposure from external sources [146]. In this study, the number of offspring weaned from the irradiated females was 74% of the control value. In contrast, irradiated male mice produced 1.4 times as many weaned young as did control males. This effect may possibly have resulted from increased libido in the males [147], or from some unspecified physiological change [148]. In the same study, rabbit lymphocytes carried an increased number of unstable chromosome aberrations, such as fragments and dicentrics, if exposed to the enhanced background radiation. It is possible that, owing to the enhanced levels of radon in the environment studied by Leonard et al. [145], the internal tissues of the mice received a substantial exposure in addition to the 2 mGy·d$^{-1}$ from external sources. These findings do not necessarily imply that populations of mammals receiving comparable dose rates would be perceptibly altered in terms of population density or general fitness.

High levels of natural radiation (4000–8000 $\mu$R·h$^{-1}$, $\sim$1–2 mGy·d$^{-1}$) have been reported for certain regions in the USSR. Turner [21] reviewed these studies [149–151] in detail. In the area of highest radioactivity, annual exposures of 70 R ($\sim$2 mGy·d$^{-1}$) were calculated for some animals; however, Turner [21] pointed out that internal doses from incorporated radionuclides were not considered. Also, possible differences in the chemical toxicity of the environment were not accounted for. For animals inhabiting these areas of high natural radioactivity, a large number of abnormalities were reported, such as aberrant mitoses, decreased body fat, lower fertility, degeneration and necrotic processes. According to Maslov et al. [150], all of these phenomena helped to explain the reduced fertility and the lower densities of animals in this area.
Apparent effects of natural background radiation exposure on the structure of bryophyte (moss–liverwort) communities in specific localities of the Sudety Mountains in Poland have also been reported by Sarosiek and Wozakowska-Natkaniec [152]. It was observed that at exposure levels around 0.91 mR·h\(^{-1}\) (∼0.2 mGy·d\(^{-1}\)), *Marchantia polymorpha* L. increased in abundance and that other species were excluded. The authors reported that the edaphic and microclimatic factors were similar across the radiation gradients studied, and thus concluded that radiation was the causal factor. This dose rate is well below that shown experimentally to produce changes in plant community structure and it would therefore be desirable for the observations to be more critically examined. One approach could be to introduce radiation stress from an external source to a normal community to see if radiation per se, at these levels, would produce a similar effect.

2.4.2. Contaminated environments

A number of field studies have been conducted at sites of enhanced environmental radioactivity from anthropogenic sources [24]. Examples of such areas include the Hanford site in the State of Washington, USA; the Irish Sea in the vicinity of the Windscale facilities; the Animas River in the vicinity of Durango, Colorado, USA, where uranium milling waste once entered the river; and the previously mentioned radioactive waste retention pond at the Oak Ridge National Laboratory. Ecological studies have also been conducted at several nuclear detonation sites. These include sites of nuclear cratering experiments and sites used for testing nuclear weapons [27, 43, 153, 154]. Other types of studies that have been conducted on populations include experimental plots that have been treated with radionuclides [19, 155].

A comparison of various biological measurements between two ecologically similar study areas with greatly differing levels of \(^{239}\)Pu contamination at Rocky Flats, Colorado, USA, has been conducted [154, 156]. Measurements included vegetation structure and biomass; litter mass; arthropod community structure and biomass; and small mammal species occurrence, population density, biomass, reproduction, organ mass, pathology and parasite occurrence. No differences attributable to radiation exposure were found for any of the measurements, even though levels of \(^{239}\)Pu in the upper 3 cm of soil were as high as \(1.5 \times 10^7\) Bq·m\(^{-2}\).

An extensive review of research on ecological effects of nuclear testing at the Pacific Proving Grounds was provided by Templeton et al. [27]. Dosimetry and observations were provided by Jackson [157] for rat populations at Eniwetok Atoll, by Cole [158] and Stone et al. [43] for *Drosophila* cultures at Bikini Island and Rongelap and Rongerik Atolls, and by Fosberg [159] and Palumbo [160] for land plants at Rongelap and Eniwetok Atolls. A similar suite of investigations was completed for marine organisms around the atolls (e.g. [161–164]). The effects of the testing programme could not, in general, be ascribed solely to radiation, because of
the concomitant effects of blast and heat. Furthermore, human exploitation of the natural resources in the area changed markedly as a consequence of the test programme. Although many significant and complex effects on these ecosystems were observed, the recovery processes following the test explosions were relatively rapid and vigorous. Deleterious effects on marine and terrestrial populations were not persistent, presumably because of the rapid decline in the intensity of radiation and other impacts, the selective elimination of defective genetic information, and the recolonization of damaged areas with healthy individuals from distant localities.

The earliest studies of mammals on the bed of White Oak Lake failed to demonstrate an effect that could be ascribed to irradiation [57, 165–167]. Lifetime doses to wild rodents in this radioactive area were probably less than 200–300 rad (<2–3 Gy), but higher than the external doses postulated for mammals in areas of elevated radiation in the USSR.

A good example of the method of purposely treating an ecosystem with radio-nuclides in order to observe effects on populations is the work on soil invertebrates by Krivolutsky [19]. Various radionuclides, including $^{90}$Sr, $^{137}$Cs, $^{106}$Ru, $^{95}$Zr, $^{239}$Pu and $^{226}$Ra, were added to soil in small plots. At various intervals thereafter, the populations of several types of soil invertebrates were assessed. Dose rates that apparently produced reductions in animal numbers were generally quite high (2–4000 rad·h$^{-1}$ or 0.5–$10^3$ Gy·d$^{-1}$); however, some effects were reported at dose rates on the order of 100 mrad·h$^{-1}$ (24 mGy·d$^{-1}$). The most sensitive organism observed was the common earthworm, of the family Lumbricidae.

The recent (1986) accident at the Chernobyl nuclear power plant in the Soviet Union provides an example in which comparatively high levels of radioactivity were found in plants and animals exposed to the fallout. In Sweden, for example, reindeer were found with levels of $^{137}$Cs as high as $1.6 \times 10^4$ Bq/kg fresh meat, and fish with up to $4.8 \times 10^4$ Bq/kg fresh tissue [44]. These levels exceeded considerably the maximum permitted concentrations of $^{137}$Cs in food products (300 Bq/kg fresh weight) for commercial sales, so general agricultural commerce was disrupted in some areas. The internal dose rate to the fish containing the maximum observed $^{137}$Cs level of $4.8 \times 10^4$ Bq·kg$^{-1}$ would be on the order of 0.2 mGy·d$^{-1}$. On the basis of other literature, this upper limit dose rate is not likely to produce observable effects on the fish population. In this particular example, human intervention took place at a level of contamination below that which would be likely to produce effects on plant or animal populations.
3. ESTIMATED DOSE TO PLANTS AND ANIMALS UNDER CURRENT RADIATION PROTECTION STANDARDS

The primary approach taken to determine whether radiation protection standards for humans will provide adequate protection for natural populations of plants and animals has involved estimating radiation doses by the use of various mathematical modelling approaches. These have consisted of:

(a) Estimating the steady state concentrations of selected radionuclides in air and water or in soil from chronic releases from nuclear installations or shallow waste repositories, respectively, that would yield a radiation dose equal to the annual dose limit for members of the public;

(b) Estimating the equilibrium dose rates to reproductive or growth tissues of aquatic and terrestrial biota that would result from the same concentrations in air, water or soil.

In this study the dose limit for members of the public is taken to be 1 mSv·a$^{-1}$ [168]. In the context of this study a member of the public is understood to be a representative individual of the critical human population group which lives in, breathes the air from and receives sustenance from the local, contaminated environment. It is assumed that the aquatic and terrestrial plant and animal populations share a geographical area in common with the critical human group and that radionuclide transfer parameters in the human food-chain pathways are the same as for other species.

The dose estimates for plants and animals have been calculated for three different scenarios: (1) controlled releases of radionuclides to the atmosphere, (2) controlled releases of radionuclides to a freshwater aquatic system, and (3) uncontrolled constant releases of radionuclides from a shallow land nuclear waste repository.

Because of unavoidable uncertainties and complexities inherent in the calculations, numerous simplifying assumptions were necessary. These assumptions were made so as to yield conservative (i.e. maximum) estimates of dose rates to plants and animals. Parameter values used in calculations are subject to uncertainty. For generic calculations and assessments it is particularly important to recognize that such uncertainties exist. Experience with various assessments (e.g. [169]) suggests that overall uncertainties of around one order of magnitude are possible for the types of calculations presented in this report. Despite such uncertainties, it is believed that the consistent application of conservative assumptions and parameter choices will ensure a high degree of confidence in the conclusions reached.

The application of the same dose calculation models to both man and other organisms produces a compensatory effect which reduces the effect of errors in model structure and parameter values on the estimated dose to plant and animal
tissues. For example, if the dose to humans from a certain level of environmental contamination is overestimated, the release to the environment will have to be decreased to achieve the exposure standard. Using the same models and parameters to assess exposure of aquatic and terrestrial biota, the dose to plant and animal tissues is similarly overestimated. However, the reduced release will compensate for the overestimate in transport to organisms other than man. Therefore, the doses to plant and animal tissues for a given dose rate to man should not vary greatly with reasonable changes in parameter values.

The dose calculations were made for a group of radionuclides that represent a diverse array of physical, chemical and biological properties. The radionuclides considered in at least one of the three release scenarios include $^3$H, $^{14}$C, $^{32}$P, $^{60}$Co, $^{90}$Sr, $^{95}$Zr, $^{99}$Tc, $^{129}$I, $^{131}$I, $^{137}$Cs, $^{226}$Ra, $^{235}$U, $^{238}$U, $^{239}$Pu and $^{241}$Am. Important fission products are represented by $^3$H, $^{90}$Sr, $^{95}$Zr, $^{129}$I, $^{131}$I and $^{137}$Cs. While $^{95}$Zr is not transferred readily through biota, the other five fission products tend to be biologically mobile. Tritium (often as HTO), $^{14}$C and $^{32}$P are metabolically active radionuclides that are commonly used in research and medicine and can be produced by nuclear activation processes. The long lived $\alpha$ emitters $^{226}$Ra, $^{235}$U, $^{238}$U, $^{239}$Pu and $^{241}$Am are less biologically mobile; this applies especially to Pu and Am. However, owing to the highly ionizing nature of their $\alpha$ emissions, these nuclides may be radiologically important when inhaled or ingested. It should be noted that the calculations were performed applying the human dose limit to the release of each radionuclide independently, i.e. the set of radionuclides in each scenario is not considered as a mixture. If the dose limit were applied to a mixture of radionuclides rather than to single radionuclides, the resulting environmental concentrations would be lower than estimated in this report and consequently the doses to biota would be lower.

3.1. CONTROLLED ATMOSPHERIC RELEASES

3.1.1. Methodology

In this scenario it is assumed that radionuclides are released to the atmosphere at a constant rate which is fixed so as to limit the exposure of humans living in the vicinity of the release point. The source term used for the dose calculations is the radionuclide deposition rate to the ground surface which produces doses equivalent to the dose limit in the critical group when all relevant exposure pathways are taken into account. These pathways are external irradiation from the cloud and from deposited radioactivity, inhalation and ingestion of food containing radionuclides. Cloud immersion doses can be shown from screening calculations to be negligible relative to the other pathways and are therefore neglected.
The deposition rate so determined is used as the basis for estimating the doses to plants and animals. These are evaluated by considering foliar deposition, root uptake and external exposure from γ emitters in soil in the case of plants. For animals the same processes as for humans, i.e. external exposure, inhalation and ingestion, are considered for dose estimation.

A basic assumption in the calculations is that atmospheric releases are chronic and constant for long periods of time, such that radionuclides in air, soil, water and organisms are also at constant, steady state concentrations. Thus, dose rates to tissues of humans and other organisms living in and obtaining sustenance from this environment would also be constant over time. This condition, in reality, would be most nearly achieved by the short lived radionuclides or by all nuclides in compartments of the environment which have rapid turnover rates and which are to some degree isolated from most soil compartments. Concentrations of longer lived radionuclides such as $^{137}$Cs, $^{90}$Sr and $^{239}$Pu would actually continue to increase in some soil compartments for many years under chronic, constant release scenarios. Thus, processes involving such compartments, such as soil to plant to animal transfers and external γ ray exposure rates from radionuclides in soil, could continue to increase for many years. However, it is believed that this situation would affect doses to humans and other organisms similarly; therefore the simplifying assumption of steady state conditions for all radionuclides should not invalidate the final conclusions.

Various assumptions were made to simplify the calculations and to provide upper estimate (conservative) predictions of dose to plant and animal tissues. Situations that would minimize human dose/intake ratios were generally adopted because this would yield maximum environmental concentrations and thus maximum doses to biota. Parameter values affecting dose/intake ratios for biota were selected to maximize such ratios.

Three approaches were employed for estimating dose. These included the use of the published results of the PATHWAY model [169], the use of ICRP Publication 30 derived limits of radionuclide concentrations in air and annual intakes [170], and computer simulations conducted in 1987 by a class in radionuclide kinetics at Colorado State University, USA. Although differences approaching an order of magnitude were encountered among the three approaches, each approach produced doses to biota which would not alter the final conclusions of this report. When more than one approach was used, results from that which produced the highest doses to biota were adopted.²

² The calculations were also reviewed by scientists of the Institut für Strahlenschutz, Gesellschaft für Strahlen- und Umweltforschung mbH (GSF), Neuherberg, Federal Republic of Germany.
FIG. 2. Structural features of the PATHWAY model. Boxes represent compartments or state variables; arrows represent transfers resulting from indicated processes; circles connect process arrows between the upper and the lower diagram [169].
3.1.1.1. Estimation of the source term

As the source term for the dose estimates, a ground deposition rate (Bq·m$^{-2}$·d$^{-1}$) was calculated that would give rise to an effective dose equivalent rate ($H_E$) of 1 mSv·a$^{-1}$ to a person as a result of living on, breathing the air above and eating the food from the land receiving the deposition. The steady state ground deposition rate (Bq·m$^{-2}$·d$^{-1}$) was derived by multiplying a constant radionuclide concentration in air (Bq·m$^{-3}$) by a deposition velocity which was assumed to be 0.2 cm·s$^{-1}$ [169, 171].

The steady state radionuclide ingestion rate by humans from all food sources for a unit deposition rate was calculated using the PATHWAY model [169]. The structural features of this model are illustrated in Fig. 2. The effective dose equivalent rate from ingestion was then obtained as the product of the ingestion rate and the dose equivalent per unit intake (Sv·Bq$^{-1}$). The latter values, obtained using ICRP [170] methodology, are summarized by radionuclide in Ref. [172].

Values adopted for food consumption and inhalation rates and dose equivalents per unit intake are for adult males. In most cases, this should provide conservatism, since dose equivalents per unit intake for younger age groups are usually higher; thus environmental concentrations and doses to biota would be lower if the factors for non-adults were applied. The effective dose equivalent rate ($H_E$) value from inhalation was calculated as the product of the radionuclide concentration in air derived from the ground deposition rate calculated above, a breathing rate of $8.4 \times 10^3$ m$^3$·a$^{-1}$ [171], and the effective dose equivalent per unit inhaled activity [172]. The $H_E$ value for external $\gamma$ ray exposure was obtained from a PATHWAY simulation [169] in which the equilibrium soil inventory for $\gamma$ emitters was estimated (30 years in the case of $^{137}$Cs). The inventory (in Bq·cm$^{-2}$) was multiplied by exposure rate conversion factors summarized in Ref. [173]. The conversion factors were modified for assumed soil relaxation depths to account for the depth distributions in soil [174]. A conversion factor for human dose equivalent per unit dose to air of 0.3 Sv·Gy$^{-1}$ was assumed to account for the shielding provided to an average individual while living and working inside buildings [146]. The $H_E$ values for ingestion, inhalation and external exposure were then summed. Finally, the ground deposition rate producing the $H_E$ value from all sources was normalized to 1 mSv·a$^{-1}$ to humans. These values are given for each radionuclide in Table II.

In the case of $^3$H, $^{14}$C, $^{32}$P and $^{95}$Zr, a different approach was used because results for these radionuclides were not available from the PATHWAY model. The derived air concentrations (DACs) that will produce an $H_E$ value of 50 mSv·a$^{-1}$ to human radiation workers are provided by the ICRP [170] and the IAEA [168]. These were used as convenient conversion factors between environmental concentrations and dose to humans. Where more than one DAC value was listed owing to there being different chemical forms of a radionuclide, the highest values were chosen because this maximized the dose to plants and animals. The DAC was adjusted to
<table>
<thead>
<tr>
<th>Nuclide</th>
<th>Deposition rate to produce an effective dose equivalent of 1 mSv for 1 year of exposure (Bq·m⁻²·d⁻¹)</th>
<th>Equilibrium concentration in plant tissues (Bq·kg⁻¹ DW⁻¹)</th>
<th>Upper estimate dose rate to plant tissues (mGy·d⁻¹)</th>
<th>Equilibrium concentration in animal tissues (Bq·kg⁻¹)</th>
<th>Upper estimate dose rate to animal tissues (mGy·d⁻¹)</th>
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<tr>
<td>H-3 (as HTO)</td>
<td>6.6 × 10⁻¹</td>
<td>1.8 × 10⁶</td>
<td>1.4 × 10⁻¹</td>
<td>1.8 × 10⁶</td>
<td>1.4 × 10⁻¹</td>
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<tr>
<td>C-14</td>
<td>NAᵇ</td>
<td>6.0 × 10⁵</td>
<td>4.4 × 10⁻¹</td>
<td>3.6 × 10⁵</td>
<td>2.7 × 10⁻¹</td>
</tr>
<tr>
<td>P-32</td>
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<td>1.6 × 10⁵</td>
<td>7.6 × 10⁻¹</td>
<td>7.0 × 10⁴</td>
<td>6.7 × 10⁻¹</td>
</tr>
<tr>
<td>Sr-90</td>
<td>5.4 × 10¹</td>
<td>3.2 × 10³</td>
<td>4.9 × 10⁻²</td>
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<td>Zr-95</td>
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<td>1.9 × 10⁵</td>
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<td>5.2 × 10²</td>
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<td>1.1 × 10⁻²ᵃ</td>
<td>6.9 × 10⁻⁵</td>
<td>2.6 × 10⁻⁴ᵃ</td>
</tr>
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</table>

ᵃ DW: dry weight.
ᵇ NA: not applicable owing to method of calculation (specific activity approach).
ᶜ A factor of 20 has been applied to account for the enhanced biological effectiveness per unit absorbed dose.
a value for a member of the continuously exposed general public by multiplying by a factor of \(4.76 \times 10^{-3}\), expressing the change from the 40 h-week\(^{-1}\) of a worker to 168 h-week\(^{-1}\) and a dose reduction from 50 to 1 mSv·a\(^{-1}\). The deposition rate corresponding to an annual effective dose equivalent of 1 mSv·a\(^{-1}\) to a member of the general public was then estimated as the product of the adjusted DAC and the assumed deposition velocity. The ingestion pathway to humans was also considered using the published annual limits on intake (ALIs) [168, 170]. The ingestion pathway is generally more restrictive for human exposure than the inhalation pathway, but in order to maximize doses to plants and animals deposition rates based on DAC values were chosen for \(\text{H}^3\), \(\text{P}^{32}\) and \(\text{Zr}^{95}\). For \(\text{C}^{14}\) the ALI approach was used because the DAC approach gives unrealistically high doses to plants and animals. (Roughly 99% of man’s carbon intake is from ingestion and since \(\text{C}^{14}/\text{C}\) ratios in air and in food will be near equilibrium, the DAC approach produces over-estimates of doses to plants and animals by about two orders of magnitude.)

The deposition rates resulting in an effective dose equivalent of 1 mSv·a\(^{-1}\) derived by these calculations are given in Table II.

3.1.1.2. Estimation of radionuclide concentrations in plants

The equilibrium concentrations of radionuclides in plant tissues were estimated using the PATHWAY model [169], in which processes such as aerial deposition, root uptake, resuspension, rainsplash and weathering are considered. In the case of \(\text{H}^3\), \(\text{P}^{32}\) and \(\text{Zr}^{95}\), for which results were not available from the PATHWAY code, the same calculations were performed by hand using the same radionuclide independent parameters contained in PATHWAY. The effective plant loss rate parameters for \(\text{H}^3\), \(\text{P}^{32}\) and \(\text{Zr}^{95}\) were estimated as 0.7, \(9.9 \times 10^{-2}\) and \(4.9 \times 10^{-2}\)·d\(^{-1}\), respectively. The loss rate parameter for \(\text{H}^3\) was conservatively chosen to provide a concentration in plant water equivalent to that in atmospheric water vapour. The loss rate parameters for \(\text{P}^{32}\) and \(\text{Zr}^{95}\) are based on the commonly used weathering half-time of 14 d. Plant concentrations and doses were calculated on a dry mass basis. For \(\text{C}^{14}\), the concentrations in plant tissues that would produce the ALI were computed using a simple specific activity model [172].

In the calculations for \(\text{H}^3\), it was assumed that the radionuclide entered the environment in the form of water or water vapour and that the conversion of HTO to organically bound tritium was not sufficient to significantly alter the computations of tissue concentrations and doses. This position was influenced by the statements in ICRP Publication 30 [170] that \“values of the committed dose equivalent to body tissues arising from an intake of tritiated water may be estimated from consideration of the retention of tritiated water alone\” and \“When tritium-labelled organic compounds are ingested, a considerable fraction may be broken down in the gastrointestinal tract producing tritiated water.\” It was also influenced by modelling results of Murphy [175], although this author pointed to large uncertainties and the need for
case specific information. Other data, however, suggest that as much as 68% of the dose to humans from environmental tritium may arise from the organically bound fraction [176]. However, it might be expected that the error introduced by neglecting organic binding of $^{3}$H would affect both man and other organisms in the environment in a roughly similar manner. If this were the case, the conclusions of this report would not be altered. For atmospheric releases, the conclusions of this report would not be altered in any case unless the doses to plants and animals were underestimated by an order of magnitude.

If $^{3}$H is released to the environment in a form other than water, a different assessment seems warranted. According to ICRP Publication 30 [170], HT is 25 000 times less radiotoxic than HTO. Therefore, the DAC is correspondingly higher for this chemical form. The oxidation of HT in the biosphere occurs mainly in soil microorganisms. Therefore, food-chains originating with soil microbes (e.g. detritus based food-chains) may receive enhanced dose rates because this effect would not apply equally to the human food-chain.

The radionuclide concentrations in plants at equilibrium derived by these calculations are given in Table II.

3.1.1.3. Estimation of dose rates to plants

The equilibrium dose rate to plant tissues was assumed to be the same for all above ground tissue types, including the critical tissues for growth (meristems) and reproduction (flowering buds). The simplifying (and conservative) assumption was made that 100% of the decay energy of $\alpha$ and $\beta$ particles is absorbed by the critical tissues (except in the case of $^{32}$P, where it was assumed that 50% of the $\beta$ decay energy is absorbed). For $\gamma$ photons, it was assumed that 10% of the decay energy is absorbed. Decay energies were taken from Ref. [171]. In the case of $^{137}$Cs, doses were multiplied by a factor of 10 to account for high bioconcentration in some plant species not used by man. Local soil properties and morphology can explain enhanced bioaccumulation in particular species or local individuals; for example, Mancanzoni [44] reports levels in mushrooms 5–10 times higher than in herbaceous vegetables. In the case of $^{239}$Pu, an $\alpha$ emitter, a factor of 20 was employed to account for the potentially greater magnitude of damage per unit of absorbed dose.

An example dose calculation for $^{32}$P in plant tissues is:

\[
\text{dose rate (Gy·d}^{-1}) = \frac{(1.6 \times 10^5 \text{ Bq·kg}^{-1}) (0.69 \text{ MeV·dis}^{-1}) (0.5) (8.64 \times 10^4 \text{ dis·d}^{-1}·\text{Bq}^{-1})}{(10^3 \text{ g·kg}^{-1}) (6.25 \times 10^9 \text{ MeV·g}^{-1}·\text{Gy}^{-1})}
\]

(1)
The external dose rates to plants from $\gamma$ emitters in the ground were assumed to be 3.3 times the respective external dose rates to humans, owing to geometry and occupancy differences [146]. The external dose rates to plant tissues from strong $\beta$ emitters (e.g. $^{90}\text{Sr}$, $^{90}\text{Y}$ and $^{32}\text{P}$) on the ground surface were shown by calculation to be less than 10% of the dose from activity on or in the foliage. This source was therefore ignored.

The most conservative (highest) dose rate estimates for plant tissues derived by these calculations are given in Table II.

3.1.1.4. Estimation of radionuclide concentrations in animals

The steady state concentrations for all radionuclides except $^{90}\text{Sr}$ in the reproductive organs of animals were assumed equal to the concentrations predicted for lamb muscle by the PATHWAY model [169]. Lamb muscle was chosen because this species generally exhibits higher concentrations of $^{137}\text{Cs}$, and probably also of other radionuclides, than other farm animals [44]. For $^{90}\text{Sr}$, the concentration in reproductive tissue (ovary) was taken as four times that in muscle, on the basis of data for stable Sr concentrations in tissues [177]. These are generally unsupported assumptions, considering the many animal species and the lack of specific data on tissue distribution within such species. However, it is believed they are conservative, on the basis of the distribution of elements in various body tissues [177]. In the case of $^3\text{H}$, $^{32}\text{P}$ and $^{95}\text{Zr}$, for which PATHWAY results were not available, the same calculations were performed by hand using the same radionuclide independent parameters. The assimilation fractions for muscle assumed for $^{32}\text{P}$ and $^{95}\text{Zr}$ were 0.4 [170] and $3.2 \times 10^{-4}$ [169], respectively. The effective loss rate constants for these radionuclides were taken as $8.6 \times 10^{-2} \cdot \text{d}^{-1}$ [170] and $1.1 \times 10^{-2} \cdot \text{d}^{-1}$ [169], respectively. For $^3\text{H}$, the conservative assumption was made that animal tissues would reach the same concentrations as plant tissues. For $^{14}\text{C}$, the concentrations in animal tissues that would produce the ALI were computed from a simple specific activity model [172].

3.1.1.5. Estimation of dose rates to animals

The same basic method used to estimate dose to plant tissues was also applied to animal tissues. It was assumed that 100% of the $\alpha$ and $\beta$ decay energy was absorbed in the tissues; for $\gamma$ photons, 30% of the decay energy was taken to be absorbed by reproductive tissues, an assumption that is believed to be conservative. In the case of $^{137}\text{Cs}$, however, the calculated internal dose rates to animal tissues were multiplied by a factor of 10 to account for unusual bioconcentration mechanisms (e.g. carnivory, or consumption of certain plants) for species not incorporated into the human food-chain calculation. An example is $^{137}\text{Cs}$ from Chernobyl fallout in Sweden, which reached sixfold higher concentrations in certain other animals than
in lamb [44]. A factor of 20 to account for the biological effectiveness of $\alpha$ radiation was assumed in the case of $^{239}$Pu. The transport to animal tissues from inhalation was assumed equivalent to that estimated for humans from inhalation. The human gonadal dose equivalent conversion factor was applied in the case of animals. This approach seemed the only feasible one, owing to the lack of data specific to other species. The dose to animal tissues from external irradiation arising from $\gamma$ emitters in the ground was considered to be the same as to plants.

The most conservative (highest) dose rate estimates for animal tissues derived by these calculations are given in Table II.

3.1.2. Results

The calculated upper estimate dose rates ranged from $\sim 10^{-2}$ to 1 mGy·d$^{-1}$ for plant tissues and from roughly $10^{-4}$ to 1 mGy·d$^{-1}$ for animal tissues. In no case did a dose rate to biota exceed 1 mGy·d$^{-1}$. It is believed that, in view of the deliberately conservative assumptions used in the calculations, the actual dose rates that might be received by terrestrial plants and animals would be substantially less than the values presented in Table II. It is believed that a sufficiently diverse array of radionuclides has been considered to reach the general conclusion that under the stated conditions of the computations it is highly unlikely that terrestrial plants or animals would actually receive dose rates exceeding 1 mGy·d$^{-1}$, regardless of the specific radionuclide released to the atmosphere. It has already been noted that for a release of tritium in the elemental form (as opposed to HTO) this conclusion may not be valid for all animals.

3.2. CONTROLLED RELEASES OF RADIONUCLIDES TO SURFACE WATERS

The disposal of radionuclides into the freshwater environment can be expected to give rise to dose rates to certain organisms that are similar to or even substantially higher than those received by people inhabiting and/or deriving sustenance from the same environment. A major question is whether the concentrations of radionuclides in the organs and tissues of the aquatic organisms, and in the components of their environment, are sufficiently high to produce deleterious effects at the population level when the radiation dose rate to humans utilizing the same environment is constrained within appropriate limits. To address this question, it has been assumed that a known release of radionuclides enters a freshwater system from either a nuclear installation or a waste disposal site and that the dose rate to humans utilizing this environment has reached the limiting value of 1 mSv·a$^{-1}$. By estimating the potential dose rate to aquatic organisms in such a situation, the consequent effects on aquatic populations may be estimated and assessed. For simplicity, it has been
assumed that the concentrations of the radionuclides in the water are constant, and that those in the sediment and in aquatic organisms have reached equilibrium values. It has also been assumed that the limiting dose rate to humans is received via the following pathways:

- Fish consumption at a rate of 100 kg·a⁻¹,
- Water consumption at a rate of 2 L·d⁻¹,  
- External exposure from contaminated sediment with an occupancy rate of 2000 h·a⁻¹.

These generic values have been adopted for adult members of the general public [171, 178].

3.2.1. Methodology

A number of approaches have been described for making estimates of the dose rates to aquatic organisms in environments contaminated with radionuclides (see Section 2.1.2). For the present purpose the BIORAD computer code [59] has been used to estimate the dose rates to fish from internal and external sources because these organisms, at the embryonic stage and in the process of gametogenesis, have been determined to be the most sensitive to long term, low level irradiation (see Section 2.3).

3.2.1.1. Estimation of the source term

The BIORAD model requires as source term the water concentration for each radionuclide to determine the dose rate to fish. First, however, it is necessary to determine the exposure of humans via each of the three pathways described above and to relate the total exposure for each nuclide to the effective dose equivalent limit of 1 mSv·a⁻¹. A nominal radionuclide concentration in water of 1 Bq·L⁻¹ was assumed and the dose rate to man for each pathway was calculated as follows:

\[
H_E(W) = C(W) \times CR(W) \times DC \tag{2}
\]

where

\(H_E(W)\) is the effective dose equivalent (Sv·a⁻¹) from drinking water,
\(C(W)\) is the radionuclide concentration in water, taken to be 1 Bq·L⁻¹,
\(CR(W)\) is the consumption rate of water equal to \(7.3 \times 10^2\) L·a⁻¹,
\(DC\) is the dose conversion factor for ingestion (Sv·Bq⁻¹).
The dose per unit intake factors for adults were used [170] as these resulted in the highest potential concentrations of the nuclides in the aquatic environment, i.e. they were the most conservative.

Fish consumption

\[ H_E(F) = C(W) \ BF \ CR(F) \ DC \]  

(3)

where

- \( H_E(F) \) is the effective dose equivalent (Sv·a\(^{-1}\)) from eating fish,
- \( BF \) is the bioaccumulation factor relating the concentration of a nuclide in the fish to that in the water (L·kg\(^{-1}\)),
- \( CR(F) \) is the consumption rate of fish equal to 100 kg·a\(^{-1}\).

The values for the bioaccumulation factor were those used in the BIORAD model [55] to provide a common basis for the estimation of the dose rates to both humans and fish.

External exposure from sediments

\[ H_E(S) = C(W) \ K_d \ DC(S) \ OF \]  

(4)

where

- \( H_E(S) \) is the effective dose equivalent (Sv·a\(^{-1}\)) from exposure to radiation originating from contaminated sediments,
- \( K_d \) is the radionuclide distribution coefficient relating the concentration in the sediment to that in the water (L·kg\(^{-1}\)),
- \( DC(S) \) is the dose conversion factor (Sv·a\(^{-1}\) per Bq·kg\(^{-1}\)),
- \( OF \) is the occupancy factor equal to 2000 h·a\(^{-1}\) = 0.228.

As the sediment is not included as a source of exposure in the BIORAD model, the required parameter values have been assembled from other publications. The \( K_d \) values for the majority of the elements considered have been taken from IAEA Safety Series No. 57 [171]; the value for Ra was derived from data given by Beneš [179], and that for U from Swanson [180]. The dose conversion factors have been derived from published data given in terms of mrem·a\(^{-1}\) per pCi·cm\(^{-3}\) at a sediment density of 1.8 g·cm\(^{-3}\) [181], using the factor 4.86 \times 10\(^{-7}\) to give values in terms of Sv·a\(^{-1}\) per Bq·kg\(^{-1}\).

Combining equations (2), (3) and (4) gives the total effective dose equivalent rate, \( H_E(T) \) (Sv·a\(^{-1}\)), for the three pathways:

\[ H_E(T) = C(W) \ [(CR(W) + CR(F) BF)DC + K_d \ OF \ DC(S)] \]
**TABLE III. CONTROLLED AQUATIC RELEASE SCENARIO: PARAMETER VALUES USED TO ESTIMATE THE EXPOSURE OF HUMANS FROM CONTAMINATED WATER VIA THE THREE DEFINED PATHWAYS AND RADIO-NUCLIDE CONCENTRATION IN WATER ESTIMATED TO DELIVER A DOSE RATE OF 1 mSv·a⁻¹ TO HUMANS**

<table>
<thead>
<tr>
<th>Nuclide</th>
<th>Bioaccumulation factor, $BF$ (L·kg⁻¹)</th>
<th>Dose conversion factor for ingestion, $DC$ (Sv·Bq⁻¹)</th>
<th>Distribution coefficient, $K_d$ (L·kg⁻¹)</th>
<th>Dose conversion factors for contaminated sediment, $DC(S)$ (Sv·a⁻¹ per Bq·kg⁻¹)</th>
<th>Water concentration required to give an effective dose equivalent rate of 1 mSv·a⁻¹ (Bq·L⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>H-3</td>
<td>$1.0 \times 10^9$</td>
<td>$1.7 \times 10^{-11}$</td>
<td>0</td>
<td>0</td>
<td>$7.1 \times 10^4$</td>
</tr>
<tr>
<td>P-32</td>
<td>$1.0 \times 10^4$</td>
<td>$2.4 \times 10^{-9}$</td>
<td>No data</td>
<td>0</td>
<td>$4.2 \times 10^{-2}$</td>
</tr>
<tr>
<td>Co-60</td>
<td>$2.0 \times 10^1$</td>
<td>$7.3 \times 10^{-9}$</td>
<td>$3.0 \times 10^4$</td>
<td>$6.1 \times 10^{-6}$</td>
<td>$2.4 \times 10^{-2}$</td>
</tr>
<tr>
<td>Sr-90</td>
<td>$5.0 \times 10^0$</td>
<td>$3.9 \times 10^{-8}$</td>
<td>$2.0 \times 10^3$</td>
<td>0</td>
<td>$2.1 \times 10^1$</td>
</tr>
<tr>
<td>Tc-99</td>
<td>$1.5 \times 10^1$</td>
<td>$4.0 \times 10^{-10}$</td>
<td>$2.0 \times 10^2$</td>
<td>$4.5 \times 10^{-13}$</td>
<td>$1.1 \times 10^3$</td>
</tr>
<tr>
<td>I-131</td>
<td>$1.5 \times 10^1$</td>
<td>$1.4 \times 10^{-8}$</td>
<td>$2.0 \times 10^2$</td>
<td>No data</td>
<td>$3.1 \times 10^1$</td>
</tr>
<tr>
<td>Cs-137</td>
<td>$4.0 \times 10^2$</td>
<td>$1.4 \times 10^{-8}$</td>
<td>$3.0 \times 10^4$</td>
<td>$1.4 \times 10^{-6}$</td>
<td>$1.0 \times 10^{-1}$</td>
</tr>
<tr>
<td>Ra-226</td>
<td>$5.0 \times 10^1$</td>
<td>$3.6 \times 10^{-7}$</td>
<td>$2.5 \times 10^3$</td>
<td>$4.2 \times 10^{-6}$</td>
<td>$2.3 \times 10^{-1}$</td>
</tr>
<tr>
<td>U-235</td>
<td>$1.0 \times 10^1$</td>
<td>$7.2 \times 10^{-8}$</td>
<td>$2.7 \times 10^3$</td>
<td>$2.4 \times 10^{-7}$</td>
<td>$3.7 \times 10^0$</td>
</tr>
<tr>
<td>U-238</td>
<td>$1.0 \times 10^1$</td>
<td>$6.9 \times 10^{-8}$</td>
<td>$2.7 \times 10^3$</td>
<td>$3.4 \times 10^{-8}$</td>
<td>$7.1 \times 10^0$</td>
</tr>
<tr>
<td>Pu-239</td>
<td>$3.5 \times 10^2$</td>
<td>$1.2 \times 10^{-7}$</td>
<td>$3.0 \times 10^4$</td>
<td>$1.8 \times 10^{-10}$</td>
<td>$2.4 \times 10^{-1}$</td>
</tr>
<tr>
<td>Am-241</td>
<td>$2.5 \times 10^1$</td>
<td>$5.8 \times 10^{-7}$</td>
<td>$3.0 \times 10^4$</td>
<td>$1.3 \times 10^{-8}$</td>
<td>$5.1 \times 10^{-1}$</td>
</tr>
</tbody>
</table>

Ref. [55] [170] [171, 179, 180] [181]
<table>
<thead>
<tr>
<th>Nuclide</th>
<th>Internal exposure (mGy·a(^{-1}) per Bq·L(^{-1}))</th>
<th>External exposure from water (mGy·a(^{-1}) per Bq·L(^{-1}))</th>
<th>(\gamma)</th>
<th>(\beta + \gamma)</th>
</tr>
</thead>
<tbody>
<tr>
<td>H-3</td>
<td>(5.1 \times 10^{-5})</td>
<td>0.0</td>
<td>1.6 \times 10^{-5}</td>
<td></td>
</tr>
<tr>
<td>P-32</td>
<td>(3.5 \times 10^{-2})</td>
<td>0.0</td>
<td>1.8 \times 10^{-3}</td>
<td></td>
</tr>
<tr>
<td>Co-60</td>
<td>(1.5 \times 10^{-1})</td>
<td>(1.3 \times 10^{-2})</td>
<td>1.3 \times 10^{-2}</td>
<td></td>
</tr>
<tr>
<td>Sr-90</td>
<td>(2.7 \times 10^{-2})</td>
<td>0.0</td>
<td>2.7 \times 10^{-3}</td>
<td></td>
</tr>
<tr>
<td>Tc-99</td>
<td>(7.0 \times 10^{-3})</td>
<td>0.0</td>
<td>2.4 \times 10^{-4}</td>
<td></td>
</tr>
<tr>
<td>I-131</td>
<td>(3.2 \times 10^{-2})</td>
<td>(1.9 \times 10^{-3})</td>
<td>2.5 \times 10^{-3}</td>
<td></td>
</tr>
<tr>
<td>Cs-137</td>
<td>(1.2 \times 10^{0})</td>
<td>(2.7 \times 10^{-3})</td>
<td>3.2 \times 10^{-3}</td>
<td></td>
</tr>
<tr>
<td>Ra-226</td>
<td>(2.7 \times 10^{1})</td>
<td>(3.5 \times 10^{-3})</td>
<td>6.8 \times 10^{-5}</td>
<td></td>
</tr>
<tr>
<td>U-235</td>
<td>(2.3 \times 10^{4})</td>
<td>(9.2 \times 10^{-4})</td>
<td>1.5 \times 10^{-2}</td>
<td></td>
</tr>
<tr>
<td>U-238</td>
<td>(2.2 \times 10^{6})</td>
<td>(1.6 \times 10^{-2})</td>
<td>2.0 \times 10^{-2}</td>
<td></td>
</tr>
<tr>
<td>Pu-239</td>
<td>(9.5 \times 10^{1})</td>
<td>(3.0 \times 10^{-6})</td>
<td>2.2 \times 10^{-5}</td>
<td></td>
</tr>
<tr>
<td>Am-241</td>
<td>(7.3 \times 10^{0})</td>
<td>(1.3 \times 10^{-4})</td>
<td>2.1 \times 10^{-4}</td>
<td></td>
</tr>
</tbody>
</table>

\(a\) Calculations performed with the BIORAD model [55].
<table>
<thead>
<tr>
<th>Nuclide</th>
<th>Total decay energy (MeV)</th>
</tr>
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<tbody>
<tr>
<td></td>
<td>α particles</td>
</tr>
<tr>
<td>H-3</td>
<td>—</td>
</tr>
<tr>
<td>P-32</td>
<td>—</td>
</tr>
<tr>
<td>Co-60</td>
<td>—</td>
</tr>
<tr>
<td>Sr-90 + Y-90</td>
<td>—</td>
</tr>
<tr>
<td>Tc-99</td>
<td>—</td>
</tr>
<tr>
<td>I-131</td>
<td>—</td>
</tr>
<tr>
<td>Cs-137 + Ba-137m</td>
<td>—</td>
</tr>
<tr>
<td>Ra-226</td>
<td>4.9 x 10^{0}</td>
</tr>
<tr>
<td>U-235 + Th-231</td>
<td>4.5 x 10^{0}</td>
</tr>
<tr>
<td>U-238 — U-234 inclusive</td>
<td>9.1 x 10^{0}</td>
</tr>
<tr>
<td>Pu-239</td>
<td>5.2 x 10^{0}</td>
</tr>
<tr>
<td>Am-241</td>
<td>5.6 x 10^{0}</td>
</tr>
</tbody>
</table>
TABLE VI. CONTROLLED AQUATIC RELEASE SCENARIO: DOSE RATES TO FISH INHABITING A WATER BODY CONTAMINATED WITH INDIVIDUAL RADIONUCLIDES AT CONCENTRATIONS WHICH WOULD DELIVER A DOSE RATE OF 1 mSv·a⁻¹ TO HUMANS VIA THE THREE PATHWAYS COMBINED

<table>
<thead>
<tr>
<th>Nuclide</th>
<th>Radionuclide concentration in water (Bq·L⁻¹)</th>
<th>Internal dose rate (mGy·a⁻¹)</th>
<th>External dose rate</th>
<th>Total dose rate, whole body (mGy·a⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>From water</td>
<td>From sediment</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>γ (mGy·a⁻¹)</td>
<td>β + γ (mGy·a⁻¹)</td>
</tr>
<tr>
<td>H-3</td>
<td>7.1 × 10⁴</td>
<td>3.6 × 10⁰</td>
<td>0</td>
<td>1.1 × 10⁰</td>
</tr>
<tr>
<td>P-32</td>
<td>4.2 × 10⁻²</td>
<td>1.5 × 10¹</td>
<td>0</td>
<td>7.6 × 10⁻⁵</td>
</tr>
<tr>
<td>Co-60</td>
<td>2.4 × 10⁻²</td>
<td>3.6 × 10⁻³</td>
<td>3.1 × 10⁻⁴</td>
<td>3.1 × 10⁻⁴</td>
</tr>
<tr>
<td>Sr-90</td>
<td>2.1 × 10¹</td>
<td>5.7 × 10⁻¹</td>
<td>0</td>
<td>5.7 × 10⁻²</td>
</tr>
<tr>
<td>Te-99</td>
<td>1.1 × 10⁻³</td>
<td>7.7 × 10⁰</td>
<td>0</td>
<td>2.6 × 10⁻¹</td>
</tr>
<tr>
<td>I-131</td>
<td>3.1 × 10⁻¹</td>
<td>9.9 × 10⁻¹</td>
<td>5.9 × 10⁻²</td>
<td>7.8 × 10⁻²</td>
</tr>
<tr>
<td>Cs-137</td>
<td>1.0 × 10⁻¹</td>
<td>1.2 × 10⁻¹</td>
<td>2.7 × 10⁻⁴</td>
<td>3.2 × 10⁻⁴</td>
</tr>
<tr>
<td>Ra-226</td>
<td>2.3 × 10⁻¹</td>
<td>6.2 × 10⁰</td>
<td>8.1 × 10⁻⁶</td>
<td>1.6 × 10⁻⁵</td>
</tr>
<tr>
<td>U-235</td>
<td>3.7 × 10⁰</td>
<td>8.5 × 10⁰</td>
<td>3.4 × 10⁻³</td>
<td>5.6 × 10⁻³</td>
</tr>
<tr>
<td>U-238</td>
<td>7.1 × 10⁰</td>
<td>1.6 × 10¹</td>
<td>1.1 × 10⁻¹</td>
<td>1.4 × 10⁻¹</td>
</tr>
<tr>
<td>Pu-239</td>
<td>2.4 × 10⁻¹</td>
<td>2.3 × 10¹</td>
<td>7.2 × 10⁻⁷</td>
<td>5.3 × 10⁻⁶</td>
</tr>
<tr>
<td>Am-241</td>
<td>5.1 × 10⁻¹</td>
<td>3.7 × 10⁰</td>
<td>6.6 × 10⁻⁵</td>
<td>1.1 × 10⁻⁴</td>
</tr>
</tbody>
</table>

a The total dose rate to the whole body has been taken to be the dose rate from internal sources plus the dose rates delivered by γ rays from external sources in the water and sediment. The β dose rate from contamination in the water is, except for H-3, small compared with that from internal sources or from γ radiation from the same radionuclide in the sediment. The β radiation from nuclides in the sediment would be a significant source of additional exposure for small fish in contact with the sediment for certain nuclides, e.g. Sr-90, Tc-99 and U-238; however, it must be remembered that the dose rate from internal sources for a small fish would be significantly less than that estimated from the BIORAD model.
The values of the parameters used in these calculations are given in Table III, together with the estimated water concentration which would result in an exposure of 1 mSv·a\(^{-1}\) to humans from the combination of the three pathways considered.

3.2.1.2. Estimation of dose rates to fish

The BIORAD model provides estimates for the exposure of fish from radionuclides in the water and those which are accumulated within the body. The model results have been published in the report of the National Research Council of Canada [55] in terms of mrem·a\(^{-1}\) per μCi·mL\(^{-1}\). The corresponding data are given in Table IV in terms of mGy·a\(^{-1}\) per Bq·L\(^{-1}\) after application of the conversion factor 2.7 \times 10\(^{-10}\).

There remains the problem of estimating the dose rate to the fish from the contaminated sediment, a source which is not included in the BIORAD model. For present purposes, it has been assumed that the \(K_d\) (Table III) can be applied to the concentration of the radionuclide in the water to estimate the concentration in the sediment, where it is conservatively assumed to be uniformly distributed with depth. The dose rate at the sediment surface \(H_E(S)\) (mGy·a\(^{-1}\)) may then be estimated from:

\[
H_E(S) = 2.52 \times 10^{-3} \, C(W) \, K_d \, E
\]

where the symbols are as defined previously except that \(E\) represents the total energy of the decay modes and radionuclides being considered. Assumptions have been made concerning the presence of radioactive daughters in the sediment and these data, together with the total β and γ decay energies [182], are given in Table V. The exposure to α particles emitted by radionuclides in the sediment has not been included in the dose rate estimates, although this source can contribute to the exposure of the skin of organisms in contact with the sediment and of the lining of the digestive tract of organisms which ingest sediment with food. The estimated dose rates are given in Table VI.

3.2.2. Results

The highest absorbed dose rate, at 120 mGy·a\(^{-1}\) (0.33 mGy·d\(^{-1}\)), is estimated to arise from \(^{90}\)Sr (and its daughter, \(^{90}\)Y, at equilibrium), mainly from the contaminated sediment. Given the short range of the β particles in water and the unlikelihood of a fish spending all of its time in contact with the sediment, this will certainly be an overestimate. In addition, it should be noted that this is the maximum dose rate at the point of contact between the fish and the sediment; the dose rate will decrease rapidly with depth in the body of the fish. Internal contamination with \(^{239}\)Pu delivers the next highest absorbed dose rate. The BIORAD model does not include...
a factor to account for the biological effectiveness of $\alpha$ particles and it may be conservatively estimated that the biologically effective dose from this nuclide could be a factor of 20 greater at 456 mGy·a$^{-1}$ (equivalent to 1.24 mGy·d$^{-1}$). Thus it can be seen that the dose rates to fish, under the defined conditions of human utilization of the environment, will be less than the 10 mGy·d$^{-1}$ and it is very unlikely, therefore, that there will be any impact on fish populations (see Section 2.3). Clearly, this conclusion is very dependent upon the assumptions made concerning the extent of human exploitation of the particular freshwater environment. If any or all of the parameters fish consumption rate, water consumption rate and occupancy rate were to be decreased, the potential exposure of the fish would increase. Thus, although it appears that in this particular case protection of man will provide protection of the aquatic environment, this cannot be a general conclusion and each site should be subject to a specific assessment.

3.3. WASTE REPOSITORIES

Of the many possible types of waste repository, only shallow landfill is considered, since this type is the only one where the possibility of intrusion by humans and by plants and animals is not remote. The scenario considered is one in which the radioactive wastes are homogeneously mixed with soil and the mixture of soil and waste extends from the ground surface to a depth of several metres. No waste covers or biobarriers are present in the assumed scenario. It is also assumed that the repository is closed and no longer under institutional control. Therefore, the land may be used for human residences and gardens. Alternatively, the land may become invaded by natural populations of soil dwelling organisms and by higher plants and animals.

3.3.1. Methodology

3.3.1.1. Estimation of the source term

As in previous scenarios a source term for the estimates of dose rates to plants and animals was determined. In this case it is the equilibrium concentration of radionuclides in soil which would deliver a dose of 1 mSv·a$^{-1}$ to humans living on it breathing the air above it and deriving their sustenance from it. All potential pathways, such as external exposure, inhalation of dust or gaseous radionuclides (e.g. $^{222}$Rn plus daughters in the case of $^{226}$Ra), and ingestion of plants, animal products and well water, require consideration in the calculation of the limiting soil concentration. Calculations of this kind have been carried out for radioisotopes of Pu by Simmonds et al. [183], and by expert groups of the NEA [184] and the IAEA [185].
From the publications cited above, the most conservative (i.e. maximum) soil concentrations, scaled to an effective dose equivalent rate to humans of 1 mSv·a⁻¹, were used. The scenarios chosen were the more limiting cases of continuous human occupancy of former waste repository sites in which inhalation, external exposure and ingestion of well water and of locally produced foods would contribute to the effective dose equivalent rate. The reference soil concentrations thus compiled (Table VII) were then used as the basis for estimating dose rates to natural organisms that might also occupy the former waste repository sites.

3.3.1.2. Estimation of dose rates to soil organisms

The first step in the calculation of dose to biota was the computation of a dose rate to soil, using the conservative assumption that all the decay energy would be absorbed in the soil. Next, the dose rate to soil organisms (e.g. invertebrate animals and microflora) was estimated. Although it was not originally anticipated that soil organisms would be as critical as higher plants and animals, owing to their comparatively high resistance to radiation [119, 186-190], it was suggested in a report by Coughtrey [191] that communities of soil organisms could be disrupted by doses on the order of "a few rad per year". Since this is in serious conflict with most literature on the subject, and since doses to soil organisms would be generally higher than doses to higher plants and animals, it was decided that the doses to soil organisms should be at least crudely estimated.

In the cases of $^3$H, $^{90}$Sr, $^{99}$Tc, $^{129}$I, $^{137}$Cs and $^{226}$Ra, the dose rates to soil organisms were assumed equal to those for soil. This approximation should be reasonable for biologically mobile radionuclides, especially since the organisms are generally small in comparison with the range of the major radiations. In the cases of $^{14}$C, $^{239}$Pu and $^{241}$Am, however, concentration ratios relative to soil of $10^{-4}$, $10^{-2}$ and $10^{-2}$, respectively, were assumed. As before, a factor of 20 was included in the case of $\alpha$ emissions from $^{226}$Ra, Ra progeny, $^{239}$Pu and $^{241}$Am to account for the greater biological effectiveness per unit absorbed dose.

The estimated dose rates to soil organisms are listed in Table VII.

3.3.1.3. Estimation of dose rates to plants

Internal dose rates to plant tissues were estimated from the reference soil concentrations using the same plant/soil concentration ratios as were used to derive the reference soil concentrations. It was conservatively assumed that all the $\alpha$ and $\beta$ decay energy and 10% of the $\gamma$ decay energy was absorbed. The factor of 20 was again used for all $\alpha$ emissions. For $\gamma$ emitters, it was assumed that external dose to plants was 0.5 times the $\gamma$ dose to soil. For strong $\beta$ emitters, the external dose was taken as 0.1 times the soil dose. The contribution of $\alpha$ and weak $\beta$ emitters to the external dose to plants was assumed to be negligible. External and internal doses
TABLE VII. SHALLOW LANDFILL SCENARIO: CALCULATED ENVIRONMENTAL QUANTITIES CONSISTENT WITH A DOSE RATE OF 1 mSv·a⁻¹ TO HUMANS a,b

<table>
<thead>
<tr>
<th>Nuclide</th>
<th>Concentration in soil to give an effective dose equivalent rate of 1 mSv·a⁻¹ to human residents (Bq·kg⁻¹)</th>
<th>Basis for reference concentration in soil</th>
<th>Upper estimate dose rate to soil organisms (mGy·d⁻¹)</th>
<th>Upper estimate dose rate to plants (mGy·d⁻¹)</th>
<th>Upper estimate dose rate to animals (mGy·d⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>H-3 (as HTO)</td>
<td>$2 \times 10^7$</td>
<td>c, d</td>
<td>$1 \times 10^0$</td>
<td>$7 \times 10^{-1}$</td>
<td>$7 \times 10^{-1}$</td>
</tr>
<tr>
<td>C-14</td>
<td>$7 \times 10^7$</td>
<td>c, d</td>
<td>$5 \times 10^0$</td>
<td>$5 \times 10^{-2}$</td>
<td>$5 \times 10^{-2}$</td>
</tr>
<tr>
<td>Sr-90</td>
<td>$1 \times 10^4$</td>
<td>c, d, e</td>
<td>$2 \times 10^{-1}$</td>
<td>$6 \times 10^{-2}$</td>
<td>$2 \times 10^{-2}$</td>
</tr>
<tr>
<td>Te-99</td>
<td>$8 \times 10^4$</td>
<td>c</td>
<td>$1 \times 10^0$</td>
<td>$2 \times 10^{-1}$</td>
<td>$4 \times 10^{-2}$</td>
</tr>
<tr>
<td>I-129</td>
<td>$3 \times 10^4$</td>
<td>d</td>
<td>$4 \times 10^{-2}$</td>
<td>$9 \times 10^{-3}$</td>
<td>$8 \times 10^{-3}$</td>
</tr>
<tr>
<td>Cs-137</td>
<td>$7 \times 10^4$</td>
<td>c, d, e</td>
<td>$8 \times 10^{-1}$</td>
<td>$4 \times 10^{-1}$</td>
<td>$4 \times 10^{-1}$</td>
</tr>
<tr>
<td>Ra-226</td>
<td>$3 \times 10^2$</td>
<td>c, f</td>
<td>$1 \times 10^0$</td>
<td>$7 \times 10^{-1}$</td>
<td>$2 \times 10^{-2}$</td>
</tr>
<tr>
<td>Pu-239</td>
<td>$4 \times 10^5$</td>
<td>c, d, g</td>
<td>$5 \times 10^0$</td>
<td>$5 \times 10^{-1}$</td>
<td>$1 \times 10^{-2}$</td>
</tr>
<tr>
<td>Am-241</td>
<td>$6 \times 10^4$</td>
<td>c</td>
<td>$9 \times 10^{-1}$</td>
<td>$1 \times 10^{-1}$</td>
<td>$1 \times 10^{-2}$</td>
</tr>
</tbody>
</table>

a All internal and external pathways are considered.
b It is assumed that wastes are homogeneously mixed in soil and there is no soil cover or biobarrier.
c [184].
d [185].
e Occupancy of the site by humans and by plants and animals begins 50 years after repository closure.
f Radium daughters in soil are considered.
g [183].
h A factor of 20 has been applied to account for the enhanced biological effectiveness per unit absorbed dose.
were summed. In the case of $^{226}$Ra, it was assumed that the chain of progeny in 50% of secular equilibrium was present and contributed to the dose.

The resulting estimates of dose rates to plants are listed in Table VII.

### 3.3.1.4. Estimation of dose rates to animals

Internal dose rates to animal reproductive tissues were generally estimated using the same approach as outlined for the atmospheric release scenario (Section 3.1.1). Intakes from both plant and soil ingestion were considered. In the case of $^{99}$Tc, it was assumed that the transfer coefficient to reproductive tissue was $10^{-2}$ d·kg$^{-1}$ [171]. The other exceptions were for $^3$H and $^{14}$C, where it was assumed that the animal tissues would be in 100% equilibrium with the $^3$H and $^{14}$C in plant tissues. Daughters of $^{226}$Ra were considered to contribute to the dose to animal tissues and $\alpha$ emissions were assigned a factor of 20 to account for their biological effectiveness. External $\gamma$ dose rates to animals were assumed equal to external dose rates to plants (i.e. 0.5 times the dose rate to soil). Inhalation was assumed to be a negligible dose contributor, except for $^{239}$Pu and $^{241}$Am.

The resulting estimates of dose rates to animal reproductive tissues are listed in Table VII.

### 3.3.2. Results

The results of the calculations are presented in Table VII. Reference soil concentrations, which ranged from $3 \times 10^2$ Bq·kg$^{-1}$ for $^{226}$Ra to $7.3 \times 10^7$ Bq·kg$^{-1}$ for $^{14}$C, were taken from previous calculations and assessments [183–185].

The reference soil concentrations were used as the basis for estimating, using very simplified and conservative assumptions, the upper estimate dose rates that might be experienced by soil organisms, higher plants, and reproductive tissues of vertebrates which might also occupy the site of the former shallow landfill waste repository (Table VII). The general lack of specific dosimetric data that could be used in such calculations made it necessary to adopt the simplistic approach used. It is judged, however, that the assumptions were conservative, so as to produce upper bound estimates of dose rates to the biota. If more accurate and realistic calculations were feasible, it is believed that the resulting dose rates to biota would be substantially lower than those presented.

The upper estimate dose rates to soil organisms were 5 mGy·d$^{-1}$ or less for all radionuclides considered. It is believed that the preponderance of scientific literature would strongly support the position that a dose rate of this order would not cause measurable perturbations in populations of soil microorganisms or the larger soil invertebrates. Literature that supports such a position includes Refs [24, 119, 186–190, 192, 193].
In the case of higher plants and animals, upper estimate dose rates resulting from the reference levels of radionuclides in soil were all less than 1 mGy·d⁻¹. Calculated dose rates to plants tended to exceed those to animals. These findings were essentially the same as those in the case of controlled atmospheric releases.

Although the number of radionuclides considered is limited, it is believed that they are sufficiently representative to allow firm conclusions to be drawn.

4. SUMMARY AND CONCLUSIONS

4.1. SUMMARY

The purpose of this report is to determine whether the statements in ICRP publications [1, 2] (Section 1.1) about the protection of non-human organisms and populations are consistent with current knowledge, and if not, to determine whether radiation protection standards for aquatic and terrestrial biota are warranted.

This has been achieved by summarizing the dose rates and total doses which have been determined, experimentally and in field situations, to produce observable effects at individual, population and community levels of biological organization for aquatic and terrestrial biota. These data were then compared with the estimated maximum possible dose rates which could be received by aquatic and terrestrial biota under current radiation protection standards for man.

While the risk of effects on some individual terrestrial and aquatic organisms would appear to be as high as, or higher than, for humans, the anthropocentric view is that these natural populations are considered and valued more as populations than as identifiable individuals. However, in certain cases emphasis may be given to individual organisms, such as those belonging to rare and endangered species, or to species of low fecundity, domesticated animals, etc. In general there are levels of radiation dose which may produce occasional harmful effects to a few individuals in a population without resulting in any noticeable deleterious effect on the maintenance of the population as a whole.

On the basis of the literature reviewed and summarized in Section 2, it is apparent that reproduction is the most radiation sensitive process in the life cycle and is therefore the limiting end point in terms of population maintenance. In the terrestrial environment, sensitivity to chronic radiation varies markedly among different taxa; certain mammals, birds and reptiles and a few tree species appear to be most sensitive. While every species of plant and animal could not be examined, there is a considerable body of data from which to derive generalized conclusions.
It would appear that chronic dose rates of 1 mGy \( \cdot \text{d}^{-1} \) or less to even the more radiosensitive species in terrestrial ecosystems are unlikely to cause measurable detrimental effects in populations and that up to this level adequate protection would therefore be provided.

In the aquatic environment developing embryos and the process of gametogenesis are considered critical from the standpoint of radiosensitivity. In general, aquatic organisms are no more sensitive than other organisms; however, because they are poikilothermic, temperature can control the time over which radiation effects are expressed. Of interest is the fact that the radiation-induced mutation rate observed for aquatic organisms lies between that of *Drosophila* and that of the mouse. From limited population studies it appears that an increased frequency of deleterious genes in the gene pool does not necessarily result in any loss of viability for populations.

In the aquatic environment it would appear that limiting chronic dose rates to 10 mGy \( \cdot \text{d}^{-1} \) or less to the maximally exposed individuals in a population would provide adequate protection for the population.

This review suggests that, provided other environmental factors are favourable, one would expect populations exposed to chronic dose rates less than those specified above to maintain the normal characteristic potential for reproduction, growth and general vigour. It is recognized that the radiosensitivity of many species has not been investigated, that many effects are based on evidence from external \( \gamma \) ray exposures only, and that natural community studies involving multiple interactive stresses are limited. However, it is believed that even with this additional information, the dose rate limits derived here would not be changed significantly.

With the information reviewed and summarized, simplified and conservative estimates of dose rates to biota for three controlled release scenarios, namely atmospheric, aquatic and disposal to a landfill, were calculated. The radiation dose rates received by representative terrestrial and aquatic biota when human exposure is constrained within the limitation of 1 mSv \( \cdot \text{a}^{-1} \) were estimated. These estimates indicate that the ICRP statements (Section 1.1) are reasonable, at least within the generic scenarios considered here.

Confidence is felt in this assertion since it is believed that the assumptions used in the estimations of doses to plants and animals were conservative. For those natural organisms which are not usually included in standard dose assessments for human exposure, additional conservative assumptions were made. As a result of this conservatism, actual doses to critical tissues could probably be as much as 1–2 orders of magnitude less than the estimates calculated in the generic scenarios.

While a value of 1 mSv \( \cdot \text{a}^{-1} \) for human exposure was used in the calculations, the prevailing radiation protection philosophy is to reduce doses to as low as reasonably achievable (ALARA) and, as a result, actual releases to the environment are typically small fractions of those assumed. Such a philosophy also minimizes the impact upon the natural populations in the same area.
Of course there may be exceptions to the scenarios presented. The human population group which is exposed may be separated geographically from a potentially exposed population of organisms, rare and endangered populations with very low fecundity may be present in the exposed area, and natural populations may be under ecological stress from a variety of natural or man-made pressures. Each of these situations should be assessed on a site specific basis and actions taken to maintain the exposure rates within the limits appropriate for the organisms' protection.

4.2. CONCLUSIONS

(a) There is no convincing evidence from the scientific literature that chronic radiation dose rates below 1 mGy·d⁻¹ will harm animal or plant populations. It is highly probable that limitation of the exposure of the most exposed humans (the critical human group), living on and receiving full sustenance from the local area, to 1 mSv·a⁻¹ will lead to dose rates to plants and animals in the same area of less than 1 mGy·d⁻¹. Therefore, specific radiation protection standards for non-human biota are not needed.

(b) In practice, when the recommended system for radiation protection is applied, the critical human group will receive doses well below the annual dose limit of 1 mSv. Furthermore, the calculations of dose to natural organisms are thought to be conservative by 1-2 orders of magnitude. For these reasons the chronic dose rate to animals and plants should be substantially less than 1 mGy·d⁻¹ under prevailing radiation protection standards.

(c) Analyses leading to the above conclusions were based on generalized information and conservative assumptions even when applied to realistic scenarios. Some situations, such as cases of prolonged exposure of a human critical group approaching 1 mSv·a⁻¹, combined with the existence of specific ecological conditions such as the presence of rare or endangered species or combined stresses, may require site specific analyses.
REFERENCES


[65] TABONE, E., Influence d'une irradiation gamma chronique sur le système sol d'une chênaie mixte méditerranéenne à Cadarache, Thèse doct. 3ème cycle, Univ. of Provence, Marseille (1986) 65 pp.


ENGEL, D.W., “Experimental approaches to demonstrate the interactions of radiation with other environmental stresses”, Methodology for Assessing Impacts of Radioactivity on Aquatic Ecosystems, Technical Reports Series No. 190, IAEA, Vienna (1979) 301.

ENGEL, D.W., SHELTON, M.G., WHITE, J.C., Jr., The effect on embryos and young of rainbow trout from exposing the parent fish to X-rays, Growth 13 (1949) 119.


ETOH, H., EGAMI, N., Damage accumulation and recovery in the fish *Oryzias latipes* exposed to fractionated or protracted radiation at different temperatures, Radiat. Res. 32 (1967) 884.

FOSTER, F.F., DONALDSON, L.R., WELANDER, D., BONHAM, K., SEYMOUR, A.H., The effect on embryos and young of rainbow trout from exposing the parent fish to X-rays, Growth 13 2 (1949) 119.

HYODO, Y., Development of intestinal damage after X-irradiation and \(^{3}H\)-thymidine incorporation into intestinal epithelial cells of irradiated goldfish, *Carassius auratus*, at different temperatures, Radiat. Res. 26 (1965) 383.


— Increased embryo production following low doses of radiation to trout spermatozoa, Radiat. Res. 51 (1972) 402.


WELANDER, A.D., DONALDSON, L.R., FOSTER, R.F., BONHAM, K., SEYMOUR, A.H., The effects of roentgen rays on the embryos and larvae of the Chinook salmon, Growth 12 (1948) 203.

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