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PROTECTION OF THE ENVIRONMENT: CURRENT STATUS AND FUTURE WORK

International Union of Radioecology

Radioecology is the scientific discipline for the study of the interactions between living organisms and radioactivity in the environment

Protection of the Environment: Current Status and Future Work

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1. Introduction

In 1997 the IUR initiated a Task Group, as part of a 2-year EC-funded co-operative study with EULEP and EURADOS (EC IVth Framework), to consider the issue of protection of the environment from the effects of ionising radiation. During this period the Task Group considered, and took note of, a number of initiatives and ideas that were being developed within the radioecological community. These included activities by regulatory bodies within different countries, international bodies and by individual researchers. It also considered the broader, socio-economic context within which these ideas were beginning to evolve.

The Task Group fairly quickly came to the conclusion that a broad and systematic approach was needed in order to develop a framework [Pentreath 1998, 1999] within which the majority – if not all – of these different initiatives could be accommodated [IUR, 2000]. It therefore promoted the development of such a system [Strand et al., 2000; Strand and Larsson, 2001] and worked with other organizations to help achieve it.

Since then, and particularly in the last three years, interest has grown, and the IUR Task Group has therefore played a role in supporting a number of groups, meetings, seminars and conferences on the subject. It has also continued to identify areas of interface between the development of such an approach, the radioecological community, and the broader scientific and research community in general. This report therefore briefly examines the current state of the subject and seeks to identify those areas in which the IUR will continue to play a vital role over the next 5 to10 years.

2. Recent Initiatives and Current Status

Since 1997, events have moved rather rapidly and the IUR, both as an organization and via its individual members, has been actively involved in them. These include:

- the development of ethical and legal aspects in general [Oughton, 2002ab]
- work by the IAEA on the ethical issues [IAEA, 2002]
- consensus building (IUR Oslo conference, 2001) [Strand and Oughton, 2002] and in dialogues with the industry and
- regulators [NEA/ICRP Taormina, 2001].

The IUR has also taken part in the work of the ICRP Task Group [ICRP, 2002], was an NGO-participant at the World Summit on Sustainable Development in Johannesburg [UN, 2002] and hosted a symposium in Monaco in co-operation with IAEA, NRPA and JER (IUR/IAEA, 2002) at which papers and discussions on environmental protection played a major part. IUR members have also been active in developing ideas and methodologies within their own countries (e.g. Domotor, Strand, Pentreath, Copplestone). Much has been learned from the work of national programmes, particularly in the USA, where the DOE has developed requirements and guidance for the radiological protection of the environment, and has currently in place a radiation dose limit for protection of aquatic biota [USDOE, 1990] and has proposed limits for protection of terrestrial biota [USDOE, 1996] for some of its own facilities. The USDOE developed screening methods using a set of reference organisms within a graded approach for demonstrating protection of biota applicable to these dose rate guidelines [USDOE, 2002; Higley et al., 2003a,b,c]. The USDOE approach employs a set of derived radionuclide concentrations, termed “Biota Concentration Guides” (BCGs), for screening sediment, water, and soil media. Principles and issues to be considered in the evaluation of radiation as a stressor to the environment as part of ecological risk assessments are also provided [Jones et. al, 2003]. Finally, both the IUR and its members have been active in two large multinational Research and Development projects: EPIC and FASSET [Larsson et al, 2002].

In view of all this activity, it is not necessary to set out, again, the basic elements of this new approach. This has been adequately described elsewhere [Pentreath, 1999, 2002; Copplestone et al., 2001]. The work of the IUR has been recognised on the political and general environmental scene and references made to the proposed IUR system (e.g. Protection of the North Sea, 2002; Arctic Monitoring and Assessment Programme, AMAP). But the IUR recognises that this is just the beginning of a new era, and one in which it is well placed to provide expert advice, data, and information both to develop the theory and methodology of a system to protect the environment – as is likely to be pursued by the ICRP [Holm and Strand, 2002; ICRP, 2002] and to put all of this into international and national practice (by way of the

IAEA and regional and national organizations).

The IUR therefore sees this future role as including input on the following:

- the basic radiobiological information and scientific understanding required;
- the basic radioecological information and scientific understanding required;
- aspects of the broader developing field of the ethical, social, legal and economic components of environmental protection in general – via the social and economic sciences;
- the interface with the developing scientific work with regard to other forms of pollution control, particularly with regard to ecotoxicology and its related fields; and
- the interface with the basic ecological sciences, particularly with regard to the new and qualitative approaches being developed within the fields of conservation, habitat protection, ecosystem health/integrity, and so on.

As yet it is clearly not possible to identify the likely needs in all of these areas, but the IUR Task Group has already considered a number of them, and these are discussed in the next sections.

3. Ethical, Legal and Social Aspects

In practice, environmental protection is influenced by a multitude of factors, and decisions made will depend on a variety of political, social, cultural and scientific issues. These are often controversial, requiring a variety of trade-offs, and thus the eventual choices made are by nature both value and science based. In many cases the conflicts are inherent in any environmental issue, such as difficulties in balancing the interests of future against present generations, of industry against the general public, and of developed against developing countries. Protection of the environment from radiation is no exception. To help put the problem into context, therefore, a good starting point is to examine briefly the legal and philosophical basis of environmental protection, and the various values, goals and principles that guide its policy and application (Figure 1).

3.1 The background of legal thinking on environmental protection

Since the 1970s there has been a growing worldwide recognition of man's impact on the natural environment and the need for measures to protect and preserve the Earth's resources. The United Nations Stockholm conference on the Human Environment in 1972 increased political awareness of the global nature of many environmental threats, and coincided with the creation of the United Nations Environment Programme. At the same time the European Economic Community (now the EU) instigated an environmental action programme. At the time of UNEPs creation in 1972 only 11 countries had environmental agencies. Ten years later that number had grown to 106, of which 70 were in developing countries. Twenty years later, the UN conference on Environment and Development in Rio saw the largest gathering of world leaders in history, with delegates from 178 countries attending. Also known as the Earth Summit, the conference produced the Rio Declaration (a 6 page statement calling for the integration of the environment with economic concerns) and Agenda 21 (a 900 page blueprint for environmental development). The related Climate Convention and the Biodiversity Convention were also drawn up, both in the form of legal agreements. At a follow-up conference in Kyoto in 1997, 160 countries signed another new legal agreement known as the Kyoto Protocol, calling for all industrialised nations to reduce emissions of global warming gasses.

A major focus of UNEP has been on **sustainable development** – increasing standards of living without destroying the environment. The UN World Summit on Sustainable Development (WSSD) held in Johannesburg, August 2002, outnumbered even Rio in terms of the actual numbers of attending delegates – over 30,000. The WSSD aimed to consolidate the intentions expressed within the Rio declaration and Agenda 21 into concrete implementation mechanisms incorporating targets and timetables. Summit participants debated a wide of issues having implications for sustainable development, including negotiations on problems related to health, energy, water, agriculture, and biodiversity. The final "Plan of Implementation", contained a number of sections relevant for environmental radiation protection including section 33.bis on radioactive wastes:

"Governments, taking into account their national circumstances, are encouraged, recalling paragraph 8 of resolution GC (44)/RES/17 of the General Conference of the International Atomic Energy Agency (IAEA) and

taking into account the very serious potential for environment and human health impacts of radioactive wastes, to make efforts to examine and further improve measures and internationally agreed regulations regarding safety, while stressing the importance of having effective liability mechanisms in place, relevant to international maritime transportation and other transboundary movement of radioactive material, radioactive waste and spent fuel, including, inter alia, arrangements for prior notification and consultations done in accordance with relevant international instruments."

and section 22 on "chemicals":

"Renew the commitment, as advanced in Agenda 21, to sound management of chemicals throughout their life cycle and of hazardous wastes for sustainable development and for the protection of human health and the environment, inter alia, aiming to achieve by 2020 that chemicals are used and produced in ways that lead to the minimization of significant adverse effects on human health and the environment, using transparent science-based risk assessment procedures and science-based risk management procedures, taking into account the precautionary approach, as set out in principle 15 of the Rio Declaration on Environment and Development, and support developing countries in strengthening their capacity for the sound management of chemicals and hazardous wastes by providing technical and financial assistance." [UN, Plan of Implementation, final version, 2002]

These new developments in environmental protection have not, however, been achieved without controversy. Regulation requirements have forced trade-offs to be made between economic development and environmental protection, between the interests of present and future generations, and between the wishes of the West and the needs of the developing countries. At Stockholm, some developing countries were considered to be fearful that a focus on environmental protection was a means for the developed world to keep the undeveloped world in an economically subservient position, worries that were still apparent in Rio and Johannesburg. On the other hand, it has also been thought that the approaches of some developed countries have proved to be amongst the strongest obstacles to the environmental process (e.g., the USA's failure, so far, to sign the

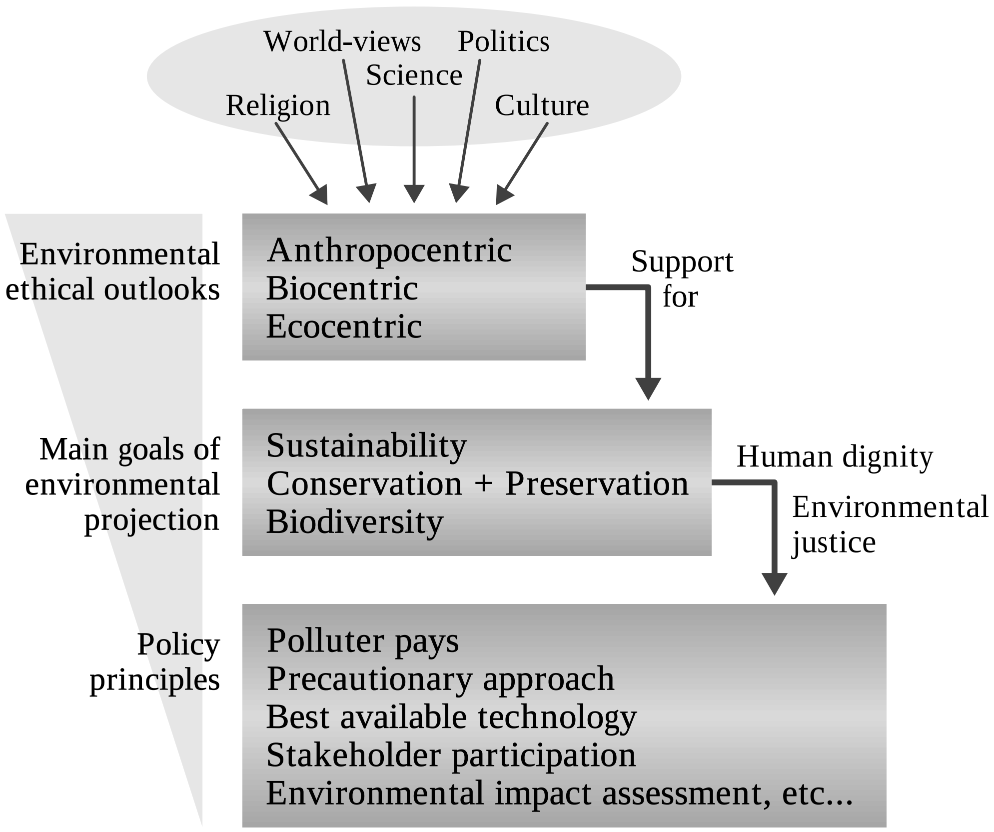

Figure 1. Links between ethical outlooks on environmental value (the "whys") and the goals and principles of environmental protection (the "hows").

Kyoto and biodiversity conventions). Conflicts between economic and human development and environmental risk are therefore not only confined to the developing countries! Targeted timetables were agreed in Johannesburg for actions related to water quality and biodiversity; negotiations on energy and climate fell short of any concrete targets, although there was a general reaffirmation of the Rio Principles. The above chemicals section attracted a large amount of controversy (and discussion time) during negotiations, with disagreements over the use and interpretation of the precautionary principle, the phrasing of "minimization of significant adverse effects", and the relation of "science" to risk assessment and management [IISD, 2002].

The recent history of environmental protection and legislation illustrates three points to bear in mind when considering protection of the environment from radiation. First, some aspects of environmental protection are relatively new and still undergoing development. Second, the issue is a global one, deemed important by both governments and the public, and has therefore stimulated action on an international scale. And third, practical solutions are not without conflicts and controversy. Not withstanding these difficulties, examples of environmental law can be found in the national laws of every country. Although their scope and detail can vary considerably, progress during the last 30 years has led to a certain amount of agreement on what we mean by environment and its protection and which principles should guide that protection.

3.2 The evolving ethical basis for environmental protection

The question of how and why one attaches value to the environment, and what exactly has moral status, is arguably the most fundamental debate within environmental ethics [Rolston, 1988; Sterba, 1994]. Central philosophical issues include whether the environment has intrinsic or inherent value (i.e., value in itself) or extrinsic or instrumental value (i.e., value because of human interest). Supporters of an anthropocentric (literally "human centred") ethic argue that only human beings have moral standing, and consequently environmental protection is only important in so far as it affects the interests or experiences of humans. Many point out that human interests can and indeed do provide a powerful set of motives for protecting nature [Wilson, 1984; Sober, 1986, Norton, 1988, 1991], but they are not concerned about human impact on other life-forms as such, suggesting that only humans are capable of "valuing" in an ethical sense [Frankena, 1973; 1979].

Proponents of non-anthropocentrism reject the notion that moral value can only be derived and justified in terms of human interests, they can be roughly divided into two main outlooks biocentric (literally "life-centred") and ecocentric ("ecosystem-centred"). Concerning the question of who or what has moral standing, for an anthropocentric the obvious answer is only humans, whereas the non-anthropocentrics have a variety of answers depending on the criteria they use for determining why something has moral standing. Biocentrists extend moral standing to all or part of the animal or biological kingdom, arguing that, like humans, some animals are sentient, i.e. capable of feeling pleasure and pain [Singer, 1991], or that some animals should be awarded "rights" [Regan, 1980]. Others contend that the fact that all living organisms have some "goal" in existence is sufficient in order to grant moral status and duties on humans to respect that existence [Taylor, 1986; Goodpaster, 1978]. Such views are also classified as an "individualistic" environmental ethic in that the locus of moral standing is put on individual living beings.

Ecocentrists suggest that moral standing should be extended to encompass whole ecosystems, including living organisms and their habitats. Many clam that mankind needs a radical change from an anthropocentric attitude of domination and exploitation of natural resources towards a greater respect for the integrity of nature and ecosystems as a whole [Callicott 1979, 1989; Næss, 1974].

One can raise such questions about value and moral standing in assessments of any environmental stressor. However, it is important to note that although philosophers might disagree about the *way* in which the environment should be valued, almost all philosophers would agree that damage to the environment should *matter* in risk assessment – be this because nature has value in itself or because of the potential consequences for human generations [Oughton, 2002a; 2003]. Sustainable development and conservation of biodiversity can be endorsed both from human interest and for the sake of the environment itself. Prevention of suffering and a respect for the integrity of nature can also be defended simply because humans wish it to be so. Recent work by the IAEA suggested that, despite the apparent diversity of values in the different ethical outlooks, one could find reasonable consensus on principles of environmental protection. They identified five common principles, namely: conservation of habitat and species; maintenance of biodiversity; sustainability; environmental justice and human dignity [IAEA, 2002]. A clear support for these five principles was obtained at the IUR consensus conference in Oslo, 2001, which was attended by participants representing a wide range of disciplines connected to radiation protection and environmental protection. Participants also identified a need for "development of policy in an open, transparent and participatory manner", that "the best available technology including consideration of economic costs and environmental benefits should be applied to control any release of radionuclides into the environment", and supported a precautionary approach to risk management [Strand and Oughton, 2002].

In summary, it is important to acknowledge that any system of protection for the environment has to be flexible enough to incorporate both anthropocentric and non-anthropocentric values, and therefore to allow for the protection of individuals, populations, or communities and ecosystems.

3.4 Bridging science and policy: decision-making and uncertainties

Even if philosophers and/or policy makers could settle the question of *why* the environment needs protecting, there still remains the problem of *how* best to achieve that protection. There is the issue of who to include in the decision making process, how costs and benefits should be distributed, and what to do about uncertainties. Although scientific knowledge is only one of many factors that need to be considered in the decision-making process, it is an important input, yet one that often seems in danger of being overlooked in policy. Part of the problem might be attributed to the lack of integration between the disciplines of risk assessment (attempts to discern what detrimental effects radiation exposures might have on the environment) and risk management (decisions on how to control, monitor and reduce those effects). It can help if scientists are aware of the social, ethical and legal influences, and provide information in a form that can best enlighten the decisions.

At present, EU environmental policy is explicitly based on the principles of best available technology, polluter pays and the precautionary principle. The recent Århus convention stresses the need for providing public information on environmental issues and for stakeholder participation in decision-making [Århus, 2001]. During the past few years, similar policy has been adopted in many countries, and support for these principles was reaffirmed in Johannesburg 2002 [UN, 2002]. All four principles call for judgements based on a mixture of scientific, legal and social knowledge.

The issue of scientific uncertainty in prediction of consequences is central to the way scientific knowledge is used in policy making. Such uncertainties operate at many different levels, but can be broadly divided into those that arise from the inherent variability and complexity of ecosystems and those that reflect basis research needs in the effects of radiation exposure in flora and fauna. Gaps in knowledge include, *inter alia*, the variable radiological sensitivity of different species to different radionuclides, effects at different ecosystem levels, and variability of responses at acute and chronic exposures. Different radionuclides, and their chemical forms, can show varying absorption, metabolism, distribution and effect amongst species. As in all risk assessments, care is needed when extrapolating from one case to another, and there can be fundamental disagreements over how one selects the criteria for determining levels of no adverse effect. When does a biological (or even non-biological) change become adverse – when it is empirically observable, statistically possible or merely biologically plausible? Does preservation of habitats mean avoiding contamination with any level of man-made radionuclides, irrespective of demonstrable harm to living organisms? Should background comparisons be made on the basis of mass concentration (which would differentiate between natural and anthropogenic radionuclides), activity concentrations (which would not necessarily make any distinction) or dose calculations (i.e., focused on biological effects)?

Such discussions have links to the debate surrounding the precautionary principle, and it should be noted that the topic is not without controversy in environmental ethics. Problems include high consequence, low probability risks (an issue familiar to radiation protection in assessment of serious nuclear accidents) and how to draw the line between a hypothetical possibility and a real, if low, probability of causing serious harm. Some might contend that, at least compared with other environmental stressors, we know enough about the effects of radiation on both humans and non-humans to conclude that the threats will not be serious or irreversible. Others might contend that the uncertainties on factors such as biological endpoints, on variability between species, and consequences for ecosystem sensitivity all support the precautionary approach.

Finally, one might point out that the rejection of the ICRP paradigm that protection of man provides an acceptable level of environment protection has arisen more from changes in social and legal thinking than due to any **new** scientific evidence. However, now that the paradigm shift has occurred, it becomes all too clear that there is a need for improvements in the basic scientific understanding if a rational and ethically defensible environmental protection policy is to be implemented. A critical factor in all of this debate, is a clear examination of the quality and validity of the science upon which such decisions can be based. This, clearly, is the area in which the IUR can make its greatest contribution, as discussed in the next section.

4. Radiobiology and Radioecology

Although the IUR considers that there is already sufficient information to start introducing an overall framework for the systematic protection of the environment from ionising radiation, drawing upon specialist reviews and interpretations of the large amount of radiobiological and radioecological information that has been gathered over the last fifty years, it nevertheless recognises the need to plug some gaps in our knowledge and to improve upon the existing data base.

The increased interest in environmental protection has highlighted a number of knowledge gaps in the scientific data on sources and effects of radiation in non-human species. Although the transfer of radionuclides is quite well known within some food-chains, there are very little data on the behaviour of radionuclides in non-temperate zones and on uptake to species that do not form

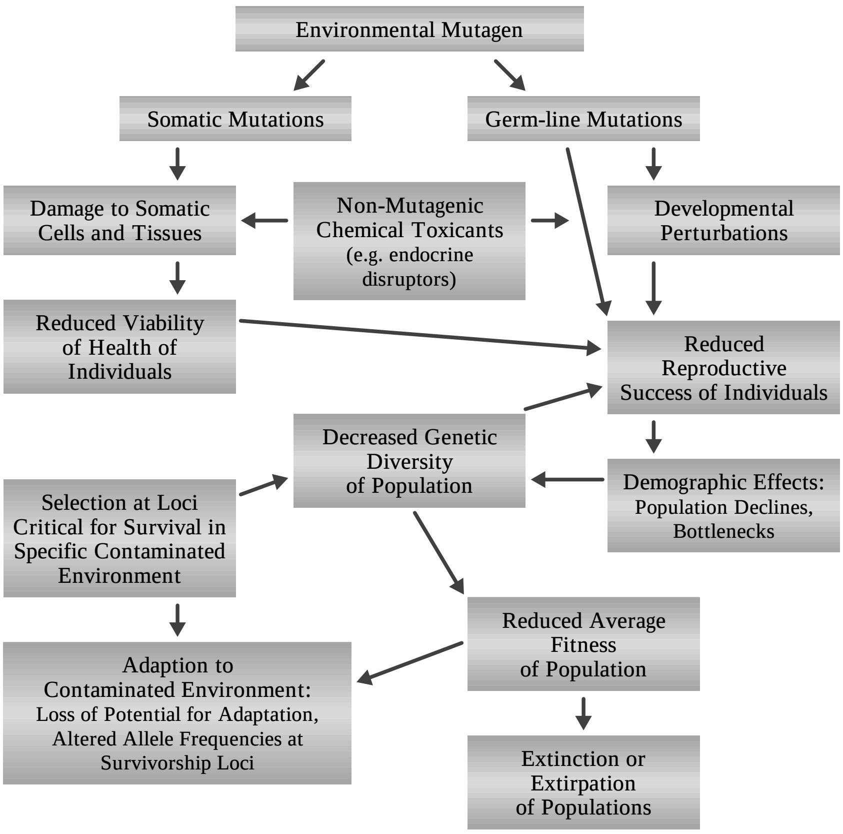

Figure 2.
Model proposed by Bickham et al [2000] to illustrate the interrelationships among factors related to chemical contamination of the environment with decreased genetic diversity of populations. The model details how mutagenic and non-mutagenic chemical contaminants affect the reproductive success of individuals by means of both heritable effects and damage to somatic tissues. Reduced reproductive success can ultimately lead to decreased genetic diversity of the population which can ultimately lead to reduced fitness and extirpation of the population (i.e. biodiversity loss).

part of the human food chain. There is a need to develop both transfer models (flux, dynamic, ecosystem, etc.) and genotoxicological biomonitoring techiniques (e.g., RNA/DNA changes, Comet assays, oxidative stress indicators) that are capable of allowing impact assessments at a variety of species, population and ecosystem levels and that could also deal with other environmental stressors (Figure 2). Mathematical models should be developed and applied to relate the effects of radiation on individuals (particularly with regard to early mortality, reproductive success, and cytogenetic damage) to potential impacts at the population level. Knowledge of the doses and effects of background radiation is lacking, as are dose-effect relationships (RBE) for a variety of species, doses and dose rates. Interaction of radionuclides with other stressors, including possible synergistic effects, is only just starting to be investigated [Suomela et al., 1999; Riekkinen and Jaakkola, 2001; Ausseil, 2001; Fraysse, 2001; Adam et al., 2002]. These and other research needs offer exciting possibilities for radioecologists in the future, and some specific areas are examined more closely below.

4.1. Quantities and unit.

THE RELEVANCE AND UTILITY OF ABSORBED DOSE (RATE)

The definition of relevant quantities, and the selection of their associated units, is fundamental to any branch of scientific endeavour and, most crucially, underlies the communication of ideas and data between practitioners and the practical application of scientific knowledge. In the radiological sciences, the required quantities relate to the ionizing radiations and their interactions with matter. From the physical viewpoint, the quantification of a radiation field, in terms of the base units of the International System of Units (SI), its interactions with matter, and its dosimetry, is reasonably well-developed [ICRU, 1998]. Indeed, mathematical expressions describing the observed initial consequences of the interactions of ionizing radiations with matter have been developed from first principles and may be applied for the purpose of radiation dose estimation [e.g., see Hine and Brownell, 1956; Attix, Roesch and Tochilin, 1968].

The current position is the culmination of a century of development and evolution in the quantities (and the units) employed to describe radiation fields. Initially, a variety of physical, chemical and biological effects of radiation were used in an attempt to quantify radiation fields and to establish correlations with biological effects. It was, however, a distinguishing physical feature of this class of radiations – its capacity to induce ionization – that emerged as the accepted basis for measurement and standardization. This led, in 1928, to the definition of the *roentgen* unit for x-rays (extended to include γ-radiation in 1937) derived from ionization measurements in air-filled chambers [Menzel and Feinendegen, 1995]. It is important to note that the *roentgen* is the unit, not the quantity; this latter has now become identified as the *exposure* [Rossi, 1990]. In most of the early studies of the biological effects of radiation, the responses are correlated with the exposure to x- and γ-rays given in *roentgen*. A rapid expansion of the use of radiation and radioactive materials, in medicine, research, and in the nuclear and other industries, took place after the second world war. This, and the concomitant requirement for radiation dosimetry for protection purposes, led to the introduction in 1953 of a more fundamental quantity – the *absorbed dose* (the energy imparted by ionizing radiation per unit mass) and its unit, the *rad* – to be correlated with the biological effects of all types of radiations [Menzel and Feinendegen, 1995].

In principle, the exposure may be converted to absorbed dose mathematically, but in practice it is difficult due to the need for detailed information concerning the incident radiation field for which the exposure is given, i.e., the energy distribution of energy radiance, and the geometrical relationships between the presumed biological target (a tissue, organ or the whole organism) and both the radiation source and the surrounding absorbing or scattering material. A convenient rule-of-thumb has been to assume numerical equivalence between the roentgen and the rad, i.e., 1 roentgen = 1 rad, although this is likely to over-estimate the absorbed dose [Hine and Brownell, 1956].

It may be taken as given that the principal biological effects of radiation are the result of ionization processes in tissue. Because ionization is the separation of orbital electrons from the parent atoms, a process that requires energy, this results in the absorption of energy from the incident radiation field. This leads directly to the definition of the radiation dose as the quantity:

$$\text{absorbed dose, } D = \frac{d\bar{\varepsilon}}{dm} \quad (1)$$

where $d\varepsilon$ is the mean energy imparted to matter of mass dm (see [ICRU, 1998] for fuller details). The quantity, absorbed dose, has SI units of J kg^{-1}, and this is given the special name gray (Gy). Although this definition relates to a limiting domain, in practical radiation protection the absorbed dose is usually determined as the average value over some specified biological entity – a tissue, organ or the whole body.

At the low dose rates and low total accumulated doses characteristic of the majority of environments contaminated by authorized releases of radioactive materials, it may well be that microdosimetric considerations become important, i.e., the distribution of absorbed energy divided by the mass of the individual cell or cell nucleus (the presumed primary targets for radiation action) becomes extremely inhomogeneous. In this case, the quantity:

$$\text{specific energy, } z = \frac{\varepsilon}{m} \quad (2)$$

where ε is the energy imparted to the matter of mass m in the defined target, may be more relevant to the determination of the consequent radiation effects.The unit of the quantity specific energy remains the J kg^{-1}, and this retains the special name gray (Gy).The specific energy may be due to one or more (energy deposition) events, i.e., the passage through the defined target mass (m) of one or more directly ionizing particle tracks.The probability that the specific energy is $\leq z$ is given by the distribution function F(z), and the probability density, f(z), is the derivative of F(z):

$$f(z) = \frac{dF(z)}{m} \quad (3)$$

Both F(z) and f(z) are dependent on the absorbed dose and radiation quality.

THE PROBLEM OF RADIATION QUALITY.

The practical application of this system of dosimetry for radiation protection purposes forces consideration of the empirical observation that the same absorbed dose of differing radiations can produce differing degrees of effect in the same biological endpoint, e.g., cancer induction in a specific tissue.That is, the radiations can differ in their *qualitative effect*. For example, there is a very substantial body of experimental evidence to indicate that the absorbed dose of high linear energy transfer (LET) radiation (α-particles) required to produce a given biological effect is less than that of low LET radiation (β-particles and γ-rays) – the relative biological effectiveness (RBE) phenomenon [e.g., Sinclair, 1985]. The relative biological effectiveness is defined as:

RBE = absorbed dose of 250 kev x-rays required to produce a given biological response/ absorbed dose of specified radiation required to produce the same effect.

For human radiological protection practice, this phenomenon is taken into account by applying dimensionless radiation weighting factors (w_r) to the absorbed doses from the different radiations, and summing, to give a quantity called the equivalent dose, where:

$$\text{equivalent dose, } H = \Sigma_r \, w_r \times D_r \quad (4)$$

The unit of the quantity, equivalent dose, remains the J kg^{-1} but it is given the special name Sievert (Sv). Although the LET of α-particles, β-particles, and the recoil electrons generated by x- and γ-rays are a function of their energy, and it would, therefore, be expected that the RBE (and the w_r) would also be continuous functions of energy, this degree of complexity has been largely avoided by defining the the w_r as set out in Table 1 [ICRP, 1991]. A continuous curve of w_r with neutron energy is given, however, for application in those situations where such precision can be justified.

The values of the radiation weighting factors for human application were chosen by the ICRP to be representative of the RBE values determined for the induction of stochastic effects (principally cancer, but to the extent that this response is initiated by somatic mutation, it might also apply to heritable mutations).The value of w_r is 1 for photons and β-particles of all energies and a value of 20 for α-particles (see Table 1). These values of w_r correlate reasonably closely with the mean LET along the corresponding charged particle tracks. In this manner, the equivalent doses to a tissue or organ from the different radiations may simply be summed to give a single measure for the total biologically effective radiation exposure.

It should be emphasised that these values of w_r

are defined for the purpose of human radiation protection; they cannot be applied without reservation to other organisms and biological endpoints (see [Sinclair, 1985]). In passing, it may be noted that the quantity *effective dose* (also with units of Sv) – the product of *equivalent dose* and tissue weighting factors, w_t, that represent the proportion of the total detriment arising from human whole body exposure that is contributed by the effects in the specified organ or tissue – is also not applicable to flora and fauna.

In terms of a quantity corresponding to *equivalent dose*, a number of approaches have already been used, and alternative proposals made, in the context of environmental protection. One existing assessment of the possible environmental impact of radioactive waste disposal has simply followed the human radiation protection practice and applied a w_r value of 20 to the α-component of the absorbed dose to obtain an estimate of the aggregate, biologically-effective dose (then termed the dose equivalent and given in Sv units) to wild organisms, although it was recognised that the procedure was open to argument [OECD-NEA, 1985]. More recently [e.g., UNSCEAR, 1996], and in explicit recognition of the inapplicability of the w_r values currently applied in human radiation protection practice, the low LET (β- and γ-radiations) and high LET (α-particles) components of the absorbed dose have been given separately.

Pentreath [1999] suggested DEFF (Dose Equivalent Flora and Fauna):

$$\mathrm{DEFF} = w_r \times D_r \qquad (5)$$

Although this quantity has the unit J kg^{-1} and could, in principle, take the special name Sv, it was additionally suggested that it be given the special name DEFF to avoid confusion with human radiation protection practice. (It should be noted that the allocation of special names for derived units in the International System of Units (SI) is at the discretion of the Bureau International des Poids et Mesures.). For the usual case of a mixture of radiation fields in a contaminated environment, the total, biologically effective, radiation exposure would then be given by:

$$\mathrm{DEFF} = w_r(\beta\gamma) \times D(\beta\gamma) + w_r(\alpha) \times D(\alpha) \quad (6)$$

A working group on “Quantities, units and compliance” at a recent meeting convened by the IAEA (Vienna, August – September 2000) suggested:

> (Radiation) Weighted Absorbed Dose (Rate) = Low LET Absorbed Dose (Rate) + w_r x High LET Absorbed Dose (Rate) (7)

where the w_r is the radiation weighting factor appropriate to the organism, effects endpoint and dose rate. Given the fluidity of this field at the present time, it was deemed inappropriate to suggest a special name to the unit for the quantity (but following the historical precedent of using acronyms (the rep and rem), the “wad” was proposed!) [IAEA, 2000].

Table 1. Radiation weighting factors.

Radiation type and energy range	Radiation weighting factors, wr
Photons of all energies	1
Electrons and muons of all energies	1
Neutrons, energy < 10 keV	5
10 keV to 100 keV	10
> 100 keV to 2 MeV	20
> 2 MeV to 20 MeV	10
> 20 MeV	5
Protons, other than recoil protons, energy > 2 MeV	5
Alpha particles, fission fragments, heavy nuclei	20

Other suggestions have since been made by other authors with respect for the need for new quantities and units [Kocher and Trabalka, 2000; Trivedi and Gentner, 2000]. In all of these approaches, the common thread is the retention of:

- the *absorbed dose* as the fundamental quantity to describe the interaction of the radiation field with the biological target. (This confirms, and provides additional support for, the conclusion reached at the end of the previous section. It might also be inferred that the retention of the special unit (Gy) for absorbed dose would be both sensible and acceptable.); and,
- the application of radiation weighting factors (w_r) to modify the absorbed dose and take account of the RBE phenomenon (including the influence of the absorbed dose rate and the biological effect of interest).

It is clear, therefore, that the assessment of the possible impact of incremental radiation exposure on the wild flora and fauna requires:

- methods for the estimation of the absorbed dose (rate), from both internal and external sources, to defined targets in a variety of reference examples of flora and fauna; and;
- values for w_r appropriate to the absorbed dose rates likely to be experienced in contaminated environments, and the radiation effects of interest in the selected types of reference organisms.

Some relevant dosimetric models already exist [e.g., IAEA, 1979; Amiro, 1997; Domotor et al. 2001] and others are in development in current programmes (e.g., the EU FASSET and EPIC projects).

As to the selection of values for the radiation weighting factor, w_r, this is less straightforward. Although there is a large literature on the differential effects of equal absorbed doses of different radiation types, it has not yet been reviewed and organised in a manner appropriate to its use in environmental radiation protection, i.e., by dose rate, species and biological effect. It may well be that a higher radiation weighting factor [Sinclair, 1985] would be appropriate for α-particles in an environmental context. This, however, presupposes that the organisms of interest, and the biological endpoints of concern, are known; although some progress is being made, this is not yet the case. It will require further research to develop a quantity corresponding to *equivalent dose* (derived from the *absorbed dose*) and the radiation weighting factors to be employed, for use in environmental impact assessments. It is becoming apparent that the endpoints that might be considered to be of significance in an environmental context could include: morbidity, mortality, reproductive capacity (encapsulating effects on fertility and fecundity), and cytogenetic damage [Pentreath, 1999; Woodhead, 2000; Pentreath and Woodhead, 2001]. These endpoints include, but are not restricted to, the cancer induction and heritable effects of major concern in human radiation protection.

The review of RBE data by Sinclair [1985] included consideration, however, of a wider variety of biological endpoints in addition to tumour induction, e.g., life shortening, chromosome aberration and mutation induction, and *in vitro* cell transformation. It also focussed on data from experiments in the low dose/dose rate domain where the responses to both the low and high LET radiations would be expected to be linear with dose and yield maximum values of RBE. Overall, the data (including some indicating that equal absorbed doses of α-radiation and neutron radiation are comparably effective) show a wide range of RBE values (~3 to ~200). Although it was found to be difficult to determine a statistically satisfactory average value from the data, it was concluded that it was probably in the range 30–50 for low dose/dose rate irradiation. For the cell-killing effects of low absorbed dose rate α-radiation in mouse ovaries and rat testes, the RBE values have been estimated to be up to 370, and in the range 10–15 (with a value of 23 for cytogenetic effects), respectively [Samuels, 1966; Searle *et al.*, 1976]. It is probable that care needs to be taken in the interpretation of these latter results as the estimation of the RBE is critically dependent on the assessment of the α-radiation dose actually received by the responding target in the cells (or cell nuclei).

At the present time, therefore, it seems reasonable to propose that a provisional w_r value of somewhere between 20 and 40 be applied in respect of the α-radiation absorbed dose rate to the tissues of wild organisms, with the recommendation that all the available data be reconsidered from an environmental protection viewpoint (this implicitly assumes that the $w_r(\beta,\gamma) = 1$). In this review, particular emphasis should be placed on the information available for species of organism of the same group that is represented by each of the reference (generic) organisms chosen to be representative of the different environments, the endpoints of relevance in an environmental context and low dose/dose rate exposure. Although, in principle, this review could lead to proposals for a number of differing $w_r(\alpha)$ for the variety of generic organisms and endpoints of interest, this level of sophistication may not be justified by the uncertainties both in the raw data, and in the assessments of the absorbed dose rates in the environment. If, however, multiple values of w_r were to be used, then any estimate of the aggregate, biologically effective dose (rate) would have to have an accompanying statement of its domain of applicability.

Canada has offered to take the lead in an IUR working group to review the available RBE data and to recommend the value(s) for w_r. While it would be inadvisable to pre-empt the overall outcome of this review, it may already be noted that it is unlikely that there is any published information concerning plants for any endpoint and that there will be data gaps relating to particular types of organism and/or endpoints (e.g., reproductive capacity in birds, etc.); in a majority of cases, the deficiency is likely to relate to information concerning the effects of high LET (i.e., α-particle) radiation exposure.

4.2. Absorbed dose (rate) – response relationships.

After an estimate has been made of the incremental radiation exposure in a contaminated environment, the next requirement is for the information allowing an assessment of the biological impact of the exposure, i.e., the absorbed dose (rate) – response relationship for the reference organism and effect endpoint of interest. All the available evidence is that the incremental absorbed dose rates arising in environments contaminated by authorised (controlled) disposals of waste radioactive materials will be low-level and chronic, i.e., generally less than ~ 100 μGy h^{-1}, but could, exceptionally, be up to several thousand (μGy h^{-1}, over significant fractions of the duration of the life-stage in the organism of interest, e.g., the period of embryonic development, but up to, and including, the whole lifetime [Woodhead, 1970, 1973a, 1973b, 1974, 1984, 1986; IAEA, 1976, 1988, 1992; OECD-NEA, 1985; Myers, 1989; NCRP, 1991; UNSCEAR, 1996; Bird et al., 2000]. It is in this dose rate range, therefore, that information is required on the effects of radiation for the purposes of environmental impact assessment. This requirement immediately raises a problem: this is the range of dose rates in which it has been found to be difficult to observe the many of the effects of radiation, even under the controlled conditions of laboratory studies that are, themselves, resource-intensive. The consequence is that directly relevant studies (in terms of the ranges of dose rates employed) are rather few. Recourse must be made to extrapolation from studies employing higher dose rates and, *in extremis*, from studies of the effects of acute exposures.

Thus far, the qualifiers "low-level", "chronic", "higher" and "acute" have been used without any definition, and it is pertinent to consider just what they might reasonably be taken to mean in an environmental context. A simple example may be used to illustrate the nature of the problem: a radiation exposure lasting several days may be effectively "chronic" for a short-lived organism such as an aphid (i.e., the exposure period might be of the same order as the development time of the parthenogenetic embryo), but effectively "acute" for a long-lived organism such as an oak tree. In practice, an exposure is usually termed "acute" if it is delivered in a time period that is shorter than that in which an acute deterministic effect (usually, but not necessarily, mortality) could appear **and** the dose rate is sufficiently high that a total dose likely to induce a deterministic effect could be accumulated in that time. Because deterministic effects show a sigmoid response curve with an effective threshold that may be as much as 2 .10^5 - 10^6 μGy, depending on the specific biological endpoint and the organism, this may be used to provide operational definitions of high doses and, if delivered within, for example, 24 hours, high dose rates (i.e., >(8 - 40).10^3 μGy h^{-1}).

From a consideration of microdosimetric factors, DNA repair processes, experimental radiobiology and epidemiological studies of tumour induction in the atom bomb survivors, it has been concluded [UNSCEAR, 1993] that low doses and low dose rates of low LET radiation (β-particles and γ-rays) are less than 2.10^5 μGy and 6.10^3 μGy h^{-1}, respectively. Below these levels, it is to be expected that the dose – response relationship for stochastic effects would be linear with dose, i.e., the effects of both the interactions between individual parti-

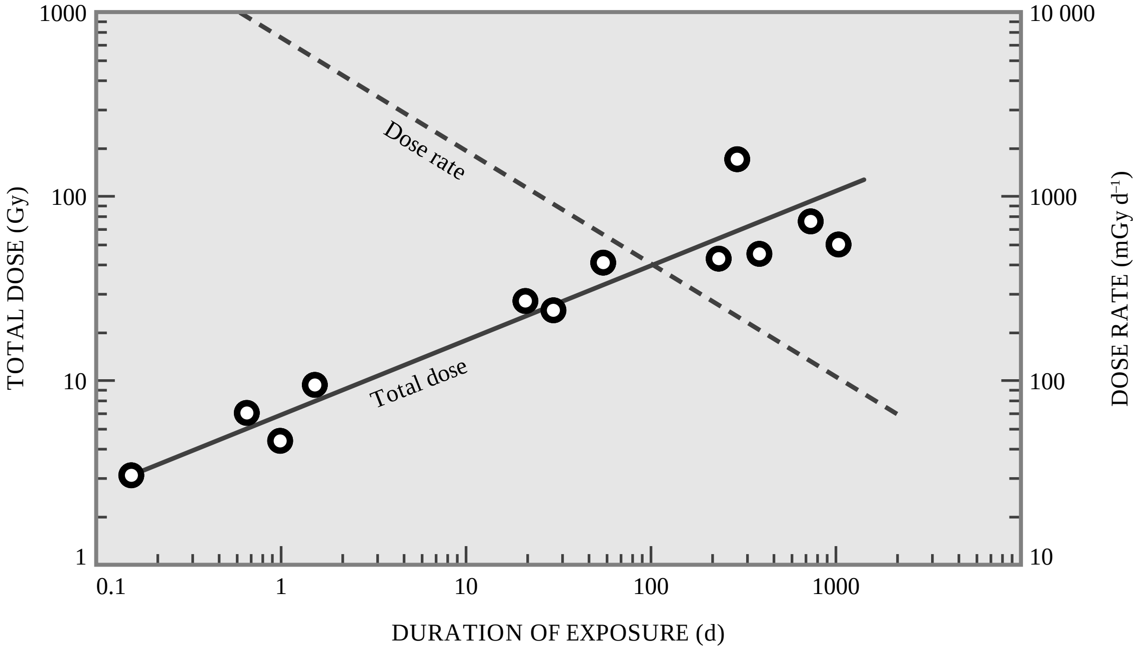

Figure 3. Relationship between total dose, dose rate and duration of continous radiation exposure for 50 % mortality in pines. (Data from IAEA 1992)

cle tracks – the D^2 component that tends to increase the response, and the induction of cell killing – a deterministic effect that tends to reduce the response – are minimal.

Two additional points may be made. First, even at “low” dose rates (i.e., <(8 - 40).10^3 µGy h^{-1}), a long-lived organism could accumulate a “high” dose if it were exposed continuously (i.e., “chronically”). Experimental studies show, however, that due to the intervention of repair processes, the onset of the deterministic effects is displaced to higher accumulated doses (see Figure 3 and examples in [Woodhead, 1998]). This factor, which will almost certainly depend on the specific biological endpoint and the organism, will make it difficult to extrapolate information concerning the appearance of deterministic effects from the “acute” to the “chronic” exposure domain. Second, in situations where “high” doses may be accumulated over extended life stages, the incidence of stochastic effects may plateau due to the induction of cell killing. Again, it is clear that extrapolation is unlikely to be a straightforward process.

The information presented in the previous two paragraphs is reasonably consistent and indicates that, for low LET radiations (β-particles and γ-rays), the boundaries between low and high (acute) absorbed doses and dose rates may be taken to be ~2.10^5 µGy and ~10^4 µGy h^{-1}, respectively. It must be emphasised that these are operational definitions, and simply indicate the dose and dose rate regions across which extrapolations should be made with both caution and due consideration of the biological effect and species being examined. In the low dose rate domain, i.e., below ~10^4 µGy h^{-1} (also the region of primary concern for environmental exposures), a “chronic” exposure is taken to mean that it is continuous over all, or at least, a significant fraction, of the life stage of interest. It should be also noted that in this domain the effects of the low LET radiation exposure depend on the dose rate **only** insofar as this determines the total accumulated dose.

It has been noted above that the endpoints that might be considered to be of significance in an environmental context could include: morbidity, mortality, reproductive capacity (encapsulating effects on fertility and fecundity), and cytogenetic damage [Woodhead, 2000; Pentreath and Woodhead, 2001]. In addition, some suggestions have been made for possible reference organisms in the European marine, freshwater and terrestrial environments [Woodhead, 2000]. Although it is likely that these lists of organisms are too long for practical application, they do provide an initial framework for organising the available information on the effects of radiation that might be applicable in an environmental context. This is the approach that has been adopted in the FASSET project, and it will undoubtedly identify significant gaps in the database that will require additional laboratory and/or field studies to fill. Finally, both the USDOE and the Canadian approaches make use of some form of generic reference "organisms" or entities for assessing compliance with predefined dose rate limits. As a general screening tool, the USDOE [2002] applies four generalised organism types (aquatic animals, riparian animals, terrestrial animals and terrestrial plants) in the derivation of limiting concentrations of radionuclides (Biota Concentration Guides) in soil, sediment and water [Higley at al., 2003 abc].

4.3. The natural background radiation regime.

There have been numerous assessments of the natural background radiation exposure of a variety of wild organisms (see [UNSCEAR, 1996] for a summary). A closer examination of the data shows that the assessments are probably partial in the sense that they do not appear to include all possible sources of internal and external exposure; this is particularly the case for the natural radionuclides taken up into the tissues of the organism, and even more so for the likely main contributor to the high LET component of the dose rate – ^{210}Po. These deficiencies arise mainly because the available data on tissue concentrations of the natural radionuclides in wild organisms are fairly limited (and many of the existing data were not obtained for the purpose of radiation dosimetry). This is particularly so for organs, such as the gonads, or the developing embryo, that are of significance from the viewpoint of possible radiation effects, and for the types of organisms that might be selected as reference flora and fauna within an impact assessment framework. It is important to identify these data gaps and take steps to fill them (some indication of the range and/or variation in the concentrations should also be obtained).

It may equally be noted that the detailed information on the distributions of the natural radionuclides in the external environments of the potential reference flora and fauna is less than complete on the scales that are relevant to the estimation of their radiation exposure. Again, this is a deficiency that needs to be addressed in the context of the use of the data for the express purpose of dose rate estimation.

The development of sophisticated dosimetry models will be wasted effort unless good radionuclide distribution data (i.e., down to the organ/tissue level, and/or on spatial scales of the same order as the ranges of the radiations in the relevant environmental media) are available as the primary input for dose rate estimation. Bioaccumulation phenomena are of paramount importance for improving the realism of dosimetric calculations, especially in chronic exposure situations.

4.4. Environmental pathways for radionuclide transport.

Radionuclides released into the environment interact with the ambient chemical, physical and biological processes according to their chemical and physical natures and are continuously redistributed on a range of spatial and temporal scales. In order to assess the possible impact of any consequent increase in the radiation exposure of the local wildlife, it is necessary to be able to quantify these radionuclide concentration distributions, using radionuclide transport models, to provide the input data for the dosimetry models. Many such transport models have been developed for the purpose of human radiation exposure assessment and, in broad terms, their output can provide an initial basis for identifying the regions of the environment where the native flora and fauna are also likely to receive enhanced exposure, i.e., a basis for the selection of reference organisms. As many of these organisms are unlikely to be of direct importance as a source of human radiation exposure, there is currently a lack of the specific transfer rate data to model the accumulation of the radionuclides into both their tissues and their local environment. Some data undoubtedly exist in the literature and need to be assembled into an accessible database; but many will be non-existent and require further research to develop the relevant information. This applies not only to the "artificial" radionuclides generated by human activities, but also to the "natural" radionuclides that can become concentrated in the wastes from certain industries, e.g., the processing of phosphate ores. It is important to emphasise that any new research should be initially tightly focussed on the specific requirements of estimating the radiation exposure of reference fauna and flora selected for inclusion in any environmental protection framework.

4.5 Genotoxicological techniques

The use of new cytogenetic techniques, or the adaptation of human cytogenetic techniques to non-human studies, offers the possibility for quantifying the effect of radiation on DNA at levels below that which cause obvious mortality or reduction in reproductive success, but which may cause chronic genetic effects in the individual or population. The use of such techniques in environmental research is already established with regard to some other genotoxins, particularly in the study of PAHs in aquatic biota [Kleinjans and van-Schooten, 2002], and may permit the comparison of the impact of environmental levels of radioactivity and other genotoxins, and in the determination of additive, synergistic or even possible antagonistic effects of radiation and other pollutants.

The available methods can be divided into two groups – those which involve DNA dosimetry to analyse the dose of the genotoxic agent at the target site (biological monitoring), and those which analyse early genotoxic responses (biological effect monitoring). Biomarkers include chemical DNA modifications such as DNA adducts and markers of cytogenetic effects such as chromosomal aberrations and micronuclei, and mutations. In studying genotoxicity in exposed species, the aim is not particularly to assess morbidity risk to the individual. Rather, by determining effects in somatic and germinal cells, predictions can be made regarding the impact of the mutagen on biodiversity and survival chance [Bickham and Smolen, 1994]. However, the potential of the techniques in environmental radiochemistry is only just being investigated, and to date there are very few published data. A brief summary of the methods and some applications is given below.

<u>Alkaline Comet Assay</u> measures DNA strand breaks by a method known as single cell gel electrophoresis (SCGE) [Singh et al. 1988], and has been used frequently in studies of the effect of heavy metals and organic pollutants on biota, usually involving aquatic organisms [Mitchelmore and Chipman, 1998; Mitchelmore et al., 1998; Large et al., 2002]. Some studies have also been made on the impact of radioactivity on humans, particularly in Chernobyl accident victims [Frenzilli et al., 1998]; a few have looked at the effects of radioactivity in biota [Sugg et al., 1996; Hingston et al., 2002].

<u>Micronuclei</u> have been used for many years as markers of biological response in cellular systems. They are formed when fragments (parts of broken chromosomes) are lost from the main nucleus of a cell during division. Interference of the DNA by radionuclides or other genotoxins increases the rate of micronuclei production. The micronucleus assay is a relatively simple method – cells cultures or samples from living organisms are fixed, stained, examined under a fluorescence microscope, and the number of micronuclei in the binuclear (dividing) cells counted.

<u>Fluorescence in situ hybridisation (FISH).</u>
Radiation can induce two types of chromosome exchange in cells, asymmetrical and symmetrical. Asymmetrical exchanges are non-viable and lethal to the cell, which is generally not a problem for the organism or the population. Symmetrical exchanges are viable and create stable, but altered, chromosomes. The effects of symmetrical exchanges in an individual are therefore cumulative. Asymmetrical exchanges such as are easily detected and quantified by conventional solid staining techniques [Natarajan, 2002]. Symmetrical exchanges only became easily detectable with the development of fluorescence in situ hybridisation (FISH) in the 1980s [Pinkel et al., 1986], whereby selected chromosomes are labelled with a fluorescent probe. The technique has been extensively in human genetic research, and in the study of the biodosimetry of human victims of the Chernobyl accident. Recently Ulsh et al., [2002] applied the technique in a study involving the yellow-bellied slider turtle (Trachemys scripta).

<u>Mini- and micro-satellites</u> are repetitive noncoding sequences in DNA, which contain base-pair patterns so different from that of the coding (protein producing) genetic material that they can be identified with special DNA probes. They are situated across the whole genome, often comprising a large proportion of it (approximately 10 % in humans, 50 % in insects). They can therefore be used, in combination with PCR techniques, as markers indicating the transfer of genetic material due to chromosome aberrations. Microsatellite markers have been used in investigation on burrowing mammals exposed to solid low-level nuclear waste [Stormberg et al., 2002] and to investigate observed phenotypic changes in barn swallows (Hirundo rustica) close to Chernobyl [Ellergren et al. 1997]

5. Toxicology and Basic Ecological Sciences

At variance of previous emphasis that was placed on high doses in acute exposure, the domain of low doses implies that radiations do not constitute anymore a dominant source of stress that can be treated separately, but only one component of a multipollutant-promoted stress that needs to be tackled as a whole [Bréchignac, 2002]. The protection of the environment from stress promoted by chemical pollutants has been an important area of research and developments worldwide yielding the field of ecotoxicology (or environmental toxicology). This has formed the scientific foundation of the current approach to Ecological Risk Assessment (ERA) for non-radioactive pollutants that obviously needs to remain compatible with any future system for environmental radioprotection. From the considerable experience accumulated on using current ERA methodologies for the sake of protecting the environment, several shortcomings have been identified which are now promoting attempts to better incorporate also more basic ecological knowledge. For environmental radioprotection to be successful, a prerequisite is to evolve in good understanding, harmonisation and compliance with the fast growing evolutions underway in dealing with non-radioactive pollutants.

5.1. The current approach to ERA

The ERA methodology has developed based on what had already been elaborated for human health. However, if the general principles relating to human health are quite recognised, the development of their application to the environment still gives rise to numerous controversies. These result mainly from the multiplicity of organisms, the diversity of their sensitivity to toxins, and their various pathways of exposure. Faced with the difficulty of obtaining sufficient information concerning the dose-effect relationship for this multitude, the shortcut consists most often in selecting species that are considered to be representative of the various taxonomic groups and in using them as surrogates to the global system.

Numerous international organisations take part in the development of the ERA methodology, which is in full growth (US EPA, OECD, SETAC). The methodology has been standardised according to three successive steps: 1) problem formulation (identification of source term, stress agents, potential effects, endpoints, receiving ecosystem, etc…), 2) analysis phase that is carried out in parallel on two fronts, exposures and effects, 3) risk characterisation (comparison with pre-set limits in order to rate the risk associated with the situation under consideration). For the key analysis phase, the approach that is currently the most widespread rests on the methods of ecotoxicology for practical reasons. It first analyses the effect of the stress agent on individuals, by means of laboratory standardised toxicity tests, sometimes completed by microcosm or mesocosm verifications, then extrapolates it to upper organisation levels (populations, ecosystems).

Whilst practical, this ascendant "bottom-up" approach holds inherent problems that give rise to a large amount of literature and to numerous discussions. One can briefly mention the difficulties that are inherent to: 1) the establishment of the effects on a population scale [Hoffman et al., 1995] and the incorporation of processes that are inherent to the ecosystem scale (predation, competition, resilience, etc…), 2) the identification of indicator species [Power and McCarty, 1997] and corresponding effect endpoints that are appropriate (mortality, morbidity, sterility, fecundity, reproduction, physiology, genetic damage, etc…), 3) the taking into consideration of speciation-bioavailability relationships [Cura, 1998]. Altogether, these have prompted the further consideration of a “top-down” ecosystem approach.

5.2. Ecological sciences

The recent ethical and sociological developments concerning the protection of the environment have emphasised the integrity of the ecosystem. Thus, the goal consists in developing a set of conservative measures intended to preserve the integrity of the ecosystems as being the centre of processes that support life (of which the human being is part) and/or as providing a certain number of services to humankind [Costanza et al., 1997; Costanza, 2000; Odum and Odum, 2000; Heal, 2000]. This widening of the context, imposed in particular by the appearance of environmental concerns of a global nature, has given rise to the development of another approach centered on the ecosystem. This new approach, seeking to overcome the problems that are inherent to the reductionist approach centered on the individual, attempts to integrate the structure and functioning of the ecosystem in the effect endpoints that it considers – stability, biodiversity, resilience, effectiveness of nutrients' recycling, primary productivity, energy use efficiency – so as to settle the pitfall of extrapolation on the ecosystem scale.

This approach is less often implemented than the reductionist approach, but a growing number of authors nevertheless advise that it not be neglected in spite of the difficulties that it presents. These difficulties lie mainly in our yet-to-be-completed understanding of complex interactions occurring in ecosystems, but this is a field in full growth sustained by the theory of complex systems that currently provides a very promising clarification on the dynamic comprehension of ecosystems' integrity [Kay, 2000].

Moreover, it is important to note that this evolution is already under way in the neighbouring fields of risk assessment. The SETAC in particular has formed a work group in the field of "Life Cycle Impact Assessment" (LCIA) [Udo de Haes, 1999] which indicates that the "top-down" perspective has not been taken into account in a sufficient manner in the past [Hofstetter, 1999]. One of the main reasons put forward is that significant omissions in assessment cannot be detected without a "top-down" analysis.

It is evident that, as a result of initiatives such as the legislation now appearing in Europe arising from UN and other Resolutions on Biodiversity, plus multi-national agreements in relation to habitat protection (e.g. the 3 EC Directives) and numeric approaches to measuring directly the biological/ecological "health" of the natural environment are soon to emerge. At present, however, it is not practical to assume what these are likely to be and the IUR Task Group therefore considers it essential to liase with, and keep under review, these new areas of research and environmental management.

6. Conclusions

- The IUR is has played a major role in promoting, and helping to develop, a systematic approach and framework for environmental protection.
- Given the current scientific knowledge, a first approach to developing systems and standards for protection of the environment from ionising radiation is now also being taken forward by the ICRP and the IAEA.
- By virtue of its broad scientific expertise, encompassing radiobiological, ecological and risk assessment fields, the IUR shall be at the forefront to identify the research needs, promote the most appropriate future directions and coordinate the corresponding scientific actions.
- In particular, the IUR is well placed to help developing and implementing this new approach, particularly in view of the need to provide data on knowledge gaps and aspects of those sciences which have not previously been considered by UNSCEAR, ICRP or the IAEA because of the sole focus on human protection.
- Furthermore, the IUR can call on a global membership, with branches in Asia, Europe, America, and Australia (an African connection is currently under development). An international network for co-operation and co-ordination of experimental work has been set up and a worldwide database covering ongoing research of relevance to environmental protection is under construction.
- In addition, the IUR is well placed to provide an interface with other relevant scientific disciplines, ranging from the social and economic sciences to other conventional environmental sciences which are supporting the development of the overall environment protection philosophy and application.

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