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FOREWORD

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Under the terms of the Treaty establishing the European Atomic Energy Community, the Community, amongst other things, establishes uniform safety standards to protect the health of workers and of the general public against the dangers arising from ionizing radiation. The standards are approved by the Council, on a proposal from the Commission, established taking into account the opinion of the Group of Experts referred to in Article 31 of the Treaty. The most recent version of such standards is contained in Council Directive 96/29/Euratom of 13 May 1996 laying down basic safety standards for the protection of the health of workers and the general public against the dangers arising from ionizing radiation.

The European Commission organises every year, in cooperation with the Group of Experts referred to in Article 31 of the Euratom Treaty, a Scientific Seminar on emerging issues in Radiation Protection – generally addressing new research findings with potential policy and/or regulatory implications. Leading scientists are invited to present the status of scientific knowledge in the selected topic. Based on the outcome of the Scientific Seminar, the Group of Experts referred to in Article 31 of the Euratom Treaty may recommend research, regulatory or legislative initiatives. The European Commission takes into account the conclusions of the Experts when setting up its radiation protection programme. The Experts' conclusions are valuable input to the process of reviewing and potentially revising European radiation protection legislation.

In 2012, the Scientific Seminar covered the issue *Protection of the Environment*. Internationally renowned scientists working in this field discussed the "why and how" protecting the environment against the dangers arising from exposure to ionising radiation. They presented current knowledge on the ecological impact of ionising radiation, protection of the environment in normal situations, the effects on non-human species in areas affected by a radiation accident, stakes and limits of bioremediation, and ethical aspects of protection of the environment. The presentations were followed by a round table discussion, in which the speakers and invited additional experts discussed potential *policy implications and research needs*.

Presentations and discussions at this scientific seminar were particularly important for the development of regulatory requirements on protection of the environment, such as those currently foreseen in the new Euratom basic safety standards Directive.

The Group of Experts discussed this information and drew conclusions that are relevant for consideration by the European Commission and other international bodies.

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1 PROTECTION OF THE ENVIRONMENT IN THE CONTEXT OF RADIOACTIVE RELEASES

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1.1 Introduction

Although virtually everyone would agree on the need to protect the environment in a general sense, the term is not easy to describe, and it means different things, in different contexts, to different people. A symposium on the subject of protection of the natural environment in relation to radiation was held in Stockholm as long ago as 1996 (Amiro et al, 1996), from which a number of further initiatives subsequently flowed. An important question was what environmental protection meant in the context of radiation protection, and how it related to the statements made at the time by the ICRP in its Publication 60 (ICRP, 1991). A fundamental task was the development of an underlying ethical basis, and this was taken up by the IAEA who drew together the ethical views of relevance (anthropocentric, biocentric, and ecocentric), plus the principles embodied in United Nations legislation relating to sustainable development, conservation of the natural world, and the need to maintain biological diversity (IAEA, 1999, 2002). A more pragmatic reason for addressing the issue of protection of the environment in relation to radiation, however, has been the increasing need for all major industries to comply with both international and national obligations relating to environmental protection.

1.2 The need to demonstrate protection of the environment in the context of ionising radiation

With regard to the nuclear industries, such requirements have been slow to evolve. The Joint Convention on the Safety of Spent Fuel Management and on the Safety of Radioactive Waste Management (IAEA, 1997) initially made reference to the environment in relation to its general safety provisions, and to the safety of the management of spent fuel and radioactive wastes. The Fundamental Safety Principles (IAEA SF-1, 2006) also established, among others, principles for ensuring the protection of the public and the environment, now and in the future, from harmful effects of ionising radiation. These principles apply to all situations involving exposure to, or the potential exposure to, ionising radiation. They state that the general intent of the measures taken is to protect ecosystems against radiation exposure that would have adverse consequences for the populations of a species. The need for consideration of environmental protection criteria and methodology for the assessment of radiological impact, taking into account explicitly the protection of flora and fauna when deemed necessary by the national authorities, is also now included in the revised Basic Safety Standards, or GSR Part 3 (IAEA, 2011).

There are also a number of European Council Directives that relate in some detail to environmental protection. Examples are the Directive on the Impact of Certain Projects on the Environment (EC, 1985); the Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (EC, 1992); the Directive on Integrated Pollution Prevention and Control (EC, 1996); the Water Framework Directive (EC, 2000); and the more recent Marine

Strategy Directive (EC 2008) which specifically mentions the introduction of radionuclides under ‘hazardous substances’.

It is therefore evident that future developments involving the actual or potential release of radionuclides into the environment will come under increasing scrutiny with regard to their actual or potential impact on the environment, with respect to all exposure situations, including those involving accidents and unforeseen events. For the vast majority of industries, environmental quality standards of one form or another have been drawn up, in the form of concentrations of chemicals in various environmental materials, based on the protection of humans via various exposure pathways, and protection of the environment itself. But these are lacking for radionuclides, which results in the nuclear industries being somewhat at a disadvantage compared with others in being able to demonstrate explicitly what the actual or potential impact on the environment is, or is not, under different operational circumstances.

1.3 Creating a practical framework

The International Commission on Radiological Protection (ICRP), following on from an earlier review of the subject (ICRP, 2003), in its 2007 Recommendations considered it appropriate to broaden its scope in order to address directly the subject of protection of the environment, and therefore included in its general aims those of: “... preventing or reducing the frequency of deleterious radiation effects in the environment to a level where they would have a negligible impact on the maintenance of biological diversity, the conservation of species, or the health and status of natural habitats, communities, and ecosystems” (ICRP, 2007). The ICRP did point out, however, that it believed that its approach to environmental protection should be both commensurate with the overall level of risk, and that it should be compatible with other approaches being made to protect the environment, particularly from those risks arising from similar human activities.

The current systematic approach to human radiological protection has evolved over many years, based on a range of knowledge on the effects of radiation on humans, supplemented by other data from studies on animals. The ICRP attempts to convert these data, together with their errors, uncertainties, and knowledge gaps, into pragmatic advice that will be of value in managing different exposure situations. The advantage of such a comprehensive and systematic approach is that, as the needs for change to any one component of the system arise (as in the acquisition of new scientific data, or changes in societal attitudes, or simply from experienced gained in its practical application) it is then possible to consider what the consequences of such a change may be elsewhere within the system, and upon the system as a whole. Such a system would not work unless it was based on a numerical framework that contained some key points of reference, particularly with respect to how best to relate exposure to dose, dose to the risks of radiation effects, and the consequences of such effects. A key step in developing this scientific framework was the creation of an entity previously known as Reference Man (which has since evolved into Reference Male, Reference Female, and Reference Person) which served as a conceptual and analytical tool for many of the ICRP’s numeric analyses and resulting conclusions (ICRP, 2007). It was therefore proposed (Pentreath, 1999, 2002, 2005) that this systematic approach be extended to include a small set of animals and plants to serve as the basis for producing and analysing numerical data in order to provide advice with regard to protection of the environment. One advantage of such an approach, it was argued, was the degree of commonality that could emerge, particularly with regard to the banding of dose-effects data as a basis for considering what to do under different exposure situations, in the context of the overall principles of radiation protection.

An important consideration was the level of generalisation to adopt, and thus how best to describe the chosen selected biota, bearing in mind that it had not been the intention to select particular species, but equally not to generalize to the extent that the characteristics of the selected biological types were of little biological meaning. The most useful biological classification level in order to meet these objectives was that of Family. Thus a Reference Animal or Plant (RAP) is a hypothetical entity with the assumed basic biological characteristics of a particular type of animal or plant, as described to the generality of the taxonomic level of Family (with defined anatomical, physiological, and life-history properties) that can be used for the purposes of relating exposure to dose, and dose to effects, for that type of living organism.

In selecting a small but practical set of RAPs, the following points were considered: that there was a reasonable amount of radiobiological information already available on them, including data on probable radiation effects; that they were amenable to future research, in order to obtain the necessary missing or imprecise data, particularly with regard to radiation effects; that they were considered to be typical representative fauna or flora of particular ecosystems and had a wide geographic variation; that they were likely to be exposed to radiation from a range of radionuclides in a given situation, both as a result of bioaccumulation and the nature of their surroundings, and because of their overall lifespan, lifecycle and general biology; that their life-cycles were likely to be of some relevance for evaluating total dose or dose-rate, and of producing different types of dose-effect responses; that their exposure to radiation could be modelled using relatively simple geometries; that there was a reasonable chance of being able to identify any effects at the level of the individual organism that could be related to radiation exposure (bacteria and unicellular organisms were excluded because of their high resistance to radiation); and that they had some form of public or political resonance, so that both decision makers and the general public at large were likely to know what these organisms actually were, in common language.

A set of RAPs was therefore identified by the ICRP (ICRP, 2008), but there is nothing sacrosanct about the set; other biotic types could have been chosen. They were all considered to be organisms that are ‘typical’ of different environments, in the sense that one might expect to find them there: earthworms in soil; ducks in estuaries; flatfish, crabs and brown seaweed in coastal waters; trout in rivers and lakes; frogs in marshland; deer, pine trees, wild grass and bees across much of the temperate part of the globe; and small mammals such as the rat being virtually ubiquitous. The set is also essentially one of ‘wild’ animals and plants rather than domesticated ones, although many of them are ‘farmed’ in some countries in one way or another. With regard to typical farm animals - primarily large mammals that live essentially in a human environment - it was considered that the use of the human animal itself was probably sufficient for such managed environmental or ecological situations.

The RAPs can therefore be regarded as playing a similar, but much simpler, role to that of the ‘Reference Man family’ in order to derive some form of numerical guidance to aid management decision making with regard to different exposure situations. Basic simple dosimetric values relating to internal and external exposure were also compiled for each type (ICRP, 2008) and a set of reference Concentration Ratios subsequently compiled to help in numerical modelling of different exposure situations (ICRP, 2009).

1.4 Relevant biological end points and dose rates

For the protection of human beings, under different exposure situations, the objectives are clear and apply to individuals, or to small groups of individuals, rather than to the population as a whole. But for environmental protection, the biological endpoints of most relevance are those that could lead to changes in population size or structure, regardless of whether they are deterministic or stochastic.Among these endpoints are early mortality (leading to changes in age distribution, death rate, and population density); some forms of morbidity (that could reduce “fitness” of the individuals, making it more difficult for them to survive in a wild environment); impairment of reproductive capacity (affecting birth rate, age distribution, number and density); and the induction of chromosomal damage. And there cannot be any effect at the population level if no effects occur in any of the individuals of that population. (But the inverse is not always the case, because detectable effects in some members of a population would not necessarily have a consequence for the population as a whole.)

Data on the effects of radiation on animals and plants at dose rates relevant to most environmental exposures (and often in relation to any level of exposure) are relatively few, and there is no equivalent of the LNT model to allow extrapolation from effects at high doses, and high dose rates, to lower ones. In general, for the higher vertebrates, there is little difference in response across a range of dose rates for mammals, and this may well also apply to birds (because they are also ‘warm blooded’ vertebrates with high metabolic rates), but there are insufficient data to draw firm conclusions. For the lower vertebrates, generalizations are again difficult to make because their lower metabolic rates – that are also temperature dependent – are seldom taken into account in studies on radiation effects. The inference is, however, that if allowances for such differences were made (essentially by allowing more time for the effects to appear, and not by drawing comparisons over such short time periods as 30 days, which are only relevant for mammals) then the differences between higher and lower vertebrates may be less than it appears to be.

In order to use existing data bases on the effects of radiation, therefore, the only pragmatic approach seemed to be to consider the existing data bases in terms of bands of dose within which certain effects have been noted, or might be expected, and then to select a band to serve as what is termed a Derived Consideration Reference Level (DCRL) - a ‘reference level’ for effects, transparently derived, that can be considered as the starting point for decision making, depending on the purpose for making decisions under a defined exposure situation. Tables were therefore constructed to cover dose rate ranges, in bands, from <0.1 mGy day^{-1} to >100 mGy day^{-1} (ICRP, 2008). (Dose rates > 1Gy day^{-1} are essentially of no environmental relevance.) Bands of DCRLs for the RAPs are shown in Figure 1.

One aspect that also needed to be considered was the question of whether or not it would be sensible to combine one or more of the values in order to simplify the data. This was not considered to be appropriate for three reasons. First, it would involve mixing up information on those effects that have been looked for but not observed, with those effects that have not been looked for at all, and thus it is not known if they are observable or not. Secondly, for some applications it is better (or necessary) to be able to refer to the specific type of organism that is directly relevant, rather than to a mixture of organisms. And thirdly, the impression should not be given that any particular Reference Animal or Plant type is intended to serve as a ‘sentinel’ type for any of the others.

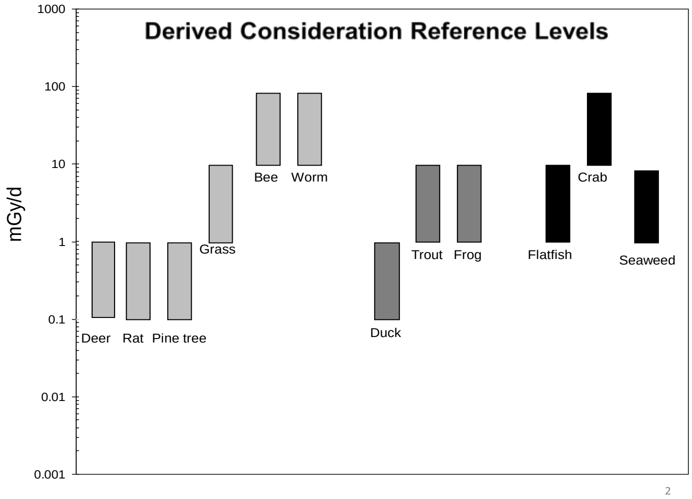

Figure 1: Derived Consideration Reference Levels for Reference Animals and Plants (from Pentreath, 2012)

1.5 Application to different exposure situations

All biota are exposed to ionising radiation from natural sources and many are, or may be, exposed to man-made sources. The processes causing the latter can be conveniently thought of as a network of events and situations, starting from a source. Radiation or radioactive material passes through environmental pathways leading to the exposure of biota that can be expressed in terms of dose. Protection can then be achieved by taking action at the source, or at points in the exposure pathways, and occasionally by modifying the location or characteristics of the exposed biota. The available points of action therefore have a substantial effect on the system of protection. The following types of exposure situations are relevant. Planned exposure situations, resulting from the operation of deliberately introduced sources, which may give rise both to exposures that are anticipated to occur (normal exposures) and to exposures that are not anticipated to occur. These include the discharge and disposal of radioactive waste, decommissioning of installations, and the activities related to eventual remediation and decontamination work of resulting contaminated sites. Emergency exposure situations are those resulting from a loss of control of a planned source, or from an accident or unexpected situation (such as a malevolent act), which requires urgent action in order to avoid or reduce undesirable consequences. And finally, existing exposure situations, which are those resulting from sources that already exist when a decision to control them has to be taken, including prolonged exposure situations after emergencies.

With regard to the use of RAPs in these different exposure situations, because they are, by definition, points of reference, it may also be necessary to identify Representative Organisms (ROs) relevant to a specific exposure situation. (In some cases there will be little choice in selecting them, because they may already have been selected by way of other existing legislation such as that for conservation or habitat protection.) When selecting ROs, differences between such biota and the RAPs should be quantifiable, in relation to their basic biology, dosimetry, or radiation effects, and such differences need to be noted and taken into account. The extent to which such factors then need to be applied, and their relevant impact on the final management decision, will depend on the nature of the implementation and application of the planning process relevant to protection of the environment. Because other regulatory bodies are likely to be involved, such as those responsible for wildlife management, it is essential to have a clearly set out logical link between any radioactive releases and any potential risk of biological effects (for which the RAP framework should be a starting point) and a clearly laid out strategy by which the relevant stakeholders can be engaged in the decision making process.

Taking the RAPs as a starting point, therefore, under those circumstances where there is, or may be, an environmental exposure of significance above the natural background locally experienced by the relevant biota, the DCRLs could be used in each exposure situation as follows. In planned exposure situations, the lower boundary of the relevant DCRL band should be used as the appropriate reference point for protection of different types of biota (Figure 2) within a given area during the planning of controls to be applied to a source.

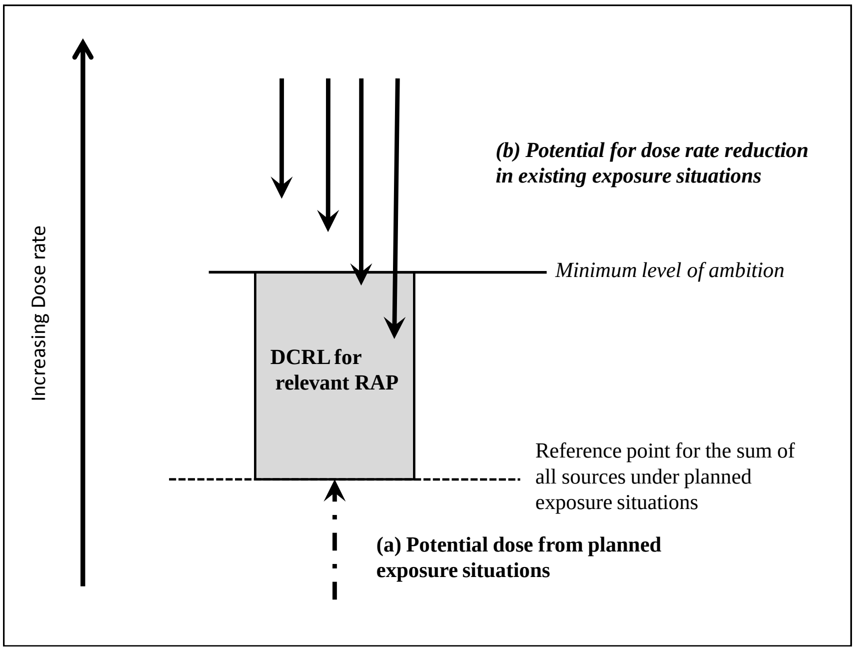

Figure 2: Relationship between DCRLs and (a) the potential dose from planned exposure situations and (b) the potential to reduce exposures in existing exposure situations

And for *existing exposure situations*, and in *emergency exposure situations* where control of the source has still not been obtained, if the dose rates are above the relevant DCRL band, then the level of ambition would be to reduce exposures to levels that are within the DCRL bands for the relevant populations, fully considering the radiological and non-radiological consequences of so doing (Figure 2).

Because the DCRL bands apply to animals and plants within a given location, however, the extent of such an area needs to be determined in advance relative to the overall conservation objectives. And in the case of multiple sources of exposure - for example from historical discharges or multiple sites - these other sources should be taken into account in comparison with the DCRLs when assessing protection options.

Planned exposures situations relating to the management of long-lived wastes are especially difficult with regard to protection of the environment, because over the long time frames that are considered, the biosphere is likely to change, and may even change substantially. Such changes may entail alterations that are natural, enhanced, or perturbed, through human action. The default case for protection, and protective actions, should therefore be the set of RAPs, bearing in mind that this set was deliberately chosen because its components are considered to be ‘typical’ biotic types of the major environmental domains of land, sea, and fresh water.

With regard to responding to an actual emergency event, or accidental release of radionuclides into the environment, consideration of environmental protection may not be an immediate priority, depending on the extent to which human exposures, or human food chains, are likely to be affected. But even where human exposure concerns predominate, consideration should nevertheless be given to the environmental consequences of the possible options available to achieve the adequate level of human protection.

1.6 Discussion and conclusions

There are still several issues to be resolved. One is the fact that different types of radiation are known to produce different degrees of effect in the same biological tissue, for the same absorbed dose, for many types of organisms.In the case of human radiological protection, it has been found useful to use other quantities to describe more accurately the expected relationship between dose and effect. Thus the equivalent dose makes use of a set of radiation weighting factors, chosen by the ICRP largely on the basis of the known relative biological effectiveness (RBE) of different types of radiation. No attempt has so far been made to allow for radiation weighting factors in RAPs, even though RBE is reasonably well studied in some small mammals; indeed, much of our knowledge of these effects comes from such studies and their relevance to environmental protection needs further consideration (Higley et al 2012). From a research needs point of view, there is also an urgent requirement for much more information on the effects of radiation on these types of biota, particularly at the egg and juvenile stage and the consequences for reproductive success.

One question often asked is: how can such a small group of RAPs be used to protect the ecosystem as a whole? Simply using twelve types of RAPs is, however, essentially a pragmatic one. It does not imply a lack of concern for other biotic types, nor for the environment as a whole (one could not ‘protect’ any of these biotic types without also conserving the habitats within which they live.) But it is impossible to know about all of the species within any ecosystem, and the ‘health’ of such areas is often assessed by studying a sub-set of its components.

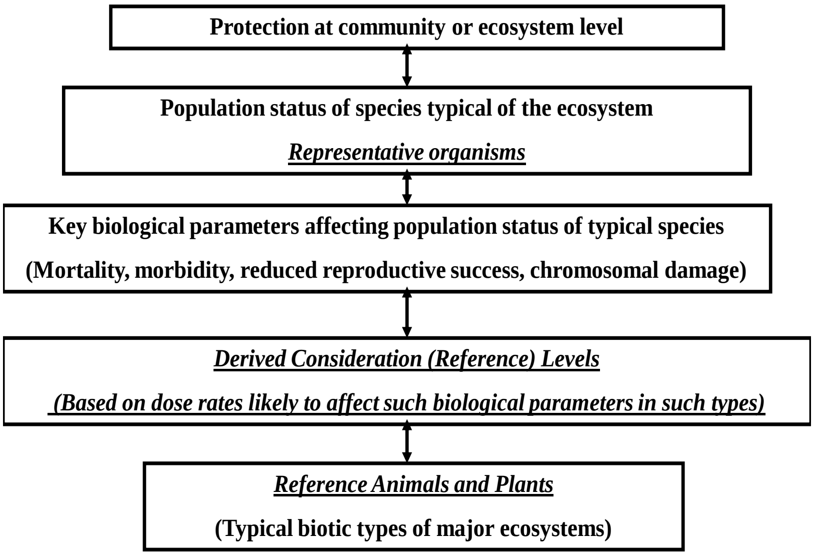

Figure 3: Relationships between the aims of protecting a community or ecosystem and the use of Representative Organisms and RAPs.

Thus it is common practice in ecological management that, in order to assess the status of a particular area, or ecosystem type, studies are made of population structures and numbers of those species that are regarded as typical members of it. If these are changing beyond an expected or desired range, then further studies are made to examine the underlying causes. These, naturally, usually relate either to physical or chemical changes to the habitat, or to biological factors that could affect the population, such as changes in food supply or to predator/prey relationships, the consequences of which are likely to lead to early mortality, reduced reproductive success and so on. Thus the RAP approach, with its choice of organisms being based on their ‘typical’ representativeness, and their set of biological effects end points, is a deliberate attempt to interface in a pragmatic way with current environmental management practice, as set out in Figure 3.A key point to note, however, is that because almost all of the information on radiation effects arises from data on small groups of individuals, if the objective is that of protecting an actual population, it will also be necessary to assess the fraction of the population of interest that is exposed to such levels of dose. This will vary from case to case.

Thus the RAP approach should inform, and provide an input to, the broader aims of environmental management, often referred to as the ’ecosystem approach’. It would be a mistake to try and ’internalise’ all of the other factors that affect ecosystems into a system designed to manage those effects relating solely to radiation. That is not to say that one should not be unaware of them. But, as is the case with human radiological protection, it is recognised that ’workers’ experience, simultaneously, many other hazards in the course of their work, possibly including exposure to other carcinogens. Similarly, the general public is exposed to many carcinogenic agents on a daily basis, as well as possibly being exposed to planned exposure situations involving radionuclides. Allowance for the totality of such exposures rests with the managers of the work force and guardians of public health and safety. And the same applies to the environment. Hopefully further knowledge and application of this approach will lead to a greater understanding of the risks (or lack of them) arising from practices relating to the nuclear industry, and place them in a better and transparent perspective with the risks and hazards relating to other human activities relating to environmental protection.

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2 ECOLOGICAL IMPACT OF IONIZING RADIATION: AN ENDPOINT ISSUE?

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2.1 Introduction

There is a wide consensus today recognizing that human health is pledged to the health status of the environment. This is why environmental protection has become an important issue that radiological protection needs to consider as well. Indeed, current civil nuclear activities are prompting fears, questions and significant efforts of regulation to prevent the occurrence of harm which would be unacceptable to society. Concern exists with respect to the effectiveness of mastering radiological risk in a robust and transparent manner. This led the International Commission of Radiological Protection (ICRP), which had for a long time subordinated environment protection to the protection of human beings, to reconsider its original paradigm and to initiate the construction of a specific system framework for the radiological protection of the environment (actually non-human biota) against ionizing radiation (ICRP, 2003).

Today, the approach towards radiological protection of the environment most considered by various national and international bodies is focused on a concept of "reference organisms" (IUR, 2002; ICRP, 2008; Larsson, 2004). Evolving from traditional toxicology, this bottom-up approach is emphasizing individual organisms for several immediate considerations: 1) the driver to operational application which leads to favour a straightforward practical approach for rapid and easy use, 2) ensuring consistency with the existing system for human radiological protection (also focused on the individual organism), and 3) the recognition that the scientific literature to date on radiation dose-effect relationships has largely considered animals and plants at individual organism level (UNSCEAR, 1996; Real *et al*., 2004). This has the merit of optimising the exploitation of the largest basic knowledge existing on the biological effects of radiation on life and ensuring consistency of protection approaches between man and other plant and animal species.

However, there is inherent reductionism with this approach which deserves critical attention because information on individual organisms only partially covers potential environmental effects, especially system level effects which have been repeatedly reported from observations in contaminated territories (IUR, 2012). Such shortcomings have already been recognized and discussed in other fields of environmental protection (Tannenbaum, 2005), and have also been stressed in the area of radiological protection (Bréchignac, 2003; Hinton *et al*., 2005; Fuma *et al*., 2003; Doi *et al*., 2005; Bréchignac and Doi, 2009; IUR, 2012).

2.2 Origin of environment protection: a move from anthropocentrism to biocentrism

Initially, environmental protection was concerned with human health, and the major driver during long periods had been protection of human life per se, without any major need to consider the environment in a broad sense. Much more recently, environmental protection evolved during the 20th century as an issue due to the planetary exponential growth of the human population. This population growth, initiated during the 19th century, prompted an associated growth of industrialisation linked to exploitation of natural resources which has proven to impact on the environment. The goods to humankind provided by these developments, in terms of economic development, have been recognised to also lead to potential deleterious side effects requiring consideration in view of ensuring "sustainability" of the processes concerned. Interestingly, concerns about the environment have not primarily evolved from the harm observed on the environment itself (air, land, water, biota), but rather from the impairments of human health that has been observed as a result of a degradation of the environment.

Quite similarly, in radiation protection, the first phase of development has been anthropocentric (Figure 1, top), restricting the consideration of the environment to a simple vector of radionuclides towards human beings, transferred through air, water, soils and sediments, and/or through animals and plants grown for agricultural purposes (as food source: vegetables, milk, meat, etc…). This phase considered human beings as the only target of concern and environment protection was directly subordinated to this goal. It formed a very linear concept which oriented research work on transfer of radionuclides essentially, and considered man as being located external to the environment.

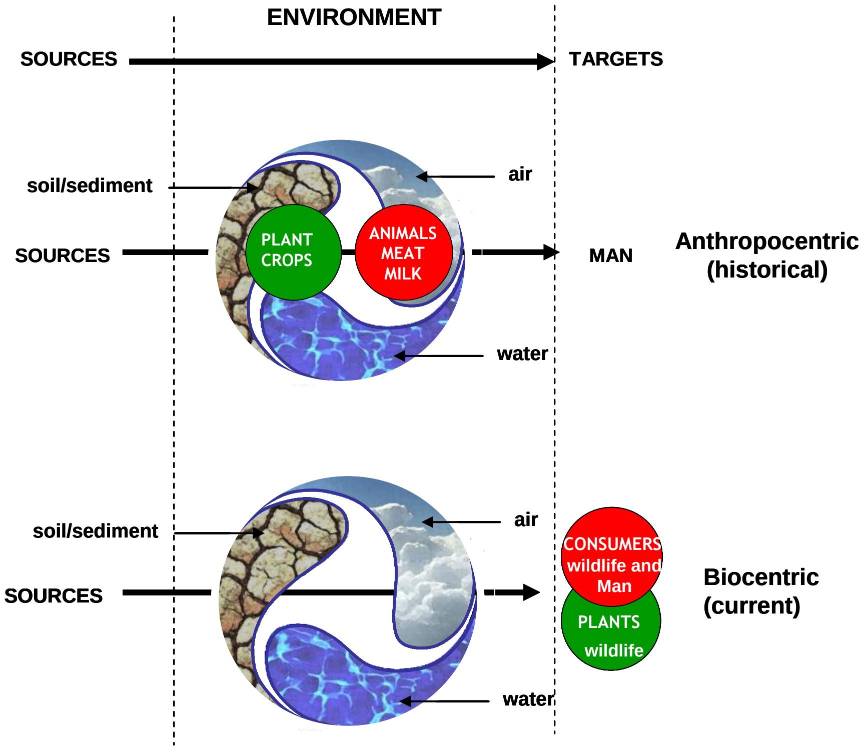


Figure 1: The anthropocentric and biocentric views over the environment for the purpose of its protection against radiation: a linear concept

Perhaps, the first reason that has driven to protecting the environment per se has been the rarefaction of biological resources (e.g., fish, game, forest logging) that had been exploited for centuries as granted for free by nature through numerous generations of harvesters, hunters and agriculture growers. This has been followed more recently by facing a rarefaction of species richness (biological and genetic diversity) which promotes fears in our societies with respect to the sustainability of future generations. Next, have appeared more or less concomitantly the growing contamination of these resources by xenobiotics (technogenic substances released by human activities and accumulated in the environment), which impact on human health, but also on other life forms. Artificial radionuclides produced within the nuclear cycle of electric power generation, from atomic weapons testing or for other industrial and medical purposes, and also "technologically enhanced naturally occurring (radioactive) materials" (TENORMs) resulting from mining and various mineral/organic resources exploitation (oil and gas), all fall in this category.

The growing size of the planetary human population drives to questions relative to the ecosystems' capacity to provide biological resource such as food in a sustainable manner. Ensuring safe food to the human population is a question of both, the nutritive quality of the biological resource itself, but also the health status of the corresponding biota assembly from which such food resources are derived, i.e. making sure that the relevant ecosystem processes which provide such resources (so called "services") are not endangered. Indeed, the potential deleterious impacts from xenobiotics introduced into the environment by human (industrial) activities has prompted a concern about the health status of non-human biota, therefore shifting the protection focus from man to other species as well. This is why radiological protection, in a second phase, has moved to a biocentric approach (Figure 1, bottom), as currently driven by ICRP Committee 5. Paralleling the system of protection designed for man, it takes now non-human biota as targets of radiation and follows a quite similar conceptual philosophy. The resulting system of protection of "non-human biota" (and not "of the environment", as often abusively stated) is dominated by the need for practical operation, leading in turn to a number of simplifications.

2.3 The biocentric approach relies on effect endpoints focused upon individual organisms

There is a high complexity in the environment: a range of abiotic components (soils, waters, gaseous atmospheres), a vast biodiversity of species interconnected within hierarchical space and temporal structures which support ecosystem functioning, a broad diversity of potential pathways and conditions of exposure to radioactivity (acute/chronic exposure, high/low doses). Simplification therefore has been one main driver to the biocentric current conceptual approach that has been grounded on the reference organism concept.

This concept takes a limited set of "reference organisms" (mimicked from the concept of "reference man" used in human radiation protection), chosen along various scientific and practical criteria, and meant to serve as points of comparison in ecological risk assessments. Each "reference organism" is documented (from a wide literature survey of radio-toxicological data) in terms of radiation induced dose-response curves through four endpoints corresponding to the individual organism level: mortality, morbidity, reproductive success and mutation. In addition, environmental exposure pathways and transfers are related to the dose received by these individual organisms based on the development of simple dosimetric calculation models. All together, these are next used to construct a scale of risk (ICRP, 2008). This concept is still linear but emphasizes now effects on non-human biota. Most current research effort is consequently oriented towards feeding the biocentric approach with more relevant data.

One basic advantage of this approach is to ensure an immediate consistency with the system of radiological protection of man due to the similarity of the built-in concepts. This is seducing as it opens the path to designing a unique system of radiological protection of both, humans and "the environment". However, one must strengthen that "the environment" here is actually restricted to "non-human biota" only, taken up to the individual organism level and not further. The evolving methodology, therefore, misses an ability to also address interactions between species, one paramount aspect of ecological relevance and concern in environment protection, where emphasis is more on higher levels of biological organisation (populations and communities of interacting species). It is further stressed that ensuring overall consistency with other conceptual approaches to environment protection is also of importance, further observing that they tend today to consider environmental problems in a more integrated and holistic manner, as detailed in the following.

2.4 Protection objectives: the driver to selecting appropriate targets and related endpoints

The "environment" notion covers a range of different realities: pristine nature, environmental media such as soils, atmosphere and water (including geological resources,…), individual organisms of endangered wildlife species, communities of interacting populations of species (fish stocks in the ocean, tropical forest,…), landscapes, habitats, ecosystems including their provision of life support functions (air regeneration, waste recycling, biomass production…) and of services (pollination, climate control,…). Depending on given particular problems deserving protective actions, various sectorial objectives of protection have been derived and translated into several legislations (endangered species, coastal environments, tropical forests, migrating birds, fish stocks, Natura 2000 habitats, etc…). It is clear that the adequate target of protection to select, along with its relevant endpoints, should be identified in accordance with the protection objectives assigned.

Harm or stress to an ecological system, via the introduction of a xenobiotic for example, as illustrated in Figure 2, has the potential to interact within the whole scale of biological complexity, from molecules up to full ecosystems. A protection system exclusively based upon organismal targets therefore necessarily misses protection objectives involving population and ecosystems attributes. This is a particular concern when facing more integrated objectives of protection expressed in terms of protection of ecosystem structure (biodiversity) and functions (life support and services), as this more and more currently relayed in the upcoming international legislation. This trend is illustrated by 1) the consensus goal that environment protection would best be served by targeting populations and their interactions within ecosystems, 2) the upcoming international legislation which often recommends to adopt an "ecosystem approach" (like the Convention on Biological Diversity, for example), 3) the clear focus on ecosystemic concepts in other fields of environment protection (biodiversity, halieutics, forestry).

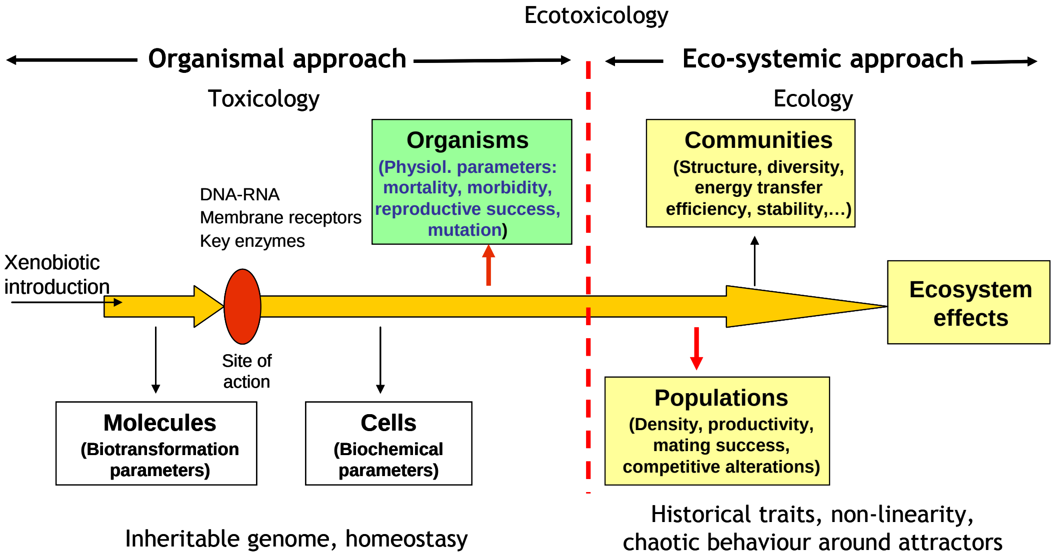


Figure 2: Parameters and indications of the interaction of xenobiotics with all levels of biological organisation within the ecosystem (adapted from Landis and Yu, 2004)

2.4.1 Organisms, populations and ecosystems: different targets requiring specific endpoints

New ecological/ecosystem theories currently develop a better description and understanding of the behaviour of complex ecological systems (Jørgensen, 2006; Kay, 2000; Müller et al., 2000). As opposed to the classical approach to presenting the impacts of toxicants upon various aspects of biological systems, a new framework is now proposed that incorporates complexity theory.

Essentially, the basic format of this framework features two distinct types of structures that concern risk assessment. Living organisms (left, on Figure 2) have a central core of information, the genome, subject to natural selection, and which drives homeostasis upon the constituents of that system. The genome of an organism is highly redundant, a complete copy existing in virtually every cell, with high communication and coordination between the various constituents within organisms. Somatic cells and structure of the organism are steadily maintained through successive generations by true inheritance through the germ line (unless DNA mutations).

Above this individual organism level, ecological (non-organismal) structures have fundamentally different properties (right, on Figure 2). Here, there is no central and inheritable repository of information, analogous to the genome, which would serve as the blueprint for an ecological system. Natural selection is selfish, working upon the phenotype characteristics of a genome and its close relatives, and not upon a structure that exists beyond the confines of a genome. Hence, the lack of a blueprint and the many interactions and non-linear relationships within an ecosystem mean that the history of past events is written into its structure and dynamics. The many non-linear dynamics and historical nature of ecosystems are characteristic of complex systems, and provide them with emergent properties which are critical to how they react to contaminants.

In this context, Cambel (1993) has emphasized the following properties: complex systems are neither completely deterministic nor stochastic, they exhibit both characteristics; they undergo irreversible processes; they are dynamic and not in equilibrium, they are constantly moving targets; their different parts are linked and affect one another in a synergistic manner; the causes and effects of the events which the system experiences are not proportional.

2.4.2 Populations attract more consensus as targets of protection than organisms

There is a consensus today to consider the population(s) as the most relevant and pertinent object of protection and many authors call for ecological risk assessment that would consider risks to populations, no more simply to individuals. The main reason for that is that all individuals eventually die, whereas populations persist in the long run. This is why interest in population-level ecological risk assessment has dramatically increased within both, the scientific and regulatory communities. The Society of Environmental Toxicity and Chemistry (SETAC) in particular is advancing the practice of population-level ecological risk assessment (Barnthouse et al., 2007). Such developments have been prompted by the consensus recognition that individual-based assessments are inadequate for the prediction of the ecological fate of a species-specific endpoint. The current rarity of assessments that focus on population characteristics does not result from the absence of a scientific foundation or understanding, but rather from the lack of concerted effort to advance their use in a risk management context.

An operational definition of the population is essential to examine the biological and ecological context necessary for risk assessments. Roughgarden (1996) defined the population as a group of individuals that are genetically and reproductively connected so that the transfer of genetic information to the next generation is greater within the group than between groups. Although the individuals provide the means, reproduction for obligate sexual organisms is a population-level property. A ramification of this definition is that the individual organism is ecologically insignificant unless placed in the context of a population. The population provides the individual mates, a gene pool for genetic recombination, social structure, modified habitat, and all other information necessary for the survival and transmission of the genetic information of the individuals to the next generation.

2.5 Towards an "ecosystem approach" featuring population/ecosystem level endpoints

2.5.1 The need for an additional ecocentric approach focused upon the ecosystem concept

Today's regulatory (and public) perceptions of the value of nature emerge from two major considerations: the need to safeguard biodiversity and the will to preserve life-supporting functions within natural systems such as to maintain safe drinking water, clean air and safe non-contaminated food, all of which depend on ecosystem level processes.

The need to view environmental problems in a more holistic manner, through the ecosystem concept, comes therefore from the recognition that human health is strongly bound to the health status of the environment itself. Toxic substances which man introduces in the environment elicit direct deleterious effects on humans, animals and plants, but also promote alterations of ecological processes which indirectly impact them, ultimately (Bréchignac, 2003; Bréchignac and Doi, 2009). This leads to considering the relationship between environment and human protection no longer through a linear view, but as a cycle/loop system within which man promotes changes in the environment (harmful or not to non-human biota), such changes in turn being capable of promoting harmful feed-back impacts in humans. The ecosystem concept best captures this holistic representation of the interactions and relationships between human beings, other species and their environmental surroundings.

2.5.2 The subsystems-to-system extrapolation in question

Controlled laboratory tests on single-species systems provide clear and easily understood linkages between stressor exposure and effects. They are typically inexpensive, quick and easy. But a population perspective invites examination of complexity and the use of experimental information to address issues associated with multiple stressors, cumulative effects and real-world population dynamics. Factors regulating populations such as disease, predation, and combination of stressors are important to consider. Criticisms of the extrapolation from laboratory single-species toxicity tests to an ecosystem effects approach state that toxicity tests do not consider bioaccumulation of contaminants and ignore both temporal changes and multiple stressors effects.

Indeed, recent observations or experimental investigations on the effects of radiation have demonstrated that overall responses at ecosystemic level may not be simply derived from local responses observed at individual organism level (Bothwell et al., 1994). This can be due to indirect effects mediated through alteration of trophic interactions between populations of different species (Fuma et al., 2003; Doi et al., 2005). But more generally, this roots from "emergent" properties of ecosystems, like resilience or resistance, which drive to non straightforward propagation of effects across levels of biological organisation (Sokolov and Krivolutsky, 1998), or through successive generations (Massarin et al., 2010). Similar responses have already been faced in other fields of environmental protection against other stressors, pushing a number of environment professionals to assign stronger emphasis on more systemic approaches.

2.5.3 Ecological impact depends on the resilience/resistance of ecosystems

Resilience and resistance are specific properties of ecosystems emerging from complexity, both referring to their stability. Resistance is defined as the capacity of a community to maintain its structure following exposure to perturbation (that is to absorb disturbance), and resilience the capacity of the ecosystem to reorganise while undergoing change so as to retain essentially the same function, structure, identity and feedbacks (ecological resilience, after Folke et al., 2004).

When exposed to gradual changes in biotic and/or abiotic factors, ecosystems usually respond to these perturbations in a smooth way or even without any externally visible response. However, in some occasions, sudden catastrophic shifts between different ecosystem states, which are called “regime shifts” around “attractors”, are observed to occur. These are caused by the combination of the magnitudes of external forces (the above-mentioned changing factors) and the internal resilience of the system. As anthropogenic disturbance (pollutant) or natural factors increase (i.e. nutrient loading, climate and habitat fragmentation), the ecosystem, becoming vulnerable to smaller disturbances or gradually changing conditions that it could previously cope with, is now at high risk of shifting to a qualitatively different state. Such shifts may be difficult or impossible to reverse.

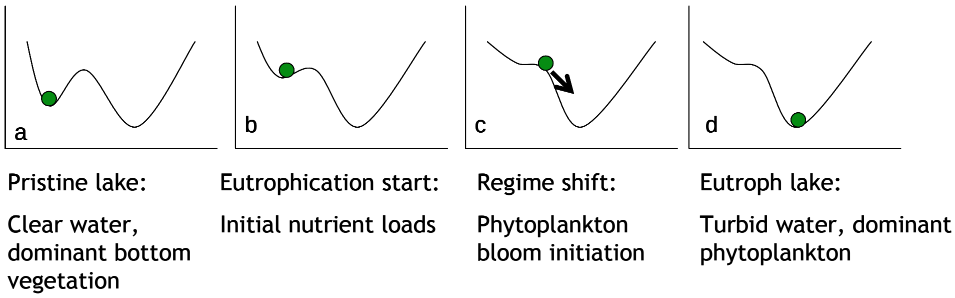


Figure 3: Regime shift of an ecosystem as resulting from its resilience decrease (adapted from Folke et al., 2004)

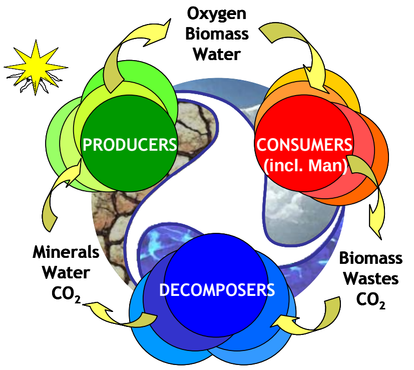
A quite commonly described example of such “catastrophic regimes” is the eutrophication observed in lakes responding to nutrient loads, an explanation of which is summarized by Bréchignac (2003). The pristine status of most shallow lakes is clear water with rich submerged bottom vegetation. After a certain degree of nutrient accumulation, the lake shifts abruptly from clear to turbid with high levels of phytoplankton and loss of bottom vegetation (Figure 3). The original status may eventually be reversed, but only after dramatic reduction of nutrients, down to much lower than the level at which the regime shift occurred (high hysteresis).

One understands from this feature of ecosystems that depending on the status of their intrinsic level of resilience/resistance, their radio-sensitivity may be quite different, absorbing radiation stress when there are resilient/resistant, or eventually undergoing catastrophic shift when their resilience/resistance is altered (by previous continuous exposure to stress, for example).

2.5.4 Designing an ecocentric view

The above-mentioned considerations lead to advocating the need to boost science and methods along an **ecocentric** approach (Bréchignac, 2002, 2003; Bréchignac and Doi, 2009), in a third phase (Figure 4). Leaving the previous linear view, the approach features now the ecosystem, with its loops of material and energy cycling, as a central concept to structure the system of radiological protection (of the environment, including man).

Ecocentric: Environment including man



Ecosystem = biotope + biocenose

biotope	biocenose
Air	**Animals (incl. Man)**
Water	**Plants**
Soil/sediment	**Microbes**

• **Services : waste recycling, provision of resources, climate regulation, …**

• **Life support :air bioregeneration, water recycling, biomass (food) production,…**

Figure 4: Ecocentric approach for environment protection featuring the ecosystem concept

There is indeed a growing awareness nowadays among policymakers and scientists that assessment studies should adopt an ecosystem approach. Started almost 2 decades ago, this trend is best illustrated by the vast literature on this subject area (Crober, 1999), demonstrating that environmental managers, the primary customers of methodologies for ecological risk assessment, have shifted towards applying an ecosystem approach to environment management.

Today, the recommendation to apply an ecosystem approach can be traced within many governmental institutions and agencies throughout the world, as related to the protection of biodiversity (Convention on Biological Diversity, 2004), of marine resources (FAO, 2005), of marine and coastal environments (Laffoley et al., 2004), among many others. Such a recommendation has also been expanded recently by IUR (2012) to the radiological protection of the environment with extensive justification and discussion.

2.6 Conclusion

There are several advantages expanding the system of radiological protection of the environment to also considering an ecosystem approach. First, it overcomes the large uncertainties generated by the otherwise necessary extrapolations from organism toxicology to impacts on ecosystems. Second, it solves a frequent ambiguity attached to using the “environment” terminology (especially when it actually refers to the organismal level only) because it embraces an overarching goal of protection: preserving life sustainability through protection of ecosystem structure and functioning. Finally, it improves the radiological protection credibility by adopting the modern concepts which the overall environment protection community is now developing to overcome and prevent man-made damages to the ecosystem-based sustainability of life.

The sustainability of life (the actual main driver for environment protection, including man), is not exclusively a question of toxicological harm to organisms. It is also a question of maintaining symbiotic-like assemblies of interacting species in ecosystems because these latter provide essential features such as life support and many services. In other words, life sustainability is best characterized in terms of ecosystems which include organisms rather than in terms of individual organisms only.

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3 PROTECTION OF THE ENVIRONMENT IN NORMAL (PLANNED) SITUATIONS

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3.1 Introduction

The UK has a duty to comply with the EC Habitats (Council Directive 92/43/EEC on the conservation of natural habitats and wild flora and fauna) and Wild Birds (Council Directive 79/409/EEC) Directives when planning and undertaking all of its regulatory and operational activities. These European Directives were introduced into UK legislation by the Conservation (Natural Habitats & c.) Regulations 1994. These Directives established and protect a network of conservation areas across the EU called 'Natura 2000'. Natura 2000 is made up of sites designated as Special Areas of Conservation (SACs) and Special Protection Areas (SPAs).

Under the Regulations, the UK environment agencies (the Environment Agency in England and Wales and the Scottish Environment Protection Agency (SEPA) in Scotland) have obligations to review relevant existing authorisations, permits, consents, licences and permissions (collectively referred to as permits) to ensure that no authorised activity or permission results in an adverse effect, either directly or indirectly, on the integrity of Natura 2000 sites. In addition, any new or varied permits must not have an adverse effect on the integrity of the Natura 2000 sites. This applies equally to chemical and non-chemical stressors (such as water abstraction). Within the UK, it has been decided that the review should also include the authorised (i.e. planned exposure situations) discharges of radioactive substances to the environment. While SEPA and the Environment Agency have adopted similar approaches to the review, this paper focuses on the approach adopted by the Environment Agency.

The Environment Agency adopted a staged approach to reviewing its' existing permits:

- Stage 1 – identify the relevant permits.
- Stage 2 – determine which permits have a potential significant effect.
- Stage 3 – undertake appropriate assessment for permits with significant effects.
- Stage4 – revise permits to ensure no adverse effects (e.g. by changing the type, amount and location of discharges).

These steps will be described in more detail in the following section. This staged approach was thought, in part, to address the fact that the Natura 2000 sites were categorised by the relevant conservation agencies into high, medium or low priority for the overall habitats assessments. It should be noted that this categorisation was not driven by the potential effects from radioactive substances or the other stressors being assessed (e.g. metals, water abstraction etc.) but was based on the sites' value to conservation. Different timescales to complete the assessments were then set by the conservation agencies:

Site priority	Complete Stage 3 by:	Complete Stage 4 by:
High	March 2004	March 2006
Medium	March 2006	March 2008
Low	March 2008	March 2010

The overall approach (section 3.2) and the results (section 3.4) obtained from the assessment in England and Wales were discussed and agreed with Natural England and the Countryside Council for Wales who are responsible for reporting on the condition of Natura 2000 sites and providing advice on conservation objectives for each Natura site to the government. Similar arrangements are in place for SEPA to consult with the relevant agencies within Scotland.

3.2 Staged Assessment Approach

The Environment Agency authorises discharges of radioactive waste to the environment from a variety of premises, including hospitals, universities, pharmaceutical companies and nuclear licensed sites. At the time of the assessments being described here these discharges were authorised under the Radioactive Substances Act 1993 (RSA 93), now, in England and Wales, they are authorised under the Environmental Permitting Regulations (2010). These disposals include discharges to air and water. All of these required some form of assessment to determine whether there was likely to be an adverse effect on the integrity of the Natura 2000 sites.

The Environment Agency completed Stages 1 and 2 of the permit review process in 2003 to identify the RSA 93 authorisations that could have a potential impact on Natura 2000 sites. Stages 1 and 2 assessments were used to determine the number of potentially impacted Natura 2000 sites following the approach described in EA, 2002. The Stage 1 assessment filtered out applications and activities authorised by the Agency that, by virtue of their nature or location, could not conceivably have an effect on the features of interest of given European sites. This was conducted using simple rules to determine the likelihood of an authorised discharge reaching a Natura 2000 site so, for example, atmospheric releases are only considered further if they occur within 1km of the Natura 2000.

Having identified a number of potentially impacted sites in Stage 1 (mainly intertidal areas of estuaries), the Stage 2 assessment then reviewed the maximum permissible radioactive discharge levels from authorised sites and compared these to defined screening levels (Allott and Dunn, 2001). Research conducted by Copplestone et al. (2001) was used to underpin the derivation of the discharge screening levels. This approach used the concept of reference organisms, which are defined as "*a series of entities that provide a basis for the estimation of radiation dose rate to a range of organisms which are typical, or representative, of a contaminated environment. These estimates, in turn, would provide a basis for assessing the likelihood and degree of radiation effects. It is important that they are not a direct representation of any identifiable animal or plant*" (Strand and Larsson, 2001). The reference organism concept therefore provides a series of organism types, which can be considered representative of different trophic levels (see Table 1). Section 3.3 discusses how the reference organism concept fits with the Reference Animals and Plants framework described by ICRP (2008).

Stages 1 and 2 identified those Natura 2000 sites for which there was a potential transfer route of authorised radionuclide discharges to reach one or more Natura 2000 sites. However following a review of the criteria used for these two stages, it was concluded that some criteria were slightly optimistic (e.g. no impact from releases to air if greater than a distance of 1 km from a Natura 2000 site) or probably inappropriate (e.g. use of human drinking water standards). Also, more scientific information became available which needed to be taken into account. For these reasons, it was decided to include all RSA 93 authorisations in the Stage 3 assessment. Revised criteria were established to select whether these RSA 93 authorisations could have an impact on Natura 2000 sites. These criteria are described in detail in Allott et al. (2009).

Table 1: Terrestrial Reference organisms as listed in Copplestone et al. (2001)

Reference Organism (and example species)	**Reference dimensions (mm)**	**Mass (kg) Fresh weight (FW)**
Lichen	100 x 5 x 5	1.31E-03
Moss	100 x 10 x 5	2.62E-03
Tree (root)	100 x 2 x 2	2.10E-04
Shrub (root)	100 x 2 x 2	2.10E-04
Herb (root)	100 x 2 x 2	2.10E-04
Germinating Seed	6 x 1 x 1	1.80E-06
Fungal fruiting body	30 x 15 x 10	2.63E-03
Caterpillar	30 x 7 x 7	7.70E-04
Social Insect - ants	5 x 3 x 3	2.00E-05
Social Insect - bee	20 x 15 x 10	2.00E-03
Wood Louse	15 x 6 x 3	1.00E-03
Earthworm	100 x 5 x 5	3.50E-03
Herbivorous Mammal (rabbit)	300 x 150 x 100	8.00E-01
Carnivorous Mammal (fox)	670 x 350 x 180	5.50E+00
Small Burrowing Rodent (mouse)	100 x 20 x 20	2.00E-02
Woodland Bird (Grouse)	350 x 150 x 150	1.50E+00

The Stage 3 assessment methodology involved the calculation of dose rates to reference organisms and feature species (those species identified by the conservation agencies as being important within each Natura 2000 site (see Table 2)) from exposure to authorised discharges of radioactive substances at Natura 2000 sites in England and Wales. Dose rate was used as the indicator of potential harm to the organism from this exposure and was calculated or measured in units of microgray/h. The dose rates for reference organisms can be related to effects data (e.g. mortality, morbidity, reproductive effects) as described by Woodhead and Zinger 2003.

The Stage 3 assessment methodology was based on Environment Agency Science (Copplestone et al. 2001, 2003), which was developed as part of the Euratom FP5 Project 'FASSET' (Larsson, 2004). It involved the calculation of dose rates to organisms from the total discharges of radionuclides at RSA 93 authorisation limits which may affect a Natura 2000 site multiplied by dose rate per unit release factors (DPUR) (see section 3.2.1).

The Environment Agency, Natural England and the Countryside Council for Wales also agreed a dose rate threshold of 40 microgray/h, below which it was concluded that there will be no adverse effect on the integrity of a Natura 2000 site. This was derived as follows:

- Research from the Euratom FP5 Project 'FASSET' (Larsson et al. 2004) indicated that, in general and from the available data, there appear to be no significant adverse effects in biota exposed at levels of up to 100 microgray/h.
- A review paper from the FASSET Project (Brown et al. 2004) indicated that wildlife might receive up to 60 microgray/h (as a weighted dose rate) from natural sources in European ecosystems.
- The threshold of 40 microgray/h for authorised discharges of radioactive substances is the difference between these two values.

Table 2: Feature species (only common names given) identified at the Natura 2000 sites

Bird species (55)	Avocet, Bar-tailed Godwit, Bewicks Swan, Bittern, Black-tailed Godwit, Brent goose, Chough, Common Scoter, Common Tern, Cormorant, Curlew, Dartford Warbler, Dunlin, Gadwall, Gannet, Golden Plover, Great Crested Grebe, Grey plover, Guillemot, Hen Harrier, Honey Buzzard, Kittewake, Knot, Lapwing, Lesser Black-backed Gull, Little Tern, Manx Shearwater, Marsh Harrier, Mediterranean Gull, Nightjar, Oystercatcher, Peregrine, Pink-footed Goose, Pintail, Puffin, Razorbill, Redshank, Ringed Plover, Ruff, Sanderling, Sandwich Tern, Scaup, Shelduck, Short-Eared Owl, Shoveler, Snipe, Stone Curlew, Storm Petrel, Teal, Tufted Duck, Turnstone, White-fronted Goose, Whooper Swan, Wigeon, Woodlark
Plant species (4)	Early Gentian, Fen Orchid, Petal Wort, Shore Dock
Terrestrial invertebrates (2)	Southern Damselfly, Stag Beetle
Amphibians (2)	Great Crested Newt, Natterjack Toad
Terrestrial mammals (4)	Bechsteins Bat, Dormouse, Greater Horseshoe Bat, Lesser Horseshoe Bat
Aquatic mammals (3)	Common Seal, Grey Seal, Otter
Aquatic invertebrate	Desmoulins Whorl Snail
Fish species (8)	Allis Shad, Atlantic Salmon, Brook Lamprey, Bullhead, River Lamprey, Sea Lamprey, Spined Loach, Twaite Shad
Reptiles (2)	Sand Lizard, Smooth Snake

This threshold of 40 microgray/h is the same as the lower 1992 guideline level for terrestrial animals published by the International Atomic Energy Agency (IAEA 1992). The IAEA stated that it is unlikely that there would be any significant effect on populations of terrestrial animals, which are chronically exposed at these levels.

The RSA 93 authorisations for the disposal of radioactive waste to the environment were collated. In September 2007, there were 700 authorisations representing nearly 4000 releases of individual radionuclides or groups of nuclides, which were assessed (section 3.4). All the assessments were conducted on the basis that discharges from a site were at the RSA93 authorisation limit. This was a conservative assumption because, in practice, actual discharges are known to be much lower than the authorised limit (often by less than 50% of the limit).

3.2.1 Calculation of dose per unit release values (DPUR)

Dose rate per unit release factors for marine, freshwater and terrestrial environments were calculated for different radionuclides and reference organisms types from dose rate per unit concentration data in air, soil and water combined with simple dispersion modelling factors (concentration per unit release). The dose per unit concentration values were determined from Copplestone et al. (2003) for the different reference organism types. These were weighted to take account of the likely effects of different radiation types. Hence, all calculated dose rates reported are weighted dose rates. It should be noted that, in 2013, a project was started within the Environment Agency to update these DPURs to reflect the developments made in the European funded Environmental Risk from Ionising Contaminants: Assessment and management (ERICA) project (contract number FI6R-CT-2004-508847) which incorporates, for example, a wider range of radionuclides than are found in Copplestone et al. (2003) and a significantly updated database of the key parameters needed to estimate the dose rate to wildlife (see Brown et al., 2008 and Beresford et al., 2007). Figure 1 outlines the steps to estimate the dose rate to reference organisms and highlights the key parameters that are required (e.g. a concentration ratio for determining the radionuclide content in the whole body of a reference organism relative to the relevant media (air, sediment, soil or water depending upon the ecosystem/reference organism of interest)).

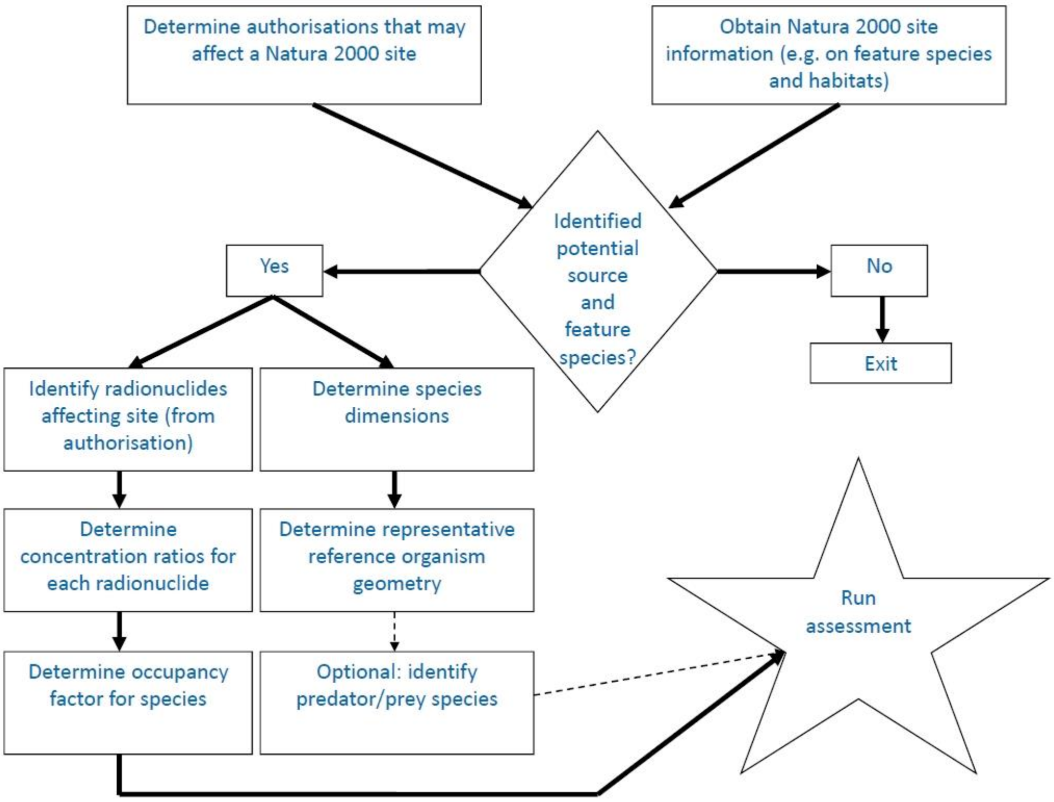

Figure 1: Overview of the assessment process given in Copplestone et al., 2003

It is possible to determine relevant ecological parameters for a species representative of the reference organism (for example, a herbivorous mammal might be selected to represent a rabbit). The ecological parameters provide information on prey, predators and the time spent in different compartments of the ecosystem, for example for a duck, how much time is spent on sediment, surface of the water, flying etc.. It is also possible to define a simplified geometry (usually as an ellipsoid) for the purposes of dosimetric calculations (full details are provided in Copplestone et al. 2001). The equations used are given below. For each reference organism a concentration factor has been defined relative to soil, water or air depending upon the type of assessment (freshwater, estuarine/marine or terrestrial) and radionuclide.

The assessment approach was developed for the following radionuclides for different ecosystems, as follows. Terrestrial only: ^{35}S, ^{41}Ar, ^{85}Kr, ^{226}Ra. Aquatic only: ^{99}Tc, ^{125}I, ^{210}Po. Both terrestrial and aquatic: ^{3}H, ^{14}C, ^{32}P, ^{60}Co, ^{90}Sr, ^{106}Ru, ^{129}I, ^{131}I, ^{137}Cs, ^{234}Th, ^{238}U, ^{239}Pu and ^{241}Am.

The dosimetry equations used are as follows:

Terrestrial ecosystems (soil)

$$\dot{H} = \sum_i (C_i^{soil} \times CF_i^{soil} \times DPUC_{total,i}^{\mathrm{int}}) + \left(\left(f_{soil} + 0.5 f_{surface}\right) C_i^{soil} \times DPUC_{total,i}^{ext}\right) \quad (1)$$

where:

$\sum_i$ represents summation over all nuclides;

C^{soil} is the activity concentration of the radionuclide in surface soil;

CF^{soil} is the concentration factor for the organism referenced to soil;

f_{soil} is the fraction of time the organism spends under the soil surface; and

$f_{surface}$ is the fraction of time the organism spends on the ground surface.

Terrestrial ecosystems (air)

$$\dot{H} = \sum_i (C_i^{air} \times CF_i^{air} \times DPUC_{total,i}^{\mathrm{int}}) + \left(\left(f_{soil} + 0.5 f_{surface}\right) C_i^{soil} \times DPUC_{total,i}^{ext}\right) \quad (2)$$

where:

$\sum_i$ represents summation over all nuclides;

C^{air} is the activity concentration of the radionuclide in air;

C^{soil} is the concentration of the radionuclide in surface soil, calculated from the air concentration and the relevant concentration ratio;

CF^{air} is the concentration factor for the organism referenced to air;

f_{soil} is the fraction of time the organism spends under the soil surface; and

$f_{surface}$ is the fraction of time the organism spends on the ground surface.

Similar equations have been defined for the aquatic situation.

Aquatic ecosystems

$$\dot{H} = \sum_i (C_i^{water} \times CF_i^{water} \times DPUC_{total,i}^{\mathrm{int}}) +$$
$$\left(\left(\left(f_{sediment} + 0.5 f_{surface}\right) C_i^{sediment} + \left(f_{water} \times C_i^{water}\right)\right) \times DPUC_{total,i}^{ext}\right) \quad (3)$$

where:

C^{water} and $C^{sediment}$ are the radionuclide activity concentrations in water and sediment respectively;

CF^{water} is the concentration factor for non-human species referenced to water;

$f_{sediment}$ is the fraction of time spent buried in sediment;

$f_{surface}$ the fraction of time spent on the sediment surface; and

f_{water} the fraction of time spent free swimming in the water column or on the water surface.

A radioactive substance habitats assessment spreadsheet tool was developed to assess the dose rates to terrestrial, freshwater and coastal organisms. In this tool the dose rate per unit release factors can be modified to take account of site-specific dispersion parameters by using the water exchange rate for releases to coastal waters, the river flow rate for releases to freshwaters, and a scaling factor for the effective release height for releases to air. This is described in more detail in Allott et al., 2009.

Derivation of concentration factor (CF) values

Probably the main source of uncertainty in the Stage 3 assessment results from the derivation of concentration factors (CFs) for the radionuclide/species combination. A review of the scientific literature revealed few data on the concentration of the radionuclides of interest in the feature species. This was not surprising given that most of the feature species are of conservation value and therefore unlikely to be included in routine sampling and monitoring programmes. Some CF data have been obtained from the literature and from species with a similar ecology but which are not of conservation value and that have been sampled.

In the absence of feature species and radionuclide specific CF values, an approach was developed, which aimed to provide a consistent method for selecting CF values for inclusion in the assessment. Essentially, the approach is as follows: determine if a CF was available for the species and radionuclide of interest. This was used if available. If not, a CF was selected from Copplestone et al. (2001) for a reference organism with similar ecology. If this was not available, the CF and K_d (aquatic only) values for the radionuclide of interest were reviewed and expert judgement used to determine which value should be recommended. Expert judgement is needed because, although the use of the K_d to equate to CF can be viewed as generally conservative in aquatic ecosystems, certain radionuclides, e.g. ^{131}I, are known to accumulate in organisms and in these situations the use of the K_d is unlikely to be appropriate. Finally, where no CF is available for another plant or animal group, the CF for the Environment Agency recommended analogue radionuclide was used (Allott et al., 2009). It should be noted that this approach might produce some highly conservative CFs (as for example in the case of using ^{137}Cs as an analogue for ^{99m}Tc). It is important to note the origins of the CFs used when interpreting the outputs of dose rate assessments. It is emphasised that the overall assessment must be supported by field measurements to produce site-specific data if there is any doubt over the results although this may be difficult if you have to measure the radionuclide content of a protected species. This is a key reason for the current project, which is updating the DPUR values using the latest scientific information.

3.3 Reference organism terminology

Since the original work to support the Environment Agency's habitat assessments, the International Commission on Radiological Protection (ICRP) has recognised the need to provide advice on environmental exposures and has outlined the development of a framework based on Reference Animals and Plants (RAPs) (ICRP, 2007). RAPs can be considered as hypothetical entities with certain assumed basic biological characteristics of a particular type of animal or plant, as described to the generality of the taxonomic level of Family, with defined anatomical, physiological and life-history properties (ICRP, 2008). They are not, therefore, necessarily the direct objects of protection themselves but, by serving as points of reference, they should provide a basis upon which some management decisions could be made. Simple dosimetric models, plus relevant data sets, are being developed by ICRP for the 12 RAPs it has identified: aquatic: brown seaweed, crab, duck, flatfish, trout; terrestrial: bee, deer, earthworm, frog, pine tree, rat, wild grass (ICRP, 2008).

The RAPs are then essentially an international subset of the 'reference organisms' described in Copplestone et al., 2001, 2003 and Larsson, 2004 and the reference organisms described are, in the ICRP terminology, equivalent to 'representative organisms'. Representative organisms, as described by ICRP (2008), are essentially the objects of assessment (equivalent of the human 'representative individual'). The actual choice of the representative organisms will depend upon the purpose of the assessment and may be specific and predetermined (e.g. named organisms in legislation – equivalent in this paper to the 'feature species' described previously) or may be selected purely on practical grounds to allow the assessment to be conducted.

3.4 Assessment results

The Environment Agency has now completed the Stage 3 assessments for the impact of discharges of authorised radioactive substances to air and water on Natura 2000 sites and the results are recorded in Allott et al., 2009. The results of the low priority site assessments have been sent to Natural England and the Countryside Council for Wales for consultation along with a reassessment of the high and medium priority Natura 2000 sites using current RSA 93 authorisations. These high and medium priority sites were originally assessed in 2004 and 2006.

The Environment Agency has assessed the impact of the radioactive discharges it authorises on Natura 2000 sites. These assessments have involved the calculation of dose rates to organisms in coastal, freshwater and terrestrial environments, taking account of the combined impact of discharges from multiple authorised releases and cautiously assuming that discharges occur at the authorisation limits. All discharges authorised under the Radioactive Substances Act 1993 that could have an impact on the Natura 2000 sites were included in the assessment. The total dose rates, calculated in the Stage 3 assessments, were compared to a threshold of 40 microgray/h, below which the Environment Agency, Natural England and the Countryside Council for Wales agreed there would be no adverse affect to the integrity of a Natura 2000 site The total dose rates to the worst affected organism are less than 40 microgray/h for all but two Natura 2000 sites (Ribble and Alt Estuaries SPA and Drigg Coast SAC) (Figure 2).

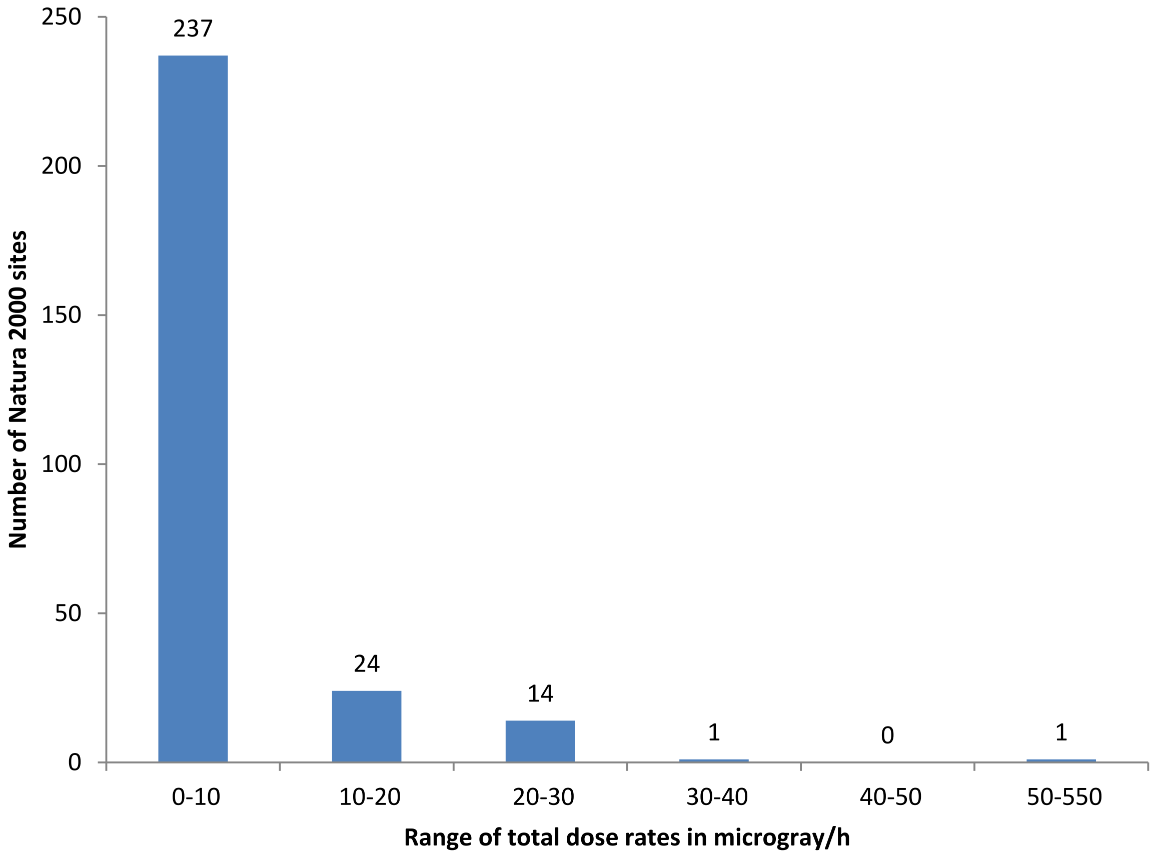

Figure 2: Overview of the habitat assessment results (Allott et al. 2009)

The calculated total dose rate to the worst affected organism for the Ribble and Alt Estuaries SAC was 520 microgray/h. This was significantly in excess of the agreed threshold and so this Natura 2000 site was included in Stage 4 of the Habitats Regulations implementation process. A separate report is available for the Ribble and Alt Estuaries which concluded that previously agreed new authorisation limits for the Springfields Fuels Ltd site (in effect from January 2008) would ensure that the dose rates to reference organisms and feature species will be less than 40 microgray/h (Allott and Copplestone, 2009). The total dose rate for the Drigg Coast SAC is just greater than the 40 microgray/h threshold. The assessment methodology is generally cautious, in particular compared to a new assessment methodology resulting from an EC funded project (ERICA assessment tool). The dose rate to the worst affected organism (phytoplankton) is 20 microgray/h using the dose rate per unit concentration data from the ERICA assessment tool. The Drigg Coast SAC was also considered in an ERICA project case study (Wood et al., 2008), which concluded that there was no indication of significant impact from ionising radiation on the sand dune biota. This Natura 2000 site will be kept under review. It was recommended that consideration be given to refining the assessment parameter data and the overall assessment approach for phytoplankton, in light of the Ribble and Alt Estuaries report. As required under the regulations the assessments for Natura 2000 sites should be kept under review and subject to detailed assessment where the predicted dose rates approach the agreed threshold.

3.5 Conclusions

To date the environment agencies have assessed the impact of radioactive discharges on the Natura 2000 sites. These assessments have involved the calculation of dose rates to organisms in coastal, freshwater and terrestrial environments, taking account of the combined impact of discharges from multiple authorised releases and using cautious assumptions like the discharges occur at the authorisation limit. All authorised discharges that could have an impact on the Natura 2000 sites were included in the assessment. The calculated total dose rates were compared to a threshold of 40 microgray/h, below which the Environment Agency, Natural England and the Countryside Council for Wales agreed there would be no adverse effect to the integrity of a Natura 2000 site. More recently there have been assessments made using the European funded ERICA tool (Brown et al., 2008), for example in an updated Sellafield wildlife dose assessment and within the UK’s generic design assessment process for new nuclear power stations. There are plans to adopt the ERICA tool for all future (re)-assessments for all Natura 2000 sites. This work is currently underway.

3.6 Acknowedgements

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4 EFFECTS OF NON-HUMAN SPECIES IN AREAS AFFECTED BY THE RADIATION ACCIDENT: IMPLICATIONS FOR RADIATION PROTECTION

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4.1 Introduction

At present, there is still a lack of quantitative information on the real long-term biological consequences of radiation exposure. Only a few studies exist that are directly relevant to understanding the response of plant and animal populations to radionuclides in their natural environments and these studies were performed mainly in areas affected by the severe radiation accidents (Alexakhin et al., 2004). Areas affected by radiation accidents provide a valuable source of the information on radiation effects in different biota species and represent an important input to assessment of acceptable dose levels for non-human species based on a variety of end-points. An assessment of the state of plant and animal populations, inhabiting contaminated territories, and the analysis of mechanisms of their adaptation to adverse environmental conditions is also of high biological importance. Field data from natural settings can improve our overall abilities for evaluating consequences of severe man-caused disasters for the environment.

A characteristic feature of severe radiation accidents is the presence of several phases, including intensive short-term radiation impact, interim phase and subsequent long period with a slow decline in the dose rate. Each such period can be characterised by specific radiation effects and only holistic assessment of these effects allow proper evaluation of the consequences of radiation contamination of the environment.

Field data on biological effects in non-human species are characterised by high uncertainty because of high heterogeneity of radioactive contamination of the territory and not adequate dosimetry or not sufficient documentation of the environmental parameters in many studies. Biota species may be exposed to different radionuclides with different physical and chemical properties and through a variety of exposure pathways, which are not always known (IAEA, 1992).

Biological objects are characterized by a huge variety of sizes, shapes, rations, locations and occupied ecological niches, which may also result in huge differences in doses and in the types of affected tissues and organs in the radiological situations of interest. Diversity of the locations of those species may also result in high uncertainty in dose assessments and problems in interpretation of the observed biological effects. Additionally, living beings in the course of individual development undergo different stages, the sensitivity of which varies by orders of magnitude (Sarapult'zev and Geras'kin, 1993). During ontogenesis the geometry and conditions of irradiation vary significantly, which also makes dose estimation rather complex and makes it necessary to consider ecological and physiological peculiarities of organisms' development (Geras'kin et al., 2008).

The purpose of this report was to review effects of non-human species in areas affected by the radiation accidents and, based on this review, to consider the lessons learned which are important for radiation protection of the environment.

4.2 Chernobyl accident in April 1986

The accident occurred in April 1986 – ata period of high sensitivity for many biota species. Long period of intensive release of radionuclides, variable meteorological conditions at that time of release and wide spectrum of deposited radionuclides resulted in formation of noticeable "spottiness" (in both amount and spectrum of radionuclides) of deposition (IAEA, 2006).

Environmental effects following the Chernobyl accident were specific to 3 distinct time periods (UNSCEAR, 1996). During the first three weeks, radiation exposures can be characterised as an acute exposure which was responsible for forming most severe radiation effects in wildlife biota. Large quantities of short lived radionuclides (^{99}Mo, ^{132}Te/^{132}I, ^{133}Xe, ^{131}I and ^{140}Ba/^{140}La) were present in the environmental contamination (IAEA, 2006). High doses to the thyroids of mammals were observed mainly during these weeks. The second phase of radiation exposure lasted during the summer-autumn of 1986, when a contribution of short lived radionuclides to the doses to wildlife biota was substantially decreased and longer lived radionuclides were transported to different components of the environment by physical, chemical and biological processes. Although the dose rates at the soil surface declined to much less than 10% of the initial values, high doses were observed at many sites affected by the accident. Around 80% of the total radiation dose accumulated by plants and animals was received within three months of the accident, and over 95% of this was due to beta radiation (IAEA, 2006). In the third (and continuing) phase of radiation exposure, dose rates have been chronic, less than 1% of the initial values, and derived mainly from ^{137}Cs (IAEA, 2006).

The consequences of the Chernobyl accident for biota varied from an enhanced rate of mutagenesis to severe damage of the ecosystems. In the course of the crucial (in terms of biological effects forming) early period radiation monitoring was insufficient. Therefore, and because of rapid changes of doses to biota species in that period (decay of the short-lived radionuclides, radionuclide redistribution in the ecosystems, changes in contributions of different exposure pathways etc.), estimations of doses absorbed by biological objects were mainly rough (UNSCEAR, 1996; IAEA, 2006; Hinton, et al., 2007; Geras'kin, 2008). Investigations of the effects of the radioactive contamination of wildlife biota started as early as May 1986, and allowed a direct evaluation of the radiation effects in a variety of biota species.

4.2.1 Forest trees

Mass mortality of pine trees was reported at an absorbed dose of 60-100 Gy over the "acute" period (Arkhipov et al., 1994; Tikhomirov and Shcheglov, 1994). Death of weakened coniferous trees and mass yellowing of needles are observed under exposure to lower doses: spruce – 4-5 Gy (Kozubov and Taskaev, 2007), pine – 8-12 Gy (Arkhipov et al., 1994; Kozubov and Taskaev, 2007). Doses, at which the recovery processes were observed, amounted to 50-60 Gy for Scots pine and 10-12 Gy for Norway spruce (Kozubov and Taskaev, 2002). Inhibition of reproductive capacity in pine was observed at doses 1-5 Gy (Fedotov et al., 2006).

Death of shrubs and deciduous trees were observed at doses above 200 Gy (Smirnov and Suvorova, 1996). Morphological changes in pine needles and underwood of deciduous trees in 1987 were observed starting from dose of 0.1-1.0 Gy (Arkhipov et al., 1994; Kozubov and Taskaev, 2002) and in herbaceous plants from dose of 1.5-2.3 Gy (Suvorova et al., 1993). Increased numbers of hollow seeds, morphological changes one year after the accident were found to be at the dose level of 0.5-1.0 Gy. The minimum dose at which morphologic effects in the Chernobyl zone (decrease in the increment of shoots, compared to the control in 1987) was observed in the majority of trees starting from a dose of 0.43 Gy, while a dose of 3.45 Gy caused full cessation of growth (Sidorov, 1994).

4.2.2 Herbaceous plants

Inhibition of photosynthesis and transpiration was observed at the dose levels of 1-5 Gy, while reduced growth and plant developmental problems were found to be at the doses up to 10 Gy. Reduced numbers of plants per square meter and reduction of species diversity were observed in herbaceous phytocenosis in 1987 starting from γ-dose rate of 17 mGy/day (Suvorova et al., 1993), enhancement of vegetative mass of heather (*Calluna vulgaris L.*) and gigantism of some plant species were observed at external dose rates more than 24 mGy/day and 36 mGy/day, respectively (Smirnov and Suvorova, 1996). Decrease in the number of peas in pods of wild vetch, increase in both fraction of sterile pods and fraction of embryonic lethals was observed at a rather low dose levels, namely, 0.4 mGy/day (Smirnov and Suvorova, 1996). Morphological changes in herbaceous plants were observed at the site with doses ranging 1.5-2.3 Gy (Suvorova et al., 1993).

4.2.3 Soil Invertebrates

The Chernobyl accident coincided with the most radiosensitive phase in the development of soil inhabitants: period of reproduction and molting of invertebrates after winter somnolence and spring warming up of the soil invertebrates. Doses of about 10-30 Gy led to a high (by a factor of 10-30) reduction in the population density and substantial change in the composition of species of forest litter invertebrates (Krivolutsky and Pokarzhevsky, 1992). In arable soils, even at accumulated absorbed dose on the surface of 86 Gy, soil animals were damaged relatively less (2-3 times, with no reporting in any of the animal groups of a catastrophic drop in the population size), probably because they were well protected against β-radiation by the soil layer (dose of β-radiation in the upper soil layer was 3-10-fold lower than in the litter), the main contributor to the total dose. Due to high radiosensitivity of the invertebrates at early stage of the development, the process of reproduction of soil inhabitants was greatly disturbed (Geras'kin et al., 2008). Among the inhabitants of pine forest litter, first instar larvae and nymphs failed to be detected. Near the Chernobyl NPP young earthworms (*Nicodrilus caliginosus S., Dendrobaena octaedra S.*) did not survive or hatch from cocoons in the autumn of 1986 (Geras'kin et al., 2008).

4.2.4 Vertebrates

The major radiation impact on farm animals was the thyroid affection due to the radioiodine accumulation in the thyroid. Two hundred forty days after the accident, in cows from the Gomel region (Belarus) the thyroid dose was a 230-fold higher than that of average whole body dose (Alexakhin et al., 1992). Destruction of thyroid as well as chronic radiation disease in agricultural animals was observed at the thyroid doses above 200 Gy (Geras'kin et al., 2008). Chronic radiation syndrome (including reduced body mass and fat reserves; increased mass of lymph nodes, liver & spleen; thickening of lower gastrointestinal lining) were observed at doses higher than 2 Gy, while reproductive failure; impaired immune response; reduced mass of offspring were found at a dose level above 1 Gy (Geras'kin et al., 2008). Five months after the accident sheep evacuated from the 30- km zone demonstrated serious hematological alterations in the peripheral circulation (Alexakhin et al., 2004). Leukopenia was reported in 89% of animals and lymphopenia in 90%; 54% of sheep exhibited initial and marked anemia and 34% — serious inhibition of hemopoiesis (Astasheva et al., 1991). Noticeable blood changes, mainly in the form of leukopenia and hyper- and normochromic anemia, were registered in pigs, dogs and cats captured in the 30-km zone in the summer and autumn of 1986. Offspring of highly exposed cows had reduced weight, decreased daily weight gains, disruptions of the hormonal status (Astasheva et al., 1991). The dogs abandoned in the 30-km zone developed alterations typical for a chronic radiation disease in the visceral organs and tissues. Pathologies such as reduced mass of muscular and fat tissue, changes in the liver, kidneys, gut, stomach with hemorrhages and local necroses (liver, kidneys) appeared (Alexakhin et al., 2004).

Mouse-like rodents are the most abundant group of mammals occurring in the vicinity of the Chernobyl NPP. During the fall of 1986, the number of small rodents on highly contaminated plots decreased by a factor of 2 to 10 (Taskaev and Testov, 1999). The numbers of animals had recovered by the spring of 1987, mainly due to migration from less affected areas. Doses of 3–6 Gy resulted in multiple destructive alterations in both the hemopoietic system and the internal organs of mouse-like rodents (Materiy, 1996; Ermakova, 1996; Materiy and Goncharov, 1996). The significantly reduced testes mass as well as irreversibly or temporary sterility in part of males from mouse-like rodents populations was observed (Pomerantseva et al., 1997) at absorbed doses by gonads of 3-4 Gy per month.

4.2.5 Freshwater species

Higher occurrence of the reproduction system alterations as well as reduced viability of progeny of silver carp is observed at dose of 9–11 Gy over 5 years (Belova et al., 1993; Makeeva et al., 1994). Analysis of young pike-perch of the 1986 generation by the number of rays in thoracic fins demonstrated a level of fluctuating asymmetry by a factor of 30 higher than in the control. In phytophilous fish, whose spawn received the major dose from aquatic plants that accumulate radionuclides, failure in the blood system (carp), reproductive system (perch and carp), as well as cytogenetic abnormalities (carp) (Polikarpov and Tsytsugina, 1995) and aneuploid-like patterns in the DNA histograms (catfish) were observed (Dallas et al., 1998).Populations of the some other freshwater species such as pond snail *Lymnaea stagnalis L.* inhabiting heavily contaminated lakes were characterized (Golubev et al., 2005) by increased frequency of cells with micronuclei in hemolymph in comparison with similar species sampled from the not so contaminated lakes. Statistically significant relationships between the severity of cytogenetic damage in the worm population and the number of individuals switching to sexual reproduction were observed at the sites with dose rate on the surface of bottom sediments of 0.3-0.4 mGy/day (Tsytsugina and Polikarpov, 2003).

4.2.6 Genetic effects

In 1986, at the most contaminated sites with external dose of γ-irradiation at the level of 10–20 Gy frequency of enzyme loci mutations was 4–17 times and frequency of aberrant cells 1.5–7.2 times higher than in the control (Fedotov et al., 2006). In most cases the dose–effect relationship was supralinear and the yield of mutations per unit dose was higher at low doses and dose rates (Fedotov et al., 2006). In 1987–1990, an increased yield of cytogenetic disturbances in needles was observed, which declined more slowly than the contamination in the area (Sidorov, 1994). Extra exposure to γ-radiation of Scots pine seeds from the control and chronically treated populations revealed a radioadaptation effect based on the frequency of aberrant cells (Fedotov et al., 2006). An essential role of genomic instability in the long-term consequences of radiation exposure was demonstrated in many studies (Geras'kin et al., 2003). In subsequent years the decline in the radiation background rate occurred faster than the reduction in the mutation rate (IAEA, 2006). Populations of both woody and herbaceous plants showed signs of adaptation to chronic radiation exposure (Shevchenko et al., 1996). From this point of view radioactive contamination may be regarded as an ecological factor that potentiates the action of natural selection by eliminating radiosensitive genotypes. Some genetic effects are still apparent: in 2000-2002, increased frequency of abnormal sperm, partial albinism, and decreased level of antioxidants in blood and liver were observed in barn swallows from Chernobyl (IAEA, 2006).

4.2.7 Post radiation recovery

When the radiation impact on the environment is reduced, the recovery of some natural ecosystems may occur. The change in species diversity of the soil invertebrate communities presented above is perhaps the most obvious published example of community level change and subsequent recovery following the Chernobyl accident. In the first year after the accident, recovery of the resident populations of soil fauna was very slow, but size of populations was increasing due to migration of insects from surrounding less contaminated areas. The abundance of earthworms was about 15% of the control but the presence of their cocoons suggested that they were reproducing even in the most contaminated areas (Sokolov et al., 1993). Within 2.5 years after the accident the population of soil invertebrates was almost completely restored in size but the species diversity of communities in the affected regions, even 10 years after the accident, was only 80% compared to that before the contamination event (Krivolutsky, 1996).

The death of pine stands close to the Chernobyl reactor and the subsequent establishment of grasslands and deciduous trees represent another example. Although, the replacement of coniferous trees by deciduous ones occurred and the ecosystem is not able to return to the state which was before the accident, new forest provides its ecological service to the surrounding populations of living beings (Geras'kin et al., 2008).

Currently, with the removal of humans, wildlife around Chernobyl is flourishing. Forty eight endangered species listed in the international Red Book of protected animals and plants are thriving in the Chernobyl Exclusion Zone, supporting the conclusion that the removal of humans alleviates one of the more persistent and ever growing stresses experienced by natural ecosystems (IAEA, 2006).

4.2.8 Assessment of the acceptable dose levels for biota

As noted earlier, data on effects in non-human species in areas affected by the radiation accident represent an important source of the information for identification of dose levels which might be considered as potentially safe for non-human species. Several publications specifically reviewed Chernobyl related data on biological effects (namely, Sokolov et al., 1993, UNSCEAR, 1996; Fesenko et al., 2005; IAEA, 2006, Geras'kin et al., 2008 and UNSCEAR, 2008) to achieve this goal.

Based on evaluation of 250 references, maximum doses at which effects were not still observed, and minimum doses at which effects were registered, were assessed and dose levels which could be potentially safe for the non-human species i.e., dose values for wild biota at which no effects can be observed (so called critical dose values – CDV_b values), were suggested (Table).

Table: Dose values for non-human species inhabiting the Chernobyl study area at which no effects are expected at the population level, Gy/a (Fesenko et al., 2005)

Biota species	CDV_b	Biota species	CDV_b
Coniferous trees (pine)	0.40	Soil invertebrates	0.90
Herbaceous plants (meadow grasses)	3.00	Phytoplankton	3.00
Herbaceous plants (cereals)	3.00	Zooplankton	2.50
Cattle	0.60 (50*)	Zoobenthos	0.90
Mouse-like rodents	0.40	Fish	0.60

* Dose to the thyroid, G

Several research were recently addressed to assess predicted no observed effect dose rate values for non-human species (IAEA, 1992, UNSCEAR, 1996; Larson, 2008, ICRP, 2009). Although, these data are presented in terms of dose rates (mGy d^{-1} or µGy h^{-1}), and intended for assessingeffects of chronic exposure of wildlife biota, it is of interest to compare these values with data given by the table.

The default screening criterion doses to non-human species assessment within the ERICA Integrated Approach is a dose rate of 10 µGy h^{-1}, to be used for all ecosystems and organisms. This value corresponds to the annual dose of 8.76×10^{-2} Gy a^{-1}. This value was derived from a species sensitivity distribution analysis performed on chronic exposure data in the FREDERICA database using a safety factor of 10. This value tends to be much lower than those presented in the Table. However, having in mind that a safety factor of 10 was used to derive this level, it should be mentioned that the actual predicted no expected effect value is around 1 Gy/year, i.e. is at the same level as given in the report, especially having in mind that the table provides these data for different groups of biota species.

4.2.9 Comparative Radiation Impact on Biota and Man

An important trend in the evolution of the current system of radiation protection is the development of principles to ensure protection of non-human species from ionising radiation. Over the last decades the scientific grounds of radiation protection of biota are based on the postulate originally formulated in ICRP Publication 26 (ICRP, 1977) “if radiation standards protect man, then biota species are also adequately protected from ionising radiation”. This approach to radiation protection has received wide acceptance in the last quarter of the XX century and been reflected in legal documents on protection of the environment of many countries (IAEA, 2000). Contribution to the evaluation of the correctness of this thesis can be made by comparing exposure doses to human and non-human species found in the same area, in particular in areas affected by the radiation accidents. Such assessments were made for the 30 km zone of the Chernobyl NPP (Fesenko et al., 2005). For the study area within the 30 km ChNPP zone doses to 10 biota groups and the population (with and without evacuation) were calculated. Ratios of these doses to the dose levels considered as save, namely, dose levels for wildlife biota given in the Table. As for the human beings, dose levels recommended by the ICRP for the emergency exposure and for existing exposure (1mSv a^{-1}) were taken. It has been found that in 1986 (early period after the accident) some biota species were affected more seriously compared to man. However, it was also found that for the long term after the accident radiation safety standards for man are shown to ensure radiation safety for biota as well. Additionally, these results have shown that the soil invertebrates and coniferous trees should be regarded as critical environmental species under the conditions of the Chernobyl accident (Fesenko et al., 2005).

4.3 Kyshtym accident in September 1957

The accident occurred in September 1957 - at a period of low radiation sensitivity for many biota species, when in most plants the metabolic processes began noticeably to slow. Rather long duration of the period of physiological dormancy (up to 7-8 first months after the accident) had mitigated the severity of “acute” radiation impact. Therefore, total dose during first year after the accident was used as an indicator of the radiation impacts. The period of the first 3-5 years was followed by a phase when exposure of the non-human species was much lower compared with the “acute” phase because of decay of relatively short-lived radionuclides as ^{95}Zr (24.9% of initial radionuclide composition), ^{144}Ce (66% of initial radionuclide composition), and dominant contribution of more long-lived ^{90}Sr (5.4% of initial radionuclide composition) (Avramenko et al., 1997)[1]. Contributions of β-irradiation to the total radiation dose accumulated by plants and animals exceeded 90 % (Tikhomirov et al., 1993). Compared to the "acute" period, annual absorbed doses in 1960-1980s dropped approximately 1,000-2,000-fold in pine crowns, 200-300-fold in grass, 100-fold in birch crowns, 10-30-fold in soil invertebrates (Sokolov et al., 1993). Thus, following the Kyshtym accident, environmental effects were specific to 2 distinct time periods 1957-1958 (acute stage) and from 1958 (recovery stage). The decrease of dose rates was not as fast as in areas affected by the Chernobyl accident because ^{95}Zr and ^{144}Ce dominated (91 %) in the activity of radionuclides released to the environment (Romanov, 1997). The spatial distribution of radionuclides in the soil was characterized by the rather narrow trail with explicitly expressed axis (maximum contamination density), which is gradually decreasing with distance from a source and sharply decreased in a transverse direction at both sides from the axis of the trace. Such pattern of spatial radionuclides distribution provided opportunity to study environmental effects at the sites with very similar environmental characteristics but, at the same time, with very different environmental contaminations and doses to wildlife species. Unfortunately, the research of the biological effects in the area affected by the Kyshtym accident, named EURT (East-Ural Radioactive Trace (Trail)), had started only after 1962 and this resulted in additional uncertainty in interpretation of the observed effects (Alexakhin et al., 2004).

4.3.1 Forest trees

The effects of radiation damage of coniferous trees were observed as early as in the spring of 1958 in pine trees, when part of buds did not burst and those survived developed short bundles of sprouts. By autumn 1959, pine stand died completely within the area with contamination density of ^{90}Sr at the level of 6.0-7.0 $MBqm^{-2}$(Tikhomirov and Romanov, 1993). Deciduous trees (birch) proved to be more radioresistant and visual effects of radiation damage were observed only in the most affected area. Thus, some 50% of trees died at the sites with ^{90}Sr contamination density around and higher than 100 $MBqm^{-2}$ (Alexakhin et al., 2004). The main radiation effects observed were delay or losses of bud and delayed sprout development and subsequent dying of crowns. The absorbed doses over the "acute" period responsible for severe damage of pine trees ranged from 5-10 Gy for needles and from 2-4 Gy for buds. Lethal effects were observed for forests exposed to doses of 20-40 and 10-20 Gy, respectively (Tikhomirov and Scheglov, 1994). In birch trees, the absorbed dose in the bud apical meristem that induced partial drying of trees reached 100-150 Gy and full death of the entire birch stand could be found at the "acute" dose of 200 Gy (Tikhomirov and Romanov, 1993).

4.3.2 Herbaceous plants

In terms of the geometry of the exposure of the generative tissues, plants may be classified into different groups, depending on the location of winter brood buds relative to the contaminated soil surface: phanerophytes – trees and shrubs with brood buds located well above the ground; chamaephytes, i.e. species such as bilberry shrubs, with brood buds wintering in the snow layer near the surface; hemicryptophytes (such as reed grass) with brood buds on the soil surface; cryptophytes (meadow grass, fescue, speedwell, thistle) with brood buds in the soil layer and therophytes – species with winter seeds. It is clear from the discussion above, that such features of the plants development may result in very different radiation effects (Smirnov, 1993).

[1]Fractions for radionuclides include progeny.

In the first vegetation period following the accident change in the structure of phytocenosis was observed demonstrating disappearance of therophytes and initial decline in the cumulative stock of the above-ground biomass approximately by 30% at the sites with contamination density by ^{90}Sr of 37 MBq m^{-2} and higher. The ratio of the projective cover between different vital groups exhibited in the first year the prevalence of hemicryptophytes (nearly 70%), although they are the species which received the highest exposure doses. Cryptophytes demonstrated suppression of growth of the projective cover with time compared to the initial value. 5-6 years after the accident, a considerable reduction was observed in the share of projective cover in hemicryptophytes (obvious result of damage to reproductive organs) and increase in the share of some cryptophytes (Sokolov et al., 1993; Alexakhin et al., 2004). In the following 5-6 years a slow recovery of the phytocenosis had been noticed. It had, however, failed to achieve the state of a pre-accidental phytocenosis. It is seen that this threshold corresponds to dose absorbed by plants of about 30-50 Gy. As the absorbed dose increases, an unambiguous change in the structure of phytocenosis arises – the contribution of hemicryptophytes declines and, accordingly, grows that of cryptophytes (Smirnov, 1993).

4.3.3 Soil Invertebrates

Research was started only in 1969 in birch forest on the site with average absorbed doses in the top 1 cm layer of soil of 5-12 mGy a day (data based on thermoluminescent dosimetry) (Smirnov, 1993). Although, the research started following 12 years after the accident, effects to invertebrates on the affected sites had been revealed. It has been found that the number of soil mesofauna populations on the experimental plot in 1969-1971 was twofold lower than in the uncontaminated sites (Krivolutsky, 1983). The highest decrease in the number of animals was reported for saprophages and polypodies and earthworms, which were practically lacking at the contaminated site (Krivolutsky, 1983). Such a sharp decrease is mainly related to relatively low mobility of saprophages, which during their life cycle are in a permanent contact with the most contaminated environment. The number of mites was not significantly different from the similar uncontaminated sites control; the number of mites fell 4 times, illustrating a tendency for simplification of the community of soil invertebrates (Alexakhin et al., 2004). The second study performed 30 years after the accident showed that even after such a long lapse of time the community of invertebrates had not been restored to the state close to that in the uncontaminated areas, and the numbers of the animals under study reached 40-70% of those specific for similar uncontaminated sites (Krivolutsky, 2000).

4.3.4 Vertebrates

Lethal effects of mouse-like rodents were observed at a dose of about 10 Gy. In these areas considerable reduction of population and substantial changes in the species structure of the community of mouse-like rodents were found (Ilyenko, 1974). However, the population numbers in that area could be restored through migration of animals from adjacent, less contaminated sites, and the populations found by the beginning of observation (1962) might have been formed from parents with different levels of irradiation (Alexakhin et al., 2004). Statistically significant radiation effects were registered in both organism and population levels inhibiting sites where the average annual dose to the red bone marrow amounted to 4-8 Gy in 1962 and 0.6-1.0 Gy in 1981 (Ilyenko and Krapivko, 1993). The substantial changes were reported for mass and size of animal and some animal organs as spleen and liver, for craniologic performance, blood and other morphophysiological parameters. The population data indicate a decrease in the life span in population of exposed animals, decrease in survival rate and increase in death rate during wintering, reduction in the reproductive potential, increased incidence of hemo- and ecto parasitism (Ilyenko, 1974; Ilyenko and Krapivko, 1993).

4.3.5 Freshwater species

Fish of local species in the most affected lakes (Urus-Kul and Berdenish) could have received during the “acute” period doses of 10-40 Gy (Alexakhin et al., 2004). It may be suggested that such absorbed doses could induce a decrease in productivity of fish populations due to radioactive damage to incubated or laid spawn. No direct evidence is, however, available. Observations of the status of fish populations in these basins that commenced several years after the accident (first in 1960) showed no reduction in fish productivity that suggested an idea of possible biological compensation of radioactive damage, if any, by the beginning of observation.

At the same time, research performed at the Urus-Kul Lake in the 1980s, where the doses to freshwater biota ranged in 1957-1958 1.0-40.0 Gy during first year after the accident, demonstrated a 1.5-2.0-fold decrease of biodiversity for the communities of the species related to all trophic levels, namely, for phytoplankton, zooplankton and benthic invertebrates (Fesenko, 1983). The degree of the development of most species and their biomass per m^3 in the affected lake was also much lower compared to those measured for similar lakes located outside of the contaminated zone (Fesenko, 1983).

4.3.6 Genetic effects

Genetic effects were observed in many non-human species inhibiting the EURT area (Alexakhin et al., 2004). In particular, in 1961, birch and pine trees in the EURT area demonstrated increased rates of chromosome aberrations, the presence of abnormal pollen, decrease in tree development, structural anomalies, and decrease in cellular mitotic index (Shevchenko et al., 1993). For herbaceous plants (*Centaurea scabiosa*) on the plots with the absorbed dose rates of 6 and 12 mGy a day measured in 1982, the frequency of alterations in the enzyme electrophoretic mobility was 6.6 and 4.5%, respectively, compared to 0.4% on the non-contaminated sites. Similar effects were observed for chlorophyll mutations (Shevchenko et al., 1993). A statistically significant increase in the aberration rate in the first mitosis of seedlings was also found to be characteristic only for the sites with doses to plants (*Crepistectorum)* of 100-180 mGy per year (Shevchenko et al., 1993). At the same time, no statistically significant effects were found for plants exposed to radiation dose of 2-50 mGy per year. Results of many studies with a provocative (additional) irradiation of plant seeds sampled in 1967 and in 1976 from the sites with doses 20-200 Gy strongly suggested that there has been some genetic adaptation of natural populations to the increased radiation enhancing their radioresistance compared to the initial one (Alexakhin et al., 2004). These findings were confirmed in 1995 in a similar experiment with additional γ-treatment of seeds of military grass (Shevchenko et al., 1998). Genetic effects were also observed for various mammal and fish species. Increased rate of chromosome aberrations and aneuploid karyotypes in cells of bone marrow was reported for mouse-like rodents. No karyotypes were, however, detected that carried any hereditary alterations, which could have pointed to cumulating of the genetic effect in generations (Shevchenko et al., 1993). Results from the studies of fish populations inhabiting the contaminated EURT water bodies allowed the conclusion that during chronic exposure a dose rate of 10-30 mGy a day likely did not provide statistically significant effects (Alexakhin et al., 2004).

4.3.7 Post radiation recovery

As some time passed (2-3 years and later), the affected forest ecosystems began to recover; however, if the recovery of birch forests was possible through sprouting of survived trees and viable stumps, the restoration of pine in the perished stands occurred much later through seed propagation from the adjacent territory (Tikhomirov and Romanov, 1993). The all-perished pine stands had never been restored. These species were replaced by either birch or mixed pine-birch forests with low percentage of pine. Thus, in the following 5-6 years a slow recovery of the phytocenosis had been noticed following both radiation accidents considered in this report. It had, however, failed to achieve the state of a pre-accidental phytocenosis. It may be suggested that the observed within 11 and 30 years after the accidents radiation effects of wildlife biota resulted from their intensive irradiation during the "acute" period, first of all, due to relative "immobility" of most populations "tied" to a small plot and not actually migrating outside it, as well as from slow rates of biological post-accident recovery, including the lack of inflow of "new blood" from other areas.

4.4 Secondary ecological effects

Radiation effects at a biocenotic level begin from the dose that induces disappearance of the most radiosensitive species (e.g., death of coniferous trees). The disturbances of ecological relationships are induced by the following factors: (1) changes in microclimatic and edaphic conditions (in affected coniferous forests, because of improvement of both light and mineral nutrition condition, more radioresistant deciduous species actively develop); (2) disturbances in the synchronism of seasonal phases in the development of ecologically connected groups of organisms (shifts in the time of leaves blossoming and eggs of leaf worms hatching); (3) imbalance in food interrelations between consumers and producers (decrease in food resources as a result of irradiation); (4) changes in biological pressure as a result of species differences in radioresistance (changes towards prevalence of more radioresistant species in meadow phytocenoses; disturbances in both host-parasites and predator–pray relationships); (5) induced by ionizing radiation changes in affected communities make open ecological niches for immigration of new species (Tikhomirov et al., 1994; Alexakhin, et al., 2004, Geraskin et al, 2008).

4.5 Conclusion

The severity of radiation effects was strongly dependent on the dose received in the early period after the accident. The most exposed phytocenoses and soil animals' communities exhibited dose dependent alterations in the species composition and reduction in biological diversity. In most cases the dose–effect relationships were nonlinear and the mutation rates per unit dose were higher at low doses and dose rates. In subsequent years, a decline in the radiation background rate occurred faster than reduction in the mutation rate. Based on the results of these studies, doses resulted in ecological and biological effects were assessed for various species, namely, forest trees, herbaceous plants, mammals, soil invertebrates, aquatic species etc. It has been found that even for severe radiation accidents, such as Chernobyl and Kyshtym accidents, the impact on non-human species at the population level was observed only on relatively small areas and highly impacted populations of the biota species were recovered during 2-3 years after the accident, except for coniferous trees. Some ecological consequences, such as the disruption of the relationships between the various species within the ecosystems can persist for long time. Genetic effects can also be observed in many areas with high contamination levels and their consequences are not still fully understood. Such effects require evaluation and special attention. Overall, evaluation of the effects of biota in areas affected by the radiation accident do not show a need in introduction of new biota related options in the regulation intended for emergency preparedness and response. At the same time, the potential impact on biota should be carefully assessed and taken into account to provide a proper ecological management on the affected areas.

4.6 References

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5 STAKES AND LIMITS OF BIOREMEDIATION OF RADIONUCLIDES

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5.1 Introduction

Bioremediation is a branch of biotechnology using biological processes to overcome environmental problems. Biological agents used for bioremediation may be organic molecules such as DNA, antibodies, dead or living organisms (bacteria, alga, fungi, or green plants). More specifically, phytoremediation consists of the use of green plants to decontaminate polluted soil or water. Contrarily to chemical pollutants such as PCB, TNT, TCE that can be metabolized by microorganisms or plant roots, radionuclides and heavy metals cannot be directly degraded. Thus, bioremediation strategies concerning radionuclides will essentially consist of their stabilization/mineralization to decrease their bioavailability through a change in their redox state or in their *in situ* extractionusing plants or their *ex situ* treatment using plants or bacteria for contaminated effluents. Compared to traditional methods used for cleaning radionuclides contaminated soils that disrupt soil structure, degrade the landscape and reduce soil fertility and productivity, bioremediation is considered as a "green" and "eco-friendly" technology. It is a cost effective technology that can be applied on large surfaces. In this chapter different examples of bioremediation strategies of heavy metals and radionuclides will be presented and the positive and negative aspects debated.

5.2 Bacterial biomineralization/biostabilization of actinides

The inventory of radionuclides generated during the past 60 years resulting from nuclear reactor operating and nuclear power plant accidents includes ^{237}Np, Pu isotopes, Am, ^{3}H, ^{14}C, ^{85}Kr, ^{90}Sr, ^{99}Tc, ^{129}I and ^{137}Cs in addition to ^{235}U from nuclear fuel. Bacteria are used i*n situ* or *ex situ* for the bioremediation of some of these radionuclides. *In situ*, bacteria immobilise radionuclides through biosorption at the cell wall (Pollmann et al., 2004; see Fig 1) which has generally the capability to bind cations. In bacteria bioaccumulation also occurs followed by reduction, operating at the surface of bacteria and resulting in a precipitation of radionuclides (Lloyd, Renshaw, 2005).As an example, bacteria such as *Shewanella oneidensis* possess an electron chain able to transfer electrons from Fe(III) to U(VI). Such mechanism results in the reduction of U(VI), soluble and highly toxic, to U(IV) insoluble and less bioavailable (Liu et al., 2009). The same species and *Geobacter metallireducens* GS-15 are also able to reduce plutonium, Pu(IV), in the stable form of reduced Pu(III). Transmission electron microscopy images of the solids obtained from the cultures after the reduction of Pu(VI) and Pu(V) show that the Pu precipitates have a crystalline structure (Icopini et al., 2009). Bacterial hydrolysis of organophosphates operating through the synthesis of phosphatases by bacteria precipitates SrII, AmIII, ThIV, PuIV and UVI as phosphates. In these conditions EXAFS spectra identify the uranyl phosphate precipitate as an autunite/meta-autunite group mineral (Beazley et al., 2007).Reoxidation along time of reduced radionuclide forms by other bacteria species as observed in the presence of oxygen or high nitrate concentrations can be a problem (Istok et al., 2004) but it can be solved by adding or stimulating already-present communities of Iron(III) or sulfate reducing bacteria (Wall & Krumholz, 2006). Biomineralization is stimulated in situ by injecting nutrients, offering a potential remediation mechanism using indigenous microorganisms. Such strategy has been applied on US DOE FRC in Oak Ridge, Tennessee, using a series of wells. When ethanol, glucose or acetate were added as electron donors, a rapid denitrification was observed together with Tc(VII) and U(VI) reduction when the electron donor resulted in Fe(III)-reducing conditions (Istok et al., 2004). Such approach was also used in Ashtabula, Ohio, to treat a contamination of soil by U(VI) resulting in natural and slightly enriched uranium leaching into the soil. In that case, the metabolic properties of the anaerobic bacterium *Clostridium* sp. was exploited to solubilize and precipitate radionuclides (Francis et al., 1994), and injections of polylactate ester or acetate were found to accelerate the process. Bacteria are also directly used on the field for bioleaching or biostabilization by spraying the contaminated zone with water and nutrients at a pH that will favor the desired process. Acidification is already used to promote the bioleaching of U from minerals containing 0.05% to 0.15% U_3O_8, a U content insufficient for classical mining. Changes in the nutrient composition and pH, and eventually introduction of specific bacterial species could result in biomineralization of U(VI) or other oxidized actinides.

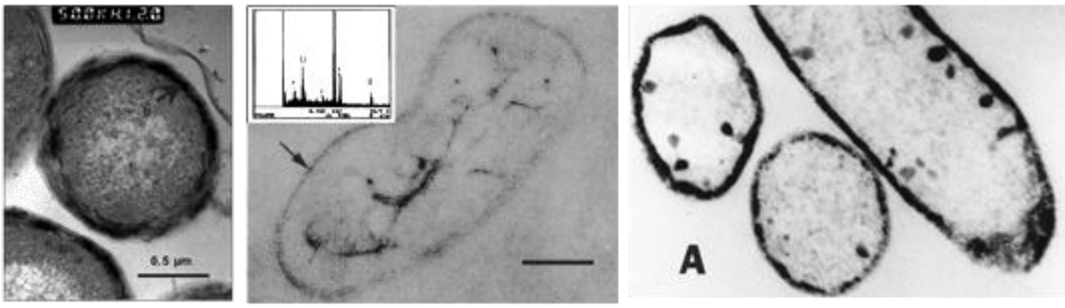

Figure 1: Interaction between radionuclides and bacteria, left panel uranium accumulated at the cell surface of Bacillus sphaericus JG-A12 (Merroun et al., 2005); center panel, granules of U associated with polyphosphates in the cytoplasm of Pseudomonas CRB5 (McLean& Beveridge, 2001; right panel, Tc Reduction and accumulation at the cell wall of Desulfovibrio desulfuricans (Loyd et al., 1999)

Such processes are also applied *ex situ* to treat contaminated soils or effluents. As an example, 300 m^3 composite ceramic tanks are used at Perring, South Africa, for the cultivation of a thermophilic bacteria to promote gold extraction from (arseno)pyrite. Bacteria are now integrated in various polymers such as polyurethane foam or ceramic filters and included in a filtration process to promote the reduction of actinides and their immobilization.

5.3 Phytoremediation of radionuclides

Phytoremediation covers different techniques (summarized in Fig. 2) using plants to immobilize or extract radionuclides from soil or water. Phytostabilization is the simplest technique used when erosion by wind or water could lead to a large dispersion of the contaminant. It consists in the (re)establishment of a vegetal cover allowing to structure the soil and to limit leaching of contaminants to the subsurface water. Such solution has been used to stabilize Arsenic leaching from an 11 million tons heap of waste resulting from gold mining at a flow rate of 300 kg/y in the subsurface water and 1.300 kg/y in a neighboring river. The use of steel shots (an industrial material containing 97% metallic Fe) to immobilize As, associated with plant species resistant to this metalloid, allowed to limit As content in leaching water below the local regulation for drinking water (www. difpolmine.org). Rhizofiltration uses the properties of plant roots with a high affinity for cations to withdraw. It has been demonstrated as simple and efficient in the treatment of U contaminated water by the pioneering work of Timofeeva-Ressovskaia in 1963 illustrated Fig. 3. This process was applied in Ashtabula by a private phytoremediation company, Phytotech. Bioaccumulation coefficients based on the ratios of U concentrations in the roots of sunflowers *vs* U concentrations in the aqueous phase reached 3 10^4 (see a review in Dushenkov et al., 1997). Using this technique in a pilote-scale rhizofiltration system, U content in contaminated water (21−874 µg/L) was reduced to less than 20 µg/L, the EPA Water Quality Standard, before discharge to the environment.

Phytoextraction is a technique which consists to grow plants to extract contaminants from the soil aim to their natural affinity for soil cations at the level of the roots for nutrient uptake and the potential of translocation of the pollutant to the shoot for an easier harvesting. It is an interesting technique since the contaminant is definitely withdrawn from the environment.However the yield of remediation is highly dependent on many parameters such as contaminant bioavailability, soil structure, depth of contamination, climatic conditions, plant species which make every case a new situation. The final yield of extraction depends on the transfer coefficient between the soil and the plant and the biomass production. The transfer coefficient is frequently the most limiting factor which led to decades of phytoremediation to reach a reasonable result. Additional treatments accelerate the process such as a judicious choice of plant species and the amendment of soil with contaminant chelators (citric acid, EDTA or DTPA). In the frame of the EEC PhyLes project, different species were used for the phytoremediation of a soil highly contaminated by lead (300-1200 mg kg^{-1}) (http://www.phyles.ge.cnr.it). The use of a specific cultivar of an efficient plant species (*Brassica juncea*) rather than sunflower and the treatment of soil with K_2EDTA have allowed decreasing the estimated duration of treatment from 300 years to 20 years. A treatment with citric acid was also found to highly increase the translocation factor of U by 30-50 fold using Indian mustard or ryegrass (Vandehove and Hees, 2004) while DTPA was able to increase a hundred fold Pu uptake by sunflower or Indian mustard.

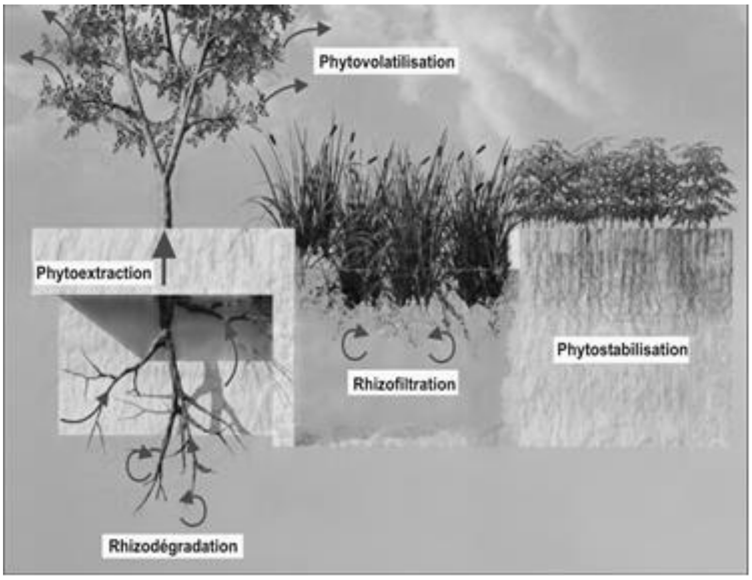

Figure 2 : Different techniques of phytoremediation (Vavasseur et al., 2009)

However, one drawback of such treatments with chelators is their eventual persistence and toxicity in soils and a higher leaching of the contaminant towards subsurface waters.

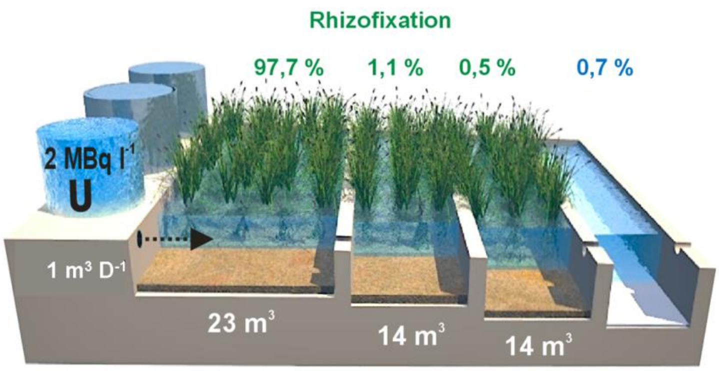

Figure 3: Rhizofiltration of uranium in successive ponds according to Timofeeva-Ressovskaia EA, 1963

5.4 Remediation of Cs, the post-Fukushima context

At the recent symposium "*International science symposium on combating radionuclide contamination in Agro-soil environment*" held in March 2012 at Koriyama in Fukushima province, a general landscape of soil contamination was drawn consisting mainly in ^{137}Cs and ^{134}Cs. The main applied remediation techniques were traditional methods: (i) removal of the most contaminated soil surface resulting in a 80-95% decrease in contamination (ii) deep plowing in order to limit the exposure of persons and the uptake by plants and (iii) potassium fertilization, potassium being known to compete with Cs to enter plant roots (Zhu &Smolders, 2000). The main problem is the very large contaminated area. It is admitted that a total area of about 1800 km^2 should be treated to reach a clean-up of the soils limiting human exposure due to radioelements spread by the Fukushima Daiichi nuclear power plant below 4 mSv/year. Cleanup of such a surface through the removal of a 5 cm soil surface would represent some 10^8 m^3 of waste contaminated with^{137}Cs and ^{134}Cs. At the moment there is no industrial means to stock and treat such an amount of contaminated material. In such a context is bioremediation of radionuclides an option? The first observation is that even if bioremediation has many limits there is no other alternative technique for large scale remediation allowing preservation of the soil properties and of the landscape. One property of Cs is to be poorly mobile in soils. The analysis, in 1992, of a contaminated soil in Ural following the Kyshtym accident in 1957 in Russia, shows that most of ^{137}Cs stays in the ten first centimeters of the soil. One reason is that Cs interacts strongly with clay composed of alumino-silicate which form a network composed of layers spaced by negatively charged cavities which tightly chelate cations. Cs, having low hydration energy, replaces potassium and ammonium at these sites and stays for years chelated in clay. Since ^{137}Cs stays in the first horizon of the soil it will be continuously an uptake by plant roots, promoting a recycling of the contaminant at the soil surface.

What is the potential of phytoremediation to treat ^{137}Cs contaminated soils? There have been some assays of phytoremediation of ^{137}Cs. A leakage from a nuclear reactor at Brookhaven has resulted in a soil contamination of 0.02 – 1.4 kBq/kg of ^{90}Sr and 0.02-110 kBq/kg of ^{137}Cs. Three plant species were tested on this site for their potential of phytoremediation, *Amaranthus retroflexus*, *Brassica juncea* and *Phaseolus acutifolius* (Fuhrmann et al., 2002). Among the three species only *A. retroflexus* was able to uptake both radioisotopes at a reasonable level. From this study it was estimated that to reach a fifty per cent decrease of the contaminants six years would be necessary for^{90}Sr and sixteen for ^{137}Cs. The duration of treatment is certainly the major limit of phytoremediation. However it can be shorten by adding by soil treatment such as addition of ammonium that competes with Cs at the chelation sites in soils. White and col. (2003) have reported that an addition of 100 kg ha^{-1} of

NH_4NO_3 resulted in a three to five fold increase in ^{137}Cs uptake by *Secale cereale* and *Brassica oleracea.* There is a large diversity between plant species in their ability to uptake Cs, Broadley and Willey (1997) compared 30 taxa and found the maximum differences between *Chenopodium quinoa* and *Koeleria macrantha* of 20-fold in Cs concentration and 100-fold in total Cs accumulated. Thus, there is likely in biodiversity a large potential to find more efficient species for the phytoremediation of Cs. Another approach is the use of molecular biology to unravel the molecular components involved in Cs uptake and translocation in plants. A high-affinity K^+ transporter, highly expressed at the level of the root epidermis during potassium starvation has been found to contribute to around fifty per cent of Cs uptake in the model plant in genetics *Arabidopsis thaliana* (Qi et al., 2008). It is likely that orthologs of this transporter are important in Cs uptake by other species and that the degree of expression of such genes could be an indicator of the potential of a particular species for the uptake of Cs, giving to these molecular markers a predictive aspect on Cs uptake by plant cultures. Other Cs transporters remain to be identified and their discovery will open the gates of plant biotechnology or traditional selection to increase Cs uptake and translocation to increase the yield of Cs phytoremediation. On another side the identification of the molecular actors of Cs uptake and translocation *in planta* will allow to select plants with lower ^{137}Cs content in their edible parts.

5.5 Stakes and limits

In this short review I tried to draw a general landscape of bioremediation techniques and to show their potential interest for the remediation of the most encountered radioactive contamination in the environment. The major advantage of bioremediation technologies is certainly that they can be applied *in situ* with a lesser impact on the environment than the classical physical-chemical techniques of remediation. One interesting point is that a technique like phytoremediation can be applied tovery large contaminated areas with moderate to low contamination and that there is at the moment no other alternative to remediate the most extended contamination such as ^{137}Cs issued from nuclear power plant accidents. The cost of bioremediation is also generally far below than that of classical remediation techniques and, coupled to an economic activity, preserves human activities in weakly contaminated areas. Recycling of contaminated biomass through the production of energy is economically interesting and creates an activity in areas where agronomy is impaired. It has been recently evaluated at the issue of a phytoremediation program consisting of lead decontamination using poplars (PHYTOPOP project), that the energy generated by burning the biomass and the use of a modern technology of cogeneration generate for the farmer a salary equivalent to 70% of that obtained by growing wheat.

It is clear that bioremediation has limits and that in the case of highly contaminated areas the duration of treatment would be prohibitive.A second limit is that each bioremediation project is different from the previous according to the high number of parameters that need to be taken in account: level of pollution, bioavailability of the contaminant, nature of the soil, climate, biotic and abiotic conditions… It results from it a difficulty to draw a general strategy due to the diversity of parameters. This is certainly a strong brake at the moment in the large development of bioremediation techniques. Another braking point is that the production of contaminated biomass needs the creation of a concomitant infrastructure to burn this waste in safe conditions meaning the creation of adapted nuclear installations. The last negative point is the duration of treatment, mainly for phytoremediation that should be shortened in the future through the creation of high potential cultivars in order to obtain acceptable results in less than a decade. To reach such objectives more basic research is needed and large scale experimentations should be encouraged together with a change from a pragmatic approach on the field to an actual engineering of bioremediation strategies.

It is reasonable to think that in the next future a mixed technology of remediation will be applied to recover a safe environment mixing classical techniques to treat the limited hot spots of contamination while bioremediation could be applied to extended regions moderately contaminated.

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6 ETHICAL ASPECTS OF PROTECTION OF THE ENVIRONMENT FROM IONISING RADIATION

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6.1 Introduction

Many of the international organisations involved in the development of tools for assessing the ecological impact of ionising radiation have recognized that producing a practical and coherent system of radiological protection raises a number of ethical questions. The IAEA produced a report on “The Ethical Aspects of Protection of the Environment from Ionizing Radiation” (IAEA, 2003), which addressed the cultural, scientific and social influences on environmental ethical worldviews, as well as links to political protection principles such as sustainability and biodiversity (Figure 1). These aspects have also been addressed in IUR and referred to in the ICRP publications on environmental protection (ICRP, 2008; IUR, 1999, 2012). All approaches appreciate the diversity in ethical and cultural views on valuing the environment, and recognise that this diversity should be respected in environmental protection frameworks.

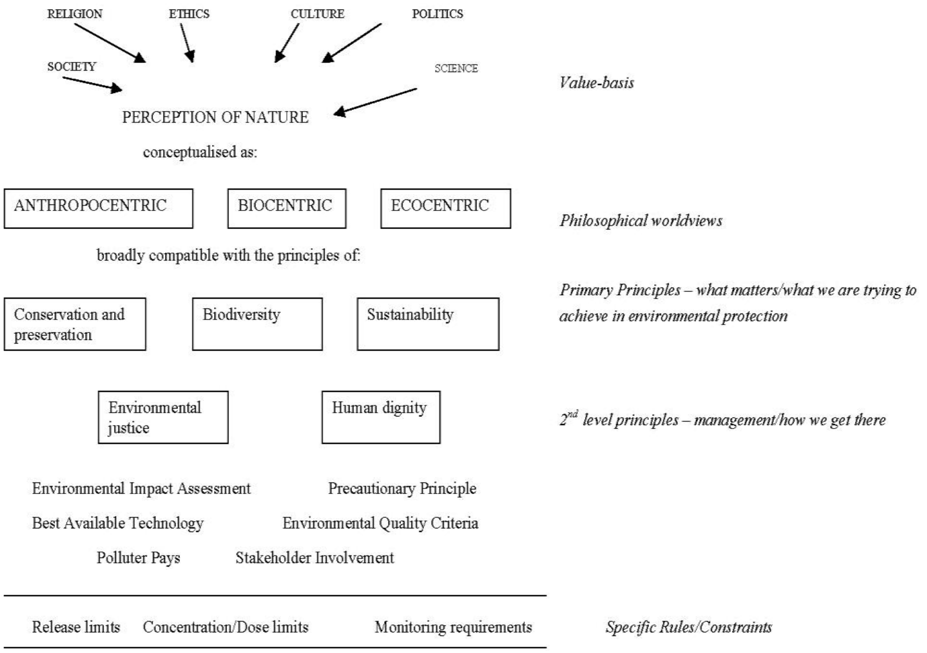

Figure 1: Links between philosophical worldviews, perception of nature and environmental protection principles (see IAEA, 2003 for further elaboration).

The present paper first summarises some of the main ethical issues concerning the protection of the environment from radiation, largely based on the IAEA report and previous papers (e.g., Oughton, 2003), and then looks at more recent developments on environmental protection in radiation risk assessment. The first part gives an overview of different philosophical worldviews on valuing the environment in a context of radiation risk. The second part addresses some of the more recent challenges to proposed environmental protection frameworks, including practical applications following the Chernobyl and Fukushima accidents. The final part of the paper offers some recommendations for how ethical evaluation can aid in producing a robust and transparent approach to protection of the environment from ionising radiation.

6.2 Valuing the environment: Philosophical theories

Environmental ethicists have been debating the matter of why one attaches value to the environment for a number of decades (Rolston, 1988; Sterba, 1994). Central philosophical issues include the question of moral standing and 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). Although environmental ethics is a relatively young field within philosophy, a number of distinct views on this question have emerged. In contemporary environmental philosophy, the most fundamental source of divergence arises between the anthropocentric and the non-anthropocentric view. An anthropocentric ethic (literally human-centred) alleges that only humans have moral standing and that environmental degradation matters only in so far as it influences human interests (Norton, 1988; Bookchin, 1991). Proponents of a non-anthropocentric ethic reject this assumption, and attribute moral standing either to other living organisms or to the ecosystem as a whole, contending that effects on the environment matter irrespective of their consequences for humans. Although a variety of different views can be found in the literature, the biocentric and ecocentric outlooks are arguably the two main contenders.

6.2.1 Biocentricsm

Proponents of biocentrism (literally “life-centred”) assert that individual life-forms other than humans can have moral standing, and should be respected for what they are—not only because they affect the interests of humans. Different biocentric views exist as to which criterion forms the basis for moral standing, and what hierarchy (if any) exists between different species. But all views derive moral value from some biological characteristic of individual members of species, such as sentience or the ability to feel pleasure or pain (Singer, 1991), self-consciousness (Regan, 1980) or inherent worth or a “good of their own” of all living things (Taylor, 1986; Goodpaster, 1978).

Biocentric outlooks can be found within supporters of both utilitarian and deontological theories of ethics. Utilitarians can include the welfare, interests or preference satisfaction of animals in their calculations; deontologists can find room for rights of or duties to animals. The Australian utilitarian philosopher, Peter Singer, is one of the most influential proponents of animal ethics, and suggests that sentience represents the fundamental criterion for moral standing (Singer, 1991). Welfare or well-being matters for any life-form with the capacity to feel pleasure and pain. In this he advances an idea first proposed by Bentham when considering who or what should count in a utilitarian evaluation: “The question is not, Can they reason? Nor, Can they talk? But, Can they suffer?” (Bentham, 1789). Although the calculation may allow a hierarchical weighting of different species, human interests are not inalienable and can be outweighed if the amount of suffering caused to animals is large enough.

Deontologists might suggest that the notion of rights and duties should be extended to the animal or biological kingdom. One of the strongest proponents of animal rights, Tom Regan argues that like humans, some non-human animals have consciousness or self-awareness and a capability for reasoning (Regan, 1980), and some form of rights attribution to animals can be found in national legislation (e.g., New Zealand). However, critics have claimed that the debate around giving “rights” to non-human species or indeed to whole ecosystems, is a futile response to the increasing tendency of human society towards environmental destruction. They draw parallels with the way that human rights have emerged as a well-meaning, and yet, to date, depressingly ineffective way of counteracting the modern day atrocities of warfare or racism (Bradford, 1993). The critique harks back to Bentham's notorious claim that "natural rights is simple nonsense; natural and imprescriptable rights, rhetorical nonsense—nonsense upon stilts (Bentham, 1824)". Nonetheless, human rights are being perceived as important by an increasing proportion of the world's population, and the possible future extension of these rights to other species is not easily dismissed.

Because biocentrism focuses on individuals rather than the diversity of species, these various outlooks have also been described as an "individualistic" environmental ethic (Sagoff, 1984; Rolston, 1991). In practical policy-making, biocentric outlooks have had the greatest influence in issues of animal welfare and the use of animals in research (Sagoff, 1984). The ICRPs Reference Animals and Plants (RAP) approach (Table 1), is consistent with a biocentric methodology for assessing radiation effects on individual non-human species. Although, as discussed below, this does not necessarily make it a biocentric value-basis for protecting those individuals. The idea of including impacts on animals in radiation protection optimization is also compatible with a broadly utilitarian approach. In this case optimization would include both the direct impacts of radiation on non-humans, as well as the more general (and often more damaging) consequences for the environment of reducing doses to human (see Oughton et al, 2004 for examples of the environmental and animal welfare side-effects of accident remediation).

Nevertheless, optimization in radiation protection rarely considers exactly why one is bothered about environmental impacts, and there can of course still be disagreements on which species and which effects matter. For example, Singer's criterion of sentience only encompasses vertebrates, whereas Paul Taylor suggests that all living organisms are equal moral subjects (egalitarian biocentrism) since each has some goal to its existence (Taylor, 1986). Note that for any biocentric view, as soon as the ethically relevant factor for assigning moral standing diverges from the "speciesist" criterion of simply being human to some trait such as rationality, consciousness or sentience, one is faced the problem of how to deal with those members of the human species that, due to some force of circumstance (accidental or otherwise), might be considered to rank lower than the higher animals.

6.2.2 Ecocentrism

Supporters of an ecocentric philosophy claim that the diversity of species, ecosystems, rivers, mountains and landscapes can have value in themselves, irrespective of the consequences on humans or other individuals of non-human species. All ecocentrics attach particular value to the diversity, dynamics and interactions within healthy ecosystems, but differ in their views on the causes of, and proper solutions to, modern environmental problems. Callicott (1979, 1989) and Næss (1974) both see the Western, instrumental view of nature as a main source of environmental problems. Ecofeminists suggest the problem lies in the history of male dominance and sexist oppression of females (Warren, 1990); others that it stems from the social and economic structure of society (Bookchin, 1991). Many link problems to theJudeo-Christian tradition, and, more specifically, in the Biblical quotation (e.g., White, Science, 1967):

> "Let us make man in our image, after our likeness: and let them have dominion over the fish of the sea and over the fowl of the air, and over the cattle, and over all the earth, and over every creeping thing that creepeth upon the earth" (Genesis 1: 26–30)

Although other philosophers have pointed out that the bible also contains examples of human obligations to respect nature (Ariannsen, 1996).

Most ecocentrics claim 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 (deep ecologists like Næss, are perhaps more radical than others). In evaluating actions, Callicot defends the land-ethic maxim of Aldo Leopold, "A thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community; it is wrong when it tends otherwise" (Leopold, 1949). The general concern for the biotic and abiotic community as a whole leads to the alternative classification of the outlook as an "holistic" ethic (Sagoff, 1984). The inclusion of the abiotic components of the environment in ecocentrism, together with the fact that most definitions of the environment in international legislation include man, biota, abiota and physical surroundings, raises the issue of how to deal with the abiotic (i.e., soil, rocks, water) in environmental protection, particularly since many environmental standards are based on concentrations in media.

In radiation protection, the ecocentric view has been linked to the ecosystem approach of environmental assessment and management (IUR, 2012), and is sometimes presented as an alternative to the reference animals and plants approach forwarded by the ICRP (ICRP, 2008). One of the criticisms of the RAP approach is that the twelve selected species do not permit an ecosystem level assessment. To do this one needs a broader range of ecologically relevant species covering producers, predators and decomposers, as well as insights in to differences in the sensitivity of species (Bréchignac et al. 2011); variability in sensitivity is a driving factor for ecosystem change since some species can prosper by the impacts on others. But this does not mean that the ICRP approach is not capable of providing relevant information; there is just a need for data on a wider variety of species. And as stated above in the discussion of biocentrism, the method of carrying out an environmental impact assessment should not be taken as the same as ascribing moral value to those entities. As discussed below, many ecosystem services approaches to environmental protection are blatantly anthropocentric in both their approach and underlying value-system.

6.2.3 Anthropocentrism

In defence of anthropocentrism, both scientists and philosophers have argued that human interests can provide a powerful set of motives for protecting nature (Wilson, 1984; Sober, 1986). Understanding the economic and social impacts of environmental damage on humans can provide a strong incentive to protect the ecosystem. On a more philosophical defence of anthropocentrism, William Frankena suggests that only humans are capable of "valuing" in an ethical sense (1973, 1979). Although, in recent years, some interesting research has been carried out on the morality of animals (Bekoff and Pierce, 2012). In reply to Leopold and Callicott, anthropocentrics ask: who is to answer the question of when a biotic community is stable and beautiful? Can such counsel ever express more than the ecological interests of humans and the species they most closely identify with? (Fritzell, 1987).

Anthropocentrics are also concerned about impacts of radiation on animals and plants (and even soil and water, should that impact on human use of the resources), but they do not consider these entities to have moral standing or value in themselves, only by virtue of the consequences to humans. As an example, Kant's philosophy was clearly human-centred, but his morality did include restrictions on what harms one might cause to animals. His rationale was that people who mistreat animals are likely to develop a habit that inclines them to treat humans in the same fashion (Kant, 1785; Regan and Singer, 1976). Similarly, looking at the bigger picture, one might argue that not showing respect for nature would foster an inclination to lose respect for one's fellow humans.

Interestingly, the anthropocentric and the non-anthropocentric ethic tend to highlight both man's uniqueness and our oneness with nature. Humans are the only ethical animal, the only "valuer"; humans are responsible for environmental destruction unmatched by any other species, population growth is a singularly human problem. Biology, evolutionary science and genetics have shown that humans are continuous with the rest of nature, "yet none of this scientific reasoning can guarantee that we will develop ethical concern or a proper relation to the biosphere, any more than the knowledge that other human beings are our genetic kin will prevent us from annihilating them in war" (Bradford, 1993). We may agree that humans have a responsibility not to damage the environment, but disagree on what measures are needed to correct human behaviour, and when intervention to protect the environment is necessary.

The ICRP statement that “if man is protected, other living species are also likely to be sufficiently protected” (ICRP 1977) is widely perceived to be an anthropocentric approach to environmental protection. This is understandable when combined with the strong historical human focus on radiation protection. Exposure experiments on animals were carried out largely to provide information on human effects; the majority of studies on environmental transfer concentrated on those food-chains with humans at the top. But whilst the statement is clearly an anthropocentric approach to risk assessment, it does not necessarily mean that radiation protection does not value the environment per se. In the 1960s, the operators of the Windscale plant took the trouble to evaluate the possible environmental impact of its radioactive discharges (Dunster et al., 1964, cited in Kershaw et al., 1992).

6.3 Common ethical principles

Despite the apparent diversity of these three theories, it is important to realize that although they may disagree quite strongly over why, exactly, certain factors are relevant to ethics, there can still be room for consensus on some common features. For evaluation of any action involving exposure of humans, animals or plants to radiation, each of the above theories would find it morally-relevant to ask: (1) who and what is being affected; (2) what is the relative size of the benefits and the harms arising from the exposure; (3) what is the distribution of the risks and the benefits; and (4) what alternative courses of action are available?

With respect to protection of the environment and non-human species, all theories can defend the principle that radiation protection should not be limited to humans. Since regulations already exist for the protection of the environment from other contaminants, all other things being equal, there is no ethically relevant reason why effects caused by radiation exposure should be treated differently. However, the different theories might disagree on which types of effects matter most, depending for example on whether harms are evaluated in terms of sentience, animal rights, consequences for existing humans or effects on future generations. Two examples of the types of challenges in practical radiation protection are discussed in the next sections. First, the question of how to correlate ecological change with risk of harm; second the issue of assigning a monetary value to environmental impacts. Both of these aspects have been the focus of recent discussions in environmental radiation protection.

6.4 Harms and values in practical radiation protection

No one disputes that exposure to radiation can cause changes in biota and the environment, but what many experts question is the long-term consequences of such doses. It is accepted that deterministic, stochastic and hereditary effects in plants, insects and animals, have been seen both in the laboratory and after serious accidents and that species can show large variations in radiological sensitivity (UNSCEAR, 2012). Scientists have documented genetic mutations in a number of species following the Chernobyl accident (Ellegren et al., 1997; Pomerantseva et al., 1997; Mosseau and Moller, 204, 205, IAEA, 2006) and damage to pine trees in the Red Forest resulted in the pine forests being replaced by the more radiation-resistant birch (IAEA, 2005; Kovalchuk et al, 2004, 2005). Similar ecosystem level effects were also reported after the Kyshtym accident, including from coniferous to deciduous forests, and population level effects on some insects and mammals (JNREG, 1997).

Immediately after the Fukushima accident, questions were raised about the possible ecosystem effects (Garnier-Laplace et al. 2011); and studies showing impacts on butterflies in contaminated areas were widely reported in both the scientific and traditional media (Hiyama et al. (2012).

But scientists disagree over whether or not these changes reflect permanent or serious ecological damage—after all the forests grew back, the wildlife returned, and genetic change is not always a bad thing (Baker et al., 1996). Indeed, many people have suggested that the ecological benefit of removing humans from the Chernobyl area might outweigh any radiation detriments (Maiyo 1999). The consequences that are deemed “harmful” depend on the level of protection awarded to the various components of the environment (individual, population, species, ecosystem). This in turn can depend on the moral standing of those components.

The regulation of human exposure to radiation takes effects on individuals very seriously. Management of environmental hazards tends to disregard low rates of stochastic effects, focusing instead on the risk of harm to populations. In this respect, most environmental risk managers make a clear moral distinction between human and non-human species: individual humans matter; individual animals tend not to. The types of radiation exposure that result in observable (and probably, therefore, unacceptable) damage on a population level are thought to be far higher than the mGy levels at which intervention to protect humans takes place. While this might be true for mortality, however, it need not be the case for other biological endpoints such as reproductive ability and genetic effects. In some cases, such as for endangered species, effects on the individual are deemed to matter—even if not quite as much as for individual humans. Of course the variety of non-anthropocentric views may offer quite different interpretations and explanations on this last point. Some might be offended by the mere presence of man-made radionuclide in the environment, irrespective of any discernible effect on humans or biota.

To conclude, supporters of both anthropocentric and non-anthropocentric ethics can agree that harms to non-human populations should be avoided. They may disagree on the level of population change that can be accepted, and which populations should be considered the most important to protect. Likewise all viewpoints could find specific cases where the individual would be the appropriate level of protection: the anthropocentric and ecocentric may focus on endangered species or habitats; the biocentric on certain individuals as having value in themselves. Both the anthropocentric and ecocentric may find it necessary to also address changes in the abiotic environment, i.e. increased concentrations of radionuclides in soil, water and air. Anthropocentric support for such views may arise from aesthetics or a wish to “preserve” “pristine” environments such as the Arctic; ecocentric support may arise fromconsiderations of the inherent value of all components of the ecosystem. To conclude, population effects can be an appropriate focus for environmental protection from ionising radiation, but not at the exclusion of effects on individuals, ecosystems or even the abiotic environment itself.

6.5 Assigning monetary value to the environment

A number of philosophers and politicians are concerned by the tendency of environmental policy to attach monetary value to the environment (Barde and Pearce, 1990; Spash, 2010). A similar issue has been raised in human radiation protection (see Valentine) since a limit on the amount of money invested to reduce one manSv can be recalculated in terms of “a price on a life” (assuming, of course, that the linear, non-threshold hypothesis holds). A more nuanced view is provided by social ecologists who suggest that economic, ethnic, and cultural issues lie at the core of the most serious environmental problems we face today (Bookchin, 1991, 1993). Hence, ecological problems cannot be understood, much less, clearly resolved, without dealing with problems within society. “An environmental philosophy that fails to recognise the interrelatedness of the social and natural crisis, will fail to uncover and confront the real sources of the ecological meltdown occurring today" (Bradford, 1993). The interrelationship between environment, economy and society is grounded in the principle of sustainable development, and a central component of an ecosystem approach to environmental protection (Costanza et al., 1997; Millennium Ecosystem Assessment, 2005). This focuses on the ecosystem, rather than single species, and the sustainable use of resources. They stress the inherent dynamic interactions between system components (including humans), potential feedback loops, and indirect effects, resilience. The concepts of ecosystem services and ecological economics are aimed predominantly at the ultimate benefits of ecosystems for humans, either financially or otherwise, while the ecosystem approach is arguably less human-centred. Nevertheless, all approaches share a fundamental recognition of the integration and interdependency of humans and the environment.

Other ecologists, however, suggest that the root of the problem is capitalism itself and, in turn, the reduction of all societal values to profits and losses. In a market economy nothing can be sacred, since to be sacred means to be "non-exchangeable" (Kovel, 1993; Spash, 2012). The challenge is that if one does not attach monetary value to the environmental consequences of actions, then it makes it difficult to account for those consequences in a cost-benefit evaluation. Honest accounting of the interests of present as well as future generations can make environmentally damaging policies unprofitable.

An analysis of the economic consequences of the Japan tsunami and Fukushima accident on fishing industries offers an interesting perspective on the issue. The ecological economist Shunsuke Managi has pointed out that since Japanese fishing industries were heavily subsidised, the government is actually saving money through fishing restrictions. Furthermore, in many areas the traditional fishing was unsustainable and outdated, hence rebuilding after the tsunami offers the opportunity for a rejuvenation of the industry (Pacchioli, 2013). There are also ecological benefits from a ban or restriction in fishing over large areas, as well as the complex social consequences caused by demographic changes in the, predominately young, people moving out of contaminated areas and not carrying on in family business. This type of holistic analysis is also in line with ecosystem approaches to environmental impact assessment, as proposed by the IUR and other environmentalists (IUR, 2012), as a possible way of reconciling anthropocentric and non-anthropocentric worldviews in practice. But recognizing some of the more fundamental concerns ecocentrics have about the links between ecological damage and monetary valuation of natural resources, perhaps the most important recommendation is that such damage is not assessed only in terms of instrumental value, assessors should also respect the idea of intrinsic value of plants, animals and the environment.

6.6 Conclusion: Relevance of the value debate to ecological radiological protection

Difficulties in defining a valuation for the environment include fundamental questions such as what exactly constitutes harming the environment and how the environment should be valued. Both of these, typically philosophical, problems arise in assessments of any environmental contaminant. 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. Furthermore, most people would agree that harms caused by exposure of non-human species to radiation should carry weight in optimization and justification—either because the species has value in itself and/or because of the potential consequences for future human generations.

Because there are no easy answers to the challenges highlighted above, any system of environmental protection should be sufficiently flexible to allow such conflicts to be addressed.Ethical evaluation can be valuable both in identifying controversies and in forcing decision makers to address the issues, and clarify the premises upon which decisions are being made. Showing that decision makers are aware of, and have considered, such conflicts is an important step in making ethical issues transparent in policy making. Ethical evaluations also encourage attempts to find alternative solutions in order to mitigate or avoid the ethical insult, and help to document the assumptions and reasons behind eventual disputes. For instance, it is helpful to know whether experts disagree on ways of managing radiation risks due to a matter of fact (e.g., they might disagree about the environmental consequences or the probable cost of remediation) or a matter of ethics (e.g., they may disagree about the relative importance of human interests against those of non-human species).

Ethicists put great weight on “treating like cases equally”. In this respect, protecting the environment from radiation will need to be put into context with the risks from other environmental contaminants and detriments. Unless there are clear, morally relevant grounds, radiation damage should not be treated differently than other hazards. The significant progress made in developing frameworks and tools for assessment of the effects of ionizing radiation over the past two decades (e.g., ICRP, ERICA) mean that decision-makers have a much more robust scientific basis for comparison of the ecological impacts of radiation with other environmental stressors.

To conclude, there is a need for a holistic evaluation of the environmental impacts of ionizing radiation that not only considers the direct consequences on the health of humans and non-human species, but also the more complex social, ethical, and economic consequences of both human and non-human exposures. Ethical risk evaluation for both humans and the environment extends the issue of whether a risk is acceptable into dimensions that go beyond its probability of harm; ethical risk management asks questions other than those connected simply to radiation dose and its economic costs.

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7 SUMMARY

Prepared by Laurence Lebaron-Jacobs on behalf of the Working Party on Research Implications on Health and Safety Standards of the Article 31 Group of Experts[2]

7.1 Introduction

This chapter provides the background, summarizes the presentations and the results of the round-table discussion, and tries to identify the potential implications of the Scientific Seminar on “Protection of the Environment”, held in Luxembourg on 20 November 2012. While it is not intended to report in an exhaustive manner all opinions that were expressed, it takes account ofthe presentations and the discussions that took place during the seminar and during the subsequent meeting of the Article 31 Group of Experts. This chapter has been submitted to the lecturersfor comments.

7.2 The Article 31 Group of Experts and the rationale of the Scientific Seminars

The Article 31 Group of Experts is a group of independent scientific experts established according to Article 31 of the Euratom Treaty to assist the European Commission in the preparation of legal initiatives under the Euratom treaty for the protection of the health of workers and the public against the dangers arising from ionizing radiation, in particular theBasic Safety Standards. This group of experts has to give priority to the protection of health, to safety and to the development of the best available operational radiation protection. For doing so, the Group of Experts need to follow carefully scientific and technological developments as well as new research data, in particular if these developments could impact the protection of exposed persons.

In this context, the European Commission organises every year, in cooperation with the Group of Experts, a Scientific Seminar on emerging issues in Radiation Protection – generally addressing new research findings with potential policy and/or regulatory implications. The Article 31 Group of Experts chooses a topic amongst issues identified by the Working Party RIHSS to be of interest for discussion at such a Scientific Seminar. Leading scientists are invited to present the status of scientific knowledge in the selected topic. Additional experts, identified by members of the Article 31 Group from their own country, take part in the seminars and act as peer reviewers. The Commission convenes the seminars on the day before a meeting of the Article 31 Group, in order that members of the Group can discuss the potential implications of the combined scientific results. Based on the outcome of the Scientific Seminar, the Group of Experts referred to in Article 31 of the Euratom Treaty may recommend research, regulatory or legislative initiatives. The European Commission takes into account the conclusions of the Experts when setting up its radiation protection programme. The Experts' conclusions are also valuable input to the process of reviewing and potentially revising European radiation protection legislation.

[2] The following members of the Working Party on Research Implications on Health and Safety Standards of the Article 31 Group of Experts contributed to the preparation of this overview: A. Friedl, R. Huiskamp, L. Lebaron-Jacobs(who was acting as rapporteur for this seminar),S. Risica, P. Smeesters (Chairperson of the WP), and R. Wakeford. They were assisted by S. Mundiglfrom the European Commission:.

7.3 Main Points arising from the Presentations

Jan Pentreath – *Protection of the Environment: Why and how?*

The Publication N°91 (2003) of the International Commission on Radiological Protection (ICRP) discussed the problem, ethics, scope, and ways of approaching the problem of the protection of the environment. The ICRP recognizes three situations of exposure: planned, existing, and emergency. These exposure situations can relate to humans and/or biota. A number of tools and approaches have been developed recently to allow assessments of the environmental impact of radiation on wildlife. The ICRP has stated an intention to provide a more inclusive protection framework for humans and the environment.

ICRP created in the 2007 Recommendations a Reference Animals and Plants (RAPs) approach, together with derived consideration reference levels (DCRLs) which attempt to define a band of dose rates for each biotic type, as starting points for considering what action to take, depending on the objectives and the exposure situation. The introduction of the RAP approach has clarified what specific environmental protection objectives may apply in different exposure situations, without confusing the issue with the protection of humans. This is a flexible and transparent approach with issues from protecting individual species, to habitats, or to ecosystems, or even to issues relating to animal welfare depending on the exposure situation. Moreover, to prevent or reduce the frequency of deleterious effects of ionizing radiation to a level where they would have a negligible impact on the maintenance of biological diversity, the conservation of species, or the health and status of natural habitats, communities, and ecosystems, the ICRP's 2007 recommendations give key elements relating exposure to dose, and dose to effect, for different types of biota in a manner that is similar to that which is used as the basis for the protection of humans.

The framework of human radiation protection is based on knowledge of the relationships between exposure and dose, and dose and effect, as reflected through the use of Reference Male, Reference Female, and Reference Person, with the protection of the Representative Person being managed by a set of dose constraints and reference levels. This new approach should clarify what has previously been a rather confused area of policy objectives, scientific interpretation, and management actions over whether or not both humans and the environment are protected, and relevant to what legislative requirements, under different exposure situations.

In conclusion, a consistency in regulatory approaches amongst large industries is needed. Thus there is an opportunity to move towards a system of discharge control, for low-level 'normal' releases, that could satisfy the objectives of both human and environmental protection, thus bringing the nuclear industry more in line with other major industries;

Moreover, research priorities emerge from this approach as development of more realistic models, improvement of dosimetric models and more information on chemical composition of pollutants.

François Bréchignac – *Ecological impact of ionising radiation: an endpoint issue*

The protection of the environment from ionizing radiation is based on a concept of reference organisms. The ICRP defines a "reference organism" as "a hypothetical entity, with the assumed basic biological characteristics of a particular type of animal or plant, as described to the generality of the taxonomic level of family, with defined anatomical, physiological, and life-history properties, that can be used for the purposes of relating exposure to dose, and dose to effects, for that type of living organism". This approach is derived from the system of human radiation protection. The risk assessment is rated through some organism-level effect endpoints using a dosimetric approach to individual organisms associated to radio-toxicological data. The "reference organism approach" is biocentric and built upon individual organism responses without accounting for higher levels of the environment organization.

Only linear transfers to biota are considered and effect endpoints (early morbidity, mortality, reproductive success, chromosome damage) are related to individual organisms. The biological impact of these endpoints is evaluated on sectorial objectives as endangered species, protection of biodiversity, pollution control, and nature conservation. The used methodology ignores interactions between species and cannot account for ecosystem-level effects as indirect effects, “cascade” effects, trans-generation propagation of effects, and propagation from individuals up to populations and ecosystem. A scale of risk (Derived Consideration Reference Levels) has been derived from the available literature. However most data come from isolated organisms tested in ideal experimental conditions. Moreover, the biocentric approach is limited and does not meet the integrated environment protection objectives, where populations or communities and, structure and functions of ecosystems should be considered as a whole. This concept is the “ecosystem approach” based on the idea that radioecology is supposed to support the radiation protection of the environment including man. The “ecosystem approach” integrates the interdependence relationships of living organism populations (plants, insects, and animals) and their natural environment (soil, sediment, atmosphere, water, light). The modification of any parameter by a chemical substance or a radionuclide may disrupt the ecological equilibrium or be hidden within a resiliency of the system. Thus indirect effects (for example response of an ecological system diatoms-chironomids in trophic interaction to U.V.), higher levels of organization and ecosystem resilience are considered in this new concept.

In conclusion, some recommendations to develop more integrated and functional endpoints, to promote overall consistency between research in ecology and environmental management, and to improve the dialogue between environmental assessors and managers have been listed. Moreover, some research priorities have been identified: studies on impact at the ecosystem level (interactions between populations…), on ecologically relevant effects (differences in radiosensitivity…) and in cross-cutting disciplines and approaches (Chernobyl, mines, Fukushima…).

David Copplestone – *Protection of the environment in normal (planned) situations*

Legislation arising from the European Commission has increased the need to consider the impacts of discharges on the environment when issuing discharge consents for both radioactive and nonradioactive substances. Implementation of the Wild Birds and Habitats Directives has led to the creation of conservation areas collectively referred to as Natura 2000 sites.

Relating to the protection of the environment from ionizing radiation in the UK, one of the most relevant regulations derived from EU Directives is the UK Habitats Regulations 1994 which implements the Habitats Directive (Council Directive 92/43/EEC on the conservation of natural habitats and of wild flora and fauna), and provides mechanisms to protect sites designated under the Birds Directive. The regulations require measures to be taken to maintain or restore to favourable conservation status of habitats and species of wild flora and fauna of Community interest. These regulations collectively offer the principle means where by protection of designated nature conservation sites from potential damaging effects from operations such as radioactive discharges is achieved. Stringent legislation concerning the use of nuclear materials, containment of radiation sources and discharges of radioactive waste exists in the UK. The Environment Agency in England and Wales has the responsibility to issue authorizations, which stipulate discharge limits and methods of disposal.

The environment agencies have obligations to protect or enhance the environment, taken as a whole, and to contribute to sustainable development through pollution prevention. Other pollution control duties are to prevent, minimize, remedy or mitigate the effects of pollution to the environment.

Radioactive discharges from nuclear installations must be authorized prior to disposal. Authorizations place limits and conditions on operators to ensure that the radiation doses to humans resulting from radionuclide discharges remain within internationally agreed limits.

The environment agencies are responsible for issuing new authorizations, varying and reviewing existing authorizations, consents, licenses and permissions for discharges affecting Natura 2000 sites. These discharges, whether directly released into the designated sites or having a potential impact on them, must exert no adverse effect on the integrity of the site. The environment agencies also report on the state of the environment based on their own independent monitoring programs.

Dose models were carried out to evaluate the impact of ionizing radiation on wildlife from authorized discharges in England and Wales. The models have been produced as spread sheets for estimating doses to biota for various radionuclides and in different ecosystems. A database of concentration factors for these organisms and radionuclides has been developed for use in the impact assessment process. Tests on the validity of these concentration factors have proved successful, giving confidence that a generic assessment performed using the concentration factors identified will provide a result which is likely to be, if anything, over cautious. It is however recommended that wherever possible site-specific information (water, air, soil or biota concentrations) should be used in the dose calculation spread sheets to improve the assessment.

Reference organisms were defined as: *“a series of imaginary entities that provides a basis for the estimation of the radiationdose rate to a range of organisms that are typical, or representative, of a contaminatedenvironment. These estimates, in turn, would provide a basis for assessing the likelihoodand degree of radiation effects. It is important to recognise that they are not a directrepresentation of any identifiable animal or plant species.”*

By using the “reference organism” approach, a standard set of models and databases of information can be developed for comparison purposes.

These reference organisms have been selected based on consideration of ecological-, and radio-, sensitivity. Radioecological data for species of similar size to the reference organisms can be used depending on the locality of the site under assessment.

For the purposes of human assessment, ICRP recommends to use the 1 mSv y^{-1} public dose limit. For the wildlife assessment, an incremental screening dose rate of 10 μGy $h^{-1}$was derived using approaches compatible with those used for setting benchmarks for chemical risk assessments. Biological effect endpoints that are relevant to population level effects have been selected. About 100 authorizations exceeded the screening level at 51 Natura 2000 sites, so further assessment and choice of other analogues were required. For example, Ribble and Alt estuaries presented an initial assessment of 520 µGy.h^{-1} and radionuclides giving dose were primarily associated with Springfields Fuels Ldt discharges. Due to the risk of direct toxicity of radioactive substances released at levels which can compromise the supply of invertebrate prey used by birds, some operations were carried out on this site to decrease alpha and beta discharges. Radioactivity in food and in the environment was then reassessed using monitoring data and ERICA tool. All results were below 40 µGyh^{-1} in 2008.

At present, spreadsheet tools combining human and wildlife assessments and a procedure for all new or variations to permits are available after updated Sellafield habitats assessment (2011).Moreover an annual update to check is still in compliance. In the UK, it is planned to adopt ERICA and replace R&D 128 to obtain targeted dose re-assessments in conjunction with recent environmental monitoring data using new numeric criteria.

Sergey Fesenko – *Effects on non-human species in areas affected by a radiation accident: implications for radiation protection*

A characteristic feature of severe radiations accidents is the presence of distinct phases which include a short phase with severe acute radiation exposure, an interim phase with redistribution of radionuclides in the ecosystem and a subsequent long period where dose is determined by long-lived radionuclides and dose rates are much lower and become chronic.

- Chernobyl accident:

The Chernobyl accident occurred in spring as plants enter their period of accelerated growth and reproductive phase, the most radiosensitive phases of their life cycle. A large number of field studies were conducted following the Chernobyl accident particularly investigating the effects on terrestrial plants and mammals. However, the lack of dosimetry data makes it difficult to compare field investigations with reported laboratory results which used clearly defined experimental approaches with known radionuclide concentrations or doses. The area suffering from the greatest impact was the 30km exclusion zone, which received doses of 80-100 Gy within the first few weeks of the accident.The high heterogeneity of contamination levels and the diversity of location of each speciesresults in a high heterogeneity in doses to non-human species.

During the first period (one month after the accident), acute adverse effects within 30-km zone were observed. These included for example leukopenia and anaemia in pigs, dogs and cats; haemorrhages and local necrosis of liver or kidneys in abandoned dogs; mortality of conifers and impacts on plant and animal reproduction (reduced weight offspring of highly exposed cows),

Ruminants were particularly exposed to high doses of ^{131}I and ^{133}I. Due to endemic deficiency of stable iodine in soils, the transfer of radioactive iodine to ruminant thyroids was 2 or 3 fold higher than in other mammals. No apparent symptoms of acute radiation sickness were observed.Soil invertebrates were highly affected at a distance of 3-7 km: their population dramatically decreased (20-30-fold on sites where soil surface dose was between 8 and 30 Gy) and reproduction was strongly impacted.The population of rodents decreased by a factor of 2 to 10 in highly contaminated plots just after the Chernobyl accident.

From 1 to 12 months after the accident (the second period), lower dose rates were recorded due to the decay of short-lived isotopes. Due to redistribution of radionuclides to soil, soil invertebrates were impacted. Morphological effects were noticed in plants and pine forests.

Finally, later than one year after the accident (third period) recovery is going on with noticeable positive impacts. Despite the mass death of pine forests and cytogenetic effects reported in mammals and birds in the immediate aftermath of the accident, it has been suggested that the sum effectofit on flora and fauna has been positive with observed increases in biodiversity and species abundance. Due to the migration of rodents from less affected areas the population recovered during the spring of 1987.

It is also established that secondary effects are essentially due to human abandonment. With the removal of humans, wildlife around Chernobyl is now flourishing: 48 endangered species listed in the international Red Book of protected animals and plants are now thriving in the Chernobyl Exclusion Zone, as Prejevalsky Horses. Plant diversity within the exclusion zone is similar to that in protected areas outside the zone. These studies indicate that the benefits of excluding man far outweigh the impact of ionizing radiation.

Although doses were decreasing 2 or 3 years after the accident, incidence of plant cell mutations increased (abnormal sperm, partial albinism in barn swallows, for example).Moreover, in spite of reciprocal translocations, the mice population recovered within 3 years after the accident.However, the long term genetic consequences are unknown at the present time.

Detailed long-term studies on genetic load, population genetics, mutation rate, life expectancy, fertility and radioresistance are required to evaluate the long-term ecological impact of the accident.

- Kyshtym accident :

This accident occurred in 1957 at a period of low sensitivity for biota species. The releases were mostly composed of â emitters (10^6 Bqm^{-2}) and heterogeneous radionuclide deposition on soils resulted in high heterogeneity of doses in non-human species. Two periods have succeeded: the acute stage (1957-1958) and the recovery stage (after 1958). Studies started in 1962 show severe effects on non-human species observed in relatively small areas: excess of chromosome aberrations, genomic instability, radioadaptation. Genetic effects are still observed in few areas.

Secondary effects result mainly from a disruption of ecological relationships between components of various ecosystems, due to changed microclimatic conditions, disturbed synchronism of seasonal phases, imbalanced food interrelations between consumers and producers (decreased food resources) and immigration of new species.

Finally, as observed after the accident of Chernobyl, when recovery is going on, noticeable positive impacts are observed but long-term genetic consequences are unknown.

In conclusion, severe effects in non-human species were observed in relatively small areas even after the Chernobyl or Khystym accident. Recovery of highly impacted populations of non-human species is observed from 2 or 3 years after. Some ecological consequences still persist such as genetic effects. However, no new approaches are needed to assess the effects of these accidents on non-human species. The impact on biota has only to be precisely assessed and considered in the management of exposed areas.

Alain Vavasseur – *Stakes and limits of bioremediation*

Bioremediation is the use of organism (bacteria, alga, fungi, or green plants) metabolism or organic molecules as DNA, antibodies to remove pollutants. Technologies can be generally classified as *in situ* or *ex situ*. *In situ* bioremediation involves treating the contaminated material at the site, while *ex situ* involves the removal of the contaminated material to be treated elsewhere. Some examples of bioremediation related technologies are phytoremediation, bioleaching, rhizofiltration.

There are a number of cost/efficiency advantages to bioremediation, which can be employed in areas that are inaccessible without excavation. Moreover, this is typically much less expensive than excavation followed by disposal elsewhere, incineration or other *ex situ* treatment strategies.

Bacteria are able to transfer electrons from actinides (U, Tc, Pu…) and to transform soluble uranium in insoluble uranium, which is less bioavailable. Bacteria are used i*n situ* or *ex situ* for the bioremediation of some of these radionuclides. Biomineralization, obtained by spraying the contaminated zone with water and nutrients in acid conditions, uses *in situ* indigenous microorganisms to enhance the process. This strategy was applied in Oak Ridge site (Tennessee, USA). Bacteria are also used *ex situ* to treat contaminated soils or effluents (Perring, South Africa): they are integrated in various polymers or ceramic filters and included in a filtration process to immobilize pollutants.

Both phytoextraction and rhizofiltration follow the same basic path to remediation. First, plants are put in contact with the contamination. They absorb contaminants through their root systems and store them in root biomass and/or transport them up into the stems and/or leaves. The plants continue to absorb contaminants until they are harvested. The plants are then replaced to continue the growth/harvest cycle until satisfactory levels of contaminant are achieved. Both processes are also aimed more toward concentrating and precipitating heavy metals than organic contaminants. The major difference between rhizofiltration and phytoextraction is that rhizofiltration is used for treatment in aquatic environments, while phytoextraction deals with soil remediation.

Phytoremediation has been employed at sites with soils contaminated with lead, uranium, and arsenic. This process was applied in Ashtabula, Ohio (US), by a private phytoremediation company, Phytotech.

Rhizofiltration is a type of phytoremediation, which refers to the approach of using hydroponically cultivated plant roots to remediate contaminated water through absorption, concentration, and precipitation of pollutants. Experiments concerning rhizofiltration are ongoing at a DOE facility in Ohio. Rhizofiltration using sunflowers are used in the remediation of radionuclides (strontium and cesium) from surface water near Chernobyl.

Rhizofiltration allows in-situ treatment, minimizing disturbance to the environment.

After the accident of Fukushima-daiichi, a very large area has been contaminated. An enormous volume of soil contaminated with^{137}Cs and ^{134}Cs should be removed if the limit 4 mSv per year for man has to be reached (10^8 m^3 of waste). Some treatments could be proposed: phytoremediation by specific plants (*Amaranthus retroflexus*, *Brassica juncea…),* high-affinity K^+ transporters.

There are some limitations to use bioremediation:

- the production of contaminated biomass needs the creation of a concomitant infrastructure to burn this waste in safe conditions meaning the creation of adapted nuclear installations
- long-term commitment is required, as the process is dependent on a plant's ability to grow and thrive in an environment that is not ideal for normal plant growth.

The cost of the phytoremediation is lower than that of traditional processes both *in situ* and *ex situ*: the plants can be easily monitored and the possibility of the recovery and re-use of valuable metals preserves the environment in a more natural state. However, the survival of the plants is affected by the toxicity of the contaminated land and the general condition of the soil.

Rhizofiltration is cost-effective for large volumes of water having low concentrations of contaminants and subjected to low (stringent) standards. Plants that are efficient at translocating metals to the shoots should not be used for rhizofiltration because more contaminated plant residue is produced. It is relatively inexpensive, yet potentially more effective than comparable technologies.

Finally, mixed classical techniques of remediation should be applied to remove pollutants from limited hot spots of contamination while bioremediation could be applied to extended regions moderately contaminated.

Deborah Oughton – *Ethical aspects of Protection of the Environment from ionising radiation*

Several points of view can currently be identified within the ethical aspects of the protection of the environment. These views arise from philosophical considerations: what has moral standing or value in the world?And why?The most fundamental source of divergence arises between the anthropocentric and the non-anthropocentric views. For convenience, they may be summarized as follows:

- *anthropocentrics*, probably the most easily recognized,for which human beings are the main or only thing of moral standing, are the only “valuers”, and thus the environment is of concern primarily as it affects humans; humans are the only entities that have moral standing;

- *biocentrics*, (literally “life-centered”) have a common feature: recognition of the moral obligations that arise from the fact that many animal species can be shown to be *sentient*, in that they can experience pleasure and pain. individual life-forms other than humans should be respected for what they are—not only because they affect the interests of humans; biocentrics disagree on the basis by which we draw a moral distinction between humans and animals and on which organisms have moral standing;in practical policy-making, biocentric outlooks have had the greatest influence in issues of animal welfare and the use of animals in research;

- *ecocentrics*, who claim that both biotic and abiotic components of the ecosystem (including landscape features such as rivers and mountains) can have moral standing: it is an “holistic” ethic; however, ecocentrics disagree on the reasons for and solutions to environmental problems (human arrogance, male dominance, social and economic hierarchy). Allecocentrics claim that a radical change from an anthropocentric attitude of domination and exploitation of natural resources towards a greater respect for the integrity of nature is needed (for example, Næssmore radical than others).

Thus biocentric and ecocentric views reflected in social, cultural and religious levels of societies.

Finally, all three theories can support the need to protect the environment. However divergences remain between the three theories about intervention and remediation after accidents.

The Western view of nature is perceived as a main source of environmental problems. For example, ecofeminists suggest the problem lies in the history of male dominance and sexist oppression of females.Other people locate the source of the problem in the Judeo-Christian tradition, and, more specifically, in Biblical quotations.

To conclude, supporters of both anthropocentric and non-anthropocentric ethics can agree that harms to non-human populations should be avoided. They may disagree on the level of population change that can be accepted, and which populations should be considered the most important to protect. In summary, population effects can be an appropriate focus for environmental protection from ionizing radiation, but not at the exclusion of effects on individuals, ecosystems or even the abiotic environment itself.

Augustin Janssens – *Legal basis for a regulation on the Protection of the Environment*

The International Basic Safety Standards reflect an international consensus on what constitutes a high level of safety for protecting people and the environment from the harmful effects of ionizing radiation. While the ICRP has published a methodology for dose assessment for biota, a publication on the application of criteria is still awaited. Pending such further guidance, it is up to national authorities to assess the doses to representative animals and plants in terms of protection of the ecosystem. Appropriate technical measures also need to be taken to avoid the environmental consequences of an accidental release and to monitor existing levels of radioactivity in the environment, from the perspectives of both environmental protection and human health.

For the moment, relevant provisions about the protection of the environment are incorporated in Chapter VIII of the BSS considering environmental criteria in discharge authorization, on the basis ofavailable scientific data. Furthermore, a methodology for the assessment is needed. Thus the European commission is waiting for ICRP environmental criteria.An amendment could be incorporated to the Directive.

7.4 Summary of the Roundtable discussion

Gilbert Eggermont (Moderator)

Before the actual round table discussion, Werner Rühm gave a short presentation. He recommended protecting man as individual, environment (where earthworms live) as population and finally, the Earth's ecosystem as a whole.

Is protection of the environment necessary? Do we need additional rules and regulations?

There is no contradiction between reference organism approach and ecosystem approach: they are complementary, looking at the same problem using different ways. This new approach should integrate multiple pollutants (chemicals are released with nuclear materials).After an accident, we have to know what we must do, to make the best choices. Thus there is a need to improve dose assessment and remediation tools to protect the environment.

What are the fundamental gaps in the knowledge?

Radioecology is constantly evolving.For example, STAR (STrategy for Allied Radioecology), a European network of excellence for radioecology, has been created in 2011 as part of the 7thEuratom framework programme for research and development for 4 years. STAR is composed of nine European organisations. Its primary objective is to initiate a sustainable, efficient, long-term integration of radioecology within Europe.

What needs to be done to arrive at a conclusion?

More research on marine species and on chronic exposure to low doses has to be developed. Furthermore, lots of fundamental gaps are to be regretted on internal exposure of flora and fauna to radionuclides alone, and also to multiple pollutants (radionuclides and chemicals).

Moreover, the hormesis or the adaptive response resulting from anexposure of species to ionizing radiation is a question raised by recent scientific studies.

What is the public perception of the problem?

Protection of the environment is needed. Populations of earthworms may be considered in this context: they give a lot of useful services to man (soils).

Remediation tools are also needed for this new ecosystem approach.

To conclude, a demonstration is needed for this new ecosystem approach: technical feasibility has to be taken into account.

8 CONCLUSIONS

Working Party on Research Implications on Health and Safety Standards of the Article 31 Group of Experts[3]

1. Everybody agrees that the environment needs to be protected. Some experts are nevertheless not convinced that there could be a real harm to the environment due to radioactivity as long as humans are adequately protected by applying the European and International Basic Safety Standards. But all experts agree that we have to "demonstrate" it, as it is the case for other industries.
2. Tools for such a demonstration are to a large extent already developed, at least for use in routine situations, but ICRP recommendations as well as practical, reliable and easily accessible screening tools are currently not yet finalized. Some experts consider that the system is mature enough to incorporate additional regulation through the currently discussed BSS EU Directive (with detailed guidance to be provided later), but other judge that it is premature.
3. There are still fundamental gaps in our knowledge and a lot of missing data, implying that the currently developed tools should be complemented, particularly to deal with existing and post-accident situations. Among these gaps, due attention should be drawn on ecosystem effects, multiple interactions, dosimetric uncertainties and long term effects (particularly genetic effects such as those reported after the Chernobyl and Kyshtym accidents). The Strategic Research Agenda of the STAR European network of excellence includes an overview of the main scientific challenges in the field. It has been suggested to attract more young researchers to this domain through Ph.D. programmes.
4. The importance of remediation has been underlined but research in this field is necessary to improve its efficiency.
5. Reasons for differences in public perception should be further explored. As the system will only be efficient if the population is convinced of its reliability, more stakeholder involvement should be encouraged.

[3] The following members of the Working Party on Research Implications on Health and Safety Standards of the Article 31 Group of Experts contributed to the preparation of these conclusions: A. Friedl, L. Lebaron-Jacobs, R. Huiskamp, S. Risica, P. Smeesters (Chairperson of the WP), and R. Wakeford. They were assisted by S. Mundigl from the European Commission.