International Union of Radioecology

TOWARDS AN ECOSYSTEM APPROACH FOR ENVIRONMENT PROTECTION WITH EMPHASIS ON RADIOLOGICAL HAZARDS

Findings of the IUR Ecosystem Approach Task Group

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Towards an Ecosystem Approach for Environment Protection with Emphasis on Radiological Hazards

François Bréchignac, Clare Bradshaw, Simon Carroll, Soichi Fuma, Lars Håkanson, Alicja Jaworska, Larry Kapustka, Isao Kawaguchi, Luigi Monte, Deborah Oughton, Tatiana Sazykina, and Per Strand

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[1] ICRP: International Commission on Radiological Protection

Executive Summary

The scientific analysis which is developed in this document results from the need, after nearly one decade of development, to review in a broad context, beyond the radiation protection circles, the trends, views and methods currently exploited to set up a system of radiological protection of the environment. Indeed, the International Commission on Radiological Protection (ICRP) expressed its objective as to propose a system of radiological protection of the environment which would be compatible not only with the current system of protection for humans, but also more generally with other systems of environment protection such as those for protecting biodiversity or protecting against chemical stressors. The International Union of Radioecology (IUR), therefore, committed a broad team of experts to tackle the issue by setting up a dedicated Task Group gathering radioecologists together with experts from various areas such as risk assessment of chemicals, biodiversity, systems ecology and fisheries. The IUR "Ecosystem approach" Task Group membership therefore included: Clare Bradshaw (Department of Systems Ecology, Stockholm University, Sweden), François Bréchignac (Institute for Radioprotection and Nuclear Safety, IRSN, France), Simon Carroll (Center for Biological Diversity, Sweden), Soichi Fuma (National Institute of Radiological Sciences, Japan), Lars Håkanson (Uppsala University, Sweden), Alicja Jaworska (Norwegian Radiation Protection Authority, NRPA, Norway), Larry Kapustka (SLR Consulting, Canada), Isao Kawaguchi (National Institute of Radiological Sciences, Japan), Luigi Monte (National Agency for New technologies, Energy and the Environment, ENEA, Italy), Deborah Oughton (Norwegian University of Life Sciences, Ås, Norway), Tatiana Sazykina (Typhoon, Obninsk, Russia) and Per Strand (Norwegian Radiation Protection Authority, NRPA, Norway).

Starting from the acknowledgement that there is a number of definitions for "environment protection" depending on the context where protection is to be applied, or from where it is evolving, an attempt is first made to identify a possible common overarching objective of protection that would be general enough to encapsulate all specific goals expressed in various particular contexts. This is achieved through analysing why environment protection became a concern and how its various objectives have evolved from various perceptions including ethical, philosophical, moral, socio-economic and legal considerations. This is also pursued by briefly reviewing the issue of environment protection in the particular context of radiation, mentioning recent undertakings and achievements from the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), the International Union of Radioecology (IUR), the International Atomic Energy Agency (IAEA) and the International Commission on Radiological Protection (ICRP). It is concluded that the ecosystem concept, the structural and functional entity defined in ecology[2], best captures all the perceptions, and that the field of environment protection against radiation would gain from considering further an ecosystem approach to better fulfil the general goals of protection.

In the context of environmental impact assessment, the ecosystem approach is conceived as a holistic strategy which integrates toxicological knowledge with ecological understanding. One of its major justifications stems from the mismatch between current methodologies which are largely based upon toxicological data gathered for individual organisms and recognition that the most widely accepted goal of protection generally sits at the population and ecosystem levels of organisation. Toxicological knowledge in organisms is indeed needed, but this is not enough to adequately meet the protection goal as ecosystem responses to stress will also be governed by interactions between populations of such organisms. How ionising radiation impacts ecosystems has been widely reported from a number of empirical studies conducted in contaminated areas. Also, ecological theories featuring ecosystems as complex systems demonstrate the importance of taking such interactions in consideration especially in view of assessing ecosystem stability and resilience. Another justification to considering an ecosystem approach for radiation protection comes from the observation that the concept is already applied in other fields of environment protection, such as those dealing with protection of biodiversity, under the noticeable pressure of environmental managers and users.

In radiation protection, ongoing developments are based upon a concept of reference organisms[3], or reference fauna and flora, for which a dosimetric approach to individual organisms together with associated radio-toxicological data allow to rate the potential risk of harm through some organism-level effect endpoints. Primarily driven by operational goals for practical application, this approach necessarily features some inherent reductionism. The pros and contras of this approach are discussed especially in the light of the approach which is currently used to assess toxicity of chemicals. In order to promote an ability for suitable comparisons, the current method in use for assessment of toxicity of chemicals is first widely reviewed and discussed. Secondly, the approach developed for the protection of biodiversity is also presented in details as it is the most advanced international concerted effort having particularly

[2] Ecology defines an ecosystem as an assembly consisting of an association or community of living beings (biocenose) and its abiotic i.e. geological, edaphic, hydrological, climatic, etc… environment (biotope), (see section 1.6). The constituting elements of an ecosystem develop a network of energy and matter exchanges allowing the development and sustainability of life. This terminology was initiated by A.G. Tansley (1935) to identify the basic unit of nature.

[3] ICRP (2008) 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."

advanced the concept of an "ecosystem approach", with underlying principles, objectives and practical tools to support it. Finally, a third example is provided with the protection of fish stocks in use to support adequate management in fisheries, where more and more efforts are devoted to include modelling of ecological complexity. Given this wide context depicted above, research priorities that would help to move on beyond the reference organism concept are discussed, especially in terms of addressing the various extrapolations required and the important need to also address (eco)systems level effects. One key conclusion from the discussion leads to acknowledging that the reference organism and ecosystem approaches are complementary and part of a continuum. The concept of reference fauna and flora has been devised for the purpose of radiological protection and does not improve our understanding of the ecosystem, but considering ecosystem processes and interactions can improve the reference fauna and flora approach to radiological protection.

An extensive review of legislation about environment protection as a whole, undertaken with particular emphasis on the ecosystem approach, illustrates the wide current spread of the concept, and how it is being applied in various fields (marine environment, fisheries, OSPAR, European Union Marine Strategy Framework Directive, RAMSAR Convention on Coastal Wetlands, Convention on Biological Diversity, European Union Habitat Directive, Canadian Environment Protection Act, Forest Management). Several methods to implement ecosystem approaches are already existing, or emerging, even in the field of radiation, and their use demonstrates that this field of development is both active and promising.

The general overview and analysis made above on how does the current "reference organism" based approach developed for radiation protection fits within the overall context of environment protection drives to support the additional development of an ecosystem approach to improve future radiation protection. In view of promoting such an improvement, and of contributing to the development of relevant international strategic research agendas, this overall discussion drives to identifying Research & Development needs to support the ecosystem approach. Featuring ecosystem-level issues, enhancement of organism-level studies that could be used more effectively in modelling ecological systems interactions and, cross-cutting field studies of radiation contaminated areas from, for example, accident areas or mine sites, the research priorities fall in three categories:

- Areas of emphasis for the systems-level research include detailing interactive responses to radiation exposure, propagation of effects, delayed effects, and resistance/resilience of ecosystems. Each of these could be designed to examine effects at a) population-, guild-, or community-levels, or b) systems functions such as primary productivity, decomposition, energy transfer, or nutrient flow.
- Additional research at the organism-level should be expanded to include representatives of trophic groups not currently included or understudied (e.g., decomposers). There should also be efforts to expand representation of taxa from multiple geographic regions to supplement the current dominance of data from northern temperate systems. Topical research that would be useful would be to develop better understanding of radiation effects that result in adaptation, acclimation, hormesis, and epigenetic effects.
- Field studies are needed to calibrate laboratory studies from both the systems- and organism-levels. In addition to the opportunities at Chernobyl and Fukushima (decidedly different in terms of ecological systems), studies should be undertaken in radionuclide mining areas. In each of these potential study areas, the investigative designs should preferably be based on gradient analyses approaches and not some attempt to compare to "reference sites[4]".

In concluding, recommendations with respect to radiation protection are finally drawn. Recognizing that the ecosystem concept has been adopted in an increasing number of other situations, it is believed appropriate for radiation protection to move in the direction of an ecosystem-based approach in order to improve the relevance of information coming to decision-makers. To that end, the following points should be considered:

- Promote the dialogue between environmental assessors and environmental managers (facilities operators, contaminated site managers, and other regulators) to increase the chances of improving the value of information flow (two-way dialogue).
- More integrated and functional endpoints to expand beyond the organism-level. This could also include consideration of additional indices that embed the existing and new endpoints (decomposition, primary productivity, etc.).
- Reference organism approach–improve to incorporate ecological functionalities, other ecological criteria, and reference species versus reference organisms, all aimed to facilitate an ecosystem approach. Better consideration of taxonomy such as insects, bacteria, fungi to cover ecological functionality and to make it more accessible to people within different geographical areas, biomes[5].

[4] The "reference site" terminology is often confusing as regional data sets are nothing else than a collection of sites in the landscape that share some similarity allowing comparisons. Both, clean and dirty reference sites can lead to good effect in site specific causal assessment (Gerritsen et al., 2010).

[5] An additional concept of "representative organisms" is currently being worked out by ICRP, contributing to address this point.

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Preface

Although many scientists published experimental results about the effects of ionizing radiation on various types of biota during the 50s up to the 70s, the issue of environment protection from the adverse effects of ionizing radiation was first raised internationally by IAEA in the context of the practice of dumping nuclear waste in the oceans (IAEA, 1988). In more recent years, after a precursor conference held in Sweden in 1996, and the launch of an IUR Task Group initiated in 1997, the issue of the protection of the environment from ionizing radiation has evolved and is now a major topic for radioecologists. Several International Organizations (IAEA, UNSCEAR, EC, ICRP) and national Institutions (US DOE, CNSC Canada, NIRS Japan, IRSN France, SCK-CEN Belgium, EA U.K., NRPA Norway, RIARAE Russia, TYPHOON Russia, SSM Sweden, AECL Canada,… only mentioning a few) are now engaged in developing systems for the radiological protection of the environment from radiation.

Ongoing developments are aimed at designing conceptual approaches and methods for ecological risk assessment of radiation in the environment in order to aid decision making, especially with respect to situations of existing or future possible environmental contamination. Most of these methods build on the radiotoxicological methods in use for the radioprotection of man on one hand (concept of reference person, radiotoxicological data used to identify dose-response relationships, focus on individual organisms, etc.), and from the classical ecotoxicological methods based on individual organisms of test species for chemical toxicants, on the other hand. Consequently, they are all based on several types of "reference organisms" designed to exploit dose-response relationships at the level of individual organisms. Most of the ongoing research work is oriented in this direction. The shortcomings of such methods, however, are evident both in the field of radioprotection and also in the field of assessment of risks from other stressors, such as chemical toxicants and physical stressors.

The use of organism-based methods may prove difficult to actually meet the environment protection objectives, especially those that have been set at the population and ecosystem levels. This is one major difference between the protection of humans, where the target of protection is set at the level of the organism, and the protection of the environment, where the most frequent goal is to protect populations and the structure and function of ecosystems. The protection of individual organisms is essentially supported by toxicology. The protection of ecosystems can similarly start by making use of ecotoxicology for animals and plants (the current development at this stage), but need to be complemented and expanded by measuring additional endpoints, adding ecologically relevant parameters, and modelling that addresses ecosystem structure and functions.

It is primarily for this reason that several areas of environmental management have already been engaged in developing "ecosystem approaches" of risk assessment, like in halieutics for the protection of fish stocks in the oceans (FAO), or for the sustainable maintenance of biodiversity (International Convention on Biodiversity). Indeed, perturbations induced by stressors within ecosystems cannot be fully grasped from the exclusive toxicological understanding of the stressor's interaction with individual organisms. Such effects only act as triggers of perturbation, which propagate within ecosystems, these later being dominated by complex inter-population relationships mostly characterized by non-linear responses, which can be quite different from the initial response observed within individual organisms. Inter-population relationships at ecosystem level, such as between predator and prey, are also capable of mediating indirect effects by means of which the population less exposed to the stressor may be the most affected, even to the point of extirpation. This is particularly relevant when considering the long-term ecological effect of chronic exposure to toxicants. When considering radiation, this leads to the finding that damage may not be most due to the direct radio-toxicological effect of radiation per se (upon individual organisms), but rather to the cascading effects among interacting populations within ecosystems as a result of differences in sensitivity to radiation.

This report, the result of a collective effort by expert scientists comprising the IUR "Ecosystem approach" Task Group considers this issue in more detail. In an attempt to capture a common overarching goal of environment protection that would help to promote consensus, and therefore better guide further developments in this field, the group argues that a stronger focus on the ecosystem concept is warranted. In particular, the group advocates the need to further develop an ecosystem approach in order to support efficient environment protection against radiation.

The scientific analysis proposed in this report is aimed at demonstrating that a more holistic view when developing the scientific foundation for the radiological protection of the environment is required. Meant to stimulate Research & Development efforts in this direction, this document is therefore particularly intended to the researcher community in this field, but also in all neighbour fields focused on advancing knowledge and understanding about impact of stressors on the environment. It is also intended to all professionals interested about, or concerned with, environmental protection, such as regulatory authorities and professionals, nuclear facilities operators including those to deal with waste storage and management, policy and decision makers and all stakeholders concerned with the potential impact of radiation on ecosystems, who will find here some clues to accessing a broader view with a widened scope over a risk that is highly ranked by society.

1. Generic and Specific Goals of Environmental Protection

Environmental protection has emerged relatively recently within modern societies due to large scale environmental problems which are suspected to be linked to some human activities. Efforts to identify the goals of environment protection therefore have been undertaken through various international initiatives and institutions, which have attempted to capture the different facets of the environmental issues in order to identify a suitable legal framework to adopt and international standards for Environmental Management Systems. Radioactivity-driven potential hazards are only one specific case within a much more general context.

One historical corner stone in environmental protection has been the emergence of the sustainable development concept first raised in the Brundtland report (Brundtland, 1987) and then stressed in the so-called Agenda 21 of the Rio declaration (UNCED, 1992) which highlighted the importance of the functional aspects of ecosystems.

1.1. Why Environmental Protection has Evolved

Environmental protection was initially concerned with human health. This anthropocentric approach has been traced to the efforts to conquer cholera in Europe through closure of contaminated wells and later through various measures to improve sanitation (Schwabe et al., 1977). Medicine, both as a scientific discipline and as a profession were influenced greatly by recognition of the role of micro-organisms as disease agents and later from practices that embraced chemical hygiene. More recently, the field of environmental medicine has returned to its roots as there is increased consideration of humankind, taken as populations of human beings, through public health efforts supported for example by epidemiology. The fundamental holistic approach was articulated in the preamble of the World Health Organization charter (1946, ratified in 1948) in the declaration that *"Health is a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity."* The major driver during long periods had been protection of human life and health per se, without any major need to consider the environment in a broad sense.

Environmental protection is an issue that has evolved much more recently during the 20th century, essentially due to exponential growth of the human population since the 19th century, resulting from the associated growth of industrialisation of various processes, including exploitation of natural resources. The demand for goods by humankind in relation to economical development has also been recognised to have limits and leads to deleterious side effects that have fostered reconsideration of societal behaviours under the philosophies of sustainability. Concerns about the environment, though often stated in terms of air, land, and water, ultimately focus on biota and the well-being of humans. This leads to consideration of the relationship between environment and human protection as a closed loop within which man promotes changes in the environment (harmful or not to non-humans), such changes in turn being capable of promoting harmful feed-back impacts in humans. Furthermore, such changes may often prove to be quite indirect, that is delayed in time and space, confounding the linkages between cause and effect.

Perhaps, the first driver for the desire to protect the environment was the decrease of biological resources (e.g., fish, game, forest) that, in the absence of assigned property rights, had been freely exploited for centuries by numerous generations of harvesters, hunters and agriculture growers (see Hardin, 1968). This was followed by a reduction of species richness (i.e., biological diversity, genetic diversity), which has caused concern in modern societies for the sustainability of life due to reduced potential for adaptation with a smaller pool of genes.

Next, and more or less concurrently, impacts on human health appeared from anthropogenically produced substances in the environment (technogenic substances, xenobiotics, etc.). These can either have direct impacts - causing toxicological harm, or indirect impacts - causing changes to the environment, often on a global scale, which in turn can be deleterious to humans or their activities (e.g., climate change). These substances can either have direct toxic effects on humans, animals and plants, or cause alterations of ecological processes that will indirectly affect animals, plants and also humans. Artificial radionuclides produced within the nuclear power cycle, from atomic weapons testing, or from other industrial and medical purposes are among these anthropogenic contaminants. This is also the case for the so called "technologically enhanced naturally occurring radioactive materials" (TENORMs) that result from mining and various mineral/organic resources exploitation (e.g. oil and gas), or which accumulate in caves of houses and buildings.

1.2. Various Perceptions of the Environment

There are a number of different perspectives with which to consider environmental issues: for example, ethical, moral, legal, socio-economic, scientific, etc… all of which lead to different perceptions of the environment.

1.2.1. Ethical, Philosophical, and Moral Considerations

An expert group that was assembled under the auspices of the IAEA extensively reviewed the ethical, moral and philosophical aspects of environment protection that could be relevant to radiation (IAEA, 2002). These experts recognised that a value can be assigned to all living beings, either on their own (biocentric) or due to their active role in shaping ecosystems including their abiotic components (ecocentric), and not necessarily to man exclusively independently from other living beings (anthropocentric). The group also acknowledged the cultural and religious dimensions and the ethics of morals, duties and justice that can influence the perception of the environment.

This triggered considerable debate between anthropocentric and non-anthropocentric proponents that usually remained opposed to each other due to lack of understanding on a common ground. In fact, the biocentric/ecocentric and anthropocentric views can be compatible within an ecosystem approach: ecological functions ensured through ecosystem assembly support all life forms, that of man as well as that of non-human species. Thus, environmental protection can be viewed as being both ecocentric and anthropocentric at the same time, as it aims at ensuring the sustainability of life in general.

1.2.2. Socio-economic considerations

Socio-economic considerations are important within the various perceptions of the environment. They lead to the goal of developing a set of conservation measures intended to preserve the flow of goods and services of ecological systems, known as ecosystem services (Costanza et al., 1997; Costanza, 2000; Odum et al., 2000; Heal, 2000), recognised as being central in providing processes that support life (including that of man). Ecosystem services can be defined as the benefits that people derive from ecosystems (Millenium Ecosystem Assessment, 2005). Of particular importance are the estimations of the economical value of ecosystem services, which have been considered so far as being granted for free by nature, in order to estimate how much it would cost society to replace them if the flow of these ecosystem services was impaired. For example, in a first estimation of the economical value of world's ecosystem services, Costanza and co-workers rated it to be in the range of US \$ 16-54 trillion (10^{12}) per year. A refinement of the calculation method is probably needed (Simpson, 2010) in order to allow for a comparison to be made with the global gross national product total (GNP) being around US \$ 18 trillion per year. Some have argued that some components of an ecosystem delivering ecosystem services could potentially be substituted without compromising the delivery of those ecosystem services, but this is still highly theoretical if at all feasible.

These considerations lead to a recognition that the environment is primarily made of **ecological systems** that deliver services (which may have economical value), and which form highly structured and complex networks by means of which life (including that of human beings) is globally supported on earth.

A non-exhaustive list of life support services provided by ecosystems includes the following:

- Provisioning:
  - Solar energy capture through photosynthesis of primary producers that leads next to the production of food, of construction materials (wood), and of various combustibles of biological origin (including fossil fuels);
  - Genetic resources for new product development (medicines, food, molecules of various interests,…) arising from genetic amelioration programmes or bioengineering;
  - Water storage, purification and distribution;
  - Agricultural soil generation and maintenance;
  - Waste decomposition (by microbial decomposers);
  - Pollination of agricultural plants;
- Regulating:
  - Pest control (via insectivorous birds, for example);
  - Air purification and cleaning;
  - Climate macro- and micro-control;
  - Buffering effects reducing the consequences of natural stresses such as floods, forest fires, epidemics;
  - Nutrients (bio)regeneration;
- Cultural and supporting services:
  - Aesthetic satisfaction.

These ecosystem services that can be valued by society rely on some ecosystem functions (Table 1) as described by Costanza et al. (1997) and Curtis (2004).

Table 1: The bundle of value-laden goods and services (attributes) provided by ecosystems that society might value (Adapted from Constanza et al., 1997; Curtis, 2004).

Goods & services	Ecosystem functions	Examples
Genetic resources	Sources of unique and ever-evolving genetic information	Genes for pathogen resistance, technology for breeding
Other raw materials	That portion of gross primary production extractable as raw materials	The production of timber, fuel, and fodder
Climate regulation	Regulation of global temperature and precipitation at global or local levels	Greenhouse gas regulation, DMS production affecting cloud formation
Gas regulation	Regulation of atmospheric chemical composition	The carbon dioxide-oxygen balance and ozone levels for UVB protection
Water regulation	Regulation of hydrological flows	Provisioning of water for industrial processes or transportation
Pollination	Movement of floral gametes	Provisioning of pollinators for the reproduction of plant populations
Biological control	Trophic-dynamic regulations of populations	Predator control of prey species
Regulation of human diseases	Ecosystems can change the abundance of human pathogens	Cholera and abundance of mosquitoes can be altered
Waste treatment	Recovery of mobile nutrients and removal or breakdown of excess nutrients and compounds	Waste treatment, pollution control, detoxification
Recreation	Providing opportunities for recreational activities	Eco-tourism, hunting and other recreational activities
Heritage	Providing opportunities for non-commercial uses	Historical, aesthetic and educational values

It is worth also mentioning that beside scientific developments derived from the theory of complex systems which provide a dynamical understanding of ecosystems (Kay, 2000), recent sociological and economical developments concerning the protection of the environment have also emphasised the concepts of "ecosystem health" and "ecosystem integrity" that are now spreading (Rapport et al., 1999; Patil et al., 2001). However, strong arguments have challenged the use of the health and integrity metaphors because neither health nor integrity are properties of ecological systems (Suter 1993; Wicklum and Davies 1995; Kapustka and Landis 1998). The policy implications of these conflicting perspectives have been discussed by Lackey (2001, 2007).

1.2.3. Legal considerations

The value judgements of a society often find expression in law, as different political groups and communities translate their concerns and chosen means to address them into authoritative norms and procedures. Environmental protection law is ultimately about the relationship between humans and the earth and is manifested in the strata of national, regional, and international norms regulating human activities that may impact on the environment (Kiss & Shelton, 1997).

Due to changing knowledge and perceptions, technological change and industrial developments, and also political and institutional change, legislative and regulatory measures relating to environmental protection and natural resource use have evolved and developed considerably during the 20th century – and especially in the second half thereof (Tarlock, 2010). National environmental protection legislation was initially adopted in a few, mostly industrialised countries, like the UK with for example the Alkali Act in 1863, leading to the world's first public pollution control agency. This was followed by the Rivers Pollution Prevention Act in 1876, and this trend disseminated in most industrialised countries up to the 1960s. By the 1990s, virtually all states in the world had adopted environmental protection legislation; together, these national laws now total in the thousands. Initially and to a large extent this was in response to visible or dramatic evidence of environmental deterioration or to high-profile catastrophic incidents. It is this period that lead to legislation aiming to address pollution of inland waters, oceans, air and soil and to protect particular habitats or species. As awareness increased about more subtle impacts and the interdependence of all elements of the biosphere, the need for new approaches to environment protection and natural resource management became apparent. This lead to development of legislative measures aiming to ensure sustainable utilisation of environmental resources and to address environmental protection in a more integrated and holistic fashion. References to the need to apply the "ecosystem approach" or the aim of ensuring ecosystem health started to appear in legislation and international agreements. As the environmental issues, the perception of these, and the priorities assigned to them continue to shift and develop, the legislative and regulatory approaches, norms, techniques, and institutions used to manage these effectively and efficiently necessarily will evolve further.

1.3. Various objectives of protection

The evolution of environmental protection, in combination with over-exploitation of resources and increased levels of anthropogenic contaminants in the environment, has resulted in a range of different objectives of environmental protection.

From a review of existing agreements and legislation that aim at protection of the environment (see Chapter 6) the most frequently cited objectives of protection can be assembled within five categories: 1) the conservation of biological diversity, 2) the protection of species, including rare and endangered species, 3) the maintenance of ecosystem structure and functioning that ensure the sustainability of the life support and services provided by ecosystems, 4) the precautionary approach in environmental management, 5) the conservation of resources (biological, geological, etc).

The objects of protection therefore, all cutting across ecosystem services, can be:

- Biological resources that are exploited for various purposes (fisheries, forestry, agriculture);
- Endangered species (with a rationale identical to that of human protection);
- Regional landscapes, which are assigned important value by society for various reasons (e.g., aesthetics, cultural or religious heritage, uniqueness, critical services);
- Whole ecosystems, which are assigned important value by society, such as in view of maintaining biodiversity, for example (coral reef, tropical forest, tundra);
- Various resources that are exploited for various purposes (biological resources for human and animal feeding, mineral resources for mining, geologically stored gas and oil resources exploited for energy production).

Environmental protection clearly targets a greater scope of protection than the narrowly focused human health protection programs that only target the organisms of one single species, *Homo sapiens* (Bréchignac, 2001), although occasionally, a human targeted legislation may have resulted in environmental protection (like the UK Public Health Act, in 1875). However, if one considers the holistic perspective of health declared by the WHO, then many of the broader environmental protection objectives could be interpreted as contributing to the general well-being of humans.

The philosophy within which the objectives of environmental protection are driven can take two formats: a curative philosophy that takes counteracting measures to face an already existing harm (existing contamination of the environment); a philosophy of prevention that aims to prevent in advance the occurrence of harm (Council of Europe, 2009). For example, the protection of endangered species belongs to a curative philosophy, such species being subject to extinction due to their exposure to stress (not mentioning “natural” extinction which also exists). This need to protect endangered species – a fairly common goal within current legislation – is somehow a proof of protection failure. Indeed, protecting endangered species would not be needed if the environment would have already been afforded an efficient level of protection against stress. The second category, prevention, should be prioritized as it places emphasis on anticipation of potential hazards, therefore reducing or avoiding them at the source. Capacity for anticipation in environmental protection is also an aspect of paramount importance because of resilience phenomena in ecosystems: harmful effects may not become apparent immediately upon exposure to the stress, but much later, and with an extent that will require a similar time scale for any counteraction, if at all possible, to become corrective. This is due to ecological processes, which often prove to have slow kinetics, as well as large delays in time and space and indirect and unexpected effects often associated with crossing thresholds of stress (see sections 2.3 and 2.4). With that respect, the experience gained so far has shown that it is often very late, if not too late, for any reasonable counteraction to successfully be opposed when harm becomes apparent.

It is therefore not surprising that different types of environmental assessments have evolved in response to this variety of objectives. Cormier and Suter (2008) have identified up to four general types of assessment - condition assessments to detect chemical, physical and biological impairment; causal pathway assessments to determine causes and identify their sources; predictive assessments to estimate environmental, economic, and societal risks, and benefits associated with different possible management actions; outcome assessments to evaluate the results of the decisions of an integrative assessment - depending on whether the analysis runs from cause to effect or from effect to cause. This is driving in particular to the need for an integrated framework by means of which practitioners of the various approaches to environmental assessment would better see how these are conceptually linked.

1.4. Environmental Protection in the Context of Radiation

With respect to radiation hazards, several international institutions are promoting various actions and initiatives.

1.4.1. Historical context

For more than one decade, various international organisations specialized in the field of radiological protection have embarked on reflections and programmes dedicated to protection of the environment against radiation. Historically, after IAEA first raised the issue in 1988 in relation to the practice of dumping nuclear waste in the ocean (IAEA, 1988), UNSCEAR published its Report entitled “Effects of radiation on the environment” (UNSCEAR, 1996) where some maximum safe dose levels were mentioned for populations of non-human biota, following which the International Union of Radioecology (IUR) assembled a Task Group in 1997. At the same time, the International Commission on Radiation Protection (ICRP) suggested that environment protection *per se* was not deemed necessary as it was implicitly covered by human radiation protection standards. One important driver to this first initiative, therefore, came from the scientific community which argued that environmental protection deserved specific attention and could not be simply covered through human radiation protection. At this time, the US DOE developed a national protection system based upon a graded approach and implemented the “RESRAD Biota” tool (DOE, 2000) using the 1996 UNSCEAR recommendations, whilst the UK Environmental Agency had already started some related work (Copplestone et al., 2000a and 2000b) and developed its R&D Publication 128 (Copplestone et al., 2001).

This initial context prompted a strong effort, especially in Europe where a series of projects (FASSET, EPIC, ERICA, PROTECT) debated the issue, proposed and worked out a systematic framework for the protection of the environment based upon a concept of “reference organisms.” This effort has been paralleled on a worldwide scale both by the International Commission on Radiological Protection (ICRP), which created its 5th Committee to tackle the issue, and by the International Atomic Energy Agency (IAEA), within its EMRAS programme.

An unprecedented coordination effort across these four organisations, based upon a clarification of the respective roles, is now in progress to achieve the best development efficiency (Bréchignac, 2007). Both IUR and UNSCEAR hold roles that deal with science. IUR operates on the anticipation side: promoting science, with strategic anticipation of new directions to stimulate. UNSCEAR operates at a later stage on the science that has been achieved: consolidating the already published knowledge and validating it through international expertise. IAEA and ICRP operate both towards more operational purposes, the former designing international standards, and the latter elaborating recommendations in a regulatory context. All four organisations therefore synergistically contribute to radiological protection with their own complementary skills and attributes (Figure 1).

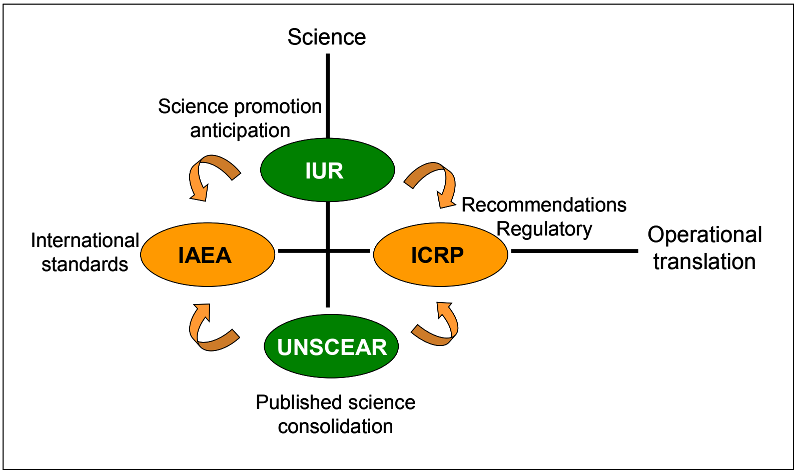


Figure 1. Illustration of the synergistic relationships between four international organisations active in the field of environmental radiological protection (International Union of Radioecology, International Commission on Radiological Protection, International Atomic Energy Agency, United Nations Scientific Committee on the Effects of Atomic Radiation) - (From Bréchignac, 2007).

1.4.2. United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR)

UNSCEAR is an international governmental entity under the UNEP programme of the United Nations which has been assigned the task to consolidate the scientific knowledge on the “effects of atomic radiation” based upon expert review of the published literature. It has published world wide authoritative scientific evaluations of the knowledge in the field, especially two reports which are relevant to environment protection (UNSCEAR, 1996; 2010). Based upon its reviews, UNSCEAR has proposed two reference values of radiation dose corresponding to maximum safe levels to populations of non-human biota, 1 $mGy.d^{-1}$ for terrestrial animals, and 10 $mGy.d^{-1}$ for aquatic animals and terrestrial plants.

1.4.3. International Union of Radioecology (IUR)

The early work of IUR identified an initial approach to environment protection featuring “reference flora and fauna,” as recommended in its report “Protection of the Environment: Current Status and Future Work" (IUR Report n° 3, 2002).

The Union also started to promote deliberations in a much larger context, which were addressed through several initiatives: a Consensus Statement obtained through the Oslo Consensus conference on “Radiation protection in the 21st century: ethical, philosophical and environmental issues” (IUR/NRPA, 2001; Oughton and Strand, 2003), a statement published by the IUR Board of Council entitled “Protection of the environment in the 21st century: radiation protection of the biosphere including humankind” (Bréchignac et al., 2003), and a continuing effort through Task Groups designed to unravel the identification and prioritisation of the research requirements in this field of protection of the environment from ionising radiation. IUR recommendations have been elaborated along these lines and published in 2006 (IUR report n°4, 2006; IUR Report n°5, 2006).

IUR also dedicated significant effort to: 1) addressing the improvement of communication, particularly at an international level, on issues related to the protection of the environment from ionising radiation via setting up a virtual network for discussion, and 2) progressing towards the ultimate general objective of constructing a Worldwide Research Network in Radioecology.

Overall, two main recommendations for improvement of understanding through research in this field have been derived and directed towards the scientific community:

- relationships between low chronic doses and their resulting effects on a number of wildlife species with proper understanding of propagation towards the ecosystem level, and
- potential interactions within multi-pollutant mixtures and proper understanding of the overall resulting effect as opposed to that from single stressors.

1.4.4. International Atomic Energy Agency (IAEA)

The IAEA is especially concerned with the international harmonisation of standards with the overall objective to enhance the capabilities of Member States to simulate radionuclide transfer in the environment and, thereby, to assess exposure levels of the public and in the environment in order to ensure an appropriate level of protection from the effects of ionizing radiation, associated with radionuclide releases and from existing radionuclides in the environment.

For a long time, the Agency has supervised work and international exercises in the field of radioecological modelling in view of reducing uncertainties that remain in the predictive capability of environmental models. Currently, its major contribution to the field of environment protection is within its EMRAS II Programme which, based upon the “reference organism” concept, aims to improve capabilities in the field of environmental radiation dose assessment for non-human biota by acquiring improved data for model testing and comparison, reaching consensus on modelling philosophies, approaches and parameter values, and developing improved methods as well as exchanging information.

The IAEA identified the goals to protect the environment from deleterious radiation effects according to some core principles (IAEA, 2002). These goals are expressed as follows:

- any radiation exposure should not affect the capability of the environment to support present and future generations of humans and biota (principle of sustainability);
- any radiation exposure should not have any deleterious effect on any species, habitat, or geographic feature that is endangered or is under ecological stress or is deemed to be of particular societal value (principle of conservation);
- any radiation exposure should not affect the maintenance of diversity within each species, amongst different species, and amongst different types of habitats and ecosystems (principle of maintaining biodiversity);
- the management of any source of radiation exposure of the environment should aim to achieve an equitable distribution of the benefits from the source of the radiation exposure and any harm to the environment resulting from the radiation exposure, or to compensate for any inequitable damage (principle of environmental justice); and
- in decisions on the acceptability and appropriate management of any source of radiation exposure of the environment, the different ethical and cultural views held by those humans affected by decisions should be taken into account (principle of respect for human dignity).

1.4.5. International Commission on Radiological Protection (ICRP)

The ICRP (a Non Governmental Organisation) provides recommendations for radiological protection that are authoritative worldwide and largely taken into account in most national and international legislation. The long standing paradigm of ICRP, which had for a long time subordinated environment protection to the protection of human beings (ICRP 60, 1991), has provided a major driver to the development of the environmental protection issue in radiation protection, as pointed by OECD/NEA (2007) which specifically stated: "*The current system of radiological protection, not having been designed for this purpose, is a weak tool to demonstrate the level of radiological protection afforded to the environment*." The OECD/NEA statement was made from the perspective that the system of radiological protection for the environmental has historically centred on the rationale that if humans are protected so too, by default, is the remaining environment.

The shift to specifically addressing the environment started in 2000 with a Task Group precursor of the creation, five years later, of a dedicated 5th Committee. The construction of a specific system framework for the radiological protection of the environment (actually non-human biota) against ionizing radiation has been formulated based upon a concept of "Reference Animals and Plants" (ICRP, 2003). Indeed, legislation within several countries now requires protection of the environment to be explicitly and transparently demonstrated, irrespective of human exposures (Pentreath, 2009).

The ICRP acknowledges that, in contrast to human radiological protection, the objectives of environmental protection are both complex and difficult to articulate. There is no simple or single universal definition of 'environmental protection' and the concept differs from one country to another, and from one circumstance to another. Other ways of considering radiation effects are therefore likely to prove to be more useful for non-human species, such as those that cause early mortality, morbidity, or reduced reproductive success (ICRP 108, 2008).

The ICRP has therefore stated that its aim is that of: *"… preventing or reducing the frequency of such radiation effects 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".* But in achieving this aim, the ICRP recognises that exposure to radiation is but one factor to consider and that, indeed, it is often likely to be but a minor one.

The goals of environmental protection have been recognised and described so far as being highly variable because they arise from a number of different situations encountered by various populations with different cultural heritages, and leading to various perceptions of what is to be protected, what protection means, and even, how to define the environment. This context has been particularly discussed by ICRP (ICRP 91, 2003) which felt it was *"reasonable to ask: can one identify any common ground for a consensus on such issues?"* The following chapter therefore is focused on attempting to answer this question.

1.5. The Ecosystem: a Core Concept for the Protection of the Environment

Central to this discussion still lies an ambiguity as to what "protection of the environment" actually means in terms of its objectives. Indeed, as described in more details above, 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 (climate control, water regeneration, provision of material and biological resources…). Confusion often arises from the fact that the "protection of the environment" terminology is often abusively employed as a generic expression where it actually only addresses a few of the specific aspects mentioned above.

As a consequence, it is of great interest to look for a possible unit within the environment that would capture at the same time most of its various perceptions, that would be common to landscapes, abiotic components (seas, rivers, lands) or resources (mining), biotic components (microbes, plants, animals, humans) or resources (logging, agricultural harvesting, fishing,…), regional biomes (tundra, tropical forest, coral reefs, swamps) etc…, that is a unit that would encapsulate all the different objects of protection mentioned above.

The **ecosystem** concept perhaps best captures all these perceptions (for a definition of the ecosystem concept, see the next section 1.6). It reconciles most of the various views, biotic and abiotic components of the environment, human and non-human related components interlinked through the provision of life support and services. The concept introduces the recognition that life is maintained through ecosystem structures and functions in such a way that no life form can evolve and survive in isolation from other life forms (in other terms, human and non-human life are not dissociable), or its abiotic surroundings (Odum, 1983; Ramade, 1994, 1995). Focusing on ecological systems has another substantial merit as it allows definition of an overarching goal of environment protection: preserving life sustainability, irrespective of it being human or non-human, as both are strongly dependent on each other (Bréchignac, 2001; Bréchignac and Doi, 2009).

All the present goals of environment protection, which are expressed either through the variety of current legislations, or founding the development of frameworks for ecological risk assessment, whether in general or in the context of radiation, are related to, or contributing to, this overarching goal, which therefore could serve as a basis for establishing a useful consensus as a starting point. This is indeed why the Convention on Biological Diversity has centred its reflexion and undertakings on the ecosystem concept that it defines as "*a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit*".

A possible formulation of this overarching goal for environmental protection, which could form the baseline policy driving all its aspects including radioprotection, could therefore be expressed as: "*preventing ecosystem ecological disturbance likely to impair the long-term maintenance of life and the sustainability of resources*."

1.6. What is an ecosystem in this context

Odum (1971), one of the major contributors to modern ecology, defines an ecosystem as "*a unit that includes all of the organisms (the "community") in a given area interacting with the physical environment so that a flow of energy leads to clearly defined trophic structure, biotic diversity, and material cycles (exchange of materials between living and nonliving parts) within the system*".

The concept stresses that an ecosystem consists of a self-sustaining community comprised of interdependent populations of living organisms (plants, insects, animals) and their natural environment (soil, sediment, atmosphere, water, light). Ecologists refer to the first as the "biocenose", and to the second as the "biotope". This defines a system of "complex interactions" between species and between species and their physico-chemical environment featuring a food chain through which energy flows, and the biological cycles recycle essential nutrients and wastes.

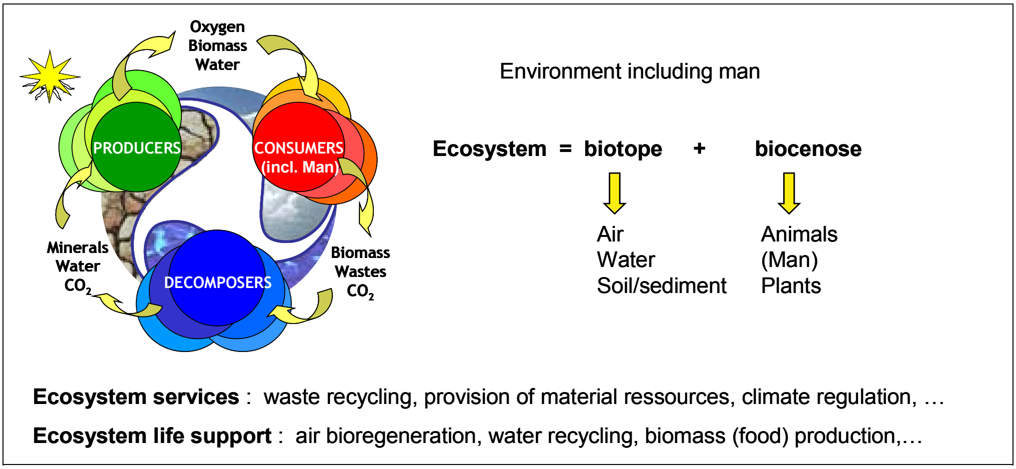


Figure 2. The interdependence relationships within an ecosystem featuring the association of a biotope (air, water, soil/sediment) with a biocenose (animals, plants, microbes) with loops of material and energy transformation which provide ecosystem services and life support functions.

One key point of ecosystems is that their functions provide life support and services. Self-sustainability of the ecosystem is paramount as it is an emerging property which founds the long-term survival of any life form. Self-sustainability is bound to the interdependence relationships which link the various elements of the ecosystem through exchanges of materials and flows of energy as depicted on Figure 2. The biotope and the biocenose therefore form a non-dissociable system, in local equilibrium, but capable of evolution and adaptation to the ecological context. The modification of any parameter within the ecosystem, as induced by a toxicant for example, is prone to yield a disruption of the ecological (metastable) equilibrium or to be hidden within the resiliency of the system (itself driven by diversity or redundance).

The scientific understanding of ecosystems has been much influenced by the theory of complex systems (addressed in more details in chapter 2.3) explaining observed features such as "hierarchy" and "emerging properties".

2. The Ecosystem Approach

There is a wide array of justifications which have led to designing an integrated concept for environment protection along a so called “ecosystem approach”.

2.1. Origin

Common regulatory perceptions of the value of nature are based on two major concepts: the need to safeguard biodiversity and the will to preserve life-supporting functions within natural systems such as maintenance of safe drinking water, clean air and safe non-contaminated food, both of which depend on community and ecosystem level phenomena. The effects of stressors on biological systems can occur at different levels of organisation: cells, organs, individuals, populations and communities. From an ecological point of view, populations and communities are of greater concerns. Nowadays, a significant interest is devoted to the protection of the environment in an ecosystem-centred perspective (Bréchignac and Doi, 2009). Even though the majority of regulatory structures and approaches for assessing the effects of toxic substances are largely based on the outcome of single-species laboratory tests, often at individual level, there is a widespread recognition of the limitations of such tests (Barnthouse, 1993; Suter and Bartell, 1993). These include criticisms of the extrapolation from single-species toxicity tests to population and ecosystem effects and from the laboratory to the field that toxicity tests do not consider bioaccumulation of contaminants, and ignore temporal changes, indirect effects and multiple stressor effects. There is a growing awareness nowadays among policymakers and scientists that assessment studies should adopt a systems approach. While there are many different definitions of the ecosystem approach within the context of environmental impact assessment, this is seen as a holistic strategy for risk characterisation which integrates toxicological knowledge with ecological understanding.

The expression “ecosystem approach to environmental protection” indicates that concern is focused on three main classes of effects:

- The effects of stressors on biota
- The effects of stressors on species populations as components of the ecosystem;
- The systemic effects of stressors on the ecosystem as a whole.

Although the first kind of effects is of paramount importance for protecting individual organisms, the analysis developed below will focus on the systemic effects that are particularly important in an ecological perspective and that more strictly comply with commonly shared assumptions such as those stated by the Convention on Biological Diversity (http://www.cbd.int/convention) :

> “Ecosystem functioning and resilience depends on a dynamic relationship within species, among species and between species and their abiotic environment, as well as the physical and chemical interactions within the environment. The conservation and, where appropriate, restoration of these interactions and processes is of greater significance… than simply protection of species.”

In principle, the protection of the ecosystem from the ionising radiation, from an ecological perspective, requires that three main issues are addressed:

- How to assess the effects of ionising radiation at level of the ecosystem.
- How to identify and select suitable *objectives* for the protection of the ecosystem.
- How to measure the level of achievement of these objectives (*indicators*).

The above listed issues are in agreement with traditional principles generally accepted within the frame of management theories based on the identification of effects, objectives and indicators.

As previously emphasised, “ecosystem” and “systemic level” are the keywords of the present analysis. Therefore, not only the structure, but also selection, evolution and the functioning mechanisms of the ecosystem are central to this analysis as they are considered to be significant beyond the response of individual organisms. Before arguing that an ecosystem approach is indeed justified in the protection of the environment from radiation, the following sections summarise the current scientific thinking on ecosystem-level processes.

2.2. The Goal to Protect Populations

In the frame of many ecological risk assessments the protection objective is set at the population(s) level as being the most relevant and pertinent object of protection. Nonetheless, the overwhelming majority of assessments of ecological risks of environmental chemicals (for example) are still based on an individual-level approach arguing that if all individuals from a given population are protected, then this population will be protected as well. However, a population may also be affected via interaction with other populations, especially if they consist of individuals that would not be explicitly protected. It is of importance to note that the relative 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, together with the relative ease of performing experiments at the individual level compared to a population level (Kapustka, 2010).

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 that 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 factors necessary for the survival and transmission of the genetic information of the individuals to the next generation.

There have been many published calls for ecological risk assessment that would consider risks to populations, not simply to individual organisms (for a review, see Barnthouse et al., 2008). The main reason for that is that all individual organisms 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. SETAC[6] in particular is advancing the practice of population-level ecological risk assessment by establishing a framework for population-level assessment that includes definition of goals, identification of appropriate assessment methods, specification of data needs for different types of assessment applications, and development of a technical framework for integrating population-level consideration into risk management decisions. 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.

2.3. New Ecological Theories Featuring Ecosystems as Complex Systems

As opposed to the classical approach to presenting the impacts of toxicants upon various aspects of biological and ecological systems, a new framework is now proposed that incorporates complexity theory (Müller et al., 2000). Essentially, the basic format of this framework features two distinct types of structures that concern risk assessment.

Living organisms (left, on Figure 3) have a central core of information, subject to natural selection, that can impose homeostasis (e.g., body temperature) or diversity (e.g., immune system) upon the constituents of that system. The genome of an organism is highly redundant, a complete copy existing in virtually every cell, and directed communication and coordination between different segments of the organism is a common occurrence. Unless there are genetic changes in the structure of the germ line, impacts to the somatic cells and structure of the organism are erased upon the establishment of a new generation.

Above this individual organism level, ecological structures (non-organismal) have fundamentally different properties (right, on Figure 3). Here there is no central and inheritable repository of information, analogous to the genome, which serves as the blueprint for an ecological system. 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 (ecological memory). The many non-linear dynamics and historical nature of ecosystems confer upon the system the property of complexity which, in turn, features specific properties that are critical to how ecosystems react to contaminants.

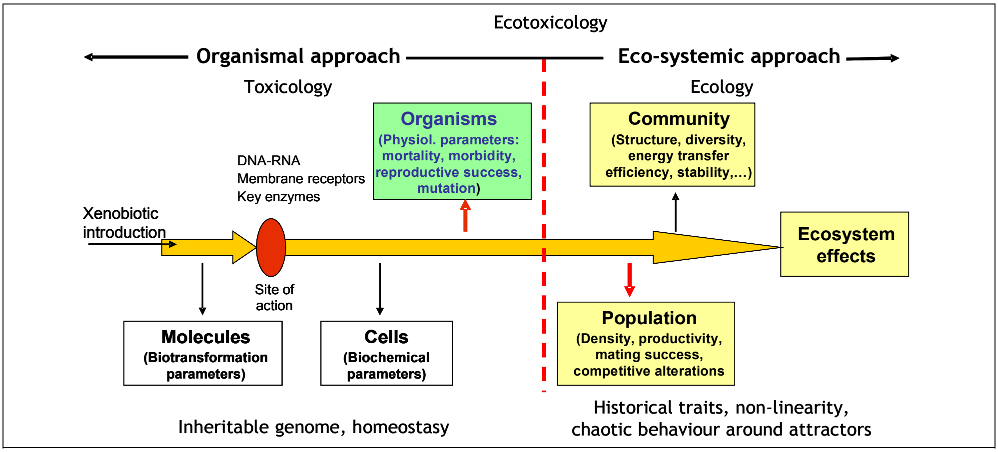


Figure 3. Parameters and indications of the interaction of xenobiotics with all levels of biological organisation within the ecosystem. Toxicological knowledge supports the organismal approach (left) whilst additional ecological knowledge is required to support the ecosystem approach (right) - (Adapted from Landis and Yu, 2004).

[6] SETAC: Society of Environmental Toxicology and Chemistry

In this context of complexity, it is worth mentioning also recent ecological/ecosystem theories, especially those referring to thermodynamics, that are currently being developed (Jørgensen, 2006a, 2006b; Müller et al., 2000; Kay, 2000) as they result in a better description and understanding of the behaviour of complex ecological systems. In addition, resilience theory describes how stressed complex systems can undergo unexpected and possibly irreversible changes (regime shifts) as they pass thresholds and move to alternative stable states (cf. further developments in Chapter 2.4).

Following Cambel (1993), the major relevant properties which are pertinent in the context of ecological risk assessment can be summarized as follows:

- Complex structures are neither completely deterministic nor stochastic, and they exhibit both characteristics.
- The causes and effects of the events which the system experiences are not necessarily proportional.
- The different parts of complex systems are linked and affect one another in a synergistic or antagonistic manner.
- Complex systems can undergo irreversible changes.
- Complex systems are dynamic and not in equilibrium; they are constantly moving targets.

2.4. Ecosystem Stability, Resilience, and Major Regime Shifts

An aim of environmental protection is to keep ecosystems as stable as possible. Terminology of "ecological stability" has been, however, confused for a long time. According to Grimm and Wissel (1997), there are 163 definitions from 70 different stability concepts. Among terms representing ecological stability, "resilience" is one of the most important terms. There are two major definitions in resilience. One is an ability (or rate) of communities to return to a pre-perturbation condition (definition 1; Allison and Martiny, 2008), and the other is the capacity of a system to absorb disturbance and reorganise while undergoing change so as to retain essentially the same function, structure, identity and feedbacks (definition 2; Folke et al., 2004) as shown in Figure 4. Definition 1 is called "engineering resilience" and definition 2 is called "ecological resilience" (Holling, 1996). For the sake of the analysis developed in this report, definition 2 has been adopted.

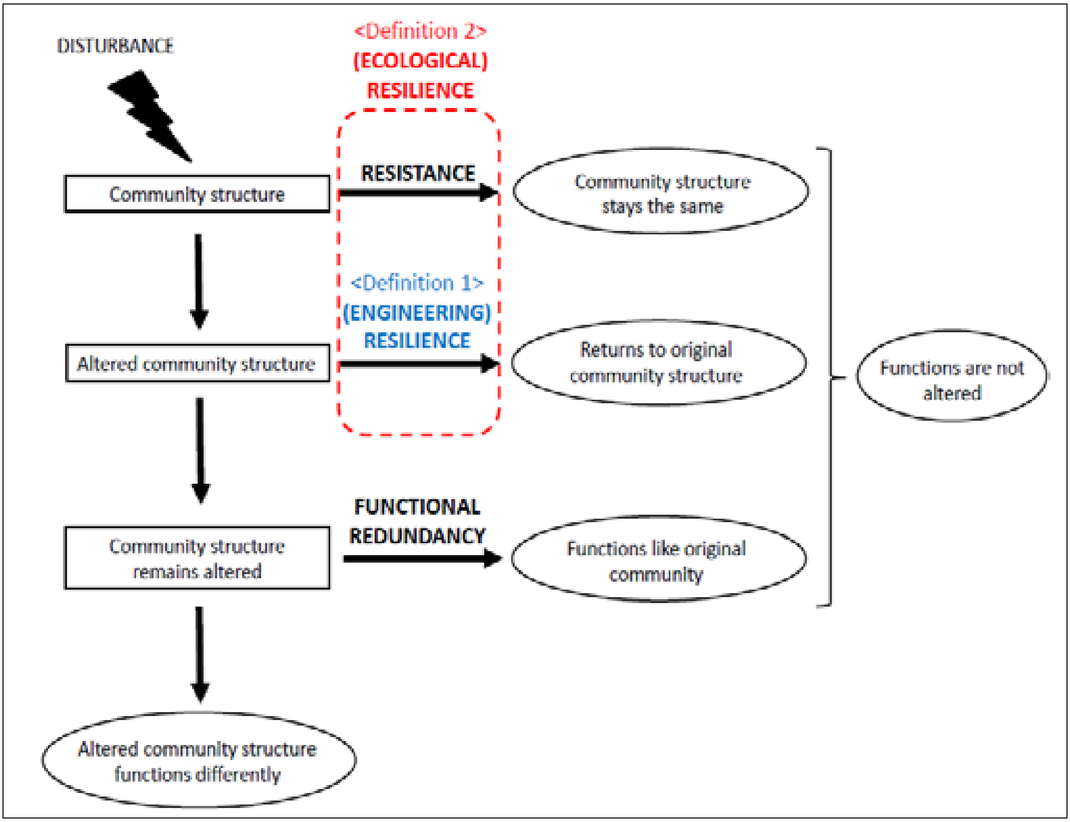


Figure 4. Two major definitions of resilience, according to Allison and Martiny (2008), with some modifications.

Ecosystems are exposed to gradual changes in biotic and abiotic factors, and usually respond to these perturbations in a smooth way. However, in some cases, sudden catastrophic shifts between different ecosystem states, which are called “regime shifts”, are caused by the combination of the magnitudes, frequency or persistence of external forces and the internal resilience of the system. As anthropogenic disturbance or natural factors (e.g., nutrient loading, climate change and habitat fragmentation) decrease resilience, the ecosystem becomes vulnerable to smaller disturbances or gradually changing conditions that it could previously cope with, and is at high risk of shifting into a qualitatively different state (Figure 5). These shifts may be difficult or impossible to reverse.

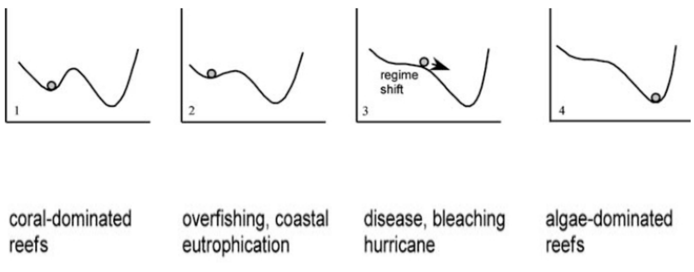
A typical example of regime shifts is a dominance change from corals to macroalgae in the Caribbean (Hughes, 1994; Knowlton, 1992; Folke et al., 2004, Figure 5). It is thought that this regime shift was a result of the following mechanisms. Overfishing and increased nutrient loading from land water run-off decreased herbivorous fish populations. As a result, sea urchins, which competed with herbivorous fish for macroalgae, dominated the coral reefs. For the next step, in 1981, a hurricane severely damaged the coral reefs, though the coral could re-colonise the reefs at this stage because the sea urchin grazed on macroalgae, which competed with the coral. In subsequent years, the sea urchins were, however, hit hard by a disease outbreak. Therefore, the macroalgae were not controlled and came to dominate the reefs. In this example, it is thought that overfishing and eutrophication decreased resilience in the coral reef ecosystem, and hurricane and sea urchin disease caused the regime shift, which still persists today.

Another commonly described example is the shift from pristine clear water lakes with rich submerged bottom vegetation to turbid phytoplankton-dominated lakes, an explanation of which is summarized by Bréchignac (2003) and Scheffer et al. (2001). The regime shift may be caused by both eutrophication (excess nutrients) and/or the addition of herbicides or pesticides. Nutrient accumulation, herbicide damaged plants which competed with phytoplankton, or pesticide-damaged zooplankton which grazed on phytoplankton, lead to decreased resilience and the lake shifts abruptly from clear to turbid with phytoplankton. Examples of regime shifts that have been observed in various ecosystems are summarised by Walker and Meyers (2004).

Resilience decline, which may cause regime shifts more easily, may occur through the following human actions (Folke et al., 2004):

- loss of biodiversity, i.e., removal of functional groups of species and their response diversity, such as the loss of whole trophic levels;
- impact on ecosystems via emissions of waste and pollutants, or climate change; and
- alteration of the magnitude, frequency and duration of disturbance regimes to which biota are adapted.

In some cases, regime shifts may be irreversible (or too costly to reverse). Irreversibility is a reflection of changes in variables with long turnover times (e.g., biogeochemical, hydrological, or climatic) and loss of biological sources and interactions for renewal and reorganisation into their previous state (Folke et al., 2004, Figure 5). Most regime shifts occur from “desired” to “less desired” states in their capacity to sustain ecosystem services to society. These suggest that strategies for sustainable management of desired ecosystems should focus on preventing regime shifts by maintaining resilience. These strategies should be applied to radiation protection of the environment: though there is as yet no demonstration of situations where ionising radiation decreases ecosystem resilience and subsequently causes regime shifts, this is a theoretical possibility, as is the possibility that radiation could be the trigger that pushes an already-stressed ecosystem over a threshold.



A ball represents a state of the ecosystem. In panel 1, the ecosystem is in a stable equilibrium, represented by a valley. In panel 2, resilience, which corresponds to the size of valley, is decreased by anthropogenic or natural factors. In panel 3, the ecosystem is moved to the unstable middle section (represented by a hill) by disturbance or gradually changing conditions. In panel 4, the ecosystem is shifted to a different stable equilibrium. Returning the ecosystem to its previous stable state is difficult (represented by a steep uphill slope).

Figure 5. Resilience decrease and subsequent regime shift from an example on coral reefs (Folke et al., 2004).

The above discussion is not to make the case that all ecosystem impacts must be prevented, but rather to have a concern as to whether a regime shift is likely to be promoted or not, as this may yield quite an undesirable state of the ecosystem.

2.5. Impact of Ionising Radiation on the Ecosystem

The previous section argued the case for considering ecosystem-level processes, as they are considered to be significant beyond the response of individual organisms. However, one basic question should be answered: can ionizing radiation affect mechanisms that are eminently of ecological rather than merely of biological nature? The following sections argue that this is indeed the case and that an ecosystem approach to environmental protection is justified. Beside theoretical considerations with respect to potential impacts of radiation at the ecosystem level, numerous ecological observations have been reported from contaminated areas, especially in the CIS countries. An overview of such observations is given in the following sections.

2.5.1. Theoretical Aspects

The peculiarities of the radiation responses in ecosystem are the result of differences between the organization of individual organisms as groups of cells and tissues, and organization of ecosystems as communities of populations.

The exposure of individual organisms to ionising radiations can cause increases in lethality and morbidity, modifications of fertility and shortening of life span. These effects directly influence the size and the age structure of the species populations and can be modelled, in principle, by determining the values of some population parameters, such as the birth and death rates, as functions of the doses to individuals of different age classes (Woodhead, 2003). By altering the relative abundances of the different species in a given ecosystem, the effects may influence the interaction among the species and, consequently, the structure of the communities and of the whole ecosystem.

In an irradiated ecosystem, absorbed doses to different species of organisms may vary considerably as a result of differences in feeding habits, home range, habitats and duration of lifespan and other factors. Populations that are sheltered from external irradiation, or species feeding on non-contaminated food items, receive lower doses than other members of the biological community. Having minimal contact with ionizinaspects"…g radiation, these species take advantage in ecological competition over more damaged species. This can also arise from differences in radio-sensitivity where absorbed doses to different species of organisms may lead to differential fitness/mortality between species or populations.

In the conditions of chronic exposure, the total doses accumulated per lifetime of organisms can vary considerably between species. Species with relatively short lifespan accumulate lower lifetime doses than long-lived species. Therefore, short-lived individuals have better chances to survive in an irradiated ecosystem.

Beside the lifetime doses, radiation effects are also determined by doses accumulated during the most sensitive periods of development. In the conditions of radiation exposure, better survival is expected for species with short period of embryonic development.

Upon acute exposure of an ecosystem, the total radiation effect depends upon what proportions of each population were in radiologically sensitive stages of ontogenesis at the period of radiation exposure (Krivolutsky, 1983). An ecosystem might survive from a massive radiation during the winter, but suffer drammatically if radiation is received in spring or early summer (i.e., during the growing or reproductive season).

Inter-specific interactions are essential features of ecosystems and form the theoretical basis for many types of indirect effect manifestations, as summarized in Table 1. The important point to make here is that due to such interactions, ecosystem response to radiation stress cannot be grasped through individual organisms.

Table 1. Emergence of potential indirect effects caused by altered sizes or fitness of populations exposed to ionising radiation (two interacting species).

Type of interaction	Induction of indirect effect
Competition Mutualism and proto-cooperation Parasitism and predation	Direct exposure of any of the two populations of species can cause indirect effects on the other
Commensalism Amensalism	Direct exposure of the neutral population of species can cause indirect effects on the benefited (or adversely affected) species
Neutralism	Direct exposure of any of the two populations of species do not cause any indirect effect on the other species

From an ecological point of view, the effects of, and exposure from, ionising radiation can be grouped in four broad categories:

- Alteration of the characteristics (i.e. size, structure, fitness) of each species population.
- Alteration of the mechanisms of interaction between different species populations.
- Alteration of the ecosystem structure (as a consequence of the two above).
- Alteration of the physical properties (abiotic part) of the ecosystem.

Currently, the ecological effects of radiation, i.e. effects on the interactions between species populations in ecosystems, are insufficiently studied both experimentally and theoretically. In the majority of radiobiological experiments conducted so far, the irradiated objects were cells, tissues and individual organisms isolated from their natural environment. The results of such experiments demonstrate the radio-sensitivity of the individual organisms, but not the sensitivity of populations or ecosystems. From the field observations on the areas highly contaminated with radionuclides in the CIS countries, it is evident that manifestations of radiation effects in natural ecosystems have some specific features which cannot be understood from the standpoint of individual radio-sensitivity of organisms.

2.5.2. Empirical studies and experimental investigations

Most of the above-mentioned theoretical considerations are supported by observations resulting from many in situ empirical studies and experimental investigations.

2.5.2.1. Literature compilation

The effects of ionising radiation on living organisms have been the subject of a great deal of studies. Information from over 1000 references on a range of animals and plants is compiled in the FASSET database (Real et al., 2004), more recently updated in the FREDERICA database (Copplestone et al., 2008). This literature review listed effects such as the increase in mortality and morbidity, life shortening, and reduced fertility and fecundity which, as we have previously emphasised, induce changes in the size and the age structure of the exposed populations and can initiate, in principle, modifications of the ecosystem structure[7].

2.5.2.2. Field evidence for various radiation effects on ecosystem structure / processes / interactions

The occurrence of several radioactively contaminated areas in the CIS countries have allowed the study of radiation damage to populations and integrity of irradiated ecosystems in the field. Such irradiated ecosystems have often been demonstrated not to follow directly the individual radio-sensitivities of biological species obtained from short-term acute exposure in laboratory conditions (Krivolutsky, 1993). Processes of competition and predation/parasitism may aggravate the radiation damage to the populations. Furthermore, various compensatory mechanisms (e.g. interactions and feedback loops) exist, which can cause an adaptation to stress caused by ionizing radiation of populations and the ecosystem as a whole. The relative role of different ecosystem processes in aggravating or veiling the effects of radiation is still poorly experimentally addressed.

Many factors directly or indirectly modify the radiation effects in natural populations and ecosystems. Among these are: differences in exposure (between different species, life stages or times of year), differences in radio-sensitivity (between species and between different life stages), recovering capacity, life history characteristics, and trophic relationships with other species.

Different species of organisms demonstrate tremendous differences in radio-sensitivity; the lethal doses (LD for adults) of acute exposure range from a few Grays (mammals) to thousands of Grays (insects, microorganisms) (Odum, 1983; UNSCEAR, 1996). This results in higher vulnerability of particular groups of organisms to radiation exposure. It should also be noted that radio-sensitivity of organisms may vary considerably at different stages of life cycle, e.g. some insects at young stages of development can be killed by a few Grays of radiation exposure, whereas the adult insects of the same species are very radio-resistant (Hertel-Aas et al., 2007; Odum, 1983; Krivolutsky, 1983).

The survival of affected populations or species also strongly depends on their capacity to restore rapidly the numbers of organisms in the population. For example, if 99% of the cells in a bacterial colony were destroyed by irradiation, the bacterial population, having high reproductive potential, can rapidly regenerate from the remaining 1% of cells. Highly productive and rapidly growing species (e.g. insects) also have fair chances to restore the damaged populations. In contrast, long-lived species with low reproduction potential are very vulnerable to losses of individuals in the population. The restoration of these species in irradiated ecosystems may occur due to migration of healthy organisms from the neighboring non-irradiated areas.

In the case of acute exposure, asexually reproducing organisms, being independent genetically, are more likely to restore their populations, than sexually reproducing species. In populations of sexually reproducing species, a

[7]Large limitations, however, still affect the pertinence of the available data which are dominated by certain types of organisms/test species, restricted to quite artificial conditions and non realistic laboratory studies, etc …

decline in the total numbers of organisms drives to lower probability of mating, which in turn, leads to further decrease of the population size. An example is given from the land highly contaminated in 1986 as a result of the Chernobyl accident[8]. From another point of view, under conditions of chronic exposure at low-to-moderate dose rates, sexually-reproducing species generate organisms with more variable characteristics, and the resulting populations have better chances for gradual adaptation to radiation.

All these potential effects on individual components of the ecosystem must be considered in the context of reality which is dominated by multi-species complexity. Even components that have a high capacity to regenerate their populations may lose the competitive advantage over other components that are less radiosensitive and are released from competitive (or predation) pressure.

Radiation effects at the population level result in ecosystem processes and interactions between species becoming perturbed, with resulting ecosystem-level effects. Many of these effects may be indirect (i.e. not a direct consequence of radiation effect), and in that sense have many similarities with effects caused by other natural or anthropogenic stressors. For some ecosystem components or processes these indirect effects may even be positive (e.g. release of prey populations from predation).

Many of these radiation effects can be considered to operate through intensification of natural selection, firstly by acting on the genetic, physiological or ecological variability between organisms within the population[9] and secondly by causing the decrease or removal of weak, or non-adapted, organisms from the population due to competition, predation or parasitism (see below).

Effects on ecosystem engineers / habitat-building species - One of the most destructive consequences for an ecosystem is associated with the radiation damage to the dominant plant species. With a loss of these key habitat-building species (e.g. pines in pine forest), the habitat is severely altered, and ultimately the whole ecosystem may be destroyed. An example is given from the territory highly contaminated in 1986 as a result of the Chernobyl accident[10].

Changes in abundance of decomposers - Appearance of considerable amounts of dead or weakened organisms in a heavily irradiated ecosystem forms an additional source of food for the organisms from the trophic level “decomposers”. Rapid increase in numbers of decomposers aggravates the damage to weakened organisms, preventing their recovery. At high levels of radiation exposure, the decomposers may themselves suffer from radiation. In this case, mineralization of dead organic matter is slowed down, reducing recirculation of biogenic elements in the irradiated ecosystem.

Disturbances in interactions between trophic levels - At levels of radiation exposure which are not high enough to cause serious damage to plants, the radioecological effects manifest themselves in relatively radiosensitive animal populations, usually belonging to the trophic levels “primary consumers”, and “predators”.

Almost all interactions in ecosystems are of “predator-prey” type. Usually, in an ecosystem, the number of preys is limited by predators; in turn, the number of predators is limited by the availability of prey. The self-maintained mutual limitation in each pair “predator–prey” pair provides an opportunity for many species to co-exist in the same ecosystem, sharing resources and forming complex webs of trophic relationships. When irradiated, either prey or predators become more depressed by radiation (different radio-sensitivities) as compared to their trophic partner. The ecological consequences of the imbalance in the “predator-prey” trophic pair are of two types, depending on the trophic position of the most damaged species.

In the situation “damaged prey - healthy predator”, the weakened organisms of damaged prey population more easily become victims for predators[11], resulting in the rapid growth of predators numbers. Intensive removal of

[8] **Example.** Area highly contaminated in 1986 as a result of the Chernobyl accident, 3 km to the south of Chernobyl NPP (Kopachi, place Izumrudnoje); total beta activity of soil 1.1E+05 $Bq.kg^{-1}$. Field studies, 1987-1989, on the local population of brown frog (*Rana arvalis)*. In 1987, over 33 % of the laid frog’s eggs within the contaminated area remained non-impregnated or partially impregnated. In 1988, the portion of non-impregnated frog eggs was 27 %; in 1989 - 3 % and stabilized at this level. Control: 0.01 % non-impregnated frogs’ eggs. (Cherdantsev et al., 1993).

[9] **Example.** Area contaminated in 1986 as a result of the Chernobyl accident (Site "Izumrudnoje" 3 km to the south-east from the Chernobyl NPP); levels of contamination: 3.33E+06 $Bq.m^{-2}$ of Cs-137 in soils. Local population of bird Great tit (*Parus major)*. Field studies of bird's eggs (1989, 1992). In eggs, laid by great tit within contaminated area, the variability in size and shape was higher comparing with the control eggs from non-contaminated areas. The variation coefficients of egg length, diameter and shape index were 5.08, 2.12 and 4.29% respectively (control: 2.15; 2.04; 2.88% respectively). Also some decrease of egg size was observed (Ryabtsev, Lebedeva, 1999).

[10] **Example.** Area contaminated in 1986 as a result of the Chernobyl accident. The highest external irradiation levels were recorded in the area 1.5-2.0 km to the west from the Chernobyl nuclear plant near the village of Yanov. The area was covered with pine stands of 40-50 years old. By the autumn of 1986, all the pines perished here (the so-called “Red forest”), the external irradiation doses were above 100 Gy (Abaturov et al., 1990; Kozubov and Taskaev, 1994).

[11] **Example.** Area highly contaminated with ^{90}Sr in 1957 as a result of the Kyshtym radiation accident, Southern Urals, Russia. Field studies during the 1990s on local populations of European wood mice *Apodemus sylvaticus*. Mice containing higher concentrations of ^{90}Sr in their bodies became victims of predatory birds more frequently than less contaminated animals. About 80% of mice eaten by predatory bird *Buteo buteo* had about 3.5E+04 $Bq.kg^{-1}$ of ^{90}Sr in their bones; whereas in mice caught by mechanical catchers in the same area only 20% had this level of ^{90}Sr in bones, and over 60% specimen had lower concentrations - about 3.7E+03 $Bq.kg^{-1}$ (Lebedeva, Ryabtsev, Beloglazov, 1996).

prey by predators lead to further decline of the damaged prey population, down to extinction in extreme cases. Following the decline in prey numbers, the predator population also decreases because of starvation, or switches to other types of prey. An example is provided from the area contaminated in 1957 as a result of the Kyshtym accident[12].

Effects on parasite-host interactions - Parasites may be considered as specific types of secondary cponsumers, which are feeding on their hosts, without killing them immediately. Normally, host organisms have some protective mechanisms against parasites, and the infestation of a host population with parasites remains limited. Radiation causes a decrease in the capacity of the immune system of the hosts to resist against parasites. Being more radio-resistant than hosts, various parasites find favourable conditions for growth in weakened hosts. Very high infestations of plants and animals with various parasites have been recorded in the areas of radiation accidents. An example is provided from the area highly contaminated in 1957 as a result of the Kyshtym accident[13].

Effects on competition between organisms, populations or species - As some organisms from a population die, the competition for resources within the population decreases. The amount of feeding and space resources available for each remaining organism increases. Having more abundant or more easily available resources, the surviving organisms produce a greater number of progeny, thus helping the population recovery. An example is provided from a water body contaminated from the industrial activity of PA "Mayak"[14]. The same may apply regarding competition between populations of different species competing for the same ressources, leading to changes in community structure (see also Example in foot note 11).

Changes in apparent radio-sensitivity in the overall ecosystem - The overall ecosystem radio-sensitivity may change in an irradiated ecosystem, through i) a gradual increase of the radio-resistant populations due to natural selection[15], or ii) replacement of the radio-sensitive populations by other species with similar ecological functions[16] or migration of unaffected populations from adjacent areas[17]. The relative area of radiation damage is of great significance. If this area is small, the local population is maintained due to regular entry of migrants from the adjacent non-damaged areas. In a chronically exposed ecosystem, the deletion of some species is registered by biologists as a decrease in biodiversity. An example is provided from the area contaminated in the 1950s as a result of the activity of PA “Mayak”[18].

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[12] **Example.** Area contaminated in 1957 as a result of the Kyshtym accident. High level of soil contamination with ^{90}Sr: 6.7-12.6E+07 Bq.m^{-2}. 11 years after the radiation accident, afield study of soil invertebrates showed that numbers of predators and plant feeders were 66% and 56.6%, respectively of the control. Numbers of litter feeders were very low, only 0.6% of the control values (Krivolutsky et al., 1993).

[13] **Example.** Area highly contaminated in 1957 as a result of the Kyshtym accident. Levels of soil contamination with ^{90}Sr: (6.66-12.6)E+07 Bq.m^{-2} (initial contamination); (4.86-9.2)E+07 Bq.m^{-2} in the late 1960s. Field study in the late 1960s (summer)on thepopulation of short-tailed vole *Microtus agrestis*. There was aweakening of the organisms’ resistance to blood parasites (*Leucocytgrenarinae mieroti*):frequency of infected animals in contaminated area (52.4 %, N=204) was more than six times greater than in the control (8.1%, N=111); concentration of parasites in blood of contaminated animals was an order of magnitude higher (40-60 parasites per 100 leucocytes) than in the control(1-8 parasites per 100 leucocytes) (Ilyenko, 1974).

[14] **Example.** Water body, contaminated from industrial activity of PA "Mayak", Southern Urals, Russia. Contamination of water: 14.8E+03 Bq.L^{-1} of ^{90}Sr and 300 Bq.L^{-1} of ^{137}Cs.. Field studies during 1983-1986 on thelocal population of pike *Esox lucius*. On artificial incubation of pike roe, the percentage of embryos with abnormalities (13%) was 10 times higher than in the control (1%). Larvae of pike having developmental anomalies died in the first month of life. At the same time, observations show that the pike population persisted in the contaminated water body for many years; growth and nutritional status of pike were good (Smagin, 1996).

[15] **Example.** Area highly contaminated in 1957 (Kyshtym accident).Levels of soil contamination with ^{90}Sr: (6.66-12.6)E+07 Bq.m^{-2} (initial contamination); (4-7.56)E+07 Bq.m^{-2} in 1978. Studies of 1978 on local population of European wood mouse *Apodemus sylvaticus*. Mice, caught from the wild population, demonstrated an increase of radio-resistance to superlethal acute dose of gamma-exposure (8 Gy; LD50/30=6.6±0.3 Gy for control animals). These mice were the 40th generation of mice living in irradiated environment. All control animals died within 14 days, whereas 3.4% of mice from the contaminated ecosystem survived till the end of observation period (30th day after exposure). The average lifetime after radiation exposure in the group of wood mice from contaminated ecosystem was 10.7±0.3 days (N=57) and in the control 8.7±0.3 days (N=31) (significant at $p<0.001$) (Ilyenko, Krapivko, 1989).

[16] **Example.** Area contaminated in 1986 as a result of the Chernobyl accident; 4-5 km to the west of NPP. Field studies during 1986-1987. The dominant rodent species changed between October 1986 and May 1987. The numbers of striped field mouse decreased by a factor of 8; numbers of house mouse increased by a factor of 4, while common redbacked vole disappeared from the rodent community by May 1987 (Kozubov, Taskaev, 1990; 1994).

[17] **Example.** Local area with high natural radioactivity (Komi region of Russia, radium site, 3 hectares). Concentrations of radionuclides in soil ash: 70.7 Bq.g^{-1} (Ra); 25.7 Bq.g^{-1} (Po-210); 50 Bq.kg^{-1} (U). External exposure at the site: 8000 microR/h. Local population of tundra vole*Microtus oeconomus P*.Numbers of vole females involved in reproduction were 2-5 times lower when compared with the control. Period of reproduction of voles reduces to 4 months (control - 6 months). Average numbers of embryos per pregnant female were lower than normal by a factor of 2 (3.8 in the radium site, 6.3 in the control, N=2900). The existence of local population was supported by migrations of animals from neighboring areas, disturbed by regular hay cuttings. The average share of migrants found in the radioactive site was about 30%. After 1 month of living on contaminated site newcomers became as contaminated as residential animals (Maslova and Verhovskaya, 1976).

[18] **Example**. Area contaminated in 1950s as a result of activity of PA “Mayak” (Southern Urals, Russia); highly contaminated sites Metlino and Lezhnevka. Field studies of 1992-1993 (summer). Decrease in species diversity of flies was observed

2.6. The Emergence of the Ecosystem Approach Concept

Many environmental professionals recommend dealing with environment protection by designing and following concepts using the ecosystem approach (Crober, 1999; Apitz et al., 2006). This is indeed concerning the field of radiation protection as well, as illustrated by an Expert group of OECD/NEA (2007) who explored the challenges to the current paradigm for the radiological protection of the environment by stating "*an approach based on this [ecosystem health] perspective emphasises consideration of the environment in an integrated manner that better reflects the actual complexity of nature.Thus, such a top-down approach is to be favoured…*"

Thus, it must be observed that for one or two decades, there has been an on-going shift towards applying an ecosystem approach to environmental management, as demonstrated by the vast literature on this subject area (Crober, 1999; Hartje et al., 2003, and see Chapter 4) that is still increasing daily in relevant international scientific journals. This is an important point to consider as risk managers are the primary customers of methodologies for Ecological Risk Assessment.

This recommendation from environmental managers to apply an ecosystem approach can be traced within many governmental institutions and agencies throughout the world, and especially in Europe (Apitz, 2006). For example, in the area of conservation of resources and anthropogenic impact assessment at an international scale, the Food and Agriculture Organisation is undertaking specific developments "to put into practice an ecosystem approach to fisheries" (FAO, 2005). There is therefore a global movement towards an ecosystem approach to the management of marine resources. Another example at a national scale arises from English Nature, a UK Governmental Agency turned into Natural England in 2006, which recommended promoting "*the ecosystem approach*" for the management of "*coherent actions for marine and coastal environments*" (Laffoley et al., 2004). Commitments are made by the UK Government to explore ways of turning the ecosystem approach into a practical reality, as well as supporting work to implement the ecosystem approach through the OSPAR convention.

2.6.1. Ecosystem Approach Recommended for ERA by Environmental Managers and Users

There is a growing awareness nowadays among policymakers and scientists that assessment studies should adopt a systems approach. This trend started almost 2 decades ago with the US Environmental Protection Agency (EPA, 1992; Barnthouse et al., 1987) who proposed and identified endpoints for the evaluation of contaminants effects at the population, the community and the ecosystem levels, and also the Health Council of the Netherlands (HCN, 1997) who recommended using food webs as a starting point when formulating assessment programs.

The international standards of Ecological Risk Assessment[19] are now consensually formulated along 4 successive phases, namely 1) problem formulation, 2) analysis of exposure and effects, 3) risk characterisation and 4) risk management. The area of risk management, the fourth and last step of ERA, is generally clearly separated from the risk characterization phase because it goes beyond the strict application of standard scientific methods of evaluation, as it integrates also social and political considerations, so as to lead to decision making in the context of a particular scenario. Environmental managers are therefore the end users of Ecological Risk Assessment, i.e. those who are also in charge of decision-making when faced with each particular scenario. It is therefore paramount that the scientific methodologies embedded within ERA fulfil the needs expressed by environmental managers.

2.6.2. Ecosystem Approach for Assessment of Radiation Effects

To conclude, an ecosystem approach for assessment radiation effects is a holistic strategy for risk assessment and characterisation that integrates radio-toxicological knowledge with ecological knowledge. The ecosystem concept is recommended for several reasons:

- it integrates the general objectives of environment protection, which overall refer to overall ecosystem structure (biodiversity) and functions (ecosystem life support and services);
- it allows to integrate radiation risks in the whole picture of ecological risks as arising from other contaminants and stressors;
- it attempts to balance the use of both, bottom-up, reductionist, mechanistic based toxicological methods and top-down, systems based, ecological methods (population dynamics, non-linear responses, interactions between populations);
- it aims at filling the gap between current methods exclusively relying on individual organism level radiotoxicology and such general objectives of environment protection; and
- it meets the end users strategy currently developed for management purposes.

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within the highly contaminated area. From 31 typical species, 29 were found in the control and only 9 species in each of contaminated sites. Fly species which normally hatch in moist and near-water vegetation were absent (Krivosheina, 1999).

[19] More details are provided in chapter 4.

3. The Concept of Reference Organisms for Radiation Protection

The leading organisation in radiation protection, the International Commission on Radiological Protection (ICRP) developed recommendations for the protection of human beings and has now widened the system of radiological protection to also include plants and animals. Having said this, it has not considered the protection of the abiotic part of the environment, but has deliberately concentrated on its biotic part, because it is the only radio-sensitive one. Optimised exploitation of already-acquired knowledge and experience for the protection of the *Homo sapiens* "animal" is a major advantage in this approach. The ICRP (2008) has outlined an approach based on “Reference Animals and Plants”[20] (RAPs). The RAPs approach extends concepts of human radiological protection to other living organisms with any modification or adjustment considered necessary.

It is worth mentioning that the IAEA, some other European-based projects (FASSET, 2004 and ERICA, 2007) and the US RESRAD BIOTA (DOE, 2000) have worked out quite similar concepts of “Reference organisms” featuring only minor variations from the ICRP-developed “RAPs” concept.

3.1. Reference Fauna and Flora

The first major challenge was to simplify the huge biological diversity found in the environment. Living forms, plant and animal species and their ways of life are indeed so numerous that it appears difficult to understand this multitude uniquely. The approach here was to reduce this complexity to a few standard cases, considered representative[21] based on criteria such as their abundance in the environment, their radio-sensitivity (with the underlying notion of bioindicator), and their ecological significance, as well as the need for a substantial amount of radiological knowledge about them. In this way, "reference" fauna and flora were defined, resulting from the best possible compromise between all such criteria as those mentioned above. This is not primarily choosing a real existing living entity, but rather defining a stylized entity, which serve as reference points for comparison purposes.

Pentreath (1998, 1999) and Pentreath and Woodhead (2001) have in particular developed this approach inspired by the "Reference Man" concept in human radiological protection (white man, weighing 70 kg, living in a temperate, western climate, with an average age of 20 to 30, or a woman of similar definition); the approach called for a restricted range (between ten and twenty) of primary reference animals and plants. They can be used in impact assessments in a first coarse screening combined with conservative (i.e., protective) adjustment factors (Figure 6). Secondary representative organisms, more finely detailed and more representative of local specific features including ecological realism, will be used subsequently, should the preliminary coarse screening not have eliminated a sufficient probability of harm.

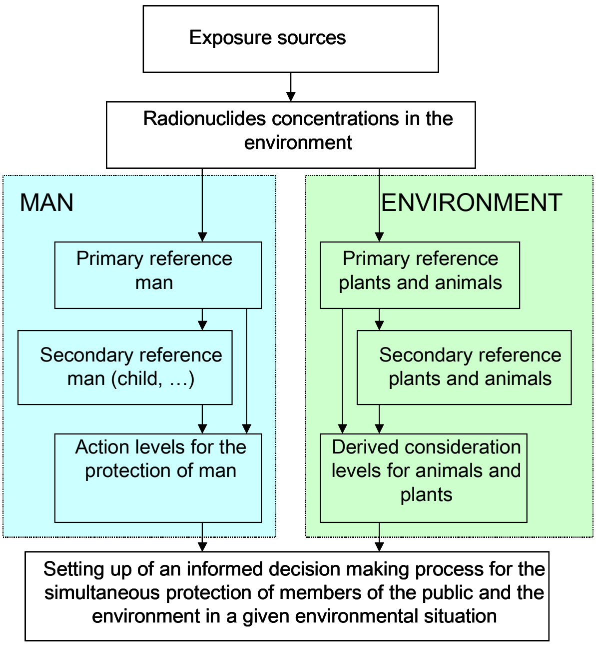

Figure 6. Homogeneous, combined approach to protecting man and the environment. (From ICRP, 2003).

[20] More details about the “reference” terminology often used in this context can be found in ICRP n°108 (2010).
[21] A conceptualisation of “representative organisms” has recently been developed by ICRP and presented during the first ICRP Symposium held in Washington DC, October 2011 (see Pentreath, 2012, in press)

3.2. Reference Units and Dosimetry Based on Simple Geometries

The concept of the absorbed dose is considered to best describe the energy deposited in the biological systems and thus forms the base unit for quantifying dose/effect relationships As for man, a dose weighting method is used to take into account both the differences in biological effectiveness of the various types of radiation (Relative Biological Effectiveness, RBE) and the differences in radio-sensitivity based on the organs being considered. This results in the definition of a single unit to qualify the so-called equivalent dose (Sievert for man), which means that all useful comparisons can be made on the same scale.

Here again the difficulty lies in the wide diversity of the animal and plant organisms, with radiation effects that are not necessarily comparable from one species to another, from one type of radiation to another or from one effect target to another. Various approaches have been suggested; the dose absorbed by the biota (Kocher and Trabalka, 2000), the dose equivalent for fauna and flora ("DEFF", Pentreath, 1999), the ecodosimetric weighting factor (Trivedi and Gentner, 2002; Thompson et al., 2003), which underlines the importance of the remaining gaps and the need for consensual standardisation; future ICRP work will place great emphasis on its development.

The reference organisms are linked to reference dosimetric calculation models based on simple geometries (e.g., sphere, ellipsoid, cylinder) that make it possible to simplify the variety of shapes for the dose calculations. These models provide access to the total dose absorbed by an organism through external and internal irradiation that should then be weighted as described above. By subsequently connecting this calculation to models describing the spread of radionuclides in the environment and the most critical exposure pathways for the reference organism being considered (provided, of course, that adequate data is available), the intention is to end up with an expressed absorbed (weighted) dose per unit of radionuclide concentration in the various environmental compartments. Tables of values are thus built up for the reference organisms which allow the assessors to situate the specific quantitative context of their study in a first global approach.

3.2.1. The Choice of a Few Effect Endpoints Focusing on Individual Organisms

A standardised organism-based dosimetric calculation can be applied to some reference organisms used as surrogates of what might happen to other species. Such an approach, selected with the intention of being protective of animals and plants is still debatable as lacking a demonstration, but it can be developed from existing tools. Without taking a specific position, ICRP considers that a pragmatic path is already being traced for two reasons. Firstly, an increasing amount of legislation has already largely defined what to protect (particularly a certain number of species considered useful or heading for extinction). Secondly, to qualify for a radiological protection system, the effects of radiation against which we wish to protect animal and plant organisms have to be defined and made accessible to measurement. These are called the effect (and/or assessment) endpoints. Through them the quantification of the effect will be related to the measurement of the radiation, or dose, intensity producing the effect in question. This is the definition of the dose-effect relationship on which the entire system is based.

Given the still-limited knowledge on the huge variety of species and the variation in their radio-sensitivity, it is considered premature to seek to distinguish between deterministic and stochastic effects, as for man. The effects of radiation are rather grouped into four major categories considered relevant for the protection of non-human species. These effect endpoints are: early mortality, morbidity (declining health linked to negative consequences on growth or behaviour), reproductive success (including fertility and fecundity) and cytogenetic effects (scorable or transmissible damage to DNA). The cytogenetic effects and effects on reproduction are usually found at the lowest doses.

All these categories produce a wide variety of effects on individual organisms, but the choice of these effect endpoints is justified by their frequent use in evaluating the impact of other toxic agents, be it for nature conservation or environmental protection. Choosing to target effects on individual organisms is also supported by the argument that the vast majority of current knowledge is at this level and that the effects at the upper organisational levels (populations, communities, ecological systems) have to come from the repercussions of effects on the organisms. The conceptual difficulties faced by problems of transposing the individual to the .ecosystem are recognised, but the organism is nevertheless favoured for practical reasons. Note lastly that .human radiological protection is also aimed at the individual organism.

3.2.2. Construction of a Scale of Risk

Currently, ICRP does not intend to recommend limit dose values, but rather to define an approach based on scientific foundations and solid ethics to guide and direct assessments. For the protection of the general public, ICRP is now tending towards considering concern levels defined by explicit reference to the natural background (ICRP, 2001). Similarly, it suggests constructing a risk scale, using for example logarithmic increments of the total annual dose relative to the background (1, 10, 100, 1000 times the natural background) for a reference terrestrial mammal (Table 2). The interest of such a scale is that, for a given dose rate range, it puts the probable effects and possible management options for situations producing such effects into perspective.

Table 2. Derived consideration levels that could be defined for a reference land mammal (modified according to Pentreath, 2002).

Derived consideration level	Relative dose level (incremented annual dose)	Probable effects on individuals	Aspects relating to taking risk into account
Level 5	> 1000 x background	Early mortality	Consideration of a potential remedial action
Level 4	> 100 x background	Reduced reproduction success	Risk dependent on type of fauna and flora likely to be affected and their total numbers
Level 3	> 10 x background	Measurable damage to DNA	Risk dependent on nature and size of affected zone
Level 2	Natural background		Little risk
Level 1	< natural background	Low	Little or no risk

3.3. Merits of the reference organism approach: a discussion

Today, the complexity of nature is putting quite a challenge on the conception of practical tools of ecological risk assessment that would ensure high level trust in subsequent measures taken for the protection of ecosystems. Simplification therefore has been one main driver to the bottom-up current conceptual approach that has been grounded on the reference organism concept. The concept, designed upon the relationship between radiation dose and its potential effects in non-human biota, also makes most optimal use of the currently existing knowledge published to date in the literature, where effects in individual organisms are most often reported on. It is to be stressed that this concept is much oriented towards regulatory purposes, therefore giving maximum priority to an immediate ability to be applied in practice.

It is worth noting in addition that traditional Ecological Risk Assessment for chemicals is also using some references which have been developed as standardised ecotoxicological test organisms for which extensive toxicological data have been explored and documented (microalgae, daphnids, barley root, worms, zebra fish…). This is still a widely used approach, especially for regulatory purposes, although its reductionism tends now more and more to be acknowledged, pushing for the development of additional approaches that would be both better integrated and more realistic.

3.3.1. A Common Approach to Radiological Protection of Man and other Organisms

Being designed with similitude to the principles of human radiological protection, the reference organism approach ensures an overall coherence that allows integration within one single system of the protection of humans and non-human biota. Both are thus rooted in a common focus on individuals, and on the use of references (reference man and reference animals and plants). This is an attractive feature because it fulfils a high level policy goal that aims at protecting at the same time man and non-human biota.

Indeed, for ICRP, it is important that the radiological protection system for non-human living organisms remains compatible with the principles of human radiological protection. A common approach towards protecting man and the other living organisms should therefore engage common methods and scientific bases to evaluate the impacts and justify the decisions. Pentreath (2002) has suggested that such a common approach could include the following objectives:

- protecting human health:
    - by eliminating all deterministic effects; and
    - by limiting the stochastic effects to individuals and by minimising them in populations;
- protecting the environment:
    - by eliminating/reducing the frequency of effects likely to cause early mortality or reduced reproduction success in individual animals and plants;
    - so that there is negligible impact on the conservation of species, maintenance of biodiversity, and the quality of habitats for the species that comprise ecological communities.

This would be consistent with established environmental protection objectives, such as pollution prevention at source and minimising of waste, which may well go beyond the single concept of dose minimisation.

The ICRP also recognises clear ethical, conceptual and practical differences between human beings and the other living organisms in terms of their radiological protection, but it notes that numerous similarities also exist that deserve to be explored. The Commission is therefore taking the exploration of this path as its starting point. A major part of the acquired scientific knowledge on interaction mechanisms between ionising radiation and living matter has come from work on non-human organisms. In addition, the development of a common approach would not only prevent the recommendations for one conflicting with those of the other, but would simplify the approach that could then be pursued in a general framework of evaluation and management.

With the accent placed deliberately on the individual organism, this approach could therefore respond to a protection objective set at this organisation scale (as for example, protection of endangered species). However, protection objectives set at higher scales of biological organisation, as increasingly mentioned in international conventions, are not clearly addressed in this approach. This forms an important shortcoming requiring further attention and appropriate developments.

3.3.2. Robustness of the Scientific Foundation

One basic driving principle of the ICRP is feasibility. A radiological protection system with concepts and methods too far removed from practice would be useless. For all that, if feasibility tends always to push towards simplification, the resultant uncertainties must be fully acknowledged, not to forget them when decisions are taken, and understood and in order to promote continuous efforts to reduce them. In this vein, it can be observed that the available knowledge on the dose-effect relationships for animal and plant organisms has been acquired from frequently dated work that focused attention on the effects of high doses in acute, external exposure (to γ radiation). Today, the relevant environmental context for radiological protection of the environment concerns above all the domain of low doses to which these organisms could be exposed chronically for several generations. At this level, the performance of the DNA repair mechanisms more than likely takes on a central role. However, an understanding of the repercussions of such effects on the functioning of the ecosystems via population dynamics is far from being acquired (Bréchignac and Barescut, 2003; Woodhead, 2003), especially as the current approach relies explicitly and exclusively on the effect data at individual organism level, for practical reasons. Transposition problems between organisation scales not only relate to the field of radiation effects, but concern all toxic agents as shown by current orientations of ecotoxicology research programmes.

The ICRP concept of reference animals and plants represents a “taxonomic” rather than a “functional” approach to environmental risk assessment. Instead of looking for benchmarks that could be used as screening values for given environmental entities, it looks in the first place for values that can be universal for all environments and all living organisms. The US and Canadian approaches are more differentiated estimating such trigger values for different environmental compartments. In the US, a screening benchmark has been developed for native aquatic animals and two different benchmarks for terrestrial animals and plants. In Canada, an even more specific system has been developed for screening benchmarks. However, both American approaches are still “taxonomic” in nature to some extent.

The reason for the choices made in Europe partly stems from the limited availability of good quality data for dose-effect relationships that can be used for environmental risk assessment. The assumption is that if the most sensitive organisms known are protected, the environment is sufficiently protected as well. However, since information is only available for a small proportion of the species present, it remains unknown if the most sensitive species are actually considered. The question may arise whether the regulators and other end users can be satisfied with screening values developed in this way. It could be foreseen also that such an approach would have some undesirable socio-economical implications.

3.3.3. Radiological-Chemical Integration

The challenges mentioned above concern not only the field of radiological protection, but also protection against all other toxic agents. From the point of view of radiological protection, since radiation interaction mechanisms with living matter are the same for man and other organisms, it seems useful to integrate these protection approaches. However, it is just as important to bring the radiological and chemical environmental protection approaches closer together. In practice it is clear that these two fields are inseparable in actual situations, where radioactive and chemical toxic agents are generally found together. In addition, this concomitance may promote synergies or antagonisms in singular effects, which could lead to the prediction of an erroneous global effect if not taken into account.

For this reason it is important to maintain consistency between the radiological protection of the environment approach and the Ecological Risk Assessment (ERA) approach, which has been developed to protect the environment against other (chemical) toxicants (Bréchignac, 2003). This has indeed been a concern for most developments currently achieved in this field so far. Such a consistency is essential, for at the end any evaluation of ecological impact has to be integrated with all stresses to produce an appreciation of their overall effect.

4. Approaches used in Environmental Protection

There are two main areas of environmental protection, namely the assessment of toxicity of chemicals and the protection of biodiversity, which have led to significant development of approaches. Modelling ecological complexity is also an area of serious current attention prone to giving rise to the emergence of new assessment tools.

4.1. Assessing Toxicity of Chemicals

For more than two decades, there has been very active work to develop strategies and methods for ecological risk assessment of the toxicity of chemicals. It is therefore useful to review their main features in order to provide a comparison with the current approach being designed to deal with the toxicity of radionuclides.

4.1.1. Environmental Risk Assessment for Chemical Exposures

Environmental risk assessment (ERA) for chemicals starts after hazard identification with **problem formulation** in a planning process, and under dialog between risk assessors, risk managers and sometimes other interested parties. An analysis of **exposure** and **effects** in a given environment is followed by **risk characterisation**, and finally by **risk management.** An illustration of the environmental risk assessment framework is given in Figure 7 (Pausterbach, 2002; see also Sergeant, 2002).

The **problem formulation** process is triggered by observations of adverse biological and health effects on organisms/populations and exposures to particular chemical agents. Such observations give background for the confirmation that the chemical is **likely** to cause the observed adverse effects. This is usually confirmed with some scientific data of the biological or molecular mechanisms for the toxic effect of the given chemical or the group of chemical stressors in some laboratory test organisms.

The next step in this methodology is to carry out **exposure assessment**. This is a process of measuring or estimating the intensity, frequency and duration of the organisms' exposure to a chemical currently present in the environment, or of estimating anticipated exposures under different conditions that might be caused by releasing new chemicals into the environment. Exposure assessment is based on field and laboratory measurements and characterisations of the populations and surrounding environment. The last should preferably describe the magnitude, duration, schedule and route of exposure; and the size, nature and classes of the animals, plants, and aquatic or wildlife populations that are affected.

In parallel, the **assessment of the concentration-effect** (or dose-effect) relationship should be performed; i.e., the assessment of whether there is any relationship between concentration (or dose) of the chemical(s) of interest and the observed effect. The concentration-effect (dose-effect) relationships are often supported with scientific data by extrapolation (for example, from high to low concentrations, from one species to another). Exposure and effect assessments should also elaborate on uncertainties in all the estimates which are described.

Risk characterisation is finally performed by combining the concentration-effect (dose-effect) and exposure assessments. It involves estimating the likelihood of incidence of the adverse effects on the organisms or environmental compartments under various conditions of exposure to a specific chemical or a group of related chemical stressors. The global effect of uncertainties for concentration-effects (dose-effects) and exposure assessments should be taken into consideration in this step (Paustenbach, 2002; EC TGD, 2003).

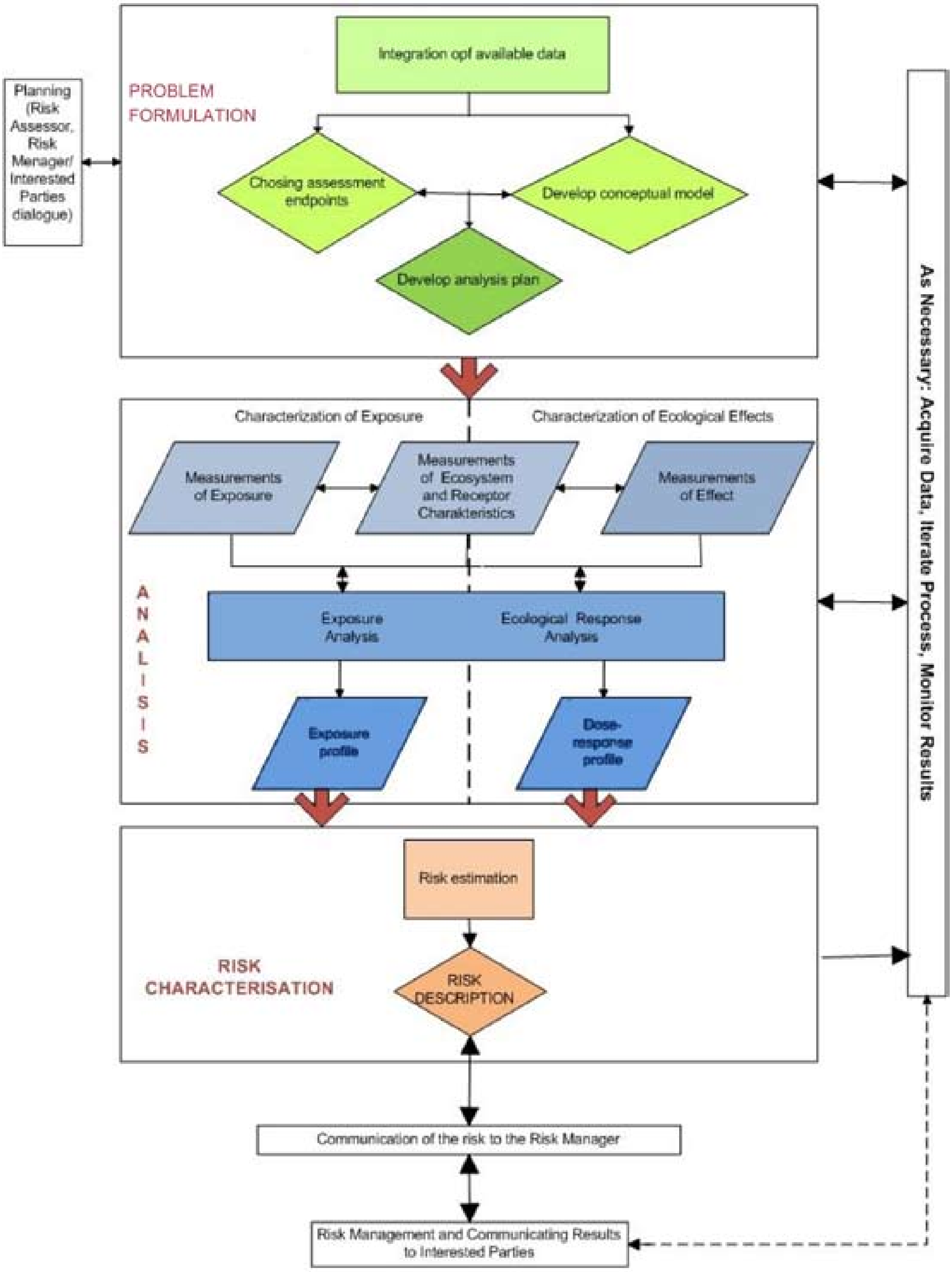

Figure 7. Ecological risk assessment framework, with expanded view of each phase. (Modified from Human and Ecological Risk Assessment (Pausterbach, 2002). Rectangles designate processes, parallelograms designate data inputs, and rhombuses designate decisions.

4.1.2. Quantitative Methods for Environmental Risk Assessment of Chemicals

4.1.2.1. Effect assessment

In order to understand the relationship between exposure and effects in the environment, it is necessary to derive a set of data for environmental recipients generated on the basis of a standardised approach for the set of test organisms. For chemicals there is an established set of test organisms with defined ecotoxicological tests (EU REACH, for example). Although most of these tests focus on sensitivity of individual species, some are tests characterising the function of the set of organisms abundant in some habitats (European Commission: Testing Methods and Directives 18.05.2004). These laboratory tests expose test organisms to a gradient of chemical concentrations in order to identify the concentrations at which toxic effects are observed. Endpoints measured may be mortality or various sub-lethal effects. Concentration-effect relationships are then fitted to the data using appropriate mathematical models (often logistic models).

All of the existing numerical methods tend to derive threshold values, (e.g., no-effect concentrations (NOEC) or lowest observable effect concentration (LOEC)), of the chemical stressors from data arising from ecotoxicity tests. Other typical effect values are EC_{50} (the concentration of the chemical causing 50% change in the observed effect typically for acute exposure conditions) or the EC_{10} (the concentration of the chemical stressor causing 10% change in the observed effect). The EC_{10} is used for chronic exposure conditions and is probably the most relevant for the majority of environmental exposures [22].

For laboratory experiments, it is very important to choose an appropriate ecological endpoint for the given environmental assessment. Common endpoints are parameters of survival, growth and reproduction. There is a strong assumption made at this point as to trusting that the laboratory data on response to the given chemical on the individual organism level can be used to estimate population responses to this chemical. Furthermore, this assumption implies that if individual species are protected, then so are the populations, communities and the ecosystem, even though many of the species, which will be potentially exposed have not been tested (Versteeg et al., 1999).

There are different ways of deriving numerical threshold values that define the acceptable level of stressors needed to carry out risk assessments. They can be based on assessment factors or statistical distribution methodologies describing the variation among a set of species in sensitivity to certain compounds like Species Sensitivity Distribution (SSD) (Posthuma, et al., 2002).

A key issue within all these approaches is that the threshold values developed should be representative for a wide spectrum of environments, albeit without an explicit claim (requirement) to be representative for all environments, and should be auditable and transparent. It is obviously highly preferable if the approach used allows the users to revisit and adjust the values when new data on exposure or dose-relationships become available.

Step-wise approaches are commonly used in risk characterisation methods, consisting of gathering data, selecting a subset of suitable exposure and effect data, estimating criteria for using ecologically relevant effects and determining final threshold values. All or any of these steps may be revisited if new information becomes available. All of these steps introduce uncertainties into the assessment, and therefore there are prerequisites and considerations regarding each of them.

Maximally exposed individuals: This approach involves basing calculations mainly on activity concentrations in the soil / water etc., corresponding to the organisms considered maximally exposed to the toxicant. One recognized difficulty of this approach stems from the associated problems of using average field concentration to calculate average doses to average individuals which may lead finally to an overall situation where a large percentage of the population could be exposed to a dose rate only intended for a small percentage of that population. This type of approach has been developed by the US DoE where concentration limits of radionuclides in soils, sediments and water have been established as “Biota concentration guides” (BCGs) to support assessments (DOE, 2000).

Species sensitivity distribution (SSD) methods have been widely used in ERA for chemicals and contaminated sites for about two decades (Posthuma, 2002). These methods use the EC10 values for several different species and ecotoxicological tests to derive the HC5 value (concentration at which 95 % of species will be unaffected) for the whole data set (see Figure 8). The methods acknowledge that species sensitivity to toxic compounds varies (without attempting to explain the cause of this variation) and describe the variation with a statistical distribution function. However, it is to be noted that this is a statistical extrapolation. The lower limit of the HC5 is always lower than the lowest reported effect in the distribution. This is important to acknowledge as the HC5 represents the best estimate of a concentration at which all species are expected to be protected (against the effects endpoints for which data supporting the statistical treatment have been obtained).

[22] In some jurisdictions the EC20 is considered better than the EC10 due to the greater calculated error bands at the lower effects concentrations.

The Species Sensitivity Distribution (SSD) method (first developed by Kooijman, 1987) is a statistical approach attempting to resolve the incongruity between individual-based data most generally available and the complex biological entities addressed in ecological risk assessment. The method predicts from single-species test data at individual organism level (for a given end-point) concentrations affecting only a certain percentage of species in a community (for this given end-point). Single species data (e.g. LC50, for example) for many species are fitted to a distribution such as the log-normal (or log-logistic). From this distribution of species sensitivities, a hazardous concentration HCp (here HC5) is identified at which a certain percentage p (here 5 %) of all species is assumed to be affected. The most conservative form of this approach uses the lower 95 % tolerance limit of the estimated percentage to ensure that the specified level of protection is achieved.

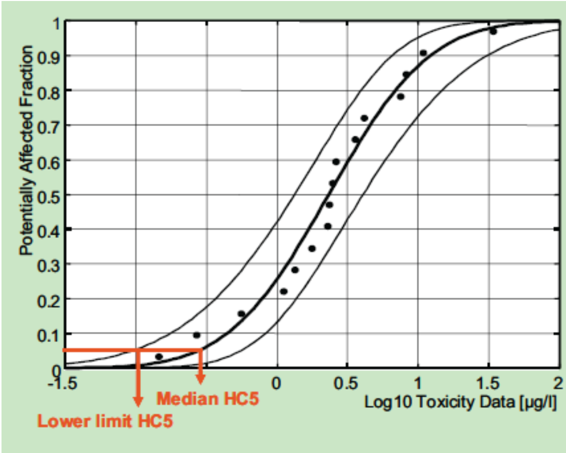

Figure 8: Species sensitivity distribution (SSD) method

Statistical extrapolation from a small set of data, obtained from a limited array of specific effects arising at individual organism level, to a real situation in the environment using SSD and associated extrapolation techniques implies that many assumptions have to be made and may include pitfalls. These limitations are well known for the researchers that developed SSD (Aldenberg and Jaworska, 2000; Newman et al., 2000). They call for scrutiny when using it for new applications, and criticism was raised on a whole spectrum of issues concerning SSD, such as the statistics, ecotoxicological issues regarding the input of data and ecological interpretation of SSD output. Such limitations call for making use of SSD methods with due caution, especially avoiding over-interpretation of the output HC5 values which they drive to, in terms of benchmarks or limits values to support regulation frameworks.

The SSD methodology can only give rise to threshold values corresponding to the few species for which data have been introduced, and are therefore likely to miss the most sensitive species. Furthermore, it considers only a few endpoints for which data have been collected, most often at individual organism level, and the tendency to mix them all for statistical treatment currently lacks any theoretical validation. One challenge indeed for risk assessments is to define other endpoints and to develop quantitative approaches as strong as SSD (van Straalen, 2000).

There are also considerations and discussions about the minimal number of species needed to extrapolate to all species in the given environment and the representativeness of the species (Aldenberg and Jaworska, 2000). Different parameters can be difficult to compare if the differences in the life history strategies are considerable. Additionally, because the SSD approach is fundamentally individual organism-based, it cannot be used to explore interspecies relationships that are central to the ecosystem concept approach.

Recently, the use of field-derived SSDs has emerged as a valuable tool for overcoming some of the weaknesses of laboratory derived SSD (US EPA, 2011; Kefford et al., 2004, 2007; Struijs et al., 2011). The pros and contras of each approach have been discussed in some more details by Cormier and Suter (2012).

Nevertheless, an important advantage of the use of SSD for deriving benchmarks needed for ecological risk management is that it is one of the most transparent and formal methods for environmental risk assessment currently available. Notwithstanding its drawbacks, the SSD technique, being better for transparency and formalism reasons, currently tends to be favoured in view of deriving screening benchmark values (i.e., values which, if exceeded, do not necessarily lead to significant consequences but should trigger further investigation into the risks).

4.1.2.2. Risk Characterisation

When all the relevant effect assessment data have been collected and analysed, a judgement must be made on the risk to the organisms in the particular environment of interest (bearing in mind however that it is actually the risk to populations and ecosystems which is ultimately needed). This is usually done by calculating Predicted No Effect Concentrations (PNECs). These can be derived, for example, from HC5 values (from the SSD method) or from LOEC values, and are simply the HC5 or LOEC divided by an Assessment Factor (AF, also referred to as Safety Factors, SF). The value for this assessment factor reflects on the degree of uncertainty of the data, and is a way of building in a measure of precaution into the assessment. Uncertainty can be due to the type, quality or quantity of the data, how 'representative' the organisms (e.g., taxonomic spread, spread among trophic groups) and endpoints (lethal, sublethal, chronic, acute etc.) are, and the statistical quality of (e.g.) the SSD curves.

The PNECs are then usually assessed by comparing them with measured or predicted environmental concentrations to produce a risk factor. (Risk Factor = environmenta / concentrations / PNEC). Values greater than 1 are thus assumed to pose a risk requiring further examination[23].

4.1.3. Comparing Risk Assessments for Chemicals and Radiation

Both assessments tend to build the risk assessment around benchmarks values, i.e. stressor levels which are expressed in terms of either concentration or dose/dose rate that can be tolerated, and follow the generic framework for risk assessment that consists of Problem Formulation, Exposure Assessment, Effect Assessment, Risk Characterisation and Risk Management.

For both approaches, the sensitivity/response of the "representative" populations or communities of organisms are based upon some reference organisms which have been identified more on the grounds of the availability of data than because of their actual ecological relevance. The reference organism approach does not incorporate any direct aspect of ecological structure or function, and most chemical assessments also are individual organism-based (i.e., not oriented toward an ecosystem concept approach). Though in both approaches there are often assumptions that information cans be interpreted to be protective of higher levels of organization (i.e., populations, communities), because neither explicitly considers the dynamic interactions among species, such assumptions should be viewed with skepticism.

The recent European project PROTECT defined a preliminary set of commonalities and differences in the environmental risk assessment process for chemicals and radiation (PROTECT Deliverable 3, Copplestone et al, 2009, see Table 3, below).

Table 3. Similarities and differences between chemical and radiological risk assessments (after PROTECT, 2007)

Risk assessment stage	Similarities and differences
Problem formulation	Scoping and protection goals common to both approaches. *A priori* definition of ecosystems and reference organisms in radionuclide risk assessment
Exposure assessment	Environmental transfer of contaminants is a common feature but attention to interactions between ambient environment and biological receptors different (chemical approaches consider factors that affect availability, e.g., pH)
Dosimetry	Major differences: this is a significant feature of radionuclide risk assessment but not chemical risk assessments where the focus is just on ambient concentrations. Possible internal and external exposure from radionuclides but only internal residues are relevant to chemicals
Effects assessment	Significant differences: assessment of chemicals is based on assessment of empirical ecotoxicological data relating concentrations to effects, whilst assessment of radionuclides uses data that relate dose to effects. Separate assessments are needed for each new chemical but radionuclide assessments need only consider a limited range of radiation types and qualities
Risk characterisation	Similar approaches for characterising risk are now being used for both chemicals and radioactive substances. For example, approaches for radiological protection of the environment have applied the SSD and assessment factor approaches to derive values to compare with predicted dose rates to determine the magnitude of any risks (Garnier-Laplace and Gilbin, 2006)

Some additional remarks can be made at this stage, however, and these are further developed in the following.

In the **Problem Formulation stage**, it must be stressed that historically, and even today, the scope and goals of radiation protection have been more about protecting humans. Chemical assessment goals have addressed non human species and ecosystems more specifically, even though with a relatively crude format so far. There is another difference regarding the answer to the question of what it is sufficient to protect in the environment and what we actually would like to protect. Risk assessment of chemicals takes into consideration both different classes of chemicals and different environmental compartments, although it is also true that there are extrapolations from one chemical to another and from a few standard test organisms to all organisms. The definition of ecosystems and reference organisms and plants (ICRP 108) or reference organisms (ERICA; PROTECT) in radiological risk assessment seems to be chosen without sufficient scrutiny in answering the

[23] Recent recommendations from Allard et al. (2010) as well as Hope (2009) and others referenced within these papers, however, urge moving beyond the NOEC and LOEC thresholds due to the many documented problems with these values. It is also to be emphasized that ratios (whether referred to as HQ, RQ, ER, or PEC/PNECs) are not risk assessments, but rather screening assessments. Moreover, exceeding 1 does not indicate that an unacceptable risk occurs, but rather that further analyses may be warranted.

question of whether they will cover all aspects of ecological risk assessment. This choice is rather driven by expert-judgement, and pragmatic decision which yield a compromise between the availability of information about the relationship between radiation exposure and sensitivity of non-human organisms, and an attempt to get representation of all biota.

A good deal of available information for chemicals was generated to meet human and environmental risk assessments needs. In many circumstances, this has been achieved by hypothesis-driven research for the purpose of environmental risk assessment, while literally all of the available information about effects that is used in environmental risk assessments for ionizing radiation and ionizing contaminants had been generated for other purposes, such as studying the mechanisms of radiation or for modelling human responses. This is why in the FREDERICA database[24] generated by the EU ERICA project (Copplestone et al., 2008) a score system addressing the quality of the data has been developed to scrutinize the enormous amount of all available data about dose-effect relationships which has been collected[25].

For the **Exposure Assessment step,** the PROTECT project made the point that environmental transfer of contaminants is a common feature, but attention to interactions between ambient environment and biological receptors is different. Chemical approaches consider several factors that affect availability to a much greater extent, e.g., pH, chemical composition, solubility, temperature effect etc. The PROTECT project concentrated only on exposures from external low LET (γ) irradiation for risk assessment due to lack of good quality experimental data for dose-response relationships for internal exposure and for exposure from the mixtures of internal emitters.

With respect to the problem of transfer of radionuclides, **dosimetry** is a significant feature of radionuclide risk assessment, a feature at variance of chemical assessments where the focus is only on ambient concentrations. The models used for environmental risk assessment of radiation are quite generic and simple (comparable with those used in human risk assessment during the 60-ties). These are likely sufficient for many organisms but some groups of organism may require more specific models (for example top predators in some ecosystem compartments). Exposure from radionuclides can arise internally and externally, but only the later is most usually described and assessed.

Internal exposure to radiation is still very poorly addressed whereas being most relevant to chronic situations of exposure (as for chemicals), and it is expected to highly depend on non-homogeneous internal distribution of radionuclides within organs and tissues. Further problems to be faced relate to the question of additivity and weighting the doses from external versus internal exposures, not mentioning the different radiation qualities which trigger different intensities of effects for a same given energy deposition within tissues.

Other critical issues relative to exposure need further consideration. The dosimetric approach fails with microscopic entities, such as micro-organisms and bacteria, because they have too small a mass. This is quite a pitfall as such organisms represent one of the three major functional groups within ecosystems, namely the “decomposers”, and are responsible for the survival or well being of many other species essentially by running the recycling of elements (closing essential biogeochemical loops). One way to overcome this important limitation might be to consider not only the microscopic individual organisms of bacteria, but the macroscopic biofilms they usually form as colonies within the environment (on aquatic surfaces, or around plant roots, for example).

It is also important to mention that in common with assessment of risks from exposure to chemicals, it is possible that large differences in radiation sensitivity among organisms represented in an individual biotope can influence the doses to other organisms. Both hormetic responses and hypersensitivity in the organisms in the food web can potentially influence the dose to other organisms (depending on life strategies, etc.). Also, the same amount of radioactivity can potentially have different impact via the chronic exposure of the same organisms in different types of environments (e.g. rural or urban, forest,…).

For the **Effect Assessment step**, there are quite significant differences. Assessment of chemicals is based on the assessment of empirical ecotoxicological data relating concentrations to effects, whilst assessment of radionuclides uses data that relate effects to dose. The PROTECT project (Protect, 2003, Deliverable 3) has pointed out that separate assessments are needed for each new chemical, whereas for radionuclide assessments there is only a need to consider a limited range of radiation types and qualities. However, radionuclides are also chemicals, and it is not always possible to ignore their chemical and metabolic properties. It is not always possible to ignore also the physical properties of radionuclides (for example the effects of temperature or particle size and deposition patterns in the organs or organisms).

In toxicological risk assessments, the focus is on the ability of particular metabolic and reparation pathways in organisms and populations to overcome the toxic insult, and chemicals also have a larger range of ‘modes of action’ than radiation. In radiobiological risk assessment, the focus is on the ability of organisms and populations to overcome/repair random biological damage which may (or may not) affect different metabolic pathways.

[24] The FREDERICA radiation effects database has been developed to provide an online search of the known effects of ionising radiation on non-human species, taken from papers in the scientific peer reviewed literature.

[25] It has however to be mention that some data generated in mammals as a model for human radiation protection, as well as data from cellular mechanistic studies were omitted in an attempt to achieve the balance between different taxonomic groups.

However, both similarly involve oxygen free radicals. A large proportion of the effects that are caused by low LET radiation is caused by oxygen free radicals involved in the indirect effect of radiation (Coogle, 1983). After exposure to low-LET radiations, about 65% of initial DNA damage is caused by indirect effects (Streffer et al., 2004). Many chemicals exert their action through the reactive oxygen species pathway as well. For chemically toxic substances, there is more knowledge available about dose-response relationships both on the individual and population levels. There are more research results available on effects at higher organisational levels, and there exist standard ecotoxicological tests.

As mentioned before, radionuclides will exert their biological function additionally by chemical toxicity mechanisms, depending on their chemical properties. This may be especially important for α or β emitters. Some radionuclides in some organisms will tend to compete with non-radioactive substances, especially in cases when they can concentrate in different parts of the organism or in particular parts of the biocenosis. Cases can be foreseen in which a radioactive contaminant can compete with a non-radioactive microelement essential for metabolic function due to, for example, its function in active centres of enzymes. Incorporation of such radionuclides into biological molecules will thus add to the complexity of dose and effects assessments, since the radioactive element may be simultaneously essential and radiotoxic[26]. The concept of biological turnover rates or biological half lives becomes important. Incorporation of radionuclides in this way can compromise the metabolism efficacy, and this may in turn influence the organism, population or even biocenosis level. The question is whether there are available models that can sufficiently cover these complex features of some radioactive contaminants.

Knowledge of radionuclides' dose-response relationships at the population and higher environmental organisational levels is fragmentary and limited, and the majority of effect data is generated at the individual organism level, a situation quite similar to that of chemical toxicants. As pointed out earlier, existing data is often generated with the aim of studying mechanisms of radiation and/or especially in vertebrates for modelling human responses. A considerable number of radiobiological studies were not performed for the purpose of setting environmental benchmarks. In many cases, when good quality radiation effect data exists, it is not accompanied by sufficient and good quality data to reconstruct the doses. Many studies were disregarded because of lack of data in the generated databases (FREDERICA, output of EPIC, etc).

By concentrating on two effect endpoints (Andersson et al., 2009), the PROTECT project attempted to derive a generic screening benchmark on the basis of the SSD methodology applied to 20 datasets for different reproduction endpoints in 20 species for which data of sufficient quality were found: the benchmark was estimated at 10 $\mu Gy.h^{-1}$. Currently, this value does not seem to be controversial. However, if one looks at the data set, several questions arise. The first one is whether the numerical values for reproduction endpoints that are compared (for 20 species) have the same impact on population maintenance. Different species have different reproduction strategies and, for some, even a substantial reduction in reproductive output may not be of ultimate importance for the population (e.g. seed production in some plants). Conversely, the same reduction of maturation of sperm in mammals might have a higher impact for reproduction success. The other observation is that although 20 different species are compared (and there is a SSD methodology requirement to analyse a large enough spectrum of species), it is still not certain that the requirement of representativeness of the species can be met. Some of the 20 species are closely related, and there is not a single data set for insects – one of the richest and arguably most functionally important groups of the animal kingdom. The 20 species are also heavily dominated by European and US species, and several are laboratory species that have special characteristics, which their wild equivalents may not.

The hypothesis of additivity of effects from different radiation types also needs attention in the risk assessment process. Most of the data testing the hypothesis of additivity are obtained from studies on external radiation effects. Biological effects have been studied after subsequent exposure to two or more types of external irradiation to cell line systems or animals, or studies that were performed for improving radiotherapy with mixed fields and especially BNCT or other neutron therapy modalities. Most of the results from such experiments indicate additivity for different types of irradiation. There is, however, a lack of data regarding the effects of biological systems that are exposed to different radiation mixtures at the same time. There is also almost no information about the dose-effect relationships and the contribution of different radiation types to biological effects exerted by the mixture of radioactive substances that have contributed to internal exposure.

Ideally it would be desirable to adapt the same standardized ecological test for effect studies from chemical and radiation exposures. There is however a caveat that effects at small doses appear rather late, and standardized short time chemical tests as well as radiological tests will not reveal such effects. This was recently shown in earthworm studies (Hertel-Aas et al., 2007).

In the risk characterisation step similar approaches are now being used for both chemicals and radioactive substances. For example, approaches for radiological protection of the environment have applied the SSD and

[26] The concentration effect relationship of the stable element is thus likely to be bell-shaped (with an optimum concentration), while the radioactive isotope will have a different relationship. Thus the 'normal' dose-effect models (linear or log-normal etc) will not work.

assessment factor approaches to derive predicted dose rates benchmarks against which assessing the magnitude of any risk (Garnier-Laplace and Gilbin, 2006).

However both assessments suffer from the insufficiency of taking into account the interactions with other organisms and the abiotic environment, or with other contaminants. Risk characterisation for radioactive substances is less developed as compared to risk assessment of chemicals as was shown above from an analysis of different stages in risk assessment methodology. This is especially visible in the problem formulation stage.

One of the ideas that appear in this context could be the need to develop screening values for environmental compartments' or biocenoses. For this purpose, it may be necessary to design studies on model environments (both field and laboratory experiments could be applied) or mesocosms using for example SSD distribution methodology for those functional entities. There were attempts made in the PROTECT project to develop the screening benchmarks for different taxonomic groups, but these failed due to lack of suitable good quality effect data from which the screening values for different taxonomic group could be derived.

One commonality for both chemicals and radiation effects assessment is that of extrapolation. In both cases, data must be used to extrapolate between species, between ecosystems, between chemicals (or radionuclides), from chronic to acute effects or from laboratory to field situations. These extrapolations are based on numerous assumptions (see chapter 5), and rarely include much ecology (see Forbes and Calow, 2002; Calow and Forbes, 2003).

A recent attempt of developing an integrated approach for comparative risk assessment between ionizing radiation and other stressors is briefly presented in the Annex.

4.2. Protection of Biodiversity

Protection of biodiversity has emerged as one of the most advanced international concerted efforts, together with climate change and global warming due to emissions of substances with greenhouse effects. Biodiversity is reported from a large array of scientific evidence as playing an important role in the maintenance of ecosystem structure and functioning. International efforts aimed at protecting biodiversity have particularly advanced the concept of an "ecosystem approach", with its underlying principles, objectives and practical tools to support it. It is worthwhile therefore to investigate its compatibility with the reference organism (also the ICRP RAPs) approach.

4.2.1. The Role of Biodiversity[27] in Ecosystem Dynamics: a Brief Review

One of the most interesting issues in conservation biology has been whether or not a species extinction propagates to other species and causes secondary extinction. Many theoretical studies have been published (summarized in Table 4 as three different theories linking biodiversity to ecosystem functioning), and such studies help to understand the effect of irradiation in ecosystems. A brief review of the relationship between biodiversity and secondary extinction is therefore given below.

MacArthur (1955) discussed the relationship between the structure of foodwebs and the chain of extinction, and concluded that a complex network rarely shows a chain of extinction because such a foodweb includes species that have the same ecological function as the extinct species and thus replace its function in the foodweb. Dunne et al. (2002) found that a foodweb consisting of large number of interactions between species is robust to secondary extinction compared with a foodweb with a small number of interactions. These studies more or less agreed with empirical observations, however, they did not considered the population dynamics because of the complexity due to huge number of species. Some studies including population dynamics with small number of species showed that removing the top predator in a complex food web is more likely to cause secondary extinction than removing producers or consumers at lower trophic levels (Pimm 1980, Ekloef and Ebenman 2006). Sole and Montoya (2001) studied the probability of secondary extinction by removing species from the three existing foodwebs. They compared two types of removal method, random removal and selective removal of highly connected species. They found that the secondary extinction risk was very low when species were removed randomly. On the other hand, the risk was very high when species were removed in order of their number of interactions. They also found that removing a species with a small number of interactions can cause extensive secondary extinctionsre depending on the location of the species in the foodweb. Ebenman and Jonsson (2005) also showed that extinction of prey species, which have few interactions, but support many species directly and indirectly, can cause extensive extinction. Quite a similar situation can occur when species are substantially decreased in number, therefore promoting extreme knock-on effects (decreases or extinctions).

The above has focused on the relationship between biodiversity and extinction, but it is also important to consider the relationship between biodiversity and ecosystem functioning (i.e., productivity of plants, material cycle and biomass), (Loreau et al. 2001, Cardinale et al.,2011). A clear-cut link between these parameters has been hard to show and has been the source of much debate. Tilman (1997) showed with mathematical models that biomass increases with biodiversity. This result is supported by long term experiments of biodiversity and ecological functioning in grass land in USA (Tilman et al., 2001). European long term experiments showed the same results as in the USA (Hector et al., 1999), however, Huston et al., (2001) reanalyzed Hector's data and

[27] We refer here to "species" biodiversity.

found that biodiversity had no significant correlation with ecosystem functioning. It is, therefore, inadequate to use only the biodiversity as the index of ecosystem functioning. Hooper and Vitousek (1997) suggested that diversity of functional groups in ecosystems might be used as an index. More recently, efforts have been focussed on exploring the importance of trait diversity (e.g, Cadotte et al., 2009, Litchman et al., 2010, de Bello et al., 2011). It is worth noting that most of such studies focused only on trophic levels and ignored the spatial effects, though this is an active area of current research. Miki et al. (2008) and Cardinale et al. (2011) showed the relationship between ecological functioning and biodiversity is influenced by spatial scale.

This short review can be summarized as follows:

- More complex networks are more resistant to secondary extinctions.
- It is important to consider not only biodiversity but also the number of pathways and connectivity (number of interactions) when secondary extinction risks are assessed.
- The removal of a top predator in an ecosystem is likely to cause secondary extinctions.
- The removal of the hub species (which interact with many species) causes extensive secondary extinction. In some cases, however, removal of non-hub species can also cause extensive secondary extinctions.
- Some studies showed a positive correlation between ecosystem functioning and biodiversity (however, others have not shown this).
- The role of functional or trait diversity is as important as species diversity.

Table 4. Different theories about the role of biodiversity with respect to ecosystem functioning, and implications for ecological risk assessment. (Adapted from Forbes and Calow, 2002).

Ecological Theory	Reference	Implications for ERA
Each time one species is removed, the structure of the ecosystem is weakened gradually resulting in functional failure	The “rivet popper hypothesis” (Ehrlich and Ehrlich 1981)	Changes in ecosystems structure and processes are closely connected each other. Either one provides relevant endpoints for risk assessment
Several species in an ecosystem perform the same process	The “redundant species hypothesis” (Walker 1992)	As certain species are removed, others take their function. Changes in structure are more sensitive than changes in process
Certain species play a much larger functional role than others	The “ecosystem engineers hypothesis” (Jones et al., 1994)	Many non-keystone species could be lost without any observed changes in function. If a single keystone species were to be removed, dramatic changes could occur in the structure and functioning.

4.2.2. The Ecosystem Approach of the Convention on Biological Diversity

The Convention on Biological Diversity (CBD) aims at the conservation and sustainable use of biological diversity[28], and the equitable sharing of benefits arising from the utilization of genetic resources. In doing so, the CBD takes a holistic, ecosystem approach to biodiversity conservation and sustainable use in contrast to other nature conservation approaches. In implementing the CBD, the ecosystem approach is used as one of the basic concepts guiding efforts to manage biological resources, along with the precautionary principle.

The ecosystem approach is the primary framework for management action under the CBD, and its application is envisaged as helping to reach a balance of all the three objectives of the convention. In the CBD context, the ecosystem approach is a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way. The ecosystem approach was adopted by the Parties to the Convention in 2000[29], and is considered essential in guiding action under the various programmes of work of the Convention, and in providing linkages between those programmes of work. It is based on the application of appropriate scientific methodologies focused on levels of biological organization, which encompass the essential structure, processes, functions and interactions among organisms and their environment[30].

[28] "Biological diversity" is defined in Article 2 of the CBD as "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems".

[29] CBD COP Decision V/6 (2000): Ecosystem approach (http://www.cbd.int/decision/cop/?id=7148, accessed 2011-01-03).

The CBD's ecosystem approach requires adaptive management to deal with the complex and dynamic nature of ecosystems and the absence of complete knowledge or understanding of their functioning. Management must be adaptive in order to be able to respond to inherent uncertainties and contain elements of "learning-by-doing" or research feedback. Scientific research aimed at providing understanding of the functioning of the broader ecosystem in terms of its component parts and their connectivity, and oriented towards the information needs of management, will ensure that management decisions are based on best available science in the context of the precautionary approach. Measures may need to be taken even when some cause-and-effect relationships are not yet fully established scientifically. The parties have also adopted a framework under the CBD for advising stakeholders on how they can ensure that their use of the components of biodiversity will not lead to the long-term decline of biological diversity[31]. In doing so, it is a fundamental assumption that the application of the practical principles and operational guidelines for sustainable use of biodiversity is set within the context of the CBD's ecosystem approach.

The Annex to the CBD's Decision on the Ecosystem Approach provides Principles[32] and points of operational guidance, reflecting the then level of common understanding of the ecosystem approach and its implementation. As the 12 Principles are complementary and interlinked, it is essential they be considered together for the purposes of considering the CBD's ecosystem approach (Table 5).

Table 5: The ecosystem approach developed by the CBD: principles and their rationale (CBD, 2000)

Principle		Rationale
1	The objectives of management of land, water and living resources are a matter of societal choice.	Different sectors of society view ecosystems in terms of their own economic, cultural and societal needs. Indigenous peoples and other local communities living on the land are important stakeholders and their rights and interests should be recognized. Both cultural and biological diversity are central components of the ecosystem approach, and management should take this into account. Societal choices should be expressed as clearly as possible. Ecosystems should be managed for their intrinsic values and for the tangible or intangible benefits for humans, in a fair and equitable way.
2	Management should be decentralized to the lowest appropriate level.	Decentralized systems may lead to greater efficiency, effectiveness and equity. Management should involve all stakeholders and balance local interests with the wider public interest. The closer management is to the ecosystem, the greater the responsibility, ownership, accountability, participation, and use of local knowledge.
3	Ecosystem managers should consider the effects (actual or potential) of their activities on adjacent and other ecosystems.	Management interventions in ecosystems often have unknown or unpredictable effects on other ecosystems; therefore, possible impacts need careful consideration and analysis. This may require new arrangements or ways of organization for institutions involved in decision-making to make, if necessary, appropriate compromises.
4	Recognizing potential gains from management, there is usually a need to understand and manage the ecosystem in an economic context. Any such ecosystem-management programme should: (a) Reduce those market distortions that adversely affect biological diversity; (b) Align incentives to promote biodiversity conservation and sustainable use; (c) Internalize costs and benefits in the given ecosystem to the extent feasible.	The greatest threat to biological diversity lies in its replacement by alternative systems of land use. This often arises through market distortions, which undervalue natural systems and populations and provide perverse incentives and subsidies to favour the conversion of land to less diverse systems. Often those who benefit from conservation do not pay the costs associated with conservation and, similarly, those who generate environmental costs (e.g., pollution) escape responsibility. Alignment of incentives allows those who control the resource to benefit and ensures that those who generate environmental costs will pay.

[30] This focus on structure, processes, functions and interactions is consistent with the definition of "ecosystem" provided in Article 2 of the CBD, whereby "ecosystem" means "a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit". This definition does not specify any particular spatial unit or scale, in contrast to the Convention definition of "habitat". Thus, the term "ecosystem" does not, necessarily, correspond to the terms "biome" or "ecological zone," but can refer to any functioning unit at any scale. Indeed, the scale of analysis and action should be determined by the problem being addressed. The CBD's approach also recognises that humans, with their cultural diversity, are an integral component of many ecosystems.

[31] The Addis Ababa Principles and Guidelines for the Sustainable Use of Biodiversity, adopted by CBD COP Decision VII/12 (2004): Sustainable Use (Article 10).

[32] The 12 Principles are set out in Annex B of the Decision, together with a rationale for each and the operational guidance points are contained in Annex C. For the full text of these, see the link provided in footnote 2 *supra*.

5	Conservation of ecosystem structure and functioning, in order to maintain ecosystem services, should be a priority target of the ecosystem approach.	Ecosystem functioning and resilience depends on a dynamic relationship within species, among species and between species and their abiotic environment, as well as the physical and chemical interactions within the environment. The conservation and, where appropriate, restoration of these interactions and processes is of greater significance for the long-term maintenance of biological diversity than simply protection of species.
6	Ecosystems must be managed within the limits of their functioning.	In considering the likelihood or ease of attaining the management objectives, attention should be given to the environmental conditions that limit natural productivity, ecosystem structure, functioning and diversity. The limits to ecosystem functioning may be affected to different degrees by temporary, unpredictable or artificially maintained conditions and, accordingly, management should be appropriately cautious.
7	The ecosystem approach should be undertaken at the appropriate spatial and temporal scales.	The approach should be bounded by spatial and temporal scales that are appropriate to the objectives. Boundaries for management will be defined operationally by users, managers, scientists and indigenous and local peoples. Connectivity between areas should be promoted where necessary. The ecosystem approach is based upon the hierarchical nature of biological diversity characterized by the interaction and integration of genes, species and ecosystems.
8	Recognizing the varying temporal scales and lag-effects that characterize ecosystem processes, objectives for ecosystem management should be set for the long term.	Ecosystem processes are characterized by varying temporal scales and lag-effects. This inherently conflicts with the tendency of humans to favour short-term gains and immediate benefits over future ones.
9	Management must recognize that change is inevitable.	Ecosystems change, including species composition and population abundance. Hence, management should adapt to the changes. Apart from their inherent dynamics of change, ecosystems are beset by a complex of uncertainties and potential "surprises" in the human, biological and environmental realms. Traditional disturbance regimes may be important for ecosystem structure and functioning, and may need to be maintained or restored. The ecosystem approach must utilize adaptive management in order to anticipate and cater for such changes and events and should be cautious in making any decision that may foreclose options, but, at the same time, consider mitigating actions to cope with long-term changes such as climate change.
10	The ecosystem approach should seek the appropriate balance between, and integration of, conservation and use of biological diversity.	Biological diversity is critical both for its intrinsic value and because of the key role it plays in providing the ecosystem and other services upon which we all ultimately depend. There has been a tendency in the past to manage components of biological diversity either as protected or non-protected. There is a need for a shift to more flexible situations, where conservation and use are seen in context and the full range of measures is applied in a continuum from strictly protected to human-made ecosystems.
11	The ecosystem approach should consider all forms of relevant information, including scientific and indigenous and local knowledge, innovations and practices.	Information from all sources is critical to arriving at effective ecosystem management strategies. A much better knowledge of ecosystem functions and the impact of human use is desirable. All relevant information from any concerned area should be shared with all stakeholders and actors, taking into account, inter alia, any decision to be taken under Article 8(j) of the Convention on Biological Diversity. Assumptions behind proposed management decisions should be made explicit and checked against available knowledge and views of stakeholders.
12	The ecosystem approach should involve all relevant sectors of society and scientific disciplines.	Most problems of biological-diversity management are complex, with many interactions, side-effects and implications, and therefore should involve the necessary expertise and stakeholders at the local, national, regional and international level, as appropriate.

Concerning the functional relationships and processes within ecosystems, the guidance notes that the many components of biodiversity control the stores and flows of energy, water and nutrients within ecosystems, and provide resistance to major perturbations. A much better knowledge of ecosystem functions and structure, and the roles of the components of biological diversity in ecosystems, is required, especially to understand: (i) ecosystem resilience and the effects of biodiversity loss (species and genetic levels) and habitat fragmentation; (ii) underlying causes of biodiversity loss; and (iii) determinants of local biological diversity in management decisions. Functional biodiversity in ecosystems provides many goods and services of economic and social importance. While there is a need to accelerate efforts to gain new knowledge about functional biodiversity, ecosystem management has to be carried out even in the absence of such knowledge. The ecosystem approach can facilitate practical management by ecosystem managers.

Concerning the use of adaptive management practices, the guidance emphasises that ecosystem processes and functions are complex and variable. The level of uncertainty is increased by the interaction with social constructs. Therefore, ecosystem management must involve a learning process, which helps to adapt methodologies and practices to the ways in which these systems are being managed and monitored. Implementation programmes should be designed to adjust to the unexpected, and ecosystem management should be envisaged as a long-term experiment that builds on its results as it progresses. This "learning-by-doing" will also serve as an important source of information to gain knowledge of how best to monitor the results of management and evaluate whether established goals are being attained.

The Conference of the Parties to the CBD has revisited the ecosystem approach in decisions taken in 2004 and in 2008. In addition, within the CBD framework, more detailed guidance has been developed for its implementation in specific contexts (e.g., agriculture, coastal and marine biodiversity, forestry).

The 2004 Decision on the CBD's ecosystem approach endorsed the earlier decision, emphasised that priority should be on facilitating the implementation of the ecosystem approach as the primary framework for addressing the three objectives of the Convention in a balanced way, and agreed that a potential revision of the principles of the ecosystem approach should take place only at a later stage, when the application of the ecosystem approach has been more fully tested[33].

The 2008 Decision on the CBD's ecosystem approach also endorsed the earlier decision and noted that the pressing need is to translate this framework into methods for further application that are tailored to the needs of specific users[34]. Global assessments suggest that the ecosystem approach is not being applied systematically to reduce the rate of biodiversity loss, but there are many examples of successful application at the regional, national and local scales. This decision identified a particular need for further development of standards and indicators for the application of ecosystem approach, called for efforts to further promote the use of the ecosystem approach in all sectors and enhance intersectoral cooperation.

4.2.3. The CBD's approach compared to that used by other organisations

The ecosystem approach as arising from the CBD has been adopted by a number of intergovernmental organisations and initiatives. For example, the ecosystem approach is referred to in the 2002 Plan of Implementation of the World Summit on Sustainable Development (paras. 30 (d) and 32 (c) in relation to fisheries, para. 44 (e) in relation to biodiversity, and 70 (b) in relation to sustainable tourism). Organizations and initiatives using the ecosystem approach include those working at the sector or biome level (e.g., on agriculture, forestry or fisheries, such as the Food and Agriculture Organization of the United Nations (FAO)) and those working more broadly on natural resources management. The role of the ecosystem approach in these organisations or initiatives varies considerably, largely depending upon their field of interest or nature of operation (see Table 6). This includes, for example, cases where activities are undertaken at the sector or biome level but which acknowledge the need to consider broader issues by referencing the ecosystem approach, and those that use the ecosystem approach as a primary framework for promoting improved integrated natural resources management.

As acknowledged in the CBD decisions V/6 and VII/11 on the ecosystem approach, the CBD's ecosystem approach is not a competing but rather complementary approach to others. Many principles of the ecosystem approach exist in other management approaches and are implemented in projects and other activities without reference to the ecosystem approach. The CBD has acknowledged that these approaches can also contribute to the goals of the Convention on Biological Diversity[35].

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[33] CBD COP Decision VII/11 (2004): Ecosystem approach (http://www.cbd.int/decision/cop/?id=7748, accessed 2011-01-03)

[34] CBD COP Decision IX/7 (2008): Ecosystem approach (http://www.cbd.int/decision/cop/?id=11650, accessed 2011-01-03)

[35] In-depth Review of the Application of the Ecosystem Approach, *Note by the Executive Secretary,* Subsidiary Body on Scientific, Technical and Technological Advice, Twelfth meeting, UNESCO, Paris, 2–6 July 2007, Document UNEP/CBD/SBSTTA/12/2, 30 March 2007, http://www.cbd.int/doc/meetings/sbstta/sbstta-12/official/sbstta-12-02-en.pdf (accessed 2011-01-15)

Table 6. Role of ecosystem approaches relevant to managing biodiversity in different UN agencies (from Mörth, 2004)

UN organization	Areas of activity in biodiversity management	On a policy, planning & programme level	By managing projects	Role of Ecosystem management
UNEP	Host to CBD Secretariat	Central function	Minor role	Key actor
	GEF implementing agency Biodivers ity focal area	Scientific, technical analysis guidance	Extensive GEF project support	Applies EA principles in biodiversity related GEF Operational Programs 1,2,3,4 and 12
	Biodiversity Planning Support Programme	Information, Guidelines, best practices	Prepares for funding proposals	Emphasizes multi-sectoral approach for planning
	Administrator of specific species conventions	Coordinating function	Minor importance	Species and habitat focus
UNDP	GEF implementing agency Scientific, technical	analysis guidance	Extensive GEF project support	Applies EA principles in biodiversity related GEF Operational Programs 1,2,3,4 and 12
	Biodiversity Planning Support Programme (BPSP)	Information, Guidelines, best practices	Prepares for funding proposal	Emphasizes multi-sectoral approach for planning
	Water governance	Capacity development, networking	Considerable unquantified no. of projects mostly in water supply/sanitation,	Emphasizes IWRM and freshwater, coastal ecosystems
	Capacity development in Sustainable Development, sustainable livelihoods	Strategy papers, guidelines	Country programmes – Small volume	Emphasis on sustainable living connection to World Social Summit 1995
FAO	Biodiversity activities within the Priority Areas for Integrated Action (PAIA)	Case studies, guidelines	Limited project activities on biodiversity directly	EA as an PAIA Major emphasis on genetic resources for food and agriculture
	Agriculture related biodiversity actions	Policy Advice, Technical Guidelines, Codes of Conduct	Limited technical cooperation activities	Little impact, ecosystem view, but EA perspective mot central, biodiversity debate dominated by access to genetic resources for agriculture issues
	Forestry related biodiversity actions	Policy Advice, Technical Guidelines, Codes of Conduct	Limited technical cooperation activities	Forestry dominated by sustainability issues linked to poverty reduction
	Fisheries related biodiversity actions	Policy Advice, Guidelines; Codes of conduct; FAO regional fisheries bodies	Limited technical cooperation activities	EA perspective central; change in fisheries management views: cf Reykjavik Declaration 2001
UNESCO	MAB Programme Secretariat	Action Plan 1984 Seville Strategy 1995	Certifies Biosphere Reserves	Biosphere Reserves emphasized as prototypes for EA
	World Water Assessment Programme	World Water Development Reports	Case studies for reports	No direct management role
	International Hydrological Programme	Technical support to water policy and planning	Limited relation to biodiversity	Hydrological cycles, water resources, management, databases and modelling

4.2.4. Compatibility with a Reference Animals and Plants (RAPs) Approach

The CBD's ecosystem approach does not preclude other management and conservation approaches, and may in fact be consistent and compatible with approaches such as sustainable forest management, integrated river-basin management, integrated marine and coastal area management, and responsible fisheries approaches. It is explicitly recognised that other related approaches, such as biosphere reserves, protected areas, and single-species conservation programmes, as well as other approaches carried out under existing national policy and legislative frameworks, can be integrated in the context of the ecosystem approach to deal with complex situations and that implementation of the ecosystem approach in various sectors can be promoted by building upon the approaches and tools developed specifically for such sectors.

The Reference Animals and Plants concept, as developed by the ICRP, has at its core the provision of information concerning relationships between radiation exposure and possible consequences for non-human species, using a set of Reference Animals and Plants. In doing so, the ICRP acknowledges that, in many circumstances, exposure to radiation is but one factor to consider and that the RAPs concept is not itself a system of biodiversity protection.

It is clear that the use of the RAPs concept might contribute information which could be usefully integrated into the ecosystem approach developed by the CBD (see, for example, principles 5 – 8, in Table 5). For the CBD, the ecosystem approach is a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way. A key word here is management. The main issue then is not to compare the RAPs approach with the CBD's ecosystem approach, but to address the challenge incorporating the RAPs concept, and any other methodologies developed for radiation protection, into what is a considerably broader and more holistic framework for ensuring protection of biodiversity.

4.3. Protection of fish stocks: Modelling Ecological Complexity

It is widely recognised that a huge number of species living in aquatic ecosystems are threatened by over-fishing and habitat degradation caused by a variety of human interventions (water pollution, construction of dams and harbours, water withdrawal, introduction of exotic species, etc). In particular, the release of contaminants, including radioactive ones, into the aquatic environment adversely affects water quality. The consequent contaminant dispersal through the abiotic components of the aquatic system and the following migration to living species can represent a severe potential risk to the biota populations.

One aspect of the sustainable management of aquatic environments is the protection of fish stocks that are of interest due to their important ecological role or as food for humans. This is achieved through sustainable exploitation strategies, such as those based on the concept of maximum sustainable yield in fisheries (Sparholt and Cook, 2010), and/or by safeguarding environmental conditions (including pollution status) needed to ensure species survival and development. One way to understand and predict changes in fish stock abundances is to construct ecological models that consider ecological complexity, contaminant transfer and population dynamics.

4.3.1. Modelling ecological complexity

The theoretical assessment of the impact of pollutant on biota populations requires the application of "*Contaminant migration - population effects*" models that account for three groups of reciprocally interdependent processes (Monte, 2009):

- the spatial and temporal dispersal of the populations within the environmental compartment concerned, for example the water body (4.3.1.1 and 4.3.1.2);
- the migration of the contaminants through the abiotic environmental components, the transfer from these to living organisms and, finally, the following dispersion of contaminant in the environment due to the biota movements (4.3.1.1);
- the influence on the population dynamics of the detrimental effects caused by the environmental contamination (4.3.1.2).

4.3.1.1. Contaminant (radionuclide) transfer modelling

As an example, we can consider contaminant transfer to the top predator in a freshwater ecosystem. First, the types and amounts of the various abiotic and biotic ecosystem components must be defined, preferably with some degree of spatial and temporal accuracy. A typical compartmental model is constructed, including the biotic compartments of a simplified lake ecosystem (top predator, two types of small fish, zooplankton, phytoplankton, algae and zoobenthos), the abiotic compartments (active sediments, passive sediments and water), and the processes and mechanisms regulating fluxes among these compartments for a pollutant (Figure 9A). The figure also gives the fluxes to the lake (direct lake load and river input related to catchment load) and from the lake (outflow and sedimentation to the passive sediment layer). This general model can apply to any substance, not just radionuclides.

For comparison, a simplified model is presented (Figure 9B) where the fluxes to the top predator, the variable to be predicted, are be estimated from a few compartments (small fish and lake water) and empirical knowledge of the factors regulating the uptake of a given radionuclide by small fish. The uptake by small fish can be described as a function[36] of, e.g., lake K-concentration (for radiocaesium), theoretical water retention (T) and the dynamic ratio of the lake (DR; DR = √Area/Mean depth; area in km^2, mean depth in m). To use the model, one would need reliable, quantitative data on many rates describing the fluxes (in mass per unit time) among the compartments and the characteristics of each compartment. Moreover, "rates" are not constant, they vary in time and space. The rates are variables, like most of the variables describing the system and its compartments (e.g., weight and age of the animals). This means that it is practically impossible to achieve good predictive power with large and complex ecosystem models like those in Figure 9A, whereas excellent predictive power has been obtained with smaller models of the type shown in Figure 9B (see IAEA, 2000).

[36] For radionuclides, the use of transfer factors is much widespread among the radioecology community in order to derive the internal concentration within a living organism from the external concentration in the medium (easier to measure) where this organism is living. For a large list of different radionuclides, a number of transfer factors have been assembled for various living species and conditions, and are currently being compiled by IAEA to support assessments.

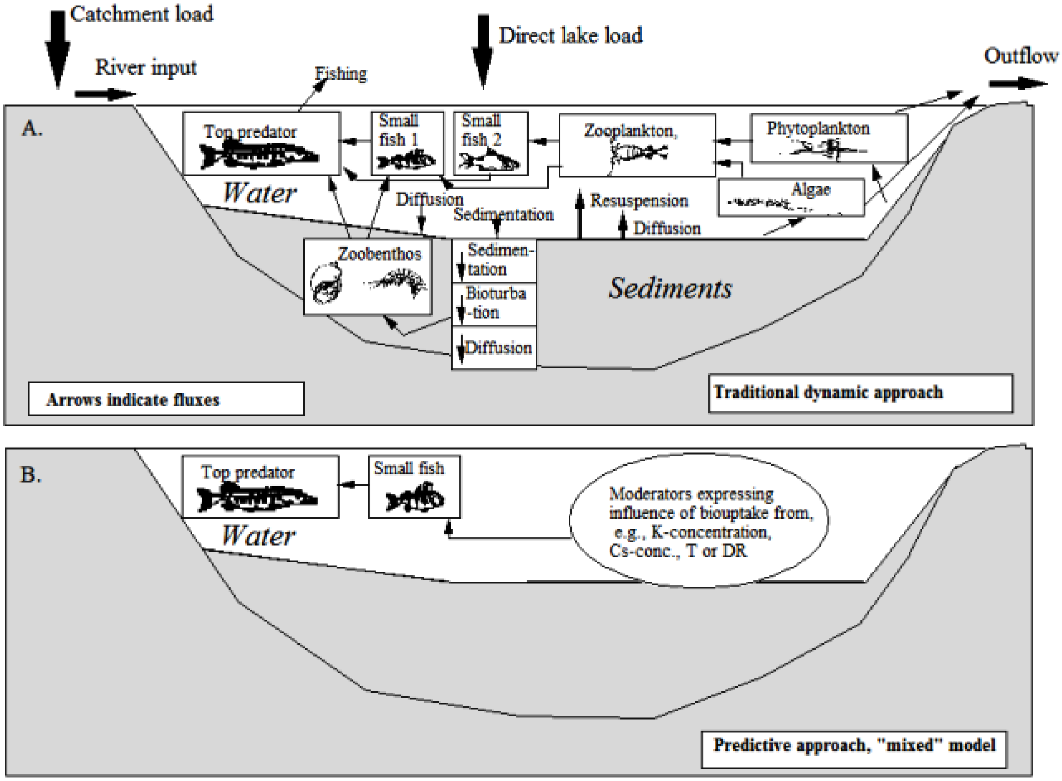

Figure 9. A. Compartmental model illustrating the fluxes (arrows; mass per unit time) in a traditional dynamic model for the type substance mercury in a lake ecosystem with compartments (mass units) for top predator, two types of small fish, zooplankton, phytoplankton, algae, benthos, water and sediments. **B.** Illustration of a "mixed" model, i.e., a model based on mass-balance and an empirical dimensionless moderator expressing the impact of environmental factors on the uptake of mercury from water to small fish. (Modified from IAEA, 2000).

Once the contaminant distribution in the ecosystem is modelled, the potential effects to the various populations of organisms can be assessed. In the case of radionuclides, both internal and external exposure (dose) to the organisms may be important.

4.3.1.2. Population modelling

In order to rationally synthesize the complicated set of processes controlling the dynamics of biota populations, it can be useful to develop generic conceptual models that evaluate the temporal behaviour of the population size and of the age distribution of the individuals by accounting for the ageing process and the balance between births and deaths per unit time.

It is well known that the dynamics of a population of living organisms is described, at least in principle, by the following equations (Magal and Ruan, 2008):

$$\frac{\partial n(a,t)}{\partial a} + \frac{\partial n(a,t)}{\partial t} = -M(a,t) \qquad (1)$$

$$n(0,t) = \int_0^A b(a)n(a,t)da, (t > 0) \qquad (2)$$

where a is the age, n(a,t) is the age density of the population at time t, M(a,t) is the mortality of individuals at age a and timet, A is the maximum age of individuals, b(a) is the fertility of individuals at age a.

The integral

$$\int_0^A b(a)n(a,t)da \qquad (3)$$

is the birth rate at time t. It is important to note that the mortality and the fertility in the above equations should be related to the size of the population of the considered species which is also influenced by intra-specific and/or inter-specific competition.

The influence of contaminants on the population dynamics of exposed organisms requires appropriate sub-models for assessing the mortality and the fertility of individuals exposed to the pollutant. To account for the morbidity effects caused by the contaminant, the above equations can be appropriately modified by subdividing the population into classes of individuals with certain health conditions.

As a simple example, the growth of a population is controlled by the following balance equation:

$$\textit{rate of variation of population} = \textit{natality rate} - \textit{death rate}$$

stating that the rate of variation of the number of individuals is equal to the number of births minus the number of deaths per unit time.

The dynamics of a population are characterised by three possible conditions: a) the natality rate is higher than the death rate; b) the death rate is higher than the natality rate; and c) the two rates are equal. The above conditions correspond, respectively, to the increase, the decrease and the stability of the population size.

On the hypothesis that the reproduction and death processes depend solely on the biological characteristics of the individuals, we can assume that both the natality and the death rates are proportional to the population size. If the above rates are constant on time, only when condition c) is fulfilled does the population reach a constant stable size (the so-called steady state). On the contrary, if condition a) is verified, the population grows indefinitely, whereas, in case of condition b) the number of individuals decreases until the population goes extinct.

However, both birth and death rates are controlled by a number of further factors that depend on the limitation of the environmental resources, on the competition among individuals and among species for the exploitation of these resources and, finally, on the adverse effects of the interactions among populations such as predation and parasitism. The whole of the above mentioned limiting processes, commonly defined as *environmental resistance*, keeps the population from reaching the maximum *biotic potential* (the maximum growth capacity of a population when the resources are unlimited).

An important consequence of environmental resistance is that the natality and death rates depend on the size of the population. Indeed, in general, when the number of individuals increases, the effects of the above-mentioned limiting factors increase correspondingly. It is worthwhile to note that, in real circumstances, the situation can be more complicated. For instance, fluctuations of the environmental conditions can cause oscillation of the population size.

Let us assume that two populations reach the same steady state size although they survive in different conditions of environmental resistance. For instance, one population has adapted to a more stringent environment by means of some competitive advantage and can maintain its size through achieving a higher biotic potential, for example, through the development of more successful reproduction strategies. Furthermore, we will assume that the individuals of the two populations show the same sensitivity to a given stressor (in other words, the increase in the probability of dying is the same for the individuals of both populations). We will try to answer the following question: should we expect that the effects, namely the decrease in the number of individuals, will be the same for both populations? The population in more competitive conditions can better respond to the stressor impact as it maintains its size thanks to a higher biotic potential. Therefore, this population, in principle, should be less affected by the adverse impact of the stressor. This shows the importance of the role that ecosystem properties, such as environmental resistance, can play in controlling the emergence of systemic behaviours. The following mathematical example will help to better understand the reasons for the emergence of the postulated systemic effects.

In the absence of limiting factors, when the population growth can reach the maximum biotic potential, we have:

$$\frac{dN}{dt} = k_n N - k_m N \qquad (1A)$$

where N is the population size at instant t, k_m is the probability per unit time of individual death and k_n is the average number of births per individual and per unit time.

To account for the effects on the population dynamics of the existing limiting factors, we should subtract a term *f(N)*, representing the environmental resistance, from the right-hand side of Eq. (1A):

$$\frac{dN}{dt} = k_n N - k_m N - f(N) \qquad (2A)$$

Let us assume, to fix our ideas, that *f(N)* is a quadratic function:

$$\frac{dN}{dt} = -k_m N + k_n N - k_2 N^2 \qquad (3A)$$

k_2 can be interpreted as a measure of the environmental resistance: the higher the value of such a parameter, the greater the environmental resistance.

When Eq. (3A) is valid, the population size reaches the steady-state value N_0:

$$N_0 = \frac{-k_m + k_n}{k_2} \quad (4A)$$

Although the biotic potentials and the environmental resistances are different, two populations can reach the same steady-state size provided that the values of the ratio in Eq.(4A) is the same for both. This means that the population with a higher environmental resistance must be characterised by a proportionally higher biotic potential. If a linear dose-response relationship without threshold is assumed, we can write:

$$k_m^* = k_m + aD \quad (5A)$$

where D is the dose rate, k_m* is the perturbed value of $k_m$and *a* is a positive quantity that measures the sensitivity of the individuals to the stressor effects.

The ratio between the perturbed and the original population sizes is

$$\frac{N_0^*}{N_0} = \frac{-k_m - aD + k_n}{-k_m + k_n} = 1 - \frac{aD}{-k_m + k_n} \quad (6A)$$

As the value of *a* is the same for the two populations (hypothesis of an equal sensitivity to the stressor) and $-k_m+k_n$ is higher for the population living in an ecosystem with a higher value of k_2, from Eq. (6A) we can see that the detrimental effects (the decrease of the population size) of a given dose depend on systemic properties (the environmental resistance that is related, in this model, to the value of k_2).
The same direct effects of a stressor on the individuals can cause different consequences on the populations due to complex responses of eco-systemic nature.

4.3.2. Consequences for risk assessment in the aquatic environment

The consideration of ecological complexity as briefly discussed above leads to the identification of some general tenets that should guide risk assessments and which can be used, in particular, to select reference species. Such principles should be considered valid for the protection of the ecosystem against chemicals and radionuclides.

4.3.2.1. Ecological role

The ecological role of the functions that different kinds of organisms have in the aquatic ecosystem is of importance for risk assessment.

Essentially, the focus is on the evaluation of the response of the so-called “functional groups” to the environmental stresses. The fundamental principle is that if there is an alteration in a functional group or in the main species characterizing a functional group, then the structure and function of the whole ecosystem will change. On the contrary, the alteration of a limited number of components of each functional group does not necessarily imply a significant modification of the ecosystem functioning. For instance, since there are generally many populations of different species of phytoplankton in a lake, the structure and/or the function of a given lake ecosystem will likely not change if one or a few phytoplankton species would be eliminated. In spite of the fact that the biological diversity of the phytoplankton group would be altered, there would remain many phytoplankton species in the system that would carry out the same work so that the structure and function of the whole ecosystem would be unaltered. Essentially, functional groups of fish are piscivores (predatory fish) that consume other fish (prey fish) and non-piscivores; the latter group includes omnivores (eating “everything”), planktivores (eating zooplankton), benthivores (eating zoobenthos) and herbivores (eating plants), (see Håkanson et al., 2010).

4.3.2.2. Species sensitivity

Aquatic organisms receive external exposure to radionuclides in water and sediments and internal exposure to nuclides accumulated in their tissues. The calculation of such doses is the goal of several dosimetric models (IAEA, 1988), which use as input the amount of radionuclides in water and sediment.

The effects on aquatic organisms from ionising radiation concern individual organism species and functional groups, and the variables of interest are, as previously noted, mortality rate, fertility rate and mutation rate (Whicker and Schultz, 1982). There are major differences between different organisms in sensitivity to ionising radiation, and there are major differences for the same organisms living in different environments (i.e., the importance of temporal events) or different organism of the same species (i.e., different life stages or genetic diversity). This is reflected in the wide intervals (UNSCEAR, 1996) of the recommended dose rate limits (Table 7) (IAEA, 1988).

Fish are the most sensitive to radiation among the organisms belonging to the non-mammalian phyla in the aquatic environment. If we assume that the acute lethal dose measures the organisms’ radiosensitivity, it is possible to rank some taxonomic groups (from the most to the less sensitive) as follows: fish > crustaceans > molluscs (Blaylock et al., 1996).

Table 7: Examples of recommended dose limits ($\mu Gy.h^{-1}$) to biota in aquatic environment

	NCRP, 1991	IAEA, 1988 and 1992	Thompson, 1999
Freshwater organisms	400	400	
Benthic invertebrates			100
Fish			50
Deep ocean organisms		1000	

4.3.2.3. Species Susceptibility

The sensitivity to radiation is not the only factor that should be accounted for to assess the damage that a species can potentially suffer from ionizing radiation. The sensitivity is indeed strictly related to the biological response of the individual organisms to the effects of radiation. However, the risk of the species experiencing adverse effects of a stressor in a given ecosystem is of importance for the assessment of the impact on populations. This risk is associated with the particular features of the organism's exposure, such as the frequency and the duration, and can be related to the specific behaviour and ecological niche occupied by a species. Like species sensitivity, species susceptibility is strictly related to the particular kind of contaminant. For example:

- It is well known that some radionuclides, such as Pu isotopes, strongly interact with bottom sediments in the marine environment. Consequently, very high concentrations of these radionuclides in sediment can be associated with relatively low contamination levels of water. In these conditions, organisms such as zoobenthos and benthivorous fish can be more exposed to the effects of contamination than pelagic fish.
- In aquatic ecosystems top predatorsare very susceptible to radiocaesium. For example in lakes of the northern hemisphere, the exposure of a pike is due to the external irradiation from radionuclide dissolved in water and to the radionuclide incorporated in the body of the fish. Whereas the external irradiation can depend on the mass and the shape of the pelagic fish and, consequently, may be similar for other fishes characterised by different feeding habits but having similar mass, the internal irradiation depends on radionuclide accumulated by the fish. It is well known that, due to the particular turn-over of radiocaesium in fish and to the uptake of radionuclides from thepredation of other fish species, pike can show persistent levels of contamination that are often higher, especially on the long term, than those observed in non-predatory fish.

4.3.2.4. Summary and Conclusions

The protection of fish populations is a key issue for the sustainable management of the marine and the fresh water environments. However, human activities can threaten the aquatic ecosystems by over-fishing, habitat degradation and destruction. Therefore, the scope of protection for the aquatic ecosystem is wider than the simple protection of fish stocks from the adverse effects of contaminant substances.

In principle, the evaluation of the impact of chemicals and radionuclides on biota living in the aquatic environment is based on the assessment of the pollutant behaviour in the ecosystem and of the alteration of the population dynamics caused by the induced changes in individual organism mortality, fertility and morbidity. Nevertheless, as noted in section 4.1.3, there are major differences on the methodological approaches to assess the environmental risk associated with the above mentioned classes of contaminants. Indeed, the focus of chemical risk assessment is on pollutant concentrations in the ecosystem components whereas radionuclides risk assessment focuses on the evaluation of the doses received by organisms. Moreover, the assessment of risk associated to radionuclide exposure is based on the relation of effects to doses rather than to contaminant concentrations. Such an approach does not necessarily require separate consideration of each radionuclide (effects are, indeed, related to the type of radiation rather than to the specific radionuclide). In general, the similarities and the differences between chemical and radiological risk assessments reported in Table 3 of section 4.1.3 are also valid for the aquatic environment.

The identification of effect threshold values (Table 7) and of reference organisms is a common approach frequently adopted for chemicals and radionuclides. In particular reference aquatic organisms recommended by ICRP 2008, include trout, flatfish and crab for which a great deal of information concerning radiosensitivity are available. Nonetheless, it should be recognised that in order to select a pertinent set of representative organisms for the different ecosystems, the objective stressed here to account for the susceptibility and the ecological role of the many species living in the aquatic environment is a high priority area for further research.

5. Using the Ecosystem Approach to identify research priorities for radiological protection

In the area of radiation protection, the ecosystem approach builds upon the reference organism approach. Using the ecosystem approach gives a top-down view of the environment including a more robust ecological understanding and realism meeting the overall protection objectives (as mentioned in section 1). As such, it can identify areas of research that are needed to complement the reference organism (or RAP) approach in order to improve radiological protection.

Although the discussion in the following sections is restricted to the specific case of radiation, it is to be stressed that very similar shortcomings are still affecting the field of environmental risk assessment of chemicals. Most traditional approaches to ecological risks associated to other stresses than radiation (such as those based upon ecotoxicology) are quite similarly suffering from the same limitations. It is therefore a priority to pinpoint the research developments that are necessary in this area, and to implement in its current approach the best flexibility such as to allow for easy later incorporation of the latest advancements.

5.1. Research priorities to complement the Reference Organism concept

The simplification of nature's complexity that is imposed by the pragmatism of the reference organism approach yields a number of shortcomings, which need to be examined in more details. One major uncertainty comes from the scarcity of data relating to the description of the various organism types and their exposure/sensitivities to ionising radiation up to a dosage that would be appropriate as input to the assessment methods. This scarcity of data may be addressed through use of extrapolations from what is known, however care is required to ensure that the extrapolations are reasonable and justified (Landis, 2002). Current critical issues in assessing the ecological effects of radionuclides include extrapolating 1) from acute exposures and high doses to chronic exposures and effects at low doses, 2) from external to internal irradiation, 3) from single radionuclides to multiple contaminants, 4) from one species to another, 5) from organism to population, to community, and to ecosystems.

5.1.1. Extrapolation Issues and Related Uncertainties

A critical issue in the assessment of the possible impacts on biota from exposure to ionising radiation in a polluted environment relates to the extrapolation of the relatively limited quantitative data that is available, to the practical and actual conditions of exposure in the environment. Most of the published data (recently assembled and evaluated within the FREDERICA data base with respect to the four relevant umbrella endpoints: mortality, morbidity, reproductive success and cytogenetic effects) relate to external exposure to gamma irradiation in acute exposures observed at the level of individual organisms. There are therefore quite significant limits of knowledge in the area of biota chronically exposed to internal irradiation from bio-accumulated α and β emitting radionuclides.

Extrapolations are used when reaching operational goals is prevented due to serious shortcomings in knowledge. Extrapolations therefore always result in associated uncertainties which need to be clearly stated when the final expression of the protection goal is given. The validity of the result obtained has a validity which is directly related to the associated uncertainty and in practice, when the result is obtained, the fact that this is at the expense of uncertainties often tends to be forgotten. Such extrapolation issues are briefly discussed below (for more detailed developments, see Garnier-Laplace et al., 2004).

5.1.1.1. Acute Versus Chronic Exposure

There is substantial evidence that a dose delivered over long time duration is less harmful than the same dose delivered acutely, probably as a result of adaptive responses and the activation/efficiency of DNA repair mechanisms (Volkert, 1988). However, there are several factors making generalisations as to the quantitative relationships between these exposure conditions difficult. Indeed, there are comparatively few data that relate directly to the chronic, low-level irradiation conditions of relevance for wild organisms, i.e., exposures at dose rates of 10-100 $\mu Gy.h^{-1}$ over the life span of the organisms, and the response endpoints most commonly assessed after acute, high dose irradiation. Furthermore, the qualifiers low-level, chronic, high-level, acute that one often reads in the radiobiological and radioecological literature are generally used without any precise definition: a radiation exposure lasting several days may be effectively "chronic" for a short-lived organism, but effectively "acute" for a long-lived organism. It is therefore not possible to provide a simple or generalised procedure for extrapolating between the two exposure situations, i.e., extrapolation from high acute dose and dose rates (several Gy at 10-100 $Gy.h^{-1}$) of low LET γ and X-rays to lower doses (<1 Gy) accumulated at lower dose rates (1-100 $\mu Gy.h^{-1}$).

5.1.1.2. External Exposure Versus Internal Exposure

External exposure to radiation is mediated through low-LET radiation whereas internal exposure occurs when radionuclides have been incorporated within biota therefore including both high-LET (α and β radiation) and low-LET radiation (γ radiation). The point of importance is that for the same absorbed dose, high- and low-LET radiations do not result in the same level of biological harm, due to differences in the damage promoted by single tracks of these different radiation qualities. This has led to the definition of the Relative Biological Effectiveness

(RBE) as a weighting factor used (in human radiological protection) to extrapolate from external exposure to internal exposure. A similar procedure is currently envisioned for the RAPs approach to be applied to the endpoints of interest to biota (mortality, morbidity, reproductive success, and mutation), but within a context of very limited published relevant data that could support a generalisation of this extrapolation (Garnier-Laplace et al., 2004).

5.1.1.3. Single Contamination Versus Multi-contamination

The knowledge about ecotoxicity of pollutant stressors is usually gained through testing each pollutant in isolation on various biota and endpoints, and then constructing the corresponding pollutant-induced dose-response curves. The global dose delivered from exposure to a mixture of several radionuclides is hence estimated from summing up all the individual doses, assuming that application of this rule of effect summation is correct. However, many real contamination scenarios will involve complex mixtures of toxicants, including for example radionuclides, metals, or pesticides, as well as potentially other concomitant stresses (e.g., UV, temperature pH, etc.), the resulting global exposure not necessarily being additive. Indeed the very scarce literature on this subject has pointed out the potential occurrence of synergistic as well as antagonistic effects of contaminant mixtures (Garnier-Laplace et al., 2002; Vanhoudt et al., submitted).

5.1.1.4. Species/Life Stage Versus Another Species/Life Stage

There is abundant evidence that there are substantial variations in the radio-sensitivities of organisms both within and among taxonomic groups. This differential sensitivity also extends to different stages of the life cycle for any given organism. The extrapolation methods require taking due consideration of the possibility that the dose response profile used in the assessment does not necessarily account for the most sensitive species or life stage. Current efforts are in progress in various research groups to apply advanced statistical treatments to sets of data (e.g., species sensitivity distribution techniques, see Posthuma et al., 2002, and uncertainty analysis) in order 1) to compare various extrapolation methods and strengthen their validity, and 2) to identify in a systematic manner the knowledge gaps generating the largest uncertainties (Iospje et al., 2003; Garnier-Laplace et al., 2004).

5.1.1.5. Individual Organisms Versus Higher Levels of Biological Organisation

The focus on individual organisms, as settled in the reference organism approach, finds its justification along three main lines of consideration. First, the vast majority of radio-toxicological data addressing the dose-effect relationship in animals and plants refer to individual organisms, useful information concerning higher levels of biological organisation still being scarce or difficult to translate into practical use. Second, if ionising radiation is to promote harm at the population or higher levels in the biological hierarchy, this is mediated through individual organisms first, or through effects at lower levels. Thus, the consequence of the two afore-mentioned points is that the development of the best feasible assessment methodology in practical terms needs today be based upon individual organisms. Altogether, this leads to the consideration of the organism as a priority driven by practical reasons, a choice which also leads to the need to address extrapolation to higher levels in the biological hierarchy if the system is to demonstrate the robustness and width of its protection goals.

Recent examination of this extrapolation issue, across levels of biological organisation, seems to indicate that measures taken to limit radiation damage in individual organisms to an acceptable degree would also provide a sufficient degree of protection for the populations of such organisms. However, this does not provide an answer as to the magnitude of population effects at a certain level of effects in individual organisms. Indeed, the sum of the parts does not necessarily equal the whole. Furthermore, the maximum level for a population effect to be considered as acceptable is not clear, either from a risk assessment perspective or an ecological point of view. Uncertainty is inherent in extrapolating from effects upon individual organisms to alteration in ecological structure or function. In particular, subtle differences in individual organisms within a population can give rise to significant evolutionary events.

It may be of relevance here to mention that in a neighbouring toxicological field, current effects of endocrine disruptors, which are observed in fish population world wide have not been first detected in individuals. Indeed, there is today a trend within the ecotoxicology community to recognise that protection of interacting populations and communities within ecosystems being exclusively based upon toxicity data gathered at the organism level, and most often in artificial/optimal laboratory conditions, is insufficient, if not sometimes misleading. This is best illustrated by the recent work of the SETAC[37] Ecological Risk Assessment Advisory Group that includes a “population level risk assessment” work group that emphasized the importance of examining population-based metrics of effects, acknowledging that this cannot be adequately done by superficially modifying the present paradigm that relies on organism-based metrics to predict consequences. In particular, it is important to stress that such an extrapolation can either underestimate or overestimate risk (a 5 % decrease in a population may have no effect on ecosystem function, or may be catastrophic, depending on the resistance/resilience status of the population ecosystem considered).

[37] SETAC: Society of Environmental Toxicity and Chemistry (www.setac.org)

5.1.2. System level issues

The endpoints considered by the reference organism approach, namely mortality, morbidity, reproductive success and chromosome alteration, are all focused at the organism level, and cannot capture the upper level population and community parameters that characterize ecosystem effects.

5.1.2.1. Ecosystem Level and Indirect Effects not Captured

The "reference organisms" approach as derived from traditional ecotoxicology that is based upon single-species testing will resolve direct effects only. The approach therefore fails to capture indirect effects, which are often reported to be responsible for long-term impacts in various fields of environmental protection. For example: atmospheric CO_2 accumulation is impacting human and organism well-being via the dynamic displacement of ecological processes that are controlled by temperature; toxic substances (pesticides, persistant organic pollutants and some organo-metals like methyl-mercury or organo-selenium) may undergo bio-magnification along trophic chains, therefore impacting top predators including man, although having little or no direct impact on species at lower trophic levels.

Indirect effects appear as a consequence of alterations in the dynamics of interacting species (De Angelis, 1996), even when the organisms affected are not directly exposed to or affected by the stressing agent. Many studies have revealed that indirect effects are important (Fleeger et al., 2003); in some circumstances, they may be more pronounced than direct effects (Paine, 1980, 1992; Yodzis, 1988; Wootton, 1994; Abrams et al., 1996). Indirect effects may also become apparent through changed ecosystem attributes such as nutrient cycling and oxygen metabolism (Kersting, 1994; Vanni, 1996). Indirect effects thus demonstrate that species that are themselves insensitive to a particular stressor might be seriously affected or even become extirpated due to diminished or lost food resources, predation or emigration that lowers the population density to a point below a critical minimum, or other effects that lower fitness. Indirect effects may also result in increases in population size (i.e., through release from predation pressure, see the example described on Figure 10 for UV radiation drawn from experiments reported by Bothwell et al.,1994) or certain ecosystem functions, which may cause imbalances in the ecosystem. Conversely, the absence of a particular species in an ecosystem is not necessarily the result of exposure to a particular stressing agent. Indirect effects may also become apparent through changed ecosystem attributes such as nutrient cycling and oxygen metabolism (Kersting, 1994; Vanni, 1996). As for non-radioactive pollutants, the potential for radiation to induce indirect effects through interaction with population, community, or ecosystem-level processes such as trophic and competitive relationships has also been stressed (Bréchignac, 2003; Doi, 2004; Hinton and Bréchignac, 2004). With quite unclear relevance to the ecosystem level of effects at this stage, it may be worth mentioning also that radiation has been shown to promote "non-targeted effects" such as the bystander effect" where radiation is shown to indirectly cause alterations to cells surrounding the actual target of irradiation (Mothersill and Seymour, 2005; Salomaa, 2008).

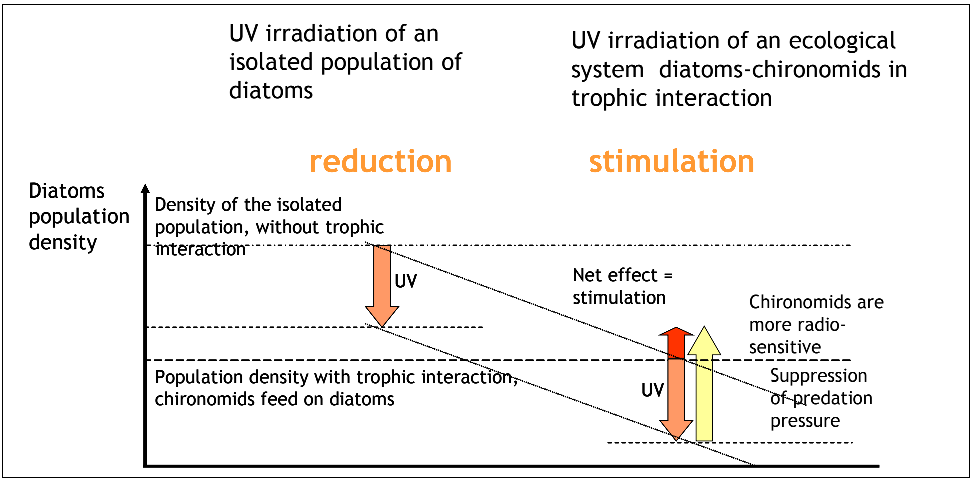

Figure 10. Ecosystem response to solar ultraviolet-B radiation: Influence of trophic-level interactions (drawn from Bothwell et al., 1994).

Furthermore, it is clear that the reference organism approach is not designed to capture ecosystem-level effects. At this level, whilst still impacting individuals, toxicants may also affect other emergent processes at this level of organisation, the ecosystem level bringing additional constraints to the system (predation, competition) which are prone to yield system dynamics quite different from its subsystems. This is because the ecosystem associates various populations of different species playing different, but complementary (symbiotic-like), functional roles where important features are hierarchical self-organisation and matter-energy cycling. At this level, interdependence between populations often overrides the single responses of individuals, a feature which can promote a loss of predictability from subsystems to system behaviour (Cairns, 1992; Suter, 1993; Taub, 1997).

5.1.2.2. Long-term Trans-generational Impacts not Captured

The reference organism approach considers reproduction as a relevant endpoint at the organism level that is meaningful for higher levels of biological organisation. However, with respect to the time scale through successive generations which makes up populations, communities and ecosystems, this is only the starting point. Figure 11 illustrates this feature with an example showing increasing severity of harm from chronic exposure of *Daphnia* to uranium through three successive generations (Massarin et al., 2010)[38].

Important features such as the inheritability of chromosomal mutation/damage or alterations in the genetic structure of population that could drive competitive and succession alterations[39], that are among important determinants of ecosystem effects, cannot be accounted for. Of particular relevance here is the manifestation of genomic instability in the long-term that a growing set of authors is reporting to be induced by radiation (Little, 2003; Salomaa et al., 2008; Vorobtsova, 2008).

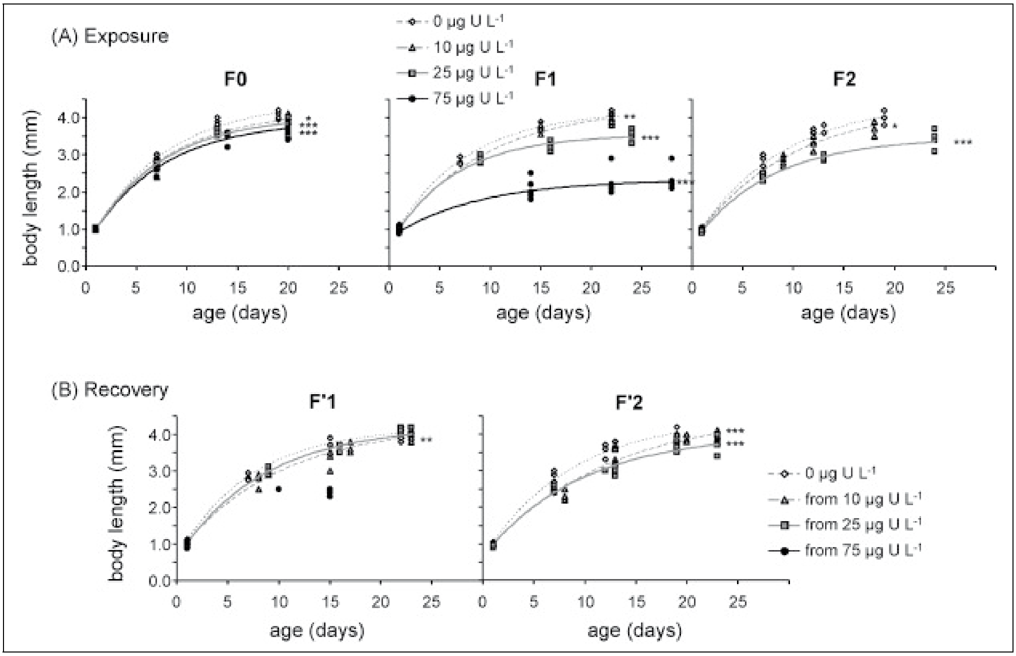

Figure 11. Daphnid body length (mm) in relation to age in the successive generations exposed to uranium concentrations of 0, 10, 25 and 75 µg.l^{-1} (F0, F1 and F2). Statistics: $^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$. (From Massarin et al., 2010).

5.1.2.3. Classical Bottom-up Approach that may Miss Important Issues

The reference organism approach takes a classical bottom-up approach, as evolved from the discipline of mammalian toxicology, that belongs to the philosophy of micro-explanation (reductionism), where “the properties and powers of individual things of materials are due to their fine structure, that is due to the disposition and interaction of their parts” (Harré, 1972). Within this context, this mode of exploring causation has proved to be very profitable. However, unique properties emerge at higher levels that are difficult or impossible to predict by micro-explanation alone. Conceding the unpredictability and importance of emerging properties in ecological systems, macro-explanation flows top-down and aims at describing consistent details and predictable behaviours of the whole without becoming completely entangled in a complex web of micro-explanation.

To favour one of these two modes of investigation will necessarily become inconsistent at some stage as it will cover only one part of the field of explanation and then miss issues that can only be grasped by the other. Any level of biological organisation faces simultaneously in two directions. It is a whole composed of parts while also being a part of a larger whole. Such a unit within a hierarchical system is now called a “holon” in modern ecological theories (cf. Chapter 2). Macro-explanation and micro-explanation are most valuable and error-free if used together and with a full understanding of their respective shortcomings.

[38] One should note however that a statistical difference (e.g. A) does not necessarily lead to a biological significance.

[39] It is to be stressed that *Daphnia*, as per the example given in Figure 11, is a clone and not a biological population proper.

5.1.2.4. Abiotic Components not Explicitly Considered

Concentration/activity of toxicants in abiotic compartments has most often been used as a metrics for assessing environmental risk, especially within regulatory contexts. Although arising from radioecology, a discipline that has for a long time promoted better description and understanding of environmental radioactivity within the various abiotic compartments, environment radiological protection is currently evolving with an almost exclusive focus on living matter. Living matter indeed constitutes the radiosensitive components of life, from molecular (DNA), through cells, tissues and organisms, up to ecosystems through further complex assemblies, and radionuclide activities in abiotic compartments are used to support dose calculation to biota within a source-to-sink unidirectional understanding.

However, ecosystems feature a bi-directional relationship that link abiotic and biotic components together, and one must realize that if radionuclide activity in abiotic compartments is susceptible to promote harm to biota, biota in turn may also change the concentration/dispersion of radionuclides in abiotic compartments based on ecologically relevant processes (biomagnification, migration, accumulation of waste, erosion, transport). Alterations to ecosystems can also change biogeochemical cycling of a wide range of elements, thus impacting the abiotic components of the ecosystem.

5.1.3. Reductionism *versus* holism

There has been much controversy in the past, and this appears to be still going on, between the tenants of reductionism addressing the mechanistic efficacy of micro-explanation, and the tenants of holism looking at ecological systems and their inherent complexity tackled through a macro-explanation path. Unique properties are indeed emerging at higher levels that are difficult or impossible to predict by micro-explanation alone. Facing the unpredictability and importance of emergent properties in ecological systems, holism aims at describing consistent details and predictable behaviours of the whole without becoming completely entangled in a complex web of micro-explanation. However, both usually agree that risk assessment should assure the value of ecosystems.

Hence, it is highly desirable to promote a move from such extreme positions, where ecosystem approaches would inherently be considered as useless because impractical, or conversely that only ecosystem approaches would be legitimate. The purpose of this report is to make the case that for pertinent and efficient ecological risk assessments both are actually needed, and therefore to pinpoint the communication and harmonisation effort that is needed to allow for their coexistence. It is clear that some of the controversy roots from the fact that those who customarily work at one level of organisation commonly develop habits and thoughts that are not appropriate to other levels of organisation. They need therefore some acculturation and reciprocal listening and understanding before reaching a successful union of efforts.

5.2. Reference Organism and ecosystem approaches are complementary

It is of great interest to place the reference organism approach in the general context of environment protection also featuring the ecosystem and its main attributes. An attempt is made (Figure 12) showing a nested (holons) representation of the scale of biological complexity with the various processes corresponding to the different levels within this scale.

First, the biotic part of the environment is no longer represented through a linear chain running from DNA up to communities and ecosystems, as has been common for decades in the ecotoxicology and radiation literature, because it is an inaccurate and misleading description. Modern ecological representations of nature recognise the nested structure of ecological systems, “holarchy”, that encapsulates one level (cells, for example, holon n) within the next (individuals, for example, holon n+1), emphasising that extrapolation from one level to the next is not straightforward and is largely governed by sets of actions and feedbacks. Such a nested structure applies to all living organisms whatever their functional role within the ecosystem (producers, consumers or decomposers), and it forms a basis for understanding equilibrium, multiple equilibrium and non-equilibrium exhibited by ecological systems as developed in the theory of hierarchical patch dynamics (Wu and Loucks, 1995).

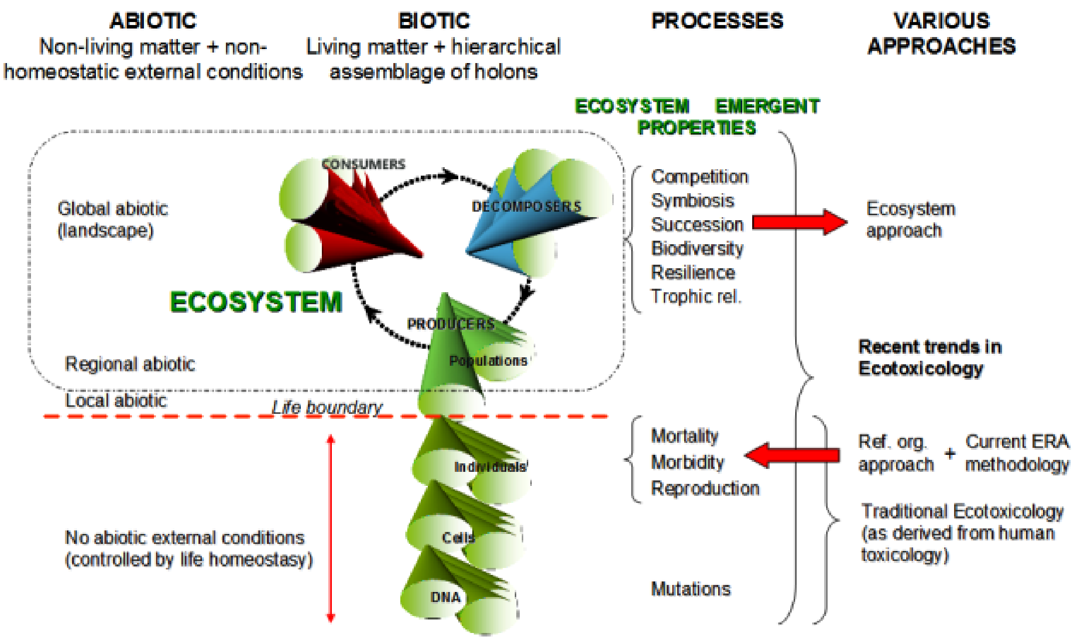

Figure 12. Structure and self-organisation of ecosystems with interacting biotic and abiotic components in relation to the major life processes occurring at the various levels of biological organisation (ERA: Ecological Risk Assessment). The horizontal dotted line called "life boundary" indicates a structural transition occurring at individual organism level on the scale of biological complexity. (From Bréchignac, 2009).

In the abiotic part of the environment, there is discontinuity at the organism level, where a transition occurs from a domain of homeostasy (self-control of abiotic conditions that occur internally to the organism) to a domain where external conditions (local, regional or global) are not controlled dominantly by life processes. From this representation, one understands better why a framework designed primarily around organisms such as the reference organism approach does not need to consider explicitly the abiotic environment. However, it is of importance to stress that the relationship between populations of living organisms and their abiotic environment is not uni-directional, one usually tending to only consider the influence of abiotic conditions on life, but bi-directional, as both can affect each other.

The scheme also reveals that there is a big conceptual jump from populations to ecosystems where additional processes appear, the so-called "emergent properties of ecosystems" which are core to the various ecosystem approaches suggested in several contexts. Finally, the emerging trend in modern ecotoxicology is to tackle ecological assessments from both ends concurrently, bottom-up along a micro-explanation pathway, and top-down along a macro-explanation pathway, as none of them can adequately represent alone the relationship between nature and toxicants[40].

[40] Another consequence from this dual approach to ecological assessment is the need to be able to evaluate multiple models.

6. Legislation about environment protection

While the focus of nuclear law has traditionally been on the protection of people and property, recent international law developments indicate that environmental law is indeed in evidence in the nuclear field and that its significance is increasing steadily[41]. Nonetheless, it has been recently observed that the influence of environmental law has not yet succeeded in ensuring that the environment is effectively protected by international nuclear law (Emmerechts and Sam, 2010). In this context, it is useful for the radiation protection community to have an overview of legal developments concerning the protection of the environment in other fields.

An increasing number of international conventions and agreements show reference to the “ecosystem approach” or references to ensuring ecosystem health or integrity. This chapter provides an introduction to environmental protection law, and then focuses on the development of ecosystem approaches in legal instruments of environmental protection and natural resource management. These trends are illustrated with reference to a few selected specific examples drawn from national, regional and international environmental instruments with the aim of showing how ecosystem approaches have been introduced in a range of contexts.

6.1. The development of environmental protection law

It is only in the 20th century - and especially in the second half thereof - that public opinion begun to demonstrate concern over the general state of the environment, leading to initiatives measures to address pollution of inland waters, oceans, air and soil, and to protect biological diversity and habitats. In recent decades, awareness has increased as to more subtle impacts and the interdependence of all elements of the biosphere with a subsequent emphasis on measures to ensure environmental sustainability. From responding to damage after it has occurred, legislation sought to prevent environmental harm through enhanced monitoring and analysis of environmental trends, the requirement to undertake assessments such as EIA/SEA (Environmental Impact Assessment / Strategic Environmental Assessment), and through deepening and broadening public access to information and participation in decision-making.

During this period, there has been a marked development of international environmental law. This can be seen in growth in the number Multilateral Environmental Agreements (MEAs) negotiated to address environmental issues, the number of states and regional organizations which have become parties to these, and the MEAs growing scope and evolving character[42].There are now literally hundreds of international treaties and other agreements, protocols, and amendments related to the environment, of which well over half date from the period between 1972 and the early 2000s (UNEP 2007), as illustrated in Figure 13. Highlights in the development of international environmental law include the outcome of the 1972 Stockholm Conference on Human Development; the 1987 World Conference on Environment and Development; the 1992 UN Conference on Environment and Development; the 1995 Jakarta Mandate; the 2001 Reykjavík Conference; and the 2002 World Summit on Sustainable Development.

[41] Environmental law entered the nuclear field both *directly* by making nuclear activities subject to international environmental law, and *indirectly* by introducing the concept of environmental protection in international nuclear law. (See: Emmerechts & Sam, 2008, pp.91-110; UNEP, 2007)

[42] For the purposes of this report, a Multilateral Environmental Agreement (MEA) is "an intergovernmental document intended as legally binding with a primary stated purpose of preventing or managing human impacts on natural resources". Such agreements are considered environmental if they seek, as a primary purpose, to manage or prevent human impacts on natural resources; plant and animal species (including in agriculture, since agriculture modifies both); the atmosphere; oceans; rivers; lakes; terrestrial habitats; and other elements of the natural world that provide ecosystem services. (See: Daily, 1997).

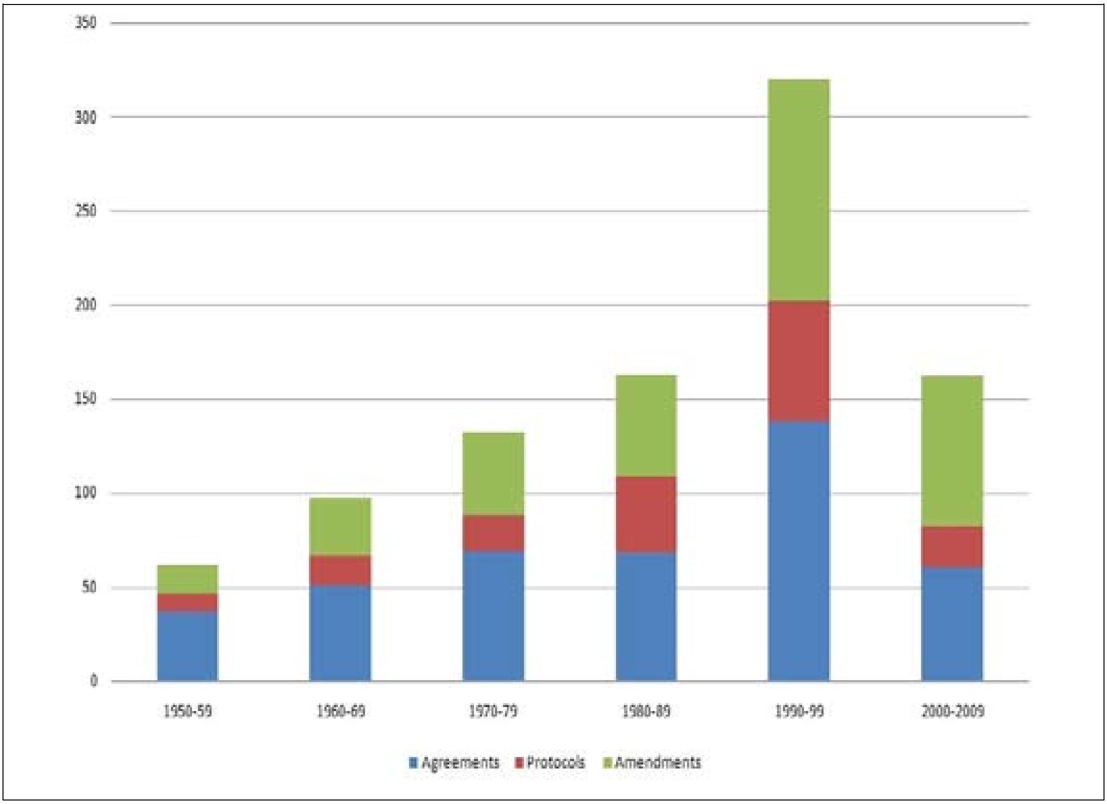

Figure 13: Number of Multilateral Environmental Agreements, 1950-2010 (Mitchell, 2011).

The MEAs are important frameworks for global environmental governance, and their development is an indication of both increasing international recognition of the need to manage human activities that have an effect on the environment and the political will to address the issue concerned. Their expanding scope and increasing sophistication reflects the growing awareness of the scale of the environmental challenges, the complexity of the underlying problems and the need for effective policy and regulatory responses. From an initial focus on particular problems or sectors, subsequent developments have moved towards addressing ecosystems in an integrated, cross-sectoral manner with the aim of ensuring protection of the environment, improving management of natural resources and promoting sustainable development. The MEAs share a common ambition to protect the environment or better manage the use of natural resources but, reflecting their varying origins, sometimes have overlapping or contradictory principles and conceptual goals. The MEAs can be placed on a continuum from traditional top-down, hierarchical hard-law treaties to the vaguest voluntary soft-law mechanisms, with the boundaries between ‘hard’ and ‘soft’ law, in practice, often blurred and difficult to differentiate (Karlsson-Vinkhuysen and Vihma, 2009; Mörth, 2004; Thürer, 2000)[43].

The further evolution of international environmental law will continue to be shaped by the need to more effectively and efficiently manage the risks of increasing environmental change. The future evolution of environmental law is also need to address what in the Malmö Declaration in 2000 was referred to as an “*alarming gap between commitments and actions*” - a gap which ten years later still prevails[44].

A detailed description of international environmental protection and natural resource management legislation is well beyond the scope of this report. Useful summaries have been published recently elsewhere; for example the OECD Nuclear Energy Agency has published an overview of international environmental protection instruments and selected national legislation aiming to protect the environment (OECD Nuclear Energy Agency, 2007), and the ERICA Project has also provided a summary of international and European environmental law and

[43] The essence of *hard law* is legally binding obligation. In contrast *soft law* at the global level can be broadly defined as rules of conduct which, in principle, have no legally binding force but which nevertheless have practical effects.

[44] First Global Ministerial Environment Forum, Malmö (Sweden), 29 to 31 May 2000. This Forum was held in pursuance of United Nations General Assembly resolution 53/242 (28 July 1999) to “enable the world’s environment ministers to gather to review important and emerging environmental issues and to chart the course for the future”.
(See: http://www.unep.org/malmo/malmo_ministerial.htm , accessed 2011-01-04)

instruments dealing with aspects of environmental protection (ERICA Project, 2007)[45]. In addition, the ERICA Project document identifies factors that need to be taken into account in order to comply with environmental instruments, grouping these into a number of functional headings:

- actions which affect the amount of radioactivity entering the environment by controlling the source and are aimed at general environmental protection;
- actions which are aimed at protection of specific ecosystems;
- actions which are aimed at the protection of specific environmental media;
- prospective and retrospective assessment of the impact of the radioactive contamination;
- monitoring or measurement of the impact;
- gathering or dissemination of information;
- decision-making;
- specific factors which relate to unusual events, i.e. radiological accidents or emergencies.

6.2. Implementing ecosystem approaches

Although there is no single accepted definition of what constitutes an ecosystem approach, the concept is generally understood to encompass the management of human activities, based on the best understanding of the ecological interactions and processes, so as to ensure that ecosystems structure and functions are sustained for the benefit of present and future generations. The concept builds on a number of elements, such as application of the precautionary approach and more integrated management strategies, with greater emphasis on consideration of goals and objectives at the level of ecosystems. The sections which follow set out examples of ecosystem approaches developed in a range of contexts.

6.2.1. The Marine Environment and ecosystem approaches

The largest cluster of multilateral environmental agreements is related to the marine environment, accounting for over 40 per cent of the total. The overarching framework is provided by the United Nations Convention on the Law of the Sea (UNCLOS) which defines the rights and responsibilities of nations in their use of the world's oceans, establishing guidelines for businesses, the environment, and the management of marine natural resources[46].

The international community has progressively agreed on principles to ensure the conservation and sustainable use of the world’s oceans and seas. Increasingly these have incorporated and emphasise ecosystem approaches. The major developments can be summarised roughly as follows:

- the United Nations Convention on the Law of the Sea (UNCLOS), which sets out the overall legal framework within which all activities in this field must be considered;
- Chapter 17 of Agenda 21, adopted in 1992, which remains the fundamental programme of action for achieving sustainable development in respect to oceans and seas;
- the 1995 Global Programme of Action for the Protection of the Marine Environment from Land-Based Activities;
- the 1995 UN Fish Stocks Agreement[47], and the FAO Code of Conduct for Responsible Fisheries;
- the 1992 Convention on Biological Diversity and Decisions II/10 (conservation and sustainable use of marine and coastal biological diversity) and V/6 (ecosystem approach) taken under it, which set out vital aims, principles and operational guidance for an equitable and integrated approach to conservation and sustainable use of the marine and coastal environment.
- the commitments made in 2002 at the World Summit on Sustainable Development[48] which highlight the issues on which action is most urgently needed, including in particular:

[45] ERICA Project, 2007. Review of international legal instruments that may influence decision-making, ERICA Project, Deliverable D8 Annex A, European Commission, Luxembourg. Available at <https://wiki.ceh.ac.uk/download/attachments/115017395/FP6+ERICA+Deliverable+D8+Annex+A+-+20+Mar+07.pdf?version=1&modificationDate=1263814194000> (accessed 2011-04-11). This annex identifies the main objectives, scope of international legal instruments of relevance in the European context, and the derived requirements and these are summarized in Table 1.1 of the D8 Annex A report.

[46] UNCLOS resulted from the third United Nations Conference on the Law of the Sea (UNCLOS III) which was concluded in 1982, replaced four 1958 treaties, came into force in 1994 and has to date 158 parties. The secretariat is provided by the UN Secretary General as performed by the UN Division for Ocean Affairs and the Law of the Sea (DOALOS). (See: http://www.un.org/Depts/los/Links/Links-home.htm, accessed 2011-01-04)

[47] The United Nations Agreement for the Implementation of the Provisions of the United Nations Convention on the Law of the Sea of 10 December 1982 relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks.

[48] http://www.un.org/jsummit/html/documents/summit_docs/2309_planfinal.htm

- encouraging the application by 2010 of the ecosystem approach;
- maintaining or restoring fish stocks to levels that can produce the maximum sustainable yield, with the aim of achieving these goals for depleted stocks on an urgent basis and where possible not later than 2015;
- putting into effect the FAO international plans of action, in particular the International Plan of Action for the Management of Fishing Capacity by 2005 and the International Plan of Action to Prevent, Deter and Eliminate Illegal, Unreported and Unregulated Fishing by 2004;
- establishing effective monitoring, reporting and enforcement, and control of fishing vessels to further the International Plan of Action to Prevent, Deter and Eliminate Illegal, Unreported and Unregulated Fishing;
- eliminating subsidies that contribute to illegal, unreported and unregulated fishing and to over-capacity, while completing the efforts undertaken at the World Trade Organization to clarify and improve its disciplines on fisheries subsidies;
- establishment of marine protected areas consistent with international law and based on scientific information, including representative networks by 2012;
- establishing a regular process under the United Nations for global reporting and assessment of the state of the marine environment, including socio-economic aspects, both current and foreseeable, building on existing regional assessment;
- promoting sustainable patterns of production and consumption, applying *inter alia* the polluter-pays principle; and
- supporting sustainable development of aquaculture.

Following the 2002 World Summit on Sustainable Development, a review by the United Nations Open-ended Informal Consultative Process on Oceans and the Law concluded that continued environmental degradation in many parts of the world and increasing competing demands required an urgent response and the setting of priorities for management interventions aimed at conserving ecosystem integrity. It recognized that ecosystem approaches to oceans management should be focused on managing human activities in order to maintain and, where needed, restore ecosystem health to sustain goods and environmental services, provide social and economic benefits for food security, sustain livelihoods in support of international development goals, including those contained in the United Nations Millennium Declaration, and conserve marine biodiversity. While recognizing that there was no single way to implement an ecosystem approach, it recommended the elements relating to ecosystem approaches and oceans, including the proposed elements of an ecosystem approach, means to achieve implementation of an ecosystem approach, and requirements for improved application of an ecosystem approach[49]. In response, the UN General Assembly recalled that States should be guided in the application of ecosystem approaches by a number of existing instruments, in particular UNCLOS and its Implementing Agreements, and encouraged States to cooperate and coordinate their efforts and take all measures, in conformity with international law, to address impacts on marine ecosystems in areas within and beyond national jurisdiction, taking into account the integrity of the ecosystems concerned. These elements were reiterated by the General Assembly in resolution 62/215.

6.2.2. The Ecosystem Approach to Fisheries (EAF)

The EAF framework has developed on the founding principles and conceptual goals emerging from the long process of elaboration of the foundations for sustainable development, aiming at both human and ecosystem well-being (Garcia et al, 2003). The 1995 FAO Code of Conduct for Responsible Fisheries provided a reference framework for sustainable fisheries, addressing ecosystem considerations, principles, and conceptual goals needed for an EAF. The ecosystem approach to fisheries (EAF) became formally accepted at the 2001 Reykjavik Conference on Responsible Fisheries in the Marine Ecosystem[50]. In 2003, FAO's Committee on Fisheries endorsed the EAF as the appropriate framework for fisheries management.

The goals of EAF are "*to balance diverse societal objectives, by taking into account the knowledge and uncertainties about biotic, abiotic, and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries*". The approach thus intends to foster the use of existing management frameworks, improving their implementation and reinforcing their ecological relevance, and will contribute significantly to achieving sustainable development. It is an extension of the conventional fisheries management paradigm. While conventional management aimed rhetorically at resource

[49] The report of the meeting is contained in document A/61/156.

[50] A good summary of the 2001 Reykjavik Conference on Responsible Fisheries can be found at: http://www.iisd.ca/download/pdf/sd/sdvol61num1.pdf. This is complemented by the output of the IOC–SCOR Working Group on Quantitative Ecosystem Indicators for Fisheries Management, see http://www.ecosystemindicators.org/.

conservation, its history shows that, in practice, it aimed at conservation of livelihoods and employment, using resource conservation as a weak constraint.

The EAF policy provides the backdrop against which the EAF is to be implemented. It defines the main orientations of fisheries and the high-level conceptual goals and constraints, connecting fisheries management to sector-development planning, integrating socio-economic and environmental considerations. High-level goals may include: food security and safety; sustainable livelihoods in remote rural areas; environment and resource rehabilitation; economic efficiency achieving human and ecosystem well-being; maintaining the ecosystem integrity; reducing uncertainty for the industry; ensuring equity within and between generations; and promoting improved stewardship. The strategy turns the conceptual goals into operational objectives, ranking them and defining the time frame within which they should be attained. Operational objectives may relate to: (i) reducing the impact on target and non-target species, e.g. reducing capacity; (ii) protection or rehabilitation of habitats and biodiversity, e.g. through zoning and MPAs; (iii) reduction of risk to the resource and to people, e.g. by improving forecasts; (iv) improvement of food safety, e.g. lobbying for reduction of pollution; and (v) improvement and/or decentralization of governance. For each of these operational objectives, some indicators and reference values should be defined, agreed, and used to determine whether the objective is being achieved. The strategy includes the instruments, measures, technical regulations, and input/output controls needed to achieve them, based on an analysis of effectiveness, available resources, risks, etc.

The role of the scientific community in collecting and organizing a wide range of data and information, analysing it in the perspective of improved policy and decision-making, and communicating the results, is fundamental. Under an EAF, more is required to extend the scope of assessments, better understand ecosystem functioning, tackle the issues of uncertainty and risk, improve forecasting capacity, apprehend natural variability, and navigate between neutral scientific advice and advocacy.

The role of indicators and reference values is as fundamental to an EAF as it is to conventional fisheries management. It would be a major error, however, to reduce implementation of an EAF to the use of some additional environmental or biodiversity indicators. However, as an EAF is an integrated approach, the availability of a set of ecological indicators and reference values is a necessary, but not a sufficient, condition for its implementation. Ecosystem indicators and reference values can be of a bio-ecological, techno-economical, and socio-cultural nature, and relate to objectives and constraints. Existing guidelines suggest that sustainability indicators for fisheries should, in principle, cover stressors on the ecosystem, state of selected ecosystem components, and responses to the measures taken (compliance and performance).

An example of implementing the EAF in practice is provided in section 7.

6.2.3. OSPAR and the ecosystem approach

The OSPAR Convention is the legal instrument currently guiding international cooperation on the protection of the marine environment of the North-East Atlantic. Work under the Convention is managed by the OSPAR Commission, made up of representatives of the Governments of 15 Contracting Parties and the European Commission, representing the European Union[51].

OSPAR is an example of an instrument specifically developed to the protection of specific areas or ecosystems. Other conventions concerned with the protection of particular regional ecosystems include conventions on the protection of the Elbe, Oder, Rhine and Danube, the Alps, the Baltic and the Mediterranean. The OSPAR Commission works under the umbrella of customary international law as codified by the 1982 United Nations Convention on the Law of the Seas (UNCLOS), especially in Part XII and Article 197 on the global and regional cooperation for the protection and preservation of the marine environment. The OSPAR Convention recognizes the jurisdictional rights of states over the seas and the freedom of the High Seas, and, within this framework, the application of main principles of international environmental policy to prevent and eliminate marine pollution and to achieve sustainable management of the maritime area. This includes principles resulting from the 1972 Stockholm United Nations Conference on the Human Environment and of the 1992 Rio de Janeiro United Nations Conference on the Environment and Development, including the 1992 Convention on Biological Diversity.

Overall, the work of the OSPAR Commission is guided by the ecosystem approach[52] - an integrated management of human activities in the marine environment. This is supported by a general obligation of Contracting Parties to apply the precautionary principle, the polluter pays principle as well as best available techniques (BAT) and best environmental practice (BEP), including clean technology. For the purpose of the OSPAR Convention, the ecosystem approach is defined as **"***the comprehensive integrated management of human activities based on the best available scientific knowledge about the ecosystem and its dynamics, in order to identify and take action on influences which are critical to the health of marine ecosystems, thereby achieving sustainable use of ecosystem goods and services and maintenance of ecosystem integrity.*"

[51] See http://www.ospar.org/

[52] It should be noticed however that a recent assessment of radionuclides performed within OSPAR used the ERICA tool based upon the reference organism approach to evaluate dose impacts to biota (OSPAR, 2009).

For OSPAR, the application of the ecosystem approach integrates conservation and management approaches, such as marine protected areas, or measures targeted on single species and habitats, as well as other approaches carried out under existing national and international policy and legal frameworks and helps to adapt the management of human activities to the complex and dynamic nature of marine ecosystems. The often limited or incomplete scientific knowledge in marine management requires the application of the precautionary principle, which is seen by OSPAR as being equally a central part of the ecosystem approach.

Following the commitment of the North Sea States at the Fifth International Conference on the Protection of the North Sea in 2002 in Bergen (Norway) to the implementation of the ecosystem approach, the Joint Ministerial Meeting of the Helsinki (HELCOM) and OSPAR Commissions held in 2003 in Bremen (Germany) adopted a Statement on the Ecosystem Approach to the Management of Human Activities “Towards and Ecosystem Approach to the Management of Human Activities”[53]. Under the Statement, the OSPAR Commission is committed to establishing a full set of management measures that are consistent with an ecosystem approach, emphasizing that this is “*crucial to conserve marine biological diversity and its intrinsic value for maintaining life on earth in order to help provide the vital resources for sustainable use to ensure well-being for present and future generations and economic prosperity, to help eradicate poverty, and to help ensure food security.*”

The Joint Statement recognises that the marine environment is both an ecosystem and an interlocking network of ecosystems. It makes reference to the CBD’s definition of an ecosystem as being “a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit”. It stresses that no particular spatial unit of scale is included in this definition, but rather that the scale of analysis and action is to be determined by the problem being addressed.

The Statement recognises that all the components of an ecosystem, including the human component, function together and interact to form an integrated network. Ensuring the integrity of the ecosystems, thereby restoring when practicable and/or maintaining their characteristic structure and functioning, productivity and biological diversity, requires a long-term integrated management of human activities, explicitly:

- managing human activities in order to respect the capacity of ecosystems to fulfil human needs sustainably;
- recognising the values of ecosystems, both in their continuing unimpaired functioning and specifically in meeting those human needs;
- preserving or increasing their capacity to produce the desired benefits in the future.

On this basis, the Statement concludes that the ecosystem approach can therefore be defined as “*the comprehensive integrated management of human activities based on the best available scientific knowledge about the ecosystem and its dynamics, in order to identify and take action on influences which are critical to the health of marine ecosystems, thereby achieving sustainable use of ecosystem goods and services and maintenance of ecosystem integrity*”. The Joint Statement also stresses that the application of the precautionary principle is equally a central part of the ecosystem approach.

6.2.4. The European Union’s Marine Strategy Framework Directive & the Ecosystem Approach

The European Union’s Marine Strategy Framework Directive **(**MSFD)[54] was adopted in June 2008, and is the environmental pillar of the EU’s Integrated Maritime Policy. The aim is “*to protect more effectively the marine environment across Europe*” and “*to achieve good environmental status of the EU's marine waters by 2020 and to protect the resource base upon which marine-related economic and social activities depend*”[55]. On 1 September 2010, the EC adopted a further decision outlining the criteria necessary to achieve good environmental status for Europe's seas[56]. The MSFD “*applies an integrated approach to ecosystems and strives to contain the collective pressure of human activities within sustainable levels*”[57].

It aims to achieve healthy European marine waters by 2020[58].

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[53] “Towards and Ecosystem Approach to the Management of Human Activities”, Statement on the Ecosystem Approach to the Management of Human Activities, adopted by the First Joint Ministerial Meeting of the Helsinki and OSPAR Commissions, Bremen, 25-26 June 2003. (See: <http://www.ospar.org/documents/02-03/JMMC03/SR-E/JMMNNEX05_Ecosystem Approach Statement.doc>, accessed 2010-12-14).

[54] Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive, (2008/56/EC)). Official Journal of the European Union, 25 June 2008, L164/19-40.

[55] “A Marine Strategy Directive to save Europe's seas and oceans” (See: http://ec.europa.eu/environment/water/marine/index_en.htm, accessed 2011-01-04)

[56] COMMISSION DECISION of 1 September 2010 on criteria and methodological standards on good environmental status of marine waters (2010/477/EU). Official Journal of the European Union, 2 September 2010, L232/14-24.

[57] “*Marine ecosystems under the weather: The European Commission’s commitment to cleaner seas and oceans by 2020*” (See: http://ec.europa.eu/environment/water/marine/pdf/leaflet091117.pdf, accessed 2011-01-04).

[58] The steps required are: by 2012, each Member State must “*provide a comprehensive assessment of the state of the environment, identifying the main pressures on their respective marine regions, and defining targets and monitoring*

The MFSD does not explicitly define an ecosystem approach, although reference is made many times in the text of the Directive to an '*ecosystem-based approach*'[59]. However the key concepts of '*ecological status*' and '*good environmental status*' are defined in the MSFD. Here reference is made to the "*overall state of the* [marine] *environment…taking into account the structure, function and processes of the constituent marine ecosystems together with the natural physiographic, geographic, biological, geological and climatic factors, as well as physical, acoustic and chemical conditions, including those resulting from human activities inside or outside the area concerned*"[60]. Good environmental status should "*allow those ecosystems to function fully and to maintain their resilience to human-induced environmental change*"[61]. The focus is not on eliminating all traces of human activity, but rather on making human activities sustainable in the sense that the quality of the ecosystem structure and their functioning are preserved[62].

Each Member State, in co-operation with other Member States and non-EU countries as appropriate, is to develop strategies for their marine waters. These marine strategies must take into account relevant geographical and environmental criteria, contain a detailed assessment of the state of the environment, a definition of "*good environmental status*" at regional level and establish clear environmental targets and monitoring programmes. The subsequent programme of measures and subsequent action should be "*based on an ecosystem-based approach to the management of human activities*" and devised on the basis of the precautionary principle and the principles that preventative action should be taken, that environmental damage should, as a priority, be rectified at source and that the polluter should pay" – i.e. the common principles to be applied by the EU for protection of the environment[63].

In summary, the Directive requires the application of an ecosystem approach, although it is never specifically defined. Fundamental to the Directive, is the notion that ecosystem structure and function should be maintained (good environmental status), their capacity to respond to change should be maintained, even if certain levels of human activity are present (and permitted). This is in the context of humans being able to continue to use marine goods and services in a sustainable way (i.e. not jeopardizing ecosystem structure and function, and good environmental status).

Parts of the MSFD that specifically refer to, or are relevant to, radionuclides/radioactivity

§ 39 mentions Euratom: "Articles 30 and 31 of the Euratom Treaty regulate discharges and emissions resulting from the use of radioactive material and this Directive should therefore not address them". However, Annex III Table 2 (that gives an 'indicative list' of Pressures and impacts) includes under 'Contamination by hazardous substances' the introduction of radionuclides. More generally, pollutants/contaminants are referred to in various places, and are one of the 11 qualitative descriptors for determining good environmental status; (Annex 1, #8: "*Concentrations of contaminants are at levels not giving rise to pollution effects*"). By pollution effects, it is assumed is meant effects on ecosystem structure and function, as outlined above.

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indicators"; by 2015, each Member State will have to develop coherent and coordinated programmes of measures; and by 2020, the target of "good environmental status of the EU's marine waters", Member States will have to achieve efficient communication and close cooperation, notably through the regional sea conventions. Note that this schedule is in line with the objectives of the European Union's Water Framework Directive (2000). The Water Framework Directive includes targets reflecting the ecological integrity of water bodies and emphasizes their ecological status defined as the quality of the structure and functioning of ecosystems. Its timeline requires surface freshwater and ground water bodies to be ecologically sound by 2015 and that the first review of the River Basin Management Plans should take place in 2020.

[59] It has been suggested that the use and understanding of the term is drawn from that outlined by the ICES Working Group on the Ecosystem Approach to Human Activities that gave specific input to the development of the MSFD. The ICES Working Group outlines the concept of the ecosystem approach and writes: "*To provide the greater specificity for the purposes of the European Marine Strategy the Ecosystem Approach could be described as 'a comprehensive integrated management of human activities based on the best available scientific knowledge about the ecosystem and its dynamics, in order to identify and take action on influences which are critical to the health of the marine ecosystems, thereby achieving sustainable use of ecosystem goods and services and maintenance of ecosystem integrity*". "*This description clearly places humans as part of natural ecosystems, and stresses that human activities in these ecosystems must be managed so that they do not compromise ecosystem components that contribute to the structural and functional integrity of the ecosystem*". ICES (2005). Guidance on the Application of the Ecosystem Approach to Management of Human Activities in the European Marine Environment. ICES Cooperative Research Report no 273. 22pp. (emphasis added).

[60] Article 3(4)

[61] Article 3(5a)

[62] The Directive calls for priority to be given to achieving or maintaining good environmental status in the Community's marine environment, to continuing its protection and preservation, and to preventing subsequent deterioration by "*applying an ecosystem-based approach to the management of human activities while enabling a sustainable use of marine goods and services*" (Preamble, paragraph 8, emphasis added). It requires that marine strategies apply an ecosystem-based approach to the management of human activities, ensuring that the collective pressure of such activities is kept within levels compatible with the achievement of good environmental status and that the capacity of marine ecosystems to respond to human-induced changes is not compromised, while enabling the sustainable use of marine goods and services by present and future generations (Article 1(3)). The Directive further states that "*adaptive management on the basis of the ecosystem approach shall be applied with the aim of achieving good environmental status*" (Article 5).Note also that this approach is similar to that set out in the section on Ecological Status in the ICES report, *supra*.

[63] See, Article 174 of the Treaty Establishing the EC (Amsterdam Treaty, 1997) Consolidated Version of the Treaty Establishing the European Community. Official Journal of the European Union, 24 Dec 2002, C 325/33-184.

6.2.5. RAMSAR Convention on Coastal Wetlands

6.2.5.1. RAMSAR and the Ecosystem Approach

Wetlands have become a focus of international conservation activity since the early 1970s, and are the subject of the 1971 'Ramsar Convention'[64]. Initially the primary aim of the treaty was to protect wetlands used by water birds. Subsequently the remit of the Convention has been systematically expanded so that now it encompasses all elements of wetland conservation and to implement Ramsar's "wise use" concept. The wise use of wetlands is defined as "*the maintenance of their ecological character, achieved through the implementation of ecosystem approaches, within the context of sustainable development*".

The Ramsar Convention is of particular importance to coastal areas, where wetlands commonly form the interface between terrestrial and marine environments. Wetlands incorporate some of the most biologically productive and diverse coastal habitats and those most under threat from development and environmental change (Fletcher *et al.*, 2011). Coastal areas can be problematic in governance terms as they are often the focal point for jurisdictional, administrative, and organisational boundaries and are also often focal points for resource exploitation (Fletcher *et al.*, 2011a). In 2002 the Contracting Parties to the Ramsar Convention highlighted the role of an ecosystem approach in coastal wetlands management, encouraging the integration of the conservation and sustainable use of wetlands in broad-scale integrated ecosystem management[65].

6.2.5.2. Management of coastal wetlands in Japan; an example

Japan ratified the Ramsar Convention in October 1980. There are currently 37 Ramsar sites designated in Japan, covering over 130,000 hectares. In Japan, the Ramsar Convention is used as a policy driver at the national level and as a leverage to encourage citizen engagement, economic benefit, and wetland conservation at the local level (Fletcher *et al.*, 2011). In designating wetlands as Ramsar sites, Japan is able to meet the international standards set by the Ramsar Convention and conduct long-term planning for nature conservation through national legislation, such as the Natural Parks Law and Wildlife Protection and Hunting Law. Prior to 2005, most Ramsar designations in Japan covered sites primarily acting as habitat for waterfowl and migratory birds, however since then, sites designated ''have expanded to include wetlands such as marshlands, lakes, salt marshes, tidal flats, sea-grass & seaweed beds, beaches, mangrove forests, and groundwater systems, which reflect the abundance and diversity of Japan's wetlands'' (Ramsar website, Japanese Ministry of the Environment)[66].

Fletcher *et al.* (2011a) present an analysis of a coastal wetland in rural eastern Hokkaido adjacent to Akkeshi Town, which was designated as a Ramsar site in June 1993. This site covers over 5000 hectares, with ownership divided between the National Government, the town of Akkeshi (which owns the largest part) and private owners. The designation of the area as a Ramsar site was driven by the Akkeshi Town Government. Akkeshi town's primary industries are dairy farming and fishing, both of which are reliant upon the maintenance of good water quality and a healthy wetland system. It was therefore considered that Ramsar designation could be used to safeguard the environmental quality of the site to allow these industries to flourish.

Kawabe (2008) has described the progressive development of coastal zone management and protection at Akkeshi. Since the 1990s, the main fisheries in Akkeshi have been the Japanese oyster (*Crassostrea gigas*) and Japanese shortneck clam (*Ruditapes philippinarum*). Akkeshi oysters are recognised as a high quality premium brand throughout Japan. The relationship between the quality of the natural environment and the productivity of the fishery is crucial, and has been accepted by Akkeshi Town government since the 1970s, which has initiated progressive environmental management measures to enhance the quality of the local environment in order to safeguard the viability of the fishery. These measures have evolved from regulation against water pollution (1970s), enhancement of water environmental protection (1980s), watershed conservation (1990s), to comprehensive environmental management (2000 onwards).

Fletcher et al. (2011a) note that the current comprehensive environmental management phase involves significant citizen and stakeholder involvement, which has been encouraged and facilitated by the Akkeshi Town government. A key aspiration of the plan and of the Town's wider efforts has been to further raise the consciousness of the population about the need for a high quality environment to support the fishing industries. In Akkeshi, the Ramsar site designation has explicitly been used by the Town to validate claims regarding the importance of the local environment. This is assisted by an explicit self-interest in maintenance of a high quality environment as it directly contributes to the longevity of the industries upon which the area depends. Akkeshi Town government has encouraged the local fisheries co-operatives to use the Ramsar brand to demonstrate the

[64] The formal name of the original agreement is the "Convention on Wetlands of International Importance especially as Waterfowl Habitat".

[65] Principles and Guidelines for Incorporating Wetland Issues into Integrated Coastal Zone Management (ICZM). Wetlands, Water, Life, and Culture: 8th Meeting of the Conference of the Contracting Parties to the Convention on Wetlands (Ramsar, Iran, 1971) Valencia, Spain, 18–26 November 2002.

[66] Japan Ministry of Environment [online] Ramsar Sites in Japan.
Available from: <http://www.env.go.jp/en/nature/npr/ramsar_wetland/pamph/index.html> (accessed 2011-04-28).

'purity' of the local environment. It is thought that this would add value to fishery and aquaculture produce by demonstrating the high standard of the product, providing national quality assurance for consumers and potentially allowing a premium to be charged for the produce, and to a lesser extent, attracting tourism.

6.2.6. The Convention on Biological Diversity and the ecosystem approach

The Convention on Biological Diversity (CBD) was adopted at the 1992 Rio Earth Summit and is dedicated to promoting sustainable development[67]. Essentially it aims to maintain the world's ecological foundations as a fundamental underpinning of economic development. The Convention establishes three main goals: the conservation of biological diversity, the sustainable use of its components, and the fair and equitable sharing of the benefits from the use of genetic resources. The CBD was the first global agreement to address these three goals in an integrative manner. It also strives to reconcile the development imperatives of the developing countries with the interests of developed countries in accessing and conserving biological diversity.

The CBD requires contracting parties to:

- establish a system of protected areas where special measures need to be taken to conserve biological diversity;
- regulate and manage biological resources important for the conservation of biological
- diversity;
- promote protection of ecosystems, natural habitats and maintenance of viable populations;
- establish or maintain means to regulate, manage or control risks;
- respect, preserve and maintain knowledge, innovations and practices of indigenous and local communities.
- place particular emphasis on certain ecosystems and habitats:
- those containing high diversity, i.e., large numbers of endemic or threatened species of social, economic, cultural or scientific importance:
- those which are representative, unique or associated with key evolutionary or other biological processes; and
- those with specific types of species and communities, e.g. wild relatives of domesticated or cultivated species, or described genomes and genes of social, scientific or economic importance.

The ecosystem approach has been adopted as the primary approach under the CBD and is described in detail in an earlier chapter of this report.

6.2.7. The European Union's Habitats Directive and the ecosystem approach

The Habitats Directive[68] (together with the 'Birds Directive'[69]) is a major component of the European Union's nature conservation policy. It requires that measures be taken to maintain or restore natural habitats of wild flora and fauna of Community interest, to 'favourable conservation status', whilst taking account of economic, social and cultural requirements.

The "Habitats" Directive is a Community legislative instrument in the field of nature conservation that establishes a common framework for the conservation of wild animal and plant species and natural habitats of Community importance; and it provides for the creation of a network of special areas of conservation. The goal is to "*maintain and restore, at favourable conservation status, natural habitats and species of wild fauna and flora of Community interest*". The directive protects over 1000 animals and plant species and over 200 so called "habitat types" (e.g. special types of forests, meadows, wetlands, etc), which are deemed to be of European importance.

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[67] See http://www.cbd.int/

[68] Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora, O.J. L206, 22.07.92

[69] Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds (this is the codified version of Directive 79/409/EEC as amended). This is the EU's oldest piece of nature legislation, creating a comprehensive scheme of protection for all wild bird species naturally occurring in the Union. It was adopted unanimously by the Members States in 1979 as a response to increasing concern about the declines in Europe's wild bird populations resulting from pollution, loss of habitats as well as unsustainable use. It was also in recognition that wild birds, many of which are migratory, are a shared heritage of the Member States and that their effective conservation required international co-operation. The directive recognizes that habitat loss and degradation are the most serious threats to the conservation of wild birds. It therefore places great emphasis on the protection of habitats for endangered as well as migratory species (listed in Annex I), especially through the establishment of a coherent network of Special Protection Areas (SPAs) comprising all the most suitable territories for these species. Since 1994 all SPAs form an integral part of the NATURA 2000 ecological network.

The interpretation of 'ecological coherence' is a key issue. When considering the ecological coherence, it is important to note that the completed Natura 2000 network, defined by the Habitats directive as the sum of all areas designated for conservation under the Birds and Habitats directives (Article 3.1 of the Habitats directive[70]), is a collection of individual protected sites. In order for these protected sites to actually form an ecologically coherent network then necessary functional connections amongst the sites and their surroundings must be maintained. Therefore management measures may need to go beyond the designated sites' boundaries and apply to the wider environment. Consequently it is important to distinguish between the established Natura 2000 network (i.e. all the protected areas) and establishing/maintaining overall ecological coherence of the Natura 2000 network (which includes the necessary functional connections amongst the designated sites).

6.2.8. The Canadian Environmental Protection Act and the Ecosystem Approach

The *Canadian Environmental Protection Act, 1999* (CEPA 1999) is part of Canada's federal environmental legislation and aims to prevent pollution and protect the environment and human health. The goal of CEPA 1999 is to contribute to sustainable development - development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs[71].

CEPA 1999 came into force in early 2000, and implements significant improvements for the protection of the environment over the previous Act. It:

- makes pollution prevention the cornerstone of national efforts to reduce toxic substances in the environment;
- sets out processes to assess the risks to the environment and human health posed by substances in commerce;
- imposes timeframes for managing toxic substances;
- provides a wide range of tools to manage toxic substances, other pollution and wastes;
- ensures the most harmful substances are phased out or not released into the environment in any measurable quantity;
- includes provisions to regulate vehicle, engine and equipment emissions;
- strengthens enforcement of the Act and its regulations;
- encourages greater citizen input into decision-making; and
- allows for more effective cooperation and partnership with other governments and Aboriginal peoples.

CEPA 1999 sets out several guiding principles in the preamble and embodies them in the administrative duties of the government. Prominent amongst these is the **Ecosystem Approach.** For CEPA 1999, the ecosystem approach is one based on natural geographic units rather than political boundaries. In doing so the CEPA 1999 ecosystem approach recognizes the interrelationships between land, air, water, wildlife, and human activities. It also considers environmental, social and economic elements that affect the environment as a whole.

Other key principles set out in CEPA 1999 include:

- **Sustainable Development**. The Government of Canada's environmental protection strategies are driven by a vision of environmentally sustainable economic development. This vision depends on a clean, healthy environment and a strong, healthy economy that meets the needs of the present generation without compromising the ability of future generations to meet their own needs.
- **Pollution Prevention.** CEPA 1999 shifts the focus away from managing pollution after it has been created to preventing pollution. Pollution prevention is "*the use of processes, practices, materials, products, substances or energy that avoid or minimize the creation of pollutants and waste and reduce the overall risk to the environment or human health*."
- **Precautionary Principle.** The government's actions to protect the environment and health are guided by the precautionary principle, which states that "*where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation*."

[70] Article 3.1 of the Habitats Directive states: "*A coherent European ecological network of special areas of conservation shall be set up under the title Natura 2000. This network, composed of sites hosting the natural habitat types listed in Annex I and habitats of the species listed in Annex II, shall enable the natural habitat types and the species' habitats concerned to be maintained or, where appropriate, restored at a favorable conservation status in their natural range*.'

[71] For more information about CEPA 1999, please refer to A Guide to Understanding the *Canadian Environmental Protection Act, 1999* (see <http://www.ec.gc.ca/lcpe-cepa/default.asp?lang=En&n=E00B5BD8-1>, accessed 2011-04-15). To read the complete Act, the full on-line version of CEPA 1999 is available at: <http://www.ec.gc.ca/lcpe-cepa/default.asp?lang=En&n=26A03BFA-1>, accessed 2011-04-15).

- **Science-based Decision-Making.** CEPA 1999 emphasizes the integral role of science and traditional aboriginal knowledge (where available) in decision-making and that social, economic and technical issues are to be considered in the risk management process.

6.2.9. Forest Management and the Ecosystem Approach

The United Nations Conference on Environment and Development (UNCED) in 1992 was a pivotal event for bringing deforestation onto the intergovernmental agenda while also highlighting considerable tensions between developed and developing countries on the issue. Ultimately, UNCED produced two documents directly related to forests: the Non-Legally Binding Authoritative Statement of Principles for a Global Consensus on the Management, Conservation and Sustainable Development of All Types of Forests (known as the "Forest Principles"), and Chapter 11 ("Combating Deforestation") of Agenda 21. The "Forest Principles" included reference to the desirability that forest management should be integrated with management of adjacent areas so as to maintain ecological balance and sustainable productivity.

Negotiations on deforestation were carried beyond UNCED within an increasing array of forest related intergovernmental processes, largely within the forest sector itself. Currently there are a number of international forest governance arrangements which directly address forests, either focusing on Sustainable Forest Management (SFM) or more specific goals, such as biodiversity conservation or climate change mitigation[72].

The 1992 UNCED "Forest Principles" defined a new paradigm for forest management, through a set of 15 principles in support of the overall objective of contributing to the management, conservation and sustainable development of forests and their multiple functions and uses. In this regard, the concept of sustainable forest management (SFM) anticipated the ecosystem approach, both of which are based on the tenet of sustainability. SFM incorporates the following key sustainability concepts: (i) stewardship; (ii) enabling environment regeneration capacity; (iii) continuous flow of goods and services without undermining the resource base; (iv) maintenance of ecosystem functioning and biodiversity; (v) maintenance of economic, social, and cultural functions; (vi) benefit-sharing; and (vii) stakeholder participation in decision-making.

Sustainable forest management (SFM) is the management of forests according to the principles of sustainable development, using very broad social, economic and environmental goals. A definition of SFM was developed by the Ministerial Conference on the Protection of Forests in Europe MCPFE), and has since been adopted by the FAO. It defines sustainable forest management as:

> *"The stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfil, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems".*

In simpler terms, the SFM concept can be described as the attainment of balance between increasing demands for forest products and benefits, and the preservation of forest health and diversity. Sustainable forest management was recognized by parties to the CBD in 2004 to be a concrete means of applying the Ecosystem Approach to forest ecosystems[73]. Specifically the CBD parties noted:

> *"[That] sustainable forest management, as developed within the framework established by the Rio Forest Principles, can be considered as a means of applying the ecosystem approach to forests… Further, there is potential for the tools developed under sustainable forest management to be used to help implement the ecosystem approach. These tools include inter alia the criteria and indicators developed under various regional and international processes, national forest programmes, "model forests" and certification schemes (as relating to decision VI/22 on forest biodiversity). There is substantial potential for mutual learning among those implementing both the ecosystem approach and sustainable forest management."*

The CBD parties further noted that, in addition to sustainable forest management, some other existing approaches, which are also relevant to other environmental conventions, including "ecosystem based management", "integrated river-basin management", "integrated marine and coastal area management", and "responsible fisheries approaches", may be consistent with the application of the CBD's ecosystem approach, and support its implementation in various sectors or biomes.

Annex II of CBD Decision VII/11 elaborates further on SFM and the ecosystem approach. It noted:

> *"SFM can be considered as a means of applying the ecosystem approach to forests. Although the concept of SFM and the ecosystem approach are not identical, the two are similar in many respects. Both need to be applied as an integrated whole. Both are also rapidly evolving. Both*

[72] The following eight policy instruments have been identified as the core components of the current international forest regime complex: Non-legally Binding Instrument on All Types of Forests (NLBI); International Tropical Timber Agreement (ITTA); forest certification schemes; world trade agreements (WTAs); forest law enforcement, governance and trade (FLEGT); Convention on Biological Diversity (CBD); Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES): the climate change regime (including REDD+, etc).

[73] Decision VII/11 of CBD COP7 (2004)

> *have a non-legally binding nature, allowing for flexibility and experimentation. SFM and the ecosystem approach are overarching frameworks--both with due consideration to societal, ecological, and governance issues--although the former has undergone substantial refinement over the last decade, being primarily an outcome-based approach. The ecosystem approach is still in need of further elaboration to be translated into good operational practice in a particular situation. As far as challenges are concerned, both SFM and the ecosystem approach need to deal with complex issues such as law enforcement, land tenure rights, and the rights of indigenous and local communities."*

The CBD identified a need to ensure further integration of the ecosystem approach and sustainable forest management, even though the ecosystem approach and sustainable forest management are broadly overlapping concepts. The CBD parties recognized that SFM is relatively more mature than the ecosystem approach in the sense of being more refined from an operational standpoint; thus it can feed on some aspects of the ecosystem approach. Specifically, the CBD parties noted there was a clear need for the ecosystem approach to adopt processes that are based upon clear statements of visions, objectives, and goals for defined regions or issues, thereby becoming more outcome-oriented (development of the ecosystem approach had emphasized a description of the content of the principles). Conversely, SFM could gain insights from the ecosystem approach concepts as cross-sectoral integration is largely missing from SFM, reflecting restricted legal mandates mostly within forest sector institutions. Thus mechanisms for inter-sectoral collaboration could be strengthened within SFM. Agro-forestry integrates the forest and agriculture sectors but other linkages between the forest sector and the agriculture sector (and other sectors such as water management, transport, and conservation) need to be strengthened.

Although there is no pre-defined scale, the ecosystem approach can be applicable over large areas (landscape level), while SFM has historically emphasized forest management-unit levels of work at typically small spatial scales. Although the Forest Principles do not indicate that forest management should be integrated with management of adjacent areas, and some larger-scale applications (e.g. landscape restoration initiatives and model forests) have been developed within the last decade, greater emphasis could be placed on SFM within a broader spatial context, including protected areas, taking into consideration conservation issues in general, and developing stronger links to adjacent land uses and/or complementary approaches, such as extraction of non-timber forest resources, agriculture, watershed management, and ecological restoration.

The CBD parties noted that there are areas where further conceptual development is needed in both SFM and the ecosystem approach. Both approaches, for example, should explicitly incorporate a principle of sustainability. The inter-generational obligation to sustain the provision of ecosystem goods and services to future generations should be clearly stated. Another area warranting further work is to incorporate issues, in both SFM and the ecosystem approach, of consideration of risks and threats. Global climate change creates risks and uncertainties for all sectors involved in applying the ecosystem approach. Concerns in the forest sector include insecure land tenure, increased forest fire incidence, and the spread of forest pests and diseases into higher latitudes.

In general, the CBD parties noted that the tools and approaches developed to implement SFM may be useful in other productive sectors as they explore ways to implement the ecosystem approach. In particular, the use of criteria and indicators is considered a key tool for implementing and monitoring SFM, and the approach is being applied both nationally and at the forest management unit level. Criteria and indicators can be used for setting goals, assessing management outcomes and policy effectiveness, and for communicating progress to policy makers.

7. Methods Used to Implement, or meant to support, Ecosystem Approaches

A number of different methods are currently in use or under development to implement ecosystems approaches in various areas. Some of them are already in use for environmental management and risk assessment purposes, whilst others are still confined to research developments but meant to ultimately support an ecosystem approach. A few are briefly presented below: mathematical models, various ecological indices, tools for fisheries management and model ecosystem experiments such as microcosms/mesocosms investigations.

7.1. Mathematical Models in ecological risk assessment

Possible methods for evaluation of community - or ecosystem-level - effects are mathematical modelling, model ecosystem experiments and natural field experiments. Among them, mathematical models are practical and integrative tools with a high potential for extrapolation (Galic et al., 2010). That is, they offer excellent tools to translate the output of standard laboratory tests, i.e., single-species ecotoxicity tests, to the higher levels of organisation such as populations and ecosystems.

Under current and the near future situations, however, mathematical model output should always be regarded in a relative sense and no absolute conclusions should be drawn in ecological risk assessment of chemicals (Galic et al., 2010). It seems unrealistic that mathematical models can completely replace mesocosm studies, which are currently the only ecosystem-level tools used routinely in the risk assessment of chemicals but which are time and resources consuming and lack some important ecological processes such as dispersals and re-colonisation. Mathematical models can be used to interpret effects observed in mesocosm experiments, while insights into community- or ecosystem-level effects could also be improved by the further development of mathematical models, using the wealth of information available from mesocosm experiments for hypothesis generation and validation.

In ecological risk assessment of chemicals, the potential of mathematical models has been recognised and has resulted in an ongoing development of various types of models for assessing risks to populations, communities and ecosystems (Galic et al., 2010). Most such models deal with population-level effects, and a small fraction of the models address the higher level of biological organisation, such as food webs, communities, and ecosystems (Table 8). In such mathematical modelling, physical, chemical and biological components of natural ecosystems and interactions among them are mathematically defined, and ecosystems are simulated in computers. For example, dynamics of each compartment are usually represented by a set of differential equations, representing the lack of structure within a compartment, and application is usually accompanied by a sensitivity or uncertainty analysis using Monte Carlo simulation. Effects on entire ecosystems are evaluated by applying single-species effect data to the mathematically constructed ecosystems. Brock et al., (2006) mention the potential use of such modelling studies in the highest tiers of risk assessment of chemicals under the WFD and 91/414/EC directives, following standard single-species tests as the first, species sensitivity distribution as the second and the model ecosystem approach as third tier. Only in the third and higher tiers were modelling studies considered as tools for risk assessment refinement.

Table 8. Community - or ecosystem-level - mathematical models (Galic et al., 2010)

Model Name	Reference
Food Web Bioaccumulation Model for Organic Chemicals in Aquatic Ecosystems	Arnot and Gobas, 2004
Deriving Water Quality Criteria	De Laender, 2007
EcoWin	Ferreira, 1995
LERAM	Hanratty and Stay, 1994
CASM	Natio *et al.*, 2002
SWACOM	O'Neill *et al.*, 1982
AQUATOX	Park *et al.*, 2005
C-COSM	Traas *et al.*, 2004
CATS	Traas and Aldenberg, 1992
Recovery of macro-invertebrates following a pulse disturbance in a river	Wantanabe *et al.*, 2005

Further to mathematical models for predicting effects at community or ecosystem level, there is also today a move towards exploiting Multi-Criteria Analysis to implement ecosystem approaches (Chen et al., 2011), such as to take account of the fact that man is part of the ecosystem and that there is a need to consider ecosystem services.

7.2. Implementing an Ecosystem Approach in Fisheries Management

Fisheries management has seen a gradual shift from a primary focus on target species and resources to a much wider focus on ecosystems, and the impacts of fisheries on them. This led to the development of the ecosystem approach to fisheries (EAF). There are many examples of implementing the ecosystem approach to fisheries, with numerous developments currently taking place internationally in this field. The paragraphs below summarises information presented by Smith *et al* (2007) on the example of implementation of the ecosystem approach to fisheries (EAF) at the Australian federal level of fisheries management jurisdiction.

7.2.1. Example in Australia

Australia began implementing an ecosystem approach to fisheries in response to a number of policy directions and initiatives. These included: (i) a national, government-wide approach to ecologically sustainable development, released in 1991; (ii) development of fisheries legislation that incorporates explicit reference to wider ecological impacts of fishing (e.g. the Fisheries Management Act 1991); (iii) new environmental legislation that assesses fisheries against environmental standards (e.g. the Environmental Protection and Biodiversity Conservation Act 1999); and (iv) Australia's Oceans Policy, which adopts an explicit ecosystem-based approach to management, with explicit requirements for regional ocean planning for all uses and users of the marine environment. More recently, the Australian Fishery Managers Forum (incorporating heads of all federal, state, and territory fishery management agencies) formally adopted the ecosystem approach to fisheries as the approach to future management.

The main elements of EAF include:

- avoiding the degradation of ecosystems;
- minimizing the risk of irreversible change;
- obtaining long-term socio-economic benefits from fishing; and
- adopting a precautionary approach to uncertainty.

7.2.2. Tools development

The adoption of the ecosystem approach to fisheries has increased the scope of fisheries management to encompass the broadened ecological focus, as well as a range of socio-economic concerns. A range of tools may be needed to support these different elements. The various tools can be seen as supporting different elements in the adaptive management cycle that characterizes fisheries management. The key steps in the cycle requiring scientific support include monitoring, assessment, and decision-making. In addition, the evaluation of the entire management cycle via management strategy evaluation (MSE) requires scientific input.

The scientific tools may adopt a variety of approaches or methods. They may be broadly classified on a spectrum from qualitative to quantitative methods, although in practice the distinctions are not always clear cut. In particular, the methods may be based on expert judgment (an informal or formal, qualitative method that reflects the predominant opinion within a group of well-informed people); they may be largely empirical; or they may be based on quantitative models, although these categories are not exhaustive.

Figure 14 illustrates the different aspects in a schematic framework for tool development. The elements of the adaptive management cycle are listed along the tools axis. The focus of the tool is on the scope axis, and the approach to be used is on the method axis. For example, traditional quantitative stock assessment would be found at the node consisting of population (scope), assessment (tool), and model (method), whereas the MSE for a single-species harvest strategy would be at the population, MSE, model node. Some tools may span multiple elements in a single axis.

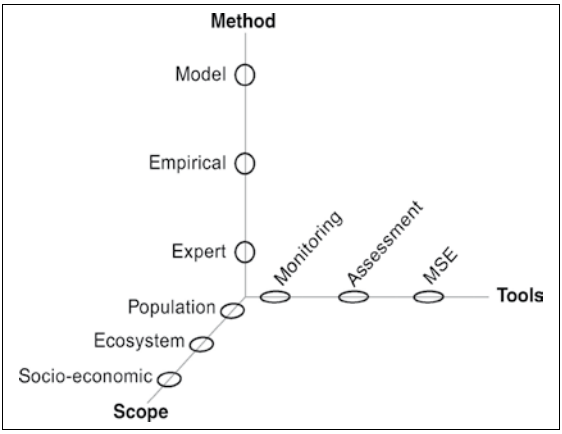

Figure 14: Framework for the ecosystem approach to fisheries tool development.

7.2.3. A hierarchical approach to ecological risk assessment for EAF

Considerable work has gone into developing ecological risk assessment methods for fisheries during the past 5 years. In general, the methods have been based either on expert judgment, or empirical.

Typically a hierarchical framework for ecological assessment for effects of fishing is used within EAF, involving a scoping stage and then up to three levels of assessment (Hobday *et al*, 2006). This spans expert-based (Level 1), through semi-quantitative or empirical (Level 2), to fully quantitative methods (Level 3), with explicit links between them (Figure 15).

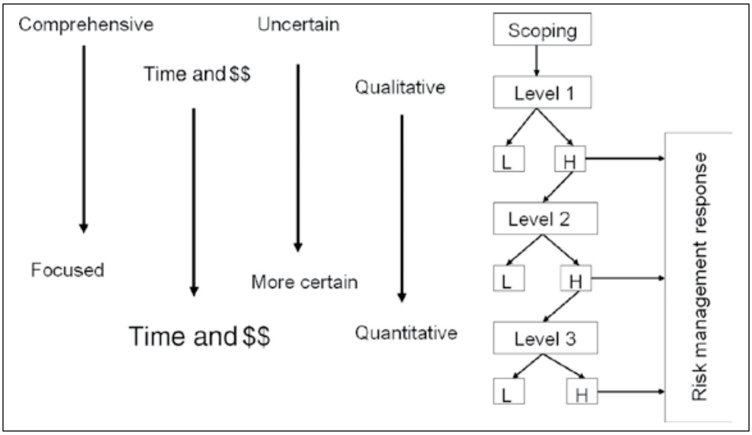

Figure 15: Hierarchical structure of ERA for effects of fishing framework illustrating the three levels of assessment. Qualifiers to the left indicate differences in specificity, costs, certainty, and quantification of the assessment (L, H: low and high risks, respectively; Smith et al, 2007)

This assessment is based on an exposure–effects approach, rather than a likelihood–consequence approach, because most fishing activities are common and deliberate rather than rare and accidental. At any stage of the assessment, identification of low (L) and high (H) risk will result in a move to a higher level of risk assessment or leads to a risk management response. The different levels of assessment provide a series of filters to screen out low risks efficiently, with the assessment extending to the next level only if the risk is judged to be above a determined threshold. A cost-effective alternative to moving to the next level of analysis may be to instigate risk-management actions to mitigate the identified risk.

The scoping stage includes four aspects:

- fishery description,
- detailed objectives,
- list of activities (potential hazards), and
- identification of units of analysis (lists of species, habitats, and communities).

The fishery description includes identification of sub-fisheries or fleets (mainly designated by fishing method), historical development, current status, and current management arrangements. This step identifies information available to support assessment at the subsequent levels. The hazards relevant to the specific fleet are identified from a specified list of activities associated with fishing, as well as six external activities that could also have impacts on the ecological system. Impacts are assessed against five ecological components representing the ecosystem: target species; by-product and by-catch species; threatened, endangered, and protected species; habitats; and communities. For each component, the relevant units of analysis are identified, comprising either a list of species, habitats, or communities. Benthic habitats are classified based on geomorphology, sediment, and faunal cover, most often using photographic images. Communities are classified using nationally agreed bioregions and biotic provinces, combined with depth classification. Depending on the nature and scale of the fishery under assessment, the units of analysis may be hundreds of species and habitat types, and tens of community types.

The Level 1 assessment uses a scale, intensity, consequence analysis (SICA) method that involves assessing the impact of each activity on each component using expert judgement and a scale from negligible to catastrophic. The potential amount of analysis required at this qualitative level is limited by taking a “plausible worst case” approach that selects the unit of analysis identified by stakeholders (including fishers, managers, environmental agencies, and NGOs) to be most vulnerable to each activity. The maximum number of scenarios required is 160 (32 activities by five components). Each scenario is carefully documented, and only activity/component combinations (hazards) for which the risk score is >2 (moderate or above) are assessed at the next level. In practice, most hazards are eliminated at Level 1. In some cases, entire components are assessed to present low risk (e.g. habitats for pelagic long-line fisheries), and excluded from further assessment at higher, more costly levels.

Level 2 assessments are based on a productivity–susceptibility analysis (PSA) approach. All units of analysis are assessed for any component not screened out at Level 1. Given that there may be a large number of potential by-catch species for a given target species (e.g. up to 500 by-catch species in a tropical prawn fishery), an efficient screening process is required. This is achieved by compiling a list of attributes for each unit of analysis that bear either on productivity (ability of the unit to recover from impact; ≈ resilience) or susceptibility (exposure of the unit to impact; ≈ vulnerability). The PSA approach has also been developed to assess habitats, but the Level 2 assessments for communities are still under development.

No new methods have been developed for Level 3 assessments, because existing methods were considered suitable. These include quantitative stock assessment for target and by-product (retained, non-target) species, population viability analysis for threatened, endangered, and protected species, and methods for community analyses.

ERA for effects of fishing also has an explicit and formal treatment of uncertainty. As already noted, the Level 1 assessment is based on a “plausible worst case” treatment of impact, and the Level 2 assessment uses a default scoring (in the absence of better information) that leads to a high risk. Therefore, the method provides the correct incentive to acquire better information to reduce risk.

Overall, the hierarchical approach leads to a cost-effective means of screening hazards, Level 1 allowing for rapid assessment with minimal information requirements, and successive levels requiring more time, resources, and information, but only being called upon if needed (Figure 16).

This approach has been applied recently to all fisheries currently managed by the Australian Fisheries Management Authority (AFMA), amounting to >30 sub-fisheries. This has demonstrated its utility and flexibility, and the consistency of the approach is leading to improved selection of research priorities, both within and between fisheries. To date, >1800 species have been screened using the PSA method, together with several hundred habitat types. AFMA is in the process of developing sets of risk management responses to the results, and intends to build ERAEF into an ongoing adaptive environmental management system with reassessment at intervals of 3–5 years.

7.2.4. The need for early and on-going dialogue to facilitate implementation the ecosystem approach to fisheries

Although considerable progress has been made in developing and applying a range of scientific tools in support of EAF, not all problems have been addressed or solved. The tools described are still evolving, not least in response to problems arising during their application. The expectation is that the tools, and others currently in development, will continue to evolve as they face the test of application in real fisheries, although this process will take time.

This problem can be traced back to how the EAF was introduced. Although fishery scientists played important roles in the policy developments leading towards introduction of the EAF in Australia, in almost all respects the policy has been running ahead of the development of the knowledge and scientific tools to support its implementation. As a consequence, the scientists had to “catch up” with the policy, and spent a great deal of effort in developing the tools, which were usually implemented immediately. This rapid development has had both benefits (e.g. considerable progress in the practical implementation of EAF approaches) and costs (e.g. diminished quality in initial results, the need to work with multiple versions of the tools, and overall delays).

Clearly ensuring early and on-going consultation between scientists, managers (operators and regulators) and other relevant stakeholders throughout the process from inception to implementation would be highly desirable. This should be a two-way process, clarifying expectations and identifying needs and practical constraints at each stage of the process.

7.3. Attempts to Identify Ecological Indices

To measure the ecological status of an ecosystem is a real challenge. Theoretically, it can be described by a mathematical vector whose components give the number of individuals (or the biomass, or the biomass density) of each living species. Once the local equilibrium status of the ecological system of interest is known, differences from the optimal values of the components of the population vector indicate the deterioration of the ecosystem.

This is, however, rather impracticable in view of the realistic achievable level of our knowledge of any ecosystem. Moreover, within certain limits, alterations of the sizes of species populations, in principle, does not necessarily involve functional failures of a complex ecosystem.

Thus, much simpler alternative options should be considered. An example of index to evaluate the ecosystem condition is the entropy H which is defined, in analogy with the corresponding concept in statistical mechanics, as follows:

$$H = -\sum p_i \, ln p_i \tag{1}$$

where p_i is the proportion of individuals of each species. Although the entropy H can be assumed as a measure of the biodiversity, definition (1) can only be used in practice if the analysis is limited to a selected number of species or communities.

7.3.1. Lake Ecosystem Index

Håkanson and Peters (1995) suggested the definition of the so called Lake Ecosystem Index (LEI) to measure the health status of a lacustrine ecosystem showing that such a concept can be successfully applied to evaluate, in an ecological perspective, the effectiveness of countermeasures aimed at reducing the impact of radiocaesium on the environment and on the human population (Håkanson et al., 2000).

The main idea in defining the LEI is to account for the amount of the most important ecological functional groups in a lake: fish, phytoplankton and benthic fauna. The fish yield (FYR), the phytoplankton biomass (PBR) and the benthic fauna biomass (BFBR) ratios are defined and a possible definition of LEI would be the average of these values:

$$LEI = \frac{FYR + PBR + BFBR}{3} \tag{2}$$

The optimal value of LEI would be 1 (unperturbed conditions). Any departure from 1 would always be considered as negative from an ecological point of view.

7.3.2. Ecological effect index for microcosms

Microcosms are experimental, very simplified, attempts to reconstruct nature which still promote debate as to their representativeness of nature. Considering them as theoretical models however, an ecological effect index (EEI) was proposed for holistic evaluation of effects on various parameters in microcosms exposed to ionising radiation and other stressors (Fuma et al., 2003a). For quantitative estimation of the EEI, degrees of differences in parameter values between exposed and control ecosystems were expressed as the Euclidean distance weighted by ecological importance of each parameter. That is, the effect index on day t after exposure (EI(t)) was defined as follows:

$$EI(t) = 100 \sqrt{\sum_{i=1}^{n} W_i \left\{ \frac{P_{i,Con}(t) - P_{i,Exp}(t)}{P_{i,Con}(t)} \right\}^2} \ [\%] \tag{3}$$

Where,

n = the number of parameters concerned.
Wi = ecological weighting factors for parameter i. This value depends on importance of each parameter for the ecosystem itself in the ecocentric principle and for man in the anthropocentric principle.

The more important the parameter i is, the larger the Wi value is. $\sum_{i=1}^{n} W_i = 1$

$P_{i,Con}(t)$ = values of parameter i in the control ecosystem on day t.

$P_{i,Exp}(t)$ = values of parameter i in the ecosystem exposed to a toxic agent on day t after the exposure.

The EI(t) should be averaged for experimental periods, because the time-dependent changes in parameters are investigated for evaluation of ecological effects in general. As such an averaged index, the ecological effect index (EEI) was defined as follows:

$$\mathrm{EEI} = \frac{1}{T}\int_0^T \mathrm{EI}(t)\,dt\ [\%] \qquad (4)$$

Where,

T = days from the exposure to the end of the observation.

The EEI was applied to population changes in the three-species and multi-species microcosms exposed to acute γ-rays. That is, the effect index for microcosm (EIM) was defined as follows:

$$\mathrm{EIM} = \frac{100}{T}\int_0^T \sqrt{\sum_{k=1}^{n} W_k \left\{\frac{N_{k,\mathrm{Con}}(t) - N_{k,\mathrm{Exp}}(t)}{N_{k,\mathrm{Con}}(t)}\right\}^2}\, dt\ [\%] \qquad (5)$$

Where,

$N_{k,Con}(t)$ = the log-transformed population of taxon k in the control microcosm on day t.

$N_{k,Exp}(t)$ = the log-transformed population of taxon k in the exposed microcosm on day t.

Assuming that all the trophic levels, i.e., producers, consumers and decomposers, had the same ecological importance and all species or taxonomic groups in each trophic level had the same ecological importance in the microcosm, the following W_k values were given to each species or taxonomic group in equation (5):

Three-species microcosm:

E. gracilis, *T. thermophila* and *E. coli*: $\frac{1}{3}$

Multi-species microcosm:

[Producers] *Chlorella sp., Scenedesmus sp. and Tolypothrix sp.*: $\frac{1}{3} \times \frac{1}{3} = \frac{1}{9}$

[Consumers] *C. glaucoma, Lecane sp.ihilodina sp.* and *A. hemprichi* : $\frac{1}{3} \times \frac{1}{4} = \frac{1}{12}$

[Decomposers] Bacteria : $\frac{1}{3}$

In general, the EIM was positively correlated with log-transformed doses of each stressor, and the relationship between them could be fitted by a sigmoid curve (e.g., Figure 16). A 50 % effect dose for the microcosm (ED_{M50}, not to be meant as an ecosystem safe level, however) at which the EIM became 50 % could be calculated in a similar manner as a 50 % effect concentration (EC_{50}) which is regarded as one important toxicity data in conventional single-species tests for chemicals.

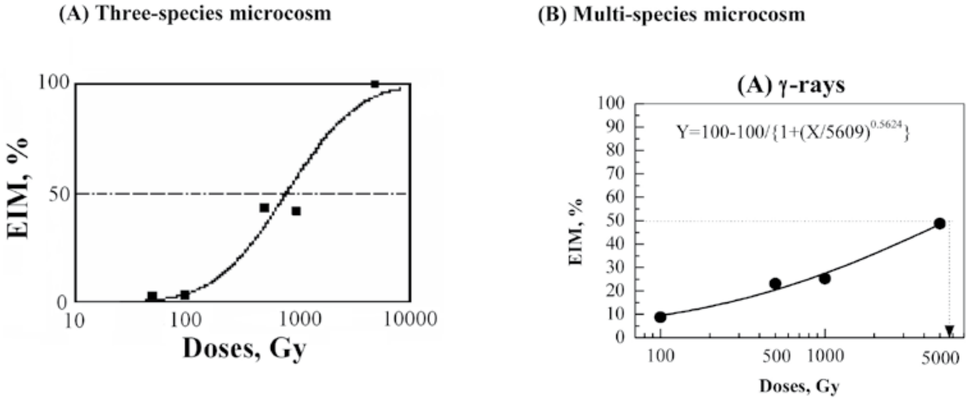

Figure 16. Dose-EIM relationship in the γ-irradiated three-species and multi-species microcosms (Fuma et al., 2003a, 2009). EIM: Effect index for microcosm.

7.4. Microcosm and Mesocosm Investigations

In ecological effects evaluation of ionising radiation, comparative studies have been made between a laboratory microcosm (a very simple experimental simulation of an ecosystem) and a mathematical model. Fuma et al., (1998) studied effects of acute gamma irradiation on the aquatic microcosm consisting of "populations" of the flagellate alga *Euglena gracilis* as a producer, the ciliate protozoan *Tetrahymena thermophila* as a consumer and the bacterium *Escherichia coli* as a decomposer. After 1000 Gy irradiation, the cell density of *T. thermophila* was increased temporarily, and then decreased compared with controls. This complicated change in *T. thermophila* populations might be an indirect response to direct effects on the other species, i.e., extinction of *E. coli* and decrease in *E. gracilis*. Doi et al., (2005) mathematically simulated a dose-effect relationship of this microcosm with an individual-based model, i.e., SIM-COSM, in which interspecies interactions were taken into consideration.

Short-term, single-species, ultraviolet-B irradiation of a benthic diatom population demonstrated reduced photosynthesis and growth, but long-term, multi-species level, tests demonstrated an increased standing crop of diatoms because the radiation also inhibited chironomids larvae, which are diatom consumers (Bothwell et al., 1994). Ionising radiation also collapses the ecosystem balance via trophic changes indirectly mediated by radiation effects, through differences in species sensitivities. For example, in a simplified model ecosystem (microcosm) consisting of populations of an algal producer (*E. gracilis, T. thermophila* and *E. Coli*, described in the previous paragraph), γ-irradiation indirectly decreased *Tetrahymena* populations by eradicating *E. coli,* on which *Tetrahymena* grazed as staple food (Doi et al., 2000, 2005; Doi and Kawaguchi, 2007; Fuma et al., 1998). After a microcosm consisting of eight taxa of aquatic microbes was irradiated with γ-rays, population increases or dose-independent population changes were observed in a blue-green alga, a protozoan, rotifers and some species of bacteria. They were likely indirect effects due to reductions of interspecies competition and grazing pressure (Fuma et al., 2010).

Mc Namara et al. (2007) γ-irradiated forest soil cores, and demonstrated that the irradiation decreased fungal abundance while it increased bacterial abundance. This induction of higher bacterial carrying capacities could be due to greater substrate availability, which likely occurred from radiation damage to organisms and soil organic matters, and less competition for the substrates between the bacteria and fungal populations. They also showed that the irradiation increased certain bacterial species, some of which were fast growing competitive species and thus could rapidly colonise the soils based on higher available resources after irradiation. These results indicate that predation and competition are ecosystem-level processes which, through balancing inter-population relationships, act on ecosystem structure and resilience.

Jones et al., (2004) γ-irradiated a soil-grass mesocosm, and detected alterations of soil microbial communities such as biomass increase. They suggested that many of these microbial responses were indirect effects mediated by the radiation effects on the plants, e.g., alteration of rhizosphere processes.

Ishii et al., (manuscript in submission) chronically γ-irradiated a flooded paddy soil microcosm at a dose rate of 1 $Gy.d^{-1}$ for five days, and found that bacterial species of the genus *Clostridium* and *Massilia* were more abundant in the irradiated microcosm than those of controls. There is a possibility that these responses were indirect effects, and this increase in the *Clostridium* and *Massilia* species might cause sulphate concentration increase in the liquid phase of the irradiated microcosm, which was detected as a radiation effect on ecological functioning, i.e., nutrient cycling (Ishii et al., 2007).

8. Research and Development needed to support the ecosystem approach

From the general overview and analysis made above on how does the current "reference organism" based approach developed for radiation protection fits within the overall context of environment protection, it is necessary to identify the Research & Development needs to support the development of an ecosystem approach. These are developed below along three lines of research: those focusing on impacts at the ecosystem level (top-down), those focusing on ecologically relevant effects on individual organisms (bottom-up), and those exploiting cross cutting studies such as field investigations *in situ*.

8.1. Ecosystem level issues

Many of the knowledge gaps and research needs at the ecosystem level will be generic to any type of stressor. Therefore, research directions recommended here to improve our understanding of radiation impacts will apply to other stressors as well. They collectively call for specific experimental approaches and *in situ* observations, concept development and modelling undertakings at ecosystem level. They include the following:

- Better knowledge on the interactions and dependencies between populations, such as predator-prey for example, and how their relative dynamic equilibrium may be influenced by a stressor.
- Modelling to understand the sensitivity of ecosystems to population changes – what impact will changes in populations have on other species and the ecosystem as a whole? Which species are "vulnerable"; which are "robust" and which are "keystone species". This will in turn help direct research needs on toxicological knowledge needed on individual species.
- Development of ecologically relevant models - incorporation of seasonal variation, migration, non-linear responses, energy budgets, etc.
- Development of methods and associated indices for assessing ecological status - e.g., methods for assessing biological and genetic diversity and ecosystem functions.
- Development of population dynamic modelling to better understand how population level effects are mediated by life-history traits in individuals, and more generally how effects mediated at lower levels of organisation propagate at higher levels.
- Development of a better knowledge of ecosystem resilience, such as masking potential deleterious radiation effects occurring at lower level of organisation, or triggering abrupt regime shifts beyond certain thresholds of stress (tipping points), therefore promoting delayed effects.

One essential feature of all these studies emphasizes a requirement for collaboration with and between modellers, ecologists, systems ecologists and ecotoxicologists. It is therefore recommended that radioecology evolves with a spirit of openness by following research developments within such fields, but also by participating in inter-disciplinary research, especially outside the nuclear world.

8.2. Reference Organism (or reference species) enhancements

The need for improved understanding of the impacts of radiation at the level of individual species is shared by both the ecosystem approach and classical toxicology followed by the reference organism approach, and thus, a number of the research needs are shared by both approaches. However, the research focus and research questions can be framed by the need to better understand the ultimate impacts at an ecosystem level. These include the following:

- Knowledge on effects on species from different ecosystem functional groups (e.g., decomposers, primary and secondary producers, consumers/predators). The various approaches based upon reference organisms are often lacking in important groups, such as fungi, bacteria and predators, hence the databases and dose-effect compilations elaborated to support assessments are missing important groups of species. A better representation of taxa from different geographical regions to supplement the current dominance of data from northern temperate systems is also warranted.
- Understanding of differences in radio-sensitivity among taxa/functional groups. Ecosystem levels effects will depend strongly on differences in sensitivity between species. This is particularly relevant to the lower functional groups (decomposers and primary producers), which are known to have the largest variability in radio-sensitivity.
- Chronic exposure studies at low doses for whole life durations, and their long-term impact on the fitness of populations. These include in particular such aspects as life-history traits of individual organisms, genetic/cytogenetic/epigenetic effects and non-targeted effects such as immune response and genomic instability, and also aspects of hormesis, adaptation and acclimatisation. It is to be noted that some of such effects might be non-linear.

- Studies of impacts from multiple stressors to evaluate whether integrated responses will turn out to be additive, synergistic or antagonistic, when assessing risk.
- Internal emitters and inhomogeneous dosimetry. Particularly for alpha and auger emitters, the internal distribution of radionuclides has important implications for dose and organ level response. Knowledge is needed on both the internal distribution and metabolism of radionuclides, as well as the biological impacts of inhomogeneous dose distribution (RBE, micro-dosimetry).

8.3. Field Experimentation

Field studies are needed to calibrate laboratory studies from both the systems and organism level. Field studies in areas contaminated from former accidents or mining activities, for example, offer excellent opportunities to study the impacts of stressors across different hierarchical levels. It is therefore recommended to develop consistent field approaches for Chernobyl, Fukushima, and mine sites, featuring gradient analyses study designs and using the reference organisms to move toward the ecosystem approach. In this context, it is worth strengthening that designing *in situ* investigations on contaminated areas based upon ANOVA approaches with reference or control sites is to be precluded. Some argue that there cannot be a reference site in any ecological context (see Landis et al., 2011), but more generally, this is because hypothesis testing is not the goal.

For *in situ* experimentation again, the collaboration of radioecologists with landscape ecologists and systems modellers in the one hand, and molecular biologists, geneticists, etc., in the other hand, is to be strongly encouraged.

9. Recommendations with respect to radiation protection

When setting out in a new direction, the concerned scientific community quite naturally seeks guidance on how to proceed in practice. Drawing from the arguments and the research priorities developed above, this section presents some major recommendations with respect to radiation protection of the environment. The recommendations are particularly addressed to both the radioecology scientific community, which is tackling the scientific foundation, and to the ICRP, which aims to translate the scientific knowledge into operational terms that could be used for regulatory purposes.

Recognizing that the ecosystem approach has been adopted in an increasing number of other situations to deal with environment protection, it is appropriate for radiation protection to move in this direction in order to improve the relevance and coherence of information coming to decision-makers. To that end, the following points should be considered:

- **Promote the dialogue between environmental assessors and environmental managers** (facility operators, contaminated site managers, and regulators) to facilitate improving the value of information flow between them (a two-way dialogue). In the methodology of ecological risk assessment, it is often stressed that the management phase should be separated clearly from the assessment phase because assessment does not involve decision making, but rather only a quantitative and robust rating of risk. However, if the separation hinders necessary dialogue between both parties, then the methodologies of risk assessment may not respond to the actual needs and expectation of risk managers. This does not mean jeopardizing the transparency of roles would a risk assessor being asked to become a risk advisor. Both roles need to be clearly distinguished. Here, it may be observed that the reference organism method, emphasising individual organisms, largely misses the objectives of protection set by risk managers at population and ecosystem levels.
- **Develop more integrated and functional endpoints to expand beyond the organism level**. There is a need to identify endpoints relevant to population and ecosystem levels in order to reduce uncertainties necessarily driven by extrapolations when attempting to reach protection objectives set at such levels. A few examples have been presented in this report with respect to radiation protection, but this could be expanded to include consideration of additional indices that embed existing and new endpoints (such as decomposition rate, rate of primary productivity, rate of energy cycling, etc).
- **Incorporate more ecological contextualisation in the reference organism approach.** The reference organism approach is an efficient tool to take account of the toxicological impacts of radiation upon individual organisms. It can be improved in the direction of the ecosystem approach by considering more such aspects as ecological functionalities (predators, consumers, producers, etc.), other ecological criteria (key-stone species, trophically-related species, etc.), and reference species versus reference organisms. An asset would be to be able to arrange sets of reference organisms which, when assembled, would resemble a model ecosystem, albeit a necessarily simplified one. It is also recommended to pay more attention to taxonomy. Insects are currently poorly represented whilst forming the widest collection of different species in wildlife, and playing important ecological roles, such as pollination, for example. Bacteria and fungi, major ecological players in the biosphere as decomposers, are also missing, which constitutes a major drawback of the current approach. It is therefore recommended to include more ecological functionalities in order to make the approach more accessible to people within different geographical areas and biomes.
- **Promote overall consistency across the broad spectrum of ecological research and environmental management** (see Field studies and cross-cutting research needs, chapter 8). The goal here is to ensure that information can be leveraged from multiple efforts outside of the radioecology fields. In particular there should be efforts to coordinate work on radiation and chemicals, as well as various other stressors. Also, efforts should be devoted to establish a continuum from very analytical investigations (microscopic organism level), through microcosm/mesocosm experimentations, to field observations and testing, and modelling, by attempting to include consideration of the same sets of species at the various possible degrees of assembly. While recognising the goal of improving environmental protection per se, this approach also should encourage advancing coordination of radio-protection approaches for human and ecological receptors – considering humans to be part of the ecological system.
- **Avoiding the misleading use of NOAEC/LOAEC metrics.** Another more detailed but important recommendation from the task group is to emphasize that various metrics of different qualities have been traditionally used to quantify effect levels (from toxicants). Among these, only "Effects Level" estimates (i.e., EC_{20}, EC_{50}) are scientifically sound and should be used. The "no observed adverse effect concentration/level" values are still employed in some published works and studies (as reported above), but such metrics are not recommended as they do not innately relate to biologically-relevant thresholds and therefore do not provide information about the actual magnitude of effects in the reported studies. Moreover, NOAECs do not necessarily equate to a "no effect" dose and may be misleading. They reflect only the test concentrations used in an individual study and are strongly influenced by factors related to statistical power (e.g. study

design, replication). The use of NOAECs and LOAECs (or equivalent terms) in Ecological Risk Assessment has been criticized widely (Bailer and Oris, 1997; Chapman et al., 1996; Hoeskstra and van Ewijk, 1993; Laskowski, 1995; OECD, 1998), yet these terms are routinely resurfacing, partly due to regulatory bodies being reluctant to update their thinking, and partly because authors, reviewers, and editors continue to be seduced by the idea that the concept of “no effect” means something. In reality, it does not, and one should remember the dangers of subjecting science to policy in this way (Kapustka, 2008). One should not forget in this context that approaches based upon weight of evidence often give quite reliable results.

10. References

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ANNEX

An Integrated approach proposed for comparative risk assessment between ionizing radiation and other stressors

The conceptual model was originally proposed for ecological risk assessment of ionising radiation (Polikarpov, 1977, 1998a, see Figure 17).

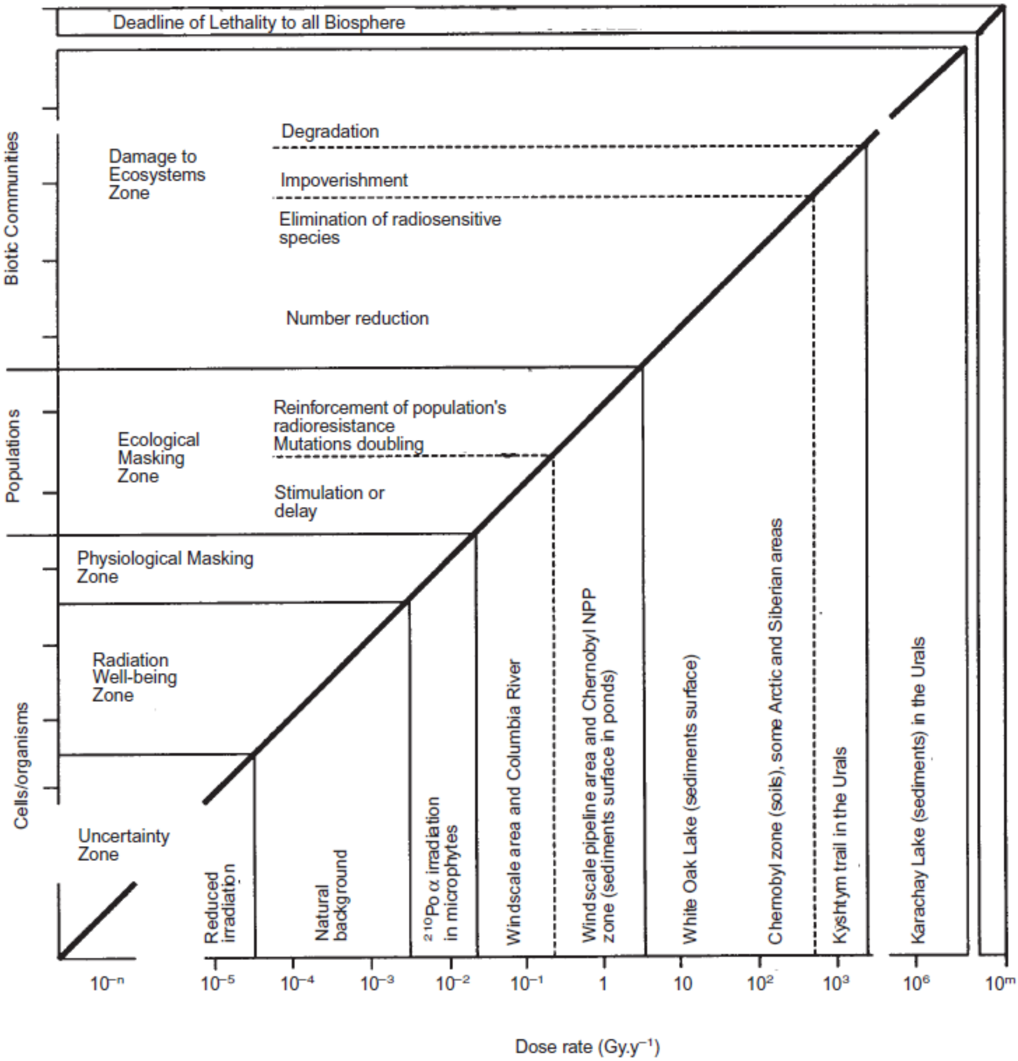


Figure 17. Zones of dose rates and their effects in the biosphere (Polikarpov, 1998a). A value expressed as a ^{210}Po alpha-irradiation dose rate was calculated multiplying an absorbed dose rate by a radiation weighting factor of 20.

This model categorises responses of organisms, populations and ecosystems to all the existing and possible dose rates in the environment as follows:

- the uncertainty zone (below the lowest natural ionising radiation background level),
- the radiation well-being zone (natural ionising radiation background range),
- the physiological masking zone (0.005-0.1 Gy.y^{-1}),
- the ecological masking zone (0.1-0.4 Gy.y^{-1}),

- the zone of damage to communities/ecosystems (>>0.4 $Gy.y^{-1}$),
- the radiation threshold for lethality to the biosphere (>>$MGy.y^{-1}$).

This model has been later extended and transformed into the "radio-chemo-ecological" conceptual model, which covers effects of both ionising radiation and chemical pollutants (Figure 18, Polikarpov, 1998b, 1999, 2001). This model was used for comparative risk assessment between radioactive and chemical pollutants in the Black Sea (Polikarpov, 1999; Polikarpov et al., 2004). Some severe impacts have been observed in various organisms such as macroalgae, mussels, fish, mammals in the Black Sea shelf and estuarine ecosystems.

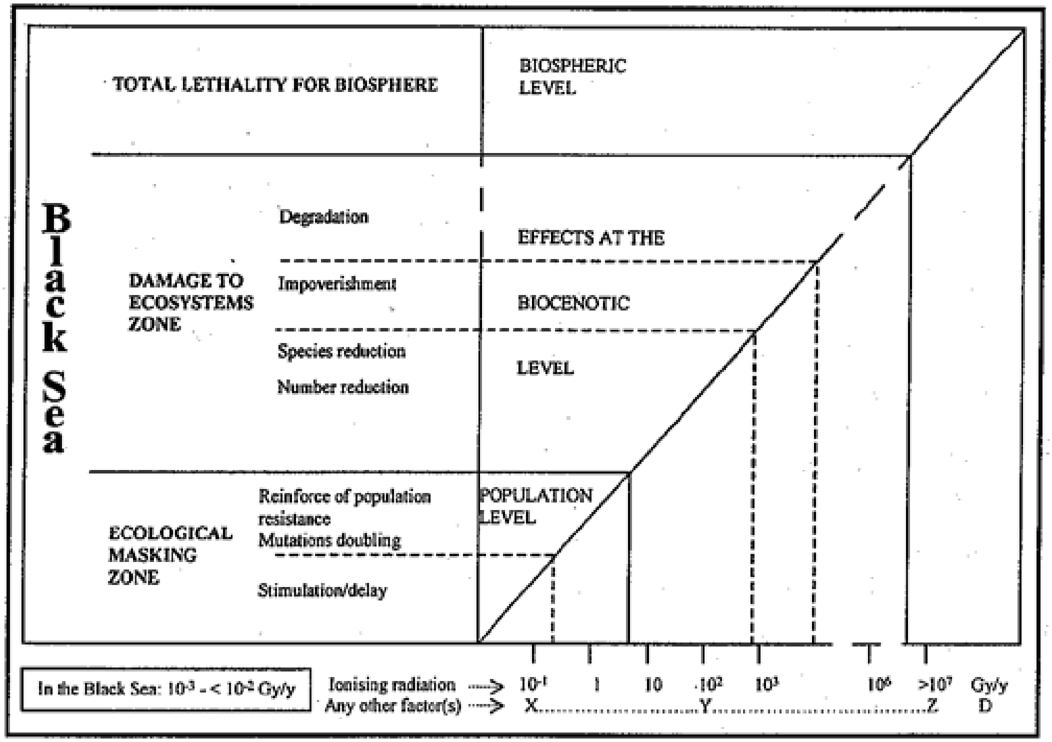
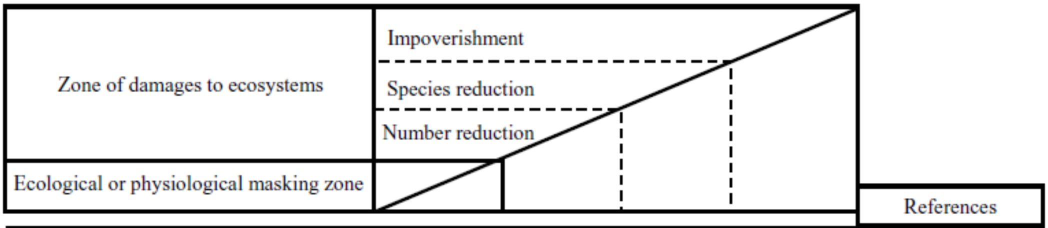


Figure 18. Long-term reactions of populations/ecosystems to ionising radiation as well as any chemical or physical factors at doses within "masking" and "damage" zones (Polikarpov et al., 2004). X, Y and Z are doses of non-radioactive factors in each zone. Ecological impacts in the Black Sea are due to non-nuclear factors because of low dose of ionising radiation.

According to this model, such chemical impacts are quantitatively equivalent to the ecological effects of nuclear pollution in the nearest zone of the Chernobyl nuclear power plant, the Kyshtym trail, the Karachay Lake and other similar areas in the world contaminated with radionuclides. Dose rates of ionising radiation to biota are, however, less than 7 $mGy.y^{-1}$ in the Black Sea, which is by three to seven orders of magnitude lower than chronic lethal doses for these organisms. The results (Polikarpov et al., 1994, 2004) indicate that the severe impacts in the Black Sea were caused by non-nuclear pollution.

This model has also proved to be useful for the purpose of comparative evaluation of effects of ionising radiation and various other stressors on the aquatic microbial microcosms. The first one was the three-species microcosm consisting of populations of the flagellate alga *Euglena gracilis* as a producer, the ciliate protozoan *Tetrahymena thermophila* as a consumer, and the bacterium *Escherichia coli* as a decomposer; the second one was the multi-species microcosm consisting of populations of the green algae *Chlorella* sp., *Scenedesmus* sp. and the cyanobacterium *Tolypothrix* sp. as producers, the ciliate protozoan *Cyclidium glaucoma*, the rotifers *Lecane* sp. and *Philodina* sp. and the oligochaete *Aeolosoma hemprichi* as consumers, and bacteria as decomposers (Fuma et al., 2003b, 2009; Polikarpov 2001, Polikarpov et al., 2004).

The resulting comparative effect doses, i.e., equi-dosimetric data, are given in Figure 19 and Table 9 for the three-species and multi-species microcosms, respectively. The results indicate that if "species reduction", i.e., extirpation of at least one species in the microcosm, is adopted as an endpoint, then 500-1000 Gy acute γ-irradiation is considered to be equivalent to 5000 J/m^2 UV-C, pH3.5 acidification, 550 $mg.l^{-1}$ Mn, 5.9 $mg.l^{-1}$ Ni, 6.4 $mg.l^{-1}$ Cu, 47 $mg.l^{-1}$ Gd and 49-91 $mg.l^{-1}$ Dy for the three-species microcosm (Figure 17), and 500-5000 Gy acute γ -irradiation is considered to be equivalent to 0.5-8 $mg.l^{-1}$ benthiocarb, 5-10 $mg.l^{-1}$ LAS and 200 $mg.l^{-1}$ soap for the multi-species microcosm (Table 9).



γ-rays (Gy)	—	50-100	500-1000	5000	Fuma et al., 1998b
UV-C (J/m^2)	100	1000	5000	10000	Takeda et al., 1998
Acidification	—	pH 4	pH 3.5	—	Miyamoto et al., 1998
Al (mg/L)	0.27	2.7-13	—	27	Fuma et al., 2003b
Mn (mg/L)	—	5.5-55	550	—	Fuma et al., 2000
Ni (mg/L)	0.59	—	5.9	59	Fuma et al., 1998a
Cu (mg/L)	0.64	—	6.4	—	Fuma et al., 2003b
Gd (mg/L)	7.9	16	47	160	Fuma et al., 2001
Dy (mg/L)	8.1-16	29	49-91	160	Fuma et al., 2005

—: Not exmained

Figure 19. Effect doses of γ-rays and other stressors for the three-species microcosm evaluated by the radio-chemo-ecological conceptual model (Polikarpov et al., 2004).

Table 9. Effect doses of γ-rays and chemicals for the multi-species microcosm evaluated by the radio-chemo-ecological conceptual model (Fuma et al., 2009).

Toxic agents	Physiological or ecological masking zone	Zone of damages to ecosystems		References
		Number reduction	Species reduction	
γ-rays (Gy)	100	-	500-5000	Fuma et al., 2009
Chemicals ($mg.l^{-1}$)				
Cu	-	0.6-1.2	-	Sugiura et al., 1982
Benthiocarb[a]	-	-	0.5-8	Takagi et al., 1994
Simazine[a]	-	0.08-1.28	-	Takagi et al., 1994
LAS[b]	1.5	2.5	5-10	Takamatsu et al., 1995
Soap	30	50-100	200	Takamatsu et al., 1995

[a] Herbicide
[b] Linear alkylbenzenesulfonate (surfactant)

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Radioecology is a highly multidisciplinary scientific discipline situated at the crossroads between risk induced by environmental radioactivity and its consequences on both, man and the environment.

The overall objective of the Union is to promote radioecology worldwide, in all its dimensions, from research activities up to expert advice and operational management.

An overarching role of the Union is to perpetuate a "think tank" capacity on radioecology issues through the maintenance of a network of scientists and professionals.

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