

Chapter 7

Risk To The Environment From Development

Introduction

Any consideration of the risk to the environment from development needs initially to confront a number of issues:

1. Does the natural environment include humans?
2. Any general procedures related to risk assessment need to be applicable to the risk to the natural environment from development associated with chemicals (to be dealt with in Chapter 8) as well as other sources of risk to the natural environment from development.
3. In the same vein, the methods adopted to deal with contaminated sites need to be applicable to the risk to the environment from development.

The Executive Director of the Environment Protection Agency is also the Supervising Scientist, responsible for protecting the environment by assessing the adequacy of environmental protection relating to uranium mining in the Alligator Rivers Region and by advising on best practice in environmental protection. The Supervising Scientist controls an Environmental Research Institute situated in Jabiru, Northern Territory, which undertakes research.

The environment and human populations

Section 3 of the Commonwealth *Environment Protection (Impact of Proposals) Act 1974* defines the environment to include all aspects of the surroundings of human beings, whether affecting human beings as individuals or in social groupings. This is generally taken to mean the ecological, economic, social and cultural aspects relevant to whatever matter is under consideration.

There are certain activities conducted by the EPA in which the term environmental risk includes the risk to human populations, and certain activities in which consideration of the risk to humans is the responsibility of another government department. In its consideration of uranium mining in the Northern Territory, the EPA needs to consider risks to the flora, the fauna, and to both present and future generations of humans who may occupy the area. When dealing with the potential risks of agricultural or industrial chemicals, or the risks associated with contaminated sites, then responsibility for human health risk assessment is with the National Health & Medical Research Council (NHMRC) so that the use within the EPA of the term environmental risk, in relation to these areas, refers to the flora and fauna only.

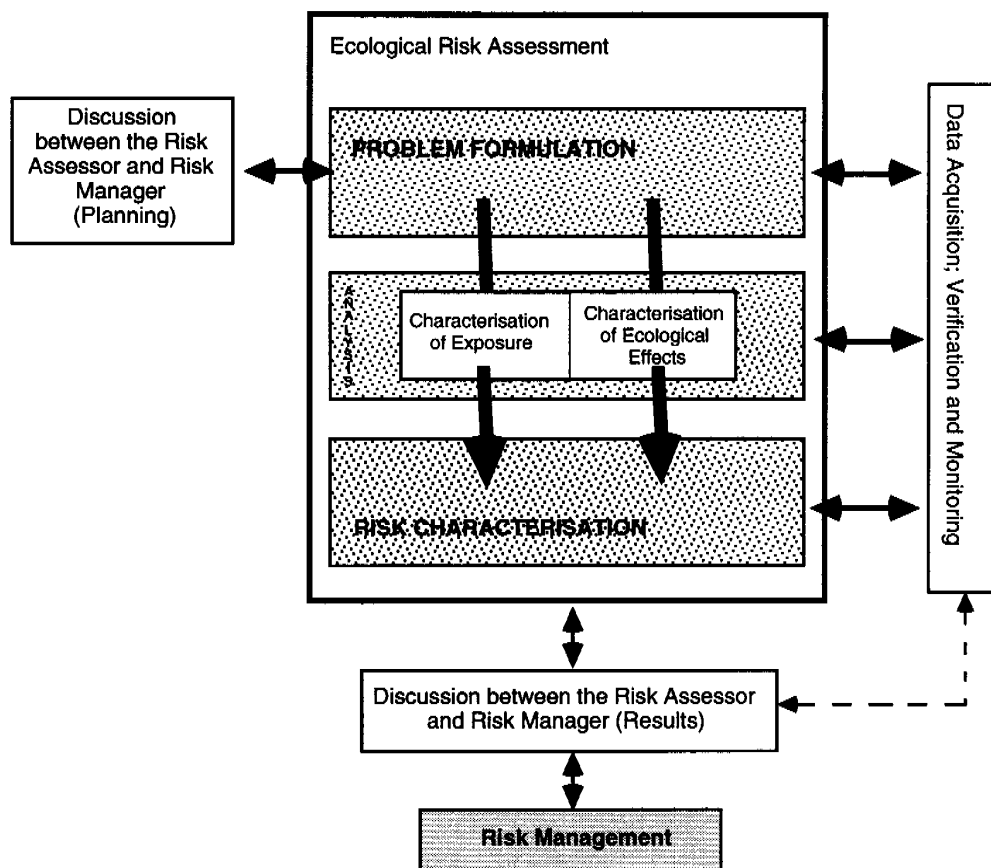
Such conflicts mean that it will be difficult, if not impossible, to find acceptable definitions for terms such as environmental risk assessment and ecological risk assessment. To some the terms are synonymous and include risks to people — after all there is a field of scientific endeavour known as human ecology. To others they are synonymous and exclude risks to people. And to a third group the first term includes people whereas the second term excludes people.

In a consultancy report to the Department of Human Services and Health (Axis Environmental, 1995) environmental health is defined as "the state of human health and well-being related to the environment and environmental factors such as air, water, soil contamination (as distinct from occupational factors, infectious diseases etc.)". Such a definition of environmental health is too narrow as it does not consider the state of the environment — a topic of crucial interest to the EPA. As mentioned in the previous chapter, there is a growing realisation that hazardous industry planning needs to include the biophysical environment

Ecological risk assessment

The US EPA uses the term ecological risk assessment to exclude human health risk assessment and has issued a framework for ecological risk assessment (US EPA, 1992). This framework, shown in Figure 7.1, was based on the principles of human health risk assessment and thus provides an entry to an integrated framework for human and ecological risk assessment. The framework of Figure 3.2 could equally well apply.

Figure 7.1 Framework for Ecological Risk Assessment (US EPA - 1992)



Whatever theoretical framework is chosen, most of the problems in conducting an ecological risk assessment are practical. The problem formulation step (i.e. dealing with concerns and their consequences) is vital because one needs to determine which part of the ecosystem is vulnerable. This is not easy because there is little agreement amongst ecologists on appropriate measures for the health of an ecosystem.

Another recent area of convergence between human health risk assessment and ecological risk assessment is in the use of probabilistic methods, such as Monte-carlo modelling, to accomplish the uncertainty analysis. Monte-carlo models have been used in systems analysis, and specifically ecosystems analysis, for about fifteen years. The method was used, as an initial screening tool, to identify the causes of eutrophication in the Peel Inlet of Western Australia (Hornberger and Spear, 1983) and has recently been advocated for human health risk assessment (Finley & Paustenbach, 1994; Copeland et al., 1994).

The Office of the Supervising Scientist (OSS) and the Environmental Research Institute of the Supervising Scientist (ERISS)

The Office of the Supervising Scientist was set up under the Environment Protection (Alligator Rivers Region) Act of 1978 (as amended in 1993), following the findings of the Ranger Uranium Environmental Inquiry into the environmental aspects of mining newly-discovered uranium mineralisation in the Alligator Rivers Region of the Northern Territory.

The purpose of the Supervising Scientist is to protect the environment in the Alligator Rivers by assessing the adequacy of existing methods of environmental protection, by developing improved ways of environmental protection and by providing expert advice.

The Environmental Research Institute of the Supervising Scientist is situated at Jabiru, in the Northern Territory. ERISS carries out scientific research on the environmental impact of mining and on tropical freshwater and estuarine ecosystems. As at December 1994 this was done within two research programs — Wetland Management and Impact of Mining — which were divided into five research groups: biology, ecosystems, chemistry, radioactivity and geomorphology.

Tasks in prioritising risk

In this section we will conduct an uncertainty analysis by using a probabilistic methodology to apply ecological risk assessment techniques to the operations of the Ranger Uranium Mine so as to prioritise risks. This will be followed by a case study, more limited in scope, that examines some issues associated with radiological pathways. In both cases, the particular probabilistic modelling tool to be used in such an ecological risk assessment is Monte-carlo modelling.

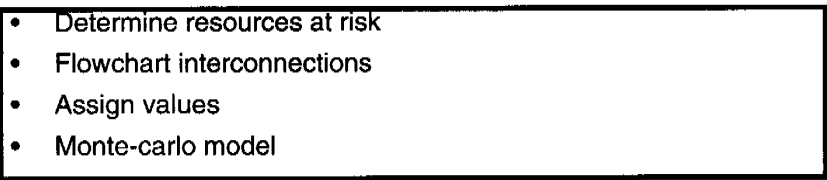
There are three areas of environmental risk that need to be considered:

- failure of engineered constructions;
- chemical contamination; and
- radiological pathways.

The case study, which serves as an example of the use of the methodology, considers only the radiological pathways.

To apply probabilistic modelling involves four steps (Fig. 7.2):

Figure 7.2 Steps in Uncertainty Analysis for Ecological Risk Assessment

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- Determine resources at risk
 - Flowchart interconnections
 - Assign values
 - Monte-carlo model

1. Develop flow charts of the systems, likely emissions, pathways and fate of materials.

This requires discussion and collaboration with experts in each of the three aspects to be studied.

2. Quantify the hazards associated with each step of the process.

When examining chemical contamination or radiological pathways this step requires collaboration with a health physicist or toxicologist able to assign hazards to particular chemicals and radioactive species.

3. Determine probability distributions for key steps in the process, and run Monte-carlo simulations to determine the expected probabilities.

This step requires collaboration between the hazards expert and a statistician/Monte-carlo modeller capable of expressing the hazards information in the form of probability distributions to be assigned to each step of the flow pathway and incorporate the distributions into a Monte-carlo model.

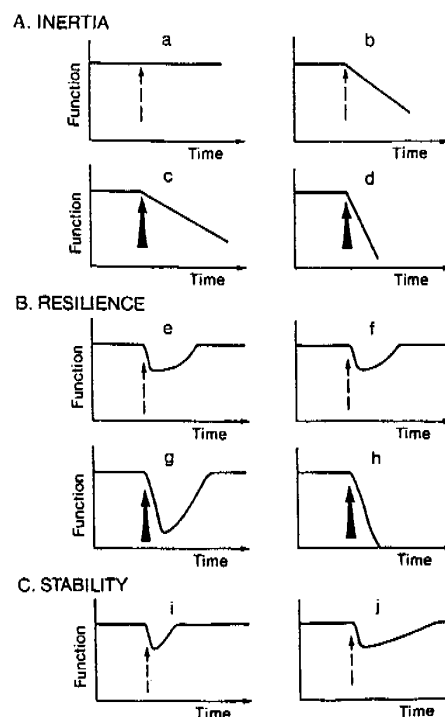
4. Rank the probabilities on the basis of some criterion such as risk/benefit, utility, or other grounds.

This step, which is the prioritisation step, requires the skills of an expert, or an expert panel, able to assign priorities (e.g. research priorities) on the basis of technical information. It may be possible to accomplish this by using an environmental economist able to assign utilities or determine risk/benefit curves. But then again, it may not. The discussion in Chapter 5 indicated that assigning numeric values to environmental damage is a difficult, if not impossible, task.

Characterising the health of an ecosystem is a difficult task and there is little agreement on which population variables to use. Ecological response indicators quantify the integrated response of ecological resources to individual or multiple stressors. Examples include measurements of the condition of individuals (e.g. frequency of tumours), populations (eg. abundance, biomass), and communities (species composition, diversity). Much of the work done in assessing the risk to the environment as a result of uranium mining in the Northern Territory has concentrated on studies of key indicator species such as freshwater snails, or the rainbow fish (*Melanotaenia nigrans*).

Research in Australia has cast doubt on the utility of community measures, diversity indices in particular (Campbell, 1990), whereas it is extremely difficult to obtain sufficient monitoring data to characterise population characteristics such as those suggested by Underwood (1990) as shown in Fig. 7.3.

Figure 7.3 Characteristics of populations relevant to monitoring programs and environmental disturbances (From Underwood, 1990)



Because of this diversity in possible ecological response indicators the methods used to conduct ecological risk assessments will differ. The US EPA has assembled case studies on eleven ecological risk assessments (US EPA, 1993, 1994) to determine whether they addressed generally accepted components of an ecological risk assessment (as defined by the US EPA) or provided an alternative approach to assessing ecological effects.

As an example, in the use of ecological risk assessment techniques to investigate the problem of radiological pathways involved in the operations of the Ranger Uranium Mine, we have re-analysed the work involved in estimating the risk associated with the land application of effluent water, given by Moroney (1992). The framework used to examine this particular problem is given in Figure 7.2.

CASE STUDY: Risk assessment of radionuclide pathways into the environment

1. Introduction

There is considerable literature on the hazards associated with radionuclide release. A scientific summary is given by Vennart (1983), whereas a popular account may be found in Caufield (1990). The absorbed dose (Gray) is defined as the energy deposited per unit mass of material. The SI units are J/kg which are given the special name of Gray (Gy). However, the biological damage produced by radiations depends not only on the absorbed dose but also on the type of radiation that delivered the dose. These differences are roughly measured by a quantity called the quality factor, which has values of 1 for X, gamma and beta radiation, 10 for neutrons, and 20 for alpha particles. To place all radiations on an equal footing with respect to biological damage a quantity known as the dose-equivalent has been introduced, which is the product of the absorbed dose and the quality factor. The unit of the dose-equivalent is the Sievert (Sv). Annual dose-equivalent limits above background, set by the International Commission on Radiological Protection (ICRP) in 1977 for the general public, were 1 mSv per year as a lifetime average, with 5 mSv permitted in any one year.

The particular problem dealt with by Moroney (1992) is the radiation exposure that would be received following the land application of effluent water from the restricted release zone of the Ranger Uranium Mine. The problem has been chosen for analysis because of the conflicting results of Moroney (1992) and Carter et al. (1994). The Ranger Uranium Mine, between 1986 and 1989, applied an average annual load corresponding to 2 kBq/m² of radium-226, and 44 kBq/m² of uranium (Moroney, 1992, Table 1). The calculations of Carter et al. (1994) indicate that after 30 years of application to the land this will lead to a dose-equivalent that is above the acceptable limit, whereas the calculations of Moroney (1992) indicate that the resulting dose-equivalent will be below the acceptable limit. Both studies used the same assumptions, but varied the parameters used to model the pathways. In this paper we wish to study the probability distribution of the dose-equivalent by: (i) allowing the model parameters to be treated as random variables; and (ii) examining the sensitivity of the results to the assumptions.

2. Pathways

The pathways for radionuclides are shown in Fig. 7.4. The risk analysis endpoint is the radiation health of local residents who live a traditional lifestyle, after the end of the land application program. Each pathway in Fig. 7.4 is, in practice, determined from a complex model. We follow a traditional systems analysis approach which brings the flow pathways to the simplest possible pathways that still incorporate the essential and relevant features of the system. This decomposition was undertaken by Moroney (1992) and Carter et al. (1994) and simplifies the problem to one of linear relations linking an input, expressed in kBq/m², with an output expressing dose-equivalent in terms of Sv per year. If we express this as

$$D = \sum c_i I_i$$

then c is the linear function linking the input I and the dose-equivalent D for each of the pathways, represented by the subscript i . Altogether ten values need to be considered for the subscript i . Each pathway has a dose-equivalent as a result of radium and a dose-equivalent as a result of uranium.

The function, c , depends on exposure to the particular activity considered under the particular pathway, as well as any physical parameters that are part of the sub-model involved in determining the relationships. Thus

$$c_i = A_i \beta_i C_i$$

where A is used to represent human activity that determines the dose-equivalent, β represents a physical parameter whose value can vary over a range, and C is a constant along any particular pathway. The values of C were determined from the estimates for c given in Moroney (1992, Table 9), which are reproduced in Table 7.1.

Table 7.1 Values of A and c used to infer β and C

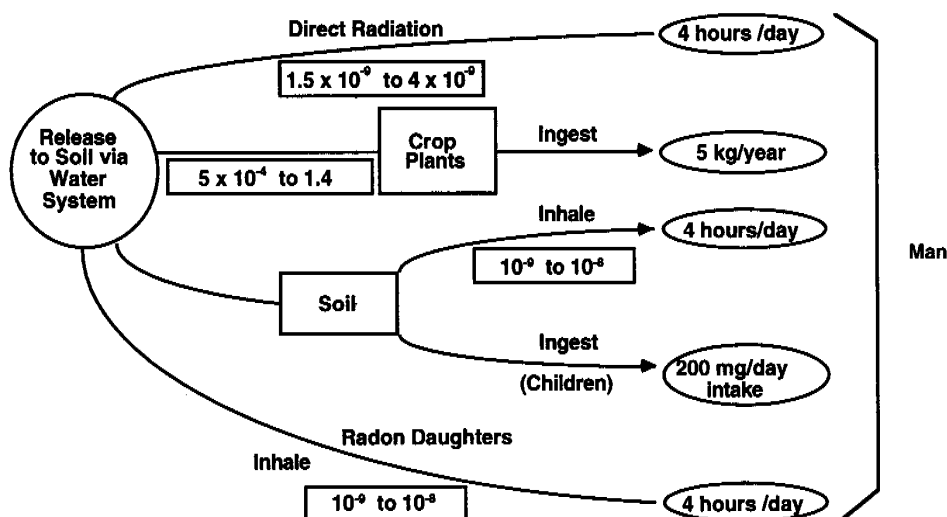
	i	A	units of A	c (Moroney)	c (Carter et al)
Direct	1 (Ra)	4	hours per day	3.5×10^{-6}	4×10^{-6}
Food	2 (Ra)	5	kg/year	1.7×10^{-6}	2×10^{-6}
Soil ingest	3 (Ra)	200	mg/day	n/e	2×10^{-7}
Inhale Rn	4 (Ra)	4	hours per day	6.1×10^{-7}	6×10^{-6}
Inhale dust	5 (Ra)	4	hours per day	1.2×10^{-9}	6×10^{-8}
Direct	6 (U)	4	hours per day	4.7×10^{-8}	n/e
Food	7 (U)	5	kg/year	3.7×10^{-8}	2×10^{-8}
Soil ingest	8 (U)	200	mg/day	n/e	1×10^{-7}
Inhale Rn	9 (U)	4	hours per day	n/a	n/a
Inhale dust	10 (U)	4	hours per day	2.0×10^{-8}	6×10^{-7}

n/e - not estimated

n/a - not applicable

Figure 7.4 Uncertainty Analysis

$$\text{TOTAL} = \sum \boxed{\text{PARAMETERS}} \times \text{ACTIVITY} \times \text{CONSTANTS}$$



2.1 Direct radiation:

Radium:

The activity is assumed to be the gathering of fruit and vegetables for 4 hours per day. This will be denoted by A_1 and will be used in a number of the other pathways. The major physical parameter uncertainty arises from the value of the effective dose-equivalent rate relating kBq/m^2 to Sv/h . According to Carter et al. (1994) this ranges from 1.5×10^{-9} to $4 \times 10^{-9} \text{ Sv/h per kBq/km}^2$ for Radium (^{226}Ra), with Carter et al. (1994) choosing a value of 3×10^{-9} , and Moroney (1992) choosing a value of 2.6×10^{-9} .

Thus

$$A_1 = 4 \text{ hours per day}$$

$$\beta_1 = 1.5 \times 10^{-9} \text{ to } 4 \times 10^{-9}$$

$$C_1 = 336$$

Uranium:

Moroney (1992) estimates direct radiation from uranium in surface soil. Values used are:

$$A_1 = 4 \text{ hours per day}$$

$$\beta_6 = 3.5 \times 10^{-11}$$

$$C_6 = 336$$

With this choice of units, C represents the number of days per year that food collection takes place. Moroney appears to have assumed 336 days which corresponds to 7 days a week for 48 weeks.

2.2 Ingestion from food

Radium

The activity is assumed to be the gathering of 5 kg per year of edible fruit and vegetables. According to IAEA (1992) 1 Bq/g of ^{226}Ra in the soil would result in a concentration factor of $4 \times 10^{-2} \text{ Bq/g}$ in edible vegetation. Moroney (1994) lists a large range for the concentration factors with a low value of 5×10^{-4} for vegetables, and a high value of 1.4 for pastures.

$$A_2 = 5 \text{ kg per year}$$

$$\beta_2 = 5 \times 10^{-4} \text{ to } 1.4$$

$$C_2 = 1.0 \times 10^{-5}$$

where C has been calculated assuming $\beta_2 = 4 \times 10^{-2}$, and $c_2 = 2 \times 10^{-6}$

Uranium

Moroney (1992) lists concentration factors for uranium as ranging from 1.6×10^{-4} for vegetables to 0.14 for pastures, and uses a value of 3.7×10^{-2} . Appropriate factors then appear to be

$$A_2 = 5 \text{ kg per year}$$

$$\beta_7 = 1.6 \times 10^{-4} \text{ to } 0.14$$

$$C_7 = 2.0 \times 10^{-7}$$

where C has been calculated assuming $\beta_7 = 3.7 \times 10^{-2}$ and $c_7 = 3.7 \times 10^{-8}$

2.3 Direct ingestion of soil

Carter et al. (1994) assume that children ingest about 200 mg per day of soil. Factors are then

$$A_3 = 200 \text{ mg per day}$$

$$\beta_3 = 1$$

$$C_3 = 10^{-9}$$

Uranium

$$A_3 = 200 \text{ mg per day}$$

$$\beta_8 = 1$$

$$C_8 = 5.0 \times 10^{-10}$$

2.4 Inhalation of radon daughters

Moroney (1992) estimates c to be 6.1×10^{-7} whereas Carter et al. (1994) obtain a value one order of magnitude larger. Both assume that the radon daughters are inhaled during the 4 hours per day that fruit and vegetables are being collected:

$$A_1 = 4 \text{ hours per day}$$

$$\beta_4 = 0.1 \text{ to } 1$$

$$C_4 = 1.5 \times 10^{-6}$$

Uranium does not emit radon daughters. Take $\beta_9 = 0$

2.5 Inhalation of resuspended soil

The inhalation of suspended soil particles occurs during the period that collection of fruit and vegetables takes place. The major parameter uncertainty arises from assumptions concerning the resuspension factor for contamination in the surface layer. Carter et al (1994) assume this to be $10^{-8}/\text{m}$ whereas Moroney (1992) assumes that it is $10^{-9}/\text{m}$. Using these values gives:

$$A_1 = 4 \text{ hours per day}$$

$$\beta_5 = 10^{-9} \text{ to } 10^{-8} / \text{m}$$

$$C_5 = 1.5$$

Uranium

$$A_1 = 4 \text{ hours per day}$$

$$\beta_5 = 10^{-9} \text{ to } 10^{-8} / \text{m}$$

$$C_{10} = 15.0$$

2.6 Total

The total effective dose equivalent rate (Sv/y) for all exposure pathways for the deposition of $1 \text{ kBq}/\text{m}^2$ of radium and uranium is the sum of the above effective dose-equivalents.

$$\begin{aligned} \text{Total dose} = & T[A_1\{P_R(\beta_1C_1 + \beta_4C_4 + \beta_5C_5) + P_U(\beta_6C_6 + \beta_5C_{10})\} \\ & + A_2\{P_R\beta_2C_2 + P_U\beta_7C_7\} + A_3\{P_R C_3 + P_U C_8\}] \end{aligned}$$

where T = effective time period of disposal

P_R = annual load in kBq/m^2 for radium

P_U = annual load in kBq/m^2 for uranium.

This is shown diagrammatically in Figure 7.4.

3. Monte-carlo modelling

Results of Carter et al. (1994) gave the total for all pathways as 1.2×10^{-5} for radium and 7.2×10^{-7} for uranium. Moroney (1992) obtains 5.8×10^{-6} for radium and 1.0×10^{-7} for uranium (or 3.2×10^{-7} using a different soil loading model).

There are two sources of parameter uncertainty in the problem. The first is the uncertainty in the physical parameters denoted by beta in the above pathways. The second is the uncertainty in future traditional use of the land. This uncertainty in lifestyle parameters is indicated by the term A in the above pathways. The above description has followed both Moroney (1992) and Carter et al. (1994) in setting fixed values for postulated traditional use of the land following cessation of land application. In practice it is widely acknowledged that these quantities are highly uncertain. The assumption of 4 hours per day spent in gathering fruit and vegetables is likely to be an upper bound and assumes virtually full-time gathering during the daylight hours of the dry season.

The choice of 5kg per year per person of flora to the Aboriginal diet is justified by Akber & Marten (1992). At first sight the figure seems low, both in terms of land productivity of the 35 ha (McQuade, 1992) irrigated site ($1/7$ kg per ha per year) and in terms of the finding by Beck (1986) who estimated that day trips for hunting provided between 0.2 and 2.0 kg of bush food. However, this bush food is primarily meat, whereas the 5kg per person per year refers only to fruit and vegetables harvested from the land application area.

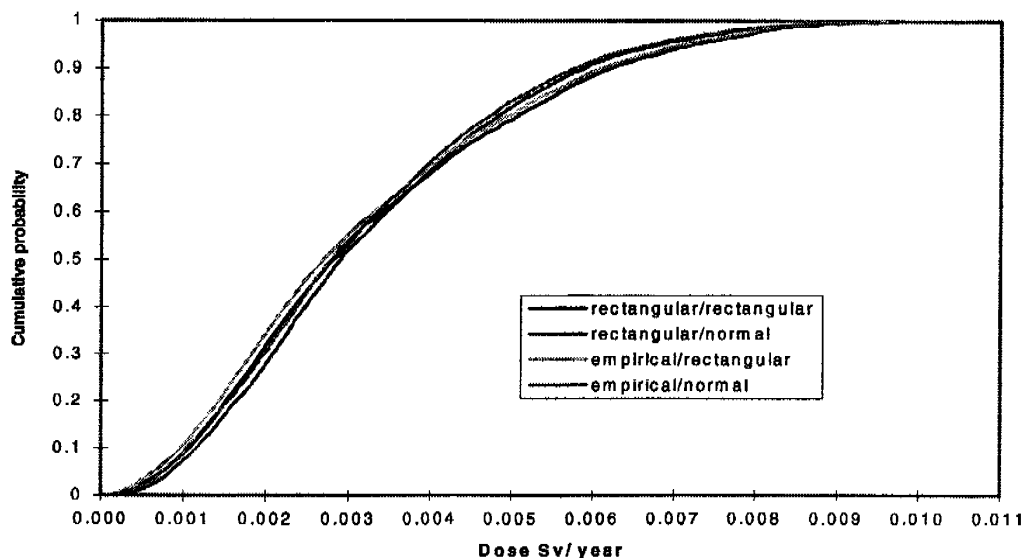
The assumed figure of 200 mg/day soil ingestion by children is also the figure used by the US EPA (US EPA, 1989). Finley et al. (1994) discuss the data on which it is based and provide a cumulative probability density function for soil ingestion rates for children that allows for the observation that most children ingest virtually no soil, but that a small group ingest very large amounts.

4. Results

Macpherson (1995) undertook an uncertainty analysis of the above problem using Monte-carlo modelling. Various probability distributions were assumed for the variation of the beta distribution. The results are based on an assumed 30 year disposal period for an average annual load corresponding to 2 kBq/m^2 of radium-226 and 44 kBq/m^2 of uranium. The results, depicted in Figure 7.5, indicate that there is a 90% probability that a dosage of 1mSv/yr will be exceeded.

Two aspects of this result are noteworthy. The first is that the result is relatively insensitive to the choice of probability distribution. The second is that the result is strongly driven by the high value of 1.4 for beta in the pathway dealing with crops and plants. This pathway dominates the subsequent analysis because of this large value for beta. However, the value of 1.4 refers to pastures. If this is considered unrealistic, and a maximum value of 0.014 is assumed, then the probability of exceeding 1 mSv/yr drops to 57%.

Figure 7.5 Cumulative Probability Results of Radiation Exposure from Land Application of Effluent Water



5. Discussion

Detailed risk analysis of the issue of land application of contaminated water reveals the importance of Aboriginal issues in determining the outcome of any Monte-carlo modelling. This result is, of course, not original. The paper by Robotham & McLaughlin (1992), of the Northern Land Council, identified the occupancy rate as a key variable, deemed the choice of 4 hours per day as appropriate and reached the conclusion that restrictions would have to be placed on the land application area to prevent it becoming a permanent camp site. What is original in our analysis is that the application of a formal ecological risk analysis methodology has led to the identification of the uncertainties associated with Aboriginal lifestyles as being the key environmental uncertainties and thus leads to the question of whether the existing research priorities of ERISS, which concentrate on reducing the uncertainties in the physical, chemical, biological and geomorphological parameters are indeed adequate to meet future issues that arise.

There seems little doubt that the Aboriginal perspective will emerge as a key variable, and a key unknown, in any risk analysis of the radionuclide risk, the chemical risk and probably even of the risk of failure of engineered constructions. The reason is that the risk, as perceived by the Aboriginal community, is so great as to preclude risk management options based on technically determined actual risks. This is exemplified by McLaughlin (1991) who states:

The history of technical man in the Alligator Rivers Region in explaining himself to Aboriginal people has not been successful as the field practices, certain field results, debate and acrimonious politics surrounding the release practices of land application and stream releases have created suspicion and cynicism among Aboriginal people as to the efficacy of such practices.

and

... contaminant encapsulation in a single or known limited number of repositories is a far more viable means of managing contaminants over time, as opposed to dispersing them beyond recall in the environment...Aboriginal people also perceive this risk in the dilute and disperse approach and insist that their future should not have to depend on the present scientific assurances of 'dispersers'.

and also

...Will the assurances of regulators today become regrettable errors in 20 years time?

The water management issue is one that best illustrates the wide discrepancy between the actual risk, as assessed using the tools of ecological risk assessment as they presently stand (Suter, 1993), and the perceived risk as illustrated by the above quotes. Armstrong & McNally (1991) point out that "Ranger has applied for approval to release water directly from the restricted release zone to Magela Creek, if required, every year since 1986....However, such releases have not received final approval from the Federal authorities ...".

In fact, the extreme rainfalls during the first two months of 1995 meant that the Federal authorities gave approval for such a release to take place. On 9 March 1995 the Northern Land Council sought an injunction against ERA (who control the Ranger Uranium Mine) to prevent release. ERA offered to hold the release for a week. During that week the heavy rains ceased and the flow rate in Magela Creek dropped below a level sufficient to ensure adequate dilution. Thus no release took place.

Social trust

There seems little chance of quantitative risk assessment being able to provide answers that reconcile such strongly held divergent views in relation to water release. The situation is one in which the Aboriginal community has a risk averse utility function that treats the possibility of any contamination of the natural water system as unacceptable. The utility function for the mining company is one that would be considered risk neutral (on the basis of the technical risk), or risk prone (on the basis of the Aboriginal communities' perceived risk). We have suggested in the chapter on setting priorities that, in such a situation, the role of research and education is to bring the two utility functions closer to the risk-neutral utility function. In this particular case, it seems that no amount of quantitative, technical work will reassure the Aboriginal community. This suggests that the Aboriginal community does not trust the work of the Supervising Authorities, presumably including the Office of the Supervising Scientist. It seems that, at the moment, the link between the Office of the Supervising Scientist (OSS) and the Aboriginal community is through the Northern Lands Council (NLC). Research work needs to be undertaken that will enable direct face-to-face contact between the Aboriginal community and research workers.

Earle & Cvetkovich (1994) distinguish two types of trust, interpersonal trust and social trust. Social trust, which is a trust in abstract systems and institutions, determines the success or failure of risk communication. The reason for this is that most members of the public lack the means by which to assess risky technologies. They therefore have to assess the institutions which appear to control technology. Questions of public trust, credibility, openness and the past track record in these respects become the key features in framing social attitudes (Wynne, 1980: p.186).

Earle & Cvetkovich (1994) argue that social trust is a risk judgement based on cultural values, rather than on notions of competency. Their ideas are governed by the United States and United Kingdom experience of community groups' opposition to hazardous projects.

The situation in relation to the release of water from the Ranger Uranium Mine appears to be one in which social trust can follow only from interpersonal trust. There has, to date, been a view that relationships between the OSS and the NLC should be conducted on an organisation to organisation basis. Such an approach, presumably designed to increase inter-organisational social trust, has failed in the case of the March 1995 attempt to release water. The traditional owners would not accept the assurances that the release would have no significant environmental impact.

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