Methodology for Ranking Restoration Options

Restoration Strategies for Radioactively Contaminated
Sites and their Close Surroundings
RESTRAT WP5

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Table of Content

List of Figures

List of Tables

1 Introduction

The distinction between *practices* and *interventions* as recommended by international radiation protection organisations may not always be clear for clean-up of land that has been contaminated with radioactive materials. However, in cases where there is existing exposure of a population from sites contaminated with the residues of past or old practices or work activities, the principles of protection for *intervention* are applicable. In the context of remediation of such sites, it is likely that social costs of disruption for those affected by the remedial measures and continuing long-term anxiety about residual levels of contamination for those continuing to live in the area will be important factors. The optimisation process of selecting the best strategy of remedial measures should therefore, in addition to the averted radiation detriment and the monetary costs, include considerations of how the measures can reduce anxiety and gain reassurance of the affected population. An optimised strategy would achieve conditions of *return to normality* without any restrictions associated with the residual contamination.

The formulation of the optimisation principle within a practice or an intervention will differ. The practical implementation of the optimisation of remedial measures for contaminated sites is, however, essentially the same process, whether it is considered in the context of the continuing operation of a practice, as part of decommissioning of a practice, or for intervention. In all cases, it includes the identification of remediation options available and how the exposures might be reduced, and choosing that remedial action which results in the greatest net benefit, considering all of the relevant factors that influence costs and benefits. These costs and benefits may include populations directly affected by the measures, both now and in the future, as well as to other parts of society. Decisions on remediation may go far beyond purely radiological protection considerations but can, however, often be limited to considerations of whether or not any of the range of possible remedial actions will itself result in a net benefit. In reaching such decisions it is important to consider carefully the benefits and disadvantages of the remedial actions because some actions can significantly disrupt the affected population or have serious impact on the environment.

For practical purposes measurable (operational) quantities such as radionuclide concentration or dose rate are needed to evaluate the effect of remedial measures in relation to radiological protection criteria. Such quantities are named *action levels* and they are related to the primary criterion, *e.g.* avertable dose, by suitable models for dose assessment from all relevant exposure pathways. Compliance with the action level would thus ensure compliance with the primary criterion.

2 Justification and optimisation principles in restoration

The system of radiation protection is based on the so-called justification and optimisation principles. When the subject is protection of the public against radiation exposure from contaminated land the justification/optimisation procedure is applied to the remedial or protection action for reducing this exposure. A short review of the justification and optimisation principles is given below.

2.1 Justification

Clean-up of contaminated land will introduce some benefit to the affected populations. The benefit of undertaking clean-up includes a large number of components or attributes, i, which quantify relative partial benefits, b_i. These partial benefits, depending on the circumstances, can be ‘positive’ benefits, or advantages, and ‘negative’ benefits, or disadvantages. Without intervention, the attributes, such as radiation doses - both individual and collective doses - and the anxieties they cause, will represent disadvantages as shown on the left side of Figure 1. After remediation, the disadvantages will have been reduced or even eliminated, and new attributes may have been introduced, as shown on the right side of Figure 1. Some of the new attributes may be advantageous, e.g. the reassurance produced by the remedial measure; others will be disadvantageous, *e.g.* the cost of the remedial measures and the collateral harm they may cause.

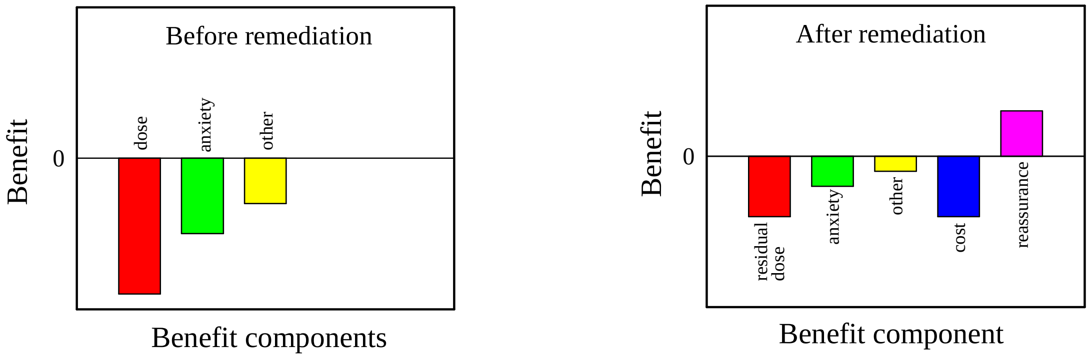


Figure 1. Benefit components, b, of clean-up operations. The left picture shows that the benefit components are all negative. The right picture shows that clean-up will reduce (or remove) some of the negative benefits, introduce new negative benefits (e.g. costs) and positive benefits (e.g. reassurance). The component 'other' includes negative benefit components such as social disruption.

Clean-up is justified when the net benefit, ΔB, is positive:

$$\Delta B = \sum_i b_i(\text{after clean - up}) - b_i(\text{before clean - up}) = \sum_i \Delta b_i > 0$$

The application of the justification principle to clean-up situations requires prior consideration of the benefit that would be achieved by the clean-up and also of the harm, in its broadest sense, that would result from it. It is emphasised that justification must consider non-radiological risks as well as radiological risks, *e.g*. chemical risks, and risks from industrial and transportation operations. Each of the benefit components, b_i, has to be expressed in the same units. These units must be in like quantities or values. For example, since costs are expressed in monetary terms, equivalent monetary values may be assigned to other parameters. Alternatively, other units of value must be used for example equivalent years of lost life.

Some *decision-aiding* techniques available for use in carrying out decision analysis have been described in detail in ICRP Publication No. 55 [3]. The primary objectives of these techniques are to identify the various factors influencing the decision, to quantify them, and systematically to examine the trade-offs between them, so that the process can be made open to the people responsible for the decision and to public scrutiny.

One decision-aiding technique that is capable of accepting input data of both a quantitative and a qualitative nature, and which can be used in a wide variety of situations, is multi-attribute utility analysis. Some of the factors to be used in such analyses are more or less quantifiable. More quantifiable factors are the avertable individual and collective risks from exposure to radiation for the members of the public and the individual and collective physical risks to the public caused by the clean-up. Others are the individual and collective risks to the workers carrying out the clean-up, and the monetary cost of the clean-up. The less quantifiable factors, including the reassurance provided by the clean-up but also the anxiety it causes, and the individual and social disruption resulting, are also factors relevant to the decision.

In analysing the inputs to the decision, it is necessary to decide on the relative importance or weight of each factor. These judgements have to be made irrespective of the decision-aiding technique used. The resultant decision will be the same provided that the database is the same and the judgements are consistent. If multi-attribute utility analysis is the technique used, then all the relevant factors can be directly included in the analysis by deriving or assigning utility functions to them, but weights still need to be assigned.

The net benefit, ΔB, of a clean-up operation will depend on several factors (attributes), *e.g.* avertable collective dose, ΔS, monetary costs of a clean-up operation, C, anxiety of the contamination, A, reassurance by the clean-up, R, etc. Thus the net benefit, ΔB, is a function of all the relevant parameters:

$$\Delta B(\Delta S, E, C, A, R, ...)$$

The individual dose, E, is often taken as the dose to the average member of the critical group. Depending on the clean-up option, collective dose may be reduced with or without changing the specified individual dose, E. Also, the critical group may change depending on the clean-up option. Thus it may be useful first to examine the effects of various levels of individual dose within a single option and among all options.

2.2 Optimisation

Normally, there would be a range of justified remediation options for which the net benefit is positive. The optimum remediation option would be the one for which the net benefit is maximised, as shown on the left side of Figure 2. Option 1 is the no-remediation option for which the net benefit is zero. In Figure 2 the options 4 to 8 are all justified because their net benefits are positive. Option 6 is the optimum because the net benefit is the maximum. The optimum remediation option does not necessarily mean the option with the lowest residual annual doses, either individual or collective, because there are additional considerations for determining the net benefit. This is illustrated in the right side of Figure 2 where options 7 and 8 entail a lower residual annual dose but give a smaller net benefit than the optimum option 6. If all remediation options have a negative net benefit, the no-remediation option would be the preferable.

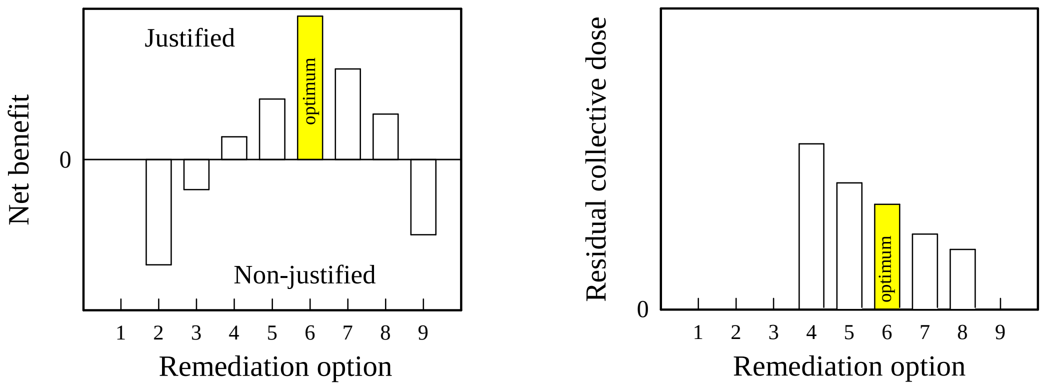


Figure 2. Net benefit of different remediation options and the corresponding residual collective dose, S, after clean-up. The left picture shows that there is a range of options, both justified and non-justified. The right picture shows the residual collective dose, S, after clean-up for the five justified options.

Most of the methods used in optimisation of protection tend to emphasise the benefits and detriments to society and the whole exposed population. Optimisation of clean-up, whether it is considered in the context of a practice or for intervention, is essentially an identical process: choosing the course of action which results in the maximum net benefit, considering all the relevant factors that influence the advantages and disadvantages of the clean-up operation.

For clean-up of contaminated land, society usually requires that the same level of protection be provided regardless of the source of exposure. Therefore, clean-up criteria that do not differ depending on whether the situation is deemed to fall within the category of practices or intervention are desirable, but may not always be possible.

The concept of optimisation of protection is practical in nature. Optimisation provides a basic framework of thinking - that it is proper to carry out some kind of balancing of the resources put into protection, and the level of protection obtained. The reduction in dose can only be achieved by the expenditure of some effort and by allocating additional resources. In such cases, it is necessary to decide whether the dose saving that is likely to result is worth the effort of achieving that saving. This is entirely consistent with the optimisation principle. In the optimisation process, two categories of radiological factors can be distinguished. The first category comprises the factors (attributes) that will always have to be included in the analytical procedure, particularly the cost of protection and the collective doses. The second category comprises the factors that may not always be necessary, such as the individual dose distribution, the time distribution of doses, the population receiving the doses, the possibility of options, etc. When all attributes that need to be considered have been specified, it may be that some of them cannot be appropriately quantified for inclusion in the analytical procedure. In this case, these factors will have to be assessed qualitatively, but the results of the qualitative analysis must be taken into account in reaching the optimum.

3 Techniques for optimisation

Decisions on clean-up in long-lasting exposure situations may well go far beyond purely radiological protection considerations. Satisfying the justification principle requires that the overall effect of the actions involved should do more good than harm, taking account of relevant radiological and non-radiological factors. The decisions can often be limited to considerations of whether or not any of the range of possible remedial actions will itself result in a net benefit. In reaching such decisions it is important to consider carefully the benefits and disadvantages because some remedial actions can significantly disrupt the exposed population.

Most decisions require multiple criteria to be taken into account. The field of multiple criteria analysis offers a number of approaches which take explicit account of multiple criteria in providing structure and support to the decision-making process. In case of restoration of contaminated sites there are several criteria or attributes that need to be considered when choosing an 'optimum' restoration strategy. When the performance and costs of all the protection options have been assessed, a comparison is needed to define the *optimum* protection option. When the optimum is not self evident, the comparison can be carried using a quantitative decision-aiding technique. The result of the application of the quantitative techniques is known as the *analytical solution*. If there are non-quantified, radiological protection factors to be taken into account, the analytical solution may *not* be the optimum solution, which then will have to be determined more intuitively. Of the different techniques available three will be described below. These are cost-effectiveness analysis, cost-benefit analysis and multi-attribute utility analysis.

3.1 Cost-effectiveness analysis

In cost-effective analyses only two factors can be included in the quantitative analyses, namely monetary cost of the different protective measures and the collective dose reduction from those measures. However, a cost-effectiveness analysis does not result in an optimisation of protection, since it does not involve the trade-off between protection costs and collective dose reduction. A cost-effectiveness analysis is rather a method to determine the best protection strategy obtainable from fixed resources. Cost-effective analyses are carried out when a specific dose reduction or the amount of money available for radiation protection is fixed. In this case, the net benefit will be maximised by either varying the monetary costs with the detriment costs as a constant, or varying the detriment costs with the monetary costs as a constant.

Cost-effectiveness analyses can therefore only define either the least costly way of achieving a specified reduction in exposure or the maximum reduction in exposure that can be attained for a fixed cost, but cannot optimise radiation protection. Cost-effectiveness analyses may, however, allow the *a priori* exclusion of available protection options and thus precede and simplify the formal optimisation analysis. For illustration of the cost-effectiveness methodology the data in Table 1 has been used.

Table 1. Collective doses and costs of protection for five protection options and for the reference case without protection. The monetary costs are given in the unit of US Dollars (USD).

Protection option	**Monetary costs [USD]**	**Collective dose [man·Sv]**
No protective measures	0	0.69
Option 1	10,000	0.56
Option 2	17,000	0.36
Option 3	23,000	0.30
Option 4	32,000	0.20
Option 5	36,000	0.18

It can be seen from the figures in Table 1 that the collective dose, *S*, decreases gradually when more efficient protection options with increasing cost, X, are being implemented. This can be seen in Figure 3 where the costs are plotted against collective dose (left-hand picture). The ratio $\Delta X/\Delta S$ is shown at the right-hand picture for each of the protection options 1 - 5.

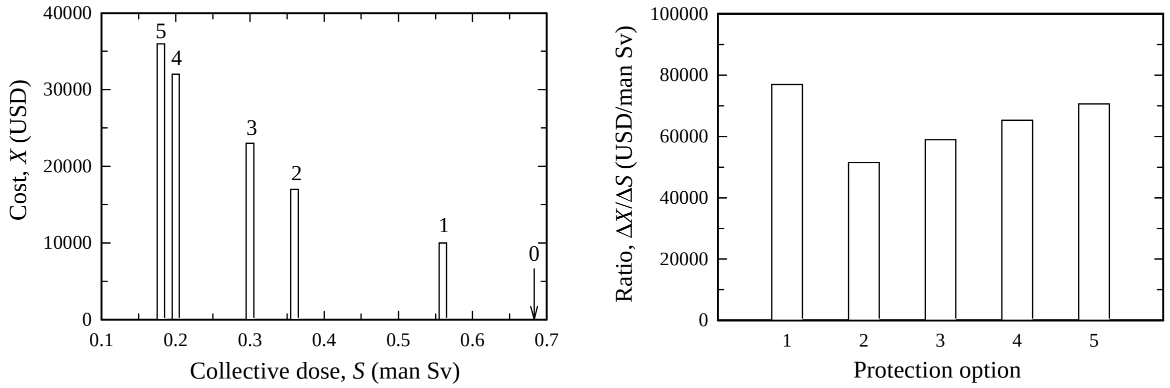


Figure 3. Protection options in terms of monetary costs and residual collective dose. The option marked "0" shown at the left-hand figure is the reference case without any protective action for which the residual collective dose is 0.69 man·Sv. The cost-effectiveness ratio $\Delta X/\Delta S$ is shown at the right-hand figure, where ΔX is the change in cost and ΔS the change in collective dose, both compared to the reference case. As $X_0 = 0$ it follows that $\Delta X_i = X_i$, and $\Delta S = S_0 - S_{residual}$.

It appears from Figure 3 that protection option 2 is the most cost-effective because this option has the lowest monetary cost per collective dose reduction.

3.2 Cost-benefit analysis

Cost-benefit analysis involves a balancing of costs in order to establish optimum levels of radiation protection. Optimisation of protection results in the best available combination of costs of radiation protection, *X*, and detriment, *Y*, so the sum of the costs (*X* + *Y*) is minimised. The optimisation process will therefore maximise the net benefit. The optimisation condition is fulfilled at a value of collective dose, S_{opt}, where the increase in cost of protection per unit collective dose balances the unit reduction of collective dose:

$$\left(\frac{dX}{dS}\right)_{S_{opt}} = -\left(\frac{dY}{dS}\right)_{S_{opt}}$$

This way of obtaining the optimisation of protection has also been called *differential cost-benefit analysis*. The level of protection defined by the above equation is such that a marginal increase in the cost of radiation protection is balanced by a marginal reduction in the cost of radiation detriment.

The principal characteristic of cost-benefit analysis is that the factors entering the analysis are commonly expressed in monetary terms. In these circumstances the collective dose is transformed into a monetary valuation using a reference value of avoiding a unit collective dose, α. This quantity can be related to the risk per unit dose, R (about 0.05 cancer Sv^{-1}), and the statistical loss of life expectancy per radiation induced cancer, l (about 15 years $cancer^{-1}$), with some allowance for loss of quality of life for non-fatal cancers and severe hereditary effects. The average loss of life expectancy per unit effective dose, L, can thus be calculated to be:

$$L = R \cdot l \quad [\text{years} \cdot \text{Sv}^{-1}]$$

giving a value of L of approximately one year per sievert.

Within the international radiation protection community it has been argued that a society for protection purposes should spend *at least* what correspond to the Gross National Product (GNP) per capita to save a statistical year of lost life and probably somewhat more. So-called *willingness-to-pay* studies have resulted in values of 200,000 USD ± 100,000 USD per saved year of statistical life, corresponding to 8 GNP ±4 GNP per capita for rich European countries. Therefore, the value of α can roughly be found from the following relation:

$$GNP \cdot R \cdot l < \alpha < 10 \cdot GNP \cdot R \cdot l$$

For rich European countries the value of GNP per capita is of the order of 25,000 USD $year^{-1}$, which would give a reference value of α between 25,000 USD $manSv^{-1}$ and 250,000 USD $manSv^{-1}$. The Nordic radiation protection authorities have recommended a maximum value of α of 100,000 USD $manSv^{-1}$ [14]. For illustration of the cost-benefit methodology the data in Table 2 has been used. The cost and collective dose data are identical to those used in the cost-effectiveness analysis.

Table 2. Collective doses and costs of protection and radiation detriment for five protection options and for the reference case without protection.

Protection option	Monetary costs [USD]	Collective dose [man Sv]	Detriment costs [USD]
No protective measure	0	0.69	55,200
Option 1	10,000	0.56	44,800
Option 2	17,000	0.36	28,800
Option 3	23,000	0.30	24,000
Option 4	32,000	0.20	16,000
Option 5	36,000	0.18	14,400

In addition to the monetary costs for the different protection options the equivalent monetary cost of the detriment, Y, are presented in Table 2. This cost component is calculated as:

$$Y = \alpha \cdot S_{\text{residual}}$$

where α is the equivalent monetary cost of averting a unit collective dose.

The upper left-hand and right-hand pictures in Figure 4 show the protection costs as a function of the residual collective dose, S_{residual}, and for each of the protection options. The lower left-hand picture in Figure 4 shows the detriment costs, Y, for the protection options, including the reference case without protection. An α-value of 80,000 USD per manSv has been used for the calculation of detriment cost. The lower right-hand picture shows for each option the sum of the protection costs and the detriment costs. It appears that option 2 has the lowest total cost and should therefore be considered as the optimum protection option. This conclusion can also be found by considering the differential cost per unit reduction in collective dose moving successively through options i to option i+1. The numerical value of $\Delta X/\Delta S$ exceeds the value of α of 80,000 USD manSv^{-1} when moving from option 2 to option 3, which appoints option 2 to be the optimum.

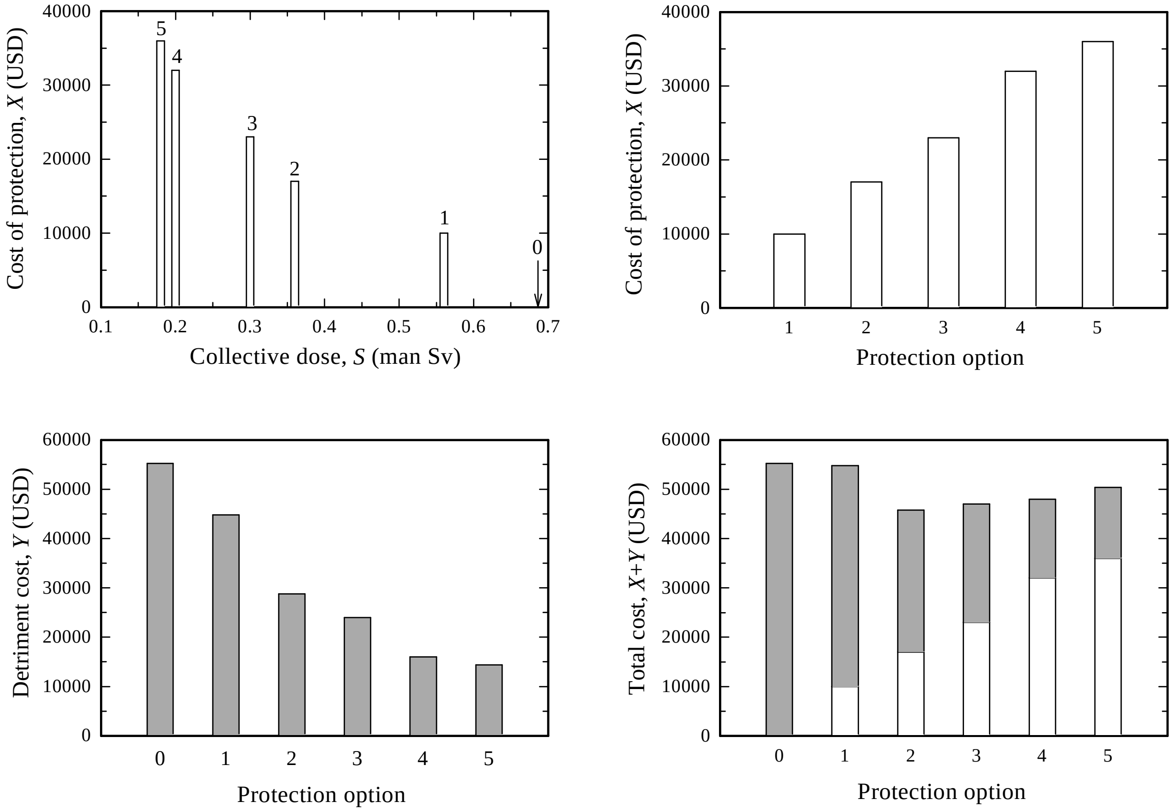


Figure 4. Costs and residual collective dose for different protection options. The option marked "0" is shown at the left figure and is the reference case for which the residual collective dose is 0.69 man Sv without any protective action. The protection costs, the detriment costs and the total costs are shown at the following figures for each of the protection options. The detriment cost, Y, is calculated as $\alpha \cdot S$ with a value of α equal to 80,000 USD per man Sv.

The cost-benefit analysis methodology is limited to quantitative comparisons between the protection costs and the detriment costs. In order to include other relevant factors, *e.g.* the distribution of individual doses within the collective dose, it is possible to extend the framework of cost-benefit analysis. This extension allows different values to be assigned to the unit collective dose through an additional component of the detriment cost depending on the individual dose levels involved. The extension can be expressed as:

$$Y = \alpha \cdot S + \sum_i \beta_i \cdot S_i$$

where S_i is the collective dose of individual doses E_i in the ith group and β_i is the additional value assigned to a unit collective dose in the ith group.

3.3 Multi-attribute utility analysis

The essence of *multi-attribute utility analysis* is to use a scoring scheme (or multi-attribute utility function) for the relevant factors with the property that if the score (or utility) is the same for two options there is no preference for one or the other. As basis for comparison between options or alternative strategies, a simple *multi-attribute* value function approach can be used. There are two major components of such value functions:

- the evaluation of each alternative strategy with respect to the considered attributes, known as *utilities*, ***u***

- scaling factors which reflect the relative importance of each of the attributes, known as the *weights*, ***w***.

The use of utility functions allows the introduction of factors that are not easy to quantify in monetary terms as is required in cost-benefit analysis. The utilities and weighting factors can be expressed in an additive form to give an overall evaluation of each of the alternative strategies, *i*, or options:

$$U_i = \sum_{j=1}^{n} w_j u_{ij}$$

U_i is the overall evaluation of option i, w_j is the weight assigned to the attribute j, and u_{ij} is the score or utility of the $\boldsymbol{n}$ factors associated with each of the alternative i on attribute j. The higher the value of U_i, the better the overall ranking of the option. Normally, weighting factors are measured on a ratio scale and normalised to sum to 1 or 100.

The aim of scoring is to assign values to each alternative reflecting the contribution to the overall evaluation from their performance on each end-attribute (sub-attribute). One way of defining the scores (utilities) is to assign the alternative which does best on a particular attribute a score of 100 (or 1) and to assign the alternative which does least well a score of 0. All other alternatives are assigned intermediate scores, which reflect their performance relative to these two end points. A major advantage of this methodology is that the utility functions need not necessarily be linear. For all non-linear utility functions, the knowledge of at least a third point (in addition to the points 0 and 100 (or 1)) is required to characterise the single utility function, $u(x)$. Depending on the relative position of the three points, the general shape can be determined as a linear, concave or convex function, either as increasing or decreasing functions. Functions of the decreasing type are shown in Figure 5 below.

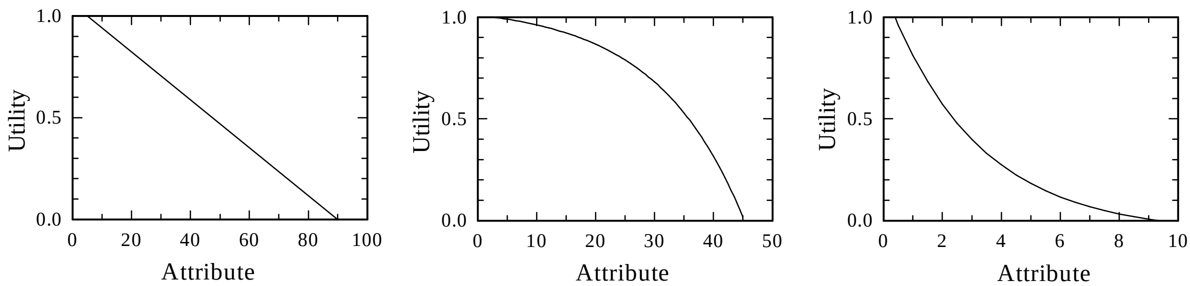


Figure 5. Examples of utility functions of the decreasing type. The left figure illustrates a risk neutral utility function (linear). The middle figure illustrates a risk averse utility function, which decreases faster nearer the worst consequences being more sensitive to variation at the upper end of the range of consequences. The right figure illustrates a risk prone utility function which decreases slower at the upper end thus being less sensitive to variation at the upper end of the range of consequences.

The data used for the cost-effectiveness analysis and the cost-benefit analysis regarding the monetary costs of protection, *X*, and for the collective doses, *S*, have been used also for the multi-attribute utility analysis. The attributes are shown in Figure 6.

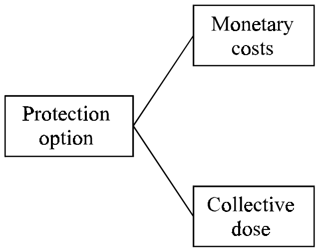


Figure 6. Attributes used in the multi-attribute utility analysis.

The utilities, *u*, for the attributes *monetary costs* and *collective dose* for each protection option have been determined from risk neutral utility functions, $u(x)$, where x describes the value of the attributes for the different options. For the monetary costs and the collective dose the utility functions has been determined from:

- monetary costs: $u(x = 0 \text{ USD}) = 1$ and $u(x = 36{,}000 \text{ USD}) = 0$
- collective dose: $u(x = 0.18 \text{ man Sv}) = 1$ and $u(x = 0.69 \text{ man Sv}) = 0$

The utility functions, u(x) can thus be expressed in the following way:

$$u_{\text{cost}}(x) = 1 - \frac{x}{36{,}000} \qquad \text{for } 0 \le x \le 36{,}000 \text{ USD}$$

$$u_{\text{dose}}(x) = 1 + \frac{0.18 - x}{0.69 - 0.18} \qquad \text{for } 0.18 \le x \le 0.69 \text{ man Sv}$$

and they are shown in Figure 7.

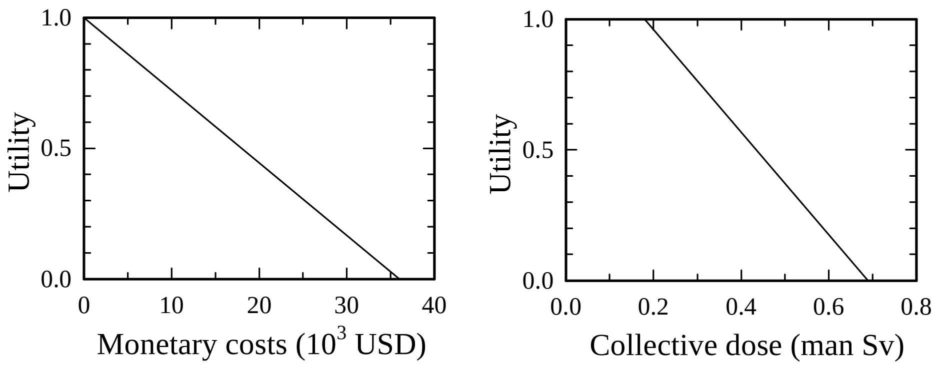


Figure 7. Utility functions for the attributes 'monetary costs' and 'collective dose'.

The utilities for each attribute and each option have been determined from the utility functions given above and the utilities are shown in Table 3.

Table 3. Utilities or scores, u(x), for five protection options and for the reference case without any protection.

Protection option	Monetary costs	Collective dose
No protective options	1	0
Option 1	0.72	0.25
Option 2	0.53	0.65
Option 3	0.36	0.76
Option 4	0.11	0.96
Option 5	0	1

The weighting factors, *w*, have been determined in the following way. If the ranges of the monetary costs and collective dose are called $R(X)$ and $R(S)$, respectively, the weighting factors can be obtained by constraining them to the same imposed criterion as for the cost-effectiveness and cost-benefit analyses described in Sections 3.1 and 3.2 as:

$$\frac{w(X)}{w(S)} = \frac{R(X)}{\alpha \cdot R(S)}$$

and then normalising so that $w(X) + w(S) = 1$. This gives the values $w(S) = 0.53$ and $w(X) = 0.47$ for an α-value of 80,000 USD manSv^{-1}.

The results are shown graphically in Figure 8. It appears that option 2 comes out with the highest score, *U*, and this protection option would thus be the optimum solution.

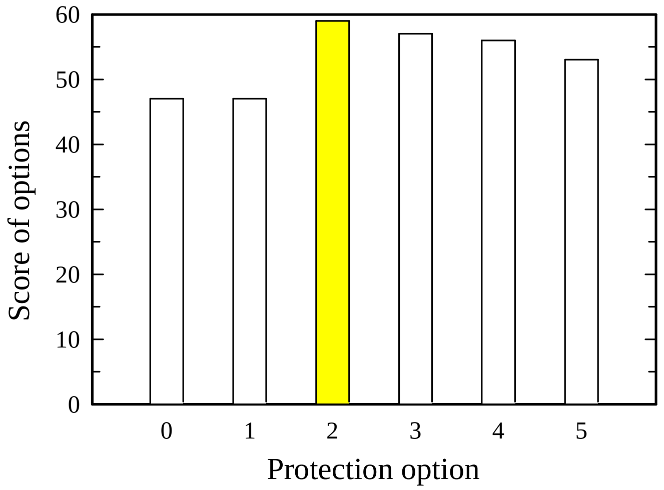


Figure 8. Overall evaluation of five different protection options with values of utilities for the attributes ‘monetary costs’ and ‘collective dose’ as shown in Table 2 and weighting factors of 0.47 and 0.53 for these attributes, respectively.

The overall score, U_i, of the different protection options, *i*, has been calculated as the sum of the products of weighting factors and utilities:

$$U_i = \sum_{j=1}^{2} w_j u_{ij} = w_{cost,i} \cdot u_{cost,i} + w_{dose,i} \cdot u_{dose,i}$$

$$U_0 = 100 \cdot (0.47 \cdot 1 + 0.53 \cdot 0) = 47$$
$$U_1 = 100 \cdot (0.47 \cdot 0.72 + 0.53 \cdot 0.25) = 47$$
$$U_2 = 100 \cdot (0.47 \cdot 0.53 + 0.53 \cdot 0.65) = 59$$
$$U_3 = 100 \cdot (0.47 \cdot 0.36 + 0.53 \cdot 0.76) = 57$$
$$U_4 = 100 \cdot (0.47 \cdot 0.11 + 0.53 \cdot 0.96) = 56$$
$$U_5 = 100 \cdot (0.47 \cdot 0 + 0.53 \cdot 1) = 53$$

It should be emphasised that it is the specification of the values of the different factors and attributes entering the analysis that determines the outcome, *not* the technique used. Therefore, it should be expected that the optimum results using different optimisation techniques would be the same if the same values of parameters were used in the analyses.

This important point can be verified by comparing the outcome from the example analyses given in the preceding sections. The outcome from the cost-benefit analysis and the multi-attribute utility analysis both appoints the protection option 2 to be the optimum. Although the cost-effectiveness technique does not present an optimised protection option because it does not involve any trade-off between collective dose and protection cost, it appears anyway that option 2 is the most cost-effective giving the highest dose reduction per invested amount of money.

There will, however, be uncertainties on the parameters used to calculate the values of the utility functions, u, and there will also be uncertainties on the values assigned to the weighting factors, w. These uncertainties can be included in the calculations of scores, U_i, by using software that is capable of building a model for the scores, $U_i(x, y, \ldots)$ in which uncertainty distributions can be assigned to the values of each of the attributes, $x, y, \ldots$, that defines the utility functions, $u_i(x), u_i(y), \ldots$, and to the weighting factors, w, for each of the attributes.

Several software systems for uncertainty analysis and decision-making between competing options are on the market. One of these systems is V•I•S•A from the company *Visual Thinking* [10]. This software system can be used to support the decision-making process. Decisions are modelled using hierarchical weighted value functions and the system has an extensive facility for visual interactive sensitivity analysis, which enables the decision-maker to explore the implications of changing or priorities and values.

Another system is Crystal Ball from the company *Decisioneering* [4]. It has the advantage of working on spreadsheets enabling the development of rather complex models; uncertainties can be assigned to model parameters and correlations made between them. Crystal Ball provides a statistical picture of the range of possibilities inherent in the parameter assumptions. Crystal Ball uses a Monte Carlo or a Latin Hypercube sampling method to generate random numbers within the assigned parameter distributions. The forecast is calculated with its own distribution from a set of, *e.g.* 5,000 - 10,000 simulations from which descriptive statistics can be interpreted. Also the sensitivity of the forecast to the different parameters can be analysed.

4 Application of methodologies to example sites

Decisions on the introduction of remedial measures in long-lasting exposure situations can often be limited to considerations of whether or not any of the possible remedial actions will result in a net benefit. If so, the optimum measure can be taken as the one having the largest net benefit. In reaching such decisions it is important to consider carefully the benefits and disadvantages because some remedial actions can significantly disrupt the exposed population. The analysis should address both radiological and non-radiological issues. Examination of the first of these will, in principle, be straightforward since it involves only the radiation detriment to be averted and the costs associated with the remedial action (including both the direct cost of the action and costs to affected parties). Examination of the second class of issues will involve, in addition to consideration of other hazards (such as those associated with chemical contaminants), economic and social considerations, some of which are beyond the scope of radiation protection. If it is determined that some remediation *is* justified on either of the above grounds then the next step is to optimise the proposed remedial action.

Most decisions require multiple attributes (criteria) to be taken into account. The field of multiple attribute analysis offers a number of approaches to provide structure and support to the decision-making process. In case of restoration of contaminated sites there are several attributes that need to be considered when choosing an 'optimum' restoration strategy. The attributes that has been considered in this study include:

- *Health attributes*
  - collective doses to population
  - doses to remediation workers
  - non-radiological health factors
- *Economic attributes*
  - costs of remedial actions (incl. costs of labour and monitoring)
  - costs of monitoring of remedial options
  - costs of disposal of generated waste (in broad categories)
  - loss/gain of taxes due to loss/gain of income
- *Social attributes*
  - reassurance of the public
  - discomfort, disturbance and anxiety from the remedial action
  - loss/gain of income

Model calculations would form the basis for determining whether to carry out remedial actions and to optimise such actions, subject to any constraints, for protection of individuals that otherwise would be exposed. The attribute hierarchy to be used for selection of an optimum restoration strategy can be structured as shown in Figure 9.

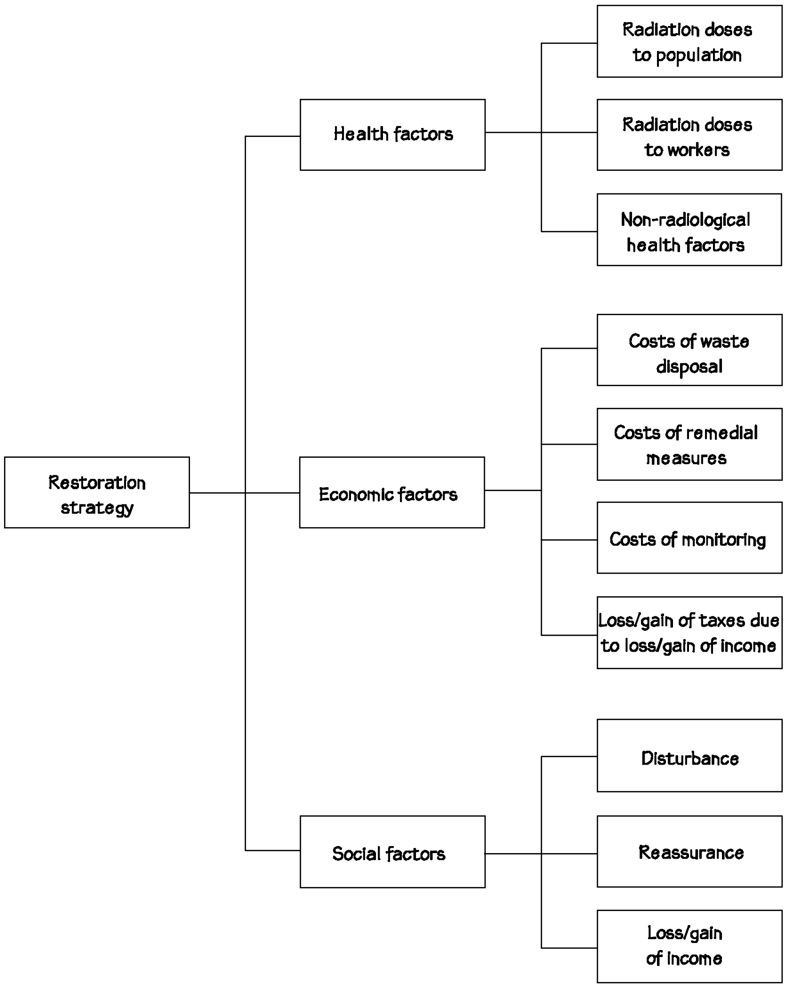


Figure 9. Attribute hierarchy for restoration of a contaminated site.

The major attributes as shown in Figure 9 are *radiation induced health effects*, *monetary costs* and *social costs* and each of these attributes are divided into sub-attributes.

A utility, u, or utility function, $u(x)$, will express the score or utility of a given attribute with value, x, for a given protective option. A risk neutral utility function can in general terms be defined as:

$$u(x) = 100 \cdot \left(1 + \frac{x_{\min} - x}{x_{\max} - x_{\min}}\right)$$

where ($x_{\min}$; $x_{\max}$) is the value range of the attribute considered.

The utilities, u, and weighting factors, w, (see Annex B) will determine the best (optimised) strategy or option amongst a set of strategies or options, i, expressed by the overall score, $U_i(x)$, which has its maximum value at the optimum:

$$U_i(x) = \sum_{j=1}^{n} w_j \cdot u_{ij}(x) \quad \text{and} \quad U_{opt} = \max \left\langle U_i(x) \right\rangle$$

The monetary costs, X, of each of the remediation options together with the averted collective doses, ΔS, for the affected population and the collective dose, S_{work}, to the workers implementing the remedial measures will determine the net benefit, ΔB, of the measures:

$$\Delta B = \alpha \cdot \Delta S - \left[\alpha \cdot S_{work} + \sum X\right] > 0$$

which should be positive for the option to be justified. All options with a positive net benefit are therefore justified on economical grounds the optimised option being the one with the largest net benefit.

4.1 Molse Nete River site

Since 1956, controlled releases of low-level radioactive effluents have been made from nuclear facilities in the region of Mol in the north-eastern part of Belgium. The Molse Nete River has been contaminated with the radionuclides ^{60}Co, ^{137}Cs, ^{239}Pu, and ^{241}Am as a result of these discharges into the river. The riverbanks have been contaminated through dredging of bed sediment out of the river. Subsequently, agricultural soils have also been contaminated through the application of the dredged sediment onto agricultural land for the purpose of soil amendment.

The following restoration options for the reduction of population doses have been identified [6]:

Discharges into the river stopped (agricultural use still possible)

A. No remediation

Discharges into the river stopped + Removal of sources (agricultural use still possible)

B. Soil/sediment removal

Discharges into the river stopped + Separation (agricultural use still possible)

C1. Physical separation (soil washing of soil/sediment)

D1. Chemical separation (chemical solubilisation)

Discharges into the river stopped + Containment (agricultural use no longer possible)

E1. Capping soil/sediment

Discharges into the river stopped + Immobilisation (agricultural use no longer possible)

F1. Physical immobilisation, *ex-situ*

F2. Physical immobilisation, *in-situ*

G1. Chemical immobilisation, *ex-situ*

G2. Chemical immobilisation, *in-situ*

4.1.1 Cost of restoration strategies

The following restoration components have been identified for the different remedial options. The monetary costs of these strategies include the costs of soil/sediment removal (including labour costs), waste disposal, loss of taxes and monitoring after or without (for option A) restoration. The costs of the different components are summarised below.

<u>Option A: No remediation</u>

- Monitoring costs of 32 kEUR$\cdot$a^{-1} in 100 years: 3,200 kEUR

<u>Option B: Soil and sediment removal</u>

- Removal of 6,120 m^3 agricultural soil; removal of 10,200 m^3 soil on river banks; removal of 10,200 m^3 bed sediment
- Excavation and transport costs for 16,320 m^3 soil: 2,040 kEUR
- Excavation and transport costs for 10,200 m^3 sediment: 1,530 kEUR
- Waste disposal and transport costs for 16,320 m^3 soil: 11,420 kEUR

- Waste disposal and transport costs for 10,200 m^3 sediment: 8,160 kEUR
- Monitoring costs of 10 $kEUR{\cdot}a^{-1}$ in 100 years: 1,000 kEUR

Option C1: Physical separation by soil washing

- Excavation and transport costs for 16,320 m^3 soil and 10,200 m^3 sediment prior to treatment: 3,570 kEUR
- Waste disposal and transport costs of contaminated fraction (5,300 m^3): 13,260 kEUR
- Washing costs for 26,520 m^3 soil and sediment (incl. labour costs): 9,300 kEUR
- Monitoring costs of 20 $kEUR{\cdot}a^{-1}$ in 100 years: 2,000 kEUR

Option D1: Chemical separation

- Excavation and transport costs for 16,320 m^3 soil and 10,200 m^3 sediment prior to treatment: 3,570 kEUR
- Waste disposal and transport costs of contaminated fraction (5,300 m^3): 13,260 kEUR
- Separation costs for 26,520 m^3 soil and sediment (incl. labour costs): 10,600 kEUR
- Monitoring costs of 20 $kEUR{\cdot}a^{-1}$ in 100 years: 2,000 kEUR

Option E1: Capping

- Capping of 34,000 m^2 agricultural soil surface, 34,000 m^2 river bank surface and 34,000 m^2 river bed surface
- Costs of capping of 68,000 m^2 soil surfaces: 2,720 kEUR
- Costs of capping of 34,000 m^2 river bed sediment: 1,530 kEUR
- Loss of taxes due to loss of agricultural production at 34,000 m^2: 1,360 kEUR (100 years)
- Monitoring costs of 32 $kEUR{\cdot}a^{-1}$ in 100 years: 3,200 kEUR

Option F1: Physical immobilisation *ex-situ*

- Excavation and transport costs for 16,320 m^3 soil and 10,200 m^3 sediment prior to treatment: 3,570 kEUR
- Costs of immobilisation of 26,520 m^3 soil and sediments (incl. labour costs): 2,650 kEUR
- Loss of taxes due to loss of agricultural production at 34,000 m^2: 1,360 kEUR (100 years)
- Monitoring costs of 32 $kEUR{\cdot}a^{-1}$ in 100 years: 3,200 kEUR

Option F2: Physical immobilisation *in-situ*

- Costs of immobilisation of 16,320 m^3 soil (incl. labour costs): 3,260 kEUR
- Costs of immobilisation of 10,200 m^3 sediments (incl. labour costs): 2,550 kEUR
- Loss of taxes due to loss of agricultural production at 34,000 m^2: 1,360 kEUR (100 years)
- Monitoring costs of 32 $kEUR{\cdot}a^{-1}$ in 100 years: 3,200 kEUR

Option G1: Chemical immobilisation *ex-situ*

- Excavation and transport costs for 26,520 m^3 soil prior to treatment: 3,570 kEUR
- Costs of immobilisation of 26,520 m^3 soil and sediments (incl. labour costs): 4,770 kEUR
- Loss of taxes due to loss of agricultural production at 34,000 m^2: 1,360 kEUR (100 years)
- Monitoring costs of 32 $kEUR{\cdot}a^{-1}$ in 100 years: 3,200 kEUR

Option G2: Chemical immobilisation *in-situ*

- Costs of immobilisation of 16,320 m^3 soil (incl. labour costs): 3,260 kEUR
- Costs of immobilisation of 10,200 m^3 sediments (incl. labour costs): 2,550 kEUR
- Loss of taxes due to loss of agricultural production at 34,000 m^2: 1,360 kEUR (100 years)
- Monitoring costs of 32 kEUR·a^{-1} in 100 years: 3,200 kEUR

4.1.2 Justified restoration strategies

The economic and radiological data for remediation of the Molse Nete River are shown in Table 4. The monetary costs, *X*, of the remediation strategies can be compared to the benefit of the collective dose reduction, ΔS. The net benefit, ΔB, is given as:

$$\Delta B = \alpha \cdot \Delta S - (\alpha \cdot S_{work} + X_{remedia} + X_{waste} + X_{tax} + X_{monitor})$$

None of the remedial options are justified on economical grounds alone when only the central estimates of collective dose are used together with an α-value of 100,000 EUR·manSv^{-1} [14]. A higher value of α (*e.g.* 200,000 EUR·manSv^{-1}) and more extreme values from the calculated collective dose distribution (*e.g.* the 95th percentile (see TD6)) would make the options E1, F1, F2, G1 and G2 economically justified when the avertable collective dose is taken over 100 years. Similarly, the options B, E1, F1, F2, G1 and G2 would be economically justified for an integration time of 500 years and more extreme values of the collective doses. It should be emphasized that the dose assessments are based on conservative assumptions concerning the habits of the affected population and the usage of the contaminated sediments. More realistic assumptions would have resulted in much lower doses.

Table 4. Remediation costs and collective doses to population and workers for different restoration strategies at the Molse Nete River site.

Restoration strategy	Collective dose to population [man Sv]		Collective dose to workers [man·Sv]	Monetary costs of restoration [kEUR]					Fraction of activity left on-site	Waste volume (m^3)
	100 y	500 y		Remediation	Monitoring	Waste disposal	Inc. loss	Tax loss		
A	16	51	0	0	3,200	0	0	0	1	0
B	1.6	5.1	$6.1 \cdot 10^{-4}$	3,570	1,000	19,580	0	0	0.1	26,520
C1	4.5	14	$1.8 \cdot 10^{-3}$	12,870	2,000	13,260	0	0	0.3	5,300
D1	1.6	5.1	$1.6 \cdot 10^{-3}$	13,970	2,000	13,260	0	0	0.1	10,600
E1	negli.	negli.	$2.6 \cdot 10^{-3}$	4,250	3,200	0	68	1,360	1	0
F1	negli.	negli.	$6.7 \cdot 10^{-3}$	6,220	3,200	0	68	1,360	1	0
F2	negli.	negli.	$1.8 \cdot 10^{-3}$	5,810	3,200	0	68	1,360	1	0
G1	negli.	negli.	$6.7 \cdot 10^{-3}$	8,340	3,200	0	68	1,360	1	0
G2	negli.	negli.	$1.8 \cdot 10^{-3}$	5,810	3,200	0	68	1,360	1	0

The individual doses would in average be of the order of 600 µSv·a^{-1} at the time of decision to introduce remediation (year 1). IAEA has proposed clean-up criteria in terms of individual dose [16]. For an individual dose range of 0.1 - 1 mSv·a^{-1} clean-up is usually needed if a constraint for controlled practices is applied. Even without the application of a constraint IAEA suggests that for individual doses of 0.1 - 1 $mSv \cdot a^{-1}$ clean-up might sometimes be needed. Based on these recommendations it can therefore be concluded that some kind of remediation would be justified for the Molse Nete River site.

4.1.3 Optimised restoration strategies

Utility functions for the attributes *monetary costs* and *radiation doses* have been calculated from the figures in Table 4 on monetary cost components and residual collective doses after remediation. Linear (risk neutral) utility functions have been used.

Utility functions for monetary costs

Utility functions have been determined for remediation costs (including labour costs), waste disposal costs (including transport costs), monitoring costs and loss of taxes due to loss of income:

$$u_{\mathrm{remedia}}(x) = 100 \cdot \left(1 - \frac{x}{13{,}970}\right) \quad \text{for } 0 \le x \le 13{,}970 \text{ kEUR}$$

$$u_{waste}(x) = 100 \cdot \left(1 - \frac{x}{19{,}580}\right) \quad \text{for } 0 \le x \le 19{,}580 \text{ kEUR}$$

$$u_{tax}(x) = 100 \cdot \left(1 - \frac{x}{1{,}360}\right) \quad \text{for } 0 \le x \le 1{,}360 \text{ kEUR over } 100 \text{ y}$$

$$u_{monitor}(x) = 100 \cdot \left(1 - \frac{x}{3{,}200}\right) \quad \text{for } 0 \le \mathrm{x} \le 1{,}000 \text{ kEUR over } 100 \text{ y}$$

Utility functions for health factors

The following utility functions for the radiological health components have been determined for the exposed population and workers implementing the remedial actions. Only radiological health factors are considered for the Molse Nete River site as no heavy metals are found.

$$u_{\mathrm{dose,\,pop,100}}(x) = 100 \cdot \left(1 - \frac{x}{16}\right) \text{ for } 0 \le x \le 16 \text{ man Sv}$$

$$u_{\mathrm{dose,\,pop,500}}(x) = 100 \cdot \left(1 - \frac{x}{51}\right) \quad \text{for } 0 \le x \le 51 \text{ man Sv}$$

$$u_{\mathrm{dose,\,work}}(x) = 100 \cdot \left(1 - \frac{x}{0.0067}\right) \quad \text{for } 0 \le x \le 0.0067 \text{ man Sv}$$

Utility functions for social factors

The utility function u_{reas} for reassurance would be linked to both the residual dose and the fraction of activity remaining on the site after the remedial measure has been implemented. However, the residual dose and remaining activity are not necessarily correlated. A remedial measure that has left all the activity on site in a contained form (capping, surface barriers etc.) might give a substantial dose reduction and thus a low value of the residual doses. Detailed information on how social factors like reassurance are linked with individual doses and activity concentration on site is not available. Therefore, utility functions for 100 years and 500 years integration time have been proposed which gives a low value only when both sub-utilities have low values:

$$u_{\mathrm{reas,100}}(x,y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 - \frac{x}{16}\right)_{\mathrm{dose}} + \frac{1}{2} \cdot \left(1 + \frac{0.1 - y}{1.0 - 0.1}\right)_{\mathrm{activity}}\right)$$

$$\text{for } 0 \le x \le 16 \text{ man Sv and } 0.1 \le \mathrm{y} \le 1$$

$$u_{\text{reas},500}(x,y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 - \frac{x}{51}\right)_{\text{dose}} + \frac{1}{2} \cdot \left(1 + \frac{0.1 - y}{1.0 - 0.1}\right)_{\text{activity}} \right)$$
$$\text{for } 0 \le x \le 51 \text{ man Sv and } 0.1 \le y \le 1$$

where y is the fraction of activity remaining on site after the remedial measures has been implemented. The value of the utility function u_{reas} will be 100 for a residual dose of 0 man Sv and a remaining fraction of the initial activity of 0.1 (best strategy) and 0 for a residual dose of 16 (51) man Sv and a remaining activity fraction of 1.0 (worst strategy).

The utility function u_{distur} for disturbance has been related to the volume of soil and sediment waste to be transported to the waste disposal site:

$$u_{distur}(x) = 100 \cdot \left(1 - \frac{x}{26{,}520}\right) \quad \text{for } 0 \le x \le 26{,}520 \text{ m}^3$$

For the remedial option B the waste volume is 26,520 m^3, for the option C1 it is 5,300 m^3 and for the option D1 it is 10,600 m^3. No waste is produced for all other options.

The utility function u_{loss} for loss of income due to loss of agricultural production facilities can be determined from the specific agricultural production pattern per unit area weighted with the market price of the production. The income loss has been determined to be about 1 EUR·m^{-2}·a^{-1}. It is, however, very likely that the farmers soon would find other income. The loss is therefore assumed to last only for two years, which will give the following utility function:

$$u_{loss}(x) = 100 \cdot \left(1 - \frac{x}{68}\right) \quad \text{for } 0 \le x \le 68 \text{ kEUR over 2 years}$$

Weighting factors for major attributes

The major weighting factors considered in this study include those for monetary costs, health and social factors as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{health} + w_{economic} + w_{social} = 1$$

The assessment of the weighting factors is discussed in Annex B where conversion/scaling constants between weighting factors for the major attributes has been expressed as:

$$C_1 = \frac{w_{economic}}{w_{health}} \cong \frac{w_{economic}}{w_{dose,pop}} = \frac{R_{economic}}{\alpha \cdot R_{dose,pop}}$$

$$C_2 = \frac{w_{social}}{w_{health}} \approx \frac{r_{psy}}{r_{rad}}$$

C_1 can be determined for a 100 and 500 years integration time for the collective dose from the ranges of monetary costs and collective doses given in Table 4:

$$C_{1,100} = \frac{(13{,}970 + 2{,}000 + 13{,}260) - 3{,}200}{100{,}000 \cdot (16 - 0) \cdot 10^{-3}} = \frac{26{,}030}{1{,}600}$$

$$C_{1,500} = \frac{(13{,}970 + 2{,}000 + 13{,}260) - 3{,}200}{100{,}000 \cdot (51 - 0) \cdot 10^{-3}} = \frac{26{,}030}{5{,}100}$$

The value of C_2 is more difficult to assess but a value of 0.2 - 0.3 has been argued for in Annex B. The weighting factors can be calculated from the scaling constants C to be:

$$w_{health,100} = \frac{1}{1+\frac{26{,}030}{1{,}600}+0.25} = \underline{0.057} \quad \text{and} \quad w_{health,500} = \frac{1}{1+\frac{26{,}030}{5{,}100}+0.25} = \underline{0.157}$$

$$w_{economic,100} = \frac{\frac{26{,}030}{1{,}600}}{1+\frac{26{,}030}{1{,}600}+0.25} = \underline{0.929} \quad \text{and} \quad w_{economic,500} = \frac{\frac{26{,}030}{5{,}100}}{1+\frac{26{,}030}{5{,}100}+0.25} = \underline{0.803}$$

$$w_{social,100} = \frac{0.25}{1+\frac{26{,}030}{1{,}600}+0.25} = \underline{0.014} \quad \text{and} \quad w_{social,500} = \frac{0.25}{1+\frac{26{,}030}{5{,}100}+0.25} = \underline{0.039}$$

Weighting factors for health sub-attributes

The weighting factors for health sub-attributes include those of radiation induced stochastic health effects to the affected population and workers and non-radiation induced stochastic health effects to the affected population as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{dose,pop} + w_{dose,work} + w_{non-rad} = 1$$

The conversion/scaling constant, C, for the health attributes can according to Annex B be expressed as:

$$w_{dose,pop} = C \cdot R_{dose,pop} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$

$$w_{dose,work} = C \cdot R_{dose,work} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$

$$w_{non-rad,pop} = C \cdot R_{non-rad,pop} \cdot l \cdot r_{non-rad}$$

Exposure to heavy metals is not relevant for the Molse Nete River site and $R_{non\text{-}rad}$ is therefore zero. The value of C can be determined from the collective dose ranges, R, given in Table 4 as:

$$C_{100} = \frac{1}{(16-0)+(0.0067-0)} = 6.25 \cdot 10^{-2}$$

$$C_{500} = \frac{1}{(51-0)+(0.0067-0)} = 1.96 \cdot 10^{-2}$$

The weighting factors can be determined from the scaling constant C to be:

$$w_{dose,pop,100} = 6.25 \cdot 10^{-2} \cdot (16-0) \cong \underline{1}$$

$$w_{dose,work,100} = 6.25 \cdot 10^{-2} \cdot (0.0067-0) \cong \underline{0}$$

$$w_{dose,pop,500} = 1.96 \cdot 10^{-2} \cdot (51-0) \cong \underline{1}$$

$$w_{dose,work,500} = 1.96 \cdot 10^{-2} \cdot (0.0067-0) \cong \underline{0}$$

Weighting factors for economic sub-attributes

The weighting factors for economic sub-attributes include those for cost of remediation, cost of waste disposal, costs of monitoring and loss/gain of taxes as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{remedia} + w_{waste} + w_{monitor} + w_{tax} = 1$$

The conversion/scaling constant, C, for the economic attributes can according to Annex B be expressed as:

$$w_{remedia} = C \cdot R_{remedia}$$

$$w_{waste} = C \cdot R_{waste}$$

$$w_{monitor} = C \cdot R_{monitor}$$

$$w_{tax} = C \cdot R_{tax}$$

The conversion/scaling constant, C, for the economic sub-attributes can be determined from the cost ranges in Table 4:

$$\mathrm{C} = \frac{1}{(13{,}970 - 0) + (19{,}580 - 0) + (3{,}200 - 0) + (1{,}360 - 0)} = 2.62 \cdot 10^{-5}$$

The weighting factors can be calculated from the scaling constant C to be:

$$w_{remedia} = 2.62 \cdot 10^{-5} \cdot (13{,}970 - 0) = \underline{0.37}$$
$$w_{waste} = 2.62 \cdot 10^{-5} \cdot (19{,}580 - 0) = \underline{0.51}$$
$$w_{monitor} = 2.62 \cdot 10^{-5} \cdot (3{,}200 - 0) = \underline{0.084}$$
$$w_{tax} = 2.62 \cdot 10^{-5} \cdot (1{,}360 - 0) = \underline{0.036}$$

Weighting factors for social sub-attributes

The weighting factors include those for reassurance, disturbance and loss/gain of taxes as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{distur} + w_{reas} + w_{loss} = 1$$

The conversion/scaling constants for the social sub-attributes can according to Annex B be expressed as:

$$C_1 = \frac{w_{reas}}{w_{distur}} \quad \text{and} \quad C_2 = \frac{w_{loss}}{w_{distur}}$$

In Annex B it is argued that $w_{reas} > w_{loss} > w_{distur}$ and that $C_1 \approx 5$ - 7 and $C_2 \approx 2$ - 3. From these values the weighting factors can be calculated as:

$$w_{distur} = \frac{1}{1 + 6 + 2.5} \cong \underline{0.11}$$

$$w_{reas} = \frac{6}{1 + 6 + 2.5} \cong \underline{0.63}$$

$$w_{loss} = \frac{2.5}{1 + 6 + 2.5} \cong \underline{0.26}$$

The calculated values of the weighting factors for each of the attributes and sub-attributes are shown in Table 5.

Table 5. Weighting factors for attributes and sub-attributes applied in the optimisation of remediation of the Molse Nete River site. The values in the left of the double columns are for an integration time of 100 years and in the right column for an integration time of 500 years.

Health factors		Economic factors		Social factors	
100 years	500 years	100 years	500 years	100 years	500 years
0.057	0.157	0.929	0.803	0.014	0.039

	100 years	500 years	Remediation costs	0.37	Reassurance	0.63
Dose population	1	1				
Dose workers	0	0	Waste disposal costs	0.51	Disturbance	0.11
Non-radiation	-	-	Monitoring costs	0.084	Loss/gain of income	0.26
			Loss/gain of taxes	0.036		

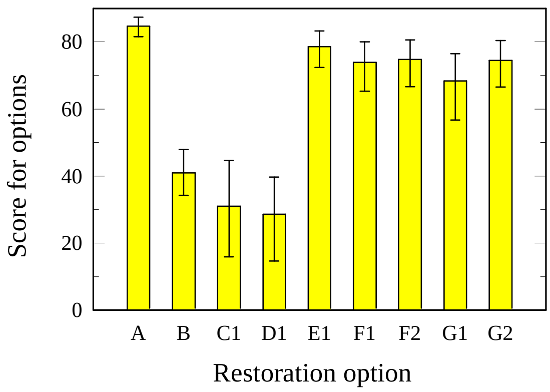
It should be emphasized that value setting of weighting factors is the crucial issue of any optimisation because subjective judgements inevitably will enter the process.

Scores for remediation options

The overall scores, U_i, of the remediation options i has been determined from the weighted sum of utilities for each of the attributes considered as shown in Figure 9:

$$\begin{aligned} U_i &= \sum_{j=1}^{3} w_j \cdot u_{ij} \\ &= w_{health} \cdot (w_{dose,pop} \cdot u_{dose,pop} + w_{dose,work} \cdot u_{dose,work}) \\ &+ w_{economic} \cdot (w_{waste} \cdot u_{waste} + w_{remedia} \cdot u_{remedia} + w_{monitor} \cdot u_{monitor} + w_{tax} \cdot u_{tax}) \\ &+ w_{social} \cdot (w_{distur} \cdot u_{distur} + w_{reas} \cdot u_{reas} + w_{loss} \cdot u_{loss}) \end{aligned}$$

The weighting factors above have all been sampled in a triangular distribution between 1.5^{-1} - 1.5 times the most probable value given in Table 5. Similarly, the values of all the utilities, $u(x)$, are determined from the utility functions in which the values of x are sampled in a triangular distribution between 1.5^{-1} - 1.5 times the central values of x given in Table 4. Negative correlation between collective doses and remediation costs has been applied with a correlation coefficient of –0.8. The evaluation of the different strategies has been made with the forecasting and risk analysis program CRYSTAL BALL [4]. Latin Hypercube Sampling technique was used and the number of trials was 10,000. The results for the scores, U_i, for the options A - G2 are shown in Figure 10. The error bars represent the 5% and 95% percentiles of the distributions of U_i.



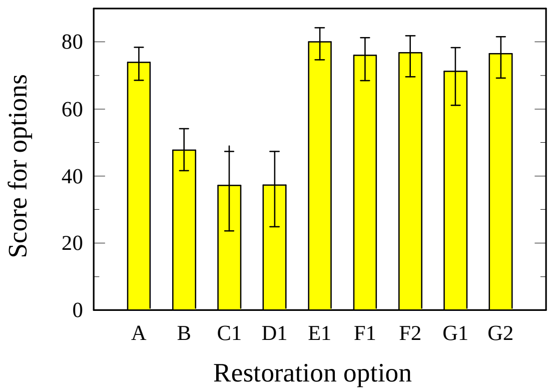


Figure 10. Overall evaluation of scores for different remediation strategies for the Molse Nete River site. The left picture shows the results for an integration time of 100 years for the collective dose and the right picture for an integration time of 500 years.

It appears from Figure 10 that there is practically no difference between the scores for an integration time of 100 and 500 years due to the low weight of the health attributes although the score for option A is somewhat lower for the longer integration time. The options A, E1, F1, F2, G1 and G2 have practically an equal score which makes it rather difficult to distinguish which is the optimum. For a 500-years integration time the option E1, capping, has the highest score, and this option can therefore be considered as the optimum.

4.2 Drigg site

The Drigg site is situated in West Cumbria about nine km south of Sellafield in the UK on the coast of the Irish Sea. The site is placed just west of the village of Drigg, 300 meters north of the tidal estuary of the River Esk. Since 1959 the site has been used for the disposal of low-level radioactive waste. It is operated by British Nuclear Fuel plc (BFNL) for the shallow burial of solid waste, mostly from the Sellafield site. Several small streams cross the site. The dominating radionuclides giving rise to the low doses to the local population, mainly from milk consumption, are ^{137}Cs, ^{238}U,^{239}Pu, and ^{241}Am.

The following restoration options to reduce population doses have been identified [7]:

- *A. No remediation*
- *C2. Filtration*
- *D1. Chemical Solubilisation*
- *D2. Ion Exchange*
- *D3. Bio-sorption*
- *E1. Capping*
- *E3. Sub-surface Barrier*
- *F1. Physical Immobilisation, ex-situ*
- *F2. Physical Immobilisation, in-situ*
- *G1. Chemical Immobilisation, ex-situ*
- *G2. Chemical Immobilisation, in-situ*

4.2.1 Cost of restoration strategies

The restoration measures fall into two categories, those, which treat solutions, and those, which treat solids. Water treatment will be an on-going process and is assumed to last 500 years. The monetary costs of the different restoration options include remediation costs, waste disposal costs and monitoring costs. The costs of the different options are summarised below.

<u>Option A: No remediation</u>

- Monitoring costs: 100,000 £·a^{-1} corresponding to 75,000 kEUR in 500 years

<u>Option C2: Filtration</u>

- Remediation of a total volume of water of about $5 \cdot 10^8$ m^3 from the Drigg stream; processing of contaminated suspended solids; with a 100% efficient filtration the amount will be 40,000 kg·a^{-1} corresponding to about 25 m^3 a^{-1}; costs over 500 years (incl. labour costs): 380,000 kEUR
- Costs of removal and disposal of 12,500 m^3 disposable waste: 31,000 kEUR
- Monitoring costs: 10,000 £·a^{-1} corresponding to 750 kEUR in 500 years

Option D1: Chemical Solubilisation

- Costs of excavation, transport and treatment off-site of $5.5 \cdot 10^5$ m^3 solid waste from the Drigg Trenches: 300,000 kEUR
- Waste disposal costs for 41,000 m^3 waste solution: 100,000 kEUR
- Monitoring costs: 10,000 £·a^{-1} corresponding to 7,500 kEUR in 500 years

Option D2: Ion Exchange

- Costs of removal of 12,500 m^3 solid material by filtration and ion exchange of $5 \cdot 10^8$ m^3 liquid: 1,000 MEUR
- Costs of removal and disposal of 12,500 m^3 disposable waste: 31,000 kEUR
- Monitoring costs: 20,000 £·a^{-1} corresponding to 15,000 kEUR in 500 years

Option D3: Biosorption

- Costs of removal of 12,500 m^3 solid material by filtration and biosorption processing of $5 \cdot 10^8$ m^3 liquid: 1,300 MEUR
- Costs of removal and disposal of 12,500 m^3 disposable waste: 31,000 kEUR
- Monitoring costs: 10,000 £·a^{-1} corresponding to 7,500 kEUR in 500 years

Option E1: Capping

- Costs of capping an surface area of 10^5 m^2: 3,500 kEUR
- Monitoring costs: 100,000 £·a^{-1} corresponding to 75,000 kEUR in 500 years

Option E3: Sub-surface Barrier

- Costs of establishing a grout curtain with a depth of 10 m: 6,300 kEUR
- Monitoring costs: 100,000 £·a^{-1} corresponding to 75,000 kEUR in 500 years

Option F1: Physical Immobilisation (*ex-situ*)

- Costs of excavation, transport and immobilisation of $5.5 \cdot 10^5$ m^3 solid waste: 55,000 kEUR
- Monitoring costs: 100,000 £·a^{-1} corresponding to 75,000 kEUR in 500 years

Option F2: Physical Immobilisation (*in-situ*)

- Costs of immobilisation of $5.5 \cdot 10^5$ m^3 solid waste: 190,000 kEUR
- Monitoring costs: 100,000 £·a^{-1} corresponding to 75,000 kEUR in 500 years

Option G1: Chemical Immobilisation (*ex-situ*)

- Costs of excavation, transport and immobilisation of $5.5 \cdot 10^5$ m^3 solid waste: 130,000 kEUR
- Monitoring costs: 100,000 £·a^{-1} corresponding to 75,000 kEUR in 500 years

Option G2: Chemical Immobilisation (*in-situ*)

- Costs of immobilisation of $5.5 \cdot 10^5$ m^3 solid waste: 55,000 kEUR
- Monitoring costs: 100,000 £·a^{-1} corresponding to 75,000 kEUR in 500 years

4.2.2 Justified restoration strategies

The economic and radiological data for remediation of the Drigg site are shown in Table 6. The monetary costs, *X*, of the remediation strategies can be compared to the benefit of the collective dose reduction, ΔS. The net benefit, ΔB, is given as:

$$\Delta B = \alpha \cdot \Delta S - (\alpha \cdot S_{work} + X_{remedia} + X_{waste} + X_{monitor})$$

None of the remedial options are justified on economic grounds alone when only the central estimates of collective dose are used together with an α-value of 100,000 EUR·manSv^{-1} [14]. A higher value of α (*e.g.* 200,000 EUR·manSv^{-1}) and more extreme values from the calculated collective dose distribution (*e.g.* the 95th percentile) would make the options E1 and E3 economically justified, but only when the avertable collective dose is taken over 500 years.

Table 6. Remediation costs and collective doses to population and workers for different restoration strategies at the Drigg site.

Restoration strategy	Collective dose to population [man Sv]		Collective dose to workers [man·Sv]	Monetary costs of restoration [kEUR]			Fraction of activity left on-site	Waste volume (m^3)
	100 y	500 y		Remediation	Monitoring	Waste disposal		
A	49	120	0	0	75,000	0	1	0
C2	0.93	3.3	$1.5 \cdot 10^{-9}$	380,000	750	31,000	0.01	12,500
D1	9.9	33	$1.7 \cdot 10^{-9}$	300,000	7,500	100,000	0.1	41,000
D2	16	51	$3.7 \cdot 10^{-10}$	1,000,000	15,000	31,000	0.2	12,500
D3	13	42	$7.1 \cdot 10^{-10}$	1,300,000	7,500	31,000	0.1	12,500
E1	0.43	1.9	$5.5 \cdot 10^{-12}$	3,500	75,000	0	1	0
E3	2.9	11	$6.9 \cdot 10^{-10}$	6,300	75,000	0	1	0
F1	4.2	10	$2.8 \cdot 10^{-9}$	55,000	75,000	0	1	0
F2	4.2	10	$1.4 \cdot 10^{-9}$	190,000	75,000	0	1	0
G1	2.9	7.2	$1.9 \cdot 10^{-9}$	130,000	75,000	0	1	0
G2	2.9	7.2	$9.4 \cdot 10^{-10}$	55,000	75,000	0	1	0

The individual doses would be of the order of 1,300 μSv·a^{-1} at the time of decision to introduce remediation (year 1). IAEA has proposed clean-up criteria in terms of individual dose [16]. For an individual dose range of 1 - 10 mSv·a^{-1} clean-up is almost always needed if a constraint for controlled practices is applied. Even without the application of a constraint IAEA suggests that for individual doses of 1 - 10 mSv·a^{-1} clean-up would usually be needed. Based on these recommendations it can therefore be concluded that some kind of remediation would almost always be justified for the Drigg site.

4.2.3 Optimised restoration strategies

Utility functions for the attributes *monetary costs* and *radiation doses* have been calculated from the figures in Table 6 on monetary cost components and residual collective doses after remediation. Linear (risk neutral) utility functions have been used.

Utility functions for monetary costs

Utility functions have been determined for remediation costs (including labour costs), waste disposal costs (including transport costs) and monitoring costs:

$$u_{remedia}(x) = 100 \cdot \left(1 - \frac{x}{1{,}300{,}000}\right) \quad \text{for } 0 \le x \le 1{,}300{,}000 \text{ kEUR}$$

$$u_{waste}(x) = 100 \cdot \left(1 - \frac{x}{100{,}000}\right) \quad \text{for } 0 \le x \le 100{,}000 \text{ kEUR}$$

$$u_{monitor}(x) = 100 \cdot \left(1 + \frac{750 - x}{75{,}000 - 750}\right) \quad \text{for } 750 \le \text{x} \le 75{,}000 \text{ kEUR over } 500 \text{ y}$$

Utility functions for health factors

The following utility functions for the radiological health components have been determined for the exposed population and workers implementing the remedial actions. Only radiological health factors are considered for the Drigg site as no heavy metals are found.

$$u_{dose,\,pop,100}(x) = 100 \cdot \left(1 + \frac{0.43 - x}{49 - 0.43}\right) \text{ for } \; 0.43 \le x \le 49 \text{ man Sv}$$

$$u_{dose,\,pop,500}(x) = 100 \cdot \left(1 + \frac{1.9 - x}{120 - 1.9}\right) \quad \text{for } 1.9 \le x \le 120 \text{ man Sv}$$

$$u_{dose,\,work}(x) = 100 \cdot \left(1 - \frac{x}{2.8 \cdot 10^{-9}}\right) \quad \text{for } 0 \le x \le 2.8 \cdot 10^{-9} \text{ man Sv}$$

Utility functions for social factors

The utility function u_{reas} for reassurance would be linked to both the residual dose and the fraction of activity remaining on the site after the remedial measure has been implemented. However, the residual dose and remaining activity are not necessarily correlated. A remedial measure that has left all the activity on site in a contained form (capping, surface barriers etc.) might give a substantial dose reduction and thus a low value of the residual doses. Detailed information on how social factors like reassurance are linked with individual doses and activity concentration on site is not available. Therefore, utility functions for 100 years and 500 years integration time have been proposed which gives a low value only when both sub-utilities have low values:

$$u_{reas,100}(x,y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 + \frac{0.43 - x}{49 - 0.43}\right)_{\text{dose}} + \frac{1}{2} \cdot \left(1 + \frac{0.01 - y}{1.0 - 0.01}\right)_{\text{activity}}\right)$$

$$\text{for } 0.43 \le x \le 49 \text{ man Sv and } 0.01 \le \text{y} \le 1$$

$$u_{reas,500}(x,y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 + \frac{1.9 - x}{120 - 1.9}\right)_{\text{dose}} + \frac{1}{2} \cdot \left(1 + \frac{0.01 - y}{1.0 - 0.01}\right)_{\text{activity}}\right)$$

$$\text{for } 1.9 \le x \le 120 \text{ man Sv and } 0.01 \le y \le 1$$

where y is the fraction of activity remaining on site after the remedial measures has been implemented. The value of the utility function u_{reas} will be 100 for a residual dose of 0.43 (1.9) man Sv and a remaining fraction of the initial activity of 0.01 (best strategy) and 0 for a residual dose of 49 (120) man Sv and a remaining activity fraction of 1.0 (worst strategy).

The utility function u_{distur} for disturbance has been related to the volume of soil and sediment waste to be transported to the waste disposal site:

$$u_{distur}(x) = 100 \cdot \left(1 - \frac{x}{41{,}000}\right) \quad \text{for } 0 \le x \le 41{,}000 \text{ m}^3$$

Weighting factors for major attributes

The major weighting factors considered in this study include those for monetary costs, health and social factors as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{health} + w_{economic} + w_{social} = 1$$

The assessment of the weighting factors is discussed in Annex B where conversion/scaling constants between weighting factors has been expressed as:

$$C_1 = \frac{w_{economic}}{w_{health}} \cong \frac{w_{economic}}{w_{dose,pop}} = \frac{R_{economic}}{\alpha \cdot R_{dose,pop}}$$

$$C_2 = \frac{w_{social}}{w_{health}} \approx \frac{r_{psy}}{r_{rad}}$$

C_1 can be determined for a 100 and 500 years integration time for the collective dose from the ranges of monetary costs and collective doses given in Table 6:

$$C_{1,100} = \frac{(1{,}300{,}000 + 75{,}000 + 31{,}000) - 0}{100{,}000 \cdot (49 - 0.43) \cdot 10^{-3}} = \frac{1{,}406{,}000}{4{,}857}$$

$$C_{1,500} = \frac{(1{,}300{,}000 + 75{,}000 + 31{,}000) - 0}{100{,}000 \cdot (120 - 1.9) \cdot 10^{-3}} = \frac{1{,}406{,}000}{11{,}810}$$

The value of C_2 is more difficult to assess but a value of 0.2 - 0.3 has been argued for in Annex B. The weighting factors can be calculated from the scaling constants C to be:

$$w_{health,100} = \frac{1}{1 + \frac{1{,}406{,}000}{4{,}857} + 0.25} = \underline{3.4 \cdot 10^{-3}} \quad \text{and} \quad w_{health,500} = \frac{1}{1 + \frac{1{,}406{,}000}{11{,}810} + 0.25} = \underline{8.3 \cdot 10^{-3}}$$

$$w_{economic,100} = \frac{\frac{1{,}406{,}000}{4{,}857}}{1 + \frac{1{,}406{,}000}{4{,}857} + 0.25} = \underline{0.996} \quad \text{and} \quad w_{economic,500} = \frac{\frac{1{,}406{,}000}{11{,}810}}{1 + \frac{1{,}406{,}000}{11{,}810} + 0.25} = \underline{0.990}$$

$$w_{social,100} = \frac{0.25}{1 + \frac{1{,}406{,}000}{4{,}857} + 0.25} = \underline{8.6 \cdot 10^{-4}} \quad \text{and} \quad w_{social,500} = \frac{0.25}{1 + \frac{1{,}406{,}000}{11{,}810} + 0.25} = \underline{2.1 \cdot 10^{-3}}$$

Weighting factors for health sub-attributes

The weighting factors for health sub-attributes include those of radiation induced stochastic health effects to the affected population and workers and non-radiation induced stochastic health effects to the affected population as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{dose,pop} + w_{dose,work} + w_{non-rad} = 1$$

The conversion/scaling constant, C, for the health sub-attributes can according to Annex B be expressed as:

$$w_{dose,pop} = C \cdot R_{dose,pop} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$

$$w_{dose,work} = C \cdot R_{dose,work} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$

$$w_{non-rad,pop} = C \cdot R_{non-rad,pop} \cdot l \cdot r_{non-rad}$$

Exposure to heavy metals is not relevant for the Drigg site and $R_{non\text{-}rad}$ is therefore zero. The value of *C* can be determined from the collective dose ranges, *R*, given in Table 6 as:

$$C_{100} = \frac{1}{(49 - 0.43) + (5.6 \cdot 10^{-9} - 0)} = 2.06 \cdot 10^{-2}$$

$$C_{500} = \frac{1}{(120 - 1.9) + (5.6 \cdot 10^{-9} - 0)} = 8.47 \cdot 10^{-3}$$

The weighting factors can be determined from the scaling constant *C* to be:

$$w_{dose,pop,100} = 2.06 \cdot 10^{-2} \cdot (49 - 0.43) \cong \underline{1}$$
$$w_{dose,work,100} = 2.06 \cdot 10^{-2} \cdot (5.6 \cdot 10^{-9} - 0) \cong \underline{0}$$
$$w_{dose,pop,500} = 8.47 \cdot 10^{-3} \cdot (120 - 1.9) \cong \underline{1}$$
$$w_{dose,work,500} = 8.47 \cdot 10^{-3} \cdot (5.6 \cdot 10^{-9} - 0) \cong \underline{0}$$

Weighting factors for economic sub-attributes

The weighting factors for economic sub-attributes include those for cost of remediation, cost of waste disposal and costs of monitoring (no loss/gain of taxes) as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{remedia} + w_{waste} + w_{monitor} = 1$$

The conversion/scaling constant, *C*, for the economic attributes can according to Annex B be expressed as:

$$w_{remedia} = C \cdot R_{remedia}$$

$$w_{waste} = C \cdot R_{waste}$$

$$w_{monitor} = C \cdot R_{monitor}$$

The conversion/scaling constant, *C*, for the economic sub-attributes can be determined from the cost ranges in Table 6:

$$\mathrm{C} = \frac{1}{(1{,}300{,}000 - 0) + (100{,}000 - 0) + (75{,}000 - 750)} = 6.78 \cdot 10^{-7}$$

The weighting factors can be calculated from the scaling constant *C* to be:

$$w_{remedia} = 6.78 \cdot 10^{-7} \cdot (1{,}300{,}000 - 0) = \underline{0.882}$$
$$w_{waste} = 6.8 \cdot 10^{-7} \cdot (100{,}000 - 0) = \underline{0.068}$$
$$w_{monitor} = 6.8 \cdot 10^{-7} \cdot (75{,}000 - 750) = \underline{0.050}$$

Weighting factors for social sub-attributes

The weighting factors include those for reassurance and disturbance as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{distur} + w_{reas} = 1$$

The conversion/scaling constant for the social sub-attributes can according to Annex B be expressed as:

$$C_1 = \frac{w_{reas}}{w_{distur}}$$

In Annex B it is argued that $w_{reas} > w_{distur}$ and that $C_1 \approx 5 - 7$. From these values the weighting factors can be calculated as:

$$w_{distur} = \frac{1}{1+6} \cong \underline{0.14} \quad \text{and} \quad w_{reas} = \frac{6}{1+6} \cong \underline{0.86}$$

The calculated values of the weighting factors for each of the attributes and sub-attributes are shown in Table 7.

Table 7. Weighting factors for attributes and sub-attributes applied in the optimisation of remediation of the Drigg site. The values in the left of the double columns are for an integration time of 100 years and in the right column for an integration time of 500 years.

Health factors		Economic factors		Social factors	
100 years	500 years	100 years	500 years	100 years	500 years
0.0034	0.0083	0.996	0.990	0.00086	0.0021

	100 years	500 years				
Dose population	1	1	Remediation costs	0.882	Reassurance	0.86
Dose workers	0	0	Waste disposal costs	0.068	Disturbance	0.14
Non-radiation	-	-	Monitoring costs	0.050	Loss/gain of income	-
			Loss/gain of taxes	-		

Scores for remediation options

The overall scores, U_i, of the remediation options i has been determined from the weighted sum of utilities for each of the attributes considered as shown in Figure 9:

$$\begin{aligned} U_i &= \sum_{j=1}^{3} w_j \cdot u_{ij} \\ &= w_{health} \cdot (w_{dose,pop} \cdot u_{dose,pop} + w_{dose,work} \cdot u_{dose,work}) \\ &+ w_{economic} \cdot (w_{waste} \cdot u_{waste} + w_{remedia} \cdot u_{remedia} + w_{monitor} \cdot u_{monitor}) \\ &+ w_{social} \cdot (w_{distur} \cdot u_{distur} + w_{reas} \cdot u_{reas}) \end{aligned}$$

The weighting factors above have all been sampled in a triangular distribution between 1.5^{-1} - 1.5 times the most probable value given in Table 7. Similarly, the values of all the utilities, $u(x)$, are determined from the utility functions in which the values of x are sampled in a triangular distribution between 1.5^{-1} - 1.5 times the central values of x given in Table 6. Negative correlation between collective doses and remediation costs has been applied with a correlation coefficient of −0.8. The evaluation of the different strategies has been made with the forecasting and risk analysis program CRYSTAL BALL [4]. Latin Hypercube Sampling technique was used and the number of trials was 10,000. The results for the scores, U_i, for the options A - G2 are shown in Figure 11. The error bars represent the 5% and 95% percentiles of the distributions of U_i.

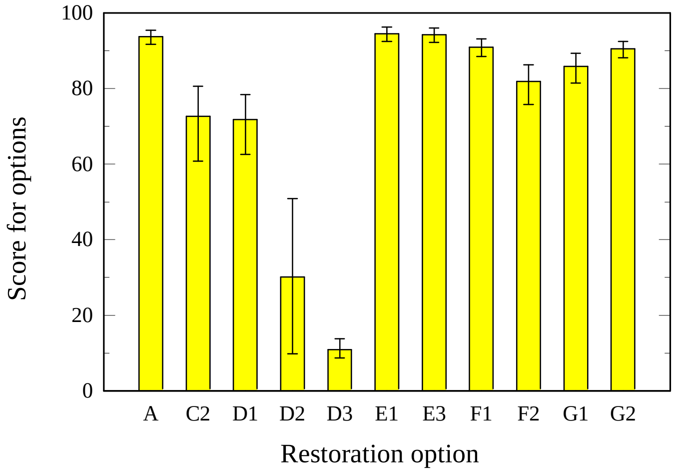


Figure 11. Overall evaluation of scores for different remediation strategies for the Drigg site for an integration time of 500 years for the collective dose. Identical scores are found for an integration time of 100 years.

It appears from Figure 11 that the options E1 and E3 have the highest score, closely followed by the option A. Also the options F1 and G2 have a high and practically an equal score. Therefore, it might be difficult to pick an optimum solution among the options E1, E3, A, F1 and G2.

4.3 Ravenglass site

The Ravenglass estuary is situated in West Cumbria on the coast of the Irish Sea. It encompasses the tidal reaches of the River Esk, Irt and Mite and its northern part directly borders on the Drigg site (see Section 4.2). The principal source of the estuary is the Irish Sea as the rivers contribute only a smaller part. The sediments are contaminated via the Irish Sea from waste discharges from the Sellafield nuclear fuel reprocessing plant. The main radionuclides of the contamination are ^{137}Cs, ^{239}Pu, and ^{241}Am.

This environment presents some problems considering the use of remediation techniques, as it is both tidal, dynamic and can be turbulent. The area is within the Lake District National Park and the public has therefore access to the area. As a consequence of the area characteristics ex-sit techniques will provide the best options for remediation of the site.

The following restoration options have been identified [8]:

- *A. No remediation*
- *B. Source removal*
- *C1. Soil Washing*
- *D1. Chemical Solubilisation*

4.3.1 Cost of restoration strategies

Remediation of the Ravenglass Estuary is primarily directed towards the muddy banks of the mud flats and salt marshes, which contain the highest levels of activity. The monetary he costs of the remediation options include remediation costs, waste disposal costs and monitoring costs. The cost components of the different options are summarised below.

<u>Option A: No remediation</u>

- Monitoring costs: 75,000 £·a^{-1} corresponding to 52,500 kEUR in 500 years

<u>Option B: Source Removal</u>

- Remediation of a total volume of about $1.3 \cdot 10^6$ m^3 sediments from different parts of the estuary; costs of excavation and transport (incl. labour): 130,000 kEUR
- Costs of disposal of $1.3 \cdot 10^6$ m^3 sediments: 780,000 kEUR
- Monitoring costs: 4,300 £·a^{-1} corresponding to 3,000 kEUR in 500 years

<u>Option C1: Soil Washing</u>

- Costs of excavation, transport and soil washing (incl. labour): 520,000 kEUR
- Costs of disposal of $2.6 \cdot 10^5$ m^3 radioactive waste assuming a reduction of the total volume with 80%: 650,000 kEUR
- Monitoring costs: 15,000 £·a^{-1} corresponding to 10,500 kEUR in 500 years

<u>Option D1: Chemical Solubilisation</u>

- Costs of excavation, transport and treatment (incl. labour): 720,000 kEUR
- Costs of disposal of $4.3 \cdot 10^5$ m^3 liquid radioactive waste assuming a concentration of ^{137}Cs of 10 MBq·m^{-3} from a total inventory of 4.5 TBq of ^{137}Cs in the estuary: 1,100,000 kEUR
- Monitoring costs: 15,000 £·a^{-1} corresponding to 10,500 kEUR in 500 years

4.3.2 Justified restoration strategies

The economic and radiologic data for remediation of the Ravenglass site are shown in Table 8. The remediation costs include the costs of labour and costs of monitoring. The monetary costs, X, of the remediation strategies can be compared to the benefit of the collective dose reduction, ΔS. The net benefit, ΔB, is given as:

$$\Delta B = \alpha \cdot \Delta S - (\alpha \cdot S_{work} + X_{remedia} + X_{waste} + X_{monitor})$$

None of the remedial options are justified on economic grounds alone when only the central estimates of collective dose are used together with an α-value of 100,000 EUR·manSv^{-1} [14]. Not even a higher value of α (*e.g.* 200,000 EUR·manSv^{-1}) and more extreme values from the calculated collective dose distribution (*e.g.* the 95th percentile) would make any of the options economically justified for any of the integration times for the collective doses.

Table 8. Remediation costs and collective doses to population and workers for different restoration strategies at the Ravenglass site.

Restoration strategy	Collective dose to population [man Sv]		Collective dose to workers [man·Sv]	Monetary costs of restoration [kEUR]			Fraction of activity left on-site	Waste volume (m^3)
	100 y	500 y		Remediation	Monitoring	Waste disposal		
A	28	29	0	0	52,500	0	1	0
B	15	15	0.92	130,000	3,000	780,000	0.05	$1.3 \cdot 10^6$
C1	23	24	1.01	520,000	10,500	650,000	0.2	$2.6 \cdot 10^5$
D1	7.7	8.2	2.29	720,000	10,500	$1.1 \cdot 10^6$	0.2	$4.3 \cdot 10^5$

The individual doses would be of the order of 1,500 $\mu Sv \cdot a^{-1}$ at the time of decision to introduce remediation (year 1). IAEA has proposed clean-up criteria in terms of individual dose [16]. For an individual dose range of 1 - 10 $mSv \cdot a^{-1}$ clean-up is almost always needed if a constraint for controlled practices is applied. Even without the application of a constraint IAEA suggests that for individual doses of 1 - 10 $mSv \cdot a^{-1}$ clean-up would usually be needed. Based on these recommendations it can therefore be concluded that some kind of remediation would almost always be justified for the Ravenglass Estuary.

4.3.3 Optimised restoration strategies

Utility functions for the attributes *monetary costs* and *radiation doses* have been calculated from the figures in Table 8 on monetary cost components and residual collective doses after remediation. Linear (risk neutral) utility functions have been used.

Utility functions for monetary costs

Utility functions have been determined for remediation costs (including labour costs), waste disposal costs (including transport costs) and monitoring costs:

$$u_{\text{remedia}}(x) = 100 \cdot \left(1 - \frac{x}{720{,}000}\right) \quad \text{for } 0 \le x \le 720{,}000 \text{ kEUR}$$

$$u_{waste}(x) = 100 \cdot \left(1 - \frac{x}{1{,}100{,}000}\right) \quad \text{for } 0 \le x \le 1{,}100{,}000 \text{ kEUR}$$

$$u_{monitor}(x) = 100 \cdot \left(1 + \frac{3{,}000 - x}{52{,}500 - 3{,}000}\right) \quad \text{for } 3{,}000 \le \text{x} \le 52{,}500 \text{ kEUR over } 500 \text{ y}$$

Utility functions for health factors

The following utility functions for the radiological health components have been determined for the exposed population and workers implementing the remedial actions. Only radiological health factors are considered for the Ravenglass site as no heavy metals are found.

$$u_{dose,\, pop,100}(x) = 100 \cdot \left(1 + \frac{7.7 - x}{28 - 7.7}\right) \text{ for } 7.7 \le x \le 28 \text{ man Sv}$$

$$u_{dose,\, pop,500}(x) = 100 \cdot \left(1 + \frac{8.2 - x}{29 - 8.2}\right) \quad \text{for } 8.2 \le x \le 29 \text{ man Sv}$$

$$u_{dose,\, work}(x) = 100 \cdot \left(1 - \frac{x}{2.29}\right) \quad \text{for } 0 \le x \le 2.29 \text{ man Sv}$$

Utility functions for social factors

The utility function u_{reas} for reassurance would be linked to both the residual dose and the fraction of activity remaining on the site after the remedial measure has been implemented. However, the residual dose and remaining activity are not necessarily correlated. A remedial measure that has left all the activity on site in a contained form (capping, surface barriers etc.) might give a substantial dose reduction and thus a low value of the residual doses. Detailed information on how social factors like reassurance are linked with individual doses and activity concentration on site is not available. Therefore, utility functions for 100 years and 500 years integration time have been proposed which gives a low value only when both sub-utilities have low values:

$$u_{\text{reas},100}(x,y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 + \frac{7.7 - x}{28 - 7.7} \right)_{\text{dose}} + \frac{1}{2} \cdot \left(1 + \frac{0.05 - y}{1.0 - 0.05} \right)_{\text{activity}} \right)$$

for $7.7 \le x \le 28$ man Sv and $0.05 \le \text{y} \le 1$

$$u_{\text{reas},500}(x,y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 + \frac{8.2 - x}{29 - 8.2} \right)_{\text{dose}} + \frac{1}{2} \cdot \left(1 + \frac{0.05 - y}{1.0 - 0.05} \right)_{\text{activity}} \right)$$

for $8.2 \le x \le 29$ man Sv and $0.05 \le y \le 1$

where y is the fraction of activity remaining on site after the remedial measures has been implemented. The value of the utility function u_{reas} is 100 for a residual dose of 7.7 (8.2) man Sv and a remaining fraction of the initial activity of 0.05 (best strategy) and 0 for a residual dose of 28 (29) man·Sv and a remaining activity fraction of 1.0 (worst strategy).

The utility function u_{distur} for disturbance has been related to the volume of soil and sediment waste to be transported to the waste disposal site:

$$u_{distur}(x) = 100 \cdot \left(1 - \frac{x}{430{,}000} \right) \quad \text{for } 0 \le x \le 430{,}000 \text{ m}^3$$

Weighting factors for major attributes

The major weighting factors considered in this study include those for monetary costs, health and social factors as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{health} + w_{economic} + w_{social} = 1$$

The assessment of the weighting factors is discussed in Annex B where conversion/scaling constants between weighting factors has been expressed as:

$$C_1 = \frac{w_{economic}}{w_{health}} \cong \frac{w_{economic}}{w_{dose,pop}} = \frac{R_{economic}}{\alpha \cdot R_{dose,pop}}$$

$$C_2 = \frac{w_{social}}{w_{health}} \approx \frac{r_{psy}}{r_{rad}}$$

C_1 can be determined for a 100 and 500 years integration time for the collective dose from the ranges of monetary costs and collective doses given in Table 8:

$$C_{1,100} = \frac{(1.1 \cdot 10^6 + 720{,}000 + 52{,}500) - 0}{100{,}000 \cdot (28 - 7.7) \cdot 10^{-3}} = \frac{1.87 \cdot 10^6}{2{,}030}$$

$$C_{1,500} = \frac{(1.1 \cdot 10^6 + 720{,}000 + 52{,}500) - 0}{100{,}000 \cdot (29 - 8.2) \cdot 10^{-3}} = \frac{1.87 \cdot 10^6}{2{,}080}$$

The value of C_2 is more difficult to assess but a value of 0.2 - 0.3 has been argued for in Annex B. The weighting factors can be calculated from the scaling constants C to be:

$$w_{health,100} = \frac{1}{1 + \frac{1.87 \cdot 10^6}{2{,}030} + 0.25} = \underline{1.08 \cdot 10^{-3}} \quad \text{and} \quad w_{health,500} = \frac{1}{1 + \frac{1.87 \cdot 10^6}{2{,}080} + 0.25} = \underline{1.11 \cdot 10^{-3}}$$

$$w_{economic,100} = \frac{\frac{1.87 \cdot 10^6}{2,030}}{1 + \frac{1.87 \cdot 10^6}{2,030} + 0.25} \cong \underline{0.999} \quad \text{and} \quad w_{economic,500} = \frac{\frac{1.87 \cdot 10^6}{2,080}}{1 + \frac{1.87 \cdot 10^6}{2,080} + 0.25} \cong \underline{0.999}$$

$$w_{social,100} = \frac{0.25}{1 + \frac{1.87 \cdot 10^6}{2,030} + 0.25} = \underline{2.71 \cdot 10^{-4}} \quad \text{and} \quad w_{social,500} = \frac{0.25}{1 + \frac{1.87 \cdot 10^6}{2,080} + 0.25} = \underline{2.78 \cdot 10^{-4}}$$

Weighting factors for health sub-attributes

The weighting factors for health sub-attributes include those of radiation induced stochastic health effects to the affected population and workers and non-radiation induced stochastic health effects to the affected population as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{dose,pop} + w_{dose,work} + w_{non-rad} = 1$$

The conversion/scaling constant, *C*, for the health sub-attributes can according to Annex B be expressed as:

$$w_{dose,pop} = C \cdot R_{dose,pop} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$

$$w_{dose,work} = C \cdot R_{dose,work} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$

$$w_{non-rad,pop} = C \cdot R_{non-rad,pop} \cdot l \cdot r_{non-rad}$$

Exposure to heavy metals is not relevant for the Ravenglass site and $R_{non\text{-}rad}$ is therefore zero. The value of *C* can be determined from the collective dose ranges, *R*, given in Table 8 as:

$$C_{100} = \frac{1}{(28 - 7.7) + (2.29 - 0)} = 0.044$$

$$C_{500} = \frac{1}{(29 - 8.2) + (2.29 - 0)} = 0.043$$

The weighting factors can be determined from the scaling constant *C* to be:

$$w_{dose,pop,100} = 0.044 \cdot (28 - 7.7) \cong \underline{0.90}$$

$$w_{dose,work,100} = 0.044 \cdot (2.29 - 0) \cong \underline{0.10}$$

$$w_{dose,pop,500} = 0.043 \cdot (29 - 8.2) \cong \underline{0.90}$$

$$w_{dose,work,500} = 0.043 \cdot (2.29 - 0) \cong \underline{0.10}$$

Weighting factors for economic sub-attributes

The weighting factors for economic sub-attributes include those for cost of remediation, cost of waste disposal and costs of monitoring (no loss/gain of taxes) as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{remedia} + w_{waste} + w_{monitor} = 1$$

The conversion/scaling constants for the economic sub-attributes can according to Annex B be expressed as:

$$w_{remedia} = C \cdot R_{remedia}$$

$$w_{waste} = C \cdot R_{waste}$$

$$w_{monitor} = C \cdot R_{monitor}$$

The conversion/scaling constant, *C*, for the economic sub-attributes can be determined from the cost ranges in Table 8:

$$\mathrm{C} = \frac{1}{(720{,}000 - 0) + (1.1 \cdot 10^{6} - 0) + (52{,}500 - 3{,}000)} = 5.35 \cdot 10^{-7}$$

The weighting factors can be calculated from the scaling constant *C* to be:

$$w_{remedia} = 5.35 \cdot 10^{-7} \cdot (720{,}000 - 0) = \underline{0.385}$$
$$w_{waste} = 5.35 \cdot 10^{-7} \cdot (1.1 \cdot 10^{6} - 0) = \underline{0.588}$$
$$w_{monitor} = 5.35 \cdot 10^{-7} \cdot (52{,}500 - 3{,}000) = \underline{0.026}$$

Weighting factors for social sub-attributes

The weighting factors include those for reassurance and disturbance as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{distur} + w_{reas} = 1$$

The conversion/scaling constant for the social sub-attributes can according to Annex B be expressed as:

$$C_1 = \frac{w_{reas}}{w_{distur}}$$

In Annex B it is argued that $w_{reas} > w_{distur}$ and that $C_1 \approx 5 - 7$. From these values the weighting factors can be calculated as:

$$w_{distur} = \frac{1}{1+6} \cong \underline{0.14} \quad \text{and} \quad w_{reas} = \frac{6}{1+6} \cong \underline{0.86}$$

The calculated values of the weighting factors for each of the attributes and sub-attributes are shown in Table 9.

Table 9. Weighting factors for attributes and sub-attributes applied in the optimisation of remediation of the Ravenglass site. The values in the left of the double columns are for an integration time of 100 years and in the right column for an integration time of 500 years.

Health factors		Economic factors		Social factors	
100 years	500 years	100 years	500 years	100 years	500 years
$1.08 \cdot 10^{-3}$	$1.11 \cdot 10^{-3}$	0.999	0.999	$2.71 \cdot 10^{-4}$	$2.78 \cdot 10^{-4}$

	100 years	500 years	Remediation costs	0.385	Reassurance	0.86
Dose population	0.90	0.90				
Dose workers	0.10	0.10	Waste disposal costs	0.588	Disturbance	0.14
Non-radiation	-	-	Monitoring costs	0.026	Loss/gain of income	-
			Loss/gain of taxes	-		

Scores for remediation options

The overall scores, U_i, of the remediation options i has been determined from the weighted sum of utilities for each of the attributes considered as shown in Figure 9:

$$
\begin{aligned}
U_i &= \sum_{j=1}^{3} w_j \cdot u_{ij} \\
&= w_{health} \cdot (w_{dose,pop} \cdot u_{dose,pop} + w_{dose,work} \cdot u_{dose,work}) \\
&+ w_{economic} \cdot (w_{waste} \cdot u_{waste} + w_{remedia} \cdot u_{remedia} + w_{monitor} \cdot u_{monitor}) \\
&+ w_{social} \cdot (w_{distur} \cdot u_{distur} + w_{reas} \cdot u_{reas})
\end{aligned}
$$

The weighting factors above have all been sampled in a triangular distribution between 1.5^{-1} - 1.5 times the most probable value given in Table 9. Similarly, the values of all the utilities, $u(x)$, are determined from the utility functions in which the values of x are sampled in a triangular distribution between 1.5^{-1} - 1.5 times the central values of x given in Table 8. Negative correlation between collective doses and remediation costs has been applied with a correlation coefficient of –0.8. The evaluation of the different strategies has been made with the forecasting and risk analysis program CRYSTAL BALL [4]. Latin Hypercube Sampling technique was used and the number of trials was 10,000. The results for the scores, U_i, for the options A - D1 are shown in Figure 12. The error bars represent the 5% and 95% percentiles of the distributions of U_i.

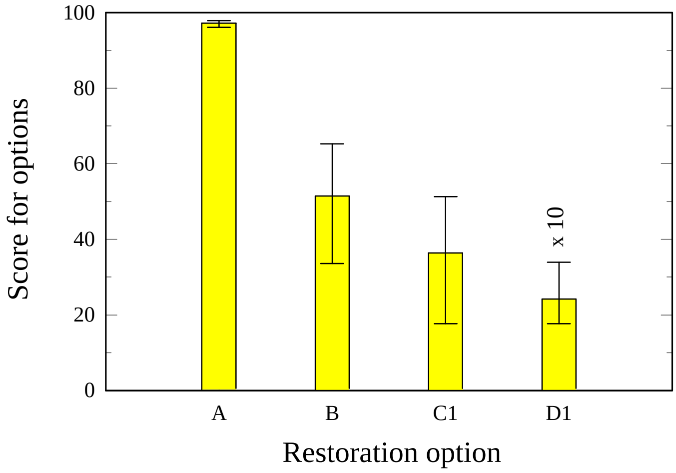


Figure 12. Overall evaluation of scores for different remediation strategies for the Ravenglass site for an integration time of 500 years for the collective dose. Identical scores are found for an integration time of 100 years ("× 10" means that the actual value is 10 times lower).

It appears from Figure 12 that option A has the highest score. The scores for options B and C1 are both significantly lower score than for option A. Due to the highest total costs for option D1 this option has the lowest score. The 'no remediation' option A is thus the optimum solution for the Ravenglass site and also the cheapest. There is no difference between the scores for the two different integration times due to the low weight of the health attributes.

4.4 Ranstad site

The Ranstad Tailing site is situated in the southern part of Sweden, in the Billingen-Häggum district about 20 km south of the city of Skövde. The tailings have been produced from a former uranium processing plant of the Swedish AB Atomenergi, which operated the uranium from a nearby open pit mine. The mill tailing consists of crushed alum shale from which uranium has been extracted by leaching. The contaminants are mainly ^{238}U in addition to significant levels of manganese and nickel.

In order to remediate a mill tailing different restoration techniques can be considered. In the case of the Ranstad mill tailing site three different categories of remediation techniques has been looked upon; containment, immobilisation and separation. Containment is a good alternative in order to reduce the amount of infiltrating water and the entrance of oxygen into the tailing. It is the percolating water together with oxygen that governs the weathering processes in the tailing. If the weathering processes stops then the amount of contaminants leaching from the tailing will be strongly reduced.

For the Ranstad tailing site two different types of capping have been considered. The first one consisting of 0.5 m of moraine, as it was on the tailing before the remediation started, and another one consisting of 1.6 m of different soil types, as was actually performed 1991-92.

Immobilisation is a technique where the aim is to reduce the mobility and solubility of contaminants. This can be done either by injecting solidifying material in the tailing, physical immobilisation, or by injecting immobilising reagents, chemical immobilisation. Since these methods would reduce the leakage from the tailing considerably they have been included in this study.

Separation techniques are useful in order to separate the contaminants from the tailing to a concentrated solution. Both physical and chemical separation can be used for this purpose. Even though such methods are not likely to be used when large amounts are to be separated, due to high costs, these techniques has been considered for the Ranstad tailing site.

The following restoration options have been identified [5]:

- A. *No remediation*
- C1. *Soil washing*
- D1. *Chemical Separation*
- E. *Containment*
  - E1. Capping 0.5 m
  - E2. Capping 1.6 m
- F2. *Physical Immobilisation, in-situ*
- G2. *Chemical Immobilisation, in-situ*

4.4.1 Cost of restoration strategies

The following restoration components have been identified for the different remedial options [5]:

Option A: No remediation

Option C1: Physical Separation (soil washing)

- Costs of remediation (incl. labour): 640,000 kEUR
- Costs of waste disposal (incl. transport): 38,000 kEUR

Option D1: Chemical Separation (solubilisation)

- Costs of remediation (incl. labour): 730,000 kEUR
- Costs of waste disposal (incl. transport): 38,000 kEUR

Option E1: Containment, capping 0.5 m

- Costs of remediation (incl. labour): 9,500 kEUR

Option E2: Containment, capping 1.6 m

- Costs of remediation (incl. labour): 16,000 kEUR

Option F2: Physical Immobilisation (*in-situ*)

- Costs of remediation (incl. labour): 23,000 kEUR

Option G2: Chemical Immobilisation (*in-situ*)

- Costs of remediation (incl. labour): 32,000 kEUR

4.4.2 Justified restoration strategies

The economic and radiological data for remediation of the Ranstad site are shown in Table 10. The remediation costs include the costs of labour and the waste disposal costs include transport costs. The monetary costs, X, of the remediation strategies can be compared to the benefit of the collective dose reduction, ΔS. The net benefit, ΔB, is given as:

$$\Delta B = \alpha \cdot \Delta S - (X_{remedia} + X_{waste})$$

None of the remedial options are justified on economic grounds alone when only the central estimates of collective dose are used together with an α-value of 100,000 EUR·manSv^{-1} [14]. Not even a higher value of α (*e.g.* 200,000 EUR·manSv^{-1}) and more extreme values from the calculated collective dose distribution (*e.g.* the 95th percentile) would make any of the options economically justified for any of the integration times for the collective doses.

Table 10. Remediation costs and collective doses to population and workers for different restoration strategies at the Ranstad site.

Restoration strategy	Collective dose to population [man Sv]		Collective intake of metals to population [man·kg]		Monetary costs of restoration [kEUR]		Fraction of activity left on-site	Waste volume (m^3)
	100 y	500 y	100 y	500 y	Remediation	Waste disposal		
			Manganese/Nickel					
A	0.59	24	12/0.88	22/58	0	0	1	0
C1	0.23	9.4	6.3/0.35	13/22	640,000	38,000	0.4	$4.5 \cdot 10^5$
D1	0.13	5.5	5.1/0.23	10/12	730,000	38,000	0.2	$1.5 \cdot 10^5$
E1	0.37	15.0	7.9/0.56	16/35	9,500	0	1	0
E2	0.19	8.1	4.4/0.31	9.4/18	16,000	0	1	0
F2	0.051	1.8	1.3/0.11	3.8/4.0	23,000	0	1	0
G2	0.034	1.1	0.73/0.075	2.9/2.5	32,000	0	1	0

The individual doses would in average be of the order of 40 µSv·a^{-1} at the time of decision to introduce remediation (year 1). IAEA has proposed clean-up criteria in terms of individual dose [16]. For an individual dose range of 10 - 100 µSv·a^{-1} clean-up is sometimes needed if a constraint for controlled practices is applied. Without the application of a constraint IAEA suggests that for individual doses of 10 - 100 $\mu Sv \cdot a^{-1}$ clean-up would rarely be needed. Based on these recommendations it can therefore be concluded that remediation would probably not be justified for the Ranstad site.

4.4.3 Optimised restoration strategies

Utility functions for the attributes *monetary costs* and *radiation doses* have been calculated from the figures in Table 10 on monetary cost components and residual collective doses after remediation. Linear (risk neutral) utility functions have been used.

Utility functions for monetary costs

Utility functions have been determined for remediation costs (including labour costs) and waste disposal costs (including transport costs):

$$u_{remedia}(x) = 100 \cdot \left(1 - \frac{x}{730{,}000}\right) \quad \text{for } 0 \le x \le 730{,}000 \text{ kEUR}$$

$$u_{waste}(x) = 100 \cdot \left(1 - \frac{x}{38{,}000}\right) \quad \text{for } 0 \le x \le 38{,}000 \text{ kEUR}$$

Utility functions for health factors

The following utility functions for the radiological health components have been determined for the exposed population and workers implementing the remedial actions. Both radiological and non-radiological health factors are considered for the Ranstad site as the heavy metals nickel and manganese would expose the population through contaminated foodstuffs.

$$u_{dose,\,pop,100}(x) = 100 \cdot \left(1 + \frac{0.034 - x}{0.59 - 0.034}\right) \text{ for } 0.034 \le x \le 0.59 \text{ man Sv}$$

$$u_{dose,\,pop,500}(x) = 100 \cdot \left(1 + \frac{1.1 - x}{24 - 1.1}\right) \quad \text{for } 1.1 \le x \le 24 \text{ man Sv}$$

$$u_{non\text{-}rad,100}(x) = 100 \cdot \left(1 + \frac{0.81 - x}{12.9 - 0.81}\right) \quad \text{for } 0.81 \le x \le 12.9 \text{ man kg nickel} + \text{manganese}$$

$$u_{non-rad,500}(x) = 100 \cdot \left(1 + \frac{5.4 - x}{80 - 5.4}\right) \quad \text{for } 5.4 \le x \le 80 \text{ man kg nickel} + \text{manganese}$$

Utility functions for social factors

The utility function u_{reas} for reassurance would be linked to both the residual dose and the fraction of activity remaining on the site after the remedial measure has been implemented. However, the residual dose and remaining activity are not necessarily correlated. A remedial measure that has left all the activity on site in a contained form (capping, surface barriers etc.) might give a substantial dose reduction and thus a low value of the residual doses. Detailed information on how social factors like reassurance are linked with individual doses and activity concentration on site is not available. Therefore, utility functions for 100 years and 500 years integration time have been proposed which gives a low value only when both sub-utilities have low values:

$$u_{reas,100}(x, y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 + \frac{0.034 - x}{0.59 - 0.034}\right)_{\text{dose}} + \frac{1}{2} \cdot \left(1 + \frac{0.2 - y}{1.0 - 0.2}\right)_{\text{activity}}\right)$$

$$\text{for } 0.034 \le x \le 0.59 \text{ man Sv and } 0.2 \le \text{y} \le 1$$

$$u_{reas,500}(x,y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 + \frac{1.1 - x}{24 - 1.1}\right)_{\text{dose}} + \frac{1}{2} \cdot \left(1 + \frac{0.2 - y}{1.0 - 0.2}\right)_{\text{activity}} \right)$$
$$\text{for } 1.1 \le x \le 24 \text{ man Sv and } 0.2 \le y \le 1$$

where y is the fraction of activity remaining on site after the remedial measures has been implemented. The value of the utility function u_{reas} is 100 for a residual dose of 0.034 (1.1) man Sv and a remaining fraction of the initial activity of 0.2 (best strategy) and 0 for a residual dose of 0.59 (24) man Sv and a remaining activity fraction of 1.0 (worst strategy).

The utility function u_{distur} for disturbance has been related to the volume of waste to be transported to the waste disposal site:

$$u_{distur}(x) = 100 \cdot \left(1 - \frac{x}{450{,}000}\right) \quad \text{for } 0 \le x \le 450{,}000 \text{ m}^3$$

Weighting factors for major attributes

The major weighting factors considered in this study include those for monetary costs, health and social factors as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{health} + w_{economic} + w_{social} = 1$$

The assessment of the weighting factors is discussed in Annex B where conversion/scaling constants between weighting factors has been expressed as:

$$C_1 = \frac{w_{economic}}{w_{health}} \cong \frac{w_{economic}}{w_{dose,pop}} = \frac{R_{economic}}{\alpha \cdot R_{dose,pop}}$$

$$C_2 = \frac{w_{social}}{w_{health}} \approx \frac{r_{psy}}{r_{rad}}$$

C_1 can be determined for a 100 and 500 years integration time for the collective dose from the ranges of monetary costs and collective doses given in Table 10:

$$C_{1,100} = \frac{(730{,}000 + 38{,}000) - 0}{100{,}000 \cdot (0.59 - 0.034) \cdot 10^{-3}} = \frac{768{,}000}{58.7}$$

$$C_{1,500} = \frac{(730{,}000 + 38{,}000) - 0}{100{,}000 \cdot (24 - 1.1) \cdot 10^{-3}} = \frac{768{,}000}{2{,}290}$$

The value of C_2 is more difficult to assess but a value of 0.2 - 0.3 has been argued for in Annex B. The weighting factors can be calculated from the scaling constants C to be:

$$w_{health,100} = \frac{1}{1 + \frac{768{,}000}{58.7} + 0.25} = \underline{7.64 \cdot 10^{-5}} \quad \text{and} \quad w_{health,500} = \frac{1}{1 + \frac{768{,}000}{2{,}290} + 0.25} = \underline{2.97 \cdot 10^{-3}}$$

$$w_{economic,100} = \frac{\dfrac{768{,}000}{58.7}}{1 + \dfrac{768{,}000}{58.7} + 0.25} = \underline{1.0} \quad \text{and} \quad w_{economic,500} = \frac{\dfrac{768{,}000}{2{,}290}}{1 + \dfrac{768{,}000}{2{,}290} + 0.25} = \underline{0.996}$$

$$w_{social,100} = \frac{0.25}{1 + \dfrac{768{,}000}{58.7} + 0.25} = \underline{1.91 \cdot 10^{-5}} \quad \text{and} \quad w_{social,500} = \frac{0.25}{1 + \dfrac{768{,}000}{2{,}290} + 0.25} = \underline{7.43 \cdot 10^{-4}}$$

Weighting factors for health sub-attributes

The weighting factors for health sub-attributes include those of radiation induced stochastic health effects to the affected population and workers and non-radiation induced stochastic health effects to the affected population as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{dose,pop} + w_{non-rad,pop} = 1$$

The conversion/scaling constant, *C*, for the health sub-attributes can according to Annex B be expressed as:

$$w_{dose,pop} = C \cdot R_{dose,pop} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$
$$w_{non-rad,pop} = C \cdot R_{non-rad,pop} \cdot l \cdot r_{non-rad}$$

The conversion/scaling constant, *C*, can be determined from the collective dose ranges, *R*, given in Table 10 as:

$$C_{100} = \frac{1}{(0.59 - 0.034)} \cong 1.80$$
$$C_{500} = \frac{1}{(24 - 1.1)} \cong 0.044$$

The weighting factors can be calculated from the scaling constant *C* to be:

$$w_{dose,pop,100} = 1.80 \cdot (0.59 - 0.034) = \underline{1.0}$$
$$w_{dose,pop,500} = 0.044 \cdot (24 - 1.1) = \underline{1.0}$$

The risk factor for ingestion of manganese and nickel, $r_{non-rad}$, is at present unknown and the weighting factor for exposure to manganese and nickel, $w_{non-rad}$, has therefore not been determined.

Weighting factors for economic sub-attributes

The weighting factors for economic sub-attributes include those for cost of remediation and costs of waste disposal (no loss/gain of taxes and no costs of monitoring) as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{remedia} + w_{waste} = 1$$

The conversion/scaling constant, *C*, for the economic sub-attributes can according to Annex B be expressed as:

$$w_{remedia} = C \cdot R_{remedia}$$

$$w_{waste} = C \cdot R_{waste}$$

The conversion/scaling constant, *C*, can be determined from the cost ranges, *R*, given in Table 10 to be:

$$C = \frac{1}{(730{,}000 - 0) + (38{,}000 - 0)} = 1.30 \cdot 10^{-6}$$

The weighting factors can be calculated from the scaling constant *C* to be:

$$w_{remedia} = 1.30 \cdot 10^{-6} \cdot (730{,}000 - 0) = \underline{0.95}$$
$$w_{waste} = 1.30 \cdot 10^{-6} \cdot (38{,}000 - 0) = \underline{0.05}$$

Weighting factors for social sub-attributes

The weighting factors include those for reassurance, disturbance and loss/gain of taxes as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{distur} + w_{reas} = 1$$

The conversion/scaling constant for the social sub-attributes can according to Annex B be expressed as:

$$C_1 = \frac{w_{reas}}{w_{distur}}$$

In Annex B it is argued that $w_{reas} > w_{distur}$ and that $C_1 \approx 5 - 7$. From these values the weighting factors can be calculated as:

$$w_{distur} = \frac{1}{1+6} \cong \underline{0.14} \quad \text{and} \quad w_{reas} = \frac{6}{1+6} \cong \underline{0.86}$$

The calculated values of the weighting factors for each of the attributes and sub-attributes are shown in Table 11.

Table 11. Weighting factors for attributes and sub-attributes applied in the optimisation of remediation of the Ranstad site. The values in the left of the double columns are for an integration time of 100 years and in the right column for an integration time of 500 years.

Health factors		Economic factors		Social factors	
100 years	500 years	100 years	500 years	100 years	500 years
$7.64 \cdot 10^{-5}$	$2.97 \cdot 10^{-3}$	1.0	0.996	$1.91 \cdot 10^{-5}$	$7.43 \cdot 10^{-4}$

	100 years	500 years				
Dose population	1	1	Remediation costs	0.95	Reassurance	0.86
Dose workers	-	-	Waste disposal costs	0.05	Disturbance	0.14
Non-radiation	-	-	Monitoring costs	-	Loss/gain of income	-
			Loss/gain of taxes	-		

Scores for remediation options

The overall scores, U_i, of the remediation options *i* has been determined from the weighted sum of utilities for each of the attributes considered as shown in Figure 9:

$$\begin{aligned} U_i &= \sum_{j=1}^{3} w_j \cdot u_{ij} \\ &= w_{health} \cdot (w_{dose,pop} \cdot u_{dose,pop} + w_{non-rad} \cdot u_{non-rad}) \\ &+ w_{economic} \cdot (w_{waste} \cdot u_{waste} + w_{remedia} \cdot u_{remedia}) \\ &+ w_{social} \cdot (w_{distur} \cdot u_{distur} + w_{reas} \cdot u_{reas}) \end{aligned}$$

It has not been possible to determine the risk factors for ingestion of manganese and nickel, and consequently no value for the weighting factor, $w_{non\text{-}rad}$, has been determined. The weighting factors above have all been sampled in a triangular distribution between 1.5^{-1} - 1.5 times the most probable value given in Table 11. Similarly, the values of all the utilities, $u(x)$, are determined from the utility functions in which the values of x are sampled in a triangular distribution between 1.5^{-1} - 1.5 times the central values of x given in Table 10. Negative correlation between collective doses and remediation costs has been applied with a correlation coefficient of –0.8. The evaluation of the different strategies has been made with the forecasting and risk analysis program CRYSTAL BALL [4]. Latin Hypercube Sampling technique was used and the number of trials was 10,000. The results for the scores, U_i, for the options A - G2 are shown in Figure 13. The error bars represent the 5% and 95% percentiles of the distributions of U_i.

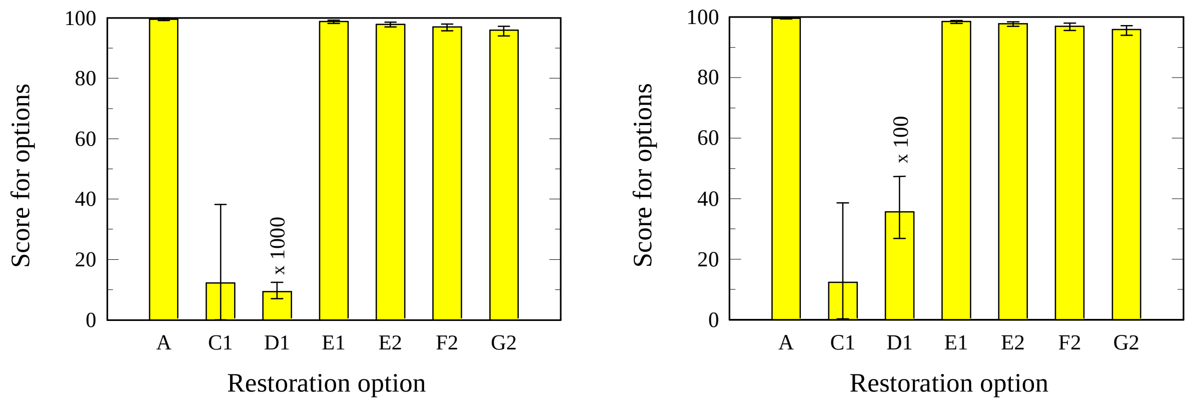


Figure 13. Overall evaluation of scores for different remediation strategies for the Ranstad site for an integration time of 100 and 500 years for the collective dose. The left picture shows the results for an integration time of 100 years for the collective dose and the right picture for an integration time of 500 years ("× 100" means that the actual value is 100 times lower).

As can be seen from Figure 13, option A has the highest score. The options E1, E2, F2 and G2 have all a more or less equal score, not significantly lower than that of option A. The options C1 and D1 both have a low score due to high remediation and waste disposal costs. The 'no remediation' option A can thus be considered as the optimum solution for the Ranstad site and also the cheapest. There is no significant difference between the scores for the two different integration times except for option D1.

4.5 Lake Tranebärssjön

The location of the Lake Tranebärssjön site is approximately 5 km east of the Ranstad tailing site. It is a former uranium mine (open pit mining) which was in operation between 1965 and 1969. The lake has been existing only since 1990, when the mine was flooded by water. Its dimensions are 2000 m length, 100-200 m width, and 15 m depth, giving an open area of 250 000 m^2.

The Lake Tranebärssjön is not considered to be a radiological problem even though the Swedish Radiation Protection Agency have decided that ^{226}Ra should be measured four times a year at the outlet of the lake. During the last three years the radioactivity has not exceeded 10 $mBq \cdot l^{-1}$. Since there is a lack of information in order to restore a lake this study has focused on the restoration of the outgoing water from the lake.

The following restoration options have been identified [5].

- *A. No remediation*
- *C2. Physical separation (filtration)*
- *D3. Biological separation (biosorption)*

4.5.1 Cost of restoration strategies

The following restoration components have been identified for the different remedial options:

Option A: No remediation

Option C2: Physical Separation (filtration)

- Costs of remediation (incl. labour): 400,000 kEUR

Option D3: Biological Separation (biosorption)

- Costs of remediation (incl. labour): 700,000 kEUR

4.5.2 Justified restoration strategies

The economic and radiological data for remediation of the Lake Tranebärssjön site are shown in Table 12. The remediation costs include the costs of labour. The monetary costs, *X*, of the remediation strategies can be compared to the benefit of the collective dose reduction, ΔS. The net benefit, ΔB, is given as:

$$\Delta B = \alpha \cdot \Delta S - X_{remedia}$$

None of the remedial options are justified on economic grounds alone when only the central estimates of collective dose are used together with an α-value of 100,000 EUR·manSv^{-1} [14]. Not even a higher value of α (*e.g.* 200,000 EUR·manSv^{-1}) and more extreme values from the calculated collective dose distribution (*e.g.* the 95th percentile) would make any of the options economically justified for any of the integration times for the collective doses.

Table 12. Remediation costs and collective doses to population and workers for different restoration strategies at the Lake Tranebärssjön site.

Restoration strategy	Collective dose to population [man Sv]		Collective intake of metals to population [man·kg]		Monetary costs of restoration [kEUR]	Fraction of activity left on-site
	100 y	500 y	100 y Manganese/Nickel	500 y Manganese/Nickel		
A	0.069	0.27	1.6/1.4	4.4/5.9	0	1
C2	0.0081	0.033	0.72/0.23	2.4/1.0	400,000	0.1
D3	0.002	0.0089	0.59/0.11	2.2/0.55	700,000	0.03

The individual doses would in average be of the order of 15 μSv·a^{-1} at the time of decision to introduce remediation (year 1). IAEA has proposed clean-up criteria in terms of individual dose [16]. For an individual dose range of 10 - 100 μSv·a^{-1} clean-up is sometimes needed if a constraint for controlled practices is applied. Without the application of a constraint IAEA suggests that for individual doses of 10 - 100 μSv·a^{-1} clean-up would rarely be needed. Based on these recommendations it can therefore be concluded that remediation would probably not be justified for the Lake Tranebärssjön site.

4.5.3 Optimised restoration strategies

Utility functions for the attributes *monetary costs* and *radiation doses* have been calculated from the figures in Table 12 on monetary cost components and residual collective doses after remediation. Linear (risk neutral) utility functions have been used.

Utility functions for monetary costs

A utility function has been determined for remediation costs (including labour costs):

$$u_{remedia}(x) = 100 \cdot \left(1 - \frac{x}{700{,}000}\right) \quad \text{for } 0 \le x \le 700{,}000 \text{ kEUR}$$

Utility functions for health factors

The following utility functions for the radiological health components have been determined for the exposed population and workers implementing the remedial actions. Both radiological and non-radiological health factors are considered for the Lake Tranebärssjön site as the heavy metals nickel and manganese would expose the population through contaminated foodstuffs.

$$u_{dose,\ pop,100}(x) = 100 \cdot \left(1 + \frac{0.002 - x}{0.069 - 0.002}\right) \text{ for } 0.002 \le x \le 0.069 \text{man Sv}$$

$$u_{dose,\ pop,500}(x) = 100 \cdot \left(1 + \frac{0.0089 - x}{0.27 - 0.0089}\right) \quad \text{for } 0.0089 \le x \le 0.27 \text{ man Sv}$$

$$u_{non\text{-}rad,100}(x) = 100 \cdot \left(1 + \frac{0.70 - x}{3.0 - 0.70}\right) \text{ for } 0.70 \le x \le 3.0 \text{ man kg nickel} + \text{manganese}$$

$$u_{non-rad,500}(x) = 100 \cdot \left(1 + \frac{2.75 - x}{10.3 - 2.75}\right) \text{ for } 2.75 \le x \le 10.3 \text{ man kg nickel} + \text{manganese}$$

Utility functions for social factors

The utility function u_{reas} for reassurance would be linked to both the residual dose and the fraction of activity remaining on the site after the remedial measure has been implemented. However, the residual dose and remaining activity are not necessarily correlated. A remedial measure that has left all the activity on site in a contained form (capping, surface barriers etc.) might give a substantial dose reduction and thus a low value of the residual doses. Detailed information on how social factors like reassurance are linked with individual doses and activity concentration on site is not available. Therefore, utility functions for 100 years and 500 years integration time have been proposed which gives a low value only when both sub-utilities have low values:

$$u_{reas,100}(x,y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 + \frac{0.002 - x}{0.069 - 0.002}\right)_{\text{dose}} + \frac{1}{2} \cdot \left(1 + \frac{0.03 - y}{1.0 - 0.03}\right)_{\text{activity}}\right)$$

for $0.002 \le x \le 0.069$ man Sv and $0.03 \le \text{y} \le 1$

$$u_{reas,500}(x,y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 + \frac{0.0089 - x}{0.27 - 0.0089}\right)_{\text{dose}} + \frac{1}{2} \cdot \left(1 + \frac{0.03 - y}{1.0 - 0.03}\right)_{\text{activity}}\right)$$

for $0.0089 \le x \le 0.27$ man Sv and $0.03 \le y \le 1$

where y is the fraction of activity remaining on site after the remedial measures has been implemented. The value of the utility function u_{reas} is 100 for a residual dose of 0.002 (0.0089) man Sv and a remaining fraction of the initial activity of 0.03 (best strategy) and 0 for a residual dose of 0.069 (0.27) man Sv and a remaining activity fraction of 1.0 (worst strategy).

The utility function u_{distur} for disturbance has been related to the remediation costs:

$$u_{distur}(x) = 100 \cdot \left(1 - \frac{x}{700{,}000}\right) \quad \text{for } 0 \le x \le 700{,}000 \text{ kEUR}$$

Weighting factors for major attributes

The major weighting factors considered in this study include those for monetary costs, health and social factors as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{health} + w_{economic} + w_{social} = 1$$

The assessment of the weighting factors is discussed in Annex B where conversion/scaling constants between weighting factors has been expressed as:

$$C_1 = \frac{w_{economic}}{w_{health}} \cong \frac{w_{economic}}{w_{dose,pop}} = \frac{R_{economic}}{\alpha \cdot R_{dose,pop}}$$

$$C_2 = \frac{w_{social}}{w_{health}} \approx \frac{r_{psy}}{r_{rad}}$$

C_1 can be determined for a 100 and 500 years integration time for the collective dose from the ranges of monetary costs and collective doses given in Table 12:

$$C_{1,100} = \frac{(700{,}000 - 0)}{100{,}000 \cdot (0.069 - 0.002) \cdot 10^{-3}} = \frac{700{,}000}{6.7}$$

$$C_{1,500} = \frac{(700{,}000 - 0)}{100{,}000 \cdot (0.27 - 0.0089) \cdot 10^{-3}} = \frac{700{,}000}{26.1}$$

The value of C_2 is more difficult to assess but a value of 0.2 - 0.3 has been argued for in Annex B. The weighting factors can be calculated from the scaling constants C to be:

$$w_{health,100} = \frac{1}{1 + \frac{700{,}000}{6.7} + 0.25} = \underline{9.57 \cdot 10^{-6}} \quad \text{and} \quad w_{health,500} = \frac{1}{1 + \frac{700{,}000}{26.1} + 0.25} = \underline{3.73 \cdot 10^{-5}}$$

$$w_{economic,100} = \frac{\frac{700{,}000}{6.7}}{1 + \frac{700{,}000}{6.7} + 0.25} = \underline{1.0} \quad \text{and} \quad w_{economic,500} = \frac{\frac{700{,}000}{26.1}}{1 + \frac{700{,}000}{26.1} + 0.25} = \underline{1.0}$$

$$w_{social,100} = \frac{0.25}{1 + \frac{700{,}000}{6.7} + 0.25} = \underline{2.39 \cdot 10^{-6}} \quad \text{and} \quad w_{social,500} = \frac{0.25}{1 + \frac{700{,}000}{26.1} + 0.25} = \underline{9.32 \cdot 10^{-6}}$$

Weighting factors for health sub-attributes

The weighting factors for health sub-attributes include those of radiation induced stochastic health effects to the affected population and workers and non-radiation induced stochastic health effects to the affected population as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{dose,pop} + w_{non-rad,pop} = 1$$

The conversion/scaling constant, C, for the health sub-attributes can according to Annex B be expressed as:

$$w_{dose,pop} = C \cdot R_{dose,pop} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$
$$w_{non-rad,pop} = C \cdot R_{non-rad,pop} \cdot l \cdot r_{non-rad}$$

The conversion/scaling constant, C, can be determined from the collective dose ranges, R, given in Table 12 as:

$$C_{100} = \frac{1}{(0.069 - 0.002)} = 14.9$$

$$C_{500} = \frac{1}{(0.27 - 0.0089)} = 3.8$$

The weighting factors can be calculated from the scaling constant C to be:

$$w_{dose,pop,100} = 14.9 \cdot (0.069 - 0.002) = \underline{1.0}$$

$$w_{dose,pop,500} = 3.8 \cdot (0.27 - 0.0089) = \underline{1.0}$$

The risk factor for ingestion of manganese and nickel, $r_{non\text{-}rad}$, is at present unknown and the weighting factor for exposure to manganese and nickel, $w_{non\text{-}rad}$, has therefore not been determined.

Weighting factors for social sub-attributes

The weighting factors include those for reassurance and disturbance as shown in Figure 9. The sum of these weighting factors should respect the following conditions:

$$w_{distur} + w_{reas} = 1$$

The conversion/scaling constant for the social sub-attributes can according to Annex B be expressed as:

$$C_1 = \frac{w_{reas}}{w_{distur}}$$

In Annex B it is argued that $w_{reas} > w_{distur}$ and that $C_1 \approx 5 - 7$. From these values the weighting factors can be calculated as:

$$w_{distur} = \frac{1}{1+6} \cong \underline{0.14} \quad \text{and} \quad w_{reas} = \frac{6}{1+6} \cong \underline{0.86}$$

The calculated values of the weighting factors for each of the attributes and sub-attributes are shown in Table 13.

Table 13. Weighting factors for attributes and sub-attributes applied in the optimisation of remediation of the Lake Tranebjärssjön site. The values in the left of the double columns are for an integration time of 100 years and in the right column for an integration time of 500 years.

Health factors		Economic factors		Social factors	
100 years	500 years	100 years	500 years	100 years	500 years
$9.57 \cdot 10^{-6}$	$3.73 \cdot 10^{-5}$	1.0	1.0	$2.39 \cdot 10^{-6}$	$9.32 \cdot 10^{-6}$

	100 years	500 years				
Dose population	1	1	Remediation costs	1	Reassurance	0.86
Dose workers	-	-	Waste disposal costs	-	Disturbance	0.14
Non-radiation	-	-	Monitoring costs	-	Loss/gain of income	-
			Loss/gain of taxes	-		

Scores for remediation options

The overall scores, U_i, of the remediation options i has been determined from the weighted sum of utilities for each of the attributes considered as shown in Figure 9:

$$\begin{aligned} U_i &= \sum_{j=1}^{3} w_j \cdot u_{ij} \\ &= w_{health} \cdot (w_{dose,pop} \cdot u_{dose,pop} + w_{non-rad} \cdot u_{non-rad}) \\ &+ w_{economic} \cdot u_{remedia} \\ &+ w_{social} \cdot (w_{distur} \cdot u_{distur} + w_{reas} \cdot u_{reas}) \end{aligned}$$

It has not been possible to determine the risk factors for ingestion of manganese and nickel, and consequently no value for the weighting factor, $w_{non\text{-}rad}$, has been determined. The weighting factors above have all been sampled in a triangular distribution between 1.5^{-1} - 1.5 times the most probable value given in Table 13. Similarly, the values of all the utilities, $u(x)$, are determined from the utility functions in which the values of x are sampled in a triangular distribution between 1.5^{-1} - 1.5 times the central values of x given in Table 12. Negative correlation between collective doses and remediation costs has been applied with a correlation coefficient of –0.8. The evaluation of the different strategies has been made with the forecasting and risk analysis program CRYSTAL BALL [4]. Latin Hypercube Sampling technique was used and the number of trials was 10,000. The results for the scores, U_i, for the options A - D3 are shown in Figure 14. The error bars represent the 5% and 95% percentiles of the distributions of U_i.

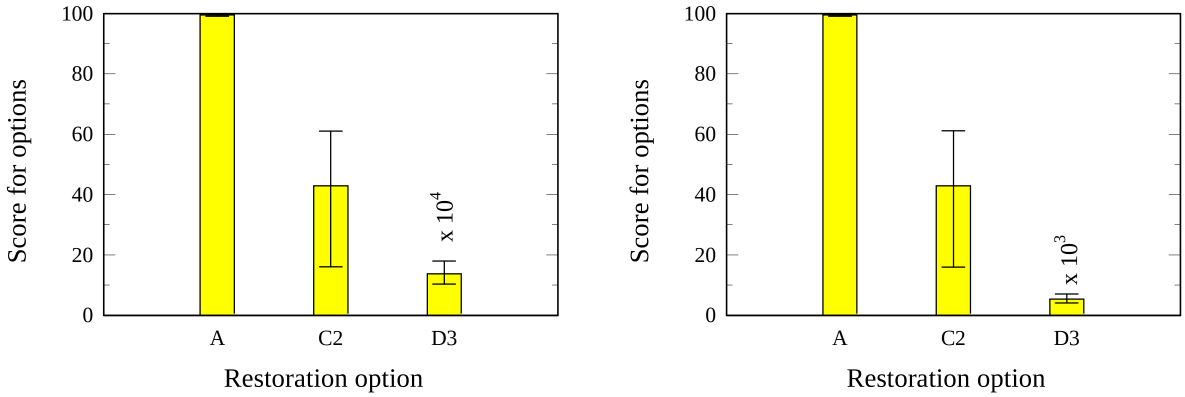


Figure 14. Overall evaluation of scores for different remediation strategies for the Lake Tranebärssjön site. The left picture shows the results for an integration time of 100 years for the collective dose and the right picture for an integration time of 500 years ("$\times 10^4$" and "$\times 10^3$" means that the actual value is 10^4 and 10^3 times lower).

As can be seen from Figure 14, option A has the highest score. The options C2 and D3 both have a significantly lower score than that of option A due to high remediation costs. The 'no remediation' option A can thus be considered as the optimum solution for the Lake Tranebärssjön site and also the cheapest.

5 Summary and conclusions

Five European sites contaminated as a result of the operation of a practice at the site have been studied. Various remediation options have been envisaged with respect to the optimisation of the protection of the populations being exposed to the radionuclides at the sites. The example sites being studied are:

- Molse Nete River in Belgium the riverbanks of which have been contaminated with the radionuclides ^{60}Co, ^{137}Cs, ^{239}Pu, and ^{241}Am from discharges from the research centre SCK•CEN in Mol
- Drigg waste disposal site in West Cumbria on the coast of the Irish Sea used for shallow burial of solid waste, mostly from the Sellafield site; the dominating radionuclides are ^{137}Cs, ^{238}U, ^{239}Pu, and ^{241}Am

- Ravenglass estuary in West Cumbria on the coast of the Irish Sea has been contaminated via the Irish Sea from waste discharges from the Sellafield nuclear fuel reprocessing plant; the main radionuclides of the contamination are ^{137}Cs, ^{239}Pu, and ^{241}Am
- Ranstad tailing site in the southern part of Sweden; the tailings have been produced from a former uranium processing plant of the Swedish AB Atomenergi, and the contaminants are mainly ^{238}U in addition to significant levels of manganese and nickel
- The Lake Tranebärssjön site which is a former open pit uranium mine; the contaminants are the same as for the Ranstad tailing site namely ^{238}U, manganese and nickel

The optimisation of protection of the exposed populations at these sites is a process of selecting among justified remediation options for the maximum net benefit, *i.e*. a comparison of options. The avertable collective dose is only one component of the net benefit. Other components include the monetary costs of the remedial measure, reassurance provided by the remedial measures, the anxiety it causes, and the resulting individual and social disruption. The collective dose is calculated from the distribution of all exposures of the entire population and it cannot, alone, be a general indicator of justification, nor does justification or collective doses provide information on the exposure of the critical group.

Limiting members of the public from being exposed inequitably is accomplished by constraining the individual dose to the average member of the critical group. Such a critical group may, or may not, be different for various remediation options. Furthermore, the relationship of a dose constraint to avertable collective dose and to justification is a complex one that is potentially different for the various remediation options and also for different contamination situations.

Multi-attribute utility analyses and cost-benefit analyses have been used to illustrate how to arrive at an optimum remediation strategy among a number of different strategies. In addition, the recommendations from the IAEA on individual dose levels above which remediation normally is justified, have been addressed. The applied attributes include monetary costs of the remedial measures, the collective dose to the clean-up workers, the collective dose to population, and social factors like reassurance and disturbance. Linear utility functions, so-called risk-neutral utility functions, have been used and uncertainties included in terms of value distributions of the attributes. The weighting factors assigned to the different attributes have been determined by use of scaling factors in terms of weighting factor ratios, and their values were sampled around a most probable value. The ranking of different remediation options at the five European example sites is summarised in Tables 14 and 15.

The ranking of the different remedial measures suggested for the example sites using multi-attribute analysis with utility functions allows the inclusion of factors that are not easy to quantify in monetary terms as is required in cost-benefit analysis. Notwithstanding this advantage of the multi-attribute method there are difficulties with the determination of weighting factors for the different attributes. Without any terms of reference for the weighting between attributes, value settings by a decision-maker could lead to ‘optimised’ results that might be useless because of a subjective bias of the decision-maker in the selection of weighting factors. Therefore, the outcome of any multi-attribute analysis, including the present study, should be judged very carefully in the light of the values assigned to the weighting factors before any firm conclusions could be drawn.

Two different methods have been used in this study to determine the weighting factors, w. For attributes at the same hierarchy level given in the same unit, *e.g.* monetary costs, the weighting between the different attributes have been related to their value ranges, R, by the relation $(w/R)_1 = (w/R)_2 = (w/R)_3 = \ldots C$. The weighting of attributes at the same hierarchy level for which the units are different, as they are for the social attributes, has been determined by assigning a value to the ratio of their weighting factors as $w_2/ w_1 = C_1$, $w_3/ w_1 = C_2$, …… $w_n/ w_1 = C_{n-1}$.

Different remediation measures have been evaluated for the five example sites. The evaluation has been based upon (a) justification of the measures by trade-off between avertable collective dose and monetary costs, (b) compliance with the recommended clean-up criteria from the IAEA [15], and (c) optimisation of scores for the different remediation measures by use of multi-attribute utility analyses. The overall results of the evaluation are summarised in the table below.

It should be emphasized that some attributes were not evaluated in detail at all the example sites. Especially some of the economic attributes have been difficult to determine. However, the potential dose savings by the suggested remedial measures are rather moderate and the overall picture is expected to remain robust with more realistic economic attributes, also because the dose estimates are on the conservative side.

None of the remedial measures considered for each site are justified from a cost-benefit point of view based on central estimates of collective dose and monetary costs. If more extreme values of collective doses are included in the cost-benefit analyses some of the remedial measures considered for the sites of Molse Nete River and Drigg would be justified. For the sites of Ravenglass, Ranstad and Lake Tranebärssjön no remedial measures are justified on economic grounds, not even if more extreme values of the collective doses are included.

Site	Justification by cost-benefit	Compliance with IAEA criteria	Optimised strategy
Molse Nete River	'No remediation' has the highest net benefit (0) on central estimates; some options are justified on extreme values of doses	Remediation usually needed (constraint) or sometimes needed (no constraint) on grounds of annual individual doses	'No remediation' (100 years); Capping soil and sediment (500 years);
Drigg	'No remediation' has the highest net benefit (0) on central estimates; some options are justified on extreme values of doses	Remediation almost always needed (constraint) or usually needed (no constraint) on grounds of annual individual doses	Capping
Ravenglass	'No remediation' has the highest net benefit (0) on central estimates and also on extreme values of doses	Remediation almost always needed (constraint) or usually needed (no constraint) on grounds of annual individual doses	'No remediation'
Ranstad	'No remediation' has the highest net benefit (0) on central estimates and also on extreme values of doses	Remediation sometimes needed (constraint) or rarely needed (no constraint) on grounds of annual individual doses	'No remediation'
Lake Tranebärssjön	'No remediation' has the highest net benefit (0) on central estimates and also on extreme values of doses	Remediation sometimes needed (constraint) or rarely needed (no constraint) on grounds of annual individual doses	'No remediation'

The individual doses to critical groups without remedial measures being introduced at each of the example sites have been compared to the IAEA criteria for clean-up of contaminated land. If it is assumed that a dose constraint for controlled practices would be applied to the outcome of the remediation process at the sites, some remediation might be needed at all sites.

Multi-attribute analyses on ranking different remediation options at each example site nearly all give the result that 'no remediation' is the best option, *i.e.* having the highest score. The reason is the dominating weight of the economic attributes compared to the health and social attributes. The rather low collective doses and the potential for only low collective dose savings by remediation together with relatively high economic costs of the remedial measures are the cause of the low weights given to health and social factors. In addition, the low health and social weights are responsible for an only marginal difference between the scores for the situations where collective doses have been determined for a time period of 100 and 500 years.

Table 14. Summary of the ranking of remediation options for the sites of Molse Nete River, Drigg, Ravenglass, Ranstad and Lake Tranebärsjön. The integration time for the collective dose is 100 years.

Ranking No.	Molse Nete River	Drigg	Ravenglass	Ranstad	Lake Tranebärssjön
1	No remediation (discharge stop[a])	Capping	No remediation	No remediation	No remediation
2	Capping soil/sediment	Sub-surface barrier	Source removal	Capping 0.5 m	Filtration
3	Physical immobilisation *in-situ*	No remediation	Soil washing	Capping, 1.6 m	Biosorption
4	Chemical immobilisation *in-situ*	Physical immobilisation *ex-situ*	Chemical solubilisation	Physical immobilisation	
5	Physical immobilisation *ex-situ*	Chemical immobilisation *in-situ*		Chemical immobilisation	
6	Chemical immobilisation *ex-situ*	Chemical immobilisation *ex-situ*		Chemical separation	
7	Soil/sediment removal	Physical immobilisation *in-situ*		Physical separation	
8	Physical separation	Filtration			
9	Chemical separation	Chemical Solubilisation			
10		Ion exchange[b]			
11		Biosorption[b]			

[(a)]All options are with discharge stop

[(b)]Very costly techniques (liquid treatment over many years)

Table 15. Summary of the ranking of remediation options for the sites of Molse Nete River, Drigg, Ravenglass, Ranstad and Lake Tranebärsjön. The integration time for the collective dose is 500 years.

Ranking No.	Molse Nete River	Drigg	Ravenglass	Ranstad	Lake Tranebärssjön
1	Capping soil/sediment	Capping	No remediation	No remediation	No remediation
2	Physical immobilisation *in-situ*	Sub-surface barrier	Source removal	Capping 0.5 m	Filtration
3	Chemical immobilisation *in-situ*	No remediation	Soil washing	Capping, 1.6 m	Biosorption
4	Physical immobilisation *ex-situ*	Physical immobilisation *ex-situ*	Chemical solubilisation	Physical immobilisation	
5	No remediation (discharge stop[a])	Chemical immobilisation *in-situ*		Chemical immobilisation	
6	Chemical immobilisation *ex-situ*	Chemical immobilisation *ex-situ*		Chemical separation	
7	Soil/sediment removal	Physical immobilisation *in-situ*		Physical separation	
8	Physical separation	Filtration			
9	Chemical separation	Chemical Solubilisation			
10		Ion exchange[b]			
11		Biosorption[b]			

(a)All options are with discharge stop

(b)Very costly techniques (liquid treatment over many years)

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Annex A. Quantification of risk from exposures to carcinogens

For the purpose of risk assessment, health effects from exposure to contaminants are generally divided into two categories.

- effects for which the probability of development is proportional to the dose (somatic and genetic effects) and for which it is assumed that exposure to even very low doses presents a non-zero risk (no threshold for effects)
- effects that only occur above a given threshold level of dose (somatic effects)

In the case of exposure to ionising radiation, the two types of effect are also referred to as stochastic and deterministic effects.

Non-threshold substances include genotoxic carcinogens and mutagens; threshold substances include non-genotoxic carcinogens and substances causing toxic effects other than cancer and genetic effects. USEPA classifies agents as carcinogenic and non-carcinogenic which can cause some confusion because non-genotoxic carcinogens are assumed to cause effects only above a certain threshold dose.

A.1 Risk from exposure to ionising radiation

During the past decade, new information about the carcinogenic effects of radiation has come from epidemiological studies of Japanese atomic bomb survivors; patients irradiated therapeutically for ankylosing spondylitis and other conditions; workers exposed to radiation in various occupations; and populations residing in areas of high natural background radiation. New data have also come from long-term studies of the carcinogenic effects of irradiation in laboratory animals and from experiments on neoplastic transformation in cultured cells. The new data have been summarised in reports from NAS/BEIR [1] and UNSCEAR [2].

In many areas of hazard assessment, specific meanings of the word *risk* are avoided and preference is given to words, which more directly indicate the relevant quantity, *e.g. probability, consequence*, and *mathematical expectation* of the consequence. This leaves the word *risk* free to be used in the everyday meaning and makes it possible to include in the risk concept a number of factors which, in addition to those more readily quantifiable, influence decisions on risk acceptance.

With this wider meaning of the word, *risk* is a concept rather than a quantity. The ICRP has in its 1990 recommendations decided to abandon its practice of always strictly using *risk* with the specific meaning of probability and attempts to use instead the more direct term *probability*. This should reduce the ambiguity when describing the probabilities and consequences of an event and makes it easier to communicate with regulatory agencies and others who deal with non-radiation risks as well. For example, the concept of *death probability rate* is used by the ICRP rather than *mortality rate*. The reason is that the rates will be integrated and the integral to be used by the ICRP is the *attributable lifetime probability* of death, related to the *average individual,* rather than the observed or expected number of deaths per 100,000.

The ICRP is mainly concerned with two quantifiable risk quantities:

- P_i which is the *probability* of each harmful effect *i*, *e.g.* lethal or curable cancer or severe hereditary effects;
- W_i which is the *consequence* if the effect occurs. The consequence can be described in a variety of ways, indicating the severity of the effect and its distribution in time.

The *mathematical expectation* of consequence, identical to the average consequence, is:

$$\overline{W} = \sum_i P_i \cdot W_i$$

This quantity is sometimes used in the effort to express the magnitude of the "risk".

A radiation dose will involve a risk commitment, *i.e.* a commitment of an increased cancer death probability rate in the future, after a minimum latent period that may be from a few years in the case of leukaemia to tens of years for other malignant conditions. Any change in the age-specific death probability rate would therefore occur later in life, when the risk of death from other causes is also higher. The risk committed by a radiation dose at a given age can therefore not be added to the background risk at the same age.

The *attributable lifetime probability of death* from radiation exposure has been used by the ICRP, and radiation risks have been expressed in *per cent per sievert*. However, our total probability of death, which is 100%, cannot be increased. The introduction of a new risk source will not change our lifetime probability of death but only the distribution of the probable causes of death. Any increment that a new risk source causes, is an increment to our *death probability rate* at any given age, provided that the person is alive at that age, *i.e*. a conditional probability rate.

A defined exposure scenario may add a *conditional* source-related increment of probability rate, to the background rate. The rate is conditional, because it will be expressed only if the individual is alive at the ages for which it is defined. From this increment, an *unconditional* probability rate can be calculated when a reference time (age) has been defined, *e.g*. the age at the onset of the exposure period. The attributable lifetime probability of death from the source under consideration must therefore be calculated from the unconditional incremental death probability rate, taking account of the probability of reaching each age by considering the likelihood of dying from other causes as well as from radiation. The unconditional incremental probability rate is obtained as the product of the conditional incremental probability rate and the *survival* probability, modified by the incremental radiation risk. Figure A.1 shows the variation of the attributable probability of death with age at the time of exposure [11]. The substantially higher risk for the youngest age group is notable. However, it must be recognised that most of this higher risk will be expressed first at high ages.

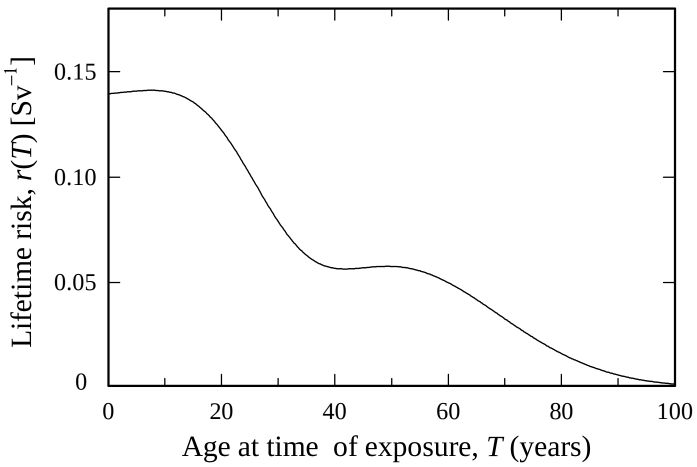

Figure A.1. The attributable lifetime probability of death from a single radiation dose at various ages at the time of exposure.

The lifetime risk function in Figure A.1 is the calculated average for both sexes. In this function the BEIR Committee have reduced the contribution from leukaemia by a dose rate effectiveness factor (DDREF) of 2 (using a linear-quadratic response) whereas for solid tumours a linear response was used, *i.e*. no DDREF-reduction. For high dose, high dose rate the leukaemia contribution should therefore be doubled.

The attributable lifetime risk due to a chronic exposure starting at a given age, T, can be calculated by proper integration of the risk function $r(T)$, the probability of survival at a given age and the chronic dose function. A lifetime dose of 1 Sv starting at age 0 will thus result in an average lifetime risk, r_{rad}, of fatal cancer of about 0.05 Sv^{-1}. The average loss of life expectancy per unit lifetime dose can be calculated as the product of the average lifetime risk, r_{rad}, and the average loss of life expectancy per cancer, l:

$$r_{rad} \cdot l = 0.05\,\text{cancer} \cdot \text{Sv}^{-1} \cdot 15\,\text{years} \cdot \text{cancer}^{-1} \approx 1\,\text{year} \cdot \text{Sv}^{-1}$$

The collective loss of life expectancy from a given collective dose, S_{rad}, can be calculated as:

$$L_{rad} = S_{rad} \cdot r_{rad} \cdot l$$

The collective exposure, S_{rad}, can be expressed over many generations as long as the age distribution of the exposed population does not deviate significantly from the one which has been used to determine the average lifetime risk, r_{rad}.

A.2 Risk from exposure to toxic chemicals

Non-radiological health effects, *e.g.* from exposure to chemical contaminants can in principle be described in the same way as the exposure to radiation as far as stochastic effects are concerned. The attribute for non-radiation exposures should be expressed in a risk scale in order to determine the total expected detriment from the exposure to non-radiological carcinogens. The attributable lifetime risk from an individual lifetime exposure to a specific chemical contaminant can be calculated by a proper lifetime integration of the exposure, the risk per unit exposure of the contaminant as a function of age and the survival function as a function of age. The available information on risk factors for exposure to non-radiological carcinogens is scarcer than for exposure to ionising radiation.

Non-threshold effects
For relatively low intakes of toxic chemicals most likely to occur from environmental exposures, a linear dose-response relationship can be assumed for estimating, R_{chem}:

$$R_{chem} = I_{day} \cdot r_{chem}$$

where R_{chem} is the probability of developing cancer, I_{day} is the exposure in terms of a chronic daily intake averaged over 70 years and per kg body mass ($mg \cdot d^{-1} \cdot kg^{-1}$) and r_{chem} is the average lifetime risk per unit exposure ($mg^{-1} \cdot d \cdot kg$). For general risk assessments, cancer risks from various exposure pathways are assumed to be additive.

The average loss of life expectancy per unit lifetime exposure can be calculated as the product of the average lifetime risk, R_{chem}, and the average loss of life expectancy per cancer, l:

$$R_{chem} \cdot l \quad \left[\text{years/mg} \cdot \text{d}^{-1}\right]$$

The average loss of life expectancy per cancer, l, is about 15 years, irrespectively of the kind of exposure that has caused the cancer. The collective loss of life expectancy from a given collective exposure ($man \cdot mg \cdot d^{-1}$) of a single non-radiological carcinogen, $S_{chem,i}$, can therefore be calculated as:

$$L_{chem,i} = S_{chem,i} \cdot R_{chem,i} \cdot l$$

The total collective loss of life expectancy from a collective exposure to several different non-radiological carcinogens, $S_{chem,i}$, in can thus be calculated as:

$$L_{chem} = l \cdot \sum_i S_{chem,i} \cdot R_{chem,i}$$

The collective exposure integral, S_{chem}, can be expressed over many generations as long as the age distribution of the exposed population does not deviate significantly from the one which has been used to determine the average lifetime risk, R_{chem}.

Threshold effects
Deterministic effects from exposure to ionising radiation are rarely of concern in the case of contaminated land. The potential for threshold effects (somatic effects) from exposure to non-genotoxic chemical substances is evaluated by comparing the exposure level with a reference threshold level for the given health effect. A reference level is defined in terms of ingestion dose, D_{ref}, or air concentration, C_{ref}, below which deterministic health effects are very unlikely. If the exposure level exceeds the reference levels there may be concern for potential deterministic effects. A toxic hazard quotient, *THQ*, has been defined by the USEPA as:

$$THQ = \frac{E}{D_{ref}} \quad \text{or} \quad THQ = \frac{E}{C_{ref}}$$

where E is the exposure level in terms of ingestion or inhalation. The exposure period at which the reference levels D_{ref} and C_{ref} have been determined should also be used for the exposure, E. The greater value of *THQ*, the greater the level of concern ought to be.

In the case of chronic exposure (exposure over a lifetime) a chronic hazard index, *CHI*, may be derived from the ratio of chronic daily intake, I_{day}, to the chronic levels of ingestion dose, D_{ref}, or air concentration, C_{ref}, as:

$$CHI = \frac{CDI}{D_{ref,chron}} \quad \text{or} \quad CHI = \frac{CDI}{C_{ref,chron}}$$

Additivity of the *CHI* for multiple pathways can be considered to be appropriate under certain conditions.

A.3 Risk from a combined radiological and chemical exposure

Combined exposure to radiation and chemical carcinogens should be expressed in a common risk scale in order to determine the total expected detriment from that exposure. The different issues and risk concepts to be addressed in a combined exposure to ionising radiation and toxic chemicals have been presented at a workshop on the effects of residues from uranium mining [3].

Some assumptions are needed in order to assess the impact of a combined exposure of ionising radiation and toxic agents like heavy metals. Two of the more important assumptions are:

- the lifetime cancer risk, r, is linearly related to the exposure, E, also known as the *linearity hypothesis* which can be expressed as $r(E) = k \cdot E$, and
- no synergetic effects exist between exposures to radiological and non-radiological carcinogens, *i.e.* the total lifetime risk of a combined exposure of $E_1 + E_2 + E_3 + \ldots\ldots$ can be described by the sum of risks as $r(E_1 + E_2 + E_3 + \ldots\ldots) = k_1 \cdot E_1 + k_2 \cdot E_2 + k_3 \cdot E_3 + \ldots\ldots$

With these assumptions the total effect of a combined collective exposure to ionising radiation and toxic heavy metals and chemicals can be described as a total collective loss of life expectancy:

$$L_{total} = L_{rad} + L_{chem,1} + L_{chem,2} + L_{chem,3} + \ldots\ldots = L_{rad} + \sum_i L_{chem,i}$$

The relative contributions to the total collective loss of life expectancy from a combined exposure are given by the ratios L_i / L_{total}.

When both deterministic and stochastic health effects involved difficulties are encountered. Several possible approaches have been discussed, *e.g.* by USEPA, CRARM (Commission on Risk Assessment and Risk Management) and WHO. However, a general consensus on a unified approach on the combination of stochastic and deterministic health risks does not yet exist.

A.4 References

1 *Health Effects of Exposure to Low Levels of Ionizing Radiation.* Report by the Committee on the Biological Effects of Ionizing Radiations (BEIR V). National Academy of Sciences/National Research Council, Washington D.C. (1990).

2 *Sources, Effects and Risks of Ionizing Radiation.* United Nations Scientific Committee on the Effects of Atomic Radiation. UNSCEAR 1988 Report to the UN General Assembly, with Annexes. United Nations, New York (1988).

3 *Integrierte Bewertung radiologisher und chemisch-toxischer Kontaminanten.* Proceedings from a workshop on *Sanierung der Hinterlassenschaften des Uranbergbaus,* Dresden, 24 November 1997. Freistaat Sachsen, Staatsministerium für Umwelt und Landesentwicklung, ISSN 0949-8540 (1997).

Annex B. Assessment of weighting factors

A structured approach to optimisation of protection is important to ensure that no important aspects are overlooked and to record the analysis for information and for assessment by others. Reduction of exposure and doses can normally only be achieved by the expenditure of some effort and by allocating additional resources. In such cases it is necessary to decide whether the likely dose saving is worth the effort of achieving that saving. An important step is to identify all options generally aimed at reducing doses and then select those, which deserve further consideration. In order to compare the performance and costs of the options, different quantitative decision-aiding techniques are available. One of these techniques is the multi-attribute utility analysis, which has evolved form several disciplines including psychology, engineering and management science. The essence of this technique is to use a scoring scheme (or a multi-attribute utility function) for the relevant factors (attributes) with the property that if the score is the same for two options there is no preference for one or the other. The option having the highest score is considered to be the best (optimum) amongst those considered in the analysis.

The use of utility functions allows introduction of factors, which are not easy to quantify in monetary terms as is required in cost-benefit analysis. The utilities and weighting factors can be expressed in an additive form to give an overall evaluation of the “total utility” for each of the alternative strategies or options, *i*:

$$U_i = \sum_{j=1}^{n} w_j u_{ij}$$

where U_i is the total utility of option *i*, w_j is the weight assigned to the attribute *j*, and u_{ij} is the utility of the *n* factors associated with each of the alternatives *i* on attribute *j*. The determination of weighting factors is a very difficult task. Different decision-makers might come up with rather different sets of weighting factors for the same attribute. Therefore, there is a need for a systematic assessment of weighting factors and a simple scaling method is proposed in the following sections.

B.1 Weighting factors for major attributes

The primary or major attributes considered in this study are the *economic*, the *health related* and the *social* attributes, which are difficult to determine as they are ‘measured’ in different units. The methodology used here is to establish conversion/scaling constants between the weighting factors that can be expressed as:

$$\frac{w_{economic}}{w_{health}} = C_1 \quad \text{and} \quad \frac{w_{social}}{w_{health}} = C_2$$

The sum of the weighting factors for the major attributes should be 1:

$$w_{economic} + w_{social} + w_{health} = 1$$

which would determine the weighting factors as:

$$w_{health} = \frac{1}{1 + C_1 + C_2} \quad \text{and} \quad w_{economic} = \frac{C_1}{1 + C_1 + C_2} \quad \text{and} \quad w_{social} = \frac{C_2}{1 + C_1 + C_2}$$

The value of C_1 can be determined from the following ratio if the population is exposed only to ionising radiation:

$$C_1 = \frac{w_{economic}}{w_{health}} \cong \frac{w_{economic}}{w_{dose,pop}} = \frac{R_{economic}}{\alpha \cdot R_{dose,pop}}$$

The parameters R_{dose} and $R_{economic}$ denote the range of the collective doses to the affected population and the range of monetary costs, including the equivalent cost of the collective dose to the workers engaged in the remediation, over the remediation options, respectively. If the affected population is exposed also to non-radiological carcinogens, *e.g.* heavy metals, the total detriment in terms of collective loss of life expectancy from cancers attributable to the combined exposure could be described in the following way (see Annex A).

Let the collective radiological and non-radiological exposure integrals to the affected population be:

$$S_{rad}\ [\mathrm{man}\cdot\mathrm{Sv}],\ S_{non-rad,1}\ [\mathrm{man}\cdot\mathrm{kg}],\ S_{non-rad,2}\ [\mathrm{man}\cdot\mathrm{kg}],\ ....\ S_{non-rad,n}\ [\mathrm{man}\cdot\mathrm{kg}]$$

The collective loss of life expectancy, *L*, from the combined exposure can be calculated as:

$$L = l\cdot\left\{r_{rad}\cdot S_{rad} + r_{non-rad,1}\cdot S_{non-rad,1} + r_{non-rad,2}\cdot S_{non-rad,2} + ...+ r_{non-rad,n}\cdot S_{non-rad,n}\right\}$$

l is here the statistical loss of life expectancy per cancer (approximately 15 years) and r is the risk factor per unit exposure integral of Sv or kg for the non-radiological exposure. If society is willing to spend an amount of money equal to the GNP (or even several times the GNP) per capita to avert a loss of one year of life expectancy ($\alpha \approx l \cdot r_{rad} \cdot GNP$) the value of the parameter C_1 can be calculated as:

$$C_1 = \frac{R_{economic}}{l\cdot GNP\cdot(r_{rad}\cdot R_{rad} + \sum_i r_{non-rad,i}\cdot R_{non-rad,i})}$$

R_{rad} and $R_{non\text{-}rad,i}$ are here the ranges of the collective radiation dose and collective non-radiological exposure integrals for each non-radiological carcinogen, *i*, over all the different remediation options.

The social factors considered in this study are *disturbance*, *reassurance* and *loss/gain of income.* It is assumed that the dominating social factor is reassurance because of its more or less permanent nature. Furthermore, it is assumed that the reassurance and radiation health factors are linked in the following way. A *decreasing* reassurance can be interpreted as an *increasing* anxiety and thus *an increasing risk* of psychological harm. A *decreasing* dose level can also be taken to result in an *increasing* reassurance and the risk of psychological harm would consequently be *proportional* to the level of residual dose, *i.e.* the larger the residual dose the larger the risk of psychological harm in the affected population. The risk of radiation induced stochastic (somatic) health effects, r_{rad}, is proportional to the residual dose (0.05 Sv^{-1}). If it were possible to determine the risk of psychological effects per unit residual dose, r_{psy}, in terms of loss of life expectancy the scaling factor, C_2, could be determined as:

$$C_2 = \frac{w_{social}}{w_{health}} \approx \frac{r_{psy}}{r_{rad}}$$

as reassurance is assumed to be the dominating social factor.

Intuitively, the value of C_2 would be expected to be less than one and probably significantly less than one. However, the experience gained after the Chernobyl accident was that socio-psychological factors were given much higher weight than radiation factors, which indicates that the value of C_2 would be higher than one. However, this value judgement will completely depend on the specific situation. In a non-accidental situation like remediation of the example sites with small exposures of the affected population the social factors would probably be given far less weight than in a major accidental situation like Chernobyl. Consequently, the value of the scaling factor C_2 is in this study assumed to be less than 1, *e.g.* 0.2 - 0.3.

B.2 Weighting factors for health sub-attributes

Health sub-attributes in relation to site restoration include health effects from exposure of the population and workers to both radiological and non-radiological carcinogens as well as from accidents due to the remedial measures at the site. The health attributes considered here include *radiation induced stochastic health effects* to the affected population and workers and *non-radiation induced stochastic* *health effects* to the affected population. The conversion/scaling constants for the health attributes can be expressed as:

$$\frac{w_{dose,pop}}{L_{dose,pop}} = \frac{w_{dose,work}}{L_{dose,work}} = \frac{w_{non-rad,pop}}{L_{non-rad,pop}} = C$$

where $L_{dose,pop}$, $L_{dose,work}$, and $L_{non\text{-}rad,pop}$ is the range of the collective loss of life expectancy from radiation exposure of the population, from radiation exposure of the work force and from non-radiological exposure of the population, respectively. The sum of the weighting factors for the health sub-attributes should be 1:

$$w_{dose,pop} + w_{dose,work} + w_{non-rad} = 1$$

which would determine the scaling constant, C, as:

$$C = \frac{1}{L_{dose,pop} + L_{dose,work} + L_{non-rad,pop}}$$

As the range of collective loss of life expectancy, L, is given as the product of the range of collective exposure, R, the risk per unit exposure, r, and the loss of life expectancy per cancer, l, the weighting factors can be determined as:

$$w_{dose,pop} = C \cdot R_{dose,pop} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$
$$w_{dose,work} = C \cdot R_{dose,work} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$
$$w_{non-rad} = C \cdot R_{non-rad,pop} \cdot l \cdot r_{non-rad}$$

The value of $l \cdot r_{rad}$ is approximately 1 year per sievert.

B.3 Weighting factors for economic sub-attributes

Economic sub-attributes include the *monetary costs of the remediation operation* including labour costs, the *monetary costs of waste disposal* including the transport of the waste, *loss/gain of taxes* to society due to loss/gain of income and *monetary costs of monitoring the remedial options*. The conversion/scaling constants for the economic sub-attributes can be expressed as:

$$\frac{w_{remedia}}{R_{remedia}} = \frac{w_{waste}}{R_{waste}} = \frac{w_{monitor}}{R_{monitor}} = \frac{w_{tax}}{R_{tax}} = C$$

where R_i is the cost range of the given sub-attribute, i over all the different remediation options. The sum of the weighting factors for the health sub-attributes should be 1:

$$w_{remedia} + w_{waste} + w_{monitor} + w_{tax} = 1$$

which would determine the scaling constant, C, as:

$$C = \frac{1}{R_{remedia} + R_{waste} + R_{monitor} + R_{tax}}$$

The weighting factors can then be determined as:

$$w_{remedia} = C \cdot R_{remedia}$$

$$w_{waste} = C \cdot R_{waste}$$

$$w_{monitor} = C \cdot R_{monitor}$$

$$w_{tax} = C \cdot R_{tax}$$

B.4 Weighting factors for social sub-attributes

The social sub-attributes considered in this study include *reassurance*, *disturbance* and *loss/gain of income*. The conversion/scaling constants for the social sub-attributes can be expressed as:

$$\frac{w_{reas}}{w_{distur}} = C_1 \quad \text{and} \quad \frac{w_{loss}}{w_{distur}} = C_2$$

The sum of weighting factors should be 1:

$$w_{distur} + w_{reas} + w_{loss} = 1$$

which would determine the weighting factors as:

$$w_{distur} = \frac{1}{1 + C_1 + C_2} \quad \text{and} \quad w_{reas} = \frac{C_1}{1 + C_1 + C_2} \quad \text{and} \quad w_{loss} = \frac{C_2}{1 + C_1 + C_2}$$

It is assumed that reassurance is given a considerably higher weight than the weight given to loss/gain of income due to permanent nature of reassurance. Furthermore, it is assumed that the weight given to disturbance is considerably lower than the weight given to loss/gain of income due to the transitional nature of the disturbance. Although loss/gain of income also is transitional, its duration would probably be longer than that for disturbance. The following hierarchy of the weighting factors for the social sub-attributes is assumed:

$$w_{reas} > w_{loss} > w_{distur}$$

and it is proposed here that $C_1 \approx 5$ - 7 and $C_2 \approx 2$ - 3.

Further research studies are needed before qualified value settings of weighting factors for social sub-attributes can be done. Such research should be performed in close collaboration between experts in the fields of radiation protection and social and psychological sciences.

Annex C. Sensitivity calculations for Molse Nete River

The best (optimised) strategy or option amongst of set of strategies expressed by the overall score, $U_i(x)$, depends on the utility functions, $u(x)$, and weighting factor, w, for each utility. Sensitivity calculations have been made in which different distributions have been assigned to the utility values, x, and the weighting factors, w. In addition, correlations between utility values have been assumed. Five different cases have been investigated:

(1) Uniform distribution function of utility values of the attributes, x, between 1.5^{-1} - $1.5 \times$ the central value of x; triangular distribution function of weighting factors, w, between 0 and 1 with central value of w as the most probable value; no correlations between utility values.

(2) Uniform distribution function of utility values of the attributes, x, between 1.5^{-1} - $1.5 \times$ the central value of x; uniform distribution function of weighting factors, w, between 1.5^{-1} - $1.5 \times$ the central value of w; no correlations between utility values.

(3) Uniform distribution function of utility values of the attributes, x, between 1.5^{-1} - $1.5 \times$ the central value of x; triangular distribution function of weighting factors, w, between 1.5^{-1} - $1.5 \times$ the central value of w; central value of w the most probable value; no correlations between utility values.

(4) Triangular distribution function of utility values of the attributes, x, between 1.5^{-1} - $1.5 \times$ the central value of x; central value of x the most probable value; triangular distribution function of weighting factors, w, between 1.5^{-1} - $1.5 \times$ the central value of w; central value of w the most probable value; no correlations between utility values.

(5) Triangular distribution function of utility values of the attributes, x, between 1.5^{-1} - $1.5 \times$ the central value of x; central value of x the most probable value; triangular distribution function of weighting factors, w, between 1.5^{-1} - $1.5 \times$ the central value of w; central value of w the most probable value; negative correlation between the collective dose and costs of remediation ($r = -0.8$).

Sensitivity ranking of the assumptions made in the calculations for the five different cases has been estimated. The results of the calculations are shown in Tables C.1 - C.5 and in Figures C.1 - C.5.

In all the cases except for Case 1 the sensitivity of the scores A - G2 is dominated (> 10%) by the weighting factors for health, economics, monitoring costs, waste disposal costs, remediation costs and social factors. Changing the variation range of the weighting factors to $[1.5^{-1} \times w_{central}; 1.5 \times w_{central}]$ from [0;1] had a dramatic influence on the scores, both regarding their value and the uncertainty band. Less dependence was observed on the type of distribution function assigned to the weighting factors (uniform or triangular distribution). Changing the distribution type from uniform to triangular for the utility values of the attributes did result in a more narrow uncertainty band of the scores as would have been expected. Introducing a negative correlation between remediation costs and collective dose did not change the uncertainty bands but resulted in more precise (smooth) distributions of each of the scores, except of course for option A for which there is no remediation costs.

Based on the conclusions from the sensitivity analysis, a triangular distribution has been used for both utility values and weighting factors for all the example sites. The triangular probability distribution of each utility value has been taken to be (0, maximum, 0) for utility values, x, of $(1.5^{-1} \times x_{central}, x_{central}, 1.5 \times x_{central})$. The triangular probability distribution of the weighting factors has been taken to be (0, maximum, 0) for weighting factor values, w, of $(1.5^{-1} \times w_{central}, w_{central}, 1.5 \times w_{central})$. The weighting factors are truncated at 0 when the value $1.5 \times w_{central}$ exceeds 1.

CASE 1

Table C.1. Distribution functions for attributes and weighting factors and sensitivity ranking of the assumptions being most important in the calculations of scores A - G2 for the Molse Nete River site.

Distribution function for attributes	Distribution function for weighting factors	Sensitivity to scores A - G2	
		Assumption	Sensitivity
Uniform distribution 1.5^{-1} - $1.5 \times$ central values in Table 4	Triangular distribution between 0 and 1 and central values in Table 5 as most probable values	Monitor costs weight	44%
		Health weight	41%
		Worker dose weight	41%
		Economic weight	29%
		Reassurance weight	21%
		Population dose weight	21%
		Disturbance weight	16%
		Income loss weight	13%
		Waste disposal costs weight	12%
		Remediation costs weight	11%
		Tax loss weight	10%
		Social weight	3%
		All remaining parametes	< 2%

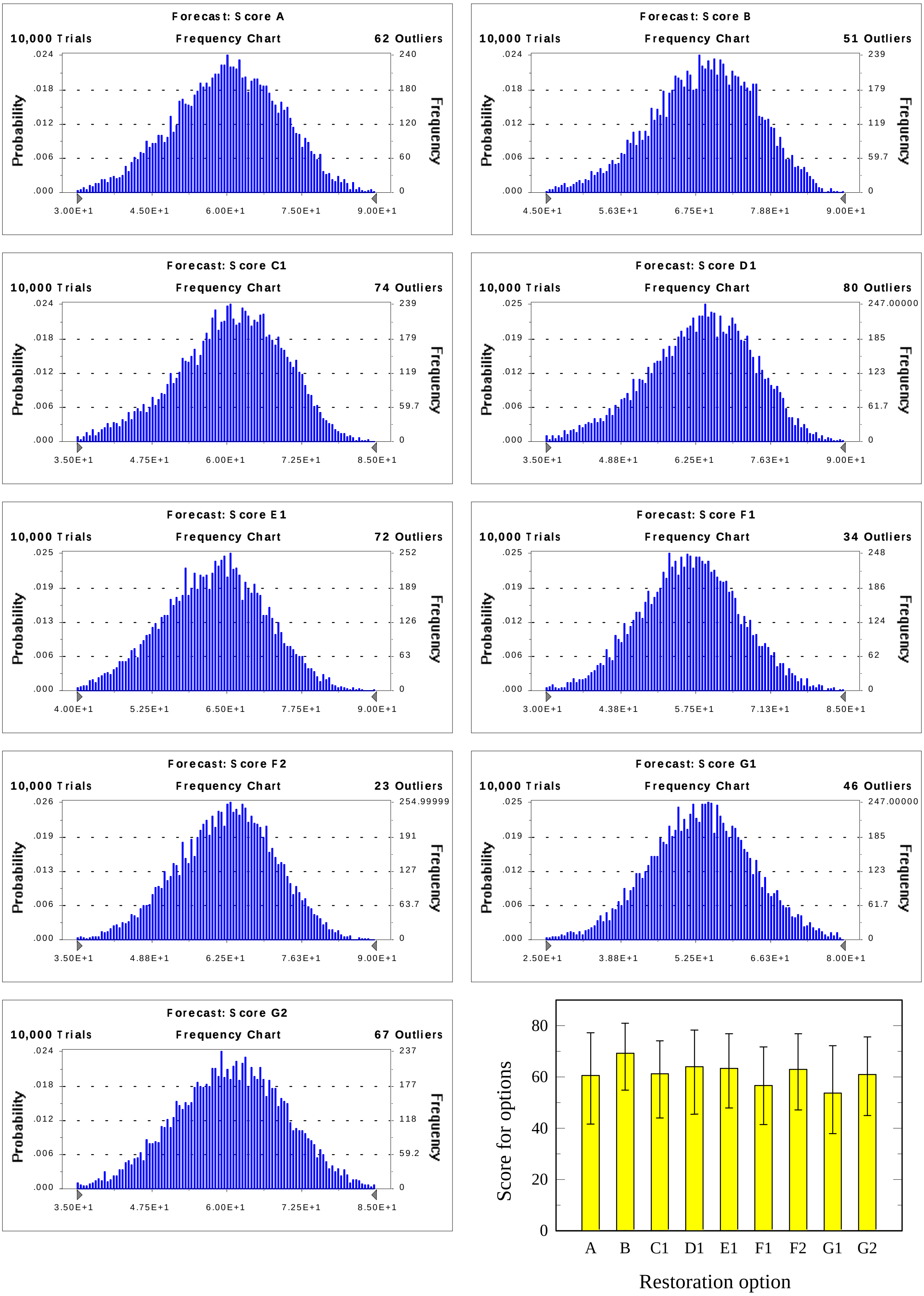

Figure C.1. Distribution functions of the scores for remediation options at the Molse Nete River site for CASE 1. The central value and the 5 - 95 percentiles for each of the options are shown in the lower right picture.

CASE 2

Table C.2. Distribution functions for attributes and weighting factors and sensitivity ranking of the assumptions being most important in the calculations of scores A - G2 for the Molse Nete River site.

Distribution function for attributes	Distribution function for weighting factors	Sensitivity to scores A - G2	
		Assumption	Sensitivity
Uniform distribution 1.5^{-1} - $1.5 \times$ central values in Table 4	Uniform distribution 1.5^{-1} - $1.5 \times$ central values in Table 5	Health weight	67%
		Economic weight	60%
		Monitor cost weight	32%
		Waste disposal costs weight	20%
		Remediation costs weight	13%
		Social weight	7%
		Reassurance weight	5%
		Remediation costs B	3%
		Income loss weight	3%
		Income loss B	2%
		All remaining parameters	< 2%

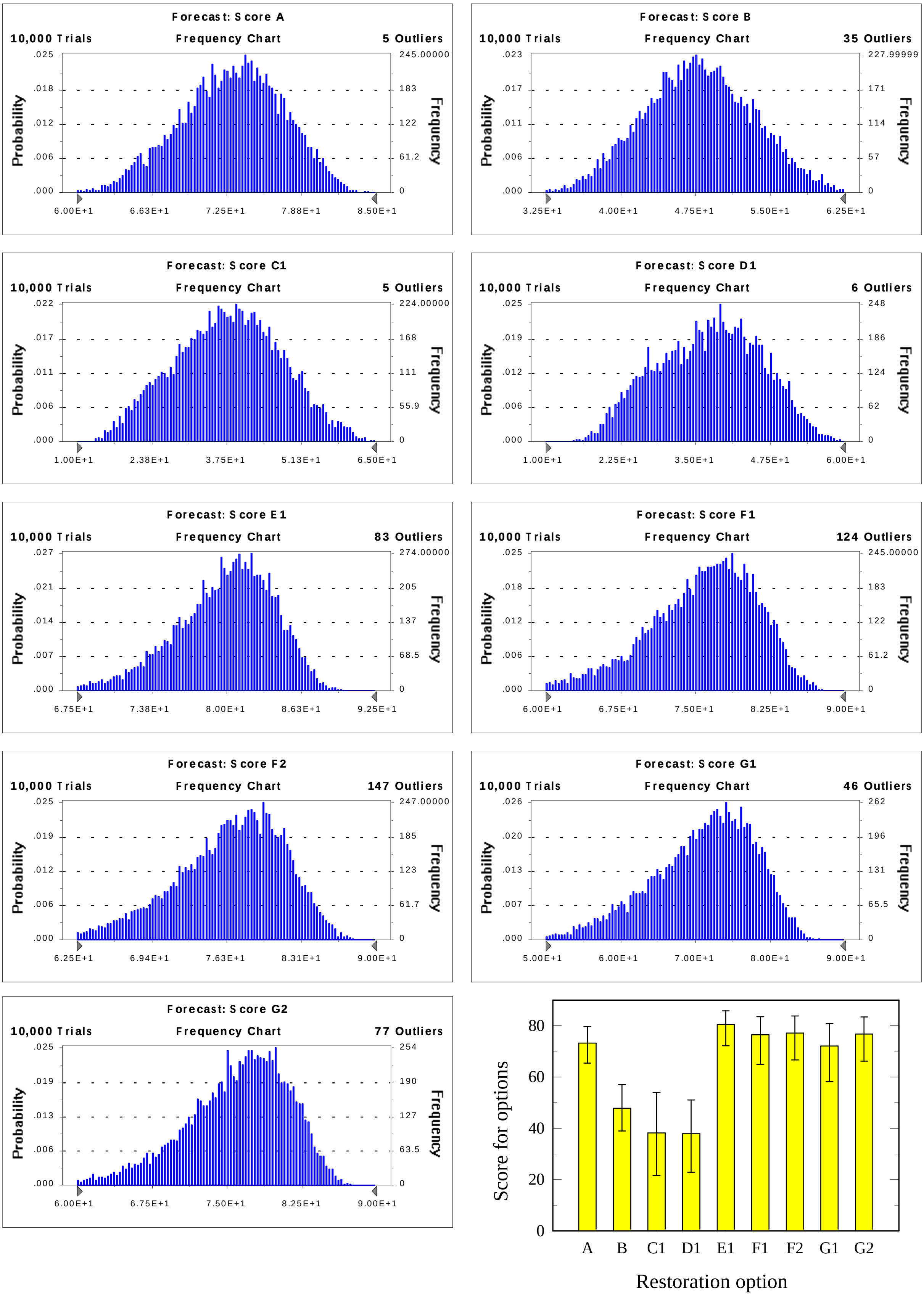

Figure C.2. Distribution functions of the scores for remediation options at the Molse Nete River site for CASE 2. The central value and the 5 - 95 percentiles for each of the options are shown in the lower right picture.

CASE 3

Table C.3. Distribution functions for attributes and weighting factors and sensitivity ranking of the assumptions being most important in the calculations of scores A - G2 for the Molse Nete River site.

Distribution function for attributes	Distribution function for weighting factors	Sensitivity to scores A - G2	
		Assumption	Sensitivity
Uniform distribution 1.5^{-1} - $1.5 \times$ central values in Table 4	Triangular distribution 1.5^{-1} - $1.5 \times$ central values in Table 5 central value in Table 5 most probable value	Health weight	68%
		Economic weight	57%
		Monitor costs weight	32%
		Waste disposal costs weight	18%
		Remediation cost weight	14%
		Social weight	10%
		Reassurance weight	6%
		Income loss weight	3%
		Tax loss F2	2%
		All remaining parameter	< 2%

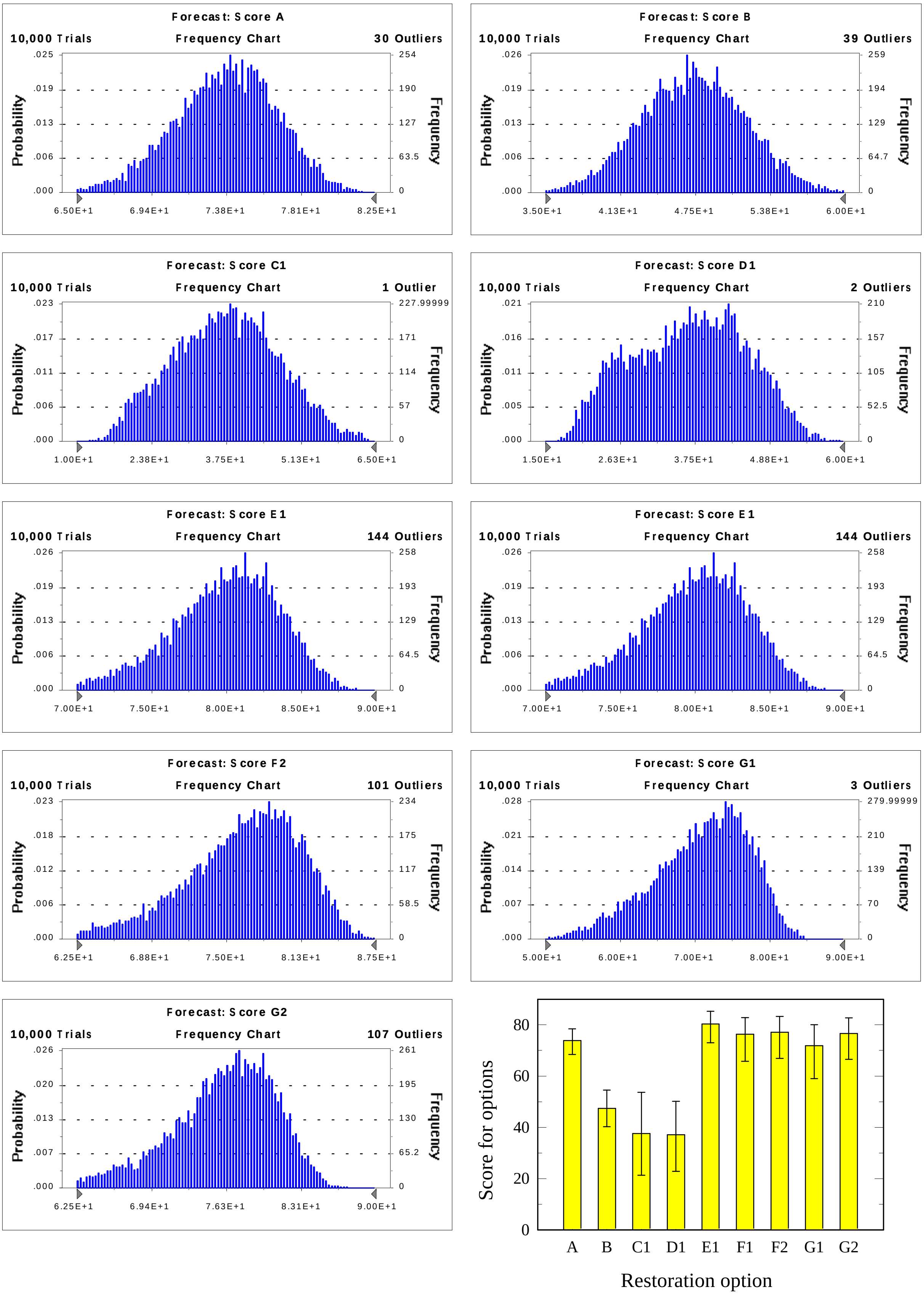

Figure C.3. Distribution functions of the scores for remediation options at the Molse Nete River site for CASE 3. The central value and the 5 - 95 percentiles for each of the options are shown in the lower right picture.

CASE 4

Table C.4. Distribution functions for attributes and weighting factors and sensitivity ranking of the assumptions being most important in the calculations of scores A - G2 for the Molse Nete River site.

Distribution function for attributes	Distribution function for weighting factors	Sensitivity to scores A - G2	
		Assumption	Sensitivity
Triangular distribution 1.5^{-1} - 1.5 × central values in Table 4 central value in Table 4 most probable value	Triangular distribution 1.5^{-1} - 1.5 × central values in Table 5 central value in Table 5 most probable value	Health weight	67%
		Economic weight	58%
		Monitor costs weight	32%
		Waste disposal costs weight	20%
		Remediation costs weight	12%
		Social weight	7%
		Reassurance weight	5%
		Tax loss weight	4%
		Income loss weight	3%
		Remediation costs F2	2%
		All remaining parameters	< 2%

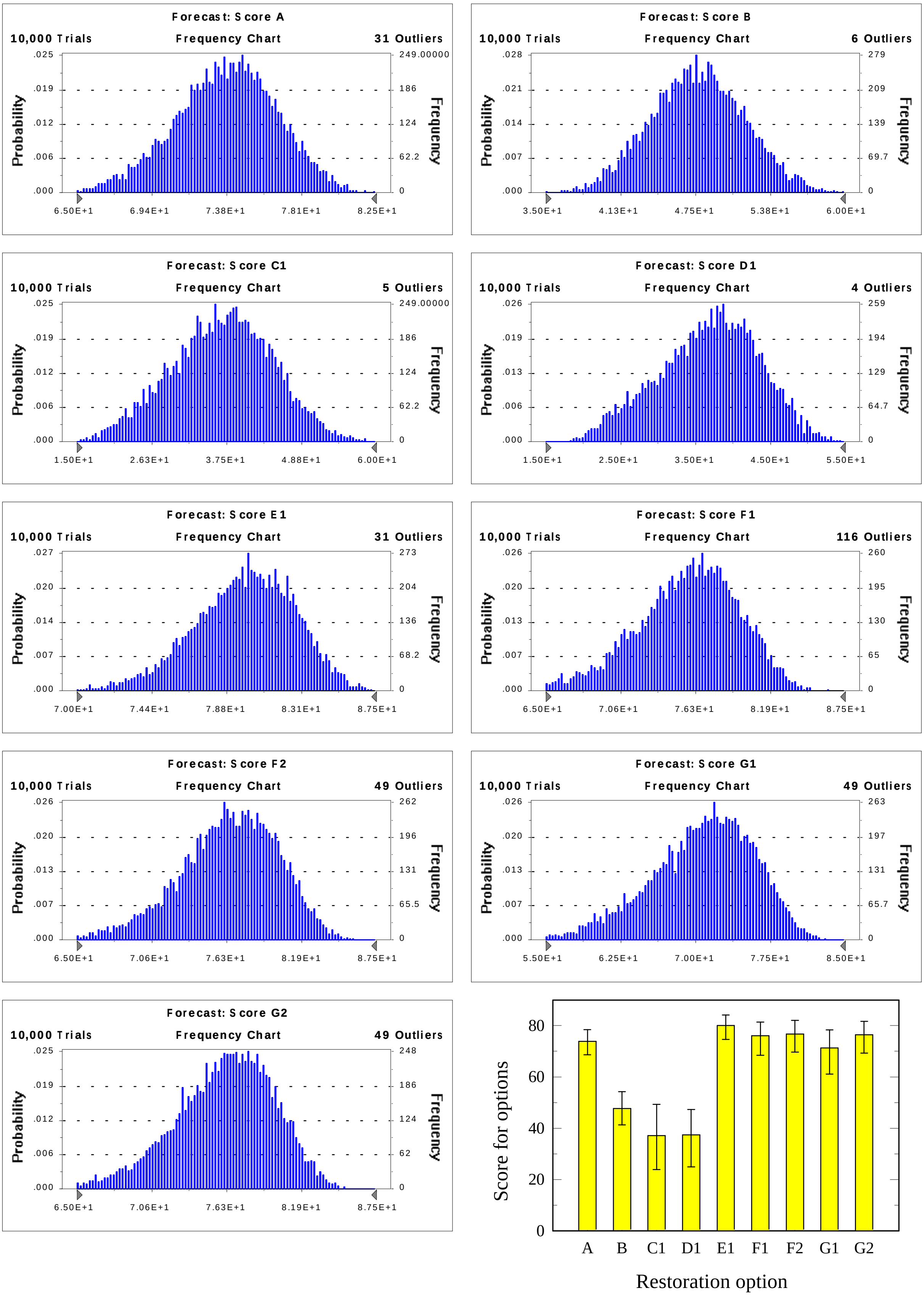

Figure C.4. Distribution functions of the scores for remediation options at the Molse Nete River site for CASE 4. The central value and the 5 - 95 percentiles for each of the options are shown in the lower right picture.

CASE 5

Table C.5. Distribution functions for attributes and weighting factors and sensitivity ranking of the assumptions being most important in the calculations of scores A - G2 for the Molse Nete River site.

Distribution function for attributes	Distribution function for weighting factors	Sensitivity to scores A - G2	
		Assumption	Sensitivity
Triangular distribution 1.5^{-1} - $1.5 \times$ central values in Table 4 central value in Table 4 most probable value Collective doses are correlated to remediation costs ($r = -0.8$)	Triangular distribution 1.5^{-1} - $1.5 \times$ central values in Table 5 central value in Table 5 most probable value	Health weight	68%
		Economic weight	57%
		Monitor costs weight	32%
		Waste disposal costs weight	18%
		Remediation costs weight	12%
		Social weight	7%
		Reassurance weight	5%
		Income loss weight	3%
		Tax loss weight	3%
		Income loss G1	3%
		Monitor costs G2	2%
		All remaining parameters	< 2%

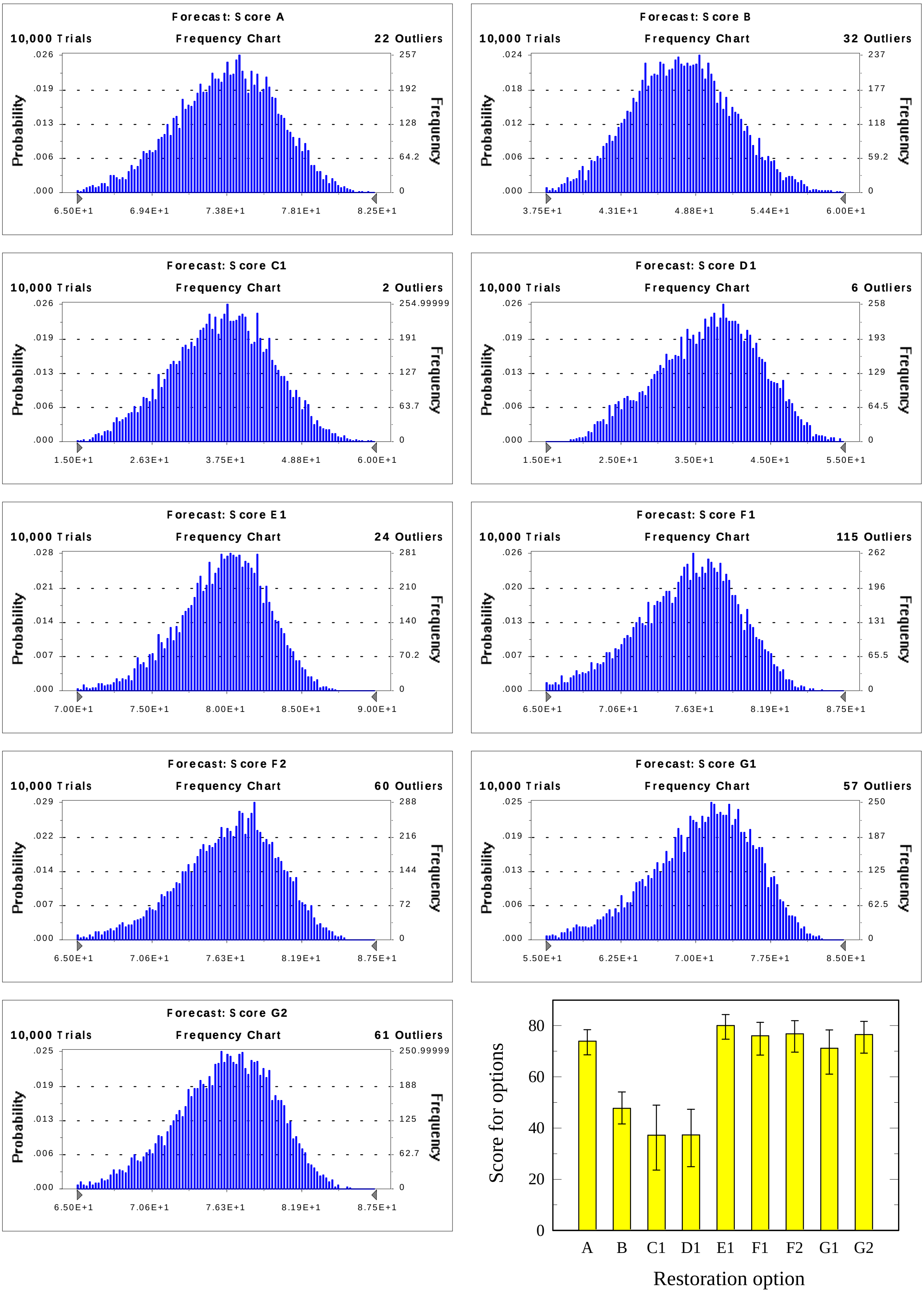

Figure C.5. Distribution functions of the scores for remediation options at the Molse Nete River site for CASE 5. The central value and the 5 - 95 percentiles for each of the options are shown in the lower right picture.