# A PROBABILISTIC RISK ASSESSMENT OF AN UNDERGROUND RADIOACTIVE WASTE DISPOSAL FACILITY

B.G.J. THOMPSON

Department of the Environment,
London, United Kingdom

ABSTRACT:

A probabilistic risk analysis is demonstrated for a single groundwater release scenario from a hypothetical repository for intermediate level wastes situated in a clay layer at a depth of about 150 metres under Harwell. This is the first stage of development of an overall methodology which will eventually treat combinations of risks due to multiple release scenarios with parameter values whose uncertainty varies with time.

It is shown that upper bound estimates of risk are unlikely to be useful and that the approach to radiological risk assessment based upon 'best estimates' is difficult to justify. Consequently, a full probabilistic risk analysis is necessary although further development of statistical sampling and data aquisition techniques and also of methods for the generation and analysis of site evolution scenarios, is necessary.

## 1. INTRODUCTION

### 1.1 Background

During 1985, the trial assessment (1) of a postulated intermediate level radioactive waste disposal facility, at a depth of 150 metres below the AERE Harwell site, was undertaken by the UK Government Department of the Environment as part of its research programme on radioactive waste management. This rehearsal (DRY RUN 1) is the first in a series designed to develop and test a methodology for probabilistic risk assessment (p.r.a) against a principal target of individual risk of $10^{-6}$ per annum during the post-closure period, as specified by the UK Regulatory Authorities (2). A single groundwater transport scenario is considered with parameter values and uncertainties that are assumed invariant with time.

An earlier paper (3) considered generic conditions and DRY RUN 1 develops the approach to the pre-site investigations stage of assessment, demonstrating the manner in which the site is characterised uncertainties estimated and iterations are made between detailed-deterministic calculations (of groundwater flow and transport, geochemical speciation etc.) and p.r.a. using simplified models in the UK SYVAC 'A/C' framework of Monte Carlo simulations. (4)

### 1.2 Previous work

Existing assessments such as KBS3 and Project Gewahr were based upon deterministic calculations of doses due to a selection of discrete scenarios. The uncertainties associated with parameter values for any given scenario, due to errors in measurement at specific locations spatial and temporal variations were not considered, nor the probabilities of occurrence of the scenarios themselves. Atomic Energy of Canada Ltd (AECL) were using their SYVAC-based approach but risk was not considered as a performance criterion. In the United States, the Performance Assessment National Review Group (PANRG) had reccomended (5) a rigorous probabilistic analysis of disposal concepts although the regulations were not formulated with risk in mind. Both risk and dose criteria are considered in a recent paper (6) by the Nuclear Energy Agency (NEA). A full risk analysis procedure is not demonstrated under realistic circumstances however and hence the present paper provides a summary of the first documented p.r.a. of an underground disposal facility (1).

### 1.3 The objectives of the trial assessments

The main objectives of the Dry Run assessment exercise were as follows:

(a) To establish an overall radiological assessment procedure bringing together different individual disciplines, models and techniques.

(b) To identify the different types and quantities of information required at each stage in an assessment, together with their sources and availability. In addition, methods of collection, justification, use and storage of data would be considered.

(c) To formulate an appropriate means of presenting the results of an assessment.

## 2. THE ASSESSMENT PROCEDURE

### 2.1 The stages of a site assessment

A detailed flow chart description is given in (1) identifying six stages once the assessment criteria and the waste inventory have been established:

(a) A generic stage before specific sites are identified, as considered earlier in Reference (3) and in the study of Best Practicable Environmental Options (7);

(b) A pre-site investigation stage when a short list of potential sites has been identified by the UK Industry Nuclear Waste Management Executive (Nirex) but for which no detailed site investigations have been made. This stage is assumed in DRY RUN 1;

(c) The site investigation stage following approval by Parliament for Special Development Orders to permit Nirex to explore the sites;

(d) Post site investigation stage. The results of (c) and of additional research are used in preparing a detailed safety case and in the subsequent independent radiological assessment by DoE. If the case succeeds on these and other grounds a Public Inquiry will decide if construction may proceed on one site. ALARA and preclosure considerations are of course involved but not treated here;

(e) Construction and emplacement phase;

(f) Post institutional management phase.

Both (e) and (f) may require radiological analyses to support authorisation to dispose of wastes and to interpret the results of any long-term monitoring that is carried out.

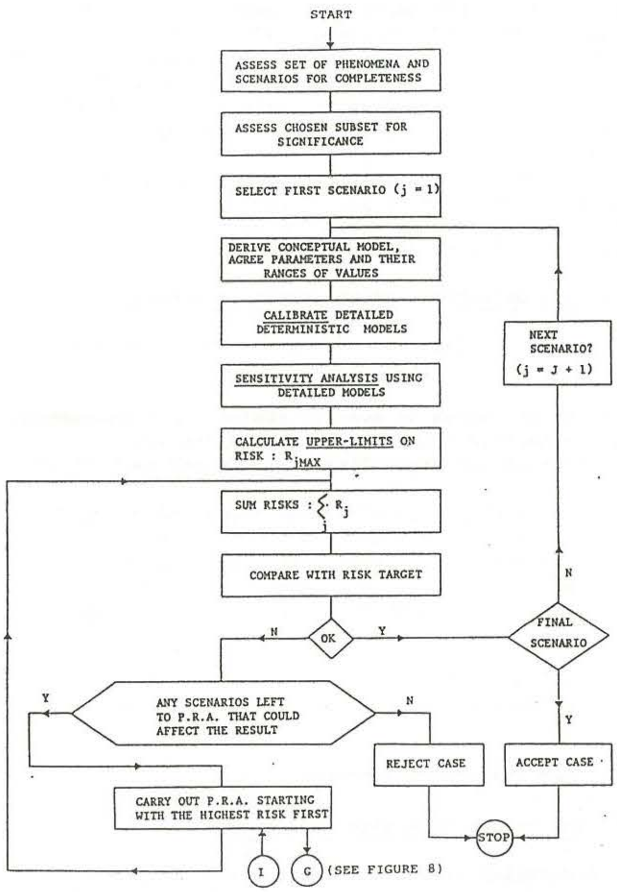


FIGURE 1 : AN OUTLINE PROCEDURE FOR POST-CLOSURE RADIOLOGICAL RISK ANALYSIS

### 2.2 Post closure radiological risk analysis

In principle, the procedure shown in Figure 1 can be used at each stage of the site assessment outlined above. For each ($j^{th}$) scenario selected an upper limit to risk ($R_j$max) may be estimated and if at any stage, the summed risk ( $\sum_j$ Rj) exceeds the risk target then a full p.r.a. must be undertaken. This must start with the highest risk scenario first. If the target is not exceeded when all scenarios have been considered the case is likely to succeed at the stage of assessment concerned.

### 2.3 Calculating risk

In the general case of a non-linear dose-parameter relationship, the dose (committed effective dose equivalent) $H(\bar{x})$ due to the average values $(\bar{x})$ of parameters (x) is not the same as the average dose $(\bar{H})$ from the full dose probability distribution. That is,

$$H(\bar{x}) \neq \bar{H} = \int_x p(x) \, . \, H(x) \, dx \qquad [1]$$

hence the expected value of risk $(\bar{R})$ cannot be obtained from deterministic calculations of dose using mean values of inputs multiplied by the ICRP risk factor ($\gamma$). That is,

$$\bar{R} = \gamma . \bar{H} \neq \gamma . H(\bar{x}) . \qquad [2]$$

Existing methods relying upon 'best estimates' of parameter values with single deterministic calculations for each chosen scenario are unsatisfactory also on two further counts:

(a) It is unclear whether the estimator is based upon median ($x_{0.5}$), modal ($x_m$), arithmetic mean ($\bar{x}$) or geometric mean ($x_{gm}$) values or indeed upon some other average measure of the joint probability distribution (p.d.f) of the defining variables, and secondly,

(b) no attempt is made to explicitly derive this p.d.f.

It is shown later in Section 6.1 that (conditional) risks calculated using extreme values from the parameter ranges are so pessimistic as to be of little practical value and so it is essential to have the capability of full p.r.a.

### 2.4 Information requirements and expert judgement

Three levels of detail of models have been identified and the information required for assessment purposes has been specified accordingly (8). It is emphasized that many quantities used in assessments over the typically long time scales involved cannot be obtained except upon the basis of expert judgements. In particular, at present, the parameters used in p.r.a. simulations are gross spatial and temporal averages over an entire biosphere compartment, or over an identifiable segment of a repository or over a migration path in the geosphere. Their probability distributions are not therefore the same as the sample distributions from a single borehole (say), but represent degrees of belief about the averages concerned. Furthermore, the probability of human intrusion into a repository seems difficult to estimate except upon the basis of judgements of this kind.

## 3. THE HYPOTHETICAL DISPOSAL CONCEPT

### 3.1 The Harwell site

The AERE site is situated approximately midway between Newbury and Oxford within the Thames catchment on the Northern flank of the Berkshire

Downs. Details of the regional geology and of the limited bore hole investigation of the site itself are given in a series of British Geological Survey (BGS) reports (eg, reference 9). The litho-stratigraphy under Harwell is summarised in Table 1.

TABLE 1 : HARWELL STRATIGRAPHY

| FORMATIONS | DEPTH BELOW GROUND LEVEL (z) metres | HYDRAULIC CONDUCTIVITY (k) |
|---|---|---|
| Lower Chalk<br>Upper Greensand | 0 - 60<br>60 - 85 | $k = 10^{-6}$ m/s |
| Gault Clay<br>Lower Greensand<br>Kinneridge Clay | 85 - 155<br>155 - 160<br>160 - 190 | $\log_{10} k = -(0.0135z + 9.3)$ m/s |
| Corallian Aquifer | 190 - 220 | $k = 10^{-6}$ m/s |
| Oxford Clay | 220 - 330 | Assumed impermeable |

The site was chosen for the Dry Run because:

(a) The geology and hydrogeology are potentially suitable for a repository at a depth of about 150 metres in the Gault Clay;

(b) Additional borehole logging data could in principle be obtained; and

(c) this site is one adopted by the Commission of European Communities (CEC) PACOMA research programme and will permit a comparison of different radiological assessment methodologies during 1987-8.

### 3.2 The repository design

For the purpose of demonstration it was assumed that Industry had proposed a repository in the Gault Clay because of economic advantages compared with the construction and operation of a facility in the deeper Oxford Clay.

The engineering concept is based upon the horizontal tunnel design scheme 'B' by Griffin (10) a section through which is shown in Figure 2. The circular crossection is stacked with 34 steel drums of intermediate level wastes encapsulated in cement and is backfilled with concrete. The liner is 1 metre thick concrete with an inside diameter of 6 metres.

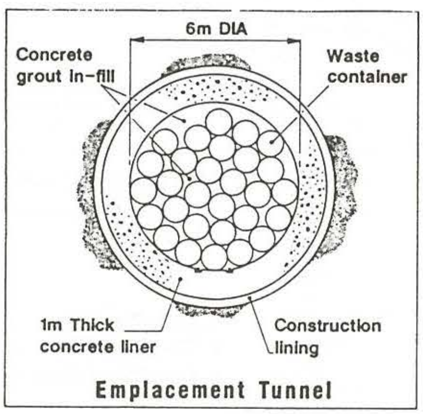


FIGURE 2 : REPOSITORY LAYOUT - SCHEME 'B'

### 3.3 The inventory of intermediate level wastes

The full derivation of radionuclide compositions for eleven waste streams is given by Thorne (11) and includes Magnox swarf and sludges, PCM, alumino-ferric hydroxide floc, THORP hulls and fines, SIXEP wastes, and $C^{14}$ from Magnox decommissioning. Sixty-four nuclides were identified but later reduced to 23, as shown in Table 2, after considering activity, radiotoxicity and half-life.

### 3.4 Chemical and geochemical characteristics

Calculations of nearfield solubility were made for Thorium, Americium, Plutonium, Neptunium and Uranium using the MINEQL program, assuming a pH of 12.7 in cement porewater for later comparison with laboratory experiments. Tc, I, Cs, $H^3$ and $C^{14}$ were assumed limited by availability rather than by solubility and their maximum concentrations set to $10^{-2}$M, whilst the values for Ni, Se, Sn, Ra and Pa were estimated from the literature because no reliable thermodynamic database was available.

TABLE 2 : INTERMEDIATE LEVEL WASTE INVENTORY

| NUCLIDE | HALF-LIFE (years) | TOTAL (GBq) ACTIVITY |
|---|---|---|
| H - 3 | 1.235E1 | 2.373E4 |
| C - 14 | 5.730E3 | 1.062E6 |
| Ni - 59 | 7.499E4 | 1.461E6 |
| Co - 60 | 5.217E0 | 7.388E7 |
| Ni - 63 | 9.600E1 | 2.793E8 |
| Se - 79 | 6.500E4 | 1.356E3 |
| Sr - 90 | 2.910E1 | 2.065E8 |
| Tc - 99 | 2.130E5 | 5.653E4 |
| Sn - 126 | 1.000E5 | 2.094E3 |
| I - 129 | 1.570E7 | 1.129E3 |
| Cs - 135 | 2.300E6 | 2.084E2 |
| Cs - 137 | 3.000E1 | 1.522E8 |
| Ra - 226 | 1.600E3 | 1.033E-2 |
| Th - 229 | 7.340E3 | 6.200E-5 |
| Th - 230 | 7.699E4 | 2.100E0 |
| Pa - 231 | 3.280E4 | 1.910E-1 |
| U - 233 | 1.580E5 | 3.670E-2 |
| U - 234 | 2.445E5 | 1.200E4 |
| U - 235 | 7.040E8 | 3.496E2 |
| Np - 237 | 2.140E6 | 4.329E2 |
| U - 238 | 4.468E9 | 1.147E4 |
| Pu - 239 | 2.410E4 | 5.334E6 |
| Am - 241 | 4.320E2 | 1.205E7 |

Little reliable information seemed to exist for sorption coefficients, especially in clays, and so subjective estimates were used based upon data for soils. No information appeared to be available concerning the detailed stable element chemistry of the wastes themselves.

### 3.5 Choosing the release scenario

After resaturation of the repository and corrosive failure of the waste containers, it was assumed that the radionuclides dissolved into flowing groundwater and were transported downwards through the clay and Lower Greensand into the Corallian aquifer for discharge into the Thames flood plain some 10 km downstream.

It was assumed that no changes occurred in the surface environment, climate or hydrogeological regime over the time period of the analysis. In a full assessment such future changes would be considered including the effect of fault zones or human intrusion (pumped wells or deep drilling) which might considerably shorten the flow path to the surface.

## 4. THE CALCULATION OF GROUNDWATER FLOW

### 4.1 A conceptual model

As indicated in Table 1, the stratigraphy was simplified to three layers: a surface layer of chalk and the upper Greensand aquifer combined, the Gault and Kimmeridge Clays combined and the underlying Corallian (limestone) aquifer. The Lower Greensand was neglected as being of little hydrogeological importance.

Study of the measured groundwater head contours in the aquifers showed that a two-dimensional vertical section flow model would be adequate (e.g. see Figure 3 for head contours in chalk), possibly overestimating groundwater velocities by neglecting any divergence of flow in plan view.

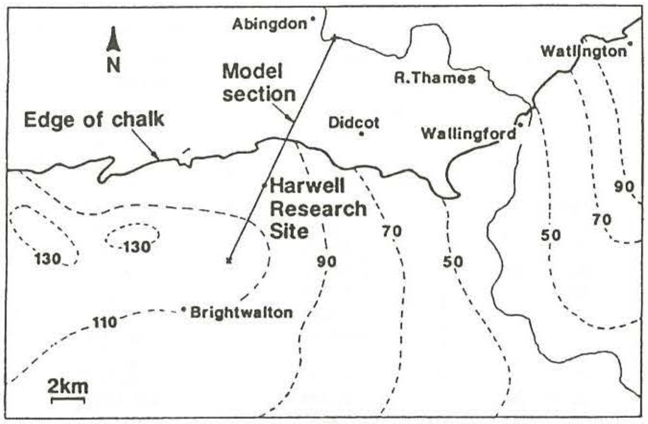


FIGURE 3 : GROUNDWATER CONTOURS IN CHALK

Figure 4 shows the section used for calculations with zero flow assumed along both vertical edges, corresponding to flow divides, and along the bottom boundary with the low permeability Oxford Clay. The upper boundary was chosen to coincide with the mean annual water table, and thus an atmospheric pressure condition could be used.

### 4.2 Calibration of the detailed model

Porous (Darcy) flow was assumed with hydraulic conductivities adjusted (see Table 1) to calibrate the model to match observed heads across the clay layer at Harwell and in the Artesian region near the Thames. The GEOHYDROFLOW finite element code was used.

### 4.3 Results of calculations

The predicted streamlines are shown in Figure 4 with very slow ($10^{-4}$ m/year) vertical flow downwards in the clay and bulk velocities in the Corallian increasing from about $5 \times 10^{-3}$ m/year under Harwell to about $8 \times 10^{-2}$ m/year under the discharge area.

### 4.4 Sensitivity to assumed boundary conditions

Calculations were repeated with small flows into the Corallian at the left hand (SSW) boundary and, as Figure 5 shows, the head difference across the clay falls markedly below the observed value, showing the zero flow assumption was justified unless the hydraulic conductivities used were seriously in error. The vertical velocity in the clay is substantially unaffected, however, and the overall water travel time from the repository to the surface remains at approximately $5 \times 10^5$ years.

### 4.5 Discussion

The calibration and sensitivity analysis activities shown in Figure 1 have been demonstrated, but in a real assessment, detailed calculations of flow and of radionuclide transport in the presence of the repository itself would of course be required. Calibration of the transport model could in theory be done by adjusting 'dispersion' to match tracer tests, for example.

## 5. THE BIOSPHERE

A 3 $km^2$ area of potentially contaminated land in the Thames floodplain was assumed from consideration of dispersion within the Corallian aquifer. Seventy-five percent of this area was allocated to permanent pasture supporting mainly cattle and sheep, 20% to cereal cultivation and 5% to general horticulture with aspersion using water from the River Thames. Run off from contaminated land was used as drinking water for man and his domestic animals. Calculations using the equilibrium biosphere code ECOS (12) indicated the majority of dose was due to ingestion of plants contaminated by root uptake and that even for poorly sorbed nuclides such as $I^{129}$, $Np^{237}$ and $Tc^{99}$ drinking water contributed only slightly to dose.

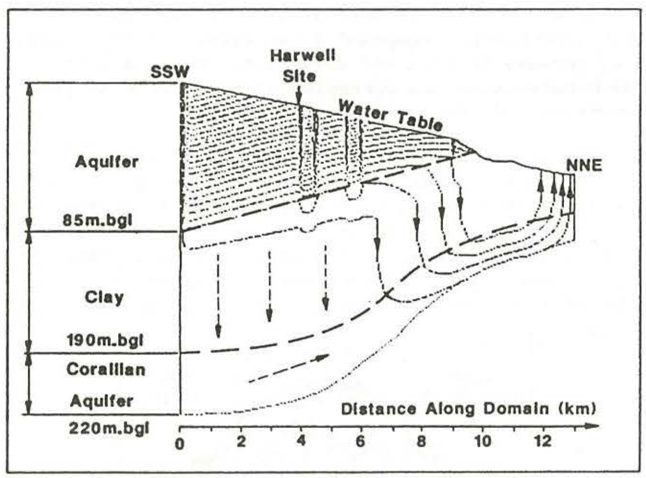


FIGURE 4: GROUNDWATER STREAMLINES USING GEOHYDROFLOW

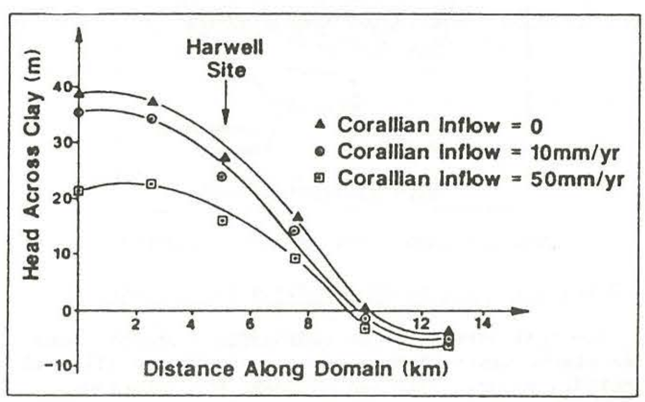


FIGURE 5 : HEAD DIFFERENCES ACROSS CLAY LAYERS

## 6. RISK ANALYSIS

### 6.1 'Worst risk' estimate

Extreme values of parameters such as hydraulic gradient, permeability and sorption in the geosphere were taken from the ranges indicated by the detailed sensitivity analyses and from the subjective p.d.f.'s used in the p.r.a. calculations (see below). SYVAC 'A/C' was then run to predict the variation of risk conditional upon these extremes actually occurring. The result is shown in Figure 6 where a maximum annual risk of about $1.3 \times 10^{-3}$ is found at 54,000 years, due to $Np^{237}$.

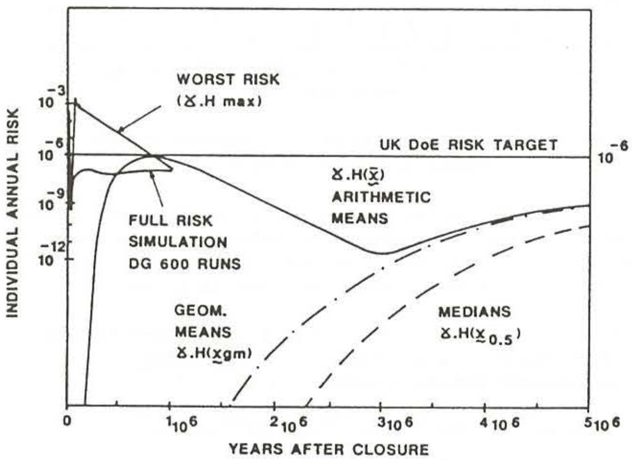


FIGURE 6 : COMPARISON OF RISK ESTIMATORS

In accordance with the approach set out earlier (Figure 1), as the worst risk exceeds the $10^{-6}$ p.a. target, it is necessary to undertake a full p.r.a. (Figure 8 summarises the activities involved.)

### 6.2 Simplified models for p.r.a. calculations

For the agreed scenario, the groundwater flow and transport model assumed is sketched in Figure 7 as a two-dimensional vertical section. The flow path from the repository has three segments; in SYVAC, a single clay layer was used of length ($L_1 + L_3$) to ensure the bulk properties for both clay segments are correlated. A steamtube bounded by the repository would contaminate only a small area of soil as Figure 6 shows. However, dispersion vertically through the aquifers is assumed, leading to contamination of the entire discharge region. The one-dimensional nuclide transport equation is solved analytically using the GEO1 code (13) in SYVAC.

The repository is assumed resaturated and all containers failed by 20 years after closure at which time leaching over the entire waste begins. In SYVAC, the numerical flow and transport code VERMIN (14) is used to model these processes, together with advection and two-dimensional dispersion through the buffer and tunnel liner. Multi-member chain decay, linear equilibrium sorption, saturation limits are also represented as is also the effect of increased permeability in the nearfield clay, damaged by the tunnelling process.

The biosphere model ECOS is used in SYVAC in the form described above in Section 5.

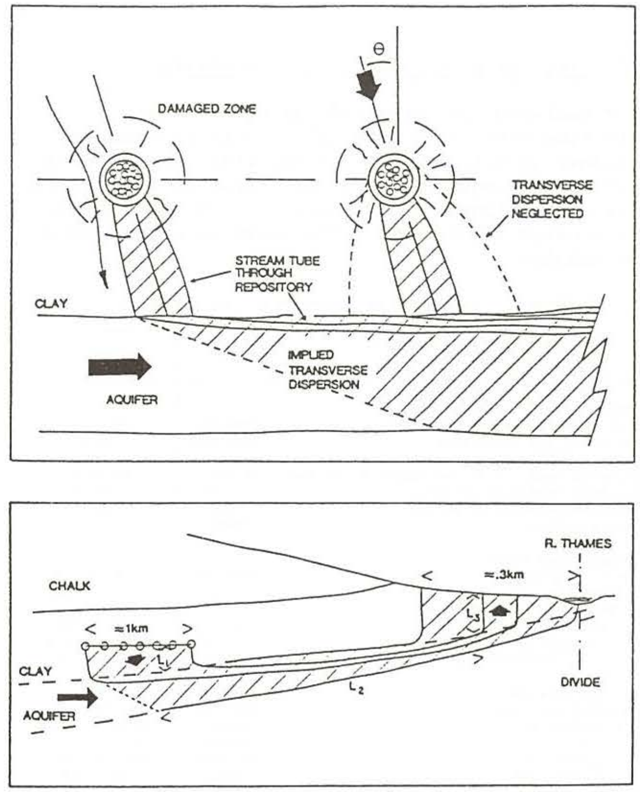


FIGURE 7 : GROUNDWATER FLOW AND NUCLIDE TRANSPORT MODEL USED IN SYVAC 'A/C' FOR HARWELL SITE

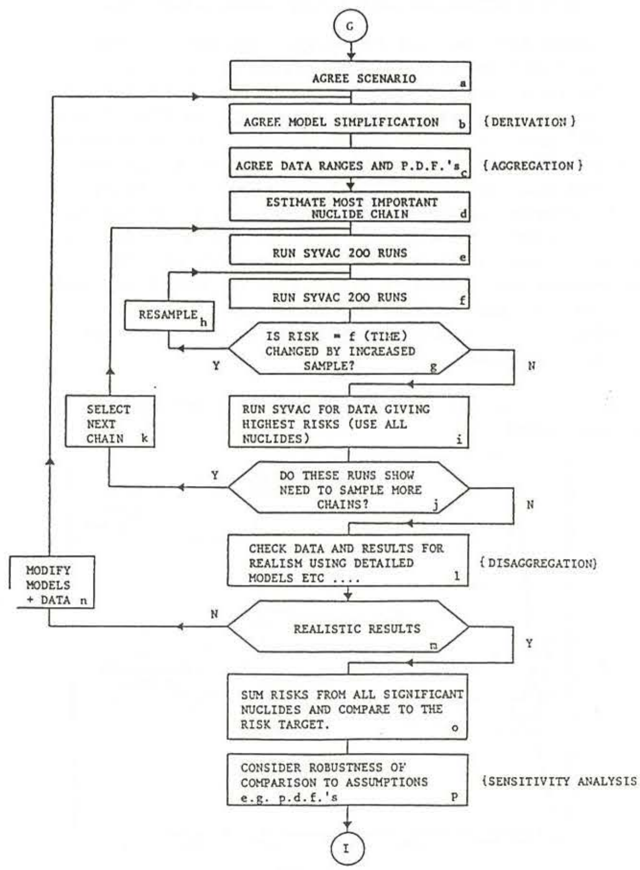


FIGURE 8 : THE CURRENT RISK CALCULATION PROCEDURE

### 6.3 Data for probabilistic risk analysis

Formalised and repeatable procedures for eliciting subjective probability distributions are examined in Reference 15 but for this initial study team discussions and individual views were used in a more conventional, intuitive manner. Table 3 shows the sampled inputs used. Biosphere parameters were not sampled.

TABLE 3 : SAMPLED DATA FOR SYVAC 'A/C'

| PARAMETERS | PDF: | RANGE: |
|---|---|---|
| VAULT: | | |
| Multiplier for buffer sorption | UNIFORM | 0.2 to 5.0 |
| Multiplier for leach rate | CONSTANT | 1.0 |
| Hydraulic conductivity of buffer (m/s) | LOGUNF | -12 to 8 |
| Hydraulic conductivity of tunnel liner (m/s) | LOGUNF | -12 to 8 |
| Nearfield diffusion ($m^2$/year) | CONSTANT | 0.02 |
| Effective porosity of buffer | CONSTANT | 0.1 |
| Liner porosity | CONSTANT | 0.1 |
| GEOSPHERE CLAY LAYER: | | |
| Hydraulic gradient | UNIFORM | 0.1 to 1.0 |
| Permeability ($m^2$) | LOGUNF | -20 to -16 |
| Porosity | UNIFORM | 0.01 to 0.3 |
| Path length (m) | UNIFORM | 50 to 170 |
| Dispersivity (m) | UNIFORM | 0.1 to 10.0 |
| Multiplier for sorption coefficient | UNIFORM | 0.2 to 5.0 |
| CORALLIAN AQUIFER: | | |
| Hydraulic gradient | UNIFORM | 0.0001 to 0.1 |
| Permeability ($m^2$) | LOGUNF | -15.0 to -11.0 |
| Porosity | UNIFORM | 0.1 to 0.3 |
| Path length (m) | UNIFORM | 9600 to 19200 |
| Dispersivity (m) | UNIFORM | 1.0 to 100 |
| Multiplier for sorption coefficient | UNIFORM | 0.1 to 1.0 |

### 6.4 The results of the SYVAC 'A/C' calculations

Iodine 129 was run first as experience suggested it would dominate risk. The variation of risk with sample size up to 1200 runs using the Deterministic Generator (DG) method (16), is shown in Figure 9 for $2 \times 10^5$ and $10^6$ years after closure. It is seen that the results for more than about 600 runs lie within the 95% confidence interval of a 8000 run study using simple random sampling (MC). The contribution of different subranges of dose to overall risk is shown in Fig 10. Sampling further into the tails of the parameter distributions gives higher doses but does not significantly increase risk thereby providing confidence that the sampling is adequate for this nuclide.

Activities (d) to (g) of Figure 8 were thus demonstrated.

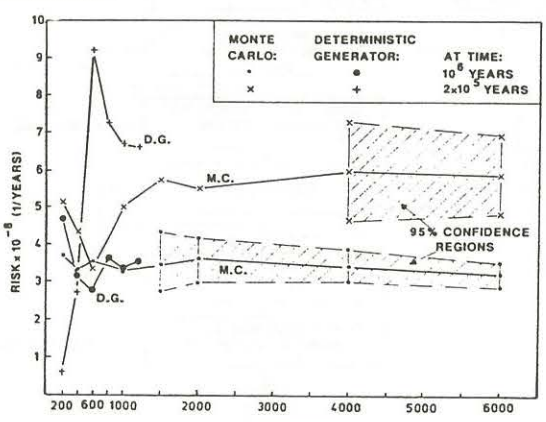


FIGURE 9 : EFFECT OF SAMPLE SIZE ON PREDICTED INDIVIDUAL RISKS FROM IODINE 129

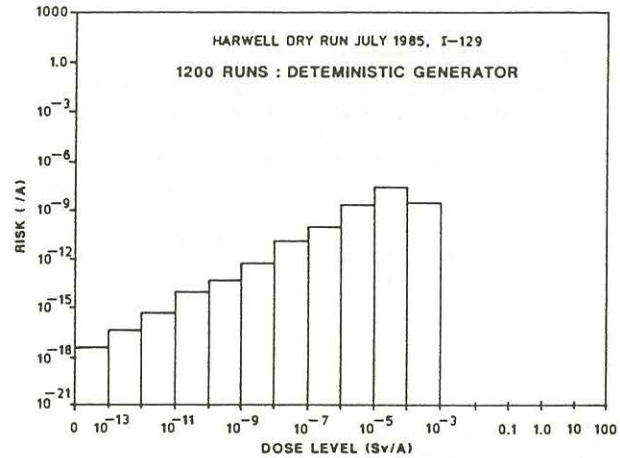


FIGURE 10 : CONTRIBUTIONS OF DOSE RANGES TO THE OVERALL ANNUAL INDIVIDUAL RISK

For those conditions leading to the highest doses from $I^{129}$, the dose variations with time were then obtained, as shown for instance in Figure 11, for all nuclides in the waste inventory [Activity (i)].

Risk was then calculated for thirteen chains and, as shown in Figure 12, the only significant contributions obtained were from $Tc^{99}$, $I^{129}$,

$U^{234} \rightarrow Th^{230} = Ra^{226}$, $Pu^{239} \rightarrow U^{235} \rightarrow Pa^{231}$,

and $Am^{241} \rightarrow Np^{237} \rightarrow U^{233} \rightarrow Th^{229}$.

The overall risk rises to about $1.3 \times 10^{-7}$ p.a at 150,000 years and falls slowly to about $10^{-7}$ p.a. at one million years. No confidence interval is shown as for the limited samples used for these other nuclides a converged value of risk has not been achieved. Typical difficulties are revealed by Figure 13 for the $Pu^{239}$ chain where between 400 and 600 runs a few very high doses are sampled leading to a large jump in predicted risk at 200,000 years, although by $10^6$ years the prediction seems reasonably well established.

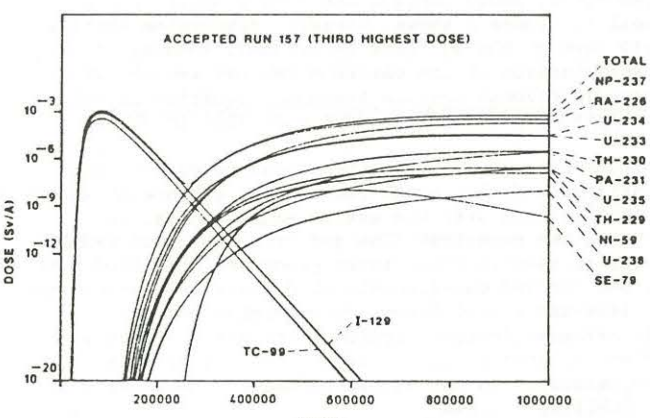


FIGURE 11 : VARIATION OF DOSE WITH TIME FOR DIFFERENT NUCLIDES

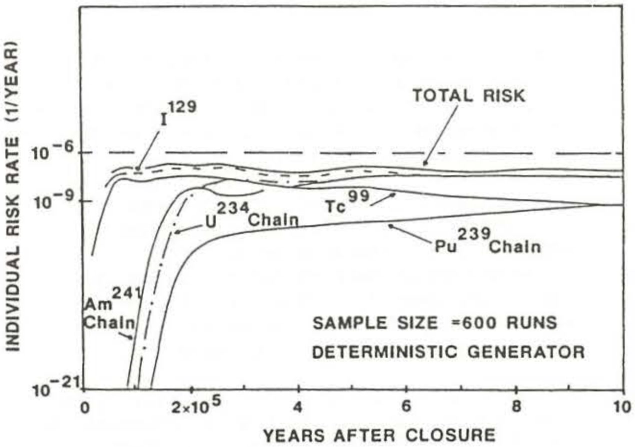


FIGURE 12: CONTRIBUTION OF INDIVIDUAL NUCLIDE CHAINS TO THE OVERALL VARIATION OF RISK WITH TIME

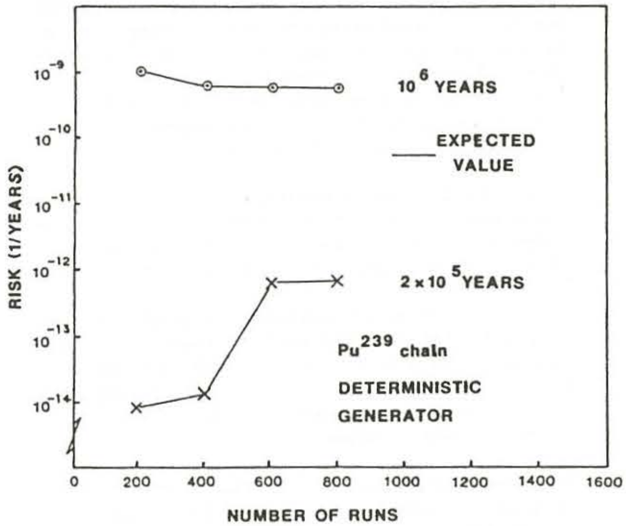


FIGURE 13: EFFECT OF SAMPLE SIZE ON PREDICTED INDIVIDUAL RISK FROM THE PLUTONIUM 239 CHAIN

## 6.5 Comparison with alternative risk estimators

In Figure 6, the result of the p.r.a. simulation is compared with the 'worst risk' from Section 6.1 and with a number of 'best estimates'. The latter appear to underestimate the true risk and could lead to complacency in assessing a disposal facility. The 'worst risk' in contrast exceeds the maximum risk from p.r.a. by about 10,000 times, because the very low probability of such pessimistic conditions has not been accounted for.

## 6.6 Reanalysis of high risk conditions

Experience shows (see for instance Fig 10) that the SYVAC runs giving the higher doses contribute most to risk even though their probabilities are small. Consequently, the conditions sampled may be very different from those assumed earlier in the detailed modelling. It is necessary, therefore, to check that the simple p.r.a. models are satisfactory by comparison with more precise detailed calculations under these extreme circumstances as indicated in activities (1,m) of the flow chart of Figure 8.

6.6.1 Derivation of detailed data sets. (DISAGGREGATION) The four runs having the highest doses were examined in detail and appear plausible in terms of the behaviour of different nuclides and in the relative importance of different hydrogeological parameters. Values of permeability, porosity, and dispersivity were obtained from these SYVAC runs and assumed uniform across the corresponding spatial domains in detailed two-dimensional flow and transport calculations.

6.6.2 Comparison with two-dimensional calculations. (VERIFICATION) The flux of radionuclides predicted by the SYVAC geosphere transport model is found to agree well with the results of the two-dimensional calculations (see Figure 14) and the assumption of rapid vertical dispersion across the depth of the aquifer was also confirmed (1).

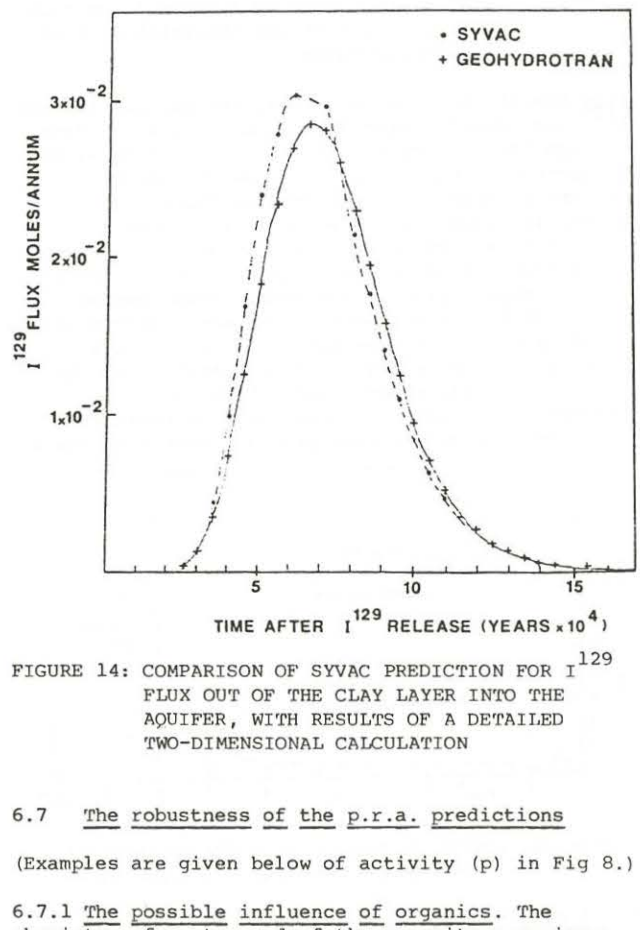


FIGURE 14: COMPARISON OF SYVAC PREDICTION FOR $I^{129}$ FLUX OUT OF THE CLAY LAYER INTO THE AQUIFER, WITH RESULTS OF A DETAILED TWO-DIMENSIONAL CALCULATION

## 6.7 The robustness of the p.r.a. predictions

(Examples are given below of activity (p) in Fig 8.)

6.7.1 The possible influence of organics. The chemistry of wastes and of the repository environment is at present poorly understood. Even at high pH values the presence of organic complexing agents, both natural (ie, humic and fulvic acids) and/or those introduced artificially (eg, EDTA), may raise saturation concentrations especially of plutonium. The present SYVAC calculation was therefore repeated with the saturation limit increased by 100 times and, as shown in Figure 15 (for a small statistical sample) this increased risk by about 30 times.

6.7.2 The choice of probability distributions. As there is a considerable influence of subjective judgement in the p.d.f's used for p.r.a., it is important to discover how the risk estimate depends

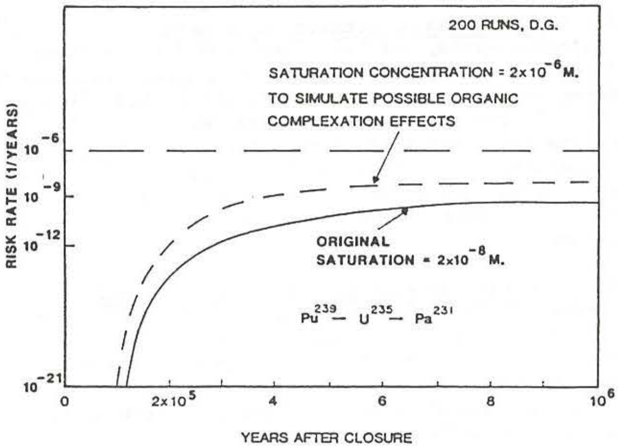


FIGURE 15: EFFECT OF CHANGES IN ASSUMED SATURATION CONCENTRATION IN THE NEARFIELD ON RISKS DUE TO PLUTONIUM

upon the shapes assumed. Hence, the calculation for $I^{129}$ was repeated with distributions changed from Uniform/log Uniform to Normal/log Normal forms but retaining the same upper and lower bounds and arithmetic mean values. Figure 16 shows the generally lower risks predicted from Gaussian distributions due to the lower probabilities in the tails that are associated with the shorter travel times. Longer travel times result from sampled values of parameters near to the modal values of the p.d.f.'s and hence give larger risks from the Gaussian family as Figure 16 indicates. The data aquisition study of Reference 15 tentatively suggests that experts prefer p.d.f.'s intermediate in shape between the alternatives considered above.

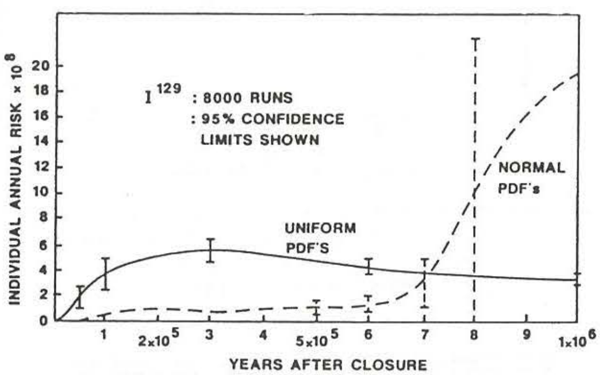


FIGURE 16: EFFECT OF CHOICE OF INPUT PROBABILITY DISTRIBUTIONS ON THE PREDICTED RISK

6.7.3 Sensitivity analysis. Other 'ad hoc' engineering comparisons can be made and in particular the effect of uncertainty in the waste inventory urgently needs investigating especially for low-level wastes. More systematic methods of sensitivity analysis have been explored (17) in connection with the Importance Sampling scheme for use in SYVAC 'A/C'.

## 7. DISCUSSION AND CONCLUSIONS.

The majority of the stages in the proposed risk assessment procedure (Figures 1 and 8) have been rehearsed for a single scenario with parameter values and uncertainties which do not change with time after closure. However, several issues need further consideration:

(a) The risk estimates require further sampling before they can be justified statistically. During 1986, Importance Sampling has been used and converged risk estimates have been obtained. This work will be published later;

(b) The influence of correlations between parameters is not fully understood and physically unreasonable combinations of (independently) sampled parametric values can occur. The second Dry Run currently being undertaken appears to have overcome this problem and will be reported soon;

(c) In order to withstand public scrutiny, good practice is essential in the verification and validation of models, in the management of case study calculations and in the aquisition and manipulation of data. Probability distributions may be elicited by formal, explicit methods (15) and Bayesian techniques are being investigated to see if they can be used in practice to improve initial estimates in the light of subsequent (eg, site) information to give data sets for p.r.a. during the later stages of assessment (Section 2.1, (c) - (f) above);

(d) Formal methods of deriving simple models and their associated data from more detailed considerations may be advantageous, and any additional systematic bias must be accounted for;

(e) Improved methods of sensitivity analysis would be beneficial if they lead to fewer sampled variables in p.r.a. although the 'adjoint' method and 'response surface methodology' have not yet been demonstrated for realistic disposal case studies;

(f) Improved modelling of significant chemical processes needs to be rehearsed but further systematic research (see Reference 18) is necessary before the effect of colloids and of organic complexing agents can be represented satisfactorily;

(g) The entire procedure needs demonstration for human intrusion scenarios and the evolution of the disposal environment with time needs to be accounted for.

## 8. THE FUTURE

The next stage (DRY RUN 2) will demonstrate the way in which different scenarios can be defined logically, selected, analysed and combined to give an overall risk prediction. Limited treatment of time-dependent parameter values will be introduced but a comprehensive approach will not be possible until DRY RUN 3 is carried out using the TIME2 scenario generator (19) showing how the repository site evolves under the influence of:

(a) a climate-driven sequence of natural proceses, both gradual and disruptive;

(b) human-induced effects; and

(c) repository-induced effects.

In addition, to ensure confidence in the various aspects of the p.r.a. procedure outlined, the UK Department of the Environment will continue to participate in international verification and validation exercises [eg, PACOMA, HYDROCOIN and the NEA PSACOIN project which includes comparison of p.r.a. codes with recent exact solutions of the transport equations (20)].

9. ACKNOWLEDGEMENTS

This work would not have been possible without the enthusiasm and cooperation of the entire contractor team. The principal contributors who should be regarded as co-authors were: Dr TW Broyd (Atkins Research and Development), Dr ZA Gralewski (Electrowatt Engineering Services Ltd.), Dr M Thorne (formerly of Associated Nuclear Services Ltd.), Dr D Read (University of Wales Institute of Science and Technology), Dr G Dalrymple (CAP Scientific) and Dr I Rae (SCICON Ltd.).

We thank the Department of the Environment for permission to publish this paper and thank also our colleagues in the Department for their advice and support. It is noted that the work described in this paper may be used in the formulation of UK Government policy but does not at this stage necessarily represent Government policy.

REFERENCES

(1) THOMPSON, B.G.J. and BROYD, T.W., "DRY RUN 1: An initial examination of a procedure for post-closure radiological risk assessment of an underground disposal facility for radioactive waste", DoE Disposal Assessments Report TR-DoE-4, Edition 1, 1986, February.

(2) Department of the Environment, "Disposal Facilities on Land for Low and Intermediate - Level Radioactive Wastes: Principles for the Protection of the Human Environment", 1984.

(3) THOMPSON, B.G.J., DUNCAN, A.G.D. and HALL, P.A., "The development of post-closure radiological assessment methods by the Department of the Environment", BNES conference on Radioactive Waste Management, London, 1984, November.

(4) LANE, G. D., HALL, P.A., "SYVAC 'A/C' User Guide, DoE SYVAC User Document UD-SC1-2, Edition 1, 1985, February.

(5) LIEBERMAN, J.A. et al, "Report of the U.S. Performance Assessment National Review Group", Prepared by Weston Inc. for U.S. DoE under contract DE-AC01-83-NE44301, 1985, February.

(6) NEA/OECD, "Long-term radiation protection objectives for radioactive waste disposal". Expert Group of NEA Committee on Radiation Protection and Public Health, and the Radioactive Waste Management Committee, 1984.

(7) RWP Division, Department of the Environment, "Assessment of Best Practicable Environmental Options (BPEOs) for Management of Low and Intermediate Level Solid Radioactive Wastes", Draft Final Report, 1985, November.

(8) THOMPSON, B.G.J., JOHNSTON, P.D., "Information required by the Department of the Environment for its post-closure radiological risk assessment of underground disposal : Preliminary advice to the Developer", DoE Disposal Assessments Technical Document TR-DOE-6, Edition 1, 1986, March.

(9) WILMOT, R.D., MORGAN, D.J., "Mineralogy and Lithochemical Studies of Strata Beneath the Harwell Research site", BGS ENPU 82-13, 1983.

(10) GRIFFIN, J.R. et al, "An Engineering design study for storage and disposal of intermediate level waste", ND/R/777(R) Issued by Risley Nuclear Power Development Establishment, Risley, Warrington, 1982, November.

(11) THORNE, M.C., "A preliminary evaluation of radionuclide inventories relevant to the disposal of low and intermediate level radioactive wastes in the UK", DoE SYVAC Technical Report TR-ANS-14, Edition 1, 1984, November.

(12) KANE, P. and THORNE, M.C., "User's Guide to the biosphere code ECOS", DoE SYVAC Report TR-ANS-9, Edition A, 1984, April.

(13) OLDFIELD, S.G. and BROYD. T.W., "User Guide for the SYVAC Geosphere Program GE01", DoE SYVAC Report TN-WSA-1, Edition 1, 1984, March.

(14) GRALEWSKI, Z.A., "Development of vault model 'VERMIN' for post closure behaviour of repositories for non-heat generating radioactive waste", DoE SYVAC Report TN-EWE-2, Edition 1, 1983, October.

(15) DALRYMPLE, G.J., JOHNSON, K.B. and PHILLIPS, L.D., "Final Report on the Acquisition of Data for use in the probabilistic risk assessment of underground disposal of radioactive wastes", DoE SYVAC Report TR-STH-19, Edition 2, 1986, January.

(16) MIDDLETON, R., "Statistical Techniques for the Development and Application of SYVAC", DoE SYVAC Report TR-STH-1, Edition 1, 1983, August.

(17) PRUST, J.O. and DALRYMPLE, G.J., "Development of SYVAC sampling techniques", DoE SYVAC Report TR-STH-14, Edition 1, 1985, April.

(18) READ, D. KROL, A.A. and THOMPSON,B.G.J., "A structured programme of research into the chemical aspects of radioactive waste disposal", Paper accepted for MRS Conference, Boston, 1-4 December, 1986.

(19) FRIZELLE, C.J.G, MUTTON, J.E., UPTON, P.S. and THOMPSON, B.G.J., "Modelling the long-term evolution of radioactive waste disposal environments for use in post-closure probabilistic risk assessment", Proc. Second International Conference on Radioactive Waste Management, CNS, 7-11 September, 1986.

(20) ROBINSON, P.C. and HODGKINSON, D.P., "Exact solutions for radionuclide transport in the presence of parameter uncertainty", AERE-R-12125, 1986, January.