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Probabilistic safety assessment model for near surface radioactive waste disposal facilities

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Abstract

A probabilistic safety assessment model has been developed for assessing the performance of near surface disposal facilities for low-level radioactive waste. Two modes of disposal such as single dump and multiple dump are considered in the model. The model is composed of four components: source term, repository failure, geosphere transport and radiological assessment. The source term contains low-level radioactive waste equivalent to 50 GWe.y energy production (1 GWe.y = 3.2×10^{16} J of electrical energy production) either disposed at an instant or distributed over 50 years. The endpoints of assessment are expressed as radioactivity release rate, radionuclide concentration in ground water, radiation dose to a member of the critical group through drinking water pathway and total risk to critical group due to disposal practice. Sensitivity analysis is carried out to identify the critical parameters, which have maximum effect on the assessment endpoints. Uncertainty analysis, based on random selection of all main parameters, is also carried out to identify the effect of overall variation of parameters used in the model in relation to a reference level. The reference level delivers a maximum annual effective dose of 1.2×10^{-2} mSv to a member of the critical group after about 1.25×10^{2} years of disposal mostly from ^{129}I, the long-lived and less sorbing radionuclide. The corresponding risk to the public due to the disposal practice (9.0×10^{-7} y^{-1}) is found to be lower than that due to natural background radiation. The most critical parameters as indicated by the sensitivity analysis are the distribution coefficient of radionuclides, seepage velocity in the unsaturated zone between the facility and the water table, dispersivity in ground water and thickness of the unsaturated zone. The uncertainty analysis shows that ^{129}I is the critical radionuclide delivering maximum dose in most cases though it constitutes a low percentage in the low-level radioactive waste inventory. The annual effective doses derived from the uncertainty analysis fall in a log normal distribution with a geometric mean of $3.6 \times 10^{-2} \pm 3.9$ mSv and the most probable annual effective dose to a member of the critical group works out to be 2.8×10^{-4} mSv. © 1999 Elsevier Science Ltd. All rights reserved.

Keywords: Radioactive waste; Near surface disposal; Safety assessment; Ground water modelling

Software availability

Name of software	REMS (Risk Evaluation Model for Safety assessment)
Developers	R.N. Nair and T.M. Krishnamoorthy, Environmental Assessment Division, Bhabha Atomic Research Centre, Mumbai-400 085, India
Email	emas@magnum.barc.ernet.in
Minimum hardware required	Pentium PC or equivalent
Hardware used	DEC-ALPHA Work Station
Program language	FORTRAN-90
Program size	365 kB
Availability and cost	Undetermined at this time

1. Introduction

Near surface disposal facilities for low-level radioactive wastes are designed to provide long term isolation of the wastes from the human environment by means of a system of barriers both natural and man-made. The basic principle of near surface disposal is to keep the radiation dose and risk from the disposal practice to lev-

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els as low as reasonably achievable. In any optimization procedure the individual doses as well as the corresponding risks must be evaluated where potential exposures are concerned. Hence it is essential to assess quantitatively the performance of the waste disposal system through safety assessment models. Probabilistic safety analysis (risk analysis) is one of the methods for safety assessment of a system. This is based on the derivation of failure rates for each component of the system. Several studies have been reported (Pritzker and Gassman, 1980; Chang and Cho, 1984; Cohen, 1984; Kim et al., 1988; Han et al., 1991; Kim et al., 1993) on the safety assessment for the radioactive waste disposal system using risk analysis. While the earlier studies on risk analysis are concerned with the radioactivity releases into the environment, later studies describe the aspects of radionuclide transport between man and the environment. Alternately, safety of radioactive waste disposal facilities can be assessed by conventional transport and dispersion models (Sullivan, 1993; Krishnamoorthy and Nair, 1994; IAEA, 1995; Nair and Krishnamoorthy, 1996, 1997) in which source term is computed based on diffusion controlled release and dissolution controlled release. The risk analysis models are useful when the release mechanism from the radioactive waste form is not clear, followed with paucity of geohydrological and geochemical data of the disposal facility. Another advantage of risk analysis models is that they can be used even for high level radioactive disposal, once the identity of the multi-barrier system is established.

A complete safety assessment of a radioactive disposal facility must include all possible scenarios of release and exposure including human intrusion. In terms of severity of risk, the human intrusion scenario may become important (Krishnamoorthy et al., 1997) in the safety analysis of radioactive waste facilities. However, in the present study emphasis is placed on the migration of radionuclides from the disposal facility through the geosphere and biosphere and the resultant radiation dose and risk to a member of the critical group based on failure rates of barriers. This approach is reasonable because under the uncertainty analysis the worst failure scenario may lead to a risk almost equivalent to that of human intrusion. Hence, a risk assessment model for near surface radioacitve waste disposal facilities has been developed based on the failure rates of multi-barriers. The barriers are assumed to be in redundancy (stand-by) and their failures are initiated by water infiltration into the facility through precipitation. The barrier failure rates have been used to evaluate the source term (radioactivity release to ground water) and the fate of released radionuclides in ground water is dealt with two-dimensional transport and dispersion model. Thus, the model is composed of a source term model, a repository failure model, a geosphere transport model and a radiological model. The endpoints of assessment are expressed as radioactivity release rate, radionuclide concentration in ground water, radiation dose to a member of the critical group through drinking water pathway and corresponding risk for each radionuclide as well as the total of each of these quantities for all the radionuclides. The critical parameters, which have maximum effect on the assessment endpoints, are identified through sensitivity analysis. Uncertainty analysis is also carried out to identify the effect of overall variation of parameters used in the model in relation to a reference level.

Two disposal modes such as single dump and multiple dump modes are considered in the study. The disposal facilities in many countries like El Cabril in Spain (Perez and Lopez, 1997) and Centre de l'Aube in France (Roch, 1997) are centralized near surface disposal facilities wherein the operation (dumping) will be over within 10 to 20 years. Such a disposal mode is referred as single dump mode. The single dump mode can represent an unplanned disposal scenario also. In countries like India, where the disposal facilities are close to nuclear power plants, the disposal operation of low-level radioactive waste starts with the commissioning of nuclear power plants and continues for a long period (> 50 to 70 years) until the permanent shutdown of the plants. Such a disposal mode is referred as multiple dump mode. The multiple dump mode requires evaluation of radionuclide source term during the dumping period and after termination of disposal (post dumping period). These two modes of disposal are used to address all possible disposal operation modes existing in different countries and also to represent unplanned disposal of radioactive waste in emergencies.

2. Multi-barrier system

The disposal site considered in this study is a typical engineered near surface disposal facility for low-level radioactive waste. Radioactive waste treated, solidified with cement and packaged in metal drums is buried in the facility. The multi-barrier system includes the engineered barriers such as top cover, waste container, waste form, backfill material, bottom cover and the natural barrier in sequence. The facility, the top cover and the bottom cover are constructed with re-inforced cement. The backfill material may include soil mixed with clay. The natural barrier is the undisturbed geological formation between the facility and the biosphere. The long-term isolation of radioactivity from the biosphere is the objective of the multi-barrier approach.

3. Barrier failure scenarios

The disposal facility is assumed to be situated in a humid region. The amount of radioactivity released to

the geosphere is calculated on the basis of sequential failure of each barrier due to water infiltration through the top cover. The failure of the top cover (Barrier a) begins with water infiltration due to rainfall. This would result in contact of the waste container (Barrier b) with water leading to corrosion of the mild steel container. As the corrosion proceeds, water will interact with the solidified waste (Barrier c) leading to leaching of radionuclides from the waste form. The leached radioactivity will begin to migrate through the backfill (Barrier d) after its saturation with ground water and after the failure of the bottom cover (Barrier e) reaches the geosphere (Barrier f). The geosphere barrier (Cho et al., 1992) is the unsaturated zone between the facility and the water table. The transit time of radionuclides in this zone is considered (Kim et al., 1993) as its mean time to failure (MTTF). After the sequential failure of all the barriers, the radionuclides migrate to ground water which is considered as the biosphere. The failure scenario for each barrier is summarized in Table 1 along with assumed mean time-to-failure.

4. Source term model

Two types of disposal modes: single dump and multiple dump of radioactivity are considered in this study. The single dump mode represents either a centralized disposal facility where the disposal practice will be completed within a short period or an unplanned disposal scenario under emergencies. The multiple dump mode allows annual disposal rates into the facility which are located within the exclusion zone of nuclear power plants and the disposal practice starts with the operation of nuclear power plants and ends with their permanent shutdown. The single dump mode considers radioactive waste inventory equivalent to 50 GWe.y and the multiple dump mode uses an inventory equivalent to 1 GWe.y for 50 years. The low-level radioactive waste inventory expected to generate due to the operation of 1 GWe.y is summarized in Table 2.

The source term, S_s (Bq), corresponding to single dump mode is given by:

$$S_s(t) = M \exp(-\lambda_p t) \quad (1)$$

where M is the inventory (Bq) of a radionuclide in the low-level radioactive waste equivalent to 50 GWe.y energy production, λ_p is its radioactive decay constant (y^{-1}) and t is the time elapsed after disposal. The subscript "s" in Eq. (1) is used to represent the single dump mode.

For multiple dump mode, source terms are required during both dumping period and after termination of disposal operation (post dumping period). The source term, S_d (Bq), during dumping period is given by:

$$S_d(t_d) = [Q/\lambda_p][1 - \exp(-\lambda_p t_d)] \quad \text{for } 0 < t_d \leq T \quad (2)$$

where t_d is the dumping period (y), T is the total disposal operation period (50 years) for the facility and Q is the annual disposal rate ($Bq.y^{-1}$) of a radionuclide. This means that at the end of the disposal period, T, low-level radioactive waste corresponding to 50 GWe.y has been dumped into the facility. During the period 0 to T years, the dumping operation is continuous and after the period, T, the dumping operation is terminated. The source term after the termination of dumping operation is represented by the residual amount of radioactivity present in the facility at time, T, after decay. Hence, the source term, S_p (Bq), during post dumping period is given by:

$$S_p(t) = S_d(T)\exp(-\lambda_p t) \quad (3)$$

where t is the time elapsed after termination of dumping operation whose origin is at the end of the disposal operation period, T. The subscripts "d" and "p" in Eqs. (2) and (3) are used to indicate the source terms for multiple

Table 1
Barriers and failure scenarios used in the model

No	Barrier	Failure scenario	Range of MTTF (y)	Reference value (y)
a	Top cover	Degradation of top cover due to water infiltration from rainfall.	10–50	25
b	Waste container	Corrosion of mild steel due to water infiltration.	5–25	12.5
c	Waste form	Degradation of cemented waste form due to ageing mechanisms including chemical attack, physical attack and leaching due to infiltrating water.	30–4000	300
d	Backfill	Infiltration of surface water or ground water due to upheaval of water table.	12–55	30
e	Bottom cover	Infiltration of surface or ground water due to upheaval of water table.	6–26	15
f	Near-field geosphere	Mobility of radionuclides through infiltrating water based on retardation factor (R_d) and travel time ($T_r = z/U_z$) where z is the thickness and U_z is the infiltrating water velocity.	R_dT_r	R_dT_r

Table 2
Radionuclide dependent parameters used in the model

Nuclide	Half-life (y)	Radioactive waste inventory (Bq/GWe.y)	Range of K_d values ($ml.g^{-1}$)	Reference K_d value ($ml.g^{-1}$)	Ingestion dose coefficient (Sv/Bq)
^{3}H	1.23×10^{1}	7.40×10^{10}	0	0	1.80×10^{-11}
^{14}C	5.73×10^{3}	4.81×10^{12}	10–50	20	6.20×10^{-12}
^{54}Mn	0.89	1.78×10^{13}	100–5000	100	7.10×10^{-10}
^{55}Fe	3.00	1.73×10^{13}	50–1000	100	3.30×10^{-10}
^{59}Ni	7.50×10^{4}	6.29×10^{11}	50–1000	100	6.30×10^{-11}
^{60}Co	5.20	5.55×10^{12}	100–5000	1000	3.40×10^{-9}
^{63}Ni	1.00×10^{2}	2.52×10^{12}	50–1000	100	1.50×10^{-10}
^{90}Sr	2.85×10^{1}	2.64×10^{13}	10–200	20	2.80×10^{-8}
^{99}Tc	2.12×10^{5}	5.55×10^{8}	1–100	10	6.40×10^{-10}
^{129}I	1.70×10^{7}	1.11×10^{8}	0.5–1	1	1.10×10^{-7}
^{137}Cs	3.02×10^{1}	5.46×10^{13}	100–1000	200	1.30×10^{-8}
^{144}Ce	0.77	1.36×10^{13}	500–5000	1000	5.20×10^{-9}
^{237}Np	2.14×10^{6}	5.18×10^{5}	100–7000	700	1.10×10^{-7}
^{238}Pu	8.96×10^{1}	1.48×10^{10}	1000–10000	2000	2.50×10^{-7}
^{239}Pu	2.44×10^{4}	1.59×10^{10}	1000–10000	2000	2.50×10^{-7}
^{241}Pu	1.32×10^{1}	3.70×10^{11}	1000–10000	2000	4.70×10^{-9}
^{241}Am	4.32×10^{2}	3.70×10^{9}	100–5000	300	2.00×10^{-7}

K_d = Distribution coefficient of radionuclides for clay.

dump mode during dumping and post dumping periods respectively.

5. Repository failure model

The repository failure distribution model is used to evaluate the radioactivity release rate into ground water through the multi-barrier system. A failure distribution is a mathematical attempt to provide information about the lifetime of barriers. The mean of failure distribution against operating time gives the mean time to failure of the barrier. If the chance failure rate is considered to be a constant, λ, the failure distribution can be represented by an exponential density distribution (Thomson, 1969; McCormick, 1981; Cho et al., 1992; Kim et al., 1993) as:

$$f(t) = \lambda \exp(-\lambda t) \tag{4}$$

where $f(t)$ is the failure distribution of a barrier, t is the operation time (y) of a barrier and λ refers to its chance (conditional) failure rate (y^{-1}) which is equal to the inverse of mean time-to-failure of the barrier when the facility operates continuously without repair.

The radioactive waste disposal facility can be considered as a system composed of independent barriers that are operated sequentially in such a manner that only one component is in operation at a time. When this component is in operation, the other components are in redundancy and it is assumed that the redundancy mode of operation never fail under normal scenarios. Thus, the latter component will be operated at the failure of the former component sequentially. The failure of a radioactive waste disposal facility means that all the unit components have failed and the radionuclides are released into the biosphere (Cho et al., 1992). Then the failure probability density of the disposal facility system can be expressed (Kapur and Lamberson, 1977; McCormick, 1981; Cho et al., 1992; Kim et al., 1993) by the following integral.

$$f_s(t) = \int_0^t \lambda_a e^{-\lambda_a t_a} \int_{t_a}^t \lambda_b e^{-\lambda_b (t_b - t_a)} \int_{t_b}^t \lambda_c e^{-\lambda_c (t_c - t_b)} \tag{5}$$

$$\int_{t_c}^t \lambda_d e^{-\lambda_d (t_d - t_c)} \int_{t_d}^t \lambda_e e^{-\lambda_e (t_e - t_d)} \lambda_f e^{-\lambda_f (t_f - t_e)} dt_e dt_d dt_c dt_b dt_a$$

where $f_s(t)$ is the exponential failure probability density of the complete barrier system (y^{-1}) and $\lambda_a \exp(-\lambda_a t_a)$ is the failure probability density of barrier "a" and so on as explained in Eq. (4). The integration in Eq. (5) has been carried out in such a manner that barrier e has failed at $t_e \leq t_f$; the barrier d at $t_d \leq t_e$; the barrier c at $t_c \leq t_d$; the barrier b at $t_b \leq t_c$; and the barrier a at $t_a \leq t_b$. The integration of Eq. (5) will lead to a solution such as:

$$f_s(t) = \left[\prod_{i=a}^{f} \lambda_i \right] \left[\sum_{i=a}^{f} \frac{e^{-\lambda_i t}}{\prod_{j \neq i} (\lambda_j - \lambda_i)} \right] \tag{6}$$

The probable release rate of a radionuclide from the waste disposal facility to ground water at time, t, can be evaluated as the product of its source term at that time and the failure probability density of the barrier system

(Cho et al., 1992; Kim et al., 1993). Thus, the release rate of a radionuclide, R_s (Bq.y^{-1}), into ground water for single dump mode is given by the equation:

$$R_s(t) = S_s(t)f_s(t) \tag{7}$$

where $S_s(t)$ is the amount of a radionuclide (Bq) in the disposal facility at time, t, which is given by Eq. (1) and $f_s(t)$ is the failure probability density (y^{-1}) of the barrier system. Similarly, the release rate of a radionuclide into ground water during dumping period for multiple dump mode, R_d (Bq.y^{-1}) is given by the equation:

$$R_d(T) = S_d(T)f_s(T) \tag{8}$$

where $S_d(T)$ is the amount of a radionuclide at time, T, in the disposal facility (Bq) which is given by Eq. (2). The release rate of a radionuclide into ground water during post dumping period, R_p (Bq.y^{-1}), is given by the equation:

$$R_p(t) = S_p(t)f_s(t + T) \tag{9}$$

where $S_p(t)$ is the amount of a radionuclide at time, t, in the disposal facility (Bq) which is given by Eq. (3).

6. Geosphere model

The geosphere model is used to translate the radioactivity release rate into radionuclide concentration in ground water. The model is based on the geochemical and hydrological data obtained at the solid waste management facility (Narayan, 1998) at Trombay in Mumbai. These data are presented in Table 3.

Table 3
Nuclide independent parameters used in the model

Parameter	Unit	Range	Reference value
Bulk density	g.cm^{-3}	–	1.7
Porosity	–	–	0.3
Thickness of unsaturated zone	cm	1.0–4.0	2.0
Seepage velocity in unsaturated zone	cm.s^{-1}	5×10^{-9}–2×10^{-7}	1.157×10^{-8}
Ground water velocity	cm.s^{-1}	1×10^{-6}–1×10^{-3}	1.157×10^{-4}
Aquifer thickness	cm	–	600.0
Aquifer lateral extent	cm	–	1700.0
Aquifer cross sectional area	cm^2	–	1.0×10^6
Dispersivity	cm	10.0–500.0	100.0
Water intake rate	L.day^{-1}	–	2.2
Location of well	km	–	1.6
Risk factor	mSv^{-1}	–	7.3×10^{-5}

The concentration of a radionuclide in ground water for single dump mode, C_{gs} (Bq.cm^{-3}) is given by the convolution integral (Nair and Krishnamoorthy, 1997):

$$C_{gs}(x,y,t) = \int_0^t S_s(t - \tau)f_s(t - \tau)C_g(x,y,\tau)\mathrm{d}\tau \tag{10}$$

where t is the time (y) elapsed after disposal, $S_s(t)$ is the source term (Bq) for single dump mode, $f_s(t)$ is the failure probability density (y^{-1}) of the multi-barrier system and $C_g(x,y,t)$ is the concentration of the radionuclide in ground water (Bq.cm^{-3}) due to an instantaneous release of unit activity as a line source.

The concentration of a radionuclide in ground water during dumping period, C_{gd} (Bq.cm^{-3}), for multiple dump mode can be evaluated using the convolution integral:

$$C_{gd}(x,y,T) = \int_0^T S_d(T - \tau)f_s(T - \tau)C_g(x,y,\tau)\mathrm{d}\tau \tag{11}$$

where T is the disposal operation period (y) and S_d (T) is the source term (Bq) during dumping period. The running index time, τ, varies from 0 to T to account for annual release of radioactivity in the facility untill the disposal operation is over. Similarly, the concentration during post dumping period, C_{gp} (Bq.cm^{-3}), can be evaluated by the convolution integral:

$$C_{gp}(x,y,t) = \int_0^T S_d(T - \tau)f_s(T - \tau)C_g(x,y,t + \tau)\mathrm{d}\tau + \int_0^t S_p(t - \tau)f_s(T + t - \tau)C_g(x,y,\tau)\mathrm{d}\tau \tag{12}$$

where t is the time elapsed (y) after the termination of dumping and $S_p(t)$ is the source term (Bq) during post dumping period. The first integral in Eq. (12) represents dispersion during the post dumping period due to release during the disposal operation period, T, and the second integral represents dispersion during the post dumping period due to release during the same period.

The solution for unit instantaneous release of radioactivity (Nair and Krishnamoorthy, 1997) for two-dimensional dispersion and transport model (line source) is given by:

$$C_g(x,y,t) = \frac{\exp(-\lambda_p t)\exp[-(x - U_x^1 t)^2/4D_x^1 t]\exp\left(-\dfrac{y^2}{4D_y^1 t}\right)}{4\pi H_g R_g \theta_g \sqrt{D_x^1 D_y^1 t^2}} \tag{13}$$

where x (≥ 0 and $\leq \infty$) is the longitudinal distance (cm), y ($\geq -\infty$ and $\leq \infty$) is the lateral distance (cm), U_x^1 ($= U_x/R_g$) is the retarded ground water velocity (cm.y^{-1}), D_x^1 ($= D_x/R_g$) is the retarded longitudinal dispersion coefficient (cm^2.y^{-1}), D_y^1 ($= D_y/R_g$) is the retarded lateral dispersion coefficient (cm^2.y^{-1}), H_g is the depth of aquifer (cm) and θ_g is the effective porosity. The retardation factor, R_g, is defined as $1 + (K_d\rho_b/\theta_g)$ where K_d (cm^3.g^{-1}) is the distribution coefficient and ρ_b is the bulk density (g.cm^{-3}). The solution represented by Eq. (13) for a line source assumes uniform vertical mixing. Since the thickness of ground water aquifer is about 6 m only, this assumption is quite reasonable.

For simplicity, analytical solutions in closed form can be obtained for the radionuclide concentration in ground water by using one dimensional dispersion and transport solution, $C_g(x,t)$, instead of two dimensional dispersion and transport solution, $C_g(x,y,t)$ in Eqs. (10)–(12) (convolution integrals). The solution for one-dimensional dispersion and transport model in ground water is given (Nair and Krishnamoorthy, 1997) by:

$$C_g(x,t) = \frac{\exp(-\lambda_p t)\exp[-(x - U_x^1 t)^2/4D_x^1 t]}{2\pi A R_g \theta_g \sqrt{D_x^1 t}}$$

where $C_g(x,t)$ is the radionuclide concentration in ground water due to instantaneous release of unit radioactivity and A (cm^2) is the cross sectional area of ground water aquifer.

Thus, the analytical solution, $C_{gs}(x,t)$, for the radionuclide concentration in ground water for single dump mode can be obtained by substituting $C_g(x,\tau)$ for $C_g(x,y,\tau)$ in Eq. (10) as:

$$C_{gs}(x,t) = \left[\frac{M\exp(xU_x/2D_x)\exp(-\lambda_p t)}{4A\theta_g R_g (D_x^1)^{1/2}}\right] \sum_{i=a}^{f}\left[\frac{\lambda_{Ri}\exp(-\lambda_i t)}{\left(\frac{(U_x^1)^2}{4D_x^1} - \lambda_i\right)^{1/2}}\right] F\left[\frac{x}{\sqrt{D_x^1}}, t, \left(\frac{(U_x^1)^2}{4D_x^1} - \lambda_i\right)\right] \quad (14)$$

where λ_{Ri} is given by the equation:

$$\lambda_{Ri} = \prod_{i=a}^{f}\left[\frac{\lambda_i}{\prod_{j\neq i}(\lambda_j - \lambda_i)}\right]$$

The F function in Eq. (14) is defined for $\gamma > 0$ as:

$$F(k,t,\gamma) = e^{-k\sqrt{\gamma}} erfc\left(\frac{k}{2\sqrt{t}} - \sqrt{\gamma t}\right) - e^{+k\sqrt{\gamma}} erfc\left(\frac{k}{2\sqrt{t}} + \sqrt{\gamma t}\right)$$

where k represents $[x/(D_x^1)^{1/2}]$ and γ represents either $\{[(U_x^1)^2/4D_x^1] - \lambda_i\}$ or $\{[(U_x^1)^2/4D_x^1] + \lambda_p - \lambda_i\}$ in the following equations. Depending on the magnitude of U and D in relation to λ_i, the value of the parameter γ can be < 0 in the F function. In such cases, the F function is defined as:

$$\frac{F[k,t,\gamma]}{i(|\gamma|)^{1/2}} = \frac{2i\sin(2xy)}{i(|\gamma|)^{1/2}}\left[\frac{\exp(-x^2)}{2\pi x} - erfc(x)\right] + \frac{4\exp(-x^2)}{i(|\gamma|)^{1/2}\pi}\sum_{n=1}^{\infty}\left\{\frac{\exp(-n^2/4)}{(n^2 + 4x^2)}[2x\sin(2xy) + n\sinh(ny)]\right\}$$

where

$$i = \sqrt{-1},\ x = \frac{k}{\sqrt{4t}},\ y = (|\gamma|t)^{1/2} \text{ and } erfc(x) = 1 - \frac{2}{\sqrt{\pi}}\int_0^x \exp(-t^2)dt$$

Using the same procedure, the analytical solution, $C_{gd}(x,T)$, for the radionuclide concentration in ground water during dumping period (multiple dump mode) can be obtained by substituting $C_g(x,\tau)$ for $C_g(x,y,\tau)$ in Eq. (11) as:

$$C_{gd}(x,T) = \left[\frac{Qexp(xU_x/2D_x)}{4A\theta_g R_g \lambda_p (D_x^1)^{1/2}}\right]\sum_{i=a}^{f}[\lambda_{Ri}\exp(-\lambda T)] \left\{\frac{F\left[\frac{x}{\sqrt{D_x^1}}, T, \left(\frac{(U_x^1)^2}{4D_x^1} + \lambda_p - \lambda_i\right)\right]}{\left(\frac{(U_x^1)^2}{4D_x^1} + \lambda_p - \lambda_i\right)^{1/2}} - \frac{\exp(-\lambda_p T)F\left[\frac{x}{\sqrt{D_x^1}}, T, \left(\frac{(U_x^1)^2}{4D_x^1} - \lambda_i\right)\right]}{\left(\frac{(U_x^1)^2}{4D_x^1} - \lambda_i\right)^{1/2}}\right\} \quad (15)$$

Similarly, the analytical solution, $C_{gp}(x,t)$, for the radionuclide concentration in ground water during post dumping period (multiple dump mode) can be obtained by substituting $C_g(x,t + \tau)$ for $C_g(x,y,t + \tau)$ and $C_g(x,\tau)$ for $C_g(x,y,\tau)$ in Eq. (12) as:

$$C_{gp}(x,t) = \left[\frac{Q\exp(xU_x/2D_x)}{4A\theta_g R_g \lambda_p \sqrt{D_x^1}} \right] \sum_{i=a}^{f} [\lambda_{Ri}\exp[-\lambda_i(t+T)]] \quad (16)$$

$$\left\{ \frac{F\left[\frac{x}{\sqrt{D_x^1}},(t+T),\left(\frac{(U_x^1)^2}{4D_x^1}+\lambda_p-\lambda_i\right)\right]}{\left(\frac{(U_x^1)^2}{4D_x^1}+\lambda_p-\lambda_i\right)^{1/2}} - \frac{F\left[\frac{x}{\sqrt{D_x^1}},t,\left(\frac{(U_x^1)^2}{4D_x^1}+\lambda_p-\lambda_i\right)\right]}{\left(\frac{(U_x^1)^2}{4D_x^1}+\lambda_p-\lambda_i\right)^{1/2}} \right.$$
$$\left. - \frac{e^{-\lambda_p(t+T)}F\left[\frac{x}{\sqrt{D_x^1}},(t+T),\left(\frac{(U_x^1)^2}{4D_x^1}-\lambda_i\right)\right]}{\left(\frac{(U_x^1)^2}{4D_x^1}-\lambda_i\right)^{1/2}} + \frac{e^{-\lambda_p t}F\left[\frac{x}{\sqrt{D_x^1}},t,\left(\frac{(U_x^1)^2}{4D_x^1}-\lambda_i\right)\right]}{\left(\frac{(U_x^1)^2}{4D_x^1}-\lambda_i\right)^{1/2}} \right\}$$

The Eqs. (10)–(12) are semi-analytical solutions and requires numerical integration techniques since they use two-dimensional dispersion and transport model in ground water. These equations can be used to compute the radionuclide concentration in ground water at any time "*t*" along any *x*, *y* directions. The maximum concentration over the lateral distance can be obtained by equating $y = 0$ in these equations. Similar results can also be obtained using Eqs. (14)–(16), which are simplified analytical solutions of the respective cases wherein one-dimensional dispersion and transport model is used in ground water. The computational efficiency of the analytical solutions is better than the semi-analytical solutions. However, Eqs. (10)–(12) are mainly used for the computations in this study and the integration in these equations have been performed using Gauss quadrature method. It is found that 256 terms of the quadrature leads to proper convergence. A comparison of results obtained by these different sets of solutions, namely semi-analytical and analytical solutions, is also included in the paper.

7. Radiological model

The radiological model is used to evaluate radiation dose to a member of the critical group through consumption of ground water for drinking and the corresponding risk. These end points are useful for a direct comparison with the dose limit (1 $mSv.y^{-1}$ to a member of the public) recommended by the International Commission on Radiological Protection (ICRP, 1990) and with the normal risk in a modern society.

The radiation dose due to a radionuclide through drinking water pathway is evaluated as the product of its concentration in ground water, drinking water intake and the ingestion dose coefficient. The ingestion dose coefficients (IAEA, 1996) applicable to general population along with a water intake of 2.2 $L.day^{-1}$ are used in the evaluation. The International Commission on Radiological Protection (ICRP, 1990) evaluated the total risk due to radiation to a member of the public (including risk due to fatal cancer, non-fatal cancer and severe hereditary effects) as 7.3×10^{-5} mSv^{-1}. The product of this risk factor and the dose received will give rise the risk to a member of the critical group due to the waste disposal practice.

8. Software

A program code (REMS) in FORTRAN-90 is produced for the probabilistic safety assessment of near surface disposal of low-level radioactive waste. The code includes options for single dump mode and multiple dump mode. The outputs of the code comprise of the radioactivity release rate into ground water, radionuclide concentration in ground water, radiation dose to a member of the critical group through drinking water and risk to the critical group. The code also generates the total of all these quantities for all the radionuclides concerned. The code is made operational in UNIX and MSDOS environments. A display system is also attached to the code in these environments for visualization of results.

9. Parameters used in the model

The failure scenario for each barrier is summarized in Table 1 along with assumed mean time-to-failure (MTTF) based on realistic assumptions (Kim et al., 1993). The inventory of low-level radioactive waste originating from PHWR reactor operation and fuel reprocessing are calculated as Bq per GWe.y (Table 2) based on operational experience (Narayan, 1998) and theoretical considerations (Cohen, 1984). The radionuclides encountered in the front end of nuclear fuel cycle like radium and thorium are not considered in this study because they are being disposed of separately in India. Table 2 contains other nuclide dependent parameters like half-lives, distribution coefficients for clay (Jiskra, 1985; IAEA, 1997) and ingestion dose coef-

ficients (IAEA, 1996) also. The nuclide independent parameters like thickness of the unsaturated zone between the facility and the water table, seepage velocity in this zone, bulk density and porosity of clay, aquifer thickness, ground water velocity and dispersion coefficients applicable for a typical coastal site at Trombay (Narayan, 1998) in Mumbai are presented in Table 3. The data represented in these tables include a reference level with range of values wherever applicable.

10. Results and discussion

The radioactivity release rates of different radionuclides into ground water (Bq.y^{-1}/GWe.y) for the reference level computed using Eqs. (8) and (9) (multiple dump mode) are presented in Fig. 1. The release rate of a radionuclide is directly proportional to its inventory and reduces if its retardation factor is high in the unsaturated zone between the facility and the water table. The release rates of short-lived radionuclides such as ^{106}Ru, ^{144}Ce (not in the figure) and ^{3}H generally persist only for few years after the termination of dumping. The long-lived radionuclides such as ^{59}Ni, ^{99} Tc and ^{129}I continue to release their activity for long periods of the order of tens to hundreds thousands of years depending on their half-lives. The lowest release rate is delivered by ^{237}Np, which has a low inventory and high retardation in the unsaturated zone. The total annual release rate of all the radionuclides attain a maximum of 4.0×10^9 Bq/GWe.y at 1.25×10^3 years and then declines to lower levels.

The maximum centre-line concentration over time (Bq.ml^{-1}) in ground water at 1.6 km away from the facility and the maximum dose to a member of the critical group through the drinking water pathway for different radionuclides due to disposal of low-level radioactive waste (single dump mode and Eq. (10)) equivalent to 50 GWe.y is presented in Table 4. This table indicates the important radionuclides with respect to the safety assessment of near surface disposal facilities as ^{3}H, ^{14}C, ^{59}Ni, ^{99}Tc, ^{129}I, ^{237}Np and ^{239}Pu. The highest concentration in ground water is observed for ^{14}C followed by ^{59}Ni and ^{129}I. However, the highest dose is due to ^{129}I since the ingestion dose coefficient of ^{14}C due to consumption of inorganic carbon through water is about 2 orders of magnitude lower than that of organic carbon (NCRP, 1985) and is about 5 orders of magnitude lower than that of ^{129}I. The period during which the maximum doses received by a member of the critical group occurs between 1.0×10^2 and 5.0×10^5 years depending on the radionuclides. A comparison between single dump mode and multiple dump mode (Eqs. (11) and (12)) in Table 4 concludes that all long-lived radionuclides show

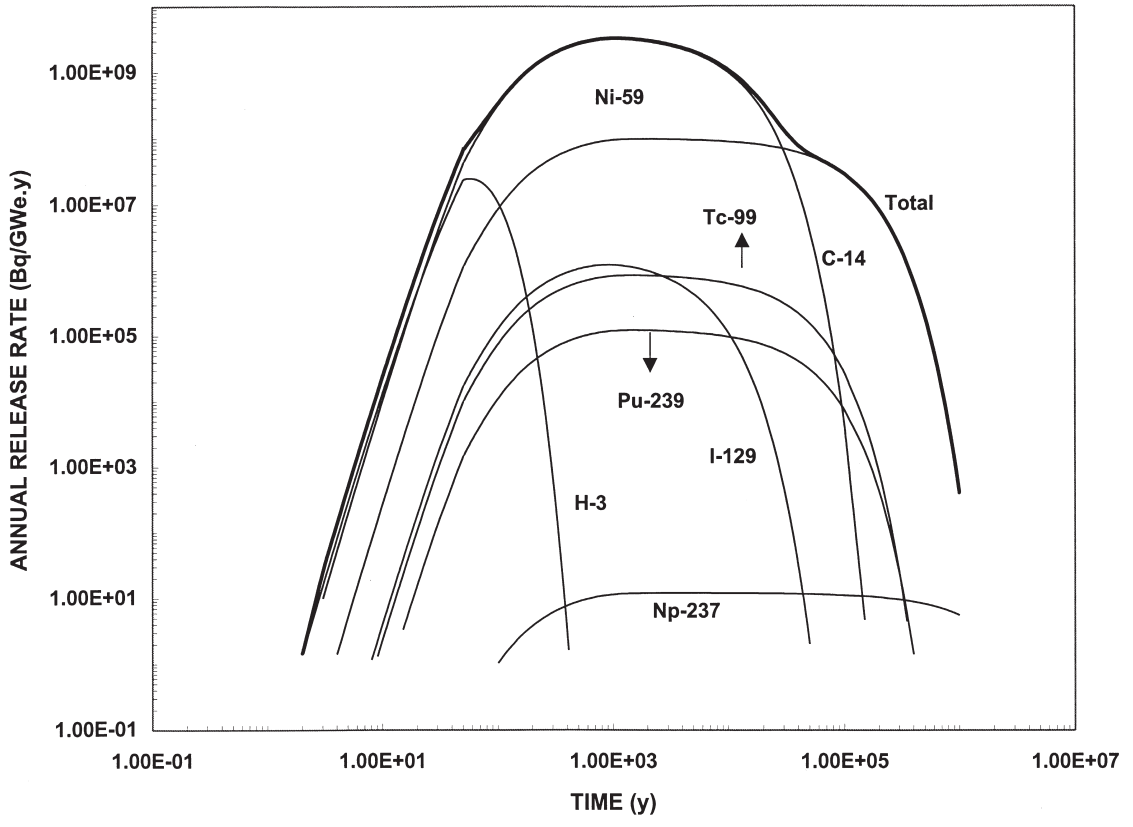

Fig. 1. Time history of radioactivity release rate into ground water from the reference level disposal facility.

Table 4
Maximum concentration in ground water obtained by semi-analytical solution (Eqs. (10)–(12)) and maximum dose to a member of the critical group through drinking water pathway

Radionuclide	Maximum concentration ($Bq.ml^{-1}$)	Maximum dose ($mSv.y^{-1}$)	Arrival time of maximum (y)
Single dump mode: Inventory corresponding to 50 GWe.y			
^{3}H	4.19×10^{-5}	6.05×10^{-7}	1.00×10^{2}
^{14}C	1.85×10^{-1}	9.30×10^{-4}	6.00×10^{3}
^{59}Ni	8.28×10^{-3}	4.19×10^{-4}	2.80×10^{4}
^{99}Tc	8.89×10^{-5}	4.51×10^{-5}	4.00×10^{3}
^{129}I	1.30×10^{-4}	1.15×10^{-2}	1.30×10^{3}
^{237}Np	1.18×10^{-9}	1.05×10^{-7}	2.00×10^{5}
^{239}Pu	5.20×10^{-12}	1.04×10^{-9}	5.00×10^{5}
Multiple dump mode: Inventory corresponding to1 GWe.y for 50 y			
^{3}H	2.28×10^{-4}	3.29×10^{-6}	1.00×10^{2}
^{14}C	1.87×10^{-1}	9.34×10^{-4}	6.05×10^{3}
^{59}Ni	8.28×10^{-3}	4.19×10^{-4}	2.81×10^{4}
^{99}Tc	8.89×10^{-5}	4.57×10^{-5}	4.05×10^{3}
^{129}I	1.30×10^{-4}	1.15×10^{-2}	1.25×10^{3}
^{237}Np	1.18×10^{-9}	1.05×10^{-7}	2.00×10^{5}
^{239}Pu	5.21×10^{-12}	1.05×10^{-9}	6.00×10^{5}

Note: Concentrations of ^{54}Mn, ^{55}Fe, ^{60}Co, ^{63}Ni, ^{90}Sr, ^{106}Ru, ^{137}Cs, ^{144}Ce, ^{238}Pu, ^{241}Pu and ^{241}Am are either $< 1 \times 10^{-50}$ or equal to zero.

more or less same concentrations for both the dump modes. Differences in concentrations are expected only for short-lived radionuclides like ^{3}H and for short distances from the facility. Similar results obtained using the analytical solutions (Eqs. (14)–(16)) are presented in Table 5. A comparison of Tables 4 and 5 reveals that the results obtained by semi-analytical and analytical methods are almost same. Any difference in these values are due to uncertainty in evaluation of the cross sectional area of aquifer as well as due to the assumption of uniform lateral mixing in the analytical case.

Table 5
Maximum concentration in ground water obtained by analytical solution (Eqs. (14)–(16)) and maximum dose to a member of the critical group through drinking water pathway

Radionuclide	Maximum concentration ($Bq.ml^{-1}$)	Maximum dose ($mSv.y^{-1}$)	Arrival time of maximum (y)
Single dump mode: Inventory corresponding to 50 GWe.y			
^{3}H	4.94×10^{-5}	7.15×10^{-7}	1.00×10^{2}
^{14}C	2.26×10^{-1}	1.12×10^{-3}	6.00×10^{3}
^{59}Ni	1.01×10^{-2}	5.09×10^{-4}	2.80×10^{4}
^{99}Tc	1.08×10^{-4}	5.56×10^{-5}	4.00×10^{3}
^{129}I	1.57×10^{-4}	1.39×10^{-2}	1.30×10^{3}
^{237}Np	1.44×10^{-9}	1.27×10^{-7}	2.00×10^{5}
^{239}Pu	6.60×10^{-12}	1.32×10^{-9}	5.00×10^{5}
Multiple dump mode: Inventory corresponding to 1 GWe.y for 50 y			
^{3}H	2.76×10^{-4}	3.99×10^{-6}	1.00×10^{2}
^{14}C	2.27×10^{-1}	1.13×10^{-3}	6.05×10^{3}
^{59}Ni	1.01×10^{-2}	5.09×10^{-4}	2.81×10^{4}
^{99}Tc	1.08×10^{-4}	5.56×10^{-5}	4.05×10^{3}
^{129}I	1.57×10^{-4}	1.39×10^{-2}	1.25×10^{3}
^{237}Np	1.44×10^{-9}	1.27×10^{-7}	2.00×10^{5}
^{239}Pu	6.54×10^{-12}	1.31×10^{-9}	6.00×10^{5}

Note: Concentrations of ^{54}Mn, ^{55}Fe, ^{60}Co, ^{63}Ni, ^{90}Sr, ^{106}Ru, ^{137}Cs, ^{144}Ce, ^{238}Pu, ^{241}Pu and ^{241}Am are either $< 1 \times 10^{-50}$ or equal to zero.

The time history of radionuclide centre-line concentrations in ground water at 1.6 km computed for the reference level using Eqs. (11) and (12) (multiple dump mode) are shown in Fig. 2. Its inventory, half-life, sorption capacity and transit period to reach that distance govern the concentration of a radionuclide at any distance. The short-lived and less sorbing radionuclides decay to innocuous levels during their transit. The long-lived radionuclides such as ^{3}H, ^{14}C, ^{59}Ni, ^{99}Tc, ^{129}I, ^{237}Np and ^{239}Pu reach 1.6 km distance with significant concentrations. The highest concentration is delivered by ^{14}C (1.9×10^{-1} $Bq.ml^{-1}$) followed by ^{59}Ni (8.3×10^{-3} $Bq.ml^{-1}$), ^{3}H (2.3×10^{-4} $Bq.ml^{-1}$) and ^{129}I (1.3×10^{-4} $Bq.ml^{-1}$). The maximum concentrations of these radionuclides occur between 1.0×10^{2} and 2.8×10^{4} years. All these four radionuclides are less sorbing and long-lived (except ^{3}H). The long-lived and high sorbing radionuclides such as ^{237}Np and ^{239}Pu show low concentrations in ground water. It is to be noted that the maximum total concentration occurs at about 6.0×10^{3} years which is predominantly contributed by ^{14}C. During the later periods (between 1.0×10^{4} and 5.0×10^{5} years) when the concentration of ^{14}C starts decreasing, the contributions from ^{59}Ni becomes significant. The peak concentrations of transuranics are low ($< 1.0 \times 10^{-9}$ $Bq.ml^{-1}$) and they are observed after about hundreds thousands years.

The time history of annual effective dose (mSv/GWe.y) through drinking water from a well located 1.6 km away from the facility is presented in Fig. 3 for multiple dump mode. The maximum annual effective dose (1.2×10^{-2} mSv) is contributed by ^{129}I at 1.25×10^{3} years after disposal followed by ^{14}C (9.3×10^{-4} mSv at 6.05×10^{3} y), ^{59}Ni (4.2×10^{-4} mSv at 2.81×10^{4} y) and ^{99}Tc (4.6×10^{-5} mSv at 4.05×10^{3} y). The maximum total annual dose (1.2×10^{-2} mSv) occur at 1.25×10^{3} years after disposal. The doses delivered by the transuranics and ^{3}H are very low due to low inventory and high K_d values in the case of transuranics or low ingestion dose coefficient in the case of ^{3}H. It can be seen that the dose during the first 4.0×10^{2} years is dominated by ^{3}H, between 4.0×10^{2} and 2.0×10^{3} years by ^{129}I, between 2.0×10^{3} and 2.0×10^{4} years by a group of radionuclides comprising of ^{129}I, ^{14}C and ^{99}Tc and beyond 2.0×10^{4} years by ^{59}Ni. This figure can be used to deduce an approximate waste acceptance criteria with respect to the drinking water pathway. If the apportioned dose limit for the near surface disposal facility is 0.05 $mSv.y^{-1}$, the source inventory may be increased by a factor of 5 (= 0.05/0.01) and the resulting dose will comply with the regulatory criteria.

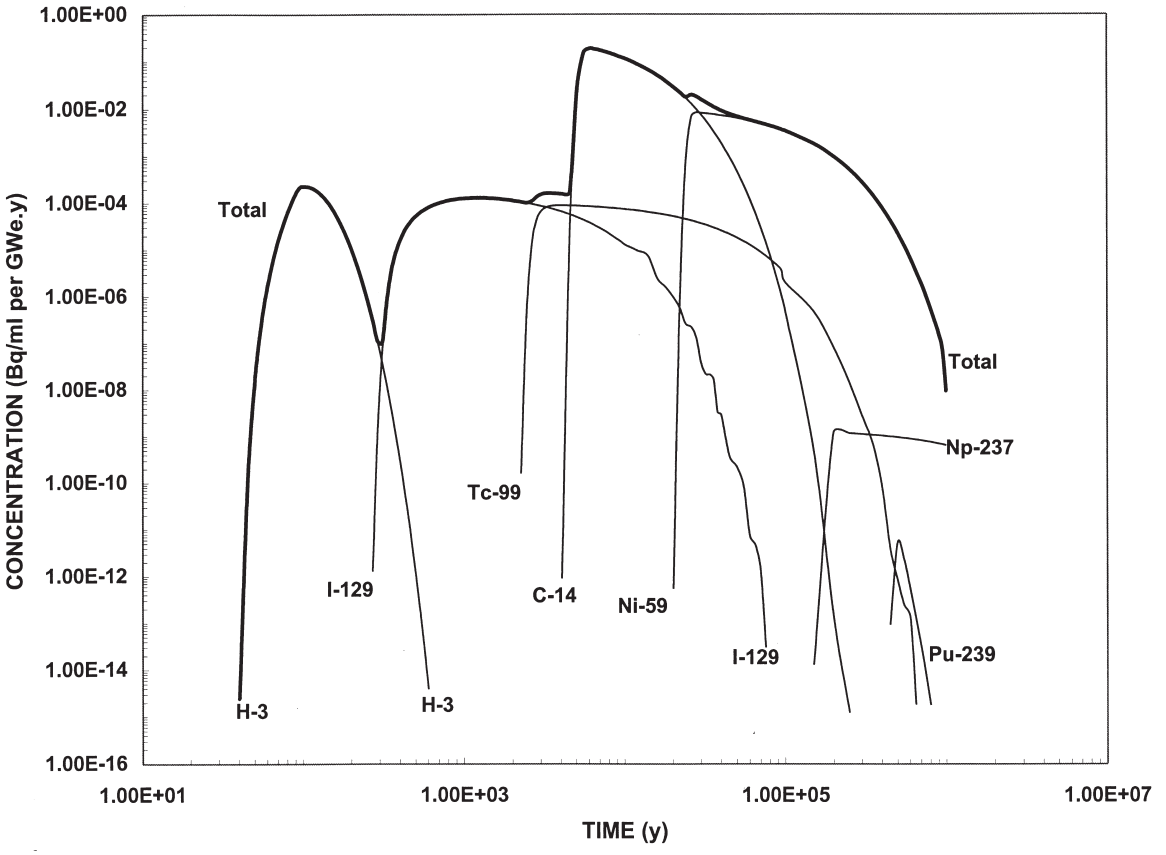

Fig. 2. Time history of radionuclide concentration in ground water at 1.6 km down flow distance from the reference level disposal facility.

The variation of total effective dose for all radionuclides with distances from the disposal facility is depicted in Fig. 4. It can be observed that the maximum effective dose decreased by a factor of 2 from 0.5 km to 1.6 km and by a factor of 30 from 0.5 km to 5.0 km.

The risk from the near surface disposal of low-level radioactive waste is computed for multiple dump mode as the product of the dose received ($mSv.y^{-1}$) and the ICRP risk factor of 7.3×10^{-5} mSv^{-1} (ICRP, 1990). The maximum risk over time is obtained as 9.0×10^{-7} y^{-1} and this is lower than the risk observed from industrial accidents and natural catastrophes (1×10^{-3}–1×10^{-4} y^{-1}). The average annual dose due to natural background radiation is estimated to be 2.4 mSv world-wide. The corresponding risk due to natural background radiation can be estimated using the ICRP total risk factor (7.3×10^{-5} mSv^{-1}) as 1.8×10^{-4} y^{-1}. It is worth mentioning that the risk due to disposal of low-level radioactive waste generated from 50 GWe.y energy production is lower by a factor of 200 than that arising from natural background radiation. The risk derived in this study is higher than that reported by Kim et al. (1993) by a factor of 250 primarily due to the higher inventory and low distribution coefficient used for ^{129}I in the study. However, it is lower by a factor of 400 than that reported by Cohen (1984) which is more probably an over estimate than an under estimate of the actual health effects. Unlike Kim et al. (1993), this study conclusively states that the dose contributions from short-lived radionuclides like ^{60}Co, ^{63}Ni, ^{90}Sr and ^{137}Cs are insignificant at all time periods after disposal. Since these are short-lived and highly sorbing radionuclides, no significant radiological impacts are expected from them. The risk factor used in this study includes risk due to fatal cancer, non-fatal cancer and severe hereditary effects. If the risk factor due to fatal cancer alone is used, as in the case of Cohen (1984), the annual risk to the critical group will be lowered by 1.5 times.

The sensitivity analysis has been carried out to identify the critical parameters, which have maximum effect on the concentration of ^{129}I in ground water at 1.6 km from the facility. The sensitivity index for the change of a candidate parameter can be defined as:

$$S_I = \left(1 - \frac{C_R}{C}\right) / P_I \tag{17}$$

where C_R is the maximum concentration for the reference level and C is the maximum concentration computed using the candidate parameter P whose impact is to be evaluated. The index P_I is defined as $[1 - (P/P_R)]$ for a decrease of the candidate parameter in relation to

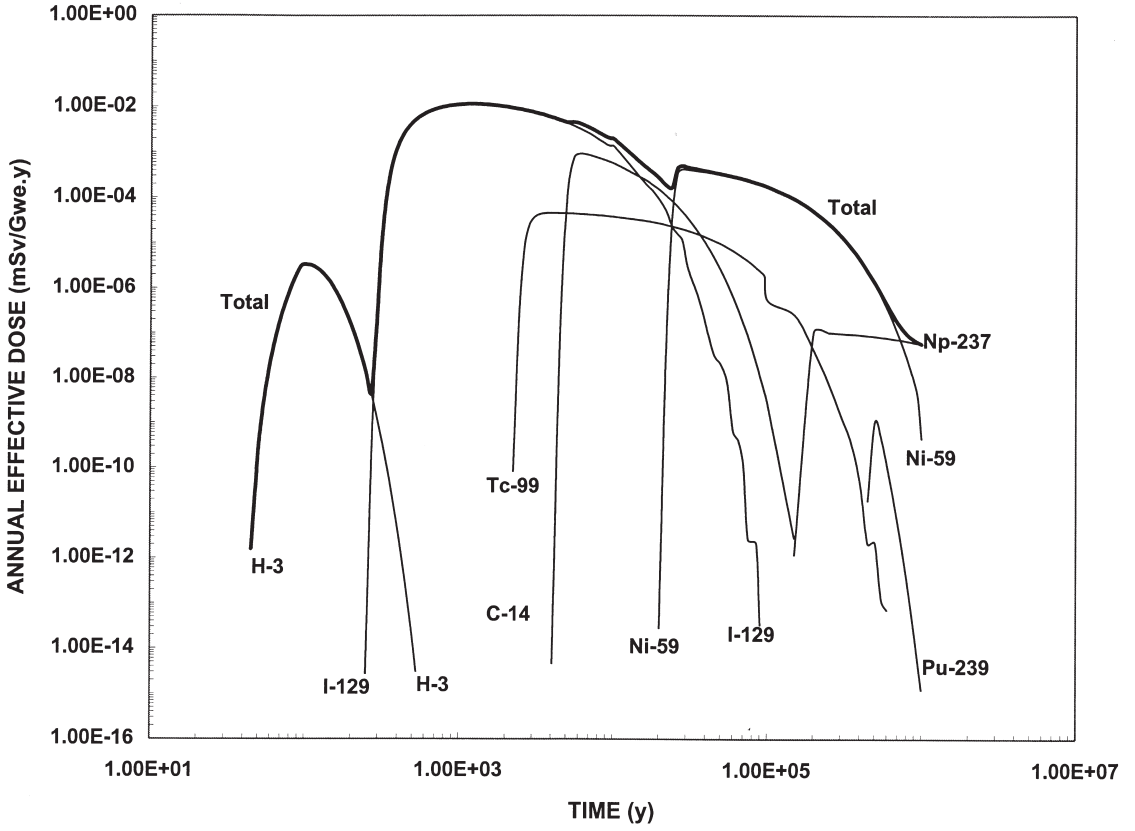

Fig. 3. Time history of annual effective dose to a member of the critical group through consumption of drinking water from a well located at 1.6 km down flow distance from the reference level disposal facility.

the reference parameter P_R and as $[1 - (P_R/P)]$ for an increase of the candidate parameter. The sensitivity indices of different parameters are given in Table 6.

The negative sign of the sensitivity index indicates that the concentration decreases with respect to the reference level whereas the positive sign indicates the reverse process. The relative magnitude of change in concentration is not constant for decrease and increase processes indicating non linearity in many cases. The most sensitive parameter is found to be as distribution coefficient followed by seepage velocity, dispersivity and thickness of unsaturated zone. The sensitivity indices shown above are more applicable for long-lived and less sorbing radionuclides. For short-lived and high sorbing radionuclides, the magnitude of these indices may vary considerably.

The uncertainty analysis provides a quantitative estimate of the range of model outputs that result from uncertainties in the structure of the model or the inputs to the model. If the analysis is carried out appropriately, the output range will contain the true value that the model seeks to predict. The analysis can also be extended to identify the sources that dominate the overall uncertainty, so that priorities can be set for work aimed at reducing the uncertainty. Uncertainty in model predictions can arise from a number of sources (IAEA, 1989), including specification of the problem, formulation of the conceptual model, formulation of mathematical model, estimation of parameter values, and calculation and interpretation of results. Of these sources, uncertainties due to estimation of parameter values in the REMS are quantified based on a random sampling program using random number generators available in the NAG FORTRAN library.

Probability density functions are constructed for parameters such as barrier integrity, distribution coefficient, thickness of unsaturated zone, seepage velocity in the unsaturated zone, ground water velocity and dispersivity. The uncertainties in these parameters are propagated through the model to generate the cumulative probability distribution of predicted radiation dose (Fig. 5) to a member of the critical group due to near surface radioactive waste disposal practice. The peak annual effective doses obtained through the uncertainty analysis fall in a log normal distribution. The statistical analysis (Hoffman et al., 1978) indicate that the range of annual effective dose arising out of random selection of 101 combination of parameters lie between 1.7×10^{-4} and 4.1 mSv with a geometric mean of $3.6 \times 10^{-2} \pm 3.9$ mSv. The most probable annual effective dose is obtained as 2.8×10^{-4} mSv. Regarding the context of this large range of effective dose, it can be stated that

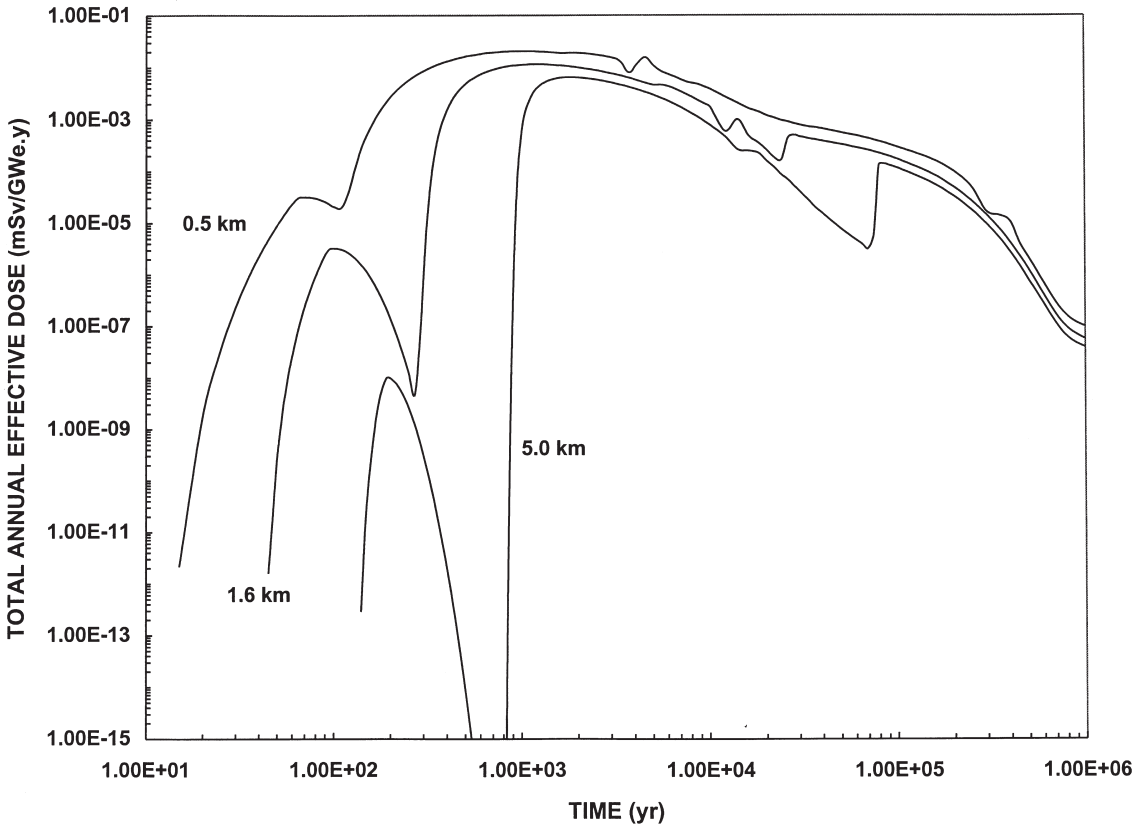

Fig. 4. Time history of total annual effective dose to a member of the critical group through consumption of drinking water from wells located at 0.5 km, 1.6 km and 5.0 km down flow distance from the reference level disposal facility.

Table 6
Sensitivity indices with respect to the reference level waste disposal facility

Parameter	Sensitivity index	Change of parameter
Ground water velocity	+ 0.67	Decrease
	− 0.66	Increase
Seepage velocity	− 6.20	Decrease
	+ 0.88	Increase
Thickness of unsaturated zone	+ 0.85	Decrease
	− 1.30	Increase
Dispersivity	+ 0.76	Decrease
	− 2.40	Increase
Porosity	+ 0.28	Decrease
	− 0.38	Increase
Distribution coefficient	+ 0.80	Decrease
	− 6.90	Increase
Barrier integrity	+ 0.18	Decrease
	− 1.10	Increase

the mean and most probable doses derived from this study are much lower than the dose limit recommended by the ICRP (1 mSv) for a member of the public. They are also lower than the authorized dose limit being practised in India (5.0×10^{-2} mSv) for terrestrial exposure route. The maximum effective dose in the range (4.1 mSv) arises out of the worst release scenario and is only two times higher than the global average of natural background radiation. The annual effective dose obtained for the reference level (1.2×10^{-2} mSv) is about 3 times lower than the geometric mean of the range.

11. Conclusions

A risk analysis methodology has been developed for the safety assessment of near surface disposal facilities for low-level radioactive waste. The model can generate the radioactivity release rate into the geosphere, the radionuclide concentration in ground water, the annual effective dose to a member of the critical group through drinking water pathway and the risk to the critical group due to waste disposal practice. The maximum annual effective dose to a member of the critical group for the reference level parameters works out to be 1.2×10^{-2} mSv. The geometric mean of annual effective dose to a member of the public generated through the uncertainty analysis is about 3 times higher than that of the reference level. However, the most probable annual effective dose obtained through the uncertainty analysis (2.8×10^{-4}

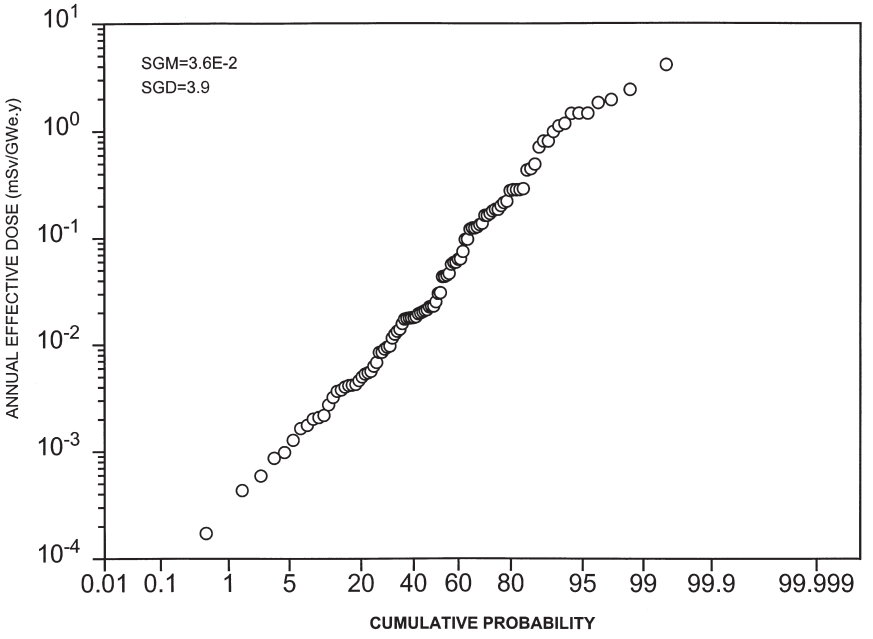

Fig. 5. Cumulative probability of annual effective dose to a member of the critical group generated through uncertainty analysis.

mSv) is much lower than the dose obtained for the reference level.

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