A SIMPLIFED PERFORMANCE ASSESSMENT MODEL FOR RADIOACTIVE WASTE DISPOSAL ALTERNATIVES

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This paper describes the development of a simplified performance assessment (PA) model, within a structured PA framework, that can be used to make fast, high-level evaluations of, and comparisons between, potential long-term radioactive waste disposal system alternatives. The utility of the of the simplified PA model is demonstrated by applying it to two published PAs: one for mined geologic disposal in clay and one for deep borehole disposal in granite.

I. INTRODUCTION

Within the U.S. Department of Energy Office of Nuclear Energy (DOE-NE) Fuel Cycle Technologies (FCT) program, the Used Fuel Disposition (UFD) campaign is sponsoring activities to explore alternative options for the long-term disposal of the nation's current and future inventory of used nuclear fuel (UNF) and high-level radioactive waste (HLW). This paper describes the development and application of a simplified PA model, within a structured PA framework, that can be used to make fast, high-level evaluations of, and comparisons between, potential long-term radioactive waste disposal system alternatives.

Over the past three decades, Sandia National Laboratories (SNL) has developed and applied a PA methodology that has informed key decisions concerning radioactive waste management both in the U.S. and internationally[1]. The SNL PA methodology includes: setting performance goals, characterizing the disposal system (identifying system components, identifying and screening features, events, and processes (FEPs)), identifying scenarios, building the system model (both conceptual and numerical), consequence modeling (e.g., source term, groundwater flow and radionuclide transport through the engineered system and geosphere, biosphere transport and health effects), and evaluating system performance including uncertainty and sensitivity analysis.

The simplified PA model was developed within the SNL PA methodology. Section II describes the identification of a conceptual model for a generic disposal system that considers: (1) a set of disposal system alternatives comprised of combinations of potential waste forms, disposal concepts, and geologic settings; (2) a set of generic disposal system components representative of the range of potential disposal system alternatives; and (3) a set of generic FEPs potentially relevant to the range of disposal system alternatives.

Section III describes the development of a simplified generic system PA model to capture the thermal-hydrologic-chemical-mechanical-biological-radiological (THCMBR) phenomena, as described by the conceptual model, that control the physical and chemical environments that influence system performance. To support the UFD objective of evaluating a range of disposal system alternatives, the simplified PA model uses mean annual dose as a system performance measure. Specific components of system performance include: (1) waste form degradation and the radionuclide source term; (2) radionuclide transport through the engineered components; (3) radionuclide transport through the host rock and surrounding geologic units; and (4) radionuclide transport, uptake, and health effects in the biosphere.

Section IV describes the application of the simplified generic PA model to two specific disposal concepts and settings (i.e., site-specific scenarios) for which published PA results are available.

II. GENERIC DISPOSAL SYSTEM CONCEPTUAL MODEL

The simplified PA model was designed to have the capability to evaluate disposal system performance for a range of disposal system alternatives. For the DOE-NE UFD campaign, the range of alternatives for long-term disposal of UNF and HLW is defined by possible combinations of four waste form types (UNF, HLW glass, HLW ceramic, HLW metal alloy) and five disposal concepts/geologic settings (mined disposal in unsaturated crystalline rock, mined disposal in saturated crystalline rock, mined disposal in saturated clay/shale, mined disposal in salt, and deep borehole disposal in saturated crystalline rock[2].

The basis for a generic disposal system conceptual model representative of these 20 disposal system alternatives was identified using a FEP analysis methodology[2]. The FEP analysis consisted of the identification of (1) a set of generic disposal system components common to the range of disposal system alternatives, and (2) a set of generic FEPs potentially relevant to the range of disposal system alternatives.

Fig. 1 shows a one-dimensional conceptualization of the key components of a generic disposal system that

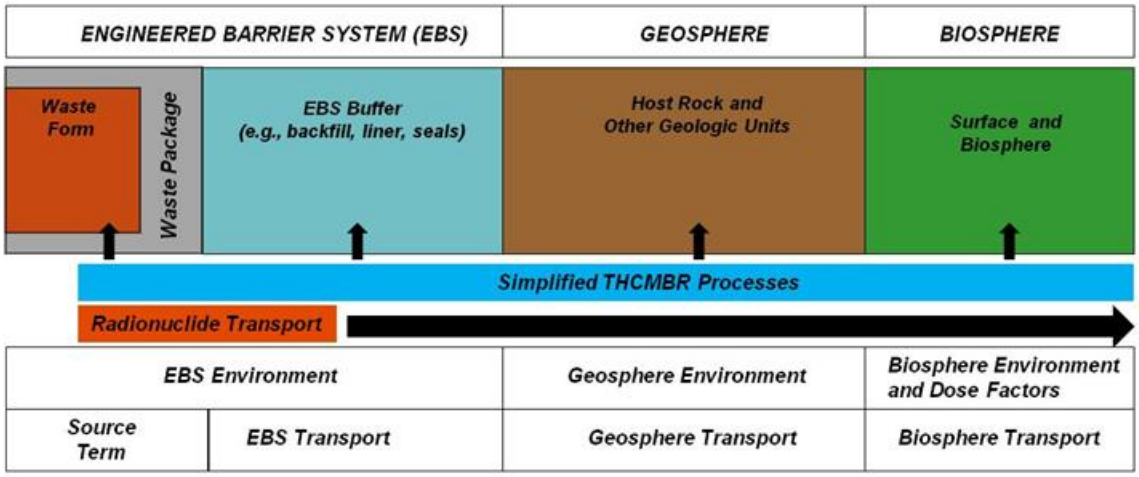


Figure 1. Key components and high-level phenomena of a generic disposal system.

includes domains, features, and phenomena common to most of the 20 disposal system alternatives. The generic system contains three domains: the engineered barrier system (EBS), the geosphere, and the biosphere. Each of these three domains contains features: waste form, waste package, buffer/backfill, tunnel/open space, and seals/liner for the EBS; excavation disturbed zone (EDZ), host rock, and other geologic units for the geosphere; and the land surface for the biosphere.

Phenomena that can affect these features, domains, and/or the disposal system as a whole are represented by FEPs. A FEP typically represents a process or event acting upon or within a feature. FEPs range from slow coupled THCMBR processes acting within or between features to discrete external events such as seismicity. A set of 208 UFD FEPs[2] potentially relevant to the 20 disposal system alternatives was developed by identifying high-level processes and events that could act upon or within the generic disposal system components shown in Fig. 1. These UFD FEPs were informed by the Nuclear Energy Agency (NEA) International FEP Database[3,4], which contains approximately 1,650 FEPs from 10 different national radioactive waste disposal programs. The NEA FEPs cover a wide range of disposal system designs and geologic settings and are organized using a hierarchical numbering scheme according to the same generic domains shown in Fig. 1.

III. SIMPLIFIED PA MODEL COMPONENTS

Radioactive waste disposal system models have been developed and applied by national programs in several countries.[5, 6, 7, 8, 9] The study objectives for each of these model applications have focused on estimating the risk to human health (e.g., annual dose) for a very specific disposal system concept, design, and geology and, as a result, these models are not very flexible to changes in designs or disposal conditions. In contrast, the simplified generic system PA model was developed to evaluate disposal system performance for a range of disposal system alternatives. To achieve this flexibility, the simplified PA model utilizes the generic conceptual model components described in Section II and applies a modular and simplified representation of the important coupled THCMBR phenomena that are described by the FEPs.

Specific high-level phenomena that are important to disposal system performance, based on FEP analysis from multiple disposal system concepts[3,4], are shown schematically in the lower part of Fig. 1. These high-level phenomena, which encompass the UFD FEPs, include: (1) waste form degradation and the radionuclide source term; (2) radionuclide transport through the EBS; (3) radionuclide transport through the geosphere; and (4) radionuclide transport, uptake, and health effects in the biosphere. In addition to the direct effects of the FEPs on radionuclide mobilization and transport captured in the four system performance components above, the FEPs also influence the physical and chemical environments in the EBS, geosphere, and biosphere, which in turn affect water movement, degradation of EBS components, and radionuclide transport.

These high-level phenomena and environments are implemented in the simplified generic PA model in a quasi-one-dimensional (1-D) form. Radionuclide transport is calculated using 1-D mathematical solutions, but geometric multipliers are employed to account for two- and three-dimensional effects and produce appropriate concentrations and doses. Specific details of the implementation are discussed in subsequent subsections.

The simplified generic PA model calculates the following performance measures, either deterministically or probabilistically: annual dose to a human receptor; radionuclide mass in place and dissolved concentration in each domain, and radionuclide fluxes across domains.

These performance measures are consistent with high-level performance metrics in most published repository performance assessments and with current U.S. high-level radioactive waste regulations.[10,11] .

III.A. Source Term

In the simplified PA model, the source term captures the processes that produce dissolved radionuclides for potential advective and/or diffusive transport to the surrounding EBS. The source term includes the effects of waste form degradation and dissolution and radionuclide dissolution within a waste package volume. The rates at which these processes occur are specified as a function of the EBS chemical environment. The source term acts within the waste form and waste package generic features and is characterized by the following properties:

- Initial radionuclide mass inventory in a waste package. Radionuclide decay (based on a decay constant, λ) and ingrowth (based on specified decay chains) are accounted for throughout the model.
- Number of waste packages and a waste package failure time distribution.
- Waste package cross-sectional area for radionuclide release (A_{WP}).
- Fractional waste form degradation rate.
- Radionuclide solubility limits.
- Groundwater pore velocity through the waste package (v_{WP}), which controls advective flux to the EBS.
- Radionuclide free water diffusion coefficients (D_m), which control diffusive flux to the EBS in conjunction with a concentration gradient.

The dissolved concentration of each radionuclide within a waste package (i.e., the source concentration) is determined by: (1) calculating the available radionuclide mass based on waste package failure and waste form degradation rates, (2) calculating an inventory concentration limit based on dissolving the available radionuclide mass into the waste package water volume; and (3) selecting the lower of the inventory limit and the solubility limit as the source concentration for each radionuclide. The source concentrations decline as the radionuclide inventories deplete due to advective and diffusive radionuclide fluxes from the waste package to the surrounding EBS. Advective fluxes into the EBS are calculated for each radionuclide based on the advective velocity and the source concentration. Diffusive fluxes into the surrounding EBS are calculated for each radionuclide based on the diffusion coefficients and the concentration gradients between the waste package and the surrounding EBS. The following radionuclide masses in the waste package are also calculated over time: undegraded mass (i.e., still part of the waste form), precipitated mass (i.e., released from the waste form, but undissolved due to solubility constraints), and dissolved mass.

III.B. EBS Transport

EBS transport captures the processes that control radionuclide movement through the system of engineered buffer, backfill, open space, seal, and/or liner materials. For a specific disposal system design, radionuclide movement in the EBS is the result of complex coupling of THCMBR processes that interact within and through an evolving set of transport pathways. In the simplified PA model, radionuclide movement in the EBS is represented as 1-D transport by advection and hydrodynamic dispersion with sorption and radionuclide decay.

The properties of the transport medium and pathway can be representative of a single dominant material (e.g., the backfill) or a combination of materials. Specific properties used to characterize EBS transport, which are dependent on the medium/material and on the thermal-chemical state of the material, include:

- EBS transport path length.
- EBS transport path cross-sectional area, A_{EBS}.
- EBS material porosity (n), tortuosity (τ), bulk density (ρ_b), and degree of saturation (s).
- Radionuclide distribution coefficients (k_d), which describe the partitioning of radionuclides between the dissolved phase and the sorbed phase in the EBS.
- Groundwater pore velocity through the EBS (v_x), which is a function of the hydraulic gradient, hydraulic conductivity, and porosity.
- Longitudinal dispersivity (α_x)

The one-dimensional form of the advection-dispersion equation for reactive radionuclides in porous medium, which includes the effects of advection, dispersion, diffusion, sorption, and radionuclide decay, is[14,15]:

$$\frac{\partial C}{\partial t} = \left(\frac{D_x}{R_f}\right)\frac{\partial^2 C}{\partial x^2} - \left(\frac{v_x}{R_f}\right)\frac{\partial C}{\partial x} - \lambda C \qquad \text{(Eqn. 1)}$$

where:

C = dissolved radionuclide concentration [M/L^3]
t = time [T]
x = distance in direction of groundwater flow [L]
D_x = coefficient of hydrodynamic dispersion [L^2/T]
= $\alpha_x v_x + \tau D_m$
R_f = retardation factor [ ]
= $1 + \rho_b k_d / n$

In Eqn. 1, the first right-hand-side term represents mass flux due to hydrodynamic dispersion (the combined

effects of molecular diffusion and longitudinal mechanical dispersion, adjusted for sorption), the second right-hand-side term represents mass flux due to advection adjusted for sorption, and the third right-hand-side term represents mass loss due to radionuclide decay.

Eqn. 1 is solved for each radionuclide to calculate dissolved concentration within the EBS transport pathway and mass flux into the geosphere as a function of time. In addition, radionuclide masses sorbed onto the EBS medium/material are calculated using the k_d values.

The dissolved concentration in the EBS calculated from Eqn. 1 for each radionuclide results from the radionuclide mass that enters the EBS from the waste package source. Conceptually, the mass flux out of waste package cross-sectional area A_{WP} enters EBS transport path transport path cross-sectional area, A_{EBS}. If A_{WP} (and the corresponding mass) is scaled up to represent multiple waste packages, then A_{EBS} is scaled up correspondingly.

III.C. Geosphere Transport

Geosphere transport captures the processes that control radionuclide movement through (1) the host rock, and (2) other geologic units.

III.C.1 Host Rock

In the simplified PA model, radionuclide movement through the host rock is represented by the same 1-D transport conceptual model described in Section III.B for radionuclide transport through the EBS. Eqn. 1 is solved for each radionuclide to calculate dissolved concentration within the host rock transport pathway and mass flux into the other geologic units as a function of time. Radionuclide masses sorbed onto the host rock are calculated using the k_d values. The specific properties used to characterize host rock transport are the same as those listed in Section III.B, but the actual parameter values are likely to be different than in the EBS because the geology, flow pathways, and thermal-chemical state of the host rock is typically quite different from the EBS.

The dissolved concentration in the host rock, C_{HR}, for each radionuclide results from the radionuclide mass that enters the host rock from the EBS. Conceptually, the mass flux out of EBS cross-sectional area A_{EBS} enters host rock transport path cross-sectional area, A_{HR}. However, the transition from 1-D EBS transport (in a localized flow field having groundwater pore velocity v_x) to 1-D host rock transport (in a regional flow field having groundwater pore velocity v_{HR}) requires consideration of two geometric/dilution factors in the simplified PA model to maintain representative dissolved concentrations.

First, the difference in the hydraulic gradients (and other flow path properties) between the localized EBS flow field and the regional host rock flow field will result in a different volumetric groundwater flux (pore velocity x cross-sectional area x porosity) through the host rock than through the EBS. These differences will result in a change in dissolved concentration at the model boundary due to dilution, even though mass is conserved. As an example, in the likely case that the volumetric flux through the host rock is greater, a volumetric flux of uncontaminated water is required to be introduced to the host rock to mix with the volumetric flux containing dissolved radionuclides coming from the EBS. This is equivalent to a host rock dilution factor.

Second, transverse spreading (Eqn. 1 only includes longitudinal spreading) will result in a reduction in the 1-D calculated dissolved concentrations as the transported mass is distributed across a larger effective cross-sectional area. Conceptually, radionuclide mass from a single waste package is transported longitudinally in the simplified PA model through a "pipe" having fixed cross-sectional area (A_{WP}, A_{EBS}, or A_{HR}). Multiple waste packages produce multiple pipes. However, in between the waste packages there will be some cross-sectional area (that is a function of waste package and/or tunnel spacing) that is not intersected by a pipe and will therefore not contain any radionuclide mass. As radionuclides are transported further longitudinally from the waste packages, it is likely to expect the mass in the pipes to be transversely spread into the areas between the pipes. In the simplified PA model, this transverse spreading is accounted for with a transverse spreading/dilution factor, D_{TS}:

$$D_{TS} = \frac{A_{TS}}{A_{WP}} \qquad \text{(Eqn. 2)}$$

where:

A_{TS} = cross-sectional area of transverse spreading [L^2]

The cross-sectional area of transverse spreading includes the cross-sectional area of the "pipes" (i.e., A_{WP}) plus the additional cross-sectional area into which spreading occurs. The cross-sectional area of transverse spreading is defined strictly as an area (i.e., it does not necessarily have a length, width, or radius), so it can represent any geometry, such as might derive from a combination of end-to-end waste package spacing and the spacing between adjacent emplacement locations (tunnels, boreholes, drifts, etc.). As an example, for a grid of 1 m by 1 m waste packages having 2 m center-to-center spacing, $A_{WP} = 1\ m^2$, $A_{TS} = 4\ m^2$ and $D_{TS} = 4$.

The transverse spreading/dilution factor is the same whether the source is a single waste package or multiple waste packages, as long as the cross-sectional areas, A_{WP} and A_{TS}, are defined consistently. If A_{WP} represents the total cross-sectional area of multiple waste packages, then A_{TS} must correspond to the total cross-sectional area of all waste packages into which spreading occurs.

In summary, while the radionuclide mass is conserved in the host rock, the associated dissolved radionuclide concentrations, C_{HR}, may be subject to a host rock dilution factor and/or a transverse spreading/dilution factor.

III.C.2 Other Geologic Units

In the simplified PA model, other geologic units are represented by an aquifer directly adjacent to the host rock. The 1-D advective-dispersive transport described in Section III.C.1 produces a time-dependent radionuclide mass flux from the host rock to the adjacent aquifer. The radionuclide mass entering the aquifer is then assumed to: (1) undergo 1-D advective-dispersive transport to the location of a groundwater withdrawal well within the aquifer, and (2) be transported to the biosphere (e.g., the land surface) under a specified groundwater withdrawal rate. The groundwater withdrawal pumping rate is based upon assumptions about the local biosphere (See Section III.D).

The dissolved radionuclide concentration at the well, C_W, is calculated based on the 1-D transport mechanisms through the aquifer (advection, dispersion, diffusion, sorption, and decay) and an aquifer dilution factor, D_{AQ}:

$$D_{AQ} = \frac{Q_W}{Q_{HR}} \qquad \text{(Eqn. 3)}$$

where:

Q_W = withdrawal (pumping) rate from aquifer [M^3/T]

Q_{HR} = groundwater flux from the host rock [M^3/T]

= $v_{HR} \cdot A_{HR} \cdot n_{HR}$

There are several aspects of this simplified groundwater pumping model that tend to underestimate the transport time between the radionuclide source and the withdrawal well, and underestimate the amount of dilution of radionuclide concentrations:

- No recharge is applied to the aquifer. In areas where there is substantial precipitation much of the groundwater captured by the pumping well would come from local recharge.
- All of the radionuclide mass in the aquifer is assumed to be captured by the pumping well. The capture zone for a pumping well located in a regional groundwater flow field only extends a finite depth. The host rock radionuclide source could be deeper than the capture zone of the pumping well.

III.D. Biosphere Transport and Dose

In the simplified PA model, radionuclide release to the biosphere is conceptualized to be through a withdrawal well and is quantified by the dissolved radionuclide concentration in the well (see Section III.C.2). The corresponding human health effects are represented by calculating an annual dose to a receptor, commonly reported in units of mrem/yr or µSv/yr.

The annual dose (DOSE) to a receptor from each radionuclide in the withdrawal well is calculated using the specific activity (S_A) and the biosphere dose conversion factors (BDCF):

$$\text{DOSE}\ (t) = C_W(t) * S_A * \text{BDCF} \qquad \text{(Eqn. 4)}$$

The BDCFs capture the lifestyle and characteristics of a hypothetical person living near the withdrawal well such as community size, water use, diet, etc. The groundwater pumping withdrawal rate specified in Eqn. 3 should be consistent with lifestyle and characteristics used to calculate the BDCFs.

III.E. System Model Implementation

The simplified PA model outlined in Sections III.A through III.D has been implemented mathematically using two different solutions.

First, the model was implemented in Microsoft Excel using an analytic solution[16] to the advective-dispersive equation (Eqn. 1). The Excel-based solution was limited to a constant source concentration, deterministic time-invariant properties in a porous medium, and bounding estimates of radionuclide ingrowth. It was applied to a preliminary hypothetical deep borehole disposal PA[13] and a generic clay repository PA[17].

Second, the model was implemented using the GoldSim Simulation Software Contaminant Transport Module[18]. GoldSim provides a numerical solution to the advective-dispersive equation (Eqn. 1) and also inherently calculates radionuclide ingrowth and the dilution factors discussed in Sections III.A through III.D. In addition the GoldSim implementation allows for a time-depleting source, time-varying conditions, both porous and fractured media, and a probabilistic solution.

IV. SIMPLIFIED PA MODEL RESULTS

The simplified generic PA model described in Section III was applied to two specific disposal concepts and settings for which published PA results are available: a clay repository in France[7] and a hypothetical deep borehole disposal concept in granite[12, 13]. As described in the following subsections, results from the simplified PA model, implemented in GoldSim with high-level input parameters representative of the more detailed parameters used in the specific published PAs, were able to approximate the published PA model results.

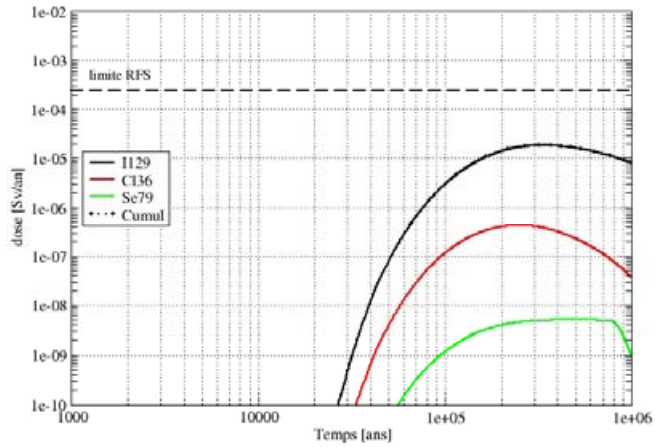
IV.A. Simulation of a Clay Repository

ANDRA (France) has published a PA for a mined repository concept located in a clay/argillite formation[7]. The following key features and processes of this repository concept were implemented in the simplified PA model using GoldSim:

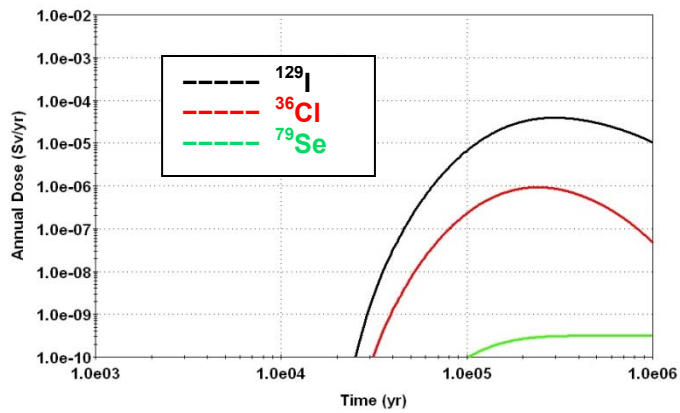
- Initial radionuclide inventory representative of UNF (UOx spent fuel in 13,500 waste packages in the ANDRA CU1 waste package group) (Ref. 7, Table 5.3-5).
- Waste form matrix fractional degradation rate of $2x10^{-5}$ yr^{-1}, corresponding to gradual releases over 50,000 yrs (Ref. 7, p. 222).
- Waste package failure time of 10,000 yrs (Ref. 7, p. 222).
- Dissolved concentrations in the waste package and EBS limited by thermal-chemical conditions (radionuclide solubilities from Ref. 7, Table 5.3-14).
- Diffusion-dominated EBS transport through a 5 m zone of bentonite and excavation-disturbed host rock having a hydraulic conductivity of $1x10^{-11}$ m/s, a porosity of 0.05, and radionuclide specific values for diffusion coefficients and retardation factors (see Ref. 7, Table 5.3-14).
- Diffusion-dominated host rock transport through a 60 m vertical section of Callovo-Oxfordian argillite having a hydraulic gradient of 0.4 m/m, a hydraulic conductivity of $5x10^{-14}$ m/s, a porosity of 0.05, and radionuclide specific values for diffusion coefficients and retardation factors (see Ref. 7, Table 5.3-16).
- Release to the Saulx Valley biosphere outlet via a groundwater pumping well located in the permeable formation overlying the Callovo-Oxfordian argillite (Ref. 7, Section 5.3.2.5) based on a pumping rate of 100 L/min (Ref. 7, Table 5.3-18). Corresponding doses are calculated using BDCFs representative of a farming community (Ref. 7, Table 5.4-1).

To demonstrate the utility of the simplified PA model to simulate the ANDRA clay repository concept, GoldSim simplified model results are compared to published ANDRA results for the case of UOx spent fuel in a CU1 waste package.

Annual doses at the Saulx Valley outlet over one million years are compared in Fig. 2. The ANDRA results in Fig 2a (reproduced from Ref. 7, Figure 5.5-18) indicate that the estimated annual dose is controlled by three non-sorbing radionuclides: primarily ^{129}I, with lesser contributions from ^{36}Cl, and ^{79}Se. Fig. 2b shows the same annual dose results calculated using the simplified PA model.



(a) ANDRA (from Ref. 7, Figure 5.5-18)

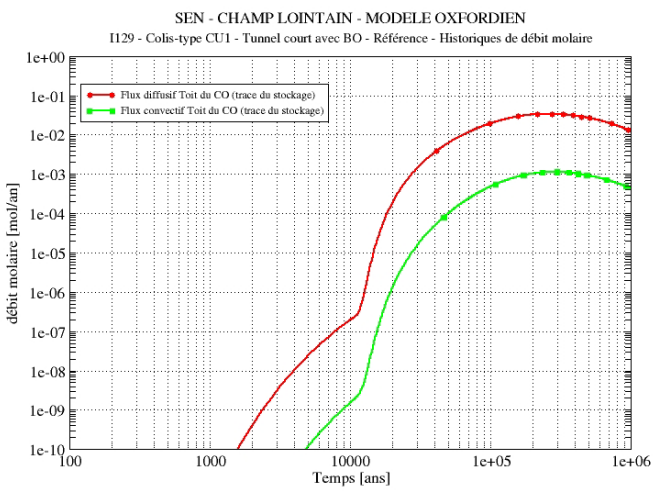


(b) Simplified PA

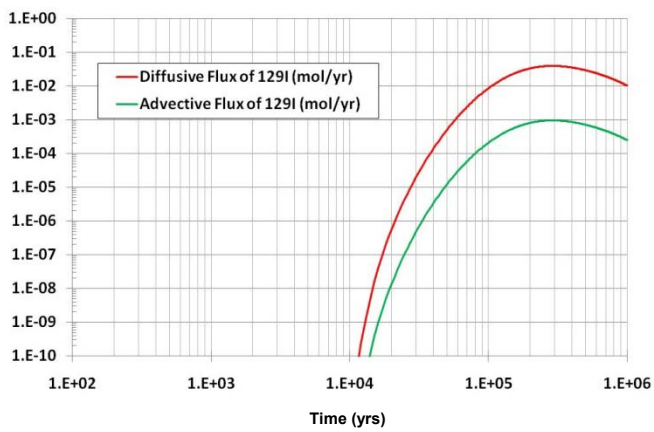
Figure 2. Comparison of annual dose at the Saulx outlet estimated with the ANDRA clay PA model and the simplified PA model

The simplified PA model dose results compare very well with the published ANDRA dose results for those three radionuclides. For ^{129}I and ^{36}Cl, the results are within approximately a factor of two. These differences are attributable to the use of multiple release pathways and BDCFs in the ANDRA biosphere model which cannot be completely captured using the single release pathway in simplified PA model. For ^{79}Se, the dose results from the simplified PA model are between one and two orders of magnitude lower than the ANDRA dose results. The simplified PA model dose results for ^{79}Se are consistent with a ^{79}Se solubility limit that is an order of magnitude lower than reported in Ref 7, Table 5.3-14).

Results from an intermediate performance measure, ^{129}I mass flux (both diffusive and advective) from the Callovo-Oxfordian host rock to the overlying permeable formation are compared in Fig. 3. This intermediate performance measure eliminates complications resulting from the ANDRA biosphere model.



(a) ANDRA (from Ref. 7, Figure 5.5-2)



(b) Simplified PA

Figure 3. Comparison of ^{129}I mass flux from the Callovo-Oxfordian host rock estimated with the ANDRA clay PA model and the simplified PA model

The ANDRA results in Fig 3a (reproduced from Ref. 7, Figure 5.5-2) indicate that the diffusive flux of ^{129}I out of the Callovo-Oxfordian is more than an order of magnitude larger than the advective flux, consistent with diffusion-dominated transport. The flux prior to 10,000 years is from a single defective waste package that fails at 200 years. The remainder of the waste packages fail at 10,000 years, leading to the significant increase in flux after 10,000 years. The ^{129}I fluxes calculated using the simplified PA model show similar behavior to the ANDRA results after 10,000 years, particularly in the magnitude and timing of the peak flux at about 260,000 years. The slightly larger fluxes in the ANDRA model prior to 260,000 years are due to two factors that are not modeled in the simplified PA model: (1) the releases from the early-failed defective waste package prior to 10,000 years, and (2) the early release of a fraction of the radionuclide mass from the waste package metal components and the gap and grain boundaries of the waste forms.

IV.B. Simulation of Deep Borehole Disposal in Granite

An alternative concept for the long-term disposal of UNF and HLW is deep borehole disposal in granite[12]. Although the deep borehole disposal concept has some significant differences from mined geological disposal concepts such as the clay repository modeled in Section IV.A, it can still be represented with the same simplified PA model components outlined in Section III. The following key features and processes of a hypothetical deep borehole disposal concept[12] were implemented in the simplified PA model using GoldSim:

- A source region containing 400 pressurized water reactor (PWR) fuel assemblies (~150 metric tons) vertically stacked down the lower 2 km of a single 5 km deep borehole. Source concentrations are limited by thermal-chemical conditions in a deep granite formation (Ref. 12, Table 4 and Appendix A).
- Thermally-driven vertically upward advection for 200 years through a 1000 m EBS region consisting of a bentonite-sealed borehole and a surrounding fractured rock annulus (Ref. 12, Section 3.2.2).
- Release to the biosphere through a groundwater pumping well located in an aquifer adjacent to the top of the 1000 m sealed borehole segment (Ref. 12, Section 3.2.2 and Figure 11).

GoldSim simplified model results for the hypothetical deep borehole are described in detail in Ref. 13. The simplified model was able to adequately represent the key processes and provide a reasonable estimate of annual dose[13].

V. CONCLUSIONS

The capability of the simplified PA model to simulate clay and deep borehole disposal concepts demonstrates its utility to represent a variety of potential long-term radioactive waste disposal system alternatives.

The simplified PA model is not intended to provide a complex licensing-based solution for a specific disposal system design and site. Rather, it provides a conceptual framework of common disposal system components that can be easily adapted to quickly perform simplified scoping-level PAs for a wide range of disposal system alternatives. The common framework allows for direct comparisons to be made between specific aspects of various disposal alternatives. Existing PA models, which are typically "hard-wired" to provide a complex evaluation of single disposal concept and site, do not have the flexibility to evaluate and compare multiple disposal system alternatives.

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