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An Assessment of the Impact of Advanced Nuclear Fuel Cycles on Geological Disposal

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1. Introduction

After a strong growth in the 1970s and 1980s, the deployment of nuclear energy has stagnated. Only a few nuclear power plants became operational during the last 20 years. More recently, rising oil prices, increasing energy demands and the need for carbon-free energy to limit global warming have resulted in a revival of interests in nuclear energy. New nuclear power plants are now under construction in, e.g., Finland, France, Japan, Korea, the Russian Federation, China and India. Especially in these last 2 countries a considerable increase of the contribution of nuclear energy to the electricity generation is expected (Nuclear Energy Agency [NEA], 2008). The nuclear renaissance made that advanced nuclear fuel cycles are being studied worldwide. They aim at making more efficient use of the available resources, reducing the risk of proliferation of nuclear weapons, and facilitating the management of the radioactive waste. However, the Fukushima accident in 2011 strongly tempered the revival of nuclear energy in several countries.

During the last 20 years the introduction of partitioning and transmutation (P&T) techniques in future nuclear fuel cycles has often been presented as a promising option that might strongly facilitate the long-term management of high-level radioactive waste, because transmutation of the actinides, which are present in spent nuclear fuel, in fast neutron reactors can strongly reduce the long-term radiotoxicity of the radioactive waste. On the other hand, the relevance of this reduction of radiotoxicity for radioactive waste management is questioned by the geological disposal community.

The impact of advanced fuel cycles on radioactive waste management and geological disposal has been studied in some national and international projects between 2003 and 2010. Highly relevant projects were the Expert Group on the Impact of P&T on Radioactive Waste Management of the Nuclear Energy Agency (NEA, 2006) and the Red-Impact project of the European Commission (von Lenza et al., 2008; Marivoet et al., 2009). In the following sections the impact of advanced fuel cycles on various aspects of radioactive waste management and, more specifically, of geological disposal in a clay formation will be investigated. The presented results were mainly obtained from SCK•CEN's participation in the Red-Impact project (Marivoet et al., 2007; Weetjens & Marivoet, 2008).

2. Considered fuel cycles and repository system

Making an assessment of the impact of advanced fuel cycles on radioactive waste management and geological disposal requires that a number of representative fuel cycle scenarios are identified. As a basis for comparison, various indicators are presented in the following sections by taking disposal in Boom Clay as example to develop the case-study. Results obtained in international projects (NEA, 2006; Marivoet et al., 2009) show that most conclusions that can be drawn from the evaluations of disposal in Boom Clay are largely valid for disposal in other clay formations and to a large extent for disposal in hard rock formations.

2.1 Considered fuel cycles

On the basis of an NEA report on advanced fuel cycles (NEA, 2002), 5 fuel cycle scenarios (3 industrial scenarios, which are based on presently available technology, and 2 innovative scenarios) were selected for detailed analyses in the Red-Impact project:

- fuel cycle A1: the reference fuel cycle, which is the "once through" cycle based on pressurised water reactor (PWR) plants with uranium oxide (UOX) fuel;
- fuel cycle A2: fuel cycle based on PWR plants with uranium oxide fuel, the generated Pu is recycled once as mixed oxide (MOX) fuel;
- fuel cycle A3: fuel cycle based on a sodium cooled fast neutron reactors with MOX fuels, in which Pu is multi-recycled;
- fuel cycle B1: fuel cycle based on a sodium cooled fast neutron reactors with MOX fuels, in which all the actinides are recycled;
- fuel cycle B2: fuel cycle based on PWR plants with uranium oxide fuel, the generated Pu is recycled once as MOX fuel, all the minor actinides and the Pu in the spent MOX fuel are recycled in a fast neutron accelerator driven system (ADS).

Results obtained for a variant of fuel cycle B1 (called B1V), in which it is assumed that the heat generating Cs and Sr are separated from the high-level waste (HLW), are also presented.

For each selected fuel cycle mass flow schemes have been prepared and the corresponding neutronic calculations have been made (von Lensa et al., 2008). The estimated waste inventories strongly depend on the assumed separation efficiency of the considered partitioning processes and the applied waste conditioning methods. It was assumed that in future reprocessing plants the reprocessing losses will be limited to 0.1% for all the actinides. For the volatile fission and activation products it was assumed that 1% of the ^{129}I and ^{36}Cl and 10% of the ^{14}C remain in the high-level waste. Specific and optimised waste matrices for conditioning the high-level waste that will arise from advanced fuel cycles have not yet been developed. Therefore, it was assumed that the conditioning methods for the waste from advanced fuel cycles will be very similar to the ones applied today in existing reprocessing facilities.

The volumes of high-level (HLW) and intermediate-level waste (ILW) that are generated in the considered fuel cycles have been estimated in the Red-Impact project (Cunado, 2007) as follows. For the 2 fuel cycles that are applied today (A1 and A2) realistic estimates of the

generated waste volumes could be made on the basis of data provided by the participating waste agencies. For the future fuel cycles (A3, B1 and B2) the volumes of the HLW were estimated on the basis of the mass of the generated fission products and assumed reprocessing losses, and by assuming that the waste will be conditioned as vitrified HLW with the same waste loading as applied today for cycle A2. The volumes of ILW were estimated on the basis of the information available on the cladding and structural components of the fuel assemblies, whereas for the technological waste from reprocessing it was assumed that future reprocessing plants will generate similar waste volumes as the present ones.

The amount of natural uranium that is needed in each of the considered fuel cycles to produce 1 TWh(e) electricity is given in Table 1 (Marivoet & Weetjens, 2008). This indicator illustrates how the introduction of advanced fuel cycles increases the efficiency of the use of uranium. The recycling of Pu in fuel cycle A2 results in a small increase, of about 10%, of the efficiency of the use of uranium for electricity production. Recycling Pu and introducing an ADS in a nuclear reactor park consisting of PWRs (fuel cycle B2) results in an increase of the generated electricity with about 25%. However, replacing PWRs by fast neutron reactors results in a drastic increase of the amount of electricity produced per kg uranium, because natural or depleted uranium can be used instead of enriched uranium for the production of the reactor fuel. In principle, fully recycling all the actinides in a fast reactor (fuel cycle B1) can result in an increase of the efficiency of the use of natural uranium with a factor 100 or more.

Fuel cycle		A1	A2	A3	B1	B2
Consumption of natural uranium	kg/TWh(e)	20723	18448	986	106	15766

Table 1. Amount of natural uranium needed to produce 1 TWh(e) electricity (Marivoet & Weetjens, 2008)

2.2 Considered repository system

The repository considered for the impact evaluations is the SAFIR 2 repository concept that has been developed by the Belgian radioactive waste management organisation ONDRAF/NIRAS (2002). This reference repository is assumed to be excavated in the Boom Clay formation at the Mol site in NE Belgium. At the considered site the host formation is about 100 m thick, of which an 80 m thick central zone consists of very homogeneous clay (in terms of its flow and transport parameters).

The considered repository will have a central access facility consisting of at least two vertical transport shafts and two horizontal transport galleries. The disposal galleries will be excavated perpendicular upon the transport galleries. The HLW canisters will be placed one after the other or with some spacing between two canisters to respect temperature limitations, along the axis of the disposal galleries. A clayey buffer will be placed between the HLW canisters and the gallery walls. Because the Boom Clay is a plastic clay, a concrete lining is required to avoid convergence of the gallery walls during the operational phase of the repository. Figure 1 shows a scheme of a disposal gallery configuration for uranium oxide spent fuel.
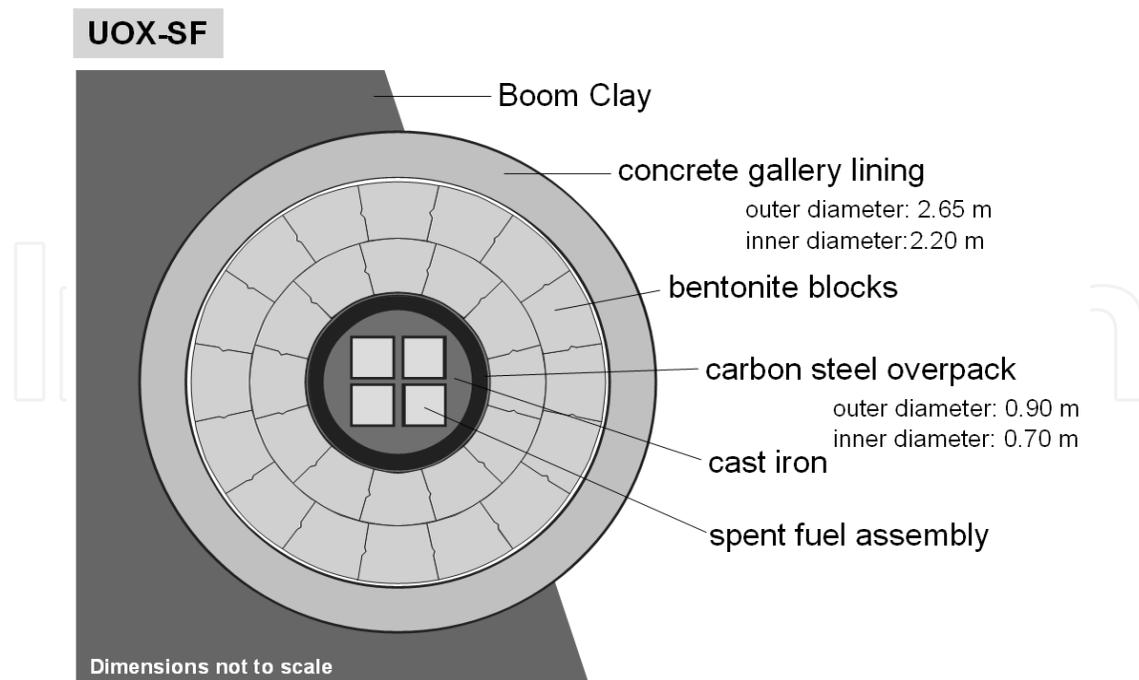
Fig. 1. Gallery configuration for disposal of uranium oxide spent fuel (4 assemblies per container)

3. Assessment of the impact of advanced fuel cycles

3.1 Impact on waste volumes

The total volumes of the waste canisters (these are the primary waste containers) containing the generated HLW and ILW are given in Table 2 (Cunado, 2007). These results show that the volume of the HLW generated in fuel cycle B1 is 5 times smaller than the volume of the spent fuel from fuel cycle A1. However, when the volume of the ILW is also considered, the volume reduction is limited to a factor 2.

Fuel cycle		A1	A2	A3	B1	B2
HLW volume	m³/TWh(e)	2.216	0.806	0.420	0.397	0.464
ILW volume	m³/TWh(e)	-	0.387	0.825	0.825	0.518
Total volume	m³/TWh(e)	2.216	1.193	1.245	1.223	0.982
Relative volume	-	1.000	0.539	0.562	0.552	0.443

Table 2. Volumes of the generated HLW and ILW (packed in canisters) estimated for the considered fuel cycles

The actual volume of HLW is not considered a meaningful indicator for geological disposal. If waste volumes were important, it would be relatively simple to reduce the volume of spent fuel by removing the void between the fuel pins by dismantling the assemblies. However, the dimensions of a geological repository for HLW disposal are mainly imposed by the thermal output of the waste and not by its volume. Therefore, the thermal output and the dimensions of the repository are discussed in the following section.

3.2 Impact on repository dimensions

Figure 2 gives the evolution of the thermal output of the HLW generated in the considered fuel cycles. After a 50 years cooling time the thermal output of the vitrified HLW from cycle A2 is only 57% of the one of spent fuel from cycle A1; however, if the MOX spent fuel from cycle A2 has to be disposed of as well, the reduction of the thermal output is only 3%. In the case of advanced fuel cycle B1 the thermal output of the HLW is reduced with a factor 3 in comparison with the reference cycle A1. It is also interesting to note that the thermal output of the HLW from fuel cycle B1 is 3 times lower after a 100 years cooling time than after a 50 years cooling time; over the same time span the thermal output of the spent fuel from cycle A1 is only reduced with a factor 2.
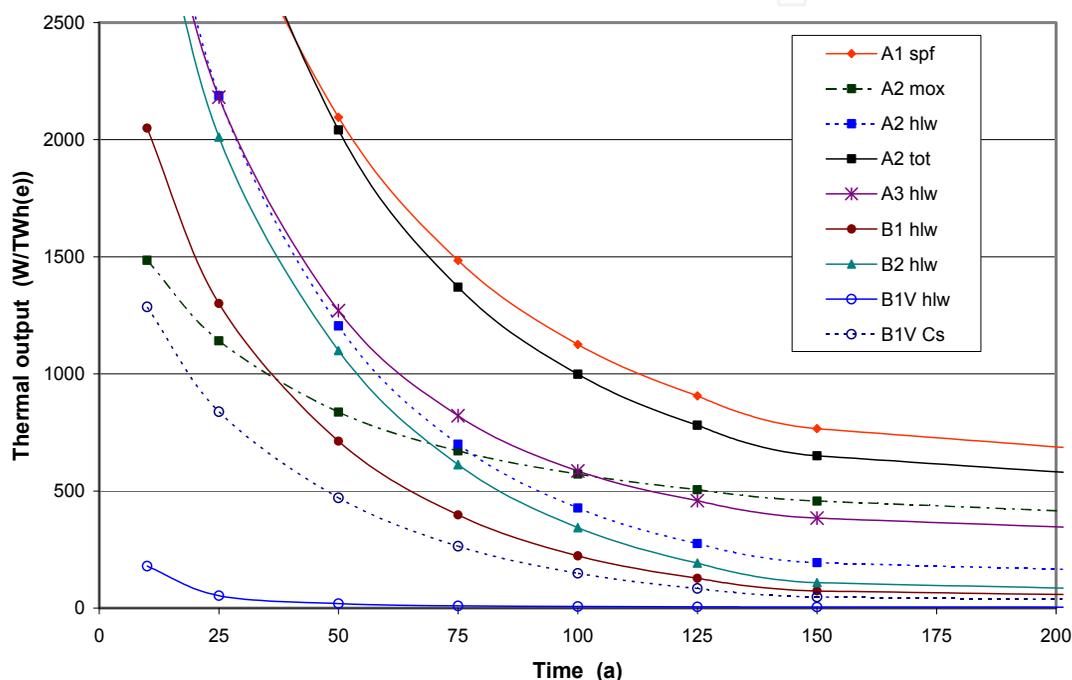

Fig. 2. Evolution of the thermal output of the high-level waste resulting from the considered fuel cycles (cycle B1V refers to cycle B1 with removal of Cs and Sr from the HLW) (Marivoet & Weetjens, 2009)

Removal of the heat generating Cs and Sr from the HLW results in a strong reduction of the thermal output of that waste; e.g., in the case of fuel cycle B1 the thermal output of the HLW is reduced with a factor 40. However, in this case the strongly heat generating Cs and Sr waste has to be stored during at least a century in a cooling facility to have a significant reduction of the thermal output, and eventually the Cs-waste, which contains the very long-lived ^{135}Cs ($T_{1/2}$ = 2.3 Ma), has to be disposed of in a geological repository.

The minimum lengths of the galleries for disposal of HLW are derived from heat dissipation calculations by ensuring that the temperature limitations are respected; for the considered disposal concept the main temperature limitation is that the temperature at the interface between the gallery liner and the host clay formation should remain below 100 °C. An overview of the estimated lengths of the HLW disposal galleries for a 50 years cooling time is given in Table 3.

Fuel cycle		A1	A2	A3	B1	B1V	B2
Allowable thermal loading (50 a)	W/m	353	332-376	365	379	-	379
HLW gallery length	m/TWh(e)	5.92	5.74	3.48	1.88	0.44	2.89
Relative HLW gallery length	-	1.00	0.97	0.59	0.32	0.074	0.49

Table 3. Allowable thermal loading per metre gallery and estimated minimum lengths of the HLW disposal galleries for a 50 years cooling time (cycle B1V refers to a variant of cycle B1 with removal of Cs and Sr from the HLW; a 100 years cooling time is assumed for the Cs-waste) (Marivoet & Weetjens, 2009)

For HLW containing only traces of actinides, which is the case for fuel cycles B1 and B2, the maximum allowable linear thermal loading at disposal time of the disposal gallery is 379 W/m; in the case of the MOX spent fuel from cycle A2, which contains a considerable fraction of actinides, this maximum thermal loading is reduced to 332 W/m. Because of the relatively small difference (about 12%) between the 2 extreme values of the allowable thermal loading, the minimum lengths of the disposal galleries are about proportional to the thermal output of the waste at disposal time. Multi-recycling of the Pu in fast reactors (cycle A3) results in a reduction of the needed length of the disposal galleries of about 40%, a PWR reactor park combined with an ADS for burning the minor actinides and the multi-recycled Pu (cycle B2) gives a reduction of about 50%, and full recycling of all actinides in fast reactors gives a reduction of about 68%. By assuming that the Cs-waste is cooled during 100 years, the removal of Cs and Sr from the HLW gives a reduction of the needed gallery length of more than 90% in comparison with the once through cycle A1.

3.3 Evolution of radiotoxicity in the waste

Radiotoxicity is a measure of how nocuous a radionuclide is to health. The type and energy of rays, absorption in the organism, residence time in the body, etc. influence the degree of radiotoxicity of a radionuclide. In the context of radioactive waste management radiotoxicity is defined as the sum over all radionuclides present in the waste of the products of the activity of a radionuclide with its dose coefficient for ingestion; values of the latter are published by the International Commission on Radiological Protection (ICRP, 1996).

The introduction of P&T techniques in a fuel cycle, i.e. recycling of all actinides in a fast neutron spectrum facility (fast reactor or ADS), results in a drastic reduction of the radiotoxicity in the high-level waste (see Figure 3).

After about 30 years cooling time the minor actinides, mainly Pu, Am and Cm isotopes, are the main contributors to the radiotoxicity. In the case of the reference fuel cycle A1 it takes more than 200 000 years before the radiotoxicity in the spent fuel has dropped below the level present in the natural uranium that was needed to produce that fuel. For fuel cycle A2 only a small reduction of the radiotoxicity can be observed because all the minor actinides and the Pu of the MOX spent fuel are present in the disposed waste. Recycling of all the generated Pu (cycle A3) makes that the radiotoxicity of the natural uranium is reached after about 20 000 years, and when all the actinides are recycled (cycles B1 and B2) it is reached already after about 400 years.

The strong reduction of the time after which radiotoxicity reaches the level of natural uranium is one of the main arguments used to justify the introduction of P&T in future fuel cycles. However, as we will see in the following section, the actinides are very efficiently confined in the geological disposal system and their releases into the biosphere are about negligible, because most actinides are poorly soluble in the chemical conditions prevailing in a geological repository in clay and, furthermore, all actinides are strongly sorbed on the clay minerals (or immobile organic matter) of the host formation. As a consequence, there is no relationship between radiotoxicity and the radiological consequences of a geological repository.
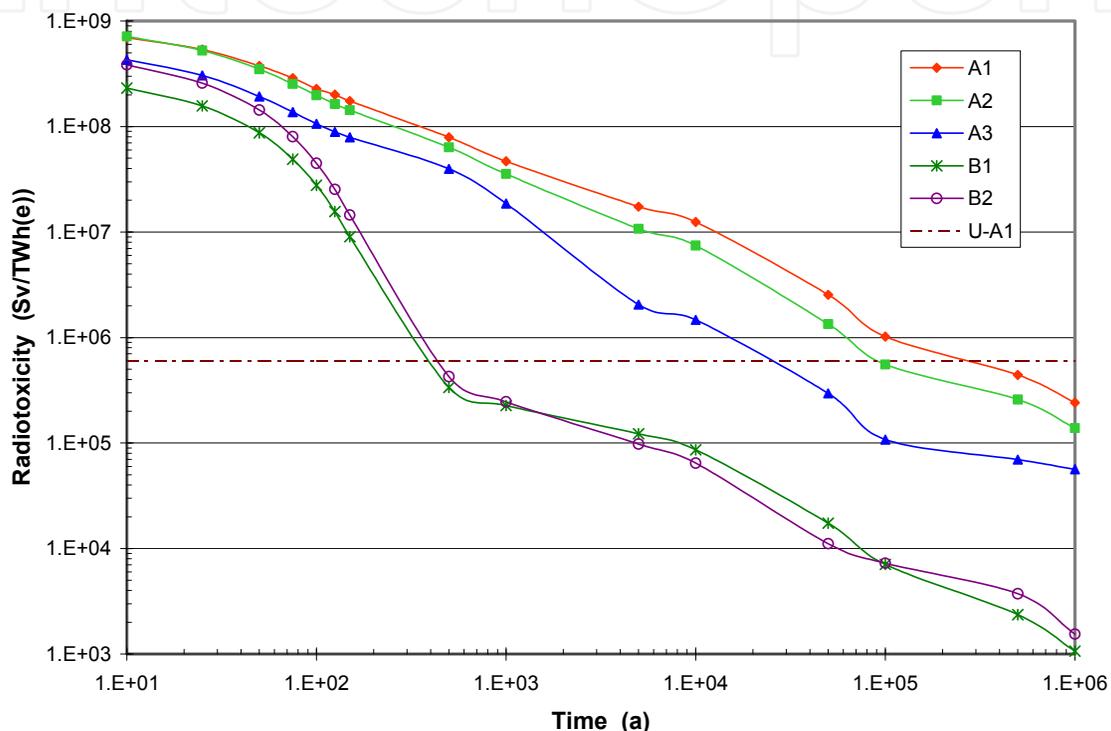
Fig. 3. Evolution of the radiotoxicity present in the high-level waste arising from the 5 considered fuel cycles (the reference line U-A1 gives the radiotoxicity present in the natural uranium needed to generate 1 TWh(e) in cycle A1) (Marivoet & Weetjens, 2009)

3.4 Impact on long-term dose in the case of the reference scenario

The main indicator to evaluate the safety of a geological repository for the disposal of radioactive waste is the effective annual dose to a representative person of the potentially most exposed group in the case of the reference (i.e. expected evolution of the repository system) scenario. The International Commission on Radiological Protection (ICRP, 2000) recommends to use a dose constraint of 0.3 mSv/a, or lower, as reference value for the effective dose. Most national radiological protection authorities are now imposing a dose constraint in the range 0.1 – 0.3 mSv/a.

The reference scenario assumes that the barriers of the disposal system will function as expected. For the considered repository system it was assumed that the overpack will have a lifetime of 2000 years, and that the waste matrices of the vitrified HLW and spent fuels will

dissolve in a 100 000 and 200 000 years period respectively. The solubility limits in the near field are taken into account. The main barrier of the disposal system is the natural barrier provided by the host clay formation; because water flow in the clay is negligible, the radionuclide transport through the clay layer is essentially diffusive; furthermore, many radionuclides are strongly sorbed on the clay minerals. After migration through the clay layer, only a small fraction of the disposed radionuclides reach the aquifers overlying the host formation and eventually the biosphere via a small river draining the contaminated part of the aquifer.

The total dose (normalised per produced electricity) and its main contributors calculated for the disposal of spent fuel from fuel cycle A1 in case of the reference scenario are shown in Figure 4. In case of direct disposal of spent fuel the main contributor to the total dose is [129]I. Other important contributors to the dose are the long-lived fission products [79]Se, ^{99}Tc and [126]Sn and the activation product [36]Cl. Actinides give contributions to the total dose after 2 million years; the maximum actinide dose is more than 2 orders of magnitude lower than the maximum [129]I dose.
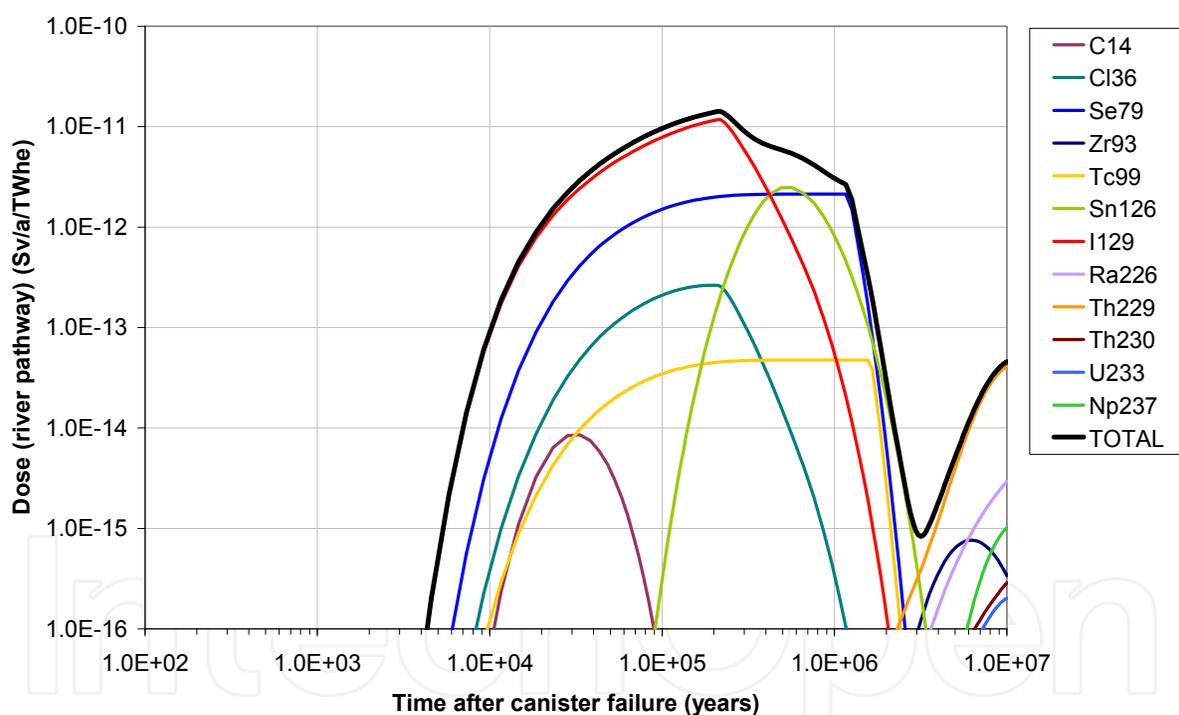

Fig. 4. Evolution of the total effective dose via the river pathway due to the disposal of spent fuel arising from fuel cycle A1 in the case of the reference scenario

Figure 5 gives the doses calculated for the disposal of the high-level and intermediate-level waste (ILW) in case of the reference scenario for the 5 considered fuel cycles. The calculated total doses (estimated by multiplying the normalised doses with the electricity generated within the existing national nuclear programme; the present Belgian nuclear programme is expected to generate about 1700 TWh(e)) are a few orders of magnitude below the dose constraint.

The impact of the transmutation of the actinides in the advanced fuel cycles on the maximum doses is negligible, because the maximum dose is essentially due to mobile long-lived fission and activation products. The amount of generated fission products is about proportional to the heat produced by nuclear fission in the reactors. The difference in the calculated maximum doses is mainly explained by the amount of ^{129}I that is present in the disposed waste. In the case of spent fuel disposal (cycle A1, and partially in the case of cycle A2 in which the MOX spent fuel is assumed to be directly disposed of), all the iodine is going into the repository, whereas in the case of reprocessing only a small fraction of the iodine, which is estimated to be about 2% (1% in the HLW and 1% in the ILW), remains in the waste; the bulk of the iodine escapes as gas from the HLW stream in the dissolver at the reprocessing plant. On the very long-term, i.e. after a few million years for disposal in clay, calculated doses are lower in the case of advanced fuel cycles, because smaller amounts of actinides are present in the HLW arising from these fuel cycles.
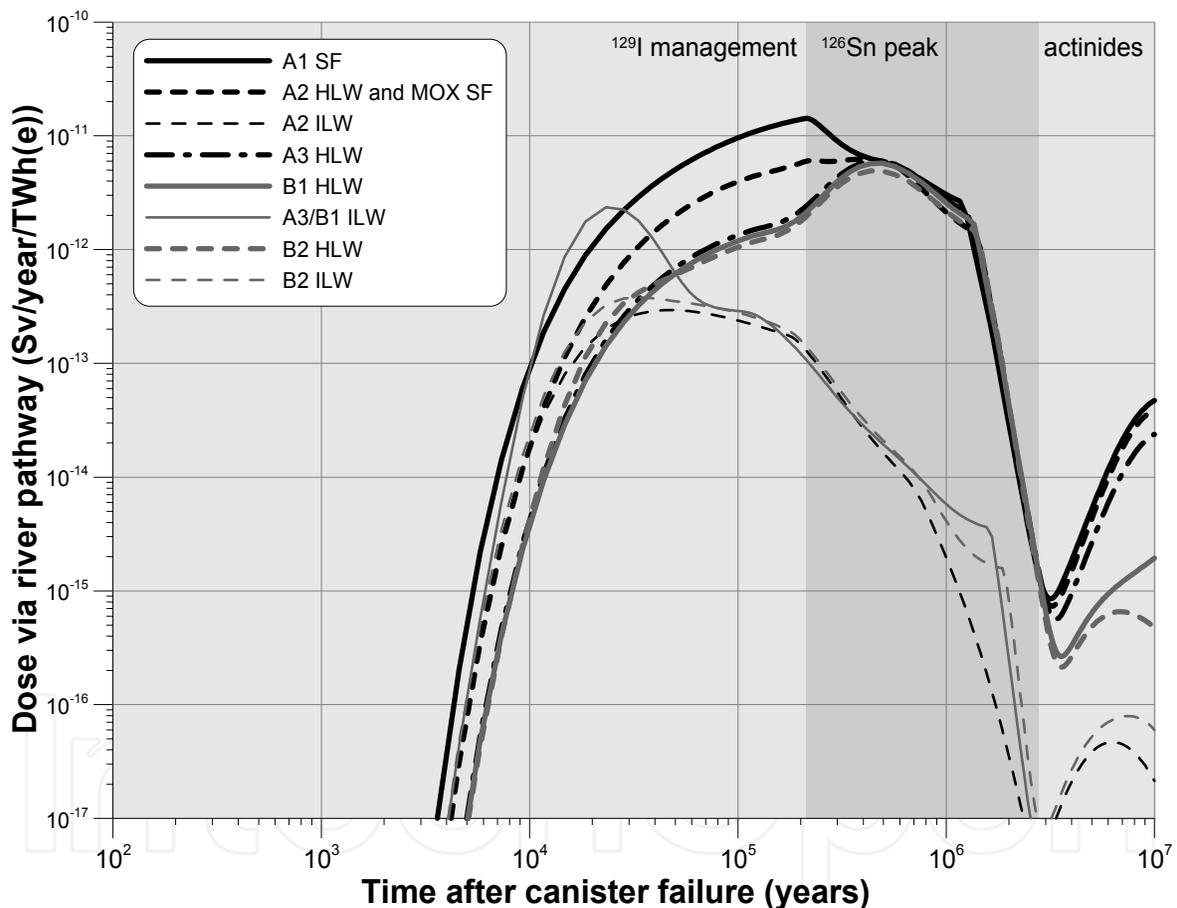

Fig. 5. Evolution of the total effective dose due to the disposal of the high-level and intermediate-level waste arising from the 5 considered fuel cycles (Marivoet & Weetjens, 2008)

Figure 5 also gives the doses due to the disposal of the ILW arising from the considered fuel cycles. For fuel cycle A2 the maximum ILW dose is more than one order of magnitude lower than the maximum HLW dose. However, the ILW dose due to cycles A3 and B1, which are based on fast reactors, shows a relatively high initial dose. This dose is due to the considerable amounts of ^{14}C which are calculated to be generated in fast reactors.

3.5 Impact on long-term dose in other cases and altered evolution scenarios

In the safety assessments made for the SAFIR 2 report and for the upcoming Belgian safety case a number of altered evolution scenarios have been analysed as well. Also for those scenarios, the total dose is mainly due to mobile fission and activation products. However, actinides can play a more important role in safety assessments of disposal in other host formations, such as hard rock (granite) and volcanic rocks (tuff or basalt).

In the case of Yucca Mountain in the United States (Department of Energy [DoE], 2008), the spent fuel is assumed to be disposed above the groundwater table in a mountain consisting of welded tuff located in an arid area. A very low amount of water is flowing through the repository. The oxidising chemical conditions in the disposal galleries result in a relatively high solubility of some actinides. As a consequence the total dose calculated for Yucca Mountain is due to mobile fission and activation products and to actinides.

Disposal in hard rock formations is studied in, e.g., Sweden and Finland and safety cases have been submitted to the radiological protection authorities by the respective waste management organisations (SKB, 2006; POSIVA, 2010). In these cases the use of a very long-lived copper canister makes that no releases of radionuclides are expected during the first hundreds of thousands of years. In the case of the reference scenario in which the failure of some canisters is assumed, the dose is mainly due to [129]I and to a lesser extent to [135]Cs and [14]C. However, for several variant scenarios, such as penetration of oxidising glacial melt water in the near field or a degradation of the bentonite buffer, [226]Ra, [231]Pa and [239]Pu can become the dominant dose contributors.

3.6 Impact on the dose in the case of a human intrusion

A few very low probability disruptive scenarios can have the potential to bring man directly in contact with the disposed waste. One of the most drastic disruptive scenarios that might be considered is a human intrusion scenario in which it is assumed that a borehole is drilled across the repository and that cores containing fragments of the disposed high-level waste are taken from that borehole and examined by a geotechnical worker, who is not aware of the presence of radioactive material (Smith et al., 1987). The dose to a geotechnical worker and its contributors calculated for an intrusion into the repository containing spent fuel from fuel cycle A1 are shown in Figure 6.

Up to 100 000 years minor actinides, successively [241]Am, [240]Pu and [239]Pu, are the main contributors to the dose to a geotechnical worker; at the long term, i.e. after 100 000 years, the main contributors to the dose are [222]Rn and [229]Th.

The doses to a geotechnical worker calculated for an intrusion into the repository containing the HLW arising from the considered fuel cycles are shown in Figure 7. This figure also gives 3 reference lines, which correspond to the ICRP intervention levels of 10 and 100 mSv and to the dose calculated for a geotechnical worker who is handling and analysing cores that are assumed to be taken from the uranium mine at Cigar Lake (Canada).

Figure 7 shows that only for 3 HLW types, arising from the advanced fuel cycles B1 and B2, the dose to a geotechnical worker drops within 1 million years under the dose associated with an intrusion into a Cigar Lake uranium ore body. For the other high-level waste and spent fuel types the doses to a geotechnical worker remain (much) higher than the dose

associated with an intrusion into the Cigar Lake ore body. Consideration of the ICRP intervention levels of 10 and 100 mSv (ICRP, 2000) can be used to contextualise the dose values calculated for the various spent fuel and vitrified HLW types. Applying a simple criterion of ranking when the calculated dose for each HLW type is bounded by the ICRP intervention levels allows a "ranking" of the high-level waste types.
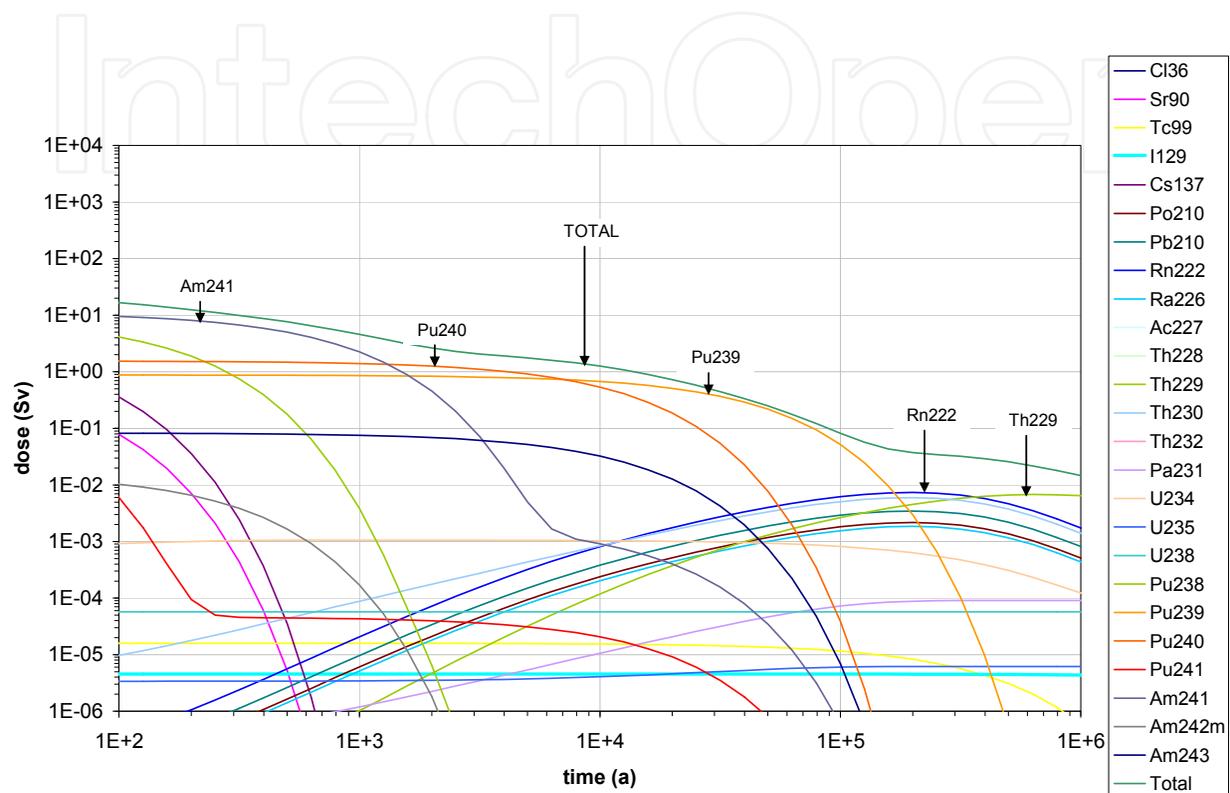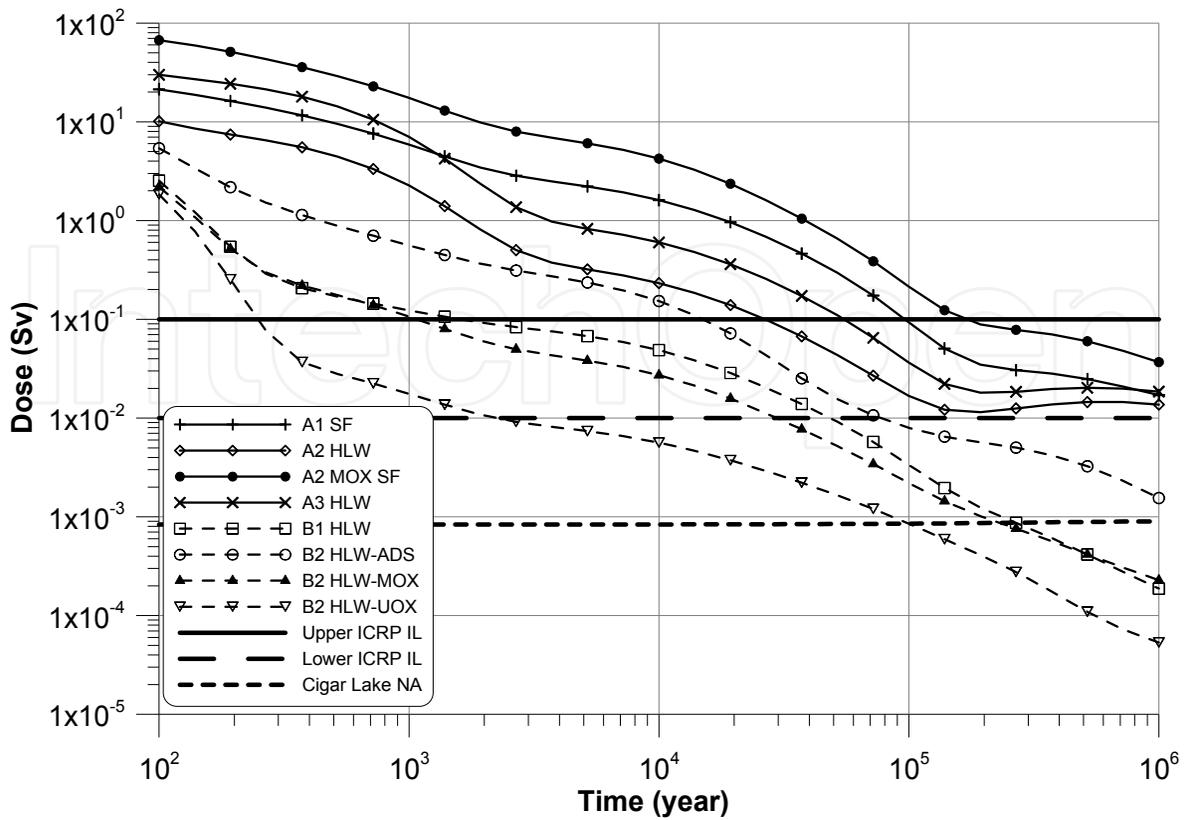

Fig. 6. Evolution of the dose to a geotechnical worker and its contributors calculated for an intrusion into a repository containing spent fuel from fuel cycle A1

Table 4 gives the estimated "required isolation times" for the main spent fuel or vitrified HLW types arising from each of the 5 considered fuel cycle scenarios. In order to verify whether the radiotoxicity present in the disposed waste can be used as an alternative indicator, Table 4 also gives the time after which the radiotoxicity in the disposed waste drops under the radiotoxicity in fresh uranium required for the fuel production in fuel cycle A1 (see Fig. 7). Comparison of the times obtained in Table 4 from the 100 mSv intervention level with the times derived from the radiotoxicity shows that these times are often of the same order of magnitude.

In the case of fuel cycles generating different HLW types, one or more waste types can occur in which a lot of radioactivity is concentrated, but of which only a very small number of waste packages are generated; this is the case for the MOX spent fuel in fuel cycle A2 and the vitrified HLW arising from the pyro-reprocessing of ADS spent fuel in fuel cycle B2. The results given in Table 4 show that the "required isolation times" obtained for the 100 mSv intervention level for the waste types of which the largest quantities are generated correspond much better with the times estimated on the basis of the radiotoxicity arising from the considered fuel cycle.

(IL: intervention level; NA: natural analogue)

Fig. 7. Evolution of the dose to a geotechnical worker for the 5 considered fuel cycles (Marivoet & Weetjens, 2009)

Fuel cycle	Comparator / Waste type	ICRP 10 mSv intervention level	ICRP 100 mSv intervention level	Radiotoxicity
Fuel cycle B1	HLW	~ 40 000 a	~ 1000 a	~ 300 a
Fuel cycle B2	HLW-UOX	~ 2000 a	~ 250 a	~ 300 a
	HLW-ADS	~ 70 000 a	~ 13 000 a	
Fuel cycle A3	HLW	> 1 Ma	~ 70 000 a	~ 24 000 a
Fuel cycle A2	HLW	~ 100 000 a	~ 20 000 a	~ 90 000 a
	MOX SF	> 1 Ma	~ 200 000 a	
Fuel cycle A1	UOX SF	> 1 Ma	~ 100 000 a	~ 200 000 a

Table 4. Ranking of high-level waste and spent fuel types, on basis of calculated dose from geotechnical worker scenario in comparison with ICRP intervention levels; for comparison the time after which the radiotoxicity in the disposed waste drops under the radiotoxicity present in the fresh uranium required for the fuel production of fuel cycle A1 is also given (Marivoet et al., 2009)

From a geological disposal point-of-view, it should be noted that the relevance of a scenario such as inadvertent human intrusion in an assessment of the post-closure consequences of a geological repository is contentious (National Academy of Sciences [NAS], 1995). For example, the considered geotechnical worker scenario assumes that, in spite of the fact that the drill-bit has to perforate a thick-wall metallic container, the geotechnical workers is not aware of the presence of very exotic materials in the underground and will not take precautions when examining the extracted cores. Recently, representatives of various radiological protection authorities discussed the treatment of human intrusion at an international workshop (NEA, 2010). They concluded that "*the results of human intrusion scenarios could also be used in a safety case to demonstrate the robustness of the repository system and the safety concept. The consequences to the intruder who comes into direct contact with the waste should not be required to meet regulatory protection goals. These consequences may need to be calculated but they should neither be evaluated against quantitative limit values nor be used as a crucial criterion for the repository optimization process.*"

3.7 Other waste management aspects

3.7.1 Presentation of fast spectrum reactors as minor actinide burners

Fast reactors or ADSs will not burn all the minor actinides in one single cycle. Only a fraction will be "burned" and multi recycling of the actinides will be necessary to reach a significant reduction. Each cycle consists of cooling of the irradiated fuel, reprocessing, fuel fabrication and irradiation in a fast neutron facility. Dufek et al. (2006) estimated that it will take about 100 years to reach a substantial decrease of the actinides.

Present PWRs are capable of achieving burn-ups up to 60 GWd/tHM. Thereafter, build-up of fission products poisons the chain reaction and the reactor must be refuelled. Much higher burn-ups, in the range of 130 to 200 GWd/tHM, are considered for fast reactors and ADSs. The spent fuels that will arise from those high burn-ups will be much "hotter" (thermal output, gamma and neutron radiation, etc.) than the spent fuels presently considered for geological disposal. As a consequence, those fuels cannot be disposed of in the repository concepts that have been developed for disposal of spent fuel. In the advanced fuel cycles it is (tacitly) assumed that the spent fuels will be infinitely recycled. It has not yet been studied how such fuel cycles can be efficiently ended in the case of a nuclear phase out.

The long duration of the P&T process spread over several generations makes it a difficult issue for decision makers (Commission Nationale d'Evaluation [CNE], 2006; LaTourette et al., 2010). On such long timescales it cannot be excluded that alternatives for nuclear fission will become available such as nuclear fusion or alternative energy resources.

3.7.2 Management of volatile fission products

In the reprocessing plants at La Hague (France) and Sellafield (UK) the iodine released from the spent fuel is discharged, together with other volatile fission products, into the sea. On the other hand, the reprocessing plant at Rokkasho (Japan) captures the iodine on Ag-filters. However, a possible final waste matrix, which remains stable during several hundreds of thousands of years in disposal conditions, has not yet been developed. Continuing the discharge of iodine into the sea cannot be considered as a sustainable solution.

3.7.3 Generation of activation products

The inventory of the activation products was calculated within the Red-Impact project (Cunado, 2007), whereas most other P&T projects focus on actinide and fission product inventories. The generation of huge amounts of ^{14}C were calculated for the advanced fuel cycles. The large amounts of stainless steel used in the considered fast reactor design (i.e. European Fast Reactor) led to the generation of a considerable ^{14}C inventory. Nitride fuel was considered as the reference fuel in the ADS fuel cycle (B2); however, because of the large amounts of ^{14}C that were calculated to be generated, it was decided to also consider an oxide fuel as alternative fuel.

4. Conclusion

The introduction of advanced fuel cycles, based on fast neutron reactors, for electricity production can increase the efficiency of the use of natural uranium with more than a factor 100, because these reactors can burn all the uranium isotopes and not only the ^{235}U as it is the case in a PWR.

The volume of the high-level waste arising from advanced fuel cycles is about 5 times smaller than the volume of the spent fuel from the reference once-through fuel cycle. However, the dimensions of a geological repository are mainly imposed by the thermal output of the waste. Therefore, it is more important that the high-level waste arising from advanced fuel cycles generates about 3 times less heat than the spent fuel from the once-through fuel cycle. As a consequence, the needed length of the disposal galleries as well as the footprint of the geological repository is significantly smaller in the case of advanced fuel cycles.

The total doses due to geological disposal of spent fuel, vitrified high-level and intermediate-level waste in clay (or granite) formations are dominated by contributions of mobile fission and activation products. Geological repositories, in whose near field reducing chemical conditions prevail, very efficiently confine the actinides, because of their high sorption on clay minerals and low solubility. As a consequence the transmutation of actinides in fast reactors or ADS would have little impact on the doses due to the disposal of the radioactive waste in the case of the expected evolution scenario. On the other hand, in the case of altered evolution scenarios, e.g., in which the chemical conditions in the near field of the repository might become oxidising, some actinides can be dominant dose contributors.

The removal of ^{129}I upon reprocessing has a strong impact on the maximum dose. The development of very stable matrices for the immobilisation of iodine is highly desirable.

The radiotoxicity present in the high-level waste from advanced fuel cycles, after a few centuries, is drastically reduced by the transmutation of the actinides. This would lead to a considerable reduction of the potential hazard in the case of some disruptive scenarios such as an inadvertent human intrusion. However, the use of this low probability scenario as a decision making tool for long-term radioactive waste management is highly debatable.

Considerable amounts of mobile activation products, such as ^{14}C, are calculated to be generated in fast neutron facilities and, consequently, to give a significant contribution to the total dose; this observation confirms the importance of developing low activation fuel matrices and construction materials for future fast neutron facilities.

5. Acknowledgment

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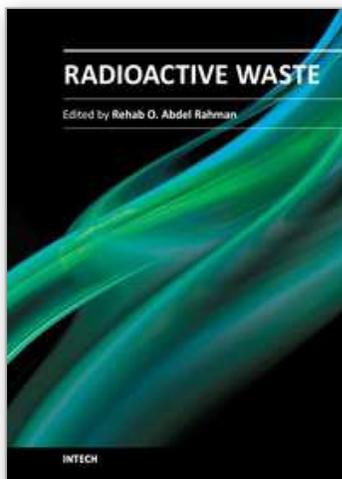
Radioactive Waste

Edited by Dr. Rehab Abdel Rahman

The safe management of nuclear and radioactive wastes is a subject that has recently received considerable recognition due to the huge volume of accumulative wastes and the increased public awareness of the hazards of these wastes. This book aims to cover the practice and research efforts that are currently conducted to deal with the technical difficulties in different radioactive waste management activities and to introduce to the non-technical factors that can affect the management practice. The collective contribution of esteem international experts has covered the science and technology of different management activities. The authors have introduced to the management system, illustrate how old management practices and radioactive accident can affect the environment and summarize the knowledge gained from current management practice and results of research efforts for using some innovative technologies in both pre-disposal and disposal activities.

How to reference

In order to correctly reference this scholarly work, feel free to copy and paste the following:

Jan Marivoet and Eef Weetjens (2012). An Assessment of the Impact of Advanced Nuclear Fuel Cycles on Geological Disposal, Radioactive Waste, Dr. Rehab Abdel Rahman (Ed.), ISBN: 978-953-51-0551-0, InTech, Available from: http://www.intechopen.com/books/radioactive-waste/an-assessment-of-the-impact-of-advanced-nuclear-fuel-cycles-on-geological-disposal

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