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# Costs and Risks of Management and Disposal of Depleted Uranium from the National Enrichment Facility Proposed to be Built in Lea County New Mexico by LES

by

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November 24, 2004

**Version for Public Release Redacted March 20, 2007**

## Preface

This report was prepared for Nuclear Information and Resource Service / Public Citizen, interveners in the LES case. The Nuclear regulatory Commission has imposed a requirement of completing the report in an unreasonably short period of time. The announcement that such a report would be required was made on 20 October 2004 and the deadline for filing is 24 November 2004. This report was prepared in the absence of full disclosure by LES of the state of information and negotiations about the expected costs of DU deconversion. The NRC has failed to provide the basis for its dose calculations of the impacts of DU disposal. Our analysis indicates that some of these dose calculations are technically incredible. They cannot be accepted as a reasonable basis for a licensing proceeding without full disclosure of the methods, assumptions, models, and all details of the actual implementation of the models used to produce them.

In addition, the NRC's Agency-wide Documents Access and Management System (ADAMS) has been unavailable since 25 October 2004 due to an ongoing security review which has prevented us from gaining access to other potentially important information. These factors have hampered the production of this report. IEER reserves the right to update and revise this report when LES makes more complete information available and the NRC publishes the requisite data and methodology for its dose calculations and restores the ADAMS database.

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Note for redacted version: This is a redacted version of the original report on the proposed National Enrichment Facility that was submitted to the NRC on behalf of the interveners, NIRS and Public Citizen. The original version contained proprietary information relating to a DU contract.

# I. Introduction

*"Waste management and disposal operations are an integral part of the practice generating the waste. It is wrong to regard them as a free standing practice that needs its own justification. The waste management and disposal operations should therefore be included in the assessment of the justification of the practice generating the waste."*[1]

*- International Commission on Radiological Protection (1997)*

Uranium enrichment, the process proposed to be carried out in the National Enrichment Facility (NEF) proposed to be built in New Mexico by LES, is the process of increasing the proportion of uranium-235 in uranium. The term "enrichment" refers specifically to increasing the concentration by weight of U-235 in a sample of uranium.

Natural uranium consists of three isotopes, U-238, U-235, and U-234, of which only U-235 is fissile (i.e. can sustain a chain reaction with zero-energy neutrons). Enrichment of the proportion of U-235 from the 0.711 percent in natural uranium to a few percent (3 to 5 percent typically) is required for nuclear fuel used in light water reactors, by far the most common nuclear power reactor in the world (including all operating U.S. power reactors). Feeding natural uranium into an enrichment plant produces two streams of uranium. One is the enrichment stream, which is used for fuel (after further chemical and physical processing). The other is depleted uranium, so-called because it is depleted in U-235.

There are various types of processes that can enrich uranium. The commercial ones require that uranium be put in a chemical form known as uranium hexafluoride, or $UF_6$. $UF_6$ when heated to modest temperatures sublimes into a gas. $UF_6$ gas when passed through a diffusion barrier or a centrifuge of appropriate design can yield the two desired streams – enriched and depleted – that are the final output of an enrichment plant.[2] Thousands of centrifuges are typically required in a commercial scale plant. The production capacity of an enrichment plant is measured in Separative Work Units or SWUs (pronounced "swooze"), which has units of kilograms. The plant that has been proposed to be built would have a capacity of 3 million SWUs per year.

Depleted uranium (DU) is currently classified as a "source material." It can be converted to plutonium-239 in nuclear reactors, and this was, at one time, thought to be the major potential use of DU. However, commercial use of plutonium is, in fact, very limited and the use of DU in this application is negligible compared to the amounts of depleted uranium that have been created over the past sixty years. As stocks of DU have continued to grow, the issue of its management and possible disposal as a waste, if it were to be so classified, has become more important. This study assesses the problems associated with the management and disposal of depleted uranium in the specific context of the proposed National Enrichment Facility (NEF).[3]

[1] ICRP 77 p 13-14

[2] For a description of enrichment technologies and plants, see Makhijani, Chalmers, & Smith 2004.

[3] In the present work we will refer interchangeably to the proposed uranium enrichment plant as the NEF or the LES facility.

# II. Depleted Uranium Characteristics and Risks

## A. Physical and Radiological Characteristics of Depleted Uranium

Depleted uranium (DU) is a byproduct of the enrichment of natural uranium as part of the nuclear fuel cycle as well as the production of nuclear weapons. As noted, DU consists of the same three radionuclides as natural uranium but in different concentrations. Typical concentrations of these isotopes by weight percent are shown in Table 1.

*Table 1: Percent composition by weight of natural and depleted uranium*

| Radionuclide | Natural Uranium | Depleted Uranium |
|---|---|---|
| U-234 | 0.005 | 0.001 |
| U-235 | 0.711 | 0.2 to 0.3 |
| U-238 | 99.284 | 99.7 to 99.8 |

The total amount of depleted uranium to be created by the proposed uranium enrichment facility as well as the specific activity of this material which must be managed and disposed of are among the most important physical quantities in considering health and environmental impacts, as well as costs. For this report, we will use the LES assumption that as much as 133,000 metric tons of depleted uranium (197,000 metric tons of $DUF_6$) would be produced by the proposed enrichment facility.[4]

The specific activity of pure natural uranium metal is about 670 nanocuries per gram. The specific activity of DU can vary, but it is always greater than 340 and less than 670 nanocuries per gram. For the purposes of this report, we will take the specific activity of DU to be 400 nanocuries per gram (metal). This is consistent with the assumption set forth in the 1997 Lawrence Livermore Engineering Analysis for the long-term management of the Department of Energy's depleted uranium stockpile and corresponds to an isotopic composition of: 99.75% U-238, 0.25% U-235, and 0.001% U-234.[5] This assumption yields a specific activity for depleted uranium oxide ($DU_3O_8$) of about 340 nanocuries per gram and a specific activity of 350 nanocuries per gram for depleted uranium dioxide ($DUO_2$). The metabolic behavior of DU in the human body and its mobility in the environment is essentially the same as that of natural uranium. The toxicological properties of depleted uranium are also essentially the same as natural uranium, while its radioactivity is roughly 40% less. Thus, the only significant difference between DU and natural uranium is its lower specific activity. We will discuss the chemical toxicity of uranium further below. Here we will focus on its radiological properties in relation to how the management and disposal of DU should be considered in comparison to other types of radioactive materials.

In terms of its radiological properties, depleted uranium (if it was to be ruled to be a waste by the Commission) would be most comparable to transuranic (TRU) waste which is similar to the

[4] NRC NEF EIS Draft 2004 p. 4-55
[5] LLNL 1997(EA) p. 2-8

classification of Greater than Class C (GTCC) waste under 10 CFR 55(a).[6] Shallow land disposal for these wastes (TRU or GTCC) is generally not appropriate and they are considered to require deep geologic disposal. Table 2 summarizes the relevant radiological properties of the three isotopes composing DU compared to a number of the most important transuranic radionuclides present in TRU wastes.[7]

*Table 2: Radiological properties of U-234, U-238 and Selected Transuranic Radionuclides*

| Isotope | Main decay mode | Alpha particle energy, MeV | Half-life, years | Comments |
|---|---|---|---|---|
| Uranium-238 | Alpha | 4.1 | 4.46 billion | |
| Uranium-235 | Alpha | 4.4 | 700 million | weak gamma emitter |
| Uranium-234 | Alpha | 4.8 | 245,000 | |
| Neptunium-237 | Alpha | 4.8 | 2.14 million | |
| Plutonium-238 | Alpha | 5.5 | 87.7 | |
| Plutonium-239 | Alpha | 5.1 | 24,110 | |
| Plutonium-240 | Alpha | 5.1 | 6,537 | |
| Americium-241 | Alpha | 5.5 | 432 | strong gamma emitter |

Note: All energies rounded to two significant figures. The alpha emitting radionuclides emit alpha particles with more than one characteristic energy, with each energy level being produced with a known probability. The alpha particle energy shown is an approximate average of these particles energies, weighted by the emission probability.

From Table 2 it is clear that in terms of its radiological properties, DU is directly comparable to these important radionuclides composing TRU waste. Adding a further consideration to this comparison, we note that the depleted uranium being considered for ultimate disposal by LES is a pure material, whereas TRU wastes are generally composed of inhomogeneous materials containing trace amounts of transuranic radionuclides, most notably Pu-238, Pu-239, and Pu-240, as well as Am-241 and Np-237. Table 3 compares the typical specific activities of DU in various chemical forms that have been proposed for long-term management with that of TRU / GTCC wastes. In addition, Table 3 also compares the specific activity of depleted uranium to the total activity of natural uranium ore that is typical of mined extraction.

[6] 10 CFR 61 final rule 1982, p.587-589 and EPA 2001

[7] For simplicity in this report we will refer interchangeably to TRU and GTCC waste. The important element of their classification with respect to this discussion is the limit of 100 nCi/gm of long lived alpha emitting transuranic elements.

*Table 3: Specific activities of various chemical forms of depleted uranium, TRU waste, and typical uranium ore with 0.2% natural U by weight.*

| Chemical form | Specific activity, nCi/gm |
|---|---|
| uranium metal (DU) | 400 |
| uranium dioxide ($DUO_2$) | 350 |
| uranium oxide ($DU_3O_8$) | 340 |
| transuranic activity in TRU or GTCC waste (Note 1) | >100 |
| 0.2% uranium ore | 4 (See note 2) |

Note: The specific activity shown for 0.2% uranium ore includes all decay products of uranium-238 up to and including radium-226, assuming they are in secular equilibrium with uranium-238. Radon-222, and its decay products are not included in this number. All values in the table are given in round numbers. There are slight differences between EPA's definition of TRU waste and NRC's definition of GTCC waste as it relates to transuranic radionuclides that are not material to this discussion since DU is clearly comparable to either one.

Tables 2 and 3 highlight the fact that the radionuclides initially present in depleted uranium share many radiological similarities with the important transuranic elements and that they are present in higher activity concentrations than the limit for TRU waste. When considering the long-term impacts of disposing of depleted uranium, however, it becomes important in many cases to also consider the impact of the in growth of uranium daughter products.[8] This was the approach that was adopted by the NRC in their analysis of the environmental impacts associated with deep disposal of the depleted uranium that would have been generated by the proposed CEC enrichment facility.[9]

The specific activity of depleted uranium will continue to grow slowly over time until secular equilibrium is reached after more than a million years. In addition to an increase in the amount of U-234 present in the DU, two of the other important daughter products of U-238 that need to be considered are thorium-230 and radium-226. Th-230, with a half-life of 77,000 years, is an alpha emitter with an average energy of 4.7 MeV. Ra-226 has a half-life of just 1,600 years, and is also an alpha emitting radionuclide with an average energy of 4.8 MeV. These two daughter products are also weak beta and gamma emitters. The inclusion of additional long-lived alpha emitting radionuclides to our considerations of depleted uranium adds further justification to the need for treating DU as analogous to TRU waste. In fact, the risks in terms of mortality per becquerel of intake of DU (including its decay products)[10] are together about four times more dangerous than that of plutonium-239, as can be seen from Table 4.

[8] NAS/NRC 2003 p. 68
[9] NRC CEC EIS final p. A-14 to A-15
[10] Th-234 and Pa-234m are omitted from the present discussion because they contribute relatively little to drinking water and other ingestion doses compared to the radionuclides under discussion here.

*Table 4: Comparison of mortality per Bq and mortality per gm of depleted uranium oxide at secular equilibrium to that of plutonium-239 contained in TRU waste at 100 nCi per gram*

| Radionuclide | Mortality per Bq for Tap Water Intake | Mortality per Bq for Dietary Intake | Ratio of Mortality per Bq versus Pu Tap Water | Ratio of Mortality per Bq versus Pu Dietary | Mortality Ratio adj for sp act per gm Tap water | Mortality Ratio adj for sp act per gm Dietary |
|---|---|---|---|---|---|---|
| | | | | | | |
| Uranium-238 | 1.13E-09 | 1.51E-09 | 0.40 | 0.42 | 1.34 | 1.41 |
| Uranium-234 | 1.24E-09 | 1.66E-09 | 0.44 | 0.46 | 1.48 | 1.55 |
| Thorium-230 | 1.67E-09 | 2.16E-09 | 0.59 | 0.60 | 1.99 | 2.05 |
| Radium-226 | 7.17E-09 | 9.56E-09 | 2.52 | 2.63 | 8.53 | 8.93 |
| **total mortality ratio at secular equilibrium** | | | **3.93** | **4.11** | **13.34** | **13.94** |
| | | | | | | |
| Plutonium-239 | 2.85E-09 | 3.63E-09 | | | | |

Note: The source for the drinking water and dietary mortality ratios is EPA Federal Guidance Report 13.[11] The two right most columns show the ratio of the mortality coefficients for uranium and its daughter products versus plutonium-239 after adjusting by 340/100 to account for the greater specific activity of bulk $DU_3O_8$ relative to that of the transuranic elements at the threshold of TRU waste.

When adjusted for the greater specific activity of $DU_3O_8$ relative to the 100 nanocuries per gram threshold of TRU waste, any one of these four components of depleted uranium exceeds the risk of plutonium-239. Together, DU and its decay products are about an order of magnitude more risky (in terms of cancer mortality per unit of mass consumed) than TRU waste with 100 nanocuries per gram of plutonium-239. Further, uranium and its decay products (with the possible exception of throium-230) have, in general, comparable or greater environmental mobility than plutonium.[12] Hence, from a regulatory point of view as well as from a scientific point of view the risks that would arise from DU disposal cannot be considered as less than those from TRU waste disposal.

The information in Tables 2 through 4 demonstrate that pure depleted uranium cannot be considered analogous to naturally occurring uranium ore and that instead it is, in fact, most directly analogous to transuranic or Greater than Class C waste. The similarity of DU to TRU waste has recently been noted in a National Research Council report, both in regards to their radiological characteristics as well in regards to the difficulties that are associated with their disposal:

> If disposal [of depleted uranium oxide] is necessary, it is not likely to be simple. The alpha activity of DU is 200 to 300 nanocuries per gram. Geological disposal is required for transuranic waste with alpha activity above 100 nanocuries per gram. If uranium were a transuranic element, it would

[11] FGR 13 1997 p. 102-103
[12] ResRad data collection manual p. 110-111

> require disposal in the Waste Isolation Pilot Plant (WIPP) based on its radioactivity. The chemical toxicity of this very large amount of material would certainly become a problem as well. One option suggested by the U.S. Nuclear Regulatory Commission (USNRC) is disposal in a mined cavity or former uranium mine. Challenges for this option would include understanding the fundamental differences between uranium ore (see Sidebar 6.1) and the bulk uranium oxide powder.[13]

The level of effort, time, and expense that LES must consider in relation to the management and ultimate disposal of the depleted uranium it proposes to generate therefore must also be assumed to be comparable to that which was required in the construction, licensing, and operation of WIPP scaled as appropriate for the relative amount of waste envisioned for ultimate disposal. We will discuss the need for adequate deep disposal and the associated issues, including costs, in further detail in Chapter IV.

**B. Health Risks of Exposure to Depleted Uranium – Well-Established and Emerging Risks**

Expect for a few important exceptions, uranium is primarily dangerous to people when it gets inside the body through ingestion, inhalation, or through breaks in the skin. One such exception is the dangers posed to uranium workers while, in the current context, a further exception is the risk posed by the external doses to a person atop an area in which large amounts of depleted uranium oxide had been disposed of in a shallow trench. Inside the body, depleted uranium creates risks both as a toxic heavy metal and as a radioactive material. In addition, there are some indications that there may be synergisms between these two types of DU effects, and further research has recommended.[14]

Current federal safe drinking water regulations limit the concentration of uranium in drinking water to 30 micrograms per liter based primarily on its chemical toxicity.[15] For natural uranium, this limit translates into 20 picocuries per liter (pCi/L) of radioactivity from uranium. For depleted uranium, the drinking water limit translates into 12 pCi/L of uranium activity. Exposure to uranium in water is regulated for chemical toxicity largely because uranium is known to be nephrotoxic (toxic to the kidneys).There remain important uncertainties with regards to the level of sensitivity of human kidney's to depleted uranium highlighted by the fact that animal studies have shown toxic thresholds that differ by more than an order of magnitude between experiments on rabbits (more sensitive) and rats (less sensitive).[16]

The science surrounding uranium's effects on the body is rapidly expanding due in large part to the concerns that have arisen in the wake of the 1991 Gulf War and the 1999 NATO bombing campaign in the former Yugoslavia and the gradual recognition of the myriad health effects that have come to be known as Gulf War Syndrome. We will discuss the emerging picture from this research further below, but as an example with particular relevance to the current drinking water limits, we note that recent experiments in mice have shown uranium effects on the brain with potential neurotoxicological importance at levels of uranium exposure that were not found to cause discernable damage to the kidneys.[17]

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[13] NAS/NRC 2003 p. 67
[14] Miller et al. 2002b p. 277, Royal Society Part I 2001 p. 15, and Royal Society Part II 2002 p. 14
[15] EPA 2004
[16] Royal Society Part II 2002 p. 15
[17] Pellmar et al. 1999 p. 791

In addition to the chemical toxicity of uranium, DU exposure via ingestion or inhalation as well as external exposures from large quantities of depleted uranium also creates a host of risks related to exposure to ionizing radiation. Currently regulated risks relate to a variety of cancers caused by radiation exposure (radiogenic cancers). The worker compensation law of 2000 (EEOICPA) recognizes all cancers except chronic lymphocytic leukemia (CLL) as radiogenic. The connection of CLL to radiation exposure is not ruled out, but CLL is the one cancer that is not currently eligible for compensation under the existing law.

The current best understanding of low-dose radiation effects (see NCRP 2002, for instance) as well as regulatory practice in the United States and Europe is that every increment of radiation exposure produces an incremental increase in the risk of cancer. Understanding the extent of this risk has changed over time and continues to evolve today in relation to such mechanisms as the bystander effect in which cells not directly transversed by an $\alpha$ or$\beta$ particle or a $\gamma$ ray can show possible genetic damage as well as in relation to such effects as radiation induced genomic instability. In general the estimated risks per unit of exposure have increased with time as more is learned about the interaction of radiation with living tissue. For example, in 1954 the AEC first set the first post war radiation limit at 5 rem per year. This was a significant reduction from the 0.1 roentgen *per day* limit that had been adopted in 1942 during the Manhattan Project. In 1959, the dose limit for the public was lowered to 0.5 rem per year and was then lowered again in the late 1980's early 1990's to 0.1 rem per year.[18]

An additional element of radiological protection that has evolved over time is our understanding of the relative risks to women and men. Currently, the overall risk to women of developing a fatal cancer from exposure to low-dose, low-LET radiation is estimated to be nearly 50 percent greater than that for men.[19] Nearly 45 percent of the additional risk to women per unit of exposure is due to the significant radiosensitivity of the female breast.[20] Current computations of risk per unit of exposure are based on the weighted average risk to men and women. The fatal cancer risk per unit of exposure to women is roughly 19 percent greater than this average. [21] If cancer incidence is considered irrespective of fatality, the comparison is grows slightly worse with women having more than a 58 percent greater risk of developing some form of cancer from radiation exposure than men.[22]

Uranium risks – current research indications

The understanding of the risks of cancer due to radiation exposure from depleted uranium and kidney damage due to its heavy metal properties has expanded greatly in recent years. In addition, evidence is amassing that raises serious concerns regarding the impact of chronic exposure to DU in relation to a number of other health issues. Studies in humans and animals have shown that

[18] IEER SDA 1997 p. 9
[19] EPA FGR 13 p. 179. The overall risk per person-Gray of exposure to women from low-dose, low-LET uniform irradiation is estimated to be $6.83 \times 10^{-2}$ while the risk to men is $4.62 \times 10^{-2}$.
[20] EPA FGR 13 p. 179. The female breast has the highest risk per unit of exposure of any individual organ listed for either men or women in this EPA guidance document.
[21] EPA FGR 13 p. 1
[22] EPA FGR 13 p. 179

uranium can concentrate in the skeleton, liver, kidneys, testes, and brain.[23] In addition, rats implanted with DU pellets have also shown uranium concentrating in the heart, lung tissue, ovaries, and lymph nodes among other tissues.[24] Research, primarily but not exclusively conducted since the 1991 Gulf War, indicates that exposure to uranium may be

Mutagenic
Cytotoxic
Tumorigenic
Teratogenic
and Neurotoxic, including in a manner analogous to exposure to lead

Additionally, as noted above, some research has also provided indications that there may be a synergistic effect between the heavy metal aspect of exposure to uranium and its radioactive effects. Research on the hazards of the heavy metal cadmium indicated a potential synergistic response when exposures were combined with gamma radiation. Current research conducted at the Armed Forces Radiobiology Research Institute (AFRBI) indicates that "[i]n the case of DU, cells not traversed by an alpha particle may be vulnerable to radiation-induced effects as well as chemically-induced effects."[25] Addition work at the AFRBI has also shown that depleted uranium can cause oxidative DNA damage and thus provides the first indication that uranium's radiological and chemical affects might potentially play both a tumor initiating and a tumor promoting role.[26]

We will discuss some of these potential aspects of depleted uranium's health effects that are emerging from a wide range of research, and we refer the reader to the publications cited for further information.

Mutagenic / Tumorigenic effects

Depleted uranium is a radioactive material and as we have discussed above, ionizing radiation is an accepted causative risk factor for all but one form of cancer. In addition, uranium is a heavy metal and many heavy metals (such as nickel) are also known to be carcinogenic in the body due to their ability to cause oxidative damage to the DNA. Since the late 1990s there has been a growing body of evidence from in vitro and in vivo studies that indicates that depleted uranium may, in fact, be genotoxic, mutagenic, and tumorigenic.[27] A significant amount of this work is currently being conducted at the Armed Forces Radiobiology Research Institute under the direction of Dr. Alexandra Miller.

Although they were not able to conclusively identify the biochemical mechanism involved, in 1998 Miller *et al.* demonstrated for the first time that internalized depleted uranium could result in "a significant enhancement of urinary mutagenicity," a common biomarker of exposure to a genotoxic agent.[28] That same year, Miller *et al.* demonstrated for the first time that exposure to

[23] WHO 2001 p. 65-66
[24] Arfsten, Still, & Ritchie 2001 p. 182
[25] Miller et al. 2002b, p. 277
[26] Miller et al. 2002c p. 251 and Miller et al. 2004 p. 254
[27] Arfsten, Still & Ritchie 2001 p. 180
[28] Miller et al. 1998 p. 646-647

DU can transform human cells into the tumorgenic phenotype, and that these transformed cells are capable of producing cancerous tumors in immuno-suppressed mice.[29] Building on this work, in 2000 Miller *et al.* again demonstrated the fact that DU could transform human cells into the tumorigenic phenotype. Significantly, their work also demonstrated that "DU can induce chromosomal aberrations that are distinctly characteristic of radiation exposure suggesting that the alpha particle component of DU exposure may play a role in the transformation and genotoxic process."[30]

The relative role of the radiological and chemical components of the genetic damage caused by the depleted uranium is a significant question given that DU is currently regulated with a primary focus on its chemical hazard and the implicit assumption that its radiation hazard can generally be treated as a secondary concern in the environment. In a trio of papers in 2002, Miller *et al.* were able to further clarify the roles of DU's chemical and radiological properties and how they relate to the observed generic damage. In the first paper they reported finding that DU caused a "small but significant increase" in the frequency of dicentric chromosomal aberrations which was not observed in the case of exposure to non-radioactive toxic heavy metals. The formation of this type of defect is known to be correlated with low-dose radiation damage from other experiments. [31] The second paper reported their finding that exposures to equal chemical concentrations of uranium with different isotopic composition caused "a specific activity dependent increase in neoplastic transformation frequency" which further suggested "that radiation can play a role in DU-induced biological effects in vitro."[32] In discussing these results Miller *et al.* recognized the significant uncertainties remaining that surround this work, but they also highlighted some its more important potential consequences. They noted that

> Although the data indicate that radiation is involved in DU effects in vitro, several questions remain unanswered. The extent to which radiation contributes to the effects exerted by DU is not known nor its mechanism(s) understood. Furthermore, one can only speculate as to whether the radiation- and chemical-effects are synergistic. Limited studies have shown that a non-radioactive metal like cadmium combined with gamma radiation can result in a synergistic response in vivo. It is intriguing to ask whether radiation actually play[s] a significant role in DU cellular effects perhaps through nontargeted effects of radiation exposure? Several recent radiation studies have demonstrated the important role that bystander effects have in cellular radiation response by causing damage in unirradiated neighboring cells. In the case of DU, cells not traversed by an alpha particle may be vulnerable to radiation-induced effects as well as chemically-induced effects. [33]

In summary, they concluded that

> Considering that conventional understanding of potential DU health effects assumes that chemical effects are of greatest concern, these results and similar future results could have a significant impact on DU risk assessments. [34]

The final paper from Miller *et al.* in 2002 examined the other end of the problem and found that DU was also capable of inducing "oxidative DNA damage in the absence of significant radioactive

[29] Miller et al. 1998b p. 465, 468-469
[30] Miller et al. 2000 p. 210
[31] Miller et al. 2002a p. 121-122
[32] Miller et al. 2002b p. 275
[33] Miller et al. 2002b p. 277
[34] Miller et al. 2002b p. 277

decay."[35] In light of their other work showing the potential for the radiological ascpect of DU to contribute to genotoxic effects in vitro, they note that "it is tempting to speculate that DU might exhibit both a tumor 'initiation' and 'promotion' component."[36] This potential dual role could result from the alpha particle radiation causing the cancerous mutation (tumor initiation) followed by a build up of oxidative damage aiding the spread of the cancer (tumor promotion).

A final example of the work being conducted at the AFRBI on these issues comes from a 2003 Miller *et al.* publication concerning the potential for DU to induce genomic instability in human cells. In this work the authors initially note that

> Studies with DU in our laboratory demonstrated neoplastic transformation of human cells under conditions where approximately 14% of the DU-exposed cells were transformed even though less than 5% were traversed by an alpha particle. These findings suggest that factors other than direct or "targeted" damage to the DNA may be involved in the transformations. Chemical effects of DU and "non-targeted" effects of radiation may also play a role. Non-targeted effects can result in damage in cells not traversed by an alpha particle. The overall level of transformation observed may result from contributions by any or all of these factors.[37]

In order to gauge the impact of radiation and heavy metal toxicity separately, the effects of depleted uranium was compared to that of nickel and to gamma irradiation. From the results of their experiments, Miller *et al.* concluded that

> In summary, we have presented data showing the production of genomic instability in the progeny of human cells exposed to DU. The findings demonstrate that DU can induce delayed cell death and genetic alterations in the form of micronuclei. Compared to gamma radiation or Ni, DU exposure resulted in a greater manifestation of genomic instability. Although animal studies are needed to address the effect of protracted DU exposure and genomic instability in vivo, results obtained from our in vitro system can play a significant role in determining risk estimates of DU exposure.[38]

While the uncertainties remain significant, the growing body of evidence that is emerging from Miller *et al.*'s laboratory cannot be ignored and their conclusions that this research is likely to play an important role in shaping future risk assessments of DU needs to be seriously considered in the present context. In light of the amount of groundbreaking work that has been conducted in just the last six to seven years, considerations for the eventual disposal of the more than one hundred and thirty thousand metric tons of depleted uranium that would be generated by the proposed LES enrichment facility must included a meaningful contingency plan for dealing with the possible outcomes of this research as well as that of the other research described throughout this section.

Finally, before moving on to the other emerging potential risks we note briefly the fact that children as well as the embryo/fetus are likely to be at particularly high risk in relation to the mutagenic and carcinogenic nature of uranium. The ICRP notes that

> It is very well known that ionising radiation interferes to a high degree with cell proliferation. Therefore, biological systems with a high fraction of proliferating cells show high radiation responsiveness. High rates of cell proliferation are found throughout prenatal development. However, although cell proliferation is a key process for the development of radiation effects, the sensitivity of the embryo and fetus is also determined through processes of differentiation and cell migration, and the radiation effects on these biological processes.[39]

[35] Miller et al. 2002c p. 251
[36] Miller et al. 2002c p. 251
[37] Miller et al. 2003 p. 248
[38] Miller et al. 2003 p. 257
[39] ICRP 90 p. 9

They go on to note that

> Tissues such as brain, thyroid, bone, and breast appear to be more susceptible if exposed during normal periods of rapid growth (i.e. early childhood or puberty).

Acknowledging these larger risks to children from radiation exposure, the 2002 supplement to EPA Federal Guidance Report 13 reports mortality and morbidity coefficients per becquerel of intake for age groups ranging from 0 to 5 years old through 25 to 70 (although it reports the age averaged value that is used in the current regulations and to which we have referred in our analysis above as ranging from 0 to 110 years of age). For the three uranium isotopes present in DU, the risk of developing a fatal cancer per unit of intake for a child under five is roughly six to eight times greater than the average risk used currently by the EPA for dietary and drinking water intake respectively.[40]

Taken together, these considerations of increased risk per unit of intake combined with the unique exposure pathways for children to environmental contaminants such as DU and the fact that uranium is known to cross the placental barrier and concentrate in the embryo/fetus, add additional uncertainties to projecting what the outcome of future research on uranium risks will be. It is plausible that far stricter requirements for the disposal of DU will need to be adopted in the event that uranium is concluded to be more carcinogenic than currently believed, and particularly if children's health is to be protected in that event.

Reproductive effects

Investigations of the reproductive effects of uranium exposure were reported as far back as the 1940's, however, these early studies do not appear to have been systematically followed up on by other researchers until many decades later.[41] Even today, there are substantial gaps in our understanding of uranium's effects on human and animal reproduction. In the early reported experiments, it was found that either continuous feeding or even just a single one time feeding of uranium to rats could detrimentally affect the animal's reproductive success. The impact of continuous feeding was significantly greater than that of the one time ingestion, but the authors noted their surprise at finding a continuing impact on the rat's reproduction even 9 months after a single exposure to uranium.[42] In summary, the authors concluded that "under the circumstances of this experiment, uranium administration adversely affected the reproductive functions in the absence of a severe derangement of nutrition."[43]

Why these provocative early studies do not appear to have been carried forward or more widely reported in not currently clear. However, the work that has been carried out quite recently on uranium has expanded these early findings, and has resulted in the identification of two distinct areas of concern in regard to the potential impact of uranium on reproductive health. The first area relates to the risks associated with exposures to men while the second relates to exposures of

[40] EPA FGR 13 CD Supplement 2002
[41] The reported experiments were apparently carried out during the Second World War as part of the Manhattan Project although they were not publicly reported until the late 1940s and early 1950s, (Voegtlin & Hodge 1953 p. vi)
[42] Voegtlin and Hodge 1953 p. 1255-1256
[43] Voegtlin and Hodge 1953 p. 1258

women. In regard to the possible effects on men, uranium is found to concentrate in the testes and has been found in the sperm of Gulf War veterans at elevated levels. While no epidemiological data yet demonstrates an impact on reproductive success from the veteran's exposure, the Royal Society noted that the concentration of DU in the testes was a potential concern given the possible synergistic effects between uranium's ability to damage DNA through both chemical oxidative stress and ionizing alpha radiation.[44] In addition, the World Health Organization has noted the observation of "unspecified degenerative changes in the testes" of rats as a result of chronic ingestion of soluble uranium compounds.[45]

Although still very limited, somewhat more work has been done on the reproductive effects of uranium exposure on females. Uranium have been shown to cross the placental barrier and concentrate in fetal tissue.[46] Experiments with animals have demonstrated that exposure to uranium either through ingestion or injection can cause "[d]ecreased fertility, embryo/fetal toxicity including teratogenicity, and reduced growth of the offspring."[47] These findings have been demonstrated in both rats and mice, and provide evidence (at least at the levels of intake examined) that uranium exposure can adversely affect the reproductive success of females.[48] The one reported experiment to use depleted uranium did not find statistically significant effects on "maternal weight gain, food and water intake, time-to-pregnancy, or the percentage of litters carried to terms," however, higher numbers of implanted DU pellets were found to lead to increasing concentrations of uranium in the placenta and whole fetus.[49]

While there are still many unknowns as to what the effects of uranium on reproductive success are, there have been a number of potential radiological and non-radiological mechanisms proposed to help explain the effects that have already been observed. These proposed mechanisms included hormonal or enzymatic disruption and behavioral changes.[50] In addition, we have already noted the ICRP's conclusions regarding the greater general radiosensitivity of the developing embryo/fetus as well as of young children which may also play a potential role in DU's effects on reproductive success. Given the high premium that is placed by many on the ability to have children, the potential for reproductive effects of depleted uranium to become an important issue as the research unfolds over the coming decades needs to be considered by LES and the NRC before the decision is made to go ahead and produce such large quantities of depleted uranium as envisioned in the case of the proposed LES enrichment facility.

Neurotoxic effects

Limited evidence raising the possibility of a link between uranium and neurological damage dates back to at least the mid-1980's.[51] These studies, however, have a number of problems that have hampered their usefulness in drawing any solid inferences regarding the neurological risks of

[44] Arfsten, Still & Ritchie 2001 p. 180, Domingo 2001 p. 606, and Royal Society Part II 2002 p. 14
[45] WHO 2001 p. 71
[46] Albina et al. 2003 p. 1072 and Royal Society Part II 2002 p. 14-15
[47] Domingo 2001 p. 603
[48] Albina et al. 2003 p. 1075-1076, Craft et al. 2004 p. 309, and WHO 2001 p. 71
[49] Arfsten, Still, & Ritchie 2001 p. 185 and Domingo 2001 p. 607
[50] Arfsten, Still, & Ritchie 2001 p. 189 and Domingo 2001 p. 605
[51] Pellmar et al. 1999 p. 785 and Craft et al. 2004 p. 307

DU.[52] One of the major concerns connected to the recent work that has been done on the potential toxic effects of depleted uranium on the brain centers around the fact that uranium's primary chemical form in the body is as the uranyl cation ($UO_2^{2+}$) which is a toxic heavy metal chemically analogous to the lead cation ($Pb^{2+}$) that has a well documented and tragic history as a neurotoxin and a particular concern in relation to children's health.[53]

In 1999 Pellmar *et al.* at the Armed Forces Radiobiology Research Institute showed that depleted uranium implanted in mice concentrated in various regions of the brain with an increasing concentration with increasing exposure. From these results they concluded that "[t]he accumulation in brain, lymph nodes, and testicles suggest the potential for unanticipated physiological consequences of exposure to uranium through this route."[54] In additional research, Pellmar *et al.* were able to further show that the "exposure to DU fragments caused neurophysiological changes in the hippocampus."[55] The hippocampus was chosen for analysis because it is the region of the brain involved in memory and learning. Reviews of these AFRBI experiments have concluded that these results provide important evidence of the potential for depleted uranium to display neurotoxic properties.[56] In addition to the work of Pellmar *et al.*, in 1998 Ozmen and Yurekli showed that following ingestion, uranium concentrated to a large extent in the brain of mice, while in 2003 Lemercier *et al.* demonstrated that a significant amount of uranium also concentrated in the brains of rats.[57] Lemercier *et al.* were also able to identify that the uranium in the brain was predominately in the form of uranyl tricarbonate.[58]

A specialized computer test designed to assess "performance efficiency" has been used to look for potential neurological effects in veterans who were exposed to depleted uranium munitions during the Gulf War.[59] These tests, conducted at the Baltimore VA Medical Center, observed a statistically significant correlation between uranium concentration in the veteran's urine and poor performance on the computerized neurocognative tests.[60] However, no measurable effects were found in this same group using traditional neurocognative tests.[61] It is important to recall in this case that the soldiers were exposed as adults, and that these tests cannot provide information on the impacts of uranium exposures during the more sensitive stages of early childhood when the brain is undergoing rapid growth and development.

In addition to the potential for uranium to play a chemically neurotoxic role in analogy to lead, radiation is also known to adversely affect the nervous system of the embryo/fetus. From a review of the atomic bomb survivors, the International Commission on Radiological Protection has concluded that

[52] Fulco, Liverman, & Sox 2000 p. 151 and 153
[53] Lemercier et al. 2003 p. 243 and Domingo 2001 p. 603
[54] Pellmar et al. 1999a p. 29
[55] Pellmar et al. 1999 p. 790 and Lemercier et al. 2003 p. 243 and 245
[56] McClain et al. 2001 p. 117-118 and Craft et al. 2004 p. 307-308
[57] Ozmen and Yurekli 1998 p. 111 and Lemercier et al. 2003 p. 245
[58] Lemercier et al. 2003 p. 245
[59] Arfsten, Still, & Ritchie 2001 p. 185
[60] Pellmar et al. 1999 p. 791, Royal Society Part II 2002 p. 13, and Craft et al. 2004 p. 307
[61] Craft et al. 2004 p. 307

> There is a clear constellation of effects of prenatal irradiation on the developing central nervous system – mental retardation, decreased intelligence scores and school performance, and seizure disorders.[62]

The ICRP elaborates further on why the prenatal period is of particular concern for radiation damage to the nervous system and why it is so important to consider in assessing risks. The Commission notes that

> Development of the central nervous system starts during the first weeks of embryonic development and continues through the early postnatal period. Thus development of the central nervous system occurs over a very long period, during which it is especially vulnerable. It has been found that the development of this system is very frequently disturbed by ionising radiation, so special emphasis has to be given to these biological processes.[63]

Prenatal exposures to lead and mercury have also shown an indication that they are capable of doing neurological damage during this period of rapid development, while the early years of childhood are generally considered to be the most critical time period for the damage from exposures to heavy metals given their greater potential for environmental exposure. As with a number of other emerging risks discussed above, there is also the potential for synergisms between uranium's chemical and radiological effects in relation to its effects on the nervous system. We will discuss further implications of this research on uranium's potential neurotxicity below, but it is important to note here that even relatively small changes in average IQ, if spread over a large number of children, will "dramatically increase the proportion of children below any fixed level of concern, such as an IQ of 80, and decrease the proportion above any 'gifted' level, such as 120."[64] Thus the effect of neurotoxic agents, even at very low levels, on an exposed population as a whole can end up being quite significant even if the effect on an "average" or "typical" member of that population does not appear so.

Skeletal effects

As with the brain, many elements of the fetal skeleton "show a complex and thus radiosensitive genesis" and the other periods of the bone's rapid development (i.e. in early childhood and during puberty) are all additional times of heightened sensitivity.[65] In experiments on rats, it has been demonstrated that both acute and chronic intakes of uranium can cause damage to bones, and the Royal Society has recommended that in light of the fact that uranium crosses the placental barrier that "the effects of maternal exposure to DU on skeletal development in the foetus may also need to be considered."[66] As noted below, the World Health Organization and the National Research Council have also recommended studies to determine what effect, if any, uranium integrated into the bone has on the bone marrow and thus on the production of new blood cells.

The future of DU effects research

[62] ICRP 90 p. 118
[63] ICRP 90 p. 9
[64] Rogan and Ware 2003 p. 1516
[65] ICRP 90 p. 30 and 149
[66] Royal Society Part II 2002 p. 13 and 67

There is clearly much that is still not understood about the array of potential deleterious effects that chronic or acute exposures to uranium, including depleted uranium, may cause. According to a recent study by the National Research Council, "[s]urprisingly there are still substantive gaps in knowledge of the non-radiological health impacts of exposure to uranium and its compounds."[67] As summed up in a recent review by Craft *et al.* from Duke University,

> Although most of the DU absorbed in the body is metabolized and excreted, enough is distributed throughout the body to raise important toxicological concerns… The long term effects of DU still have to be definitely resolved, and there is an obvious need for continued studies.[68]

In its 2001 review of depleted uranium, focusing in particular on the impact of military munitions, the World Health Organization concluded that there is inadequate information available concerning the potential impact of uranium in the following areas, and that additional research needs to be undertaken:

> - Neurotoxicity: Other heavy metals, e.g. lead and mercury are known neurotoxins, but only a few inconsistent studies have been conducted on uranium. Focused studies are needed to determine if DU is neurotoxic.
>
> - Reproductive and developmental effects have been reported in single animal studies but no studies have been conducted to determine if they can be confirmed or that they occur in humans.
>
> - Haematological effects: Studies are needed to determine if uptake of DU into the bone has consequences for the bone marrow or blood forming cells.
>
> - Genotoxicity: Some in vitro studies suggest genotoxic effects occur via the binding of uranium compounds to DNA. This and other mechanisms causing possible genotoxicity should be further investigated.[69]

A 2003 National Research Council report concerning management of the Department of Energy's depleted uranium stockpile adopted all four WHO recommendations for research.[70] This decision by the NRC strengthened the 2000 recommendation from the Institute of Medicine's Committee on Health Effects Associated with Exposures During the Gulf War that additional animal studies should be conducted to investigate the biological effects of depleted uranium with a particular focus on "studies of cognitive function, neurophysiological responses, brain DU concentrations, and the transport kinetics of DU."[71] In addition, the Royal Society has also endorsed further research on many areas of DU's effects in the body, including studies concerning its potential neurocognitive and reproductive health effects.[72]

As a final example of the current unknowns, we note two additional uncertainties that have been highlighted by the World Health Organization and the National Research Council concerning the potential health effects of DU. In 2001, WHO noted that there is "[v]ery limited information" available on the variability of sensitivity to uranium toxicity among different individuals.[73] Related to this uncertainty is the fact that, as WHO concluded,

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[67] NAS-NRC 2003, p. 67.
[68] Craft et al. 2004 p. 315
[69] WHO 2001 p. 148-149
[70] NAS/NRC 2003 p. 67-68
[71] Fulco, Liverman, & Sox 2000 p. 327
[72] Royal Society Part II 2002 p. 67-68
[73] WHO 2001 p. 80

> Children are not small adults and their exposure may differ from an adult in many ways. Unfortunately, despite their obvious importance little definitive data exists concerning how their uranium exposure differs from that of adults.[74]

In this same vein, the review by the Royal Society in 2002 noted that animal models "suggest that absorption of uranium from the gut of neonates might be higher than in older children or adults."[75] The World Health Organization, the National Research Council, and the Royal Society have all recognized the need for additional studies to better assess the impacts of uranium exposure on children.[76]

Despite the still somewhat limited amount of available data, the pattern of risks that are emerging clearly indicate the need for great caution in managing and disposing of depleted uranium, particularly in the large quantities envisioned to be generated by the proposed LES facility and its extremely long lived nature. There are clear indications that uranium toxicity for at least some effects, including its neurotoxic effects on fetuses and young children, might be better understood if uranium was considered to be analogous to a kind of radioactive lead, in which the damage from the alpha radiation occurs in conjunction with heavy metal induced damage to produce a variety of health problems at relatively low levels of exposure. This analogy between uranium and lead was made in 2003 by Lemercier *et al.* in reporting their study demonstrating the concentration of uranium in the brain of rats.[77] While this way of thinking has obvious limitations in regards to understanding the detailed biological mechanisms involved in the damage caused by uranium as compared to lead, the ability of uranium to chemically induced oxidative stress, to cross the blood brain barrier and alter electrical activity in parts of the higher brain, and to potentially interrupt neurotransmitters through chemical replacement of calcium in the interneuron gaps all in combination with the high levels of local cellular damage caused by alpha radiation raise significant warning signs about the potential impact of this material on a child's developing brain.

In light of the analogy of uranium to lead it should be noted that despite evidence of lead's damaging effect on the brain dating back more than two millennia and lead poisioning being first recognized in children as early as the 1890's, it was not until 1979 that leaded gasoline was finally taken off the US market after being widely sold for several decades.[78] As with the general trend in radiation protection standards, the Centers for Disease Control (CDC) has chosen to lower the guideline level it considers to be an indicator of "elevated" levels of lead in the blood of children four times since the late 1960s. The level today is one-sixth of where is stood 35 years ago.[79] In addition, the CDC has adopted the position that there is no safe level of exposure to lead and that any intake will result in some level of harm. Unfortunately, despite significant reductions in exposure since 1979, the current levels of lead in children's blood are still roughly 100 to 1000 times larger than the estimated pre-industrial levels, and as of 2000 the CDC estimated that nearly half a million children in the U.S. still exceeded their guideline for elevated levels of lead in the blood.[80] Adding to these concerns, continuing research on the effects of lead have shown that children's intellectual function is adversely affected by exposures to lead that are roughly half of

[74] WHO 2001 p. 30
[75] Royal Society Part II 2002 p. 17
[76] WHO 2001 p. 148, NAS/NRC 2003 p. 68, and Royal Society Part II 2002 p. 15
[77] Lemercier et al. 2003 p. 243
[78] Koller et al. 2004 p. 987
[79] Rogan and Ware 2003 p. 1515 and Canfield et al. 2003 p. 1518
[80] Koller et al. 2004 p. 993

the CDC and WHO level of concern further supporting the conclusion that there is likely no threshold for lead's damaging effects.[81]

In addition to lead's neurotoxicity, recent research has also shown that both prenatal and postnatal exposure to lead is associated with retarded growth in animals and humans and that exposure to lead can also alter sex hormone production and delay puberty in rats.[82] An epidemiological study published in 2003 has shown that even relatively low average levels of lead (roughly a third of the CDC/WHO level of concern) caused a measurable delay in puberty in African-American and Mexican-American girls, while no statistically significant delay in Caucasian girls was found.[83] This effect on the girl's sexual development was attributed, at least in part, to potential "alterations in endocrine function."[84] While many questions as to how lead caused the observed delay and whether or not the children had been exposed to higher levels in the past before the study's screening began remain unanswered, the potential for uranium to play an analogous role in effecting hormonally mediated processes in developing children could add further to its list of health concerns and add significant new avenues for potential synergisms with its other chemical and radiological heath effects.

The lessons of lead's tragic history in relation to children's health, including the decades long denial of every growing evidence of the risks by industries producing lead based products, as well as the systematic and progressive tightening of health guidelines specifically targeting children once they were finally introduced should be closely examined in relation to the direction uranium research is now unfolding. The enrichment plant that LES proposes to build will generate significant quantities of DU over the coming decades which will also likely be a time of rapid and significant expansions in the understanding of uranium and its various health effects both in isolation and in combination with other environmental stressors. In this context LES and the NRC, which is legally charged with protecting the public health, must pursue a management and disposal strategy that will have a high probability of doing just that and they must also be prepared to modify and adapt this plan in the event that radiation risks in general and uranium risks in specific are found to be greater than previously considered and that provisions are be undertaken to specifically protect both women and children's health. The inclusion of a specific cost contingency as one necessary component of building in this flexibility is discussed in Chapter IV.

## C. Regulatory Aspects of Depleted Uranium Disposal

The staff of the NRC currently takes the position that DU is Class A low-level radioactive waste.[85] However, the Commission itself has made no ruling on this question, since LES withdrew its application for a license to build and operate a uranium enrichment plant in Louisiana. Uranium is still officially classified as a source material by the U.S. Department of Energy (DOE) as well as by the NRC. This will remain the case in the absence of a specific ruling from the Commission that depleted uranium is a waste. No final disposal strategy has been chosen or fully analyzed by the DOE in relation to the management of its depleted uranium stockpile since the DOE is still

[81] Canfield et al. 2003 p. 1521 and 1525
[82] Selevan et al. 2003 p. 1528
[83] Selevan et al. 2003 p. 1527
[84] Selevan et al. 2003 p. 1535
[85] NRC NEF EIS Draft 2004 p. 2-27

considering possible, but unlikely, uses for its DU.[86] LES has also not definitively decided whether it considers the depleted uranium to be generated by the proposed enrichment facility to be a resource or a waste, though it claims that it can decide this question without reference to any regulatory authority. However, the Atomic Energy Act requires source material to be regulated:

> The processing and utilization of source, byproduct, and special nuclear material must be regulated in the national interest and in order to provide for the common defense and security and to protect the health and safety of the public.[87]

A change in the status of depleted uranium cannot be made by a corporation. It would require a specific ruling by the Commission.

The idea put forth by the NRC staff and LES that DU, with a specific activity of more than 300 nanocuries per gram arising from long-lived alpha-emitting radionuclides, can be considered as Class A low-level waste if it is to be disposed derives from 10 CFR 61.55(a)(6), which states that "[i]f radioactive waste does not contain any nuclides listed in either Table 1 or 2, it is Class A."[88] Since uranium is not listed in either table, the argument is that it would therefore be considered Class A waste suitable for shallow earth disposal.

However, 10 CFR 61.55(a) is internally inconsistent. Paragraph (3) defines waste containing up to 100 nanocuries per gram of long-lived alpha-emitting transuranic radionuclides as Class C low-level waste. Waste with a concentration of these radionuclides greater than 100 nCi/gm is therefore defined as greater than Class C (GTCC) waste and generally cannot be disposed of in shallow land burial. In the Claiborne Enrichment Center case, the NRC staff took the position that depleted uranium in the form of $U_3O_8$ powder was Class A LLW despite the fact that the NRC staff's own analysis in the FEIS showed that shallow land disposal would result in doses in excess of 25 mrem per year limit specified in the LLW regulation and that it would therefore likely require deep geologic disposal.[89]

While uranium was considered in a proposed draft of the rule, the final version of 10 CFR 61.55(a) explicitly removed it from consideration. This decision was made, in part, because (in addition to uranium being considered a source material) the NRC concluded that "the types of uranium-bearing wastes being disposed of" at that time did not present "a sufficient hazard to warrant limitation on the concentration of this naturally occurring material."[90] However, this consideration is clearly not applicable to the disposal of bulk $DU_3O_8$ powder as has been proposed by LES. Our analysis presented in Chapter II highlighted many of the reasons that disposing of depleted uranium cannot be considered to be less risky than the disposal of TRU waste at the threshold of 100 nCi/gm. This type of reasoning was also adopted in 1992, during the Claiborne Enrichment Cednter Case, by Kozak *et al.* at Sandia National Laboratory in a report entitled "Performance Assessment of the Proposed Disposal of Depleted Uranium as Class A Low-Level Waste." In this report, the authors concluded that

> Given the large inventory and form of the depleted uranium wastes, and the fact that this type of waste was not included in the Environmental Impact statement (EIS) analyses supporting 10 CFR

[86] DOE Paducah EIS 2004 p. 2-11, 2-17, and 2-25
[87] AEA vol. 1, p. 1.
[88] 10 CFR 61 final rule 1982, p.588
[89] NRC CEC EIS Final 1994 p. A-9
[90] 10 CFR 61 final rule 1982, p.57456

> 61, further analysis is necessary to demonstrate whether the disposal of this material in a 10 CFR 61 disposal facility [shallow land burial] will be acceptable in terms of public health and safety.[91]

As already indicated, the NRC, which made use of the Sandia analysis, recognized that its own analysis indicated that shallow disposal would not be able to meet the 25 mrem annual dose limit for the conditions then assumed. The option of deep mine disposal is not covered in the regulation and paragraphs (3) and (4) indicate that anything requiring deep disposal is not LLW at all, but is, in fact, GTCC waste.

In the present LES case, the NRC staff has again taken the position that DU is Class A low-level waste and that it might be disposed of by shallow land burial in a dry location. Although a number of low-level waste disposal sites were noted in the LES DEIS, no specific option was chosen and none of the indicated sites would likely be able to safely dispose of the DU in shallow trenches. Significantly, no estimates of the possible doses under dry conditions for any locations are given in the DEIS in support of this proposed disposal option despite the failure of the eastern site considered for shallow disposal in the CEC case to meet the 25 mrem annual dose limit. The NRC also states that doses from deep disposal of depleted uranium in a mine would be low and provides estimates of doses under a well water and river water scenario. As presented in the DEIS these estimates are greatly below the regulatory limit of 25 mrem per year for LLW disposal.[92] The estimates as provided are stated to be based on the CEC estimates in the FEIS of 1994. However, despite this assertion, the NRC has failed to provide IEER with the methods and assumptions underlying the dose calculation and the details of the CEC FEIS calculation are apparently no longer available, even to the NRC itself. Moreover, the doses in the current LES DEIS are not broken down by radionuclide and the totals as presented are different from those reported in the CEC FEIS by nearly a factor of 2 with one notable exception. The difference in most of the results may be explained, at least in part, by the fact that the proposed LES enrichment facility will generate roughly twice the amount of depleted uranium tails that must be disposed of. However, the estimate for the drinking water dose in the river drinking water scenario following disposal in a sandstone/basalt site are almost 54,000 times lower in the LES DEIS than the results presented in the CEC FEIS.[93] A definitive resolution of these discrepancies is not now possible given that, as we have noted, the NRC has been unable to produce a detailed justification for their current dose estimates.

The doses from U-238 estimated in the CEC FEIS for deep disposal are incredibly low (literally). The annual background dose due to drinking water with approximately 0.1 pCi/liter of uranium in it amounts to about 0.02 mrem EDE (effective dose equivalent). The drinking water dose estimated from the disposal of pure $DU_3O_8$ powder in a mine was estimated by the NRC in the CEC case to be a million to a trillion times lower than this typical background level. Indeed, the highest well water dose estimated by the NRC is less than that caused by the ingestion of an amount of uranium that would result in just the disintegration of six uranium atoms in the entire body over an entire year. The lowest drinking water dose for U-238 reported would imply that the total amount of energy deposited in a 70 kilogram adult from the uranium absorbed through the drinking water would be equal to less than the amount of energy required to ionize a single hydrogen atom.[94] In other words, this means that the amount of uranium that the NRC predicts

[91] Kozak et al. 1992 p. 1
[92] NRC NEF EIS Draft 2004 p. 4-59
[93] NRC NEF EIS Draft 2004 p. 4-55, 4-59 and NRC CEC EIS Final 1994 p. A-1, A-14 to A-15
[94] NRC CEC EIS Final 1994 p. A-14 to A-15

would accumulate in a person's body as a result of living on top of a mine containing tens of thousands of metric tons of pure depleted uranium oxide and drinking the ground water would over time amount to just 30 trillionths of the amount naturally in the body that accumulates from the uranium leaching out of ordinary rocks and dirt. IEER's estimates of dose indicate that the NRC's dose estimates are likely to be wrong by many orders of magnitude though a more thorough demonstration of any discrepancy is not now possible due to the failure of the NRC to provide the methods and details of its calculations in the present case concerning the proposed LES facility.

In an attempt to examine the potential impacts of deep disposal in a more general way, we have also made simple estimates of potential doses from DU disposal in a mine based on the assumptions that water would enter the mine and reach equilibrium with the depleted uranium powder, with and without the presence of carbon dioxide (i.e. air). For the disposal of $DU_3O_8$ powder under the conditions analyzed, the solubility of the uranium in the water was found to be between $7.5 \times 10^{-7}$ and $2.9 \times 10^{-6}$ moles per liter, with a corresponding drinking water dose from the U-238 alone in the range of tens of millirem per year. The dose from the other uranium isotopes and the other decay products would add to the total potential dose. In addition, we found that the amount of U-238 in the water in the mine would be approximately 6 to more than 20 times the current EPA drinking water limit of 30 μg/L. [95]

For a comparison of our results to those presented by the NRC in the CEC FEIS we note that if just one part in a million of the uranium dissolved in the water filling a mine with a volume of 20,000 $m^3$ reached the drinking water[96], the implied dilution required to reduce the dose from U-238 to the levels given by the NRC calculation for disposal in a sandstone/basalt mine would exceed the volume of water in all of the Great Lakes combined.[97]

The method we adopted for performing this screening analysis is generally consistent with the approach taken by Kozak *et al.* in the analysis carried out at Sandia National Laboratory during the CEC case. In the Sandia study the authors ignored the effects of all other elements that might influence uranium solubility and considered the $DU_3O_8$ to be equilibrium with both water and air. Their analysis found a minimum solubility of $10^{-6}$ moles/liter while over a wider range of pH values they found that it was likely that the uranium solubility in their case would be less than $10^{-5}$ moles/liter.[98] Calculations using this range of solubilities would again correspond to effective doses on the order of tens of mrem per year. Importantly, this range of solubilities would also lead to uranium concentrations in the water 8 to 80 times larger than the current EPA drinking water limit and would again imply the incredibly large dilution values noted above in order to be consistent with the NRC dose estimates for drinking well water following disposal in a sandstone/basalt mine.

In addition to the considerations of deep disposal, we have also made a number of test runs of various scenarios using ResRad to estimate doses under various assumptions for near-surface

[95] EPA 2004

[96] Assuming a density of 3 gm/$cm^3$, the volume of the DU3O8 powder that was proposed for disposal in the CEC case would have alone totaled approximately 20,000 $m^3$. (NRC CEC EIS Final 1994 p. A-1, A-7)

[97] For reference, the Great Lakes system contains approximately 23,000 km3 of water accounting for roughly 18% of the entire world's supply of fresh, surface water. (EPA Great Lakes Atlas)

[98] Kozak et al. 1992 p. 31

disposal of DU in a presently arid climate.[99] The modeling program ResRad created at Argonne National Laboratory has been accepted by the NRC for use in performing dose calculations in support of nuclear facility decommissioning. In these test runs we have retained the default assumption of ResRad that considers a "reference man" in its evaluation of radiation exposure. This assumption does not take into account the fact that children are likely to be a more sensitive population to both the chemical and radiological effects of uranium. However, as we will show below, shallow disposal of uranium is highly unlikely to be able to meet health and safety regulations even for the reference man.

In our screening calculations, the depleted uranium is assumed to be disposed of as a layer of bulk $DU_3O_8$ powder more than 26,000 $m^2$ in area and two meters thick under 7.6 meters of cover material with an unsaturated zone 20 meters thick separating the DU from the underlying aquifer. Due to the long time scale that must be considered for the radionuclides in depleted uranium (including the decay products) as well as the size of the land area required for disposal of the projected volume of DU that would be produced from the proposed LES facility, we calculated a resident farmer scenario with the assumption that all food and water would be derived from on site. The results of these very basic screening calculations are shown in Table 5. Significantly, all of the results are several orders of magnitude greater than the regulatory limit of 25 mrem per year making it extremely unlikely that this type of disposal strategy, even in a dry climate, would be acceptable.

In all but the lower $K_d$ and moderate effective moisture scenarios, the peak dose reported in Table 5 is dominated by direct plant uptake through the roots occurring after the cover material has eroded away to a thinner level, while in the other two scenarios (which result in far larger peak doses) drinking contaminated water is the dominant exposure pathway. Importantly, however, we found that for the scenarios in which the uranium does not reach the aquifer with the 100,000 year timeframe analyzed by ResRad, the external radiation dose at the time of the peak dose would alone exceed the 25 mrem annual limit by 1,270 to nearly 3,000 times. These numbers are generally consistent with the external dose calculation from the Sandia study for the intruder scenario. At 10,000 years, Kozak *et al.* found that the external dose would be 13.5 rem per year with Ra-226 contributing more than three-fourths of that dose.[100] This annual dose rate would be 540 times greater than the 25 mrem limit. Given that the Sandia analysis used a different type of dose calculation and different assumptions regarding such things as the dilution of the waste in the disposal site in addition to the fact that most of our peak doses were evaluated after the Ra-226 had had longer to form, these sets of results are in satisfactory agreement.

Finally, we also examined the radon-222 dose that would be received by someone on top of such a shallow disposal site for this amount of $DU_3O_8$. For the scenarios in which the uranium did not reach the aquifer, the peak indoor and outdoor radon emanations calculated by ResRad exceeded the EPA limit for any source at Department of Energy facilities of 20 $pCi/m^2$-s by up to 400 times.[101] While this regulation would not directly apply to a commercial disposal facility

[99] Over the very long timescales that must be considered for the disposal of depleted uranium the impacts of natural and potentially anthropogenic climate change need to be considered in determining the land use scenario and meteorological parameters for use in dose modeling. Although it is outside the scope of the present report, this consideration argues for the use of the resident farmer scenario as part this type of generic screening analyses.
[100] Kozak et al. 1992 p. 13
[101] 40 CFR 61.192 p. 143

disposing of depleted uranium produced at a commercial enrichment plant, the fact that the radon emanations from shallow land burial could exceed this limit by orders of magnitude needs to be considered in relation to the acceptability of such a strategy.

*Table 5: Summary of our ResRad dose calculations for shallow earth disposal of $DU_3O_8$ powder under a variety of assumptions for an arid climate. The annual doses for the uranium isotopes as shown include the contribution from their respective decay products as well. It is important to note that the doses are given in rem per year as opposed to mrem per year.* (All numbers have been rounded)

| Scenario | U-238 Dose | U-235 Dose | U-234 Dose | Total Peak Dose (rem per year) | Time at Peak Dose (years after emplacement) |
|---|---|---|---|---|---|
| higher $K_d$ (clay)<br>low effective moisture<br>low erosion | 88 | 47 | 200 | 336 | 17,412 |
| higher $K_d$ (clay)<br>moderate effective moisture<br>low erosion | 78 | 42 | 185 | 306 | 17,412 |
| higher $K_d$ (clay)<br>low effective moisture<br>moderate erosion | 72 | 30 | 109 | 210 | 9,807 |
| higher $K_d$ (clay)<br>moderate effective moisture<br>moderate erosion | 67 | 28 | 104 | 199 | 9,807 |
| | | | | | |
| lower $K_d$ (sand)<br>low effective moisture<br>low erosion | 32 | 26 | 121 | 179 | 17,403 |
| lower $K_d$ (sand)<br>moderate effective moisture<br>low erosion | 658 | 14 | 124 | 795 | 13,433 |
| lower $K_d$ (sand)<br>low effective moisture<br>moderate erosion | 38.4 | 21 | 81 | 141 | 9,807 |
| lower $K_d$ (sand)<br>moderate effective moisture<br>moderate erosion | 658 | 14 | 124 | 795 | 13,433 |

Notes:

- The $K_d$ values used for the uranium and its decay products were taken from the ResRad data collection manual for the indicated soil type. All other soil parameters not set to default values remained at the values appropriate to clay.[102]
- low effective moisture = rain of 0.178 m/yr and an evaotransportation coefficient of 0.9
- moderate effective moisture = rain of 0.356 m/yr and an evaotransportation coefficient of 0.7
- low erosion = 0.0005 m/yr
- moderate erosion = 0.001 m/yr

[102] ResRad data collection manual p. 110-111

In addition to the failure of shallow land burial to meet the 25 mrem per year dose limit at the time of peak dose by three to four orders of magnitude and its failure to meet the radon flux limit for DOE facilities by one to two orders of magnitude, in the two scenarios where the DU and daughter products did break through into the ground water within the first 100,000 years (lower $K_d$ and moderate effective moisture), the concentration of uranium estimated to be in the drinking water at the time of the peak dose is over 450,000 times larger than the 30 μg/L limit set by the EPA.

It is worth noting that in all of the shallow burial scenarios in our test runs the peak doses occur well before full in-growth of the daughter products in the depleted uranium can occur. Even after 100,000 years (far longer than the longest time at peak dose in Table 5), the U-234 component of the depleted uranium will have reached less than 40% of its equilibrium activity, while the Th-230, Ra-226, and Pb-210 will all have reached just 18% of their equilibrium value. The further in-growth of decay products would add to the unsuitability of the shallow land disposal strategy given that even if the soil conditions were such that either erosion or transport to the water table was slowed, the peak dose would generally just occur at a later time and increase in magnitude from the already extremely high values listed in Table 5 as additional daughter products slowly built up in the soil from the decay of long-lived uranium isotopes.

In other words, when the depleted uranium is lock up more tightly in the soil (i.e. larger $K_d$ values and less effective moisture), the cover has time to erode away and allows for both the direct plant uptake exposure pathway and eventually the external gamma pathway to contribute to a gross violation of the 25 mrem per year peak dose limit. On the other hand, when the uranium is more mobile in the soil (i.e. lower $K_d$ values and more effective moisture), the DU will eventually reach ground water and contribute to an even higher peak dose from drinking water intake. This type of problem with the disposal of such large amounts of pure depleted uranium as a powder was also observed in the results from the 1992 Sandia study. In reviewing the dose calculations for the various scenarios in their analysis, Kozak *et al.* noted that

> The varying conditions illustrate an important point about narrowing the uncertainty in solubility; if we are confident that the solubility is low, high intruder doses will be calculated; if we are confident about a high solubility, off-site doses will be high. Intermediate solubilities are likely to produce relatively high doses in both analyses. Furthermore, if the intrusion analyses take credit for some waste leaching into the groundwater, the analysis should probably include an on-site well in the evaluation.[103]

The high and slowly increasing specific activity of the waste, the large volume envisioned to be produced by the proposed LES facility, and the long lived nature of the major radionuclides present all combine to make near surface disposal extremely unlikely to be a suitable option for disposing of depleted uranium, even in an arid climate.

We do not claim these calculations to be in anyway exhaustive or definitive. On the contrary, they are simple screening calculations with many default and generic assumptions aimed at helping to address three important questions regarding the disposition of the depleted uranium tails from the proposed enrichment facility:

[103] Kozak et al. 1992 p. 49

1. Are the NRC calculations in the CEC FEIS reasonable and supportable given that they are claimed to have been derived by the same mechanism as the estimates in the current LES DEIS?
2. Are the generic calculations presented by LES and the NRC for the doses resulting from DU disposal a reasonable basis for estimating compliance?
3. Can shallow land disposal be ruled out as an option for DU disposal?

As regards the first question, the analysis we have done indicates that it is reasonable to consider the NRC's well water scenarios in the CEC case (and therefore also in the present case) as scientifically improbable and likely wrong in one or more respects. A final determination of this issue cannot be made until the NRC actually provides all the details concerning their calculations including which models were used, how those models were run, and all of the various assumptions that were made in regards to the site, the model parameters, and the exposure scenarios.

As regards the second issue, our screening calculations show that generic calculations are not a plausible basis for assuming that disposal in some unspecified mine can meet radiation dose criteria or other health standards. In this regard, it is worth quoting the conclusions of the International Commission on Radiological Protection at length. In issuing its 1998 guidance on disposal of long-lived radioactive wastes, the ICRP concluded that

> Site specific assessments are essential in order to evaluate the radiological consequences of waste disposal. They are also necessary to understand, describe, quantify, and optimise the role of the different barriers of the disposal system and its subsystems. Assessments consider a number of scenarios where a scenario is defined as one possible combination of specified processes affecting the disposal system that could lead to radiological consequences. Typically, an assessment consists of the following elements, which are usually dealt with in an iterative manner: system understanding, scenario analysis, development of conceptual and detailed system models, consequence analysis, uncertainty and sensitivity analysis, and interpretation of the calculated results. An integrated assessment will evaluate the expected system evolution as well as less likely system evolutions including those caused by disruptive events of natural origin or as a result of human intrusion.[104]

In addition, Kozak *et al.* concluded that in the particular case of considering the disposal of $DU_3O_8$ powder in a mine that

> Uranium solubilities can vary widely, even under fairly well established ground-water chemical conditions. As an example, a recent performance assessment was performed of an arid site for which substantial site-specific ground-water characterization was available in this performance assessment the uranium solubility ranged over five orders of magnitude. It is clear that on a generic basis little can be done to specify a solubility range with much confidence.[105]

As a final example, a 1994 analysis of DU disposal options conducted for the Department of Energy noted that the distribution coefficient for uranium has been found to vary by four orders of magnitude "indicating a very large dependence on local soil conditions."[106] These examples highlight the need expressed by the ICRP for detailed site-specific evaluations when considering the disposal of depleted uranium.

[104] ICRP 81 p. 7
[105] Kozak et al. 1992 p. 49
[106] Hertzler et al. 1994 p.12

Given the specific activity of uranium, its increasing radioactivity over time due to the ingrowth of decay products, and uranium's other chemo-toxic characteristics, it will likely be difficult to find an adequate site for the disposal of DU, whatever classification it might be given by the Commission, that will be able to demonstrate compliance with the 25 mrem dose criteria and all other health restrictions with reasonable assurance. Thus the proposal of a generic site in lieu of a detailed investigation of a particular site cannot be considered a plausible strategy for the ultimate disposal of the large amount of depleted uranium that would be generated by the proposed LES enrichment facility. In regard to this conclusion, IEER sought an independent opinion from Dr. John Bredehoeft, one of the most eminent hydrogeologists in the United States and a member of the National Academy of Engineering, for inclusion in this report.[107] His statement is quoted in full below:

> Any processing facility must somehow dispose of the waste stream that contains radioactive constituents in a safe manner. A number of investigators, including me, have suggested strategies that can lead to safe geologic disposal facilities for nuclear wastes (Bredehoeft et al., 1978; Bredehoeft and Maini, 1981). However, the devil is in the details of how safe facilities, are designed, engineered, and built.
>
> The U.S. Department of Energy (DOE) opened one facility that is now receiving nuclear wastes generated by the U.S. weapons program—WIPP. WIPP was licensed for operation after several decades of investigation and scientific review, including building an exploratory mine in which experiments were conducted in-situ. The scientific community, as represented by the National Academy of Sciences/National Research Council, went on record indicating that the facility was safe. However, it took several decades of scientific work to reach this consensus.
>
> DOE is currently attempting to license a repository for high-level nuclear wastes at Yucca Mountain in Nevada. Investigations at Yucca Mountain have also gone on for several decades. This work includes an exploratory tunnel into the mountain.
>
> At both WIPP and Yucca Mountain data from the tunneling in the subsurface revealed unexpected results—*surprises*. At WIPP the original concept, going back to a National Academy of Sciences Committee in the mid 1950s, was that salt was a good medium for nuclear waste disposal because it was thought to be dry. Once the salt at WIPP was tunneled into, it was found to contain brine—1 to 3% in the interstices between salt crystals. Experiments in the mine demonstrated that this brine would migrate into the mine rooms. A mine that was originally conceived of a dry now was observed to be damp. This caused a rethinking of the conceptual model for WIPP.
>
> At Yucca Mountain chlorine 36 and tritium produced by bombs were found in the underground tunnel. This suggested that there existed fast paths for moisture movement in the mountain that the prevailing theory for moisture movement in unsaturated media does not predict. The theory has had to be modified to accommodate the fast paths for moisture movement.
>
> Both of these site-specific examples demonstrate the level of scientific and engineering effort necessary to license a nuclear waste facility. One cannot simply draw upon generic calculations to justify that nuclear wastes can be disposed of safely. Prudent design would

[107] Dr. Bredehoeft worked for the U.S. Geological Survey for 32 years before starting The HydroDynamics Group, a consulting firm, in 1995. He was a member of the National Academy of Sciences/National Research Council Committee on the Department of Energy's Waste Isolation Pilot Plant (WIPP) as well as a member of the NAS/NRC Panel responsible for reviewing groundwater issues at the Yucca Mountain Nuclear Repository. The complete Curriculum Vita of Dr. Bredehoeft accompanies this report along with runs of ResRad, and other materials.

dictate that one must propose a specific site and method of sequestering long-lived nuclear wastes. Only after a specific site and design are proposed can one assess its safety.

I reviewed the discussion of the two disposal sites in the 2004 draft environmental impact statement for the National Enrichment Facility (NEF) and the longer discussion of such sites in the text and appendix to the 1994 final environmental impact statement for the Claiborne Enrichment Center. The results (i.e., releases) for the two sites reported in these documents are calculations for hypothetical sites, not actual sites under investigation to receive the wastes of the NEF. No actual site for radioactive waste disposal of NEF wastes is identified in these documents—both are hypothetical sites.

As suggested above, to identify a suitable disposal site requires years of investigation, modeling, and additional investigation along with further modeling. It is an iterative process that typically includes construction of a site conceptual model, attempts to calibrate the model, and concurrent investigations to determine whether the conceptual model is appropriate or, perhaps, must be drastically revised or reconstructed. There is a continuing risk during the investigation that the site may fail to meet basic criteria for suitability.

The type of site required for disposal of depleted uranium from NEF is roughly comparable to the WIPP site in terms of the level of isolation required. All three isotopes contained in depleted uranium have very long half-lives, with the half-life of the principal one, U-238 extending to the billions of years. The specific activity of depleted uranium exceeds 300 nanocuries per gram of alpha-emitting radionuclides, and radium 226 and thorium 230 would build up over time to exceed 100 nanocuries per gram. The transuranic waste disposed of at WIPP has a concentration of at least 100 nanocuries per gram of alpha-emitters. The WIPP project involves deep disposal in a sealed mine in bedded salt more than 2000 feet below the surface. The plan for WIPP was examined in a detailed performance assessment, which was reiterated several times. It required well over 20 years of analysis by a large team of scientists and engineers to achieve a level of understanding such that a consensus was reached that the WIPP facility is safe and could receive waste.

Only after a specific site and design are proposed can one assess its safety. It would be prudent to assume that, before a site could be qualified to receive depleted uranium waste, a similar amount of time, effort, expense, and scrutiny to that which went to qualify WIPP would be required.

REFERENCES

Bredehoeft, J.D., England, A.W., Stewart, D.B., Trask, N.J., and Winograd, I.F., 1978, *Geologic Disposal of High-Level Radioactive Wastes--Earth Science Perspectives:* U.S. Geological Survey Circular 779, 15 p.

Bredehoeft, J.D., and Maini, T., 1981, *Strategy for radioactive waste disposal in crystalline rocks:* Science, v. 213, p. 293-296.

Finally, in regard to the third question regarding the suitability of shallow land disposal, as we have already noted, our test ResRad runs as well as the work of Kozak *et al.* at Sandia National Laboratory have shown that under a variety of assumptions for both arid and wet climates, that shallow disposal is extremely unlikely to be acceptable based on the 25 mrem per year peak dose limit or the EPA drinking water standard or even on the EPA radon flux guideline relating to DOE facilities. The fact that the NRC has not presented its own calculations for the expected doses from shallow land disposal in this case and the fact that no suitable, specific site has been

definitively identified or fully and adequately characterized in the DEIS for the disposal of depleted uranium makes further such considerations impossible at this time.

In addition to the considerations of expected doses that would potentially result from disposal of DU in a shallow trench, we have shown above that depleted uranium is most analogous to transuranic or Greater than Class C waste, and that therefore if the Commission was to rule that DU is, in fact, a waste, it is unlikely to be included in the definition of LLW that is suitable for shallow land disposal under 10 CFR 60.55(a). Finally, adding further to these issues of waste classification is the possibility that the recognized chemical toxicity of uranium and the recent work exposing additional concerns regarding DU's effects in the body described in Chapter II may cause depleted uranium to be treated as mixed waste that would, in part, be covered by the considerations of the Resource Conservation and Recovery Act (RCRA).

The possibility of uranium requiring disposal under RCRA requirements in addition to other radioactive waste regulations was raised a decade ago in a 1994 study prepared for the U.S. Department of Energy. The study "Depleted Uranium Disposal Options Evaluation" concluded that DU was currently a source material regulated by the Atomic Energy Act of 1954.[108] However, the report also concluded that if DU was to be declared a waste that a worst case scenario for consideration of the disposal costs would be that both the depleted uranium oxide and the contaminated calcium fluoride from the deconversion process (discussed in Chapter IV) would have to be disposed of as RCRA waste.[109] Considering this range of serious concerns, it is not plausible to consider shallow land burial as a possible disposal strategy for the depleted uranium that would be generated by the proposed LES enrichment facility.

[108] Hertzler et al. 1994 p. iv-v
[109] Hertzler et al. 1994 p. v and 56

## III. Some Considerations Relative to the Environmental Impacts of $DUF_6$ Deconversion and Depleted Uranium Disposal

The specific steps in the deconversion of $DUF_6$ to a more stable chemical form for long-term disposal depends on the final choice for which potential form is to be produced. In particular, the steps for producing uranium oxide ($U_3O_8$) or uranium dioxide ($UO_2$) are different and result in different impacts such as the level of contamination in the resulting hydrofluoric acid or calcium fluoride. The choice of disposal strategy will have a significant impact on the choice of which deconversion process is to be pursued.

LES has stated that its preferred option is the deconversion of the $DUF_6$ to $DU_3O_8$ followed by its disposal as a bulk powder in an abandoned mine or potentially at a shallow land disposal facility. In addition, LES has stated that it will consider the following reactions for producing the $DU_3O_8$

$$UF_6 + 2\ H_2O \rightarrow UO_2F_2 + 4\ HF$$

followed by

$$3\ UO_2F_2 + H_2 + 2\ H_2O \rightarrow U_3O_8 + 6\ HF$$

On the other hand, depleted uranium hexafluoride may also be converted into $UO_2$ instead by the following reactions

$$UF_6 + 2\ H_2O \rightarrow UO_2F_2 + 4\ HF$$

followed by

$$UO_2F_2 + H_2 \rightarrow UO_2 + 2\ HF$$

The choice of deconversion process that is to be pursued involves important trade-offs that require additional analysis by LES and the NRC. As we have discussed in Chapter II, the depleted uranium that would be produced as a result of the proposed LES enrichment facility is analogous to transuranic waste and, if ultimately declared a waste by the Commission, will likely require fabrication into a suitable waste form and disposal in a mined repository such as the Waste Isolation Pilot Plant. The $DU_3O_8$ that would result from the first deconversion process shown above would be less dense and less uniform in particle size than the $DUO_2$ that would result from the second process. These properties make it less suitable for processing into a waste form that would aid in the development of a disposal strategy protective of the public health and capable of meeting the existing regulatory limits for uranium exposure. On the other hand, the smaller more uniform particle size of the $DUO_2$ that is an advantage in waste form processing also adds to the level of uranium contamination in the resulting byproducts (i.e. the hydrofluoric acid and the calcium fluoride that would result from neutralizing the HF) as well as adding to the airborne releases of uranium from the process building stack of the deconversion facility. The estimated stack releases of uranium for a $DUO_2$ facility are more than three and a half times those of a $DU_3O_8$ facility.[110]

A possible waste form that should be examined for the ultimate disposition of depleted uranium is the encapsulation of $DUO_2$ in an engineered ceramic that locks up the material on the atomic scale and has been demonstrated to have a very low leach rate. An example of such a waste for would

[110] LLNL 1997 (EA) p. 6.4-7-2, 6.5-7-2, 6.6-7-2, and 6.7-7-2

be Synroc or an equivalent titanate ceramic as has been proposed for the immobilization of high level waste as well as for plutonium waste. Potential unknowns surrounding this option include the fact that little industrial experience exists with these ceramic materials and the experience that does exist is for a relatively low throughput facility.[111] In considering the impacts that this type of waste form preparation would have on the mobility of the depleted uranium, and thus on the peak doses that would be expected, the analysis needs to also examine the environmental impacts that would accompany the mining and processing of mineral sands in sufficient quantities to manufacture the large amounts of ceramic material needed for the disposal of such a large quantity of depleted uranium as that which would be generated by the proposed LES facility.

Returning to the impacts of the deconversion facility, the generation of hydrofluoric acid (HF) in large amounts from either process would result in an exhaust gases that are highly acidic and chemically hazardous if sufficiently concentrated. Therefore, a scrubber system is proposed to remove most of the HF that will be produced during routine operations. According to engineering analysis performed by Lawrence Livermore National Laboratory for depleted uranium deconversion facilities, the proposed type of scrubber would be able to remove up to 99.9 percent of the HF from the exhaust gases. A consideration of the impacts for lower filter efficiency should be included in the assessment of the routine impacts of the deconversion facility. Low scrubber efficiency was frequently experienced in the scrap recovery operations at the uranium plant near Fernald, Ohio for instance.[112] The estimated composition of the exhaust gases under four scenarios as presented in the LLNL engineering analysis is shown in Table 6.

*Table 6: Estimated concentration of hydrofluoric acid and uranium oxide in the exhasust gas from the process building under a variety of assumptions regarding the chemical form of the uranium oxide and whether the HF is neutralized with lye to calcium fluoride or processed for resale as anhydrous HF. The implied uranium concentration in the HF is calculated assuming 99.9% of the HF is removed by the scrubber prior to release at the stack and that no uranium oxide is removed in that process.*

| Scenario | HF pounds per year emitted after scrubbing | Pounds of uranium oxide per year in scrubbed exhaust | Implied contamination of the HF, ppm of U |
|---|---|---|---|
| $U_3O_8$ with HF sale[(a)] | 900 | 3.3 | 3.1 |
| $U_3O_8$ without HF sale[(b)] | 300 | 3.3 | --- |
| $UO_2$ with HF sale[(c)] | 900 | 12 | 11.7 |
| $UO_2$ without HF sale[(d)] | 300 | 12 | --- |

Notes:
(a) LLNL 1997 (EA) p. 6.4-7-2
(b) LLNL 1997 (EA) p. 6.5-7-2
(c) LLNL 1997 (EA) p. 6.6-7-2
(d) LLNL 1997 (EA) p. 6.7-7-2

[111] LLNL Wilt 1997 p. 11
[112] Viollequé et al. 1995, Appendix I. See especially Table I-10 through I-13, which indicate highly variable scrubber performance, ranging from better than manufacturer specifications to nearly complete failure of scrubbers. Sodium hydroxide was the srub fluid. Thus, even if a 99.9 percent efficiency scrubber is installed, maintaining the efficiency at such a high level would be difficult and expensive due to the corrosive nature of HF.

These releases correspond to annual airborne emissions of approximately 0.51 to 1.9 millicuries of uranium under routine operation. A private conversion facility built to handle the smaller amount of depleted uranium that would be generated at the proposed LES enrichment facility in comparison to the DOE stockpile which formed the basis of the LLNL analysis would be expected to have proportionally lower absolute levels of these emissions assuming the same scrubber efficiencies. In either case, the amount of uranium in the inlet gases (and thus in the hydrofluoric acid that would have to be managed), however, is likely to be greater than that in the exhaust gases which will have passed through HF scrubbers involving a potassium hydroxide (KOH) slurry.[113] This fact must be considered when analyzing the potential fate of the highly corrosive and dangerous hydrofluoric acid that results from the deconversion process.

Currently there are no DOE or general NRC guidelines that govern the free release of contaminated hydrofluoric acid or calcium fluoride.[114] The NRC has granted a license to the Framatome Advanced Nuclear Power, Inc. uranium fuel fabrication facility in Richland, Washington for the release of HF containing up to 6.4 ppm of uranium and the European limit for release of HF from the Cogema Pierrelatte deconversion plant is 5 ppm.[115] The cost analysis of a uranium deconversion plant intended to process the DOE's stockpile of $DUF_6$ conducted by Lawrence Livermore National Laboratory concluded, however, that

> In addition to the uncertain market, there is concern about possible public reaction to uranium contaminants. If the fluorine chemical is to be sold in North America, it may be subjected to higher purity standards due to the source material.[116]

The implied uranium concentrations of uranium in the hydrofluoric acid given in Table 6 assume that no uranium oxide was removed by the HF scrubber and, therefore, the actual total contamination of the acid is likely to be higher than these levels. Given the fact that the value for the $DU_3O_8$ facility is close to the existing U.S. and European benchmarks and the fact that the value for the $DUO_2$ facility is roughly twice as large, as well as the caution raised by the LLNL analysis regarding the potential for even tighter standards in the U.S. in the future, suggests that it should be assumed that the hydrofluoric acid resulting from the deconversion of the $DUF_6$ from the proposed LES facility will not be able to be resold on the open market.

One possibility for the use of this material that would not be hampered by the projected levels of contamination would be its reuse in manufacturing new $UF_6$ from natural uranium. However, in the present context this is not likely to be a plausible option for LES given the very large amounts of hydrofluoric acid that will be being produced by the government's deconversion facility for the DOE stockpile of depleted uranium. In particular, the suggested use of the HF by the uranium fuel facility in Metropolis, Illinois is not likely to be attractive given the proximity of the Paducah deconversion plant to be operating in nearby Paducah, KY. The Portsmouth deconversion plant in Piketon, OH which would also generate large amounts of HF is also much closer than the proposed LES facility in southeastern New Mexico. These facts were explicitly considered by the NRC and in the DEIS for the proposed LES facility when it concluded that CaF2 disposal was the only scenario that was reasonable to include in the DEIS:

[113] LLNL 1997 (EA) p. 6.4-7-7, 6.5-4-8, 6.6-4-8, 6.6-4-11, 6.7-4-10, and 6.7-4-13
[114] DOE Paducah ROD 2004 p. 44657 - 44658 and DOE Portsmouth ROD 2004 p. 44652 - 44653,
[115] DOE Paducah EIS 2004 p. E-13 and LLNL Cost Analysis 1997 p. 50-51
[116] LLNL Cost Analysis 1997 p, 50-51

> Because conversion of the large quantities of DUF6 at the DOE Portsmouth and Paducah Gaseous Diffusion Plant sites would be occurring at the same time the proposed NEF would be in operation, it is not certain that the market for hydrofluoric acid and calcium fluoride would allow for the economic reuse of the material generated by the proposed NEF. Therefore, only immediate neutralization of the hydrofluoric acid by conversion to calcium fluoride with disposal at a licensed low-level radioactive waste disposal facility is considered in this analysis.[117]

The potential need for disposing of the calcium fluoride ($CaF_2$) as LLW comes from the fact that it is expected to be contaminated by the presence of the uranium in the hydrofluoric acid.[118] From Table 6 these numbers would correspond to an implied contamination level of between approximately 0.62 and 2.3 pCi/gm depending upon whether $U_3O_8$ or $UO_2$ was the end product of deconversion.

Assuming that, other than the presence of uranium, the calcium fluoride can be considered non-hazardous waste, the contaminated $CaF_2$ would qualify as Class A low-level waste that could likely be disposed of in a suitable 10 CFR 61.55(a) facility. The treatment and disposal of this waste stream would add to the environmental impacts of the routine operation of the deconversion facility.

If any consideration is to be given by LES to the possible production and sale of anhydrous hydrofluoric acid for reuse, than an examination of this option's environmental impacts should also be carried out. In the analysis of proposals to construct and build the DOE deconversion facility it was determined that the accident scenarios with the largest consequences were primarily those involving hydrofluoric acid.[119] In considering the differences between the properties of aqueous HF and anhydrous HF, the EIS for the Paducah deconversion facility points out that

> It should be noted that there may be differences in the accident impacts between releases of AHF and aqueous HF, and that these differences were not fully evaluated in the critique… Anhydrous HF has a much higher volatility than aqueous HF, and therefore would result in a larger amount of material being dispersed to the environment if equal amounts were spilled. At this time, it is not clear if production of aqueous HF would result in a significant reduction in accident risk.[120]

In the same EIS, it was also reported that an accident involving a railcar in an urban setting under unfavorable weather conditions could potentially cause irreversible damage to people within an area covering seven square miles downwind with up to 300 casualties. For comparison, this is an area roughly one-fifth of the size of Santa Fe, New Mexico. The DOE analysis goes on to conclude that, "[a]s noted above, shipment of aqueous HF may have different risks than shipment of AHF."[121]

If the preferred option of neutralizing the HF and disposing of the calcium fluoride as LLW is replaced by a decision to produce and shipment of anhydrous HF, the potential impacts on the environment are likely to be higher. However, given that no existing facility for $UF_6$ deconversion currently produces AHF, the fact that the cumulative transportation distances considered for the DOE facilities are different from those that may be required for shipping the material generated by

[117] NRC NEF EIS Draft 2004 p. 2-29
[118] Paducah EIS p. E-5
[119] Paducah EIS from Appendix D page 18-19
[120] Paducah EIS from Appendix D page 19
[121] Paducah EIS from Appendix D page 20

the proposed LES facility[122], as well as the fact that the health and environmental impacts on routine operation from the greater volatility and general hazards posed by anhydrous HF versus aqueous HF were not analyzed by the DOE EIS for the Paducah or Portsmouth facilities cited by the NRC in the LES DEIS analysis[123], it is not possible at this time for us to quantify the potential impacts of such a decision.

Finally, as noted in the introduction, the management and long-term disposal of the depleted uranium must be considered as an integral part of the environmental impact of operating the deconversion facility. We have already addressed many of the environmental and health effects of DU disposal in Chapters II and III and will discuss their impacts on the cost of the waste management strategy in the following chapter. In summary, the disposal of depleted uranium through shallow land burial is extremely unlikely to be able to satisfy health and safety standards even under arid conditions and the disposal of depleted uranium in a deep repository should proceed under the assumption that DU is at least as risky as TRU waste at the 100 nCi/gm threshold and that the DU must therefore be disposed of with a similar level of care in order to minimize the long-term impacts. This means that, as Dr. Bredehoeft summed up (see full text above),

> It would be prudent to assume that, before a site could be qualified to receive depleted uranium waste, a similar amount of time, effort, expense, and scrutiny to that which went to qualify WIPP would be required.

[122] Currently no commercial deconversion facility exists in the U.S. that would be able to accept the DUF6 from the proposed LES enrichment facility and thus no quantification of this potential impact was attempted.
[123] NRC NEF EIS Draft 2004 p. 2-30

# IV. $DUF_6$ Deconversion and Depleted Uranium Disposal Costs

In this chapter, we first outline the LES approach to estimating the costs of deconversion and disposal of the DUF6 that would be generated by the proposed enrichment facility. We consider the LES estimate to be unrealistically low an several counts and present their estimate only for reference. Finally, we then develop three alternate deconversion and disposal scenarios that we conclude to be more reasonable projects. The three scenarios that we propose are :

1. The $DUF_6$ is converted to $DUO_2$, grouted in cement, and then disposed of in a mine that is half the size of the mine considered for the depleted uranium in the DOE stockpile. The hydrofluoric acid generated would be neutralized into calcium fluoride ($CaF_2$) which would then be disposed of as low-level waste at an appropriate site.
2. The $DUF_6$ is converted to $DUO_2$, put into a ceramic type of waste form, and then disposed of in a deep geologic repository similar to the Waste Isolation Pilot Plant (WIPP) in New Mexico. As in Case 1, the hydrofluoric acid generated would be neutralized into calcium fluoride ($CaF_2$) which would then be disposed of as low-level waste at an appropriate site. This case also includes a contingency for the added costs that should be prepared for in light of the evidence of potential risks of depleted uranium emerging for recent research.
3. A variation of Case 2 above with different assumptions regarding the costs of $CaF_2$ storage and disposal as well as about the costs of operating the deep geologic disposal site.

It is important to note that our highest cost estimate (Case 3) is not intended to be a worst case analysis and represents our conclusions regarding a realistic scenario that would be prudent to considered given the uncertainties involved in planning for the handling and eventual disposal of such large quantities of depleted uranium over the next three decades in light of the issues highlighted throughout Chapters II, III, and IV.

The cost analysis conducted by the Lawrence Livermore National Laboratory in support of the DOE program to manage its stockpiles of depleted uranium does not provide ranges from the costs within each scenario. It covers some elements of uncertainty by developing a variety of scenarios built on different assumptions for model input parameters. Some crucial uncertainties such as exchange rate risks and emerging science concerning uranium health risks are not taken into account in any way in the LLNL analysis and their database for deconversion is quite old relative to more recent confidential cost data. In this chapter we will use a small amount of information from the LLNL cost analysis for reference, but we have found it an inadequate basis for developing the uncertainties and contingencies discussed below.

As discussed, in all of the deconversion / disposal cases we analyze in this chapter, we have developed contingency cost estimates relating to various foreseeable and unforeseeable uncertainties that surround this type of operation. The six items that we consider are:

1. Production of anhydrous hydrofluoric acid (AHF) instead of aqueous HF.
2. Scaling issues
3. Conversion to UO2 instead of U3O8
4. Exchange rate uncertainties

5. Uranium risks relating to existing ICRP and EPA guidance as well as to the emerging results of research on DU
6. Licensing uncertainties and unforeseen circumstances

We will consider each of these contingencies in turn below.

Before turning to our discussion of the uncertainties and the associated cost contingencies, however, we note that we will not consider the possibility of using the DOE deconversion facilities built to manage the government stockpiles of DU at the Portsmouth, Paducah, and Oak Ridge complexes in the present chapter. This is supported by the fact that, as we will discuss below, we do not consider a reliance on the Department of Energy to be a plausible strategy for dealing with the depleted uranium tails that would be generated by the proposed LES enrichment plant. In addition, LES has publicly stated that

> For many reasons, including the large volume of byproduct already in storage in the US, the DOE deconversion facilities are not LES's path of choice for byproduct deconversion. LES has continually supported the development of a commercial, private deconversion facility. In fact, the company will seek to develop long-term supply contracts with potential deconversion operators in order to assist in their financing and licensing efforts to build such a facility.[124]

From these considerations, we concluded that a detailed economic analysis of the DOE option was not warranted in the present context.

**1. AHF Production**

There is at present no commercial depleted uranium deconversion facility that produces anhydrous hydrofluoric acid (AHF) as noted above. In fact, there is currently only a single large-scale facility in Western Europe or the United States that has been operating from some time – the Cogema facility at Pierrelatte. The references in LES's literature to AHF do not contain specific cost estimates; nor can such costs be derived from actual experience of deconversion plants. We have already pointed out that the production and transport of AHF vs aqueous HF or neutralized $CaF_2$ are all likely to have different operational impacts and potential safety concerns, and these difference would likely effect the lifetime cost of the facilities as well. We will not attempt to quantify the additional costs that might be associated with the production of AHF directly in this report. According to the analysis done by the Lawrence Livermore National Laboratory (LLNL) the costs of deconversion with AHF production are comparable to the costs of neutralizing the HF into $CaF_2$ if no credit is taken for the sale of the AHF or $CaF_2$[125] although the LLNL engineering analysis of these plants does note that

> any uncertainties with the specific distillation process and its integration assumed for the engineering analysis (see 3.2.1.1) would be addressed in a subsequent engineering development phase of the Program.[126]

In all three scenarios constructed by IEER, we assume that the HF will be neutralized to $CaF_2$ and disposed of as low-level radioactive waste. As noted above, this is in line with the conclusion of

[124] LES NEF UF6 info sheet p. 3
[125] LLNL Cost Analysis 1997, p. 52
[126] LLNL 1997 (EA) p. 3-7

the NRC in this regard.[127] In the LES assumptions case presented for reference only, we include the LES assumption that aqueous HF will be generated and sold (or used internally to the nuclear fuel cycle for the production of new $UF_6$ from natural uranium), as is done in the case of the HF generated at Cogema's Pierrelatte plant.[128] AHF production would increase costs relative to the LES assumptions case, for this and other reasons discussed throughout this report this reference case must therefore be regarded as the minimum cost case, based on information and analyses available to date.

**2. Scaling Issues**

There are two issues of cost related to the scale of operations: the scale of the deconversion plant and that of the disposal facility. Cogema's Pierrelatte plant has a nominal processing capacity of 20,000 metric tons of $DUF_6$ per year. Between 2000 and 2003 the plant was operating at an average capacity factor of nearly 92%.[129] This scale is roughly comparable to that considered as the base case in the analyses of cost done by the LLNL, which was determined to correspond approximately to the requirements of being able to completely deconvert the accumulated depleted uranium stockpile at DOE facilities 20 years.[130]

The LLNL analysis indicates that the costs per kilogram of material processed may increase significantly if the scale of either the deconversion plant or the disposal facility is reduced to 50 percent of the base case and even more dramatically if it is reduced to 25 percent of the base case. The enrichment facility proposed by LES would generate up to 7,800 metric tons of $DUF_6$ per year during operation, which is less than 28 percent of the LLNL base case and less than 40 percent of the capacity of the Cogema Pierrelatte plant.[131] The LLNL analysis estimated that the added unit cost of a facility producing $DUO_2$ scaled to handle 50 percent the $DUF_6$ would be about $0.83 per kg U and that of a 25 percent facility would be about $2.37 per kg U as compared to the base case full scale facility.[132]

**3. Conversion to $DUO_2$ instead of $DU_3O_8$**

Uranium dioxide, because of its smaller and more uniform particle size, is a suitable chemical form of uranium for grouting or for conversion to various kinds of ceramic waste forms, such as Synroc or zircons. As we have discussed above, generic analysis indicate that the disposal of DU in the form of $DU_3O_8$ powder through near surface disposal would produce doses far in excess of the regulatory limit of 25 mrem per year and therefore that disposal in a deep geologic repository similar in containment to WIPP will likely be required. Processing of $DUF_6$ into $DUO_2$ and then

[127] NRC NEF EIS Draft 2004 p. 2-29 to 2-30
[128] LLNL Cost Analysis 1997 p. 50-51
[129] ASN 2000 p. 288, ASN 2001 p. 300, ASN 2002 p. 370, and ASN 2003 p. 374
[130] The plant size analyzed in the Livermore cost study was 28,000 tons of $UF_6$ per year. (LLNL Cost Analysis 1997 p. 2) Note that we have not used LLNL's estimates of conversion costs since more recent, contract-based estimates from Urenco are available. See section below on LES cost estimates.
[131] NRC NEF EIS Draft 2004 p. 2-15
[132] LLNL Cost Analysis 1997 p. 100. All numbers are rounded as indicated. Costs are in 2002 dollars unless otherwise stated. GDP deflators are used to convert costs.

converting this into a ceramic waste form, or at a minimum into grout is likely to be needed to meet dose requirements over the long timescales necessary for consideration, whatever the eventual status of the classification of DU, unless a specific site is located and sufficiently well characterized to show that $DU_3O_8$ could meet regulatory dose criteria (see discussion in Chapters II and III). The added costs of conversion of $DUF_6$ to $DUO_2$ are estimated from the information in the LLNL study for suitably sized plants to be about $[redacted] per kg U and are included in each of the three scenarios developed by IEER for this report.

There are, however, additional uncertainties in regard to the costs of converting to $DUO_2$ in lieu of $DU_3O_8$ as noted in the Livermore engineering analysis. As an example of a possible unknown relating to this option the authors note that despite this type of conversion process being common in reactor fuel fabrication facilities

> Due to the fact that the oxide [UO2] throughput is an order of magnitude higher than that for nuclear fuel fabrication plants, the preconceptual design assumes much larger sintering furnaces than those used in commercial fuel fabrication plants. Furnaces of this size and with these performance specifications are not presently available, but furnaces with one or two of the features (high capacity, high temperature, and special gas atmosphere) are common. It is believed that sintering furnaces combining all of these features can be engineered and fabricated with moderate risks.[133]

These and other potential risks concerning the cost of pursuing a plant design that deviates in significant ways from the design of the operating Cogema Pierrelatte plant must be considered in the overall economic analysis.

**4. Exchange rate uncertainties**

We use a base case of [redacted] euros per kg U in this report for the cost of conversion of $DUF_6$ to $DU_3O_8$ and its subsequent transport and storage. This is the amount that Urenco is paying Cogema to convert [redacted] metric tons per year of $DUF_6$ to $DU_3O_8$ per year, put the oxide into cylinders, transport it to Holland, and store it there. No disposal cost is included in the [redacted] euros per kilogram since the European Union has no disposal facility for DU. (Some depleted uranium from Europe is also sent to Russia for "re-enrichment." The costs of this arrangement are not transparent since the enrichment of the DU as well as the costs of conversion and disposal of the DU remaining after re-enrichment are unknown.)

The Urenco-Cogema contract is an experimental contract for five years. The [redacted] euro estimate does not include a correction for future escalation, although there is reported to be an escalator in the contract that will be applied periodically throughout the timeframe contract. Hence the [redacted] euros per kg U is, in effect, a present value of conversion costs. This is the most reliable cost estimate to date since it is the one cost estimate that is based on a contract with an operating facility in which $DUF_6$ has actually changed hands and been processed.[134] We have used it as the base case figure for this study because it is a contract that Urenco is now a party to and Urenco is the main company in the LES consortium. The [redacted] euros per kg U includes two components.

[133] LLNL 1997 (EA) p. 3-11

[134] The Uranium Disposition Services contract with the DOE to build and operated the deconversion facilities for Paducah and Portsmouth over the next six years is not based on operating experience and it does not include a provision for the cost of ultimate disposal of the $DU_3O_8$, only storage until the contract is re-evaluated in 2010. (DOE Paducah ROD 2004 p. 44657 – 44658 and DOE Portsmouth ROD 2004 p. 44652 – 44653)

The conversion component is about [redacted] euros per kg U and the rest is for transportation to and storage in Holland.[135] This is also compatible with an estimate provided by Cogema to LES for conversion only of about [redacted] euros per kg.[136]

Since Cogema is the only company with significant deconversion experience, and since this same company has figured prominently in LES statements about costs of deconversion including their public acknowledgement that "[d]iscussions have recently been held with Cogema concerning a private deconversion facility",[137] it is necessary to take account of exchange rate uncertainties in the overall economic analysis. In 2002, an assumption of one euro per dollar (as assumed by LES) appeared reasonable. This conversion is also approximately equivalent to the purchasing power parity between the dollar and euro. However, exchange rates in the real world are not set according to purchasing power parity but according to capital flows, current account deficits, relative interest rates, and other factors. The euro to U.S. dollar exchange rate has been very volatile ever since the euro was introduced and has ranged from a low of 0.82 dollars per euro to 1.30 dollars per euro, which is approximately the rate at the time of this writing (24 November 2004).

Persistent large U.S. current account deficits are expected to create further down pressure on the dollar. The analysis attributed in a recent article from the New York Times to officials at the International Monetary Fund concerning the future of the dollar in relation to the euro is worth quoting at length:

> But a third school, which includes officials at the International Monetary Fund, worries about a collapse in the dollar that would send shock waves through the global economy.
>
> That group argues that the dollar needs to depreciate another 20 percent against the other major currencies but warns about a run on the dollar that could reduce its value by 40 percent.
>
> A collapse of that size would severely affect Europe and Asia, which have relied heavily on exports to the United States for their growth.
>
> A steep drop in the dollar could lead to higher interest rates for the federal government and American private borrowers, as foreign investors demanded higher returns to compensate for higher risk. And it could expose hidden weaknesses among financial institutions and hedge funds caught unprepared.[138]

These concerns regarding the future strength of the dollar can be found in more formal (and hence restrained) language in the International Monetary Fund's World Economic Outlook from April 2004. In this report, the IMF officially and publicly concludes that

> In addition, the prospect of a higher foreign debt in the future can weaken the exchange rate even in the short term if it diminishes foreign investors' appetite for U.S. assets. As discussed in Chapter I, the prospect of continuing large U.S. fiscal and external deficits and the implied external borrowing adds to concerns about international imbalances, increasing the chances of a disorderly resolution, including a rapid fall in the value of the dollar and a rise in U.S. long-term interest rates.[139]

As a final example, both Alan Greenspan, the chairman of the Federal Reserve, and John Snow, the U.S. Treasury Secretary, have publicly warned members of the international financial

[135] Deposition Chater et al. 2004/10/04, pp. 23-25.
[136] LES Business Study 2004, pp. 13-14. A range of [redacted] to [redacted] euros per kg U is given in this study for conversion costs only.
[137] LES NEF UF6 info sheet p. 3
[138] NYT November 16, 2004
[139] IMF April 2004 p. 73

community to expect little relief from the declining position of the dollar relative to the euro and other major international currencies.[140]

As noted, LES has converted [redacted] euros to [redacted] dollars at a one-to-one exchange rate. We have used this as a base case estimate, but added an exchange rate contingency on deconversion of between 10 percent and 30 percent. Since the dollar may fall to $1.50 or even $1.80 per euro (a 20 to 40 percent fall relative to $1.30 per euro), as implied by some of the above commentary, the contingencies that we have adopted in the IEER scenarios are modest compared to the actual uncertainties in the financial world. These larger losses by the dollar relative to the euro could add a few dollars per kg U to the cost of conversion depending on the nature of the contract and whether the contract is actually with a European corporation directly, or if a European corporation is going to fully or partially supply technology or licenses only.

If the LES deconversion contract is with a U.S. corporation, then the exchange rate contingency would only apply to the portion of conversion costs that deal with acquisition of technology or equipment in Europe. A decline in the dollar could also lead to higher interest rates in the United States as noted above. This would mean far higher U.S. costs for all companies, along with depressed demand. We have addressed this uncertainty by using three different deflators, 2%, 4%, and 7% to estimate costs per SWU of deconversion and disposal. These discount costs would apply if the DU deconversion costs were to be recovered from customers as the enriched uranium is sold. They would not apply if a cash guarantee (or financial equivalent) is provided upfront (see the discussion below). Finally, we note that this does not cover the full range of interest rate and inflation risks, however, if the crisis of 1979-81 is taken as a point of comparison. At the peak of that crisis, the U.S. prime interest rate rose to 21.5 percent (December 19, 1980) and the Federal Reserve raised the rediscount rate to a record 14 percent on May 4, 1981.[141] This rise stabilized the dollar, but helped precipitate a widespread debt crisis and trigger a severe recession in the United States.

**5. Uranium risks**

We have discussed the established risks of uranium as well as concerns and risks indicated by current research that are far more wide ranging than the current assumptions recognize. The new research, much of which has been done in the context of attempting to understand the ill-health suffered by veterans of the 1991 Persian Gulf War, indicates that in addition to other areas of concerns such as its impact on the skeleton, on reproductive success, and on cancer induction and/or promotion, uranium may also be functioning as a kind of radioactive lead in relation to neurological impacts. In all of these areas there is some indication of the potential for the heavy metal damage and the radiation-induced damage to be interacting in a synergistic manner. While we do not expect that regulations will be changed in the context of the LES application, the NRC must take into account the range of the health risks indicated by current research, especially as they are far more wide-ranging than health risks as presently estimated. Financial contingency provisions to address these emerging risks are a minimum action that should be taken in the

[140] NYT November 19, 2004
[141] Harry Cleaver and Joshua Freeze, Chronology of International Monetary Affairs, http://www.eco.utexas.edu/Homepages/Faculty/Cleaver/357Lmoneychrono.html

present case. The specific amounts will depend on the risks that are taken into account and the method for allocating a monetary contingency amount to them.

As noted above, Atomic Energy Act requires the NRC to protect public health. The lowest level of financial contingency provision relates to the existing reasonably well-established scientific conclusion that women are at a greater overall risk of cancer due to radiation exposure than men. This higher risk is due in large measure to the greater radiosensitivity of the female breast, but is not limited to just this one organ. These conclusions are drawn from the work that has been published by the EPA in Federal Guidance Report 13; "Cancer Risk Coefficients for Environmental Exposure to Radionuclides" from 1999.[142] FGR 13 is based in part on two publications of the International Committee on Radiological Protection; ICRP 60 from 1991 and ICRP 66 from 1994. While these ICRP publications are not yet the basis of NRC regulations, they do form the basis of the regulations that govern the implementation of the worker compensation law of 2000.[143] As noted above, standards governing the nuclear industry have been updated from time to time over the past several decades to reflect current understanding of risks. The licensing of facilities such as that proposed by LES should take note of this trend and include contingencies for this fact.

Another area where the difference between men and women in relation to radiation protection is becoming apparent can be found in the recommendations of the ICRP. The historical evolution of the tissue weighting factors assigned to the various organs between 1977 and the present by the ICRP are shown in Table 7.

*Table 7: Tissue weighting factors adopted by the International Commission on Radiological Protection in the two previous and current draft recommendations that have been issued over the past 30 years.*

| Organ | ICRP 26 (1977) [(a)] | ICRP 60 (1990) [(b)] | Draft ICRP 100 (2005) [(c)] |
|---|---|---|---|
| Gonad | 0.25 | 0.20 | 0.05 |
| Breast | 0.15 | 0.05 | 0.12 |
| Stomach | --- | 0.12 | 0.12 |
| Lung | --- | 0.12 | 0.12 |
| Thyroid | 0.03 | 0.05 | 0.05 |
| Bone Surface | 0.03 | 0.01 | 0.01 |

Notes:
(a) ICRP 26 p. 21
(b) ICRP 60 p. 86
(c) Draft ICRP 2005 p. 25

It is easy to see that the relative importance of the female breast compared to the gonads have increased between 1977 and the draft ICRP 100 publication by a factor of four. This factor of four is not yet reflected in NRC regulations. It is therefore reasonable to make a contingency provision that management and disposal of DU generated by LES will, within the several-decade period from the present, reflect evolving understanding of radiation risks at least to the minimal extent of

[142] EPA FGR 13 p. 179
[143] Federal Register 42 CFR Part 82, pages 22314 to 22336.

recognizing that women are at greater risk per unit of exposure and that it does not make biological sense to regulate to the average of men and women given this fact. We have added a contingency factor of about 19 percent to deconversion and disposal costs in the IEER scenarios where the costs of management and disposal are taken to be comparable to estimates based on the experience of WIPP (Case 2 and Case 3). This factor was derived from the excess risk to females per unit of exposure compared to the average of men and women that is the basis of current regulations. The corresponding contingency provision is over $[redacted] per kg U in both of the IEER cases.

Furthermore, new data and new understandings are emerging in areas whether little attention has been paid before. The need for fetal protection from heavy metal and radiotoxic materials has been highlighted by a number of national and international reviews. While the regulations that will govern the LES licensing process are those that are currently in effect, a contingency provision reflecting the possibility that uranium (including DU) may turn out to be viewed as analogous to a kind of radioactive lead, with a variety of highly deleterious effects, would also seem prudent. The NRC's DEIS is seriously deficient in not having mentioned the existing evidence pointing to such health risks, let alone evaluating them in the context of DU management and disposal.

Given this failure of the DEIS to consider the results of recent research on DU risks, it is not possible within the time and resources available for this study to make an estimate of their potential environmental and health damage, and therefore of the potential costs of DU disposal. It is quite possible that if the large quantities of DU anticipated to be generated from the LES facility is found to be deleterious in some or all of the ways discussed in Chapter I, that the cost of disposal may be significantly higher than estimated the highest estimates provided here. The difficulties of estimating costs in case children in the future are saddled with a legacy similar to that of the sorry history of lead poisoning over the past three generations, but this time a kind of radioactive lead, are clearly evident. Nonetheless, the NRC is bound under the Atomic Energy Act to analyze these potential health and environmental effects to the extent that the published literature allows. We note in this context, that an argument that the amounts of DU expected to reach the human environment as implied by the dose estimates presented in the LES DEIS and the CEC FEIS are not technically credible, as we have already discussed. Site specific analysis as well as a publication of all the details of the NRC dose calculations in the DEIS as well as the earlier FEIS in the CEC case is essential before any argument regarding disposal risks (and thus costs) can be made on this basis.

**6. Licensing uncertainties and unforeseen circumstances**

Licensing of nuclear facilities is a very uncertain process as the past forty years has starkly demonstrated. The experience of LES in the CEC case illustrates this uncertainty, since in that case LES expended large sums of money and then decided to withdraw the license application. There is the risk that this kind of licensing problem may occur in the case of the application to construct and operate a private deconversion facility, even if the uranium enrichment facility is granted a license. In addition, the mined repository that would be planned to accept the depleted uranium if it is ultimate declared a waste would also face serious uncertainties in regards to its licensing, particularly in light of the long-lived nature of the uranium isotopes and the fact that no

disposal facility for large quantities of DU have been opened anywhere in the world. The Lawrence Livermore engineering analysis also noted this potential regulatory uncertainty and concluded that

> The licensing of new low-level waste (LLW) disposal facilities under the AEA would be a major compliance issue. Licensing under the AEA by NRC or authorized states may be difficult due to the extensive regulatory requirements and the inherently controversial nature of the subject. Approvals under the AEA by DOE for new LLW disposal facilities may be difficult due to extensive performance assessment requirements. Disposal facilities could potentially be required to comply with RCRA storage and permitting requirements if offsite treatment and disposal options for mixed waste continue to be limited.[144]

Hence, it is essential to make contingency provisions in relation to the costs of the deconversion facility as well as for finding and licensing a disposal site for the DU as well as for the contaminated $CaF_2$ at a low-level waste disposal facility.

The NRC requires the application of "a contingency factor of at least 25 percent to the sum of all estimated costs" as part of its decommissioning guidance.[145] The reason for making this provision is stated by the NRC as follows:

> Because of the uncertainty in contamination levels, waste disposal costs, and other costs associated with decommissioning, the cost estimate should apply a contingency factor of 25 percent to the sum of all estimated decommissioning costs. The 25 percent contingency factor provides reasonable assurance for *unforeseen* circumstances that could increase decommissioning costs, and should not be reduced or eliminated simply because foreseeable costs are low.[146]

At least in part, the cost items that we have discussed above are reasonably foreseeable, though we have not been able to quantify some aspects of these costs (as, for instance, with possible synergistic risks posed by the heavy metal and radioactive aspects of uranium).

LES has so far refused to acknowledge the necessity for making any provision for contingency costs in relation to their estimate for the cost of deconverting the $DUF_6$ of $[redacted] per kg U, which they claim is based on the existing Urenco contract with Cogema. The costs of DU management and disposal could easily be in excess of those discussed in this report for a variety of reasons ranging from heightened uranium risks to Federal Reserve interest policies to the lack of industrial experience manufacturing ceramic waste forms for DU. The range of these possible but uncertain impacts have not been fully captured in the contingency costs that we have otherwise quantified. As another example, we have not considered the potentially dramatic cost increases of licensing delays such as those experienced by WIPP, the construction of which was essentially complete in 1988 but which took about a decade longer to be put into operation with a concurrent escalation in capital costs. The emerging risks of DU that are not factored into present regulations also increase the risk of licensing delays and/or of rising costs after the start of operation, because of increasing public demands for better health protection. The quantified contingency for uranium risks in the costs presented in this report includes only those possible costs relating to the higher risks faced by women compared to men which are already published in EPA guidance documents. The far higher risks for children are not factored into the analysis here. If the same calculational procedure used with regard the risk to women were applied to the greater risk of children

[144] LLNL 1997 (EA) p. 2-13
[145] NUREG-1757, Vol. 3, p. 4-10.
[146] NUREG-1757, Vol. 3, Appendix A, p. A-29, emphasis in the original.

compared to adults as published in the 2002 supplement to EPA FGR 13, it would increase the cost by several more dollars per kilogram of uranium.

Hence, we have retained the NRC contingency factor of 25 percent despite that fact that we have quantified some of the anticipated uncertainties in the three scenarios. This is because the NRC guidance explicitly and emphatically states that the 25 percent provision relates to "*unforeseen* circumstances." The potential for uranium to be viewed as radioactive lead would fall into this category today in terms of a financial risk provision since the future of regulation in this regard cannot be foreseen. Nor can we adequately forecast the future evolution of global monetary crisis that is developing. It is our view, for the reasons discussed above, that the range of costs in the IEER scenarios does not capture the range of uncertainties, making a contingency provision for unforeseen costs necessary. Finally, while the contingency factor of 25 percent for unforeseen costs is justified, we also discuss costs for the three IEER scenarios without this contingency provision.

**Existence of a plausible strategy?**

LES has described two options for deconversion and disposal of the $DUF_6$ tails that would be produced by the proposed enrichment facility – (1) the use of the DOE facilities once the government's stockpile of $DUF_6$ has been deconverted and (2) the construction of a private deconversion plant to handle just the volume of DUF6 to be produced by the proposed LES enrichment plant.[147] We will consider whether these options as presented in the DEIS in detail below.

In relation to the definition of what a plausible strategy entails for the disposition of the depleted uranium from an enrichment facility, the NRC has ruled

> Thus, in assessing the plausible tails disposal strategy adopted by the Applicant as part of its decommissioning funding plan, we must first determine whether the funding plan contains a reasonable or credible plan to dispose of the $DUF_6$ tails generated at the CEC and then determine whether the Applicant's cost estimates for the components of the plan are reasonable.[148]

The analysis we will present in this section regarding the reasonableness and credibility of the proposed disposition strategies as well as our cost estimates in the following section are offered in this vein. From this work we have concluded that neither option as stated in the DEIS can be considered to be a plausible strategy for the disposition of the depleted uranium tails that would result from operation of the proposed enrichment facility.

*The DOE Option*

One option claimed by LES for the disposition of its $DUF_6$ would be to send it to the DOE on the grounds that the DOE would be required to take the DU under the law covering privatization of the Paducah and Portsmouth uranium enrichment plants. However, this provision in the law presumes that DU has been declared a waste (which it has not) and that a payment be made to the

[147] NRC NEF EIS Draft 2004 p. 2-28
[148]

DOE for taking the waste for management and disposal. LES has not provided any documentation that DOE is ready to take the depleted uranium from the proposed enrichment facility or at what cost if it was so ready. As noted, the cost figures in the UDS contract to build and operate the Portsmouth and Paducah deconversion facilities do not cover the disposal of the DU since DOE is reserving the option of treating it as a source material and the contract is only for the first few years of operation. Further, as we have shown, shallow land disposal of DU even in a dry environment is highly unlikely to meet the dose limit criterion of 25 mrem per year, indicating that this option remains unviable today, as it was judged to be by the NRC in the CEC case a decade ago.

Further, even a formal commitment by the DOE to take the depleted uranium from the proposed LES facility pursuant to the privatization law would not be very meaningful or credible, given DOE's poor track record in regards to such commitments. The DOE has broken its contracts with nuclear electric utilities to take the spent fuel from nuclear power plants beginning in 1998 even though these contracts were signed pursuant to the 1982 Nuclear Waste Policy Act. What is worse, the utilities have paid billions of dollars in fees to the federal government so that this commitment could be met and so far no spent fuel has yet been transferred and the DOE refused to acknowledge its legal liability when the 1998 deadline passed. The utilities have had to sue to recover damages, and the facts on the ground remain that the spent fuel is still at the power plant sites. Nearly seven years after the legal deadline to begin accepting the spent fuel, DOE has not yet even made an application for a license for Yucca Mountain. Legal uncertainties in regard to the licensing process concerning Yucca Mountain have grown in the past year, given that a portion of the EPA's disposal standard was invalidated in federal court. Whether DOE can get a license to dispose of spent fuel at Yucca Mountain remains an open question more than two decades after the passage of the NWPA and after utilities started paying for its disposal.

DOE's performance in other areas is also far from stellar. The problems in regard to some of its environmental management programs were detailed in a study by the Institute for Energy and Environmental Research published in 1997.[149] At DOE request, IEER worked with DOE staff to help the department get a better understanding of its waste-related problems.[150] Yet the DOE environmental management and waste disposal program remains mired in controversy, delays, high costs, and changing standards.[151] These complications of the record of the DOE are compounded in the present case because DOE has a vast backlog of its own depleted uranium to process and dispose of that would likely leave the LES material at the back of a long line.

Given DOE's poor track record, notably in the Yucca Mountain case, the uncertainties arising from a DOE option are such that it could not be considered a credible or plausible strategy even if a written commitment could be obtained from the DOE to the effect that it would accept the DU from LES that would be generated by the proposed enrichment facility.

[149] Fioravanti and Makhijani 1997.
[150] A three-year long effort of joint DOE and IEER work followed the publication of the 1997 report. See DOE 1998, Huntoon July 2000, DOE June 2000, Makhijani October 2000, and the website of IEER at http://www.ieer.org/reports/cleanup/cln-supp.html and
[151] Fioravanti and Makhijani 1997, Makhijani and Boyd 2001, Makhijani and Boyd 2004, and Smith 2004.

*The Private Option*

LES has also postulated a private option for deconversion and disposal of its $DUF_6$. In view of the difficulties that are likely to be faced both in regard to deconversion but most particularly in regard to DU disposal, no strategy for DU management can be considered plausible until the following conditions are met:

- A firm and binding contract, with penalties for non-performance, comparable to the costs of deconversion, for the deconversion of $DUF_6$ to an oxide form by a specified date.
- A location for the deconversion plant and a commencement of the licensing process by the party committing to build it.
- A design of the deconversion plant that corresponds to a firm disposal strategy that has been approved by the NRC at the Commission level. This NRC approval is necessary because the end point of the deconversion depends on the final waste form of the DU and the disposal strategy. Specifically, whether the final form would be $U_3O_8$ or $UO_2$ and whether disposal would be as a powder, grout, or ceramic form would be needed for a design of the plant, even if all the processing did not take place there. For instance, processing into a zircon waste form would mean that $UO_2$ powder would be produced but it would not be compacted. By contrast, compaction would likely be required if the DU were to be disposed of as a powder.
- A specific, firm location for a DU disposal site that has a certified characterization and licensing process to assure compliance with the appropriate regulations and with the protection of the public health.

LES has initiated discussions with several private parties in regard to deconversion but none of the above conditions have yet been met. Cogema has provided a cost estimate of about [redacted] euros per kg U for deconversion alone (and it is noteworthy that it is euros). Other corporations have not yet provided estimates. No disposal or storage costs are included in these estimates and no specific deep disposal site has been firmly and adequately identified.

While the Urenco-Cogema contract for deconversion, transport and storage provides a reasonable starting point for cost estimates of deconversion and $DU_3O_8$ storage in theory, it does not amount to a plausible strategy in the specific instance of the proposed plant since it dose not take into account reasonable contingency costs and does not include considerations of the ultimate disposal of the depleted uranium.

There is also the need to consider the track record of private parties, in a manner analogous to the scrutiny given to the DOE above. For instance, if the technological track record of Cogema in Europe is relevant (and we agree that it is) then the legal and political track record is also relevant. A French parliamentarian and author of France's nuclear waste law, has criticized Cogema for trying to set itself up "above the law." Cogema has also tried to unilaterally redefine the meaning of zero discharges from its reprocessing plant at la Hague, flying in the face of all established radiation protection norms and regulations.[152]

Nor does the track record of BNFL, an LES partner, inspire great confidence. BNFL, which is owned by the British government, nonetheless shed tens of billions of pounds of waste-related cost

[152] Makhijani, Gunter, and Makhijani 2002.

liabilities in the lap of the British taxpayer because it was essentially insolvent so long as those liabilities were on its books.[153]

These factors need to be considered in the financial assessment as well as in the regulatory and environmental contingencies in that they speak to both the reasonableness of the proposed strategy as well as to its credibility. The NRC cannot assume that commitments in regard to DU deconversion and disposal will be met simply because corporations, some of which may be majority owned by foreign governments (as is the case with Cogema), make paper commitments to deconvert $DUF_6$.

The ultimate disposal of the depleted uranium presents even more difficult issues than does deconversion. As we have discussed, no credible environmental analysis can be done on a generic basis. We can make rough estimates as to possible costs by analogy with WIPP, but the uncertainties are great, given the delays and licensing difficulties faced by WIPP and given the emerging health concerns related to DU and the nature of public attitudes regarding DU in the wake of the growing recognition of Gulf War Syndrome. A plausible strategy necessarily includes identification of a specific site and a process for its thorough characterization and licensing, as well as a reasonable scientific expectation that it will be able to meet the established dose limits.

**Deconversion and DU disposal cost estimates**

We have developed three scenarios for costs relating to management and disposal of DU from the LES facility. We also present cost assumptions as set forth by LES, for the purpose of comparison. LES has given a variety of figures, as noted above. We have used [redacted] euros per kg U for deconversion and storage, for reasons already discussed. LES assumes 1 dollar = 1 euro, which is unrealistic, but we preserve this assumption, as also other LES assumptions, since the LES estimates are set forth for comparison only. We reject their numbers as a basis for determining the required financial guarantees needed for disposition of depleted uranium tails. We added $[redacted] per kg U for disposal, a low estimate for mine disposal developed by LES,[154] for a total cost of $[redacted] per kg U. [redacted]. LES assumes that it will not have to make any provision for contingency costs in relation to its use of $[redacted] per kilogram of uranium based on the Urenco-Cogema contract for deconversion, transport, and storage.

The assumptions for the three scenarios that IEER has developed are set forth in Table 8 below. Note that in all these scenarios, a 25 percent contingency for unforeseen circumstances is included. IEER's estimates of reasonable costs for DU disposal for the purposes of licensing are represented by Cases 2 and 3, which assume disposal of DU on a basis similar to WIPP, whatever its waste classification might be. The notation of the scenarios as GTCC simply refers to the physical, radiological, and risk properties of DU, as discussed in Chapter I.

[153] Bellona 2003. This reference contains links to various estimates of BNFL liabilities.
[154] ERI 2003 Draft, pp. 5-6.

*Table 8: Summary of the assumptions for the three IEER scenarios for DU management and disposal.*

| Scenario | Financial assumptions | U risk | Deconversion assumptions | Disposal assumptions | Comments |
|---|---|---|---|---|---|
| Case 1: grouted DU | 10 percent exchange rate risk for deconversion | none | $UO_2$, grouted | deep mine, [redacted]% of scale required for DOE DU | $UO_2$ deconversion costs higher but disposal costs lower, and overall disposal costs lower |
| Case 2: GTCC, WIPP | 30 percent exchange rate risk for deconversion | 19 percent increase in deconversion and disposal cost items | $UO_2$, ceramic waste form | equivalent to WIPP for DU, $CaF_2$ cost \$2/kg[(a)] | Low end of WIPP cost projections, low ceramic costs (\$2 per kg U), U risk corresponds to increased risk to females |
| Case 3: GTCC, WIPP | 30 percent exchange rate risk for deconversion | 19 percent increase in deconversion and disposal cost items | $UO_2$, ceramic waste form | equivalent to WIPP for DU, $CaF_2$ cost \$4//kg[(b)] | Medium estimate of WIPP cost projections, low ceramic costs (\$2 per kg U), U risk corresponds to increased risk to females |

Notes:

(a) LLNL Cost Analysis 1997 p. 119

(b) NAS-NRC 1996 p. 176

(c) Los and medium estimate WIPP costs are estimated as follows. Repository capital costs of \$350 million (about one-third of WIPP capital costs to 1988, not including costs of the decade-long delay in opening WIPP), since a smaller repository would be required and somewhat lower than the estimated capital costs of \$500 million estimated by LLNL for a full-scale mine for grouted disposal of DOE DU. Operating costs are for TRU waste disposal only, exclusive of other costs, such as transportation and waste characterization associated with disposal of TRU waste in WIPP. The low estimate is based on DOE projections of operating costs over the entire lifetime of WIPP and the medium estimate is based on the operating costs and planned disposal volumes to 2002. The actual disposal volumes were lower. Sources: LLNL Cost Analysis 1997, SRIC 2002, and DOE Waste Plan 2002.

*Discussion of some cost details*

We have used an estimate of only \$2 per kg U for production of the waste form from UO2. This is likely to be a low estimate. For instance, the proposed conversion of DUO2 to and aggregate form, called DUAGG, for radioactive waste casks is on the order of \$2 per kg U with using UO2 as feed.[155] This is the lowest order of magnitude cost for a DU waste form for disposal on a basis that would be equivalent to GTCC. Ceramic waste forms have been considered for surplus weapons plutonium. The Department of Energy estimates that the required investment to develop and build a plant designed to process 38.2 metric tons of plutonium into a waste form analogous to the titanate ceramic Synroc would be over \$500 million.[156] Since plutonium processing must be done in a glove box environment, it is far

[155] ORNL 2002, p. xiii.

[156] DOE 1996 p. ES-1 and 4-10

more expensive than DU processing. But even if the conversion to Synroc for depleted uranium cost $500 million in total for all 133,000 metric tons of DU (well over 3,000 times cheaper per kg than for plutonium), the costs per kilogram would still be greater than those assumed here and this number did not include the operational costs of actually immobilizing the waste which would add significantly to the result. Given that the technologies for producing Synroc or zircons are not yet commercialized, a firm estimate of costs is not possible at the present time. We have therefore used a very minimal estimate of $2 per kilogram total for preparation of the waste form. This assumption allows an estimate of the minimum financial guarantees that must be given by LES for DU deconversion and disposal.

Table 9 shows the cost estimates for converting and disposing of DU under the three IEER scenarios. The LES assumptions are shown for comparison in the left column. The total costs even under the LES assumptions, without exchange rate or any other contingencies and assuming disposal as a bulk $U_3O_8$ powder would be about $[redacted] million even if no NRC contingency is applied and it would be nearly $[redacted] million if the lower [redacted] percent contingency factor would be included. (All numbers in this text discussion are present value unless otherwise noted and have been rounded for convenience.) If the costs are recovered from customers, rather than paid up front, the charge per kg SWU would range from $[redacted] to $[redacted], depending on assumptions about the discount rate (2 to 7 percent) and the contingency factor applied (0 to [redacted] percent). Hence, even the LES assumptions case results in charges for DU disposition that represent a significant fraction of the recent market price for a SWU of $100 to $120.[157]

For a more realistic scenario, but still corresponding to a lax regulatory regime of allowing disposal of DU in grouted form in a mine, the estimated cost ranges from $29 to $50 per kg SWU. This estimate includes some contingencies for the exchange rate and for unforeseen risks as required by the NRC. The total present value of the costs, corresponding to the financial guarantees that would have to be given in this case would be about $1.8 billion. Without the 25 percent contingency required by the NRC, the total amount is only reduced to $1.5 billion. If this option had been analyzed with deconversion to $DU_3O_8$ as proposed by LES and then grouted for disposal in a mine, the cost would have been somewhat higher given the larger volume of the depleted uranium that would have to be placed in the mine, so the current result should be looked on a lower range for these assumptions regarding a more realistic but still lax regulatory regime.

In the two cases that DU is treated in a manner that respects the risks it poses, the costs range from $50 to $111 per kg SWU. The total present value of the financial guarantee to be posted would range from about $3.2 billion to $4.0 billion. Even without the 25 percent contingency required by the NRC for unforeseen circumstances, the total amount (present value) is estimated to still total $2.7 to $3.4 billion with an associated charge of $42 to $93 per SWU.

Costs corresponding to any of the three scenarios would likely make the proposed plant uneconomical, increasing the likelihood of default on deconversion and disposal obligations. This makes it imperative that financial guarantees are required to be posted up front as part of

[157] The price per SWU is variable. $100 to $120 is a reasonable range. The spot price per SWU in the summer of 2004 was about $110 per SWU. UX Weekly 2004, p. 3 at http://www.uxc.com/products/UxW18-41.pdf.

the conditions imposed for granting a license. In this analysis, a cash equivalent guarantee of about $1 billion to $4 billion can be justified, depending on assumptions about deconversion and disposal of DU and the associated uncertainties therein. Given the known risks of DU and its radiological and chemical characteristics, a financial guarantee of at least $2.5 billion (present value) would be prudent if offered on a basis that could be readily encashed by the government in case of corporate default on waste management and disposal liabilities. This value is the lower IEER estimate rounded down to the nearest $500 million without taking into account the addition of the NRC contingency factor. Given the risk of high interest rates in the future and the associated economic turmoil, there should be an adequate escalation clause for the financial guarantee that is linked to Federal Reserve rediscount rate.

A guarantee based on the recovery of cost of deconversion and disposal of the depleted uranium tails from customers would be too risky given the large amounts of money that will have to be recovered relative to the price of a SWU. This approach cannot be relied on to provide an adequate financial guarantee to ensure that the costs of safely disposing of the accumulated depleted uranium will be covered. This is because LES is unlikely to be able to pass on costs of $40 or more per SWU to its customers, given that planned NEF capacity is a small fraction of the global total enrichment capacity.[158] The provision of $[redacted] per SWU on which LES's calculations are based[159] is grossly inadequate and is in the context of government owned European facilities. It does not include the costs of DU disposal; nor does it take account of the many health risks and exchange rate risks that would be associated with a new enrichment plant in the United States. A license based on the premise that LES could recover anything like the costs of DU disposal elaborated here (and they are not the highest reasonable cost estimates that could be made) would run a considerable risk of saddling U.S. taxpayers and future generations with the immense liabilities of DU management and disposal. An encashable financial guarantee up front of at least $2.5 billion, appropriately escalated, *that is not dependent on the financial health of the proposed enrichment plant or its sales*, is essential to protect the people of New Mexico, U.S. taxpayers, and future generations from the liabilities associated with the DU that proposed LES uranium enrichment plant would generate.

[158] The proposed plant's capacity of 3 million SWU kg per year is well under 10 percent of global enrichment supply. See http://www.cameco.com/media_gateway/news_releases/2002/2002-july-22backgrounder.php.
[159] Deposition Chater et al. 2004/10/04, pp. 24-25.

*Table 9: Summary of the cost estimates for the scenarios considered in our analysis. Unless noted all values are given in current dollars per kilogram of uranium. As noted, we do not consider the LES assumptions case appropriate for use as a cost basis and include it in this table only for reference.(*)*

| Cost element | LES Assumptions *NOT IEER* *Not Prudent* | Class A grouted $UO_2$ | IEER GTCC Case 2 | IEER GTCC Case 3 |
|---|---|---|---|---|
| Conversion and storage | $[redacted] | $[redacted] | $[redacted] | $[redacted] |
| Disposal | $[redacted] | $[redacted] | $[redacted] | $[redacted] |
| $CaF_2$ disposition | $0.00 | $2.00 | $2.00 | $4.00 |
| Contingency – financial, exch. Rate | $0.00 | $[redacted] | $[redacted] | $[redacted] |
| Contingency - NRC-related | $[redacted] | $[redacted] | $[redacted] | $[redacted] |
| Contingency - U risk | $0.00 | $0.00 | $[redacted] | $[redacted] |
| Total per kg U | **$7.44** | **$13.59** | **$23.79** | **$30.41** |
| Total base case 133 million kg U metal basis), PV | 9.89E+08 | 1.81E+09 | 3.16E+09 | 4.04E+09 |
| Annual revenue, current dollars (2% discount rate) | 4.74E+07 | 8.67E+07 | 1.52E+08 | 1.94E+08 |
| Charge per SWU for DU deconversion and disposal. | **$15.81** | **$28.89** | **$50.59** | **$64.65** |
| Annual revenue, current dollars (4% discount rate) | 6.00E+07 | 1.10E+08 | 1.92E+08 | 2.45E+08 |
| Charge per SWU for DU deconversion and disposal | **$19.99** | **$36.52** | **$63.95** | **$81.72** |
| Annual revenue, current dollars (7% discount rate) | 8.16E+07 | 1.49E+08 | 2.61E+08 | 3.34E+08 |
| Charge per SWU for DU deconversion and disposal | **$27.21** | **$49.72** | **$87.06** | **$111.26** |
| Principal amount | -1 | -1 | -1 | |
| Discount rate per year | 2.0% | 4.0% | 7.0% | |
| Payment period, years | 27 | 27 | 27 | |
| Monthly payment | $0.0040 | $0.0051 | $0.0069 | |

(*) The individual cost elements have been redacted to protect proprietary information. The total costs are presented to allow for comparison among the scenarios and do not carry information sufficient to reveal confidential contractual information.

Note: Total DUF6 expected to be generated = 197,000 metric tons, which is equivalent to about 133,000 metric tons of elemental U.

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