RAS 11054

Protection against a secondary potential cause of cylinder overheating, immersion in a fire, is implemented through limitation of transporter fuel inventory.

### 4.2.2.8 Radiological Impacts of $DUF_6$ Conversion and $U_3O_8$ Disposal

Depleted uranium tails will be shipped offsite for conversion and disposal. Conversion of $DUF_6$ to $U_3O_8$ will result in releases of radioactive material to air and water. In addition to $U_3O_8$, large quantities of calcium fluoride ($CaF_2$), containing low uranium concentrations, will also be generated. Disposal of $U_3O_8$ could result in slow releases of uranium and uranium decay products to the groundwater infiltrating the disposal facility. Members of the public could come into contact with the groundwater due to natural flow and mixing or through intrusive activities. This section summarizes the analytical methods used and the estimates of potential impacts of conversion of $DUF_6$ to $U_3O_8$ and disposal of the $U_3O_8$. Detailed descriptions of the analysis and results are presented in Appendix A.

Impacts of Conversion from $DUF_6$ to $U_3O_8$

Because there are no facilities in the U.S. which convert $DUF_6$ to $U_3O_8$, the staff has assumed a representative conversion site and plant. Normal operation of the $DUF_6$ conversion process may result in small releases of uranium to the atmosphere and surface water. The magnitudes of these impacts were evaluated using the dose estimation procedures described in Section 4.2.2.1. Atmospheric release dose pathways include inhalation of air and ingestion of crops. Liquid release dose pathways include ingestion of drinking water and crops. Receptors considered were a maximally exposed individual living near the plant and the population surrounding the plant out to a distance of 80 km (50 miles). For a generic conversion plant, the maximally exposed individual was assumed to be located at a residence 0.5 km (.31 mile) from the plant's exhaust stacks, and the population surrounding the plant was assumed to include approximately 400,000 people.

Release rates of uranium to the atmosphere and surface water were estimated as $3.0\times10^{+7}$ Bq/yr ($8.0\times10^{-4}$ Ci/yr) and $1.9\times10^{+8}$ Bq/yr ($5.2\times10^{-3}$ Ci/yr), respectively. Potential dose to the maximally exposed adult due to atmospheric releases was estimated as $7.0\times10^{-9}$ Sv/yr ($7.0\times10^{-4}$ mrem/yr). Collective dose was estimated as $3.2\times10^{-5}$ person-Sv/yr ($3.2\times10^{-3}$ person-rem/yr). The critical individual for the atmospheric pathway is an infant located at the nearest residence. The estimated CEDE for the infant is $8.8\times10^{-8}$ Sv/yr ($8.8\times10^{-3}$ mrem/yr) and the largest tissue dose is $5.4\times10^{-7}$ Sv/yr ($5.4\times10^{-2}$ mrem/yr) to the bone.

The potential dose to the maximally exposed adult as a result of releases to surface water was estimated as $3.4\times10^{-8}$ Sv/yr ($3.4\times10^{-3}$ mrem/yr). Collective dose was estimated to be $1.0\times10^{-3}$ person-Sv/yr ($1.0\times10^{-1}$ person-rem/yr). The critical individual for the liquid pathway is an infant located at the nearest residence. The estimated CEDE for the infant due to liquid releases was estimated as $2.9\times10^{-7}$ Sv/yr ($2.9\times10^{-2}$ mrem/yr) and the largest tissue dose was $1.8\times10^{-6}$ Sv/yr (0.18 mrem/yr) to the bone. For both the atmospheric and liquid

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U.S. NUCLEAR REGULATORY COMMISSION
4-65 In the Matter of Louisiana Energy Services, LP
Docket No. 70-3103-ML Official Exhibit No. Staff 46
OFFERED by: Applicant/Licensee Intervenor
NRC Staff Other
IDENTIFIED on 10/27/05 Witness/Panel Staff Disposal
Action Taken: ADMITTED REJECTED WITHDRAWN
Reporter/Clerk Bethany Engel

releases, external exposures are a factor of a million less than the internal exposures. In addition to the atmospheric and liquid pathway exposures, an individual may be subject to direct and skyshine radiation from cylinders stored at the conversion site. Direct and skyshine doses (EDEs) imparted to the maximally exposed individual were estimated as $1.8 \times 10^{-7}$ Sv/yr ($1.8 \times 10^{-2}$ mrem/yr) and $2.6 \times 10^{-5}$ Sv/yr (2.6 mrem/yr), respectively. The critical organ for the direct/skyshine pathway is the thyroid with an annual dose of $2.9 \times 10^{-5}$ Sv/yr (2.9 mrem/yr). All estimated doses are small fractions of applicable limits and of the dose received from background radiation.

Impacts of Disposal of $U_3O_8$

There are currently no disposal facilities for large quantities of $DUF_6$, but it is plausible to assume that $U_3O_8$ may be disposed by emplacement in near-surface or deep geologic disposal units. The quantity of uranium to be disposed is the 30-year CEC tails inventory or $9.1 \times 10^{+7}$ kg ($2.0 \times 10^{+8}$ lb) expressed as $U_3O_8$ (equivalent to a volume of $3.0 \times 10^{+4}$ $m^3$ [$1.1 \times 10^{+6}$ $ft^3$]). A near-surface disposal unit is assumed to be an earth-mounded bunker (tumulus) with a $U_3O_8$ matrix measuring approximately 61 m (200 ft) long by 61 m (200 ft) wide and 8 m (26 ft) high. A deep geological disposal unit is assumed to be a pre-existing cavity such as an abandoned mine, with a $U_3O_8$ matrix measuring approximately 100 m (320 ft) long by 100 m (320 ft) wide and 3 m (9.8 ft) high. The radiological dose limits specified in 10 CFR Part 61 are adopted as a basis for comparative evaluation. Under this regulation, annual dose to any member of the public is limited to $2.5 \times 10^{-4}$ Sv (25 mrem) to the whole body, $7.5 \times 10^{-4}$ Sv (75 mrem) to the thyroid, and $2.5 \times 10^{-5}$ Sv (25 mrem) to any other organ.

The tails disposal impact analysis method included selection of assumed generic disposal sites, development of undisturbed performance and deep well water use exposure scenarios, and estimation of potential doses. Exposure pathways used for the near-surface disposal case include drinking shallow well water and consuming crops irrigated with shallow well water. Evaluation of the deep disposal case included undisturbed performance and deep well water exposure scenarios. In the undisturbed performance scenario, groundwater flows into a river which serves as a source of drinking water and fish. For the well water use exposure scenario, an individual drills a well into an aquifer downgradient from the disposal facility and uses groundwater for drinking and irrigation.

The release of uranium isotopes and their daughter nuclides from the disposal facility is limited by their solubility in water. Solubilities were estimated using the PHREEQE computer code (Parkhurst et al., 1980) developed at the U.S. Geological Survey (USGS). Maximum inventories of nuclides present in a $U_3O_8$ matrix were estimated using computerized evaluation of the Bateman equation (Benedict et al., 1981). Concentrations of radionuclides in groundwater and corresponding doses were estimated using a combination of an analytic solution to the one-dimensional flow, three-dimensional dispersion, transport equation developed by the USGS (Wexler, 1992) and the unit soil contamination dose factors developed with the RESRAD computer code (Gilbert et al., 1989). The RESRAD factors incorporate the effects of inadvertent soil ingestion and

ingestion of crops, meat, and milk. The dose estimates were corroborated using the PRESTO-EPA (Fields et al., 1987) code to recalculate the near-surface scenario impacts.

Using the environmental characteristics of a humid southeastern U.S. site and the methods of this EIS, drinking water and agricultural doses were estimated to be $5.7 \times 10^{-3}$ Sv/yr (570 mrem/yr) and $3.1 \times 10^{-4}$ Sv/yr (31 mrem/yr), respectively, for a near surface disposal facility. These doses exceed 10 CFR Part 61 limits, making it likely that a deep disposal site will be selected.

In order to compensate for the lack of knowledge of a specific deep disposal site, two representative sites whose geological structures have previously been characterized were selected for analysis. Site 1 (Rechard, 1993) is located in a granite formation overlain by a thin layer of glacial till. Emplacement of the $U_3O_8$ matrix at Site 1 is assumed to be at a depth of approximately 290 m (950 ft). Site 2 (Stottlemyre et al., 1979) is located in a sequence of interbedded sandstone and basalt layers. Emplacement of the matrix at Site 2 is assumed to be at a depth of 635 m (2,070 ft). At each of the sites, vertical fractures intersect the emplacement horizon, allowing water to be carried upward to an aquifer and then to a river. The fracture sizes, hydraulic conductivities, and hydraulic gradients used in the EIS analysis are the same as those used in the referenced studies (Rechard, 1993 and Stottlemyre et al., 1979).

Potential consequences of emplacement of $U_3O_8$ in a geological disposal unit include intake of radionuclides from drinking water, irrigated crops, and fish. Under the assumed conditions for the undisturbed performance scenario, groundwater would be discharged to a river. Under conditions not expected to occur, an individual would obtain groundwater by drilling a well downgradient from the disposal unit. The well take-off point was located at the center elevation of the aquifer. The horizontal location of the well was the distance of maximum dose (200 meters) for a take-off point at the selected elevation.

The analysis considers nuclides present in the emplaced $U_3O_8$ and nuclides produced by decay during transport. By evaluating solubility-limited releases, the upper bound of potential dose was estimated. In the year of maximum exposure at the granite site (Site 1), the estimated doses for the well scenario were $1.6 \times 10^{-7}$ Sv ($1.6 \times 10^{-2}$ mrem) and $2.3 \times 10^{-6}$ Sv (0.23 mrem) for drinking water and agriculture pathways, respectively. For the river scenario at the granite site (Site 1), expected doses were estimated to be $5.3 \times 10^{-16}$ Sv ($5.3 \times 10^{-11}$ mrem) and $1.0 \times 10^{-15}$ Sv ($1.0 \times 10^{-10}$ mrem) for drinking water and fish ingestion pathways, respectively. In the year of maximum exposure at the sandstone/basalt site (Site 2), expected doses for the well scenario were estimated to be $1.3 \times 10^{-10}$ Sv ($1.3 \times 10^{-5}$ mrem) and $1.8 \times 10^{-9}$ Sv ($1.8 \times 10^{-4}$ mrem) for drinking water and agriculture pathways, respectively. For the river scenario at the basalt site, expected doses were estimated to be $1.6 \times 10^{-9}$ Sv ($1.6 \times 10^{-14}$ mrem) and $3.0 \times 10^{-14}$ Sv ($3.0 \times 10^{-9}$ mrem) for drinking water and fish ingestion pathways, respectively.

All estimated impacts for a deep disposal facility are less than the 0.25 mSv/yr (25 mrem/yr) level adopted from 10 CFR Part 61 as a basis for comparison. For the well

scenario, the estimated dose is dominated by $^{226}Ra$ originating at the disposal facility and its relatively short-lived daughter radionuclides which develop during transport. For the river scenario, almost all of the $^{226}Ra$ originating at the disposal facility decays during transport and the estimated dose was dominated by the less soluble $^{238}U$ parent nuclide and its daughter radionuclides developing during transport. The assumptions used in the analysis, including neglect of potential engineered barriers, mass transfer limitations in releases, and decay and retardation during vertical transfer contribute to a conservative analysis.

If for some reason a deep geological facility is not selected for tails disposal, the tails can be stored indefinitely in a retrievable surface facility with minimal environmental impacts. The environmental impacts associated with such storage would be commitment of the land for a storage area, and a small offsite radiation dose.

### 4.2.3 Cumulative Environmental Impacts

The cumulative environmental impacts as a result of facility operation include:

- drawdown of the Sparta Sand Aquifer due to water withdrawals for plant operation;
- possible reduction in shallow groundwater levels beneath the site;
- reduction in water levels in Lake Avalyn;
- possible accumulation of contaminants in surface water sediments;
- possible bioaccumulation of contaminants in surface water biota;
- reduction in ozone content of stratosphere.

Drawdown is expected in the Sparta Sand Aquifer due to water use at the site. Depending on the scenario considered, drawdown due to site activities ranges from 0.03 to 1.2 m (0.1 to 4 ft) at the facility boundary. The drawdown at the site is small and ranges from 10 to 12 percent of the total drawdown at the nearest public water supply well, Claiborne Well No. 4. Normal facility operations are expected to pump water at the lower rates considered, so that the effect of site water usage on the total aquifer is expected to be minimal (LES, 1993a).

A potential decrease in the levels of shallow groundwater also exists at the site due to construction activities. Approximately 28 hectares (70 acres) of the site will be developed and largely paved during facility construction (LES, 1994a). Water which would have infiltrated over this area will become part of the site drainage through the Yard Drains System and, therefore, be unavailable for shallow groundwater recharge. The developed

# APPENDIX A

## ASSESSMENT OF THE ENVIRONMENTAL IMPACTS OF DEPLETED $UF_6$ DISPOSITION

The depleted $UF_6$ ($DUF_6$) exiting the separation cascades will contain between 0.2 and 0.34 weight percent of $^{235}U$ and may be a potential resource. However, given the current large supply and limited market for this material, it is likely that the tails will ultimately require long-term disposal. The tails possession limit for the Claiborne Enrichment Center (CEC) will be 80,000 metric tons (88,200 tons) of $DUF_6$, or the amount produced after 15 years of production of $DUF_6$, whichever is less. Thus, no later than 15 years after commencement of CEC operations, the depleted tails will begin to be transported offsite. Due to the reactivity of $DUF_6$ with water, long-term disposal of $DUF_6$ will require conversion to a more chemically stable form. The following analysis provides a conservative assessment of the potential impacts of converting $DUF_6$ to triuranium octoxide ($U_3O_8$) and disposal of the $U_3O_8$.

### A.1 Chemical Conversion of $UF_6$ to $U_3O_8$

A variety of uranium compounds, including uranium tetrafluoride ($UF_4$), uranium dioxide ($UO_2$), uranium trioxide ($UO_3$), and $U_3O_8$, are more stable in a geological environment than $UF_6$. The uranium fluorides, including $UF_4$, are less stable than the uranium oxides and produce hydrogen fluoride (HF) in reaction with water. Thus, the oxide forms are more favorable for long-term disposal. In the presence of oxygen ($O_2$), as is unavoidable in the conversion process, $UO_2$ and $UO_3$ are oxidized to $U_3O_8$ (Katz and Rabinowitch, 1951). $U_3O_8$ is readily produced from $UF_6$ and has potential long-term stability in a geological environment.

Three primary chemical processes are available for conversion of $UF_6$ to $U_3O_8$. In the first process, $UF_6$ is reduced to $UF_4$ through reaction with hydrogen. The $UF_4$ is subsequently reacted with water to produce $U_3O_8$. In the second process, $UF_6$ is reacted with water to produce uranyl fluoride ($UO_2F_2$), which is subsequently converted to ammonium diuranate [$(NH_4)_2U_2O_7$]. The $(NH_4)_2U_2O_7$ is then calcined to form $U_3O_8$. In the third process, $UF_6$ is reacted with steam in the gas phase to produce $UO_2F_2$, which is then reacted with hydrogen ($H_2$) and $O_2$ to produce $U_3O_8$. Each of these three processes will generate HF containing small amounts of uranium. The second process entails aqueous phase reactions, resulting in more complicated waste management. Both the first and third processes use gas phase reactions which produce byproduct streams which are more readily managed. The third process entails reaction steps used by commercial facilities in the U.S. and Europe. Based on these considerations, the third chemical process was selected as the basis for this analysis.

### A.1.1 Generic Conversion Plant Site Description

The generic conversion plant site selected for evaluation covers an area of approximately 405 hectares (1,000 acres) and is located in the midwestern U.S. The site has a relatively mild continental climate with warm summers and mild winters. Total annual precipitation is approximately 104 cm (41 in), and winds are moderate with an annual average speed of 3.3 m/s (7.4 mph). Meteorological conditions are generally neutral, in Pasquill Classes C and D, 50 percent of the time. Stable meteorological conditions, Pasquill Class F, occur 12 percent of the time. The conversion plant is located adjacent to the banks of a major river which has an average flow of 45 $m^3/s$ (1,590 $ft^3/s$). Population density in the vicinity of the plant is low, and the total population within an 80 km (50 mi) radius is approximately 400,000 people. The nearest resident is located near the plant boundary approximately 500 m (1,640 ft) from the plant's gaseous effluent release point.

### A.1.2 Generic $DUF_6$ Conversion Process Description

The generic process for conversion of $UF_6$ to $U_3O_8$ utilizes a two-step reaction scheme supported by effluent controls for particulates released to the atmosphere and dissolved species released to surface water. The nominal capacity of the plant is 5,700 metric tons of $UF_6$ per year (6,270 tons/yr), allowing conversion of the expected inventory of CEC $DUF_6$ in approximately 20 years of operation. A process flow diagram of the conversion process is presented in Figure A.1 and the reaction stoichiometry is summarized in Table A.1

**Table A.1 Reaction Stoichiometry for Conversion of $DUF_6$ to $U_3O_8$**

Reaction 1 : Hydrolysis of $UF_6$

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

Reaction 2 : Conversion of $UO_2F_2$ to $U_3O_8$

$$3\,UO_2F_2 + 3\,H_2 + O_2 \rightarrow U_3O_8 + 6\,HF$$

Reaction 3 : Neutralization of HF

$$2\,HF + Ca(OH)_2 \rightarrow CaF_2 + 2\,H_2O$$

Feed $DUF_6$ is transported to the conversion plant site and stored on the site in Type 48G cylinders, each containing up to 12.7 metric tons (14 tons) of $DUF_6$. In the initial step of the process, $DUF_6$ is vaporized in an autoclave and fed to a hydrolysis reactor. The hydrolysis reactor operates at temperatures in excess of 300° C (570° F) (Chemical Abstracts, 1986), producing solid $UO_2F_2$ and gaseous HF. The solids and gases are separated in a series arrangement of porous metal filters. The particulate/gas separation efficiency of each filter is on the order of 99.9 percent (NRC, 1984). After the gases are cooled, they are routed to the HF scrubbers. The solid $UO_2F_2$ is fed to the second stage reactor, the conversion reactor. In the conversion reactor, the $UO_2F_2$ combines with $H_2$ and

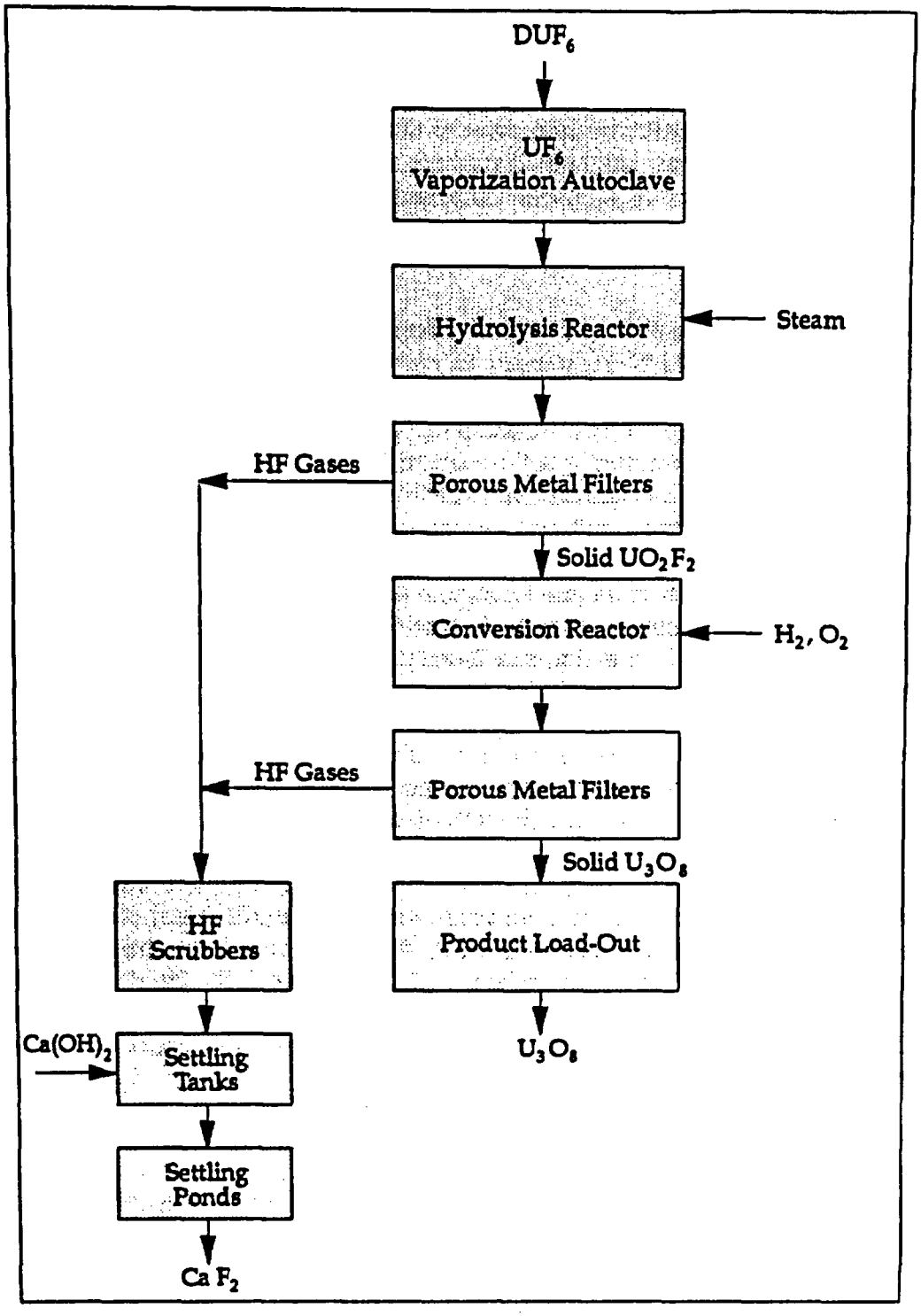


**Figure A.1 Process Flow Diagram of the $DUF_6$ Conversion Process**

$O_2$ at temperatures in excess of 750° C (1,380° F) (Harrington and Ruehle, 1959). Additional HF is generated in this reaction step. The $H_2$ used in the process is generated by catalytic dissociation of ammonia ($NH_3$) while the $O_2$ is introduced as air. Porous metal filters in series are again used to separate the $U_3O_8$ product from the reaction gases. The gas is combusted to consume residual $H_2$, cooled, and transferred to the scrubber system. Product $U_3O_8$ is transferred from the conversion reactor by a conveyor system and loaded into drums for storage and ultimate disposal. At the specified $UF_6$ feed rate, the plant would produce approximately 4,550 metric tons/yr (5000 tons/yr) of $U_3O_8$.

### A.1.3 Generic $DUF_6$ Conversion Process Waste Management

The HF scrubber system uses a spray tower and packed tower operated in series to remove uranium particulates and HF from gases which are released to the atmosphere. The HF absorbed into the alkaline scrubber solution contains small amounts of uranium. As the scrubber solution is contacted with slaked lime [$Ca(OH)_2$] in a series of settling tanks, the HF is neutralized and converted to solid calcium fluoride ($CaF_2$). The reaction solution is transferred to lined settling ponds where the solids and water are separated. On a dry weight basis the maximum uranium content of the $CaF_2$ is estimated to be 0.05 Bq/g (1.4 pCi/g). Approximately $3.4 \times 10^{+7}$ liters per year ($9.0 \times 10^{+6}$ gal/yr) of water are estimated to be released to the environment from the scrubber system. In order to provide a conservative basis for impact analysis, all of the uranium entering the scrubber system is assumed to be released in the liquid effluent. Thus, the liquid release source term is approximately $1.92 \times 10^{+8}$ Bq/yr (5,200 $\mu$Ci/yr).

The primary sources of releases to the atmosphere are the HF scrubber off-gas and dust from the product load-out system. Particulate removal efficiencies are 80 percent and 99 percent for the spray tower and packed tower, respectively (Cheremisinoff and Young, 1976). Assuming approximately $1.92 \times 10^{+8}$ Bq/yr (5,200 $\mu$Ci/yr) enters the HF scrubbers, approximately $3.8 \times 10^{+5}$ Bq/yr (10 $\mu$Ci/yr) will be released in the scrubber off-gas. Product drum loading operations will be conducted in an area vented through fabric filters. Based upon experience with similar systems (NRC, 1984) and assuming a fabric filter efficiency greater than 95 percent (Perry and Chilton, 1973), approximately $3.0 \times 10^{-5}$ weight percent of the $U_3O_8$ will be lost to the atmosphere. Based upon a $U_3O_8$ production rate of 4,550 metric tons (5,000 tons), approximately $2.89 \times 10^{+7}$ Bq/yr (780 $\mu$Ci/yr) of uranium will be released to the atmosphere from this source.

### A.1.4 Dose Estimates for Generic $DUF_6$ Conversion Plant Operation

Radioactive material would be released to the atmosphere from the generic conversion plant at a height of 20 m (65 ft). The source term for the release is approximately $2.96 \times 10^{+7}$ Bq/yr (800 $\mu$Ci/yr) as described in Section A.1.3. Expected exposure pathways include inhalation of air, consumption of crops, direct exposure to the effluent plume and soil, and inadvertent ingestion of soil. Potential internal doses to the maximally exposed adult individual and the population surrounding the plant site for the atmospheric pathway

are presented in Table A.2. External dose to the whole body of an adult individual due to airborne and deposited uranium is estimated to be approximately $5.2 \times 10^{-15}$ Sv/yr ($5.2 \times 10^{-10}$ mrem/yr). The critical individual for this pathway is an infant located at the residence nearest the plant. The estimated committed effective dose equivalent (CEDE) for the infant is $8.8 \times 10^{-8}$ Sv/yr ($8.8 \times 10^{-3}$ mrem/yr), and the largest tissue dose is estimated to be $5.4 \times 10^{-7}$ Sv/yr ($5.4 \times 10^{-2}$ mrem/yr) to the bone. For both the maximally exposed individual and the surrounding population, the exposures are only a small fraction of both background radiation and applicable limits, including the limits specified in 10 CFR Part 20, 40 CFR Part 61, and 40 CFR Part 190.

**Table A.2 Potential Internal Doses to the Maximally Exposed Adult Individual and the Population Surrounding the Generic $DUF_6$ Conversion Plant**

| | Atmospheric Pathway | | Liquid Pathway | |
|---|---|---|---|---|
| Affected Organ | Individual (Sv) | Population (Person-Sv) | Individual (Sv) | Population (Person-Sv) |
| Gonads | $2.5 \times 10^{-10}$ | $1.2 \times 10^{-6}$ | $1.2 \times 10^{-9}$ | $3.8 \times 10^{-5}$ |
| Breast | $1.7 \times 10^{-10}$ | $7.6 \times 10^{-7}$ | $7.8 \times 10^{-10}$ | $2.4 \times 10^{-5}$ |
| Red Bone Marrow | $6.4 \times 10^{-9}$ | $2.9 \times 10^{-5}$ | $3.1 \times 10^{-8}$ | $9.3 \times 10^{-4}$ |
| Lung | $2.3 \times 10^{-9}$ | $1.3 \times 10^{-5}$ | $1.3 \times 10^{-9}$ | $4.1 \times 10^{-5}$ |
| Thyroid | $2.5 \times 10^{-10}$ | $1.2 \times 10^{-6}$ | $1.2 \times 10^{-9}$ | $3.8 \times 10^{-5}$ |
| Bone Surface | $9.8 \times 10^{-8}$ | $4.4 \times 10^{-4}$ | $4.9 \times 10^{-7}$ | $1.5 \times 10^{-2}$ |
| Stomach | $2.9 \times 10^{-11}$ | $1.3 \times 10^{-7}$ | $5.4 \times 10^{-10}$ | $1.6 \times 10^{-5}$ |
| Small Intestine | $3.9 \times 10^{-11}$ | $1.6 \times 10^{-7}$ | $1.2 \times 10^{-9}$ | $3.5 \times 10^{-5}$ |
| Upper Large Intestine | $1.3 \times 10^{-10}$ | $4.7 \times 10^{-7}$ | $7.4 \times 10^{-9}$ | $2.3 \times 10^{-4}$ |
| Lower Large Intestine | $4.2 \times 10^{-10}$ | $1.5 \times 10^{-6}$ | $2.3 \times 10^{-8}$ | $6.7 \times 10^{-4}$ |
| Kidney | $4.7 \times 10^{-8}$ | $1.8 \times 10^{-4}$ | $2.2 \times 10^{-7}$ | $6.6 \times 10^{-3}$ |
| CEDE | $7.0 \times 10^{-9}$ | $3.2 \times 10^{-5}$ | $3.4 \times 10^{-8}$ | $1.0 \times 10^{-3}$ |

Radioactive material would be released to surface water from the plant's waste management systems. The upper limit on these releases is estimated to be $1.92 \times 10^{+8}$ Bq/yr (5,200 $\mu$Ci/yr) as discussed in Section A.1.3. Potential exposure pathways include ingestion of drinking water, crops and fish, and external exposure from boating and swimming. Potential internal doses for the maximally exposed adult individual and the surrounding population for the liquid pathway are presented in Table A.2. Liquid pathway external exposures are a small fraction of the internal exposures. The critical individual for the liquid pathway is an infant located at the residence nearest the plant. The estimated CEDE for the infant is $2.9 \times 10^{-7}$ Sv/yr ($2.9 \times 10^{-2}$ mrem/yr), and the largest tissue dose is estimated

to be $1.8 \times 10^{-6}$ Sv/yr ($1.8 \times 10^{-1}$ mrem/yr) to the bone. Individual and collective doses are both small fractions of background radiation and applicable limits, including those specified in 10 CFR Part 20 and 40 CFR Part 190.

The potential effect of depleted uranium storage at the conversion site is exposure of a receptor to gamma rays, bremsstrahlung, and x-rays due to direct and atmosphere-reflected (skyshine) transmission of radiation. For the purposes of this analysis, all cylinders of $DUF_6$ produced during 30 years of CEC operation are assumed to be stored in an unstacked rectangular array, on level ground, located at a distance of 1 km (0.6 mile) from the resident nearest the conversion site. The inter-cylinder spacings proposed for the CEC were adopted as a representative basis for this conversion plant analysis. In this storage configuration, the front row of cylinders would contribute almost all of the direct exposure, as radiation from cylinders in the interior of the array would be absorbed in surrounding cylinders. All cylinders would contribute to skyshine exposures.

Surface dose rates from a single cylinder containing $DUF_6$ are estimated to be less than $2 \times 10^{-5}$ Sv/hr (2 mrem/hr) (Friend, 1991). Using this estimate of dose rate, the average annual dose to the nearest resident from direct radiation from all cylinders is estimated to be $1.8 \times 10^{-7}$ Sv ($1.8 \times 10^{-2}$ mrem). Doses from skyshine were estimated using a computer code which applies the point kernel approach to calculate photon scattering dose. The annual dose (EDE) due to skyshine from all cylinders of $DUF_6$ is estimated to be $2.6 \times 10^{-5}$ Sv (2.6 mrem) while the maximum annual tissue dose is estimated to be $2.9 \times 10^{-5}$Sv (2.9 mrem) to the thyroid. The combined direct and skyshine dose from all cylinders is a small fraction of background radiation and of applicable limits, including those specified in 10 CFR Part 20 and 40 CFR Part 190.

### A.1.5 Cumulative Impacts of Generic $DUF_6$ Conversion Plant Operation

Nuclear Regulatory Commission (NRC) regulations (10 CFR 20.1301) require that the total effective dose equivalent (TEDE) for releases related to routine operations should not exceed 1 mSv/yr (100 mrem/yr). In addition, Environmental Protection Agency (EPA) regulations (40 CFR Part 190) require that for routine releases to the general environment, the annual dose equivalent should not exceed 0.25 mSv (25 mrem) to the whole body, 0.75 mSv (75 mrem) to the thyroid, and 0.25 mSv (25 mrem) to any other organ. For releases to the atmosphere, EPA regulations (40 CFR Part 61) require that the annual effective dose equivalent should not exceed 0.1 mSv (10 mrem). For the critical individual, the cumulative annual CEDE (atmospheric and liquid pathways) is estimated to be $3.8 \times 10^{-4}$ mSv ($3.8 \times 10^{-2}$ mrem) and the cumulative annual tissue dose is estimated to be $2.3 \times 10^{-3}$ mSv ($2.3 \times 10^{-1}$ mrem) to the whole bone. The whole bone dose represents the effect on the entire bone tissue, including both the bone surface and the red bone marrow. The annual TEDE (atmospheric, liquid, and direct pathways) is estimated as $2.6 \times 10^{-2}$ mSv (2.6 mrem) while the maximum annual tissue dose is estimated as $2.9 \times 10^{-2}$ mSv (2.9 mrem) to the thyroid. Each of these doses is significantly lower than applicable limits and background radiation. Based on the analysis results presented above, it is concluded that

operation of the $DUF_6$ conversion plant is expected to have negligible radiological impacts on the environment.

## A.2 Disposal of $U_3O_8$

$U_3O_8$ may be disposed by emplacement in near-surface or deep geological environments. Technologies applicable for near-surface disposal units include lined trenches, above- and below-grade vaults, and tumuli. This analysis assumes that the near-surface disposal unit is a tumulus with a 2-m thick compacted clay cover. Deep disposal facilities appropriate for disposal of uranium compounds include pre-existing mines and facilities engineered specifically for disposal. This analysis assumes that the disposal facility is pre-existing (that is, an abandoned mine or natural formation) with a minimum of engineered barriers. The objective of this analysis is to develop estimates of impacts for conditions which may be expected to occur at a carefully selected site. The analysis is not intended to assess generic impacts under all possible geological conditions.

The quantity of uranium assumed to be disposed is the 30-year CEC tails inventory (the amount adopted for analyzing the conversion of $DUF_6$ to $U_3O_8$), or approximately $9.1 \times 10^7$ kg of $U_3O_8$. The crystal density of $U_3O_8$ is reported as 8.3 g/cm$^3$ (Katz and Rabinowitch, 1951), while bulk density can be as low as 3.0 to 4.0 g/cm$^3$ (Chemical Abstracts, 1986). In order to provide a conservative analysis, a bulk density of 3.0 g/cm$^3$ was used in this analysis. The $U_3O_8$ disposal volume is thus approximately $3.0 \times 10^{+4}$ m$^3$. Initial activities of $^{238}U$ and $^{234}U$ are estimated to be approximately $9.6 \times 10^{+14}$ and $2.7 \times 10^{+14}$ Bq ($2.6 \times 10^{+4}$ and $7.3 \times 10^{+3}$ Ci), respectively. Consistent with the assumption of production in a fluidized bed process, the particles are assumed to have a small mean size with diameters on the order of 50 microns. For the near-surface case, the thickness of the disposed material is assumed to be 8 m, covering an area approximately 61 m long and 61 m wide. For deep disposal cases, the $U_3O_8$ is assumed to be emplaced at a thickness of 3 m, covering an area approximately 100 m long and 100 m wide.

The following sections present discussions of the approach used for the analysis, analysis methods and models, and the results of the analysis. The dose limits specified in 10 CFR Part 61 are adopted as a basis for comparative evaluation. Under this regulation, annual dose to any member of the public is limited to $2.5 \times 10^{-4}$ Sv (25 mrem) to the whole body, $7.5 \times 10^{-4}$ Sv (75 mrem) to the thyroid, and $2.5 \times 10^{-4}$ Sv (25 mrem) to any other organ.

### A.2.1 Disposal Analysis Methods

The tails disposal impact analysis approach includes selection of representative disposal sites, development of undisturbed performance, exposure scenarios, and selection of consequence estimation models. The characteristics of the sites selected for near-surface and deep disposal are described in the following paragraphs. Exposure scenarios selected for evaluation of near-surface disposal included drinking of well water and consumption of crops irrigated with water drawn from the well. Evaluation of the deep disposal case included undisturbed performance and well water exposure paths. In the undisturbed performance case, groundwater flows to a river which serves as a source of drinking water

and fish for an individual member of the public. For the well water exposure scenario, an individual drills into a deep aquifer down gradient from the disposal facility and uses the water for drinking and irrigation. Limits were not placed on the length of the evaluation period, and doses reported are the maximum that would be estimated for any time in the future.

The release rate of uranium and daughter radionuclides from the disposal facility is limited by their solubility in water or by the total inventory of radionuclide present at the time of release. In the case of a solubility limited release, the amount of radionuclide transported from the disposal facility is equal to the solubility multiplied by the flow rate of water through the facility. If the amount of a radionuclide present in the facility at a given time period is less than the amount of that radionuclide which could be removed during the time period by solubility limited release, the release could be considered inventory limited. For example, because an extremely small quantity of $^{226}Ra$ would be initially present in $DUF_6$, initial release of $^{226}Ra$ from the disposal facility would be inventory limited. As time passes, the facility inventory of $^{226}Ra$ would increase and a transition from inventory limited to a solubility limited release would occur. In order to provide a conservative assessment of potential impacts, inventory limited releases were not considered. In this analysis, solubilities are estimated using the PHREEQE computer code (Parkhurst et al., 1980) developed at the U.S. Geological Survey (USGS). Maximum concentrations of radionuclides present in the disposal unit are estimated using a computerized evaluation of the Bateman equation (Benedict et al., 1981). Concentrations of radionuclides in groundwater and corresponding doses were estimated using a code developed for this analysis. This code uses a combination of an analytic solution to the one-dimensional flow, three-dimensional dispersion equation developed by the USGS (Wexler, 1992), and unit soil contamination to dose factors developed with the RESRAD computer code (Gilbert et al., 1989). The code is capable of modeling retardation and decay during transport. The RESRAD factors incorporate the effects of direct exposure, inadvertent soil ingestion and ingestion of crops, meat, and milk. In the analytic solution code, drinking water doses are estimated as the product of predicted radionuclide water concentration, water intake rate (0.73 $m^3/yr$), and radionuclide ingestion dose conversion factor. Doses from fish ingestion are estimated as the product of water concentration, bioaccumulation factor, consumption rate, and ingestion dose conversion factor. Bioaccumulation factors are the same as those used in NRC analysis of decommissioning scenarios (Kennedy and Strenge, 1992). The dose conversion factors are consistent with present Federal regulatory guidance (Eckerman et al., 1988). The estimations were corroborated using the PRESTO-EPA (Fields et al., 1987) computer code to recalculate near-surface disposal scenario impacts. PRESTO is a pathway analysis code developed for analysis of impacts of disposal of radioactive waste.

For deep disposal, intrusion into the emplacement horizon is not an expected event. However, direct, inadvertent drilling into the $U_3O_8$ as a result of resource exploration was considered. The EPA has proposed occurrence frequencies for intrusion events due to exploratory drilling for resources into a deep disposal unit (EPA, 1994). The recommended occurrence frequencies are $3x10^{-3}$ boreholes/$km^2$/yr for sedimentary formations and

$3x10^{-4}$ boreholes/km$^2$/yr for other geological formations. Due to the relatively small area of the disposal facility, the estimated frequency of occurrence of this event is small, on the order of $3x10^{-6}$/yr. Also, groundwater flow rates through the facility would be low and the water extracted as a result of exploratory drilling would not likely be used for potable or irrigation purposes. Thus, the radiological consequences of groundwater brought to the surface through direct intrusion into the $U_3O_8$ would be insignificant. The consequences of drilling into the $U_3O_8$ could include transport of approximately 0.2 m$^3$ (7.7 ft$^3$) of $U_3O_8$ to the surface with the drilling mud and cuttings. The material would most likely be deposited in the drilling mud pond. The potential impacts, including exposure of workers to resuspended, contaminated dust would be small and transient. For these reasons, the impacts of drilling into the $U_3O_8$ were not evaluated in detail.

In the well scenario, the horizontal and vertical locations of the well and its take-off point, respectively, are intrinsically uncertain elements of the scenario. A water well is more likely to be screened in the upper rather than the lower portion of a deep, homogeneous, hard-rock aquifer. In this analysis, the well take-off point is assumed to be at the center elevation of the aquifer. One horizontal location of the well is as likely as the site of a well as any other. In order to provide a conservative analysis, the well was located at the distance of maximum dose for the center elevation of the aquifer.

### A.2.2 Near-Surface Disposal of $U_3O_8$

The estimates developed in this analysis focus on the impacts of disposal of $U_3O_8$. Prior analysis (Kozak et al., 1992) considered potential impacts of disposal of $UF_6$, $UF_4$, and $U_3O_8$ in near-surface disposal facilities. The characteristics of the disposal site were those of the humid southeastern U.S. This prior analysis noted that reaction of $UF_6$ and $UF_4$ with water would produce quantities of HF which could compromise the integrity of the disposal facility and significantly disturb the environment. Consequently, drinking water, intruder construction, and intruder agricultural scenario doses were unacceptably high for the fluorinated waste forms. The Kozak analysis also included consideration of potential impacts of near-surface disposal of $U_3O_8$. Release of uranium was modeled as being controlled by its solubility, which in the oxygenated near-surface environment was estimated as less than $2.4x10^{-3}$ g/L. The Kozak analysis concluded that doses would exceed the $2.5x10^{-4}$ Sv/yr (25 mrem/yr) limit specified in 10 CFR Part 61. Using infiltration rate and aquifer flow rate for the humid southeastern site, the doses presented in Table A.3 were estimated using the methods of this Environmental Impact Statement (EIS) for release from a near-surface $U_3O_8$ disposal facility. It should be noted that the estimated doses are significantly above the limits specified in 10 CFR Part 61, even though the reported results do not include the potential effects of ingrowth of uranium daughters or of intruder construction scenarios. The analytic model and PRESTO results are consistent, indicating similarity of the pathway models. Because for near-surface disposal of $U_3O_8$, projected doses exceed 10 CFR Part 61 limits, a deep disposal site is most likely to be selected for ultimate disposition of depleted uranium.

**Table A.3 Drinking Water and Agricultural Scenario Doses for Near-Surface Disposal of $U_3O_8$, (Sv/yr)**

| | Analytic Flow/ RESRAD Model | PRESTO |
|---|---|---|
| Drinking Water Dose | $5.7 \times 10^{-3}$ | $5.8 \times 10^{-3}$ |
| Agricultural Dose | $3.1 \times 10^{-4}$ | $3.7 \times 10^{-4}$ |

### A.2.3 Deep Disposal of $U_3O_8$

At the present time, candidate sites for the deep disposal of $U_3O_8$ have not been identified. In order to compensate for lack of detailed knowledge of a specific site, two sites, whose geological structures have previously been characterized, have been assumed and analyzed. The characteristics of these sites are representative of natural variability and expected conditions for deep disposal. For each of the sites, release of radionuclides would be controlled by solubility limited dissolution in water flowing through the disposal facility. The effects of potential engineered barriers and retardation during vertical transport are neglected.

Generic Deep Disposal Site Descriptions

The characteristics of the two sites used for evaluation have been developed in prior studies of radioactive waste disposal. The sites are assumed to be located in the U.S. and have the geological structures depicted in Figure A.2. Site 1 (Rechard, 1993) is located in a granite formation overlain by a thin layer of glacial till. The disposal horizon is located at a depth of 290 m (0.18 mi) below ground surface. It is intersected by vertical fractures, allowing transmission of water upward through the $U_3O_8$ matrix to a horizontal fracture zone (deep aquifer), which in turn carries water toward a river. Site 2 (Stottlemyre et al., 1979) is located in a sequence of interbedded sandstone and basalt layers. The $U_3O_8$ is emplaced in a sandstone layer 635 m (0.39 mi) below the ground surface. Local upward flow carries water through the $U_3O_8$ matrix to a cemented sand and gravel strata (deep aquifer) which intersects with the river. The fracture sizes, densities, hydraulic conductivities, and permeabilities used in this evaluation are the same as those reported in the original studies (i.e., Rechard, 1993 and Stottlemyre et al., 1979). A list of the groundwater flow path parameters and the values used for analyzing the transport paths is presented in Table A.4.

Solubility Estimates

The solubility of a radionuclide in groundwater depends on the concentrations of naturally occurring ions in the groundwater and on the physical/chemical characteristics, for example, pH, eH, and temperature of the water. Thus, in order to predict representative concentrations of a dissolved specie, the characteristics of the groundwater must be established. The chemical analysis for an actual near-surface groundwater (WVNS, 1993),

*a) Site 1: Granite Formation*

| Thickness* (m) | Lithology |
|---|---|
| 6 | Glacial till |
| 125 | Pink granite |
| 75 | Upper fracture zone |
| 175 | Gray granite (emplacement horizon) |
| 10 | Lower fracture zone |

290 m

*b) Site 2: Sandstone/Basalt Layers*

| Thickness* (m) | Lithology |
|---|---|
| 125 | Sand and gravel |
| 200 | Cemented sand and gravel |
| 300 | Basalt |
| 25 | Sedimentary interbed (emplacement horizon) |
| 200 | Basalt |

635 m

** Not to scale*

**Figure A.2 Geologic Structure of Generic Deep Disposal Sites**

Table A.4 Flow Path Parameters for Generic Deep Disposal Sites

| | Hydraulic Conductivity (m/yr) | Flow Area ($m^2$) | Gradient (m/m) |
|---|---|---|---|
| Site 1 (Granite) | | | |
| Vertical Path | 8.03 | 4.0 | 0.02 |
| Horizontal Path | 30.9 | $7.5 \times 10^3$ | 0.005 |
| Site 2 (Sandstone/Basalt) | | | |
| Vertical Path | 0.04 | $1.0 \times 10^4$ | 0.05 |
| Horizontal Path | 308.7 | $2.0 \times 10^4$ | 0.005 |

which has the characteristics similar to deep groundwater, was selected for this assessment. The most significant characteristics of the groundwater selected for this analysis are presented in Table A.5. Also included in the table, for comparison purposes, are ranges of characteristics of deep groundwater and of uranium mine water reported in a study of radioactive waste disposal (KBS, 1978). The literature values indicate that the selected groundwater analysis is representative of conditions expected for deep disposal locations. Solubilities for individual radionuclides based upon that groundwater were calculated using the PHREEQE (Parkhurst et al., 1988) computer code and thermodynamic data from the CODATA data set maintained by the International Atomic Energy Agency (Muller, 1985). The calculation procedure involves identifying dominant solid phases, and the sum of the concentrations of all aqueous forms of an element is reported as the solubility of that element. The results of the calculations are presented in Table A.6.

Table A.5 Characteristics of Ground Water Used for Deep Disposal

| Constituent | Selected Ground Water | Deep Ground Water Ranges | Uranium Mine Water Ranges |
|---|---|---|---|
| $Cl^-$ (mg/L) | 4.4 | 5-50 | 5 - 16 |
| $HCO_3^-$ (mg/L) | 205.7 | 60-400 | 183 - 441 |
| $SO_4^{-2}$ (mg/L) | 178.2 | 1-15 | 15 - 863 |
| pH | 7.8 | 7.2-8.5 | 6.5 - 7.8 |
| eH (mv)* | -100 | - | 60 - -89 |

* *Redox Potential*

Table A.6 Calculated Solubilities of Elements in Selected Deep Ground Water[a]

| Nuclide | Solubility (mg/L) | Dominant Solid Phase | Dominant Aqueous Phase |
|---|---|---|---|
| Uranium | $1x10^{-4}$ | $UO_2$ | $U(OH)_4$ |
| Thorium | $5x10^{-13}$ | $ThO_2$ | $Th(OH)_3^+$ |
| Radium | $1x10^{-2}$ | $RaSO_4$ | $Ra^{+2}$ |

[a]Deep groundwater has the characteristics given in Table A.5.

Radionuclide Transport

Groundwater seeping vertically through the disposal facility is assumed to carry dissolved radionuclides upward to a more permeable unit (aquifer). After entering the aquifer, the radionuclides are dispersed upward and transported horizontally through the aquifer by the predominantly horizontal flow. The effects of mixing in the horizontal flow are represented by inclusion in the model equations of a term for hydrodynamic dispersion. The magnitude of the mixing is quantified by longitudinal and transverse dispersion coefficients whose values for fractured granite and for sandstone formations are estimated based on review of field experiments (Waldrop, 1985). Values of longitudinal dispersion coefficients of 30.9 and 61.7 $m^2/yr$ were selected for the granite and sandstone/basalt sites, respectively. Values adopted for transverse dispersion coefficient are one-tenth the magnitude of the longitudinal coefficients based on data review (Waldrop, 1985) and theoretical considerations (Bear, 1972). Radionuclides dissolved in groundwater are adsorbed and exchanged through contact with the surrounding solid phase and thus travel at a lower velocity than the groundwater. Experimental observation of uranium, thorium, and radium in fractures at a mine site (Dearlove et al., 1989) and at hard rock sites in general (KBS, 1978 and National Research Council, 1983) indicate that the ratio of water to radionuclide velocity for these radionuclides ranges from several thousands to tens of thousands in these environments. The recommendation (KBS, 1978) of retardation coefficients greater than 1,200 for uranium, thorium, and radium was adopted for this assessment.

Impacts of Deep Disposal of $U_3O_8$

Potential radiological exposure pathways related to emplacing $U_3O_8$ in deep geological environments include consumption of drinking water, irrigated crops, and fish. Under expected conditions the groundwater would discharge to a river prior to intake. In this analysis, the river is assumed to be located 5 kilometers from the disposal facility. Under conditions which are not expected to occur, an individual would obtain water by drilling a deep well downgradient from the disposal facility. The analysis established that maximum dose for the mid-aquifer elevation well take-off point would occur at a distance of 200 meters at both the granite and sandstone/basalt sites.

The analysis considers radionuclides present in the emplaced $U_3O_8$ and radionuclides which may be produced by decay of parent radionuclides during transport. Because of low solubility and short half-lives, decay daughters of radium originating at the disposal facility would not make a significant contribution to dose at the 200 m well and river locations. Due to the relatively high solubility of radium and the relatively low solubility of uranium and thorium, radium originating at the disposal facility and its shorter half-life daughters growing in during transport would dominate dose at the 200 m well location. At the 5 kilometer river location, radium originating at the disposal facility would have decayed to comparably insignificant levels and radionuclide concentration levels would be controlled by the uranium isotopes. The assumptions of secular equilibrium of the daughters of radium with radium at the 200-meter well and other daughters of uranium with uranium at the 5 kilometer river locations are applied to assess the contribution of daughter ingrowth during transport.

Estimates of doses for the well scenario for the granite and sandstone/basalt sites are presented in Table A.7. Dose estimates for the river scenario for both sites are presented in Table A.8. At the 200-meter well, the parent radionuclide which dominates the estimated dose is $^{226}Ra$, while at the river, $^{238}U$ is the dominant radionuclide. Each of these estimates is the maximum annual dose that would be predicted for any time in the future.

For all of the results presented, estimated impacts are less than the 0.25 mSv/yr (25 mrem/yr) level adopted from 10 CFR Part 61 as a basis for comparison. The assumptions applied in this analysis, including neglect of engineered barriers, inventory limitations, mass transfer limitations in release, and decay and retardation during vertical transport contribute to a conservative analysis.

**Table A.7 Estimated Peak Doses for Well Scenario (Sv/yr)**

| | Granite Site | | Sandstone/Basalt Site | |
|---|---|---|---|---|
| Nuclide | Drinking Water Dose | Agricultural Dose | Drinking Water Dose | Agricultural Dose |
| $^{238}U$ | $5.0 \times 10^{-14}$ | $2.3 \times 10^{-15}$ | $2.3 \times 10^{-20}$ | $1.0 \times 10^{-21}$ |
| $^{234}U$ | $5.3 \times 10^{-14}$ | $1.6 \times 10^{-15}$ | $2.5 \times 10^{-20}$ | $7.5 \times 10^{-22}$ |
| $^{230}Th$ | $3.1 \times 10^{-17}$ | $1.8 \times 10^{-16}$ | $1.6 \times 10^{-23}$ | $9.0 \times 10^{-23}$ |
| $^{226}Ra$ | $4.9 \times 10^{-8}$ | $1.3 \times 10^{-6}$ | $4.0 \times 10^{-11}$ | $1.0 \times 10^{-9}$ |
| $^{226}Ra$ Daughters | $1.1 \times 10^{-7}$ | $1.0 \times 10^{-6}$ | $8.8 \times 10^{-11}$ | $8.0 \times 10^{-10}$ |

## Table A.8 Estimated Peak Doses for River Scenario (Sv/yr)

| Nuclide | Granite Site | | Sandstone/Basalt Site | |
|---|---|---|---|---|
| | Drinking Water Dose | Fish Ingestion Dose | Drinking Water Dose | Fish Ingestion Dose |
| $^{238}U$ | $2.6 \times 10^{-17}$ | $3.7 \times 10^{-17}$ | $7.9 \times 10^{-16}$ | $1.1 \times 10^{-15}$ |
| $^{234}U$ | $2.9 \times 10^{-17}$ | $4.1 \times 10^{-17}$ | $8.8 \times 10^{-16}$ | $1.2 \times 10^{-15}$ |
| $^{230}Th$ | $5.6 \times 10^{-17}$ | $1.6 \times 10^{-16}$ | $1.7 \times 10^{-15}$ | $4.7 \times 10^{-15}$ |
| $^{226}Ra$ | $1.3 \times 10^{-16}$ | $2.6 \times 10^{-16}$ | $4.0 \times 10^{-15}$ | $7.8 \times 10^{-15}$ |
| $^{226}Ra$ Daughters | $2.9 \times 10^{-16}$ | $5.1 \times 10^{-16}$ | $8.8 \times 10^{-15}$ | $1.5 \times 10^{-14}$ |

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