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National Radiation Exposures and Risks Caused by Implementing EPA's Proposed Revised National Primary Drinking Water Regulations

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ABSTRACT

This report estimates risks to workers and the public associated with treatment processes and their associated waste products that would be mandated under proposed regulations of radium, radon, and uranium in drinking water. Three scenarios were examined: (1) all wastes flushed to the sanitary sewer; (2) all wastes disposed on land; (3) similar to (2) but radon removal by granulated activated carbon rather than packed tower aeration. Risks considered included accidental injury and cancer. Worker risks for both scenarios I and II were estimated to be 0.025 and 0.01 deaths per year of operation for radium-226 and radium-228, respectively. Worker risks for uranium were estimated to be 0.13 deaths/year of operation for scenario I and 0.5 deaths/year of operation for scenario II. Worker risks for radon removal were estimated with be 1.7 deaths/year of operation for scenario I and 2.2 deaths/year of operation for scenario II. Risks to the public for scenarios I and II for radium-226 were 4×10^{-4} and for radium-228 were 9×10^{-5} deaths/year of operation. Risks to the public for scenarios I and II for uranium were 7.3×10^{-2} and 2×10^{-4} , respectively. Risks to the public for scenario I and II for radon were 24 deaths/year of operation and for scenario III were nil. Public risks were quantified only for people exposed during a year of operation. For example, effects of public exposures in future years via groundwater contamination associated with landfill of treatment waste were not considered.

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1. INTRODUCTION

To meet revised maximum contaminant levels (MCLs) in proposed revised national primary drinking water regulations for radionuclides (56 FR 33050, July 18, 1991), EPA projects that operators of water supply systems will need to construct thousands of additional water treatment systems applying best available technology (BAT) to remove hazardous radiological constituents. EPA calculates the risks they believe will be avoided by consumers of water from these systems as a result of this treatment. EPA ignores, however, many potential sources of risk imposed by compliance with the MCLs. These risks include:

- Risks to workers associated with the construction of water treatment systems;
- Risks to workers operating the water treatment processes;
- Risks to workers handling sludges and other brines and solid wastes associated with the treatment systems;
- Risks to the public living near the water treatment plant (from radon emissions);
- Risks to workers and the public associated with the transport of solid wastes from water treatment systems to disposal facilities;
- Risks to workers and the public at disposal facilities.

EPA suggests that some of these wastes might be discharged to sanitary sewers. There are no Federal regulations that specifically limit discharge of water treatment plant wastes into sanitary sewers, but regulations do prohibit discharges that would cause the sewage plant to violate an NPDES permit or would interfere with sewage plant operation or sludge disposal (EPA, 1990). In areas that are sewered, only about 60% of the water processed in treatment plants enters the sewerage system. Flushing *all* radioactive material from all water processed directly to the sewer would result in a factor of 1.7 increase in radioactive material in the sewage (neglecting the role of possible infiltration in sewers). In the case of ion exchange, however, a substantial fraction of the radioactive material may be permanently complexed with the ion exchange medium, eventually degrading its effectiveness. The degraded medium will be disposed in a landfill. If the amount of material permanently complexed and diverted to a landfill is greater than 60% of the radioactive material in the raw water, then the amount of radioactive material in the influent to the sewage plant will decrease rather than increase. Risks associated with a change in radioactive material reaching the sewage treatment plant include:

- Risks to sewer workers;
- Risks to sewage plant workers;
- Risks to workers and the public in the disposal of sewage sludge;
- Risk to the public from sewage plant discharges to surface waters.

This report attempts to calculate these new risks imposed by treatment processes that would be mandated under the proposed regulation for uranium, radium, and radon. Three scenarios are examined.

- I. The first scenario assumes EPA's proposed BATs are constructed on all facilities exceeding the standards. Treatment for radon is limited to packed tower aeration. All liquid wastes generated in treatment processes are disposed to the sanitary sewer. Spent ion-exchange media are landfilled.
- II. The second scenario is the same as the base case, but it assumes that discharges to the sanitary sewer are prohibited. All waste generated by treatment processes are assumed to be landfilled at an off-site disposal facility. Risks associated with transport and disposal facilities are added, but no additional risks are associated with the sewage treatment plant (indeed, risks associated with sewage treatment are lowered due to the diversion of radioactive materials in the water).
- III. The third scenario is the same as the discharge prohibited scenario, except that treatment of radon is assumed to require granulated activated carbon (GAC) instead of packed tower aeration. This adds additional worker risk at the water treatment plant, and additional risk associated with packaging, transport and disposal of wastes, but eliminates the risk posed to the surrounding population from the air discharge from the packed tower aeration treatment process.

The background information EPA used as the basis for its calculations supporting the proposed standard was extrapolated from a small sample of water treatment systems. No information was available on factors such as population density, distance to landfills, hydrogeological characterization of landfill sites, what kind of sewage collection and treatment facilities exist, and the character of the water body receiving the sewage discharge. All of these are important in assessing potential risk and estimates must be made. The highly generic and preliminary nature of the information available dictates the level of risk analysis that is appropriate. Although we generally prefer probabilistic methods to explicitly include uncertainties in a risk analysis, in this case the variability of potential exposure situations was too poorly known to estimate probability distributions with any confidence. Typical or illustrative values seemed a better way to convey a sense of the level of risk without providing a false sense of our degree of understanding. We have made reasonable assumptions where appropriate, used typical or illustrative values where averages were not available, and parameterized the results where we felt the variability among sites was likely to be too large to justify broad assumptions. These assumptions are stated explicitly in the following section.

2. CHARACTERIZATION OF FACILITIES AND PROCESSES

A wide variety of facilities and treatment processes are likely to be constructed and operated. For purposes of this analysis, assumptions were made to characterize facilities and processes. These assumptions are described below.

2.1 Water Treatment plants

The analysis includes water treatment plants that use various processes for removal of radioactive materials from water. The number of water supplies of various sizes that will be required to remove uranium, radium, and radon, as well as the number of people served by these systems, was estimated from EPA's censored lognormal extrapolation of a stratified 2.5% sample of the nation's water supplies (Wade Miller, 1990a, 1990b). The estimates were made by interpolation of EPA's tables, or by direct estimation using the parameters of the lognormal distribution given by EPA in the cited references. Smaller systems were aggregated to a greater degree than in EPA's analyses, but larger systems were kept at a disaggregated level because of the wide variation in system size among the large and very large systems.

Characterization data for water supply systems and treatment plants are given in Tables 1-5. The average number of people served by each subgroup was calculated from the data in the tables and water use per system in million gallons per day (MGD) was estimated assuming 140 gallons per capita per day. Ground water supply systems up to substantial size can be operated by one person (personal communication, K. Keeley, American Water Works Association). We assumed that very small systems (serving less than 500 people) can be maintained by a part-time person. We also assumed that, because of the extra effort, sampling, and analysis requirements, addition of treatment processes under the proposed regulation would require an extra person. Similarly, because of the additional treatment processes typically required to treat surface water sources, we assumed higher staffing levels for those systems.

Table 1. Characterization of Treatment Systems - Uranium in Ground Water						
Population Served	Number of Systems	People/ System	Average MGD	Capital Cost - BA	Avg No. Workers	Total No. Workers
25-500	1224	148	0.02	\$75,000	1	1224
501-3,300	214	1360	0.2	\$250,000	2	428
3,300-10,000	18	5767	0.9	\$330,000	2	36
10,001-25,000	1	23,000	3.4	\$130,000	2	2
> 25,001	none					
Total	1457					1690

Table 2. Characterization of Treatment Systems - Uranium in Surface Water						
Population Served	Number of Systems	People/ System	Average MGD	Capital Cost - BA	Avg No. Workers	Total No. Workers
25-500	51	409	0.06	\$40,000	2	102
501-3,300	34	3127	0.5	\$110,000	3	102
3,300-10,000	8	8824	1.3	\$100,000	3	24
10,001-25,000	1	22,214	3.3	\$250,000	3	3
25,001-50,000	1	48506	7.2	\$550,000	5	5
> 50,000	none					
Total	95					236

Table 3. Characterization of Treatment Systems - Radon-222 in Ground Water						
Population Served	Number of Systems	People/ System	Average MGD	Capital Cost - BAT	Avg No. Workers	Total No. Workers
25-500	23789	126	0.02	\$24,000	1	23789
501-3,300	3284	1319	0.20	\$70,000	2	6568
3,300-10,000	584	5771	0.86	\$100,000	2	1168
10,001-25,000	204	15,961	2.38	\$280,000	2	408
25,001-50,000	85	28529	4.25	\$500,000	2	170
50,000-75,000	20	63300	9.43	\$1,100,000	2	40
75,000-100,000	5	75200	11.20	\$1,300,000	2	10
100,000-500,000	12	179000	26.67	\$3,100,000	2	24
>500,000	none					
Total	27983					32177

Table 4. Characterization of Treatment Systems - Radium-226 in Ground Water						
Population Served	Number of Systems	People/ System	Average MGD	Capital Cost - BAT	Avg No. Workers	Total No. Workers
25-500	18	122	0.02	\$75,000	1	18
501-3,300	2	1925	0.29	\$250,000	2	4
3,300-10,000	31	5926	0.88	\$330,000	2	62
10,001-25,000	18	16,278	2.43	\$940,000	2	36
25,001-50,000	6	35500	5.29	\$2,055,000	2	12
50,000-75,000	2	53000	7.90	\$3,070,000	2	4
75,000-100,000	0	0	0.00	\$0	2	0
100,000-500,000	1	170000	25.33	\$3,100,000	2	2
> 500,000	none					
Total	78					138

Table 5. Characterization of Treatment Systems - Radium-228 in Ground Water						
Population Served	Number of Systems	People/ System	Average MGD	Capital Cost - BAT	Avg No. Workers	Total No. Workers
25-10,000	40	4100	0.61	\$330,000	1.5	60
501-3,300	none					

Best available technology (BAT) was assumed for all systems exceeding the proposed MCLs. We assumed that ground water systems and small surface water systems would use ion exchange processes for uranium and radium while medium and large surface water systems would use coagulation and filtration to meet BAT requirements for uranium (the only radioactive material for which EPA expects BAT removal will be required for surface water systems). In the first two scenarios, packed tower aeration (PTA) was assumed to be the BAT choice for radon removal. The third scenario substituted granulated activated carbon (GAC) as the BAT choice for radon removal.

Risks of treatment are largely related to the amount of radioactive material removed. Extrapolated estimates were available on the number of water systems having given concentrations of each radionuclide considered. But treatment processes and associated risk also can depend on total water flow through the plant and on other factors. In general, we used average or typical levels of operation in risk calculations and multiplied by the number of water systems requiring treatment rather than examining separately the levels of treatment at plants having different raw water concentrations.

Estimates of accidental injury during treatment plant construction were based on capital cost. The capital cost of the BAT processes by size of system was estimated from EPA (FR 56: 33050-33127, July 18, 1991). Estimates for the individual categories of large systems were made in proportion to the population served.

2.2 Handling of Brines, Sludges and Solid Wastes at Water Treatment Plants

Ion exchange processes require frequent back-washing, producing a liquid brine waste water. Mangelson and Lauch (1990), point out that ion exchange in water treatment using standard water-softening type resins is well understood, but may be unsatisfactory for radium removal. They describe a demonstration of a two-stage process for radium removal in which the regeneration (backwash) water for the first ion exchange process passes through a bed of radium-selective complexer (RSC) resin. Their demonstration plant had a raw water concentration of radium of 35 to 40 pCi/L. Based on the process flows described by Mangelson and Lauch (1990), it was assumed that 79% of radium is permanently complexed on the ion-exchange resins, 12% is in the backwash water, and 8% remains in the treated water. There is some confusion in the paper, a flow diagram suggesting that backwash water was about 2.5% by volume of input water, while the accompanying table suggests 0.15%. Mangelson and Lauch (1990) use the latter to estimate the useful life of the resin; we use their

brines and sludges are flushed to a lined evaporation pond through a completely enclosed piping system. It was assumed that the residuals are gathered mechanically (e.g., by front-end loader) and either packaged in containers or transported in bulk in covered trucks. The mode by which wastes are transported will affect the number of shipments and the effects at the disposal site.

2.3 Transport and Disposal of Water Treatment Plant Wastes

It was assumed that residuals from sludges and dried brines will be transported from the water treatment plant to an appropriate landfill (when called for in a scenario) in closed containers or in covered trucks. It was assumed that exhausted ion-exchange resins will be dewatered or mixed with cement and packaged in steel drums. The former is the most prevalent form of disposal of ion exchange resins in the nuclear power industry. In both cases, transport to an appropriate landfill by truck was assumed. Because most of the water systems requiring treatment for removal of radioactive materials under the proposed regulation are small, packaging and transport may be done by an outside contractor rather than plant personnel. We do not distinguish between contractors and treatment plant employees in our calculation of occupational exposure and risk.

Table 6. Range of travel distance to disposal sites by type (from Cohen and Eastern, 1992).	
Type of Disposal Facility	Range of Travel Distance (miles)
Municipal Incinerator	30 -200
Municipal Landfill	10 - 1000
Hazardous Waste Disposal	60 - 600
Low-Level Radioactive Waste	100 - 2500
Source: Cohen and Eastern (1992) except for low-level radioactive waste, for which the distance to Tooele County, Utah was calculated.	

Disposal sites may include municipal incinerators or landfills, hazardous waste disposal facilities, or low-level radioactive waste disposal sites, depending on the character of the waste and Federal, State, or local regulations (see further discussion below on disposal site options). The distance from treatment plant to landfill site likely varies considerably based on the location of the water treatment plant and the appropriate classification of landfill. Municipal landfills are likely to be much closer to treatment plants than are hazardous waste or low-level waste disposal sites. Ranges of travel distance to different disposal facilities are shown in Table 6 (Cohen and Eastern, 1992).

2.4 Sewerage Systems and Sewage Treatment Plants

For Scenario I, sludges and brines are disposed to the sanitary sewer. Many of the small water supply systems that require treatment undoubtedly service areas that are not served by sanitary sewers. Flushing to a sanitary sewer, in these cases, would probably mean disposal to an underground septic system, an inappropriate approach. Lack of a sewage treatment plant would virtually require an alternative disposal technique, presumably forcing one to the approach of Scenario II. For the purposes of this analysis, in Scenario I we assumed the existence of sewage treatment plants. The amount of radioactivity in the sewage plant influent would be altered by an amount depending on the treatment process. This will change the exposure of sewer workers and sewage treatment plant workers. It will also change the public exposure downstream of the plant effluent, via swimming or other water recreation or by uptake through the food chain. This effect will also depend on the size of the stream or other water body receiving the sewage plant effluent. Since this must vary considerably among treatment systems, we parameterize our calculations over a range. We assumed existing methods of handling and disposal of sewage sludge is unchanged, that is, that the change in radioactivity in the sludge does not lead to any requirement to modify existing practice. We consider exposure and risk of three alternative methods of sewage sludge disposal: landfill, land application, and incineration. About 42% of sewage sludge is landfilled, 21% incinerated, and 16% applied to land (Reale, 1991).

3. EXPOSURE AND RISK IN SCENARIO I: BASE SCENARIO

3.1 Accidental Injury in Facility Construction and Operation

We generalized the labor component of costs from data in the EPA documents supporting the proposed rule (Gumerman et al., 1984; Hansen et al; 1979) to obtain the relationship between construction labor requirements and system size shown in Figure 1. These were assumed for this analysis to be representative of all such systems. We took this approach because capital and operating costs calculated by Culp/Wesner/Culp for EPA were inadequate to derive the labor component of the specific costs reported by EPA (Cummins, 1992).

Occupational health risks for building and operating sanitary facilities in 1990 are shown in Table 7. The statistics combine operational fatality rates for public utilities with those for transportation, which are much higher, so we assumed that the fatality rate for operating sanitary facilities is like that for the Service Sector, which is relatively low. The risk factors from Table 6 were combined with the labor requirements of Figure 1 to yield the estimates of occupational risks of building and operating treatment facilities for radionuclides in drinking water shown in Table 8. Total worker-days lost for all systems are estimated to be about 9,000 during construction and 8,000 per year during operation. Total occupational fatalities for all systems are estimated to be less than one during construction and 1.7 per year during operation.

CONSTRUCTION LABOR

Figure 1a. Derived relationship between water treatment plant capacity and construction labor required.

OPERATION LABOR

Figure 1b. Derived relationship between water treatment plant capacity and operating labor required.

Table 7. Occupational Risks in Construction and Operation of Sanitary Services, 1990.

Industry	SIC Code	Lost Workday Cases	Lost Workdays	Fatalities
CONSTRUCTION				
Nonresidential building construction	154	7.0	148.2	
Heavy construction, except highways	162	6.2	142.3	
Special trade contractors	17	6.9	153.1	
<u>Mean</u>		<u>6.7</u>	<u>148</u>	<u>0.015</u>
OPERATION				
Water supply	494	9.4	101.8	0.005

SOURCE: Survey of Occupational Injuries and Illnesses, 1990. Bureau of Labor Statistics, U.S. Department of Labor, Washington, DC. January 1992.

Table 8. Health risks of building and operating treatment facilities for radionuclides in drinking water.

POPULATION SERVED (1000's)										
0.025-0.1	0.1-0.5	0.5-1.0	1.0-3.3	3.3-10	10-25	25-50	50-75	75-100	100-500	SUM

DESIGN CAPACITY (mgd)^a

0.06 0.14 0.31 0.96 3.1 7.5 15 25 44 124

NUMBER OF SYSTEMS

Uranium^b

Ground	754	470	114	100	18	1					1,457
Surface	28	23	15	19	8	1	1				95
Radium ^c	32	20	4	4	31	18	6	2	0	1	118
Radon ^d	14,652	9,137	1,755	1,529	584	204	85	20	5	12	27,983

LABOR REQUIREMENTS FOR CONSTRUCTION (Worker-years = 0.42x[MGD]^{0.408})

Uranium

Ground	100	89	30	41	12	1					273
Surface	4	4	4	8	5	1	1				27
Radium	4	4	1	2	21	17	8	3	0	3	62
Radon	953	1,721	457	632	387	195	109	31	10	36	5,530

Table 8 (Cont.).

POPULATION SERVED (1000's)											
	0.025-	0.1-	0.5-	1.0-	3.3-	10	25	50	75	100	
	0.1	0.5	1.0	3.3	10	25	50	75	100	500	SUM

WORKER FATALITIES IN CONSTRUCTION (@ 0.0015 Fatalities/100 Worker-Yr)

Uranium											
Ground											0.04
Surface											
Radium											0.01
Radon	0.3	0.2									0.8
TOTAL											0.88

WORKER DAYS LOST IN CONSTRUCTION (@ 148 WDL/100 Worker-yr)

Uranium											
Ground	149	131	44	61	18	1					404
Surface	6	6	6	12	8	1	2				40
Radium	6	6	2	2	30	25	11	5	0	4	92
Radon	2,890	2,546	676	935	573	289	161	46	15	53	8,185
TOTAL											8,721

LABOR REQUIREMENTS FOR OPERATION AND MAINTENANCE

(Worker-years/yr = $0.38 \times [\text{MGD}]^{0.158}$)

Uranium											
Ground	184	131	36	38	8	1					397
Surface	7	6	5	7	4	1	1				30
Radium	8	6	1	2	14	9	4	1	0	1	45
Radon	3,570	2,545	554	577	265	107	50	13	3	10	7693

Table 8 (Cont.).

POPULATION SERVED (1000's)										
0.025-	0.1-	0.5-	1.0-	3.3-	10	25	50	75	100	
0.1	0.5	1.0	3.3	10	25	50	75	100	500	SUM
ANNUAL WORKER FATALITIES IN OPERATION (@ 0.005 Fatalities/100 Worker-Yr)										
Uranium										
Ground										0.08
Surface										0.01
Radium										0.01
Radon	0.7	0.4	0.1	0.1						1.6
TOTAL										1.71

ANNUAL WORKER DAYS LOST IN OPERATION (@ 101.8 WDL/100 Worker-yr)

Uranium									
Ground	768	478	232	204	37	2			1,720
Surface	57	47	46	58	24	3	5		240
Radium	33	20	8	8	63	37	12		181
Radon	14,916	9,301	3,573	3,113	1,189	415	173		32,681
TOTAL									34,822

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- a. Malcolm Pirnie, Inc., 1987.
 - b. Wade Miller Assoc. Inc, 1991.
 - c. Wade Miller Assoc. Inc, 1990a
 - d. Wade Miller Assoc. Inc, 1990b
-

3.2 Exposure and Risk at Water Treatment Plant

3.2.1 Backwash of Ion-Exchange Resins

Backwashing is entirely contained in the piping system. Radiation exposure to workers from this source was assumed to be insignificant. In this Scenario, the backwash water is flushed to the sanitary sewer.

3.2.2 Maintenance of Process Equipment in Water Treatment Plant

Waters containing high radium concentrations are most likely to be hard water, producing scale on piping and the interior of processing equipment. This scale is likely to concentrate radium and may pose a risk to workers involved in maintenance. Even in oil field operations, however, it has been observed that there is little risk to workers as long as the radioactive materials remain within pipes and equipment (White, 1992). Moreover, upstream of the treatment process, any scaling effect would be unchanged from before application of the proposed rule. Holding tanks for backflush water and piping directing backflush water from ion-exchange units to sewers or evaporation ponds may be subject to increased scaling. If any risk exists, it is more likely in disposal of the pipe than to the workers in the plant. Possibilities of the latter are too speculative to warrant exclusion here. Water treatment plants of the size of concern here are generally too small to have pipe cleaning yards where exposures can occur, nor are process vessels large enough that workers will enter them for maintenance. We assumed no extra exposure in plant maintenance.

3.2.3 Ion-Exchange Treatment Processes for radium-226 - Direct Radiation.

Mangelson and Lauch (1990) describe a full-scale demonstration project of radium removal using a two-stage ion-exchange process for a water supply system processing 100 gallons per minute (0.144 MGD). From their data, we calculate it is necessary to backwash every 7.5 hours for this throughput, using about 2.5% by volume of processed water as backwash. They reported that the radiation level 3 inches from the radium-selective complexer (RSC) resin tank was 0.813 mrem/h and estimated that a plant worker spends 3.5 hours/week at this location. This totals 142 mrem/y, assuming a 50 week work year, or 6.4 rem over a 45-year working lifetime. This clearly is not a whole-body dose, but is probably not too much of an overestimation. Applying a lifetime cancer risk factor of 5×10^{-4} cancers per man-rem (ICRP60, 1991) gives a lifetime risk of 3.2×10^{-3} , a risk usually considered above the threshold of regulatory concern. Extrapolating to the estimated 138 workers of the 78 affected water systems yields 19.6 man-rem per year and 0.01 lifetime cancers produced per year of operation.

Mangelson and Lauch's (1990) estimate of the duration of worker exposure appears high. It would be necessary to reduce it to 6 minutes/week, however, to put it within an acceptable range (below a risk of 10^{-4}). Because of the variability among plants, further uncertainty is introduced when extrapolating to all treatment plants. In addition, if an unacceptable exposure situation exists, changes in operational procedures or equipment design might mitigate it.

3.2.4 Replacement of Ion-Exchange Resin for radium-226 and Packaging of Exhausted Resins.

Removing and handling the depleted (radium-saturated) ion-exchange resins involves some degree of exposure at a rate higher than the 0.8 mrem/h outside the RSC tank. We consider two options for packaging of these waste resins: solidification with concrete and dewatering. Mangelson and Lauch (1990) describe a treatment plant in which these resins are placed in a 55-gallon drum, mixed with an almost equal amount of concrete, then loaded on a truck. We assumed that two workers will be involved in the operation. Because of the weight of the materials, most of the handling will be done with mechanical assistance. Assuming there will be no particular effort to shield the workers from radioactivity, some handling or other close contact is likely. The radioactivity level 3 inches from the drum is assumed to be half of that measured at the RSC tank, or about 0.4 mrem/h. Even higher exposures might be experienced depending on what shielding, if any, exists in the operation of transferring resins from the RSC tank to the waste container. The entire process is assumed to take about 0.5-1 hour for each 4 cubic foot resin package and to involve an exposure of 0.5-1 mrem for both workers. As discussed in section 2, this operation was assumed to be required for every 400 million gallons of water processed. The number of resin changes required per year, associated exposure and lifetime cancer risk nationwide is:

$$\text{Resin Changes/y} = (\text{number of systems}) \times \text{MGD} \times 365 \text{ days/y} / 400 \text{ MG}$$

$$\text{Man-rem/y} = (\text{resin changes/y}) \times 2 \text{ workers} \times (\text{exposure level})$$

$$\text{Lifetime cancer risk} = \text{man-rem/y} \times 5 \times 10^{-4} \text{ lifetime cancers per man-rem}$$

Total national exposure and cancer risk to water treatment plant workers per year of operation from this source is calculated in Table 9. From this table, average individual risk ranges from 4×10^{-7} to 8×10^{-7} and nationwide lifetime cancers per year of operation from this source are estimated to range from 7×10^{-5} to 1.3×10^{-4} .

If the resins are dewatered, we assume a process similar to that described in section 2 above for nuclear power plants may be used in water treatment plants. The process itself is highly automated. In most cases, these containers would be smaller (perhaps 25% of the size) than the 50-cubic foot containers used in power plants. The principal source of occupational exposure would be in handling, storage and loading of the solidified material in the containers. As a rough estimate, we assume exposure levels would be at the low end of the range assumed for solidification with concrete, or about 0.5 mrem per resin exchange for 2 workers. The total occupation exposure would be 0.13 man-rem per year of operation nationally, associated with 7×10^{-5} lifetime cancers.

Table 9. Radiation exposure and Risk in changing ion exchange resins for radium-226 treatment.						
Population Served	Number of Systems	Average MGD	Tot. Resin Changes/y	Man-Rem/Year	Life-time Cancers per Year of Operation	
25-500	18	0.02	0.30	2.99E-04 - 5.97E-04	1.49E-07	2.99E-07
501-3,300	2	0.29	0.52	5.23E-04 - 1.05E-03	2.62E-07	5.23E-07
3,300-10,000	31	0.88	24.98	2.50E-02 - 5.00E-02	1.25E-05	2.50E-05
10,001-25,000	18	2.43	39.84	3.98E-02 - 7.97E-02	1.99E-05	3.98E-05
25,001-50,000	6	5.29	28.96	2.90E-02 - 5.79E-02	1.45E-05	2.90E-05
50,000-75,000	2	7.90	14.41	1.44E-02 - 2.88E-02	7.21E-06	1.44E-05
75,000-100,000	0					
100,000-500,000	1	25.33	23.11	2.31E-02 - 4.62E-02	1.16E-05	2.31E-05
>500,000	none					
Total	78		132.12	0.13 0.26	6.61E-05	1.32E-04

3.2.5 Radium-228 Treatment: Ion-Exchange Treatment Processes - Direct Radiation

Extrapolating from the analysis with radium-226, the same process used to remove radium-228 would produce a radiation level 61% of that produced by radium-226. Calculated results would be reduced in proportion. That is, 88 mrem/y, posing a lifetime cancer risk of 4.4×10^{-5} . This cumulates to 4.0 rem over a 45-year working lifetime for the individual worker, leading to a lifetime risk of 2×10^{-2} . The total effect on the total of estimated 60 workers in the 40 water supply systems requiring control of radium-228 would be 5.28 man-rem per year, producing an estimated 0.003 lifetime cancers per year of operation.

3.2.6 Radium-228 Treatment: replacement of Ion-Exchange Resin and Packaging of Exhausted Resins.

Again extrapolating from the analysis for radium-226, removing and handling the depleted (radium-saturated) ion-exchange resins involves some degree of exposure at a rate higher than the 0.5 mrem/h outside the RSC tank. For solidification with concrete, It was assumed two workers will be involved in the operation. It was assumed that these resins will be placed in a 55-gallon drum mixed with an almost equal amount of concrete (Mangelson and Lauch, 1990) then loaded on a truck. Because of the weight of the materials, most of the handling will be done with mechanical assistance, although it was assumed there will be no particular effort to shield the workers from radioactivity and some man-handling or other close contact is likely. The radioactivity level 3 inches from the drum is assumed to be half of that measured at the RSC tank, or about 0.25 mrem/h. The entire process is assumed to take about 0.5-1 hour for each 4 cubic foot resin package and to involve an exposure of 0.3-0.6 mrem. Given 40 systems averaging 0.61 MGD, resin changes per year would be:

$$40 \times 0.61 \times 365 / 400 = 22.$$

Total occupational exposure nationwide in the 40 plants requiring treatment of radium-228 would be 0.022 to 0.045 man-rem, producing nationwide lifetime cancers per year of operation from this source estimated to range from 1.1×10^{-5} to 2.2×10^{-5} and individual risk ranging from 14×10^{-7} to 28×10^{-7} .

3.2.7 Uranium Treatment: Ion-Exchange Treatment Processes - Direct Radiation

For water systems using ground water, ion exchange appears to be the likely choice for removing uranium. A process similar to that used for radium is assumed. Extrapolating from the analysis with radium-226, the same process used to remove uranium would produce a radiation level at the point measured that is 7.3×10^{-8} times that produced by radium-226 because of the much lower gamma constant. Alpha and beta radiation was assumed not to reach workers at a point 3 inches beyond the steel skin of the RSC tank. Calculated results would be reduced in proportion. This totals 1×10^{-5} mrem/y, assuming a 50-week work year, or 4.6×10^{-4} mrem per 45 year working lifetime. Summing over the 1457 water systems requiring removal of uranium from ground water sources, yields 2.9×10^{-5} man-rem per year of operation. Applying a lifetime cancer risk factor of 5×10^{-4} cancers per man-rem (ICRP60, 1991) to this yields an estimate of 1.5×10^{-8} lifetime cancers produced per year of operation.

3.2.8 Uranium Treatment: Replacement of Ion-Exchange Resin and Packaging of Exhausted Resins.

Applying the same assumptions to replacement of exhausted resins in uranium treatment described above for radium, with the exposure rate lower by a factor of 7.3×10^{-8} , yields the results shown in Table 10. Nationwide lifetime cancers per year of operation associated with workers involved in replacement, packaging, and loading of ion-exchange resins for uranium are estimated to be below 1×10^{-10} .

Table 10. Radiation exposure and Risk in changing ion exchange resins for uranium treatment.							
Number of Systems	Average MGD	Tot Resin Changes/y	Man-Rem/Year		Lifetime Cancers/y Operation		
			Low	High	Low	High	
1224	0.02	22.34	1.63E-09	3.26E-09	8.15E-13	1.63E-12	
214	0.2	39.06	2.85E-09	5.70E-09	1.43E-12	2.85E-12	
18	0.9	14.78	1.08E-09	2.16E-09	5.40E-13	1.08E-12	
1	3.4	3.10	2.26E-10	4.53E-10	1.13E-13	2.26E-13	
1457		79.28	5.79E-09	1.16E-08	2.89E-12	5.79E-12	

3.2.9 Uranium Treatment: Coagulation and Filtration - Direct Radiation

Sedimentation, coagulation and filtration are commonly used treatment processes in surface water systems that can be augmented to meet BAT for uranium

removal. Coagulants are mixed with the water causing a sludge to settle. For Scenario I, it was assumed that this sludge is flushed to a sewer. The latter is assumed for the purposes of this scenario. Water then passes through filters where additional contaminants are removed. Any radium removed at this stage is flushed to a sewer as part of the backflush water. Although coagulation, sedimentation and filtration are often open processes, there is little opportunity for radiation exposure to the operators. Sludge settles to the bottom of sedimentation ponds to be automatically pumped and the filter backwash is usually automatic. Some residual uranium may be included in pipe scaling and further exposure may be associated with other maintenance operations on the system, but the level of exposure will be low and these are infrequent operations.

3.2.10 Radon Treatment: Exposure to Workers from Packed Tower Aeration

Packed tower aeration units for radon gas are designed to produce 90% removal with an air:water ratio of 15:1 (Malcolm Pirnie, Inc., 1987). Thus, the expected concentration of radon in the exit gases from the towers is 0.06 pCi/l per pCi/l in the water treated. Since the design air:water ratio is standard, the concentration in the exit gases depends only on the concentration in the water treated.

Because of the small size of the treatment facilities involved, we assumed that even low wind speeds will carry the exhaust gases from an aeration tower beyond the borders of the facility. Under these conditions, no occupational exposures are produced. Exposures off-site are included in estimates of overall population exposure below.

Occupational exposures occur only when there is almost no wind, in which case the exit gases, which are relative heavy from evaporative cooling, will slump to the ground or be subject to down washing behind buildings. Under these conditions (ignoring recirculation of a small portion of the plume) the concentration of radon gas to which workers are exposed will be essentially the same as the exit concentration from the tower.

Only fresh radon gas is extracted by the aeration tower, so that exposure to radon progeny depends on the age of the plume. Figure 2 shows radon decay and production of radon progeny in a young plume with associated development of working levels of exposure as a function of time (Rowe, 1992). The age of a gravity-slumping plume in an environment full of buildings and similar obstructions is highly location-specific, and essentially impossible to generalize. We assumed for this analysis that the average age of the radon is about 5 minutes, which yields a convenient exposure to radon progeny of 0.001 working level (WL) per pCi/l in the raw water.

Figure 2. Radon decay and production of radon progeny with time.

The frequency of the calm conditions required for occupational exposure to exist ranges from 0 to 10% and is highly location-specific (Murry and Brown, 1988). For this analysis, we assumed that conditions leading to slumping or down washing of the plume occur 5% of the time. This yields an annual dose to a worker who was present during that 5% of the year of 1.5×10^{-4} WLM per pCi/l in raw water. Table 11 shows the distribution of radon concentrations in drinking water supplies from ground water, the expected values of the portions of the distributions in excess of 300 pCi/l estimated using equations for truncated normal distributions (Johnson and Kotz, 1970), and associated mean occupational doses to treatment plant workers under the above assumptions. Worker dose was calculated assuming slumping or down washing of the plume 5% of the time and that workers are always there under these conditions; this is probably an overestimate of exposure. Assuming a risk factor of 2.2×10^{-4} lifetime excess cancer deaths per WLM (Puskin, 1992), Table 12 shows the resulting estimated health risk to workers from exposure to radon progeny in slumping plumes from packed tower aeration systems. Total risk from all such systems is estimated to be 0.22 lifetime excess cancer deaths per year of operation. Slumping conditions occur mostly at night, a time when especially the smaller treatment plants are unlikely to be staffed full time, so this is an upper estimate. Based on manning levels, the risk may range from zero to 0.22.

Table 11. Distribution of radon concentrations in drinking water supplies from ground water.

System Size	Geometric mean radon in water (pCi/l)	Geometric standard deviation	Expected Value > 300 pCi/l	Worker dose (WLM/year)
25-500	243	4.53	934	0.14
501-3,300	124	3.62	645	0.097
3,300-10,000	134	2.54	503	0.075
> 10,000	134	2.52	498	0.075
Source: Wade Miller Assoc., Inc. 1990.				

Table 12. Risk to workers from occupational exposure to radon from packed tower aeration.

System Size	Workers (Person-y/y)	Radon dose (WLM/y)	Total dose (Person-WLM/y)	Lifetime excess cancers per year of operation ^a
25-500	6115	0.14	856	0.19
501-3,300	1131	0.097	110	0.02
3,300-10,000	265	0.075	20	0.004
> 10,000	183	0.075	8	0.002
Total	7693		994	0.22
^a At 220×10^{-6} lifetime excess cancer deaths per WLM exposure to radon progeny (Puskin, 1992).				

3.2.11 Radon Treatment: Exposure to the Public from Packed Tower Aeration

EPA assessed the risk of exposure of the population to radon emission from PTA units (FR 56:33086-33087, July 18, 1991). They estimated a maximum individual risk of 4×10^{-5} and an estimated total of 0.4 cancer cases per year nationwide. Assuming a 70-year average lifetime, this is equivalent to 28 lifetime cancers per year of operation.

3.3 Worker Exposure and Risk in Waste Transport

3.3.1 Radium-226 Transport of Ion-Exchange Resins to Landfill - Worker Exposure and Risk

Spent resins will be mixed with concrete in 55-gallon drums (7.4 cubic feet) or dewatered and solidified in containers of 7-50 cubic feet. For containers of the smaller size, the number of containers to be shipped per year vary from less than one for the smallest plants to over 20 for the largest plant. If disposed in a nearby municipal landfill, transport would probably be by small or medium trucks. Were the transport to be to a more distant location, as would typically be the case for a hazardous waste disposal site or a low-level radioactive waste disposal site (the latter is suggested by Mangelson and Lauch (1990), wastes might be stored on site or at an intermediate facility and waste from several water treatment plants consolidated for shipment in 15 cubic yard (405 cubic feet) containers on large trucks.

We assumed that resins will be transported from the water treatment plant to the landfill by truck with 2 occupants in the cab of the truck, located about 3 feet from the waste container. We assumed this distance is sufficient to apply an inverse-square relationship to the dose. Neglecting any shielding of the cab itself, these workers receive 0.0016 mrem/hour. We assumed an average speed of 53 mph, the average speed on rural collector roads, for trips under 100 miles and 60 mph, the average speed on rural interstate highways, for longer trips (USDOT, 1987). Exposure and risk would be inversely proportional to speed, e.g., if average speed were reduced from 60 to 30 mph, exposure and risk would double. The truck returns from the disposal site empty. Workers are exposed to radiation on the way to the disposal site and exposed to the risk of routine traffic accidents in both directions. Trucks average 1.4×10^{-8} occupant deaths per vehicle mile (USDOT, 1991).

Accidental deaths/trip = Distance * 1.4×10^{-8} occupant deaths/vehicle mile

Lifetime cancer deaths/trip = (distance/mph) x 1.6×10^{-9} rem/h x 2 occupants x 5×10^{-4} cancers/man-rem

where

mph = 53 for distances > 100 miles

mph = 60 for distances 100 miles and over.

The distance between the treatment plant and the land fill varies considerably from place to place depending on local circumstances and the kind of landfill required. The character of the waste, the shielding characteristics of the container, and the safety and other conditions of the trip may vary by the official designation of the waste and/or by

the kind of disposal site. We do not take these factors into consideration since they might vary in unpredictable ways. Dose rate received by the truck occupants could easily vary by a factor of 2 depending on the container and the configuration of the load. Table 13 calculates the radiation exposure, cancer risk and traffic accident risk per trip. For example, if the average distance between a treatment plant and a landfill were 50 miles, the estimated risk per trip of the workers would be 1.5×10^{-9} lifetime cancer and 1.4×10^{-6} from traffic accidents.

Total effects nationwide for the 78 estimated plants depends on the transport strategy. Three options are considered below:

3.3.1.1 Separate Trips for Each Package.

A total of 132 resin exchanges were calculated in Table 9. These packages will be generated at different times and different places. Assuming each package is transported individually to a disposal site, radiation exposure, cancer risk and traffic accident risk nationwide for all 132 trips are calculated in Table 14. For example, if the average distance between a treatment plant and a landfill were 50 miles, the estimated risk of the workers would be 2×10^{-10} for lifetime cancer deaths and 1.8×10^{-4} for traffic accident deaths nationwide for each year of operation.

Table 13. Transport of radioactive materials to landfill, lifetime risk per trip.
5E-4 cancer/man-rem.

Distance (miles)	Lifetime Cancers per Trip			
	All Radionuclides Accidents	Ra-226	Ra-228	Uranium
10	2.80E-07	3.02E-10	1.84E-10	2.20E-17
20	5.60E-07	6.04E-10	3.68E-10	4.41E-17
30	8.40E-07	9.06E-10	5.52E-10	6.61E-17
40	1.12E-06	1.21E-09	7.37E-10	8.82E-17
50	1.40E-06	1.51E-09	9.21E-10	1.10E-16
60	1.68E-06	1.81E-09	1.10E-09	1.32E-16
70	1.96E-06	2.11E-09	1.29E-09	1.54E-16
80	2.24E-06	2.42E-09	1.47E-09	1.76E-16
90	2.52E-06	2.72E-09	1.66E-09	1.98E-16
100	2.80E-06	2.67E-09	1.63E-09	1.95E-16
200	5.60E-06	5.33E-09	3.25E-09	3.89E-16
300	8.40E-06	8.00E-09	4.88E-09	5.84E-16
400	1.12E-05	1.07E-08	6.51E-09	7.79E-16
500	1.40E-05	1.33E-08	8.13E-09	9.73E-16
600	1.68E-05	1.60E-08	9.76E-09	1.17E-15
700	1.96E-05	1.87E-08	1.14E-08	1.36E-15
800	2.24E-05	2.13E-08	1.30E-08	1.56E-15
900	2.52E-05	2.40E-08	1.46E-08	1.75E-15
1000	2.80E-05	2.67E-08	1.63E-08	1.95E-15
1500	4.20E-05	4.00E-08	2.44E-08	2.92E-15
2000	5.60E-05	5.33E-08	3.25E-08	3.89E-15
2500	7.00E-05	6.67E-08	4.07E-08	4.87E-15
3000	8.40E-05	8.00E-08	4.88E-08	5.84E-15

Table 14. Transport of radioactive materials to landfill.

National lifetime risk of 1 year's activity.

	Ra-226		Ra-228		Uranium	
trips/year:	132		22		79	
Distance (miles)	Cancers	Accidents	Cancers	Accidents	Cancers	Accidents
10	3.98E-11	3.70E-05	6.64E-12	6.16E-06	2.38E-11	2.21E-05
20	7.97E-11	7.39E-05	1.33E-11	1.23E-05	4.77E-11	4.42E-05
30	1.2E-10	1.11E-04	1.99E-11	1.85E-05	7.15E-11	6.64E-05
40	1.59E-10	1.48E-04	2.66E-11	2.46E-05	9.54E-11	8.85E-05
50	1.99E-10	1.85E-04	3.32E-11	3.08E-05	1.19E-10	1.11E-04
60	2.39E-10	2.22E-04	3.98E-11	3.70E-05	1.43E-10	1.33E-04
70	2.79E-10	2.59E-04	4.65E-11	4.31E-05	1.67E-10	1.55E-04
80	3.19E-10	2.96E-04	5.31E-11	4.93E-05	1.91E-10	1.77E-04
90	3.59E-10	3.33E-04	5.98E-11	5.54E-05	2.15E-10	1.99E-04
100	3.52E-10	3.70E-04	5.87E-11	6.16E-05	2.11E-10	2.21E-04
200	7.04E-10	7.39E-04	1.17E-10	1.23E-04	4.21E-10	4.42E-04
300	1.06E-09	1.11E-03	1.76E-10	1.85E-04	6.32E-10	6.64E-04
400	1.41E-09	1.48E-03	2.35E-10	2.46E-04	8.43E-10	8.85E-04
500	1.76E-09	1.85E-03	2.93E-10	3.08E-04	1.05E-09	1.11E-03
600	2.11E-09	2.22E-03	3.52E-10	3.70E-04	1.26E-09	1.33E-03
700	2.46E-09	2.59E-03	4.11E-10	4.31E-04	1.47E-09	1.55E-03
800	2.82E-09	2.96E-03	4.69E-10	4.93E-04	1.69E-09	1.77E-03
900	3.17E-09	3.33E-03	5.28E-10	5.54E-04	1.90E-09	1.99E-03
1000	3.52E-09	3.70E-03	5.87E-10	6.16E-04	2.11E-09	2.21E-03
1500	5.28E-09	5.54E-03	8.80E-10	9.24E-04	3.16E-09	3.32E-03
2000	7.04E-09	7.39E-03	1.17E-09	1.23E-03	4.21E-09	4.42E-03
2500	8.8E-09	9.24E-03	1.47E-09	1.54E-03	5.27E-09	5.53E-03
3000	1.06E-08	1.11E-02	1.76E-09	1.85E-03	6.32E-09	6.64E-03

3.3.1.2. Consolidation of Waste Packages.

Each resin exchange yields about 8 cubic feet of waste for transport. Instead of shipping the waste from each exchange individually to a disposal site, waste from several water treatment plants might be consolidated. The annual waste volume is

$$132 \text{ packages} \times 8 \text{ cubic feet/package} = 1056 \text{ cubic feet} = 39 \text{ cubic yards.}$$

If all the waste could be consolidated, it would require only 2.6 trips per year in 15 cubic yard containers. Since the water treatment plants are dispersed, consolidating all the waste would be the same as independent shipments to a disposal site. Consolidation also implies unloading, handling, storage, and reloading at the

consolidation points adding additional exposure (perhaps doubling or tripling the exposure from handling and loading calculated in Table 9). In addition, risk associated with transport from the water treatment plants to the consolidation point must be included. There might be consolidation among several local water treatment plants to a regional disposal site or consolidation among all plants within a region for transport to a remote disposal site. A wide range of possibilities exist.

To explore the effect of consolidation on health risk, we constructed a simple example. We assumed 5 consolidation points in different regions where waste is collected and stored for re-shipment to a single disposal site ranging from 500 to 3000 miles distant. The average distance from water treatment plants to the consolidation points is assumed to be 100 miles. For simplicity, we assumed each 8 cubic foot package is transported separately to the consolidation point, that is, there is no storage at the water treatment plants. Estimated effects are tabulated in Table 15. The cancer risk associated with loading and unloading the materials (twice that calculated in Table 9) is included and dominates the cancer risk estimate for all distances.

Table 15. Nationwide risk of transport of radioactive materials to remote disposal site with regional consolidation. Includes risk of radiation exposure associated with handling and storage at consolidation site.						
Distance to remote site (miles)	Ra-226		Ra-228		Uranium	
	Accidents	Cancers	Accidents	Cancers	Accidents	Cancers
500	3.84E-04	2.00E-05	7.56E-05	2.00E-05	2.35E-04	2.00E-05
1000	3.98E-04	2.00E-05	8.96E-05	2.00E-05	2.49E-04	2.00E-05
1500	4.12E-04	2.00E-05	1.04E-04	2.00E-05	2.63E-04	2.00E-05
2000	4.26E-04	2.00E-05	1.18E-04	2.00E-05	2.77E-04	2.00E-05
2500	4.40E-04	2.00E-05	1.32E-04	2.00E-05	2.91E-04	2.00E-05
3000	4.54E-04	2.00E-05	1.46E-04	2.00E-05	3.05E-04	2.00E-05

3.3.1.3 Storage at Water Treatment Plants.

Storage of the waste at the water treatment plant could be combined with the consolidation option above to reduce the number of local trips. Additional handling in and out of storage would be required. If we assume the waste was not recognized as a radioactive waste, exposure during storage would be uncontrolled. Only the largest size plants, those serving over 50,000 people, could individually accumulate a full 15 cubic yard container of waste in less than a decade, so storage times would be large.

3.3.2 Radium-228 Transport of Ion-Exchange Resins to Landfill - Worker Exposure

Table 13 also calculates per trip risks of transport of radium-228, using the dose rate reduction of 61% estimated in the calculation of worker exposure from direct

radiation at the treatment plant, above, and the same exposure situation as for radium-226. As an example, for a treatment plant to landfill distance of 50 miles, we obtained a risk estimate of 9.2×10^{-10} lifetime cancers per trip and 1.4×10^{-6} from traffic accidents. The latter is independent of the truck's cargo.

From paragraph 3.2.6, above, 22 resin changes per year nationwide were estimated as necessary for the 40 systems required to treat for radium-228. Assuming each resin change generates a single trip to a disposal site, nationwide lifetime risk for 1-year's operation are tabulated in Table 14. For example, if the average distance between a treatment plant and a landfill were 50 miles, the estimated nationwide risk per trip of the workers from one year of operation would be 3×10^{-11} lifetime cancers and 3×10^{-5} deaths from traffic accidents per year of operation. Were the trips consolidated as discussed in paragraph 3.3.1.2, above, the nationwide risks are tabulated in Table 15 for different distances to a remote disposal site. For example, for a haul distance of 1,000 miles from the consolidation point to the disposal site, the overall risk nationwide of a consolidation option for one year of operation was estimated to be 9×10^{-5} for accidental deaths and 3×10^{-5} for lifetime cancers. Cancer risks are increased due to additional handling of the material at the storage point.

3.3.3 Uranium Treatment: Transport of Ion-Exchange Resins to Landfill - Worker Exposure

Table 13 also calculates per trip risks of transport of uranium, using the dose rate reduction estimated in the calculation of worker exposure from direct radiation at the treatment plant, above, and the same exposure situation as for radium-226. As an example, for a treatment plant to landfill distance of 50 miles, we obtained a risk estimate of 1×10^{-16} lifetime cancers per trip and 1.4×10^{-6} per trip from traffic accidents. The latter is independent of the truck's cargo.

In Table 10 it was calculated that 79 resin changes would be required nationwide for the 1457 systems required to treat for uranium in ground water. Assuming each resin change generates a single trip to a disposal site, nationwide lifetime risk for 1-year's operation are tabulated in Table 14. For example, if the average distance between a treatment plant and a landfill were 50 miles, the estimated nationwide risk per trip of the workers from one year of operation would be 1×10^{-10} lifetime cancers and 1×10^{-4} deaths from traffic accidents per year of operation. Were the trips consolidated as discussed in paragraph 3.3.1.2, above, the nationwide risks are tabulated in Table 15 for different distances to a remote disposal site. For example, for a haul distance of 1,000 miles from the consolidation point to the disposal site, the overall risk nationwide of a consolidation option for one year of operation was estimated to be 2.5×10^{-4} for accidental deaths and 2×10^{-5} for lifetime cancers. Cancer risks are increased due to additional handling of the material at the storage point.

3.4 Public Exposure and Risk in Waste Transport

The public would be exposed to routine traffic accidents, direct radiation during transport, and possibly radiation exposure to spilled materials in the event of an accident. Since 58% of traffic deaths involve collisions with other vehicles or pedestrians, it seemed reasonable to assume that fatalities among the public in traffic accidents is at least 58%, and probably over 100%, of truck occupant fatalities (USDOT, 1991, p. 612). We assumed that public fatalities from routine accidents are equal to occupant fatalities. Given this assumption public accidental fatalities for transport are the same as those given in Tables 14 and 15 for occupational fatalities.

Individuals in the public would be further from the cargo and exposed for a much shorter time than the occupants of the truck. For example, for a 50 mile trip, a person following the truck carrying ion-exchange material from Ra-226 removal at 35 feet from the cargo, assuming no shielding, lifetime cancer per trip would be about 5×10^{-12} and for all trips nationwide in a year, 1×10^{-8} . Corresponding values for Ra-228 and uranium would be, respectively, 1×10^{-9} and 5×10^{-13} lifetime cancers per year of operation.

Were the truck to be involved in an accident, radioactive material in the waste containers could be released. If the material were known to be radioactive, the area of the spill would be cordoned off quickly and cleaned-up with a minimum of public exposure. If the materials were not labeled as radioactive, higher exposures might occur among the public and clean-up workers. Effects would vary widely by circumstance. The Department of Transportation requires any material that has an activity of 2 nCi/g or higher to be labeled as radioactive material (49 CFR 173.403). Some of the water treatment plant wastes may fall under this limit, although in their analysis, Mangelson and Lauch (1990) could only say that the containers they hypothesize would contain less than 10 nCi/g. The Nuclear Regulatory Commission (NRC) set a concentration limit of 0.05% for material containing uranium, below which the material can be received, possessed, used, transferred or delivered without being classified as a source material. This converts to 334 pCi/g waste (10 CFR 20.1003, cited by Jarvis et al., 1992).

3.5 Landfill of Ion-Exchange Media

Although we assumed here, in Scenario I, that wastes from the water treatment plants are flushed to the sewer, it was unreasonable to assume that spent ion-exchange media will be flushed to the sewer. We assumed that the material will be dewatered and/or solidified in containers. We assumed this material will be buried. Depending on the level of radioactivity of the material, state or local regulations, and individual landfill rules, this material may be sent to a municipal landfill, a hazardous waste disposal facility, or a low-level radioactive waste disposal facility. Requirements for operating the different type facilities vary considerably.

- Hazardous waste sites require double liners with leachate collection systems. The latter must be operated post-closure until no more leachate is produced. There are 31 commercial hazardous waste facilities with capacity ranging from 3000 to 8.4 million cubic yards and area of 1 to 200 acres. Operating

personnel at hazardous waste facilities range from 2 to 200 with a median value of 19 (Cohen and Eastern, 1992). For storage of containerized wastes, a facility must have an impervious floor that allows collection of leakage, a system to prevent precipitation run-on into the containment system, a collection and removal system for any released waste (Wagner, 1991).

- Many states have required liners and leachate collection systems for municipal landfills for decades. Sixty to 70% of municipal landfills now have liners. Capacities range from 300,000 to 364 million cubic yards and area from 1 to 5,200 acres. In general, 1 employee is needed per 70 tons/day received (Cohen and Eastern, 1992). Under Subtitle D or RCRA, all municipal landfills are required to have liner systems and cover systems to minimize infiltration of water into the refuse mass. Groundwater monitoring and monitoring of surrounding soil is now required nationally (Ham, 1993).
- We assume that only the ENVIROCARE site in Utah will be available for disposal of radioactive material from water treatment plants if the waste must be treated as mixed (hazardous and radioactive) waste. This site is designed for the management of naturally occurring radioactive material (NORM), including uranium and radium-226.

Worker exposure from unloading and emplacing the waste containers at any of the landfill types would probably involve no more exposure than was estimated above for packaging and loading the material at the treatment plant.

During operation of most facilities and for some time post-closure, no public effects are expected. Years following closure, however, some landfills may be open to uncontrolled uses such as residential or even farming. Moreover, with time, both the cover system designed to prevent infiltration and the liner or alternative designed to prevent leachate reaching ground water may be expected to fail, allowing leachate to reach ground water and, potentially, drinking water wells. Time to failure will depend in part on current requirements that vary by type of landfill, and in part on local conditions that are impossible to predict. In the long run, all engineered protective measures can be expected to eventually fail, given a lapse of institutional control.

3.6 Sewerage and Sewage Treatment Plant Worker Exposure and Risk

Since 79% of the radium and uranium in the raw water was assumed to be complexed with the ion-exchange media and landfilled, the flow of radium and uranium to sewage plants will be decreased by 67% in systems that use ion exchange. Where sedimentation, coagulation, and filtration is used to remove uranium (assumed here for surface water facilities only), the amount of uranium reaching the sewage plants will increase by 60%. We consider only the incremental increase in exposure associated with uranium from surface water facilities.

Sewerage workers do routine maintenance on sewer lines and pumping stations and clean out blockages in sewage lines. They wear protective clothing to avoid chemical and biological hazards. This may offer some degree of protection from

radiation exposure also. Nonetheless, these workers do come into close contact with sewage, exposing themselves to direct radiation, skin contact, and possible ingestion to a small degree. Much the same is true of general sewage plant workers. In small plants, the largest exposure is likely to come from 1-2 hours per day cleaning screens on the sewage feed line. Again, workers wear protective clothing and, although they are in close proximity to the sewage, they have little actual contact with it. Any risk these workers face from radiation exposure would change due to the change in concentration of radioactive material in the sewage, but both the pre-rule exposure and the increment that would result from imposing the rule are likely to be negligible.

Since the removal rates in the sewage plant are unlikely to change, we assumed that the amount of radioactive material in the sewage sludge and in the wastewater plant discharge changes in the same proportion as the influent wastewater.

Sewage sludge is initially generated by means of a series of sedimentation processes and from flushing of filters. It is then pumped to digesters where much remaining organic material is gasified. Then it is either dried or spread on land. Drying may be mechanical or in evaporation ponds. In both drying methods, workers handle the sludge only with machinery. Some airborne dust may be generated in the process. We assess the potential effect of this using empirical models developed for estimating emissions of loading and unloading contaminated soil and described in EPA (1992). Generation and exposure to airborne dust generated by handling sewage sludge is not easily measured directly and several assumptions were necessary to the calculation. The equation for emissions from loading and unloading is

$$E = K_p (0.0009) (s/5) (u_w/2.2) (DH/1.5) (M/2)^{-2} (YD/4.6)^{-0.33}$$

where

E = kg particulate emitted/Mg soil loaded or unloaded

K_p = particle size multiplier (dimensionless (assume 0.75)

s = percent silt (assume 10%)

u_w = mean wind speed (m/s) (assume 2.2)

DH = drop height (m) (assume 1 m)

M = material moisture content (%) (assume 10%)

YD = dumping device capacity (m^3) (assume 0.25)

We assume dried sludge is picked up in a front-end loader and dumped into a truck for transport. It takes 30 seconds for a single dump, but 2 workers are exposed for 3 minutes for each dump. Assuming a specific density of the sludge of 2,

$$E = 1.9 \times 10^{-5} \text{ kg particles emitted/Mg sludge dumped.}$$

The uranium is only a fraction of the total particle mass. Assuming, as an example, the original concentration in the influent sewage was 60 pCi/L uranium and this was increased by 60% (an increase of 36 pCi/L), applying the ratio of 23 pCi/g sludge related to 15 pCi/L in influent sewage, an increment of 36 pCi/L yields an increment of 55 pCi/g sludge. The emission rate of uranium is thus,

$$E = 1.9 \times 10^{-5} \times 55 = 1 \times 10^{-3} \text{ pCi uranium/Mg sludge dumped.}$$

To estimate the resulting concentration in the air to which the workers are exposed, we apply a box model described in EPA (1992) as appropriate to estimate atmospheric concentrations for observations points within 100 meters of the source. The equation of this model is:

$$C = M_a / (u_w \times A_y \times Z_m)$$

where

C = Concentration in air

M_a = mass flux into the atmosphere (g/s)

u_w = wind speed (m/s) (assume 2.2)

A_y = width of waste management area (assume 100 m)

Z_m = Mixing height, fixed by the model to the height of the point of interest (the worker's nose, in this case, assumed to be at 2.5 m)

Applying the emission rate of 1×10^{-3} pCi uranium/Mg sludge dumped in 30 seconds as a mass flux rate of 1×10^{-3} pCi/30 seconds, or 3.33×10^{-5} pCi/s, the resulting concentration was,

$$C = 3.2 \times 10^{-4} \text{ pCi/m}^3$$

The duration of exposure is dependent on the amount of sewage. Based on the typical suspended solids content of sewage, we estimate 0.2 g sludge produced per L sewage. Assuming 60% of the drinking water supply enters the sewage system (Linsey and Franzini (1979), sewage quantity, sludge quantity, and exposure duration is estimated in Table 16. Assuming the "reference man" estimate of 20 L/min breathing rate, the two exposed workers inhale uranium at a rate of

$$3.2 \times 10^{-4} \text{ pCi/m}^3 \times 0.001 \text{ m}^3/\text{L} \times 20 \text{ L/min} = 6.4 \times 10^{-6} \text{ pCi/min.}$$

Effective dose equivalent for inhaled uranium (class Y) is

3.2×10^{-5} Sv/Bq or 1.2×10^{-4} rem/pCi (Till and Meyer, 1983). This is used in Table 16, along with the cancer risk factor of 5×10^{-4} lifetime cancers per man-rem, to calculate lifetime cancer risk associated with one year of operation. The nationwide total for all plants is 8×10^{-8} cancers per year of operation.

Table 16. Calculation of exposure duration of uranium dust from loading sewage sludge. from sewage plants receiving uranium waste from surface water treatment plants.

Population Served	Number of Systems	Avg. MGD	Tot. Sludge Mg/y	No. Loader Dumps	Exposure Minutes/y	Lifetime Cancer/y
25-500	51	0.06	609	2435.0	7305.1	5.61E-09
501-3,300	34	0.5	3382	13527.9	40583.7	3.12E-08
3,300-10,000	8	1.3	2069	8275.9	24827.7	1.91E-08
10,001-25,000	1	3.3	657	2626.0	7878.0	6.05E-09
25,000-50,000	1	7.2	1432	5729.5	17188.4	1.32E-08
>50000	0		0	0.0	0.0	0
Totals			8149	32594.3	97782.8	7.51E-08

Below, additional risks associated with the proposed rule are discussed for each of three methods of sewage sludge disposal. In each case, it is assumed that one method is applied to all the effected sludge.

The amount of sludge per unit intake of sewage and the concentration of radionuclides in the sludge per unit concentration in the intake sewage depend on the character of the sewage and the treatment processes used. Typical domestic sewage contains 655 mg/l solids, divided into 140 mg/l settleable solids, 95 mg/l non-settleable suspended solids, and 420 mg/L dissolved solids (Fair and Geyer, 1963). Sedimentation and chemical coagulation generally remove 100% of settleable solids and 70-90% of other suspended solids; secondary treatment removes 50-90% of the remaining suspended solids. EPA (1986) reports studies of radium-226 in sewage sludge averaging 23 pCi/g in areas with elevated levels of radium in water (not quantified). Were the radium divided proportionally among the classes of solids in the sewage, the above removal rates would suggest a sewage concentration of 15 pCi/l, consistent with the statement that the water concentrations were "elevated." This undoubtedly does not represent the real distribution of the radium in the sewage or the mechanism by which it is passed into the sludge during treatment. Lacking other information, however, we assumed that 35%-50% of the radium in the sewage enters the sludge. With removal of the radium at the water treatment plant and flushing of the ion exchange regeneration water to the sewer, the concentration in sewage, and consequently the concentration in sludge, will then be assumed to decrease by 67%. We assumed the same for uranium in the case of ground water supply, where ion exchange was assumed to be used. For surface water sources removing uranium, we assumed the uranium concentration in the sewage plant influent increases by 60%.

3.6.1 Incineration of Sewage Sludge

It was assumed that sewage sludge was burned in a municipal incinerator. Worker exposure results from the mechanical handling of the sludge and from maintenance of the incinerator and its pollution control equipment. Jarvis et al. (1992) estimated workers at a hazardous waste incinerator were exposed to dust at concentrations averaging 8×10^{-6} g/m³ over a year of operation. We assumed municipal incinerator workers are similarly exposed. Cohen and Eastern (1992) found that municipal incinerators average 550 tons/day throughput and estimated that a plant of this size would have about 6 operating workers. In Table 16, we calculated that the amount of sludge produced by sewage plants receiving uranium waste from water treatment plants with raw water sources was 8150 Mg/y and assumed that the excess uranium concentration in that sludge due to uranium removal at the water treatment plant was 1×10^{-3} pCi/Mg. If all of this sludge were incinerated, it would constitute $(8150/365 \times 1.1/550)$ 4% of the input waste to the incinerator. Average worker exposure to uranium from this source would be

$$(8 \times 10^{-6} \text{ g/m}^3 \times 1 \times 10^{-9} \text{ pCi/g} \times 0.04) = 3.2 \times 10^{-16} \text{ pCi/m}^3.$$

Six operating workers breathing this concentration at a rate of 20 L/min 8 hours/day for a year (2000 hours) combined with an effective dose equivalent of

1.2×10^{-4} rem/pCi inhaled (Till and Meyer, 1987) and the cancer risk factor of 5×10^{-4} yields a lifetime cancer risk of

$$\begin{aligned} & 6 \text{ workers} \times 20 \text{ L/min} \times 60 \text{ min/hr} \times 2000 \text{ hr} \times 0.001 \text{ m}^3/\text{L} \times 3.5 \times 10^{-16} \\ & \text{pCi/m}^3 \times 1.2 \times 10^{-4} \text{ rem/pCi} \times 5 \times 10^{-4} \text{ cancers/man-rem} \\ & = 3 \times 10^{-19} \text{ lifetime cancers/year of operation for the nationwide incineration of} \\ & \text{the uranium removed in water treatment.} \end{aligned}$$

Since the hot exhaust gases from the incinerator will quickly rise and move off the sewage plant site, this exposure would occur only under unusual conditions and is expected to be small compared to other exposures.

3.6.2 Landfill of Sewage Sludge.

3.6.2.1 Worker Benefit of Landfill of Sewage Sludge for Radium-226 Treatment

For radium-226, the decrease of radium in the sludge will reduce the exposure of workers to upward seepage of radon gas through the landfill. If, as an example, the pre-ruling sludge contains 23 pCi/g of radium-226, when treatment is introduced in the water treatment plant, the concentration will be reduced by 33%, or 7.6 pCi/g. Sewage sludge typically represents only 5% of landfill material, so the incremental radium-226 content of the landfill is 0.4 pCi/g. This assumes the sludge is uniformly spread; while this is probably not true, workers probably are uniformly distributed. Using RESRAD, we calculated a radon exposure level associated with 0.4 pCi/g of 0.5 mrem/y. Reducing this by a factor of 0.23 to account for a 8-hour work day, 5 days/week for 50 weeks/year yields a decrease in exposure of 0.1 mrem/y to a landfill worker. This reduces lifetime cancer risk by 0.5×10^{-7} for each year of operation. How many workers would be subject to this risk increment? The amount of sludge produced by sewage plants associated with water supply systems that will remove radium-226 by ion exchange, assuming all such water systems had sewage treatment, is calculated in Table 17 as 29,000 Mg/y. The calculation is the same as in Table 16 and the system data (number of systems and average MGD) are from Table 9. Assuming 1 employee per 70 tons/day total waste input at municipal landfills (Cohen and Eastern, 1992) and sewage sludge representing 5% of the total contribution to the landfill, this gives

$$(29,000 \times 1.1/365) \text{ tons/day sewage sludge}/0.05 = 1750 \text{ tons/day total waste to landfill}$$

$$(1750 \text{ tons/day}) \times 1 \text{ worker}/70 \text{ tons/day} = 25 \text{ workers.}$$

$$\begin{aligned} & 25 \text{ workers} \times 0.4 \times 10^{-7} \text{ lifetime cancer risk increment per year of operation} \\ & = 1 \times 10^{-6} \text{ lifetime cancer increment (cancers averted) per year of operation.} \end{aligned}$$

EPA rules (FR 56:50978-51039) require 6 inches of earthen cover to be applied daily. Worker exposure to radioactive material included in airborne dust from sewage sludge would be limited to periods of dumping and spreading only. Using the emissions estimation model from EPA (1992) described above, we calculated the decreased exposure to workers from dumping sewage sludge containing decreased radium-226. We assumed a 4.6 m³ capacity dump truck, a 1.5 m load drop distance, 1 minute dump time associated with a 15 minute duration exposure.

Table 17. Calculation of amount of sewage sludge produced by sewage plants serving water distribution systems projected to use ion exchange to remove radium-226.

Number of Systems	Average MGD	Total Sludge Mg/y
18	0.02	72
2	0.29	115
31	0.88	5427
18	2.43	8702
6	5.29	6314
2	7.9	3143
0	0	0
1	25.33	5039
78		28812

The calculated concentration was 1.9×10^{-5} g/m³ total particulate. This compares to an average of 1.05×10^{-5} g/m³ estimated by Jarvis et al. (1992). The incremental radium concentration of sludge was 7.6 pCi/g, so the exposure concentration to radium would be 1.4×10^{-4} pCi/m³. Total sludge production associated with radium-226 treatment was 29,000 Mg/y, the equivalent of 3150 truck loads. Effective dose for inhalation is 8.5×10^{-5} rem/pCi (Till and Meyer, 1987). Total lifetime cancer risk for one year operation of all radium removal is

$$\begin{aligned}
 & 3150 \times 2 \text{ workers} \times 15 \text{ min} \times 20 \text{ L/min} \times 0.001 \text{ m}^3/\text{L} \times 1.4 \times 10^{-4} \text{ pCi/m}^3 \\
 & \times 8.5 \times 10^{-5} \text{ rem/pCi} \times 5 \times 10^{-4} \text{ cancer/man-rem} \\
 & = 1.1 \times 10^{-8} \text{ decrease in lifetime cancers per year of operation.}
 \end{aligned}$$

From EPA (1992), an empirical model for estimating particulate emissions from spreading sewage sludge on the landfill is

$$E = 5.38 k_p s^{0.6}$$

where

E = emission rate in kg/ha

k_p = 0.33 for particles less than 30 μm

s = silt content (assume 10%)

We assume one 4.6 m³ dump truck load is spread over about 0.25 acres (0.1 ha). Since the landfill is covered at the end of each day, the average duration of emission would be 4 hours. Emission rate was calculated to average 0.049 g/s. Using this to drive the box model to calculate air concentration gives 1.1×10^{-3} g/m³. This is equivalent to 8.4×10^{-3} pCi/m³ for Ra-226. Again calculating the total lifetime cancer risk for one year operation of all radium removal,

$$\begin{aligned}
 & 3150 \times 2 \text{ workers} \times 240 \text{ min} \times 20 \text{ L/min} \times 0.001 \text{ m}^3/\text{L} \times 8.4 \times 10^{-3} \text{ pCi/m}^3 \\
 & \times 8.5 \times 10^{-5} \text{ rem/pCi} \times 5 \times 10^{-4} \text{ cancer/man-rem} \\
 & = 1.1 \times 10^{-5} \text{ decrease in lifetime cancers per year of operation.}
 \end{aligned}$$

3.6.2.2 Increased Worker Risk at Landfill associated with Uranium

The increased uranium in sewage sludge associated with removal of uranium from surface waters in water treatment plants adds additional risk to workers at the landfill associated with the dust generated from dumping and spreading the sludge. Calculations are parallel to those for radium, above. Number of dump truck loads was estimated to be

$$8149 \text{ Mg sludge/y} / 9.2 \text{ Mg/truck} = 886 \text{ loads/y.}$$

The added risk associated with dumping sludge with increased uranium concentration was estimated to be

$$886 \times 2 \times 15 \times 20 \times 0.001 \times 1.9 \times 10^{-13} \times 1.2 \times 10^{-4} \times 5 \times 10^{-4} = 6 \times 10^{-18} \text{ lifetime cancers per year of operation nationwide.}$$

The added risk associated with spreading this sludge was estimated to be

$$886 \times 1.1 \times 10^{-12} \times 2 \times 240 \times 20 \times 0.001 \times 12 \times 10^{-4} \times 5 \times 10^{-4} = 6 \times 10^{-16} \text{ lifetime cancers per year of operation nationwide.}$$

3.6.3 Land Application of Sewage Sludge.

Land application involves either piping the sludge to the location of application for spray or other type of application or piping the sludge to a tank truck for spray application. The low level of radiation in the sludge, its liquid form that prevents dust formation, and the lack of close worker contact suggests worker exposure would be small compared to the parallel risks at the landfill that were themselves small as discussed above. The principal risk would involve traffic accidents in a truck used for spreading sludge on land and that operation would be unchanged by the proposed rule.

3.7 Public Exposure and Risk from Sewage Sludge and Sewage Plant Discharge

All treatment to remove radionuclides at the water treatment plant affects the level of radioactivity in sewage. If all radionuclides were removed and sent directly to the sewer, the sewage plant would see an increase of about 70% because radioactive material in water not entering the sewers would be added to existing material entering the sewers. This percentage decreases to the extent that some radioactive material remains in the treated water leaving the water treatment plant and, more importantly, some radioactive material is complexed in ion-exchange media and landfilled, even in Scenario I. Sufficient radioactive material is landfilled with spent ion-exchange media to result in a decrease in radioactivity at the sewage plant. Only water systems drawing on surface water sources and removing uranium are assumed to use sedimentation, coagulation and filtration as a treatment process rather than ion exchange. Thus, only for these systems is an increase in radioactivity at the sewage plant and in sewage sludge expected.

3.7.1 Landfill of Sewage Sludge.

It was assumed that sludge will be co-disposed with other municipal waste. It was shown above that concentrations of airborne dust from dumping and spreading

sewage sludge on the landfill led to very low risk levels to workers on the landfill itself. Public exposure beyond the landfill boundary would be much lower. EPA rules (FR 56:50978-51039) require 6 inches of earthen cover to be applied daily and a minimum of 18 inches of cover upon closure that must be maintained for 30 years. This should effectively eliminate any airborne dust containing radioactive materials from reaching the public during operation and for 30 years post closure.

Leachate from the landfill is controlled by liners and leachate collection systems or by alternative designs to meet allowable concentration limits in ground water and approved on a site-specific basis. Sewage sludge itself actually impedes the transport of metals from the municipal landfill. Required monitoring, while not specific for uranium or radium, would detect the existence of leachate breaking through the designed barriers, providing an opportunity for remediation. Public exposures during operation would be expected to be nil. Future populations may experience exposures qualitatively similar to those described above for public effects of landfilling of ion-exchange resins, but to a lesser extent.

3.7.2 Incineration of Sewage Sludge.

EPA (1989) indicates that uranium is enriched in fly ash relative to bottom ash by a factor of 2, while radium is enriched by a factor of 1.5. The amount of enrichment is variable and uncertain, however. Another report (Pacyna, 1980), for example, concurs on uranium but suggests that radium is more likely to concentrate in the bottom ash. Fly ash is assumed to be discharged to the air while bottom ash is landfilled. We assumed that 67% of radium and 60% of uranium is emitted as flyash. This may overestimate actual emissions if some of the flyash is collected in baghouses or other particulate emissions control device. Risk of public exposure to airborne uranium emitted from sewage was calculated by estimating the increase in uranium entering sewage sludge annually for each size system, assuming 60% is emitted from the incinerator stack (i.e., no fly ash collection system), and, using a Gaussian plume model, estimating the average concentration to which people are exposed. The population at risk was considered to be the population size associated with the water distribution system. From this, annual intake was calculated and multiplied by committed dose equivalent factors (Till and Meyer, 1983). The calculation is shown in Table 18.

Table 18. Calculation of risk of sewage sludge incineration: uranium.

Population Served/system	0.025-0.5K	0.5-3.3K	3.3-10K	10-25K	25-50K
Number of Systems	51	34	8	1	1
Average Population/system	409	3127	8824	22214	48506
Average MGD/system	0.06	0.5	1.3	3.3	7.2
Sludge, g/d	5.44E+04	4.53E+05	1.18E+06	2.99E+06	6.53E+06
Delta uranium, pCi/s	5.06E+01	4.22E+02	1.10E+03	2.78E+03	6.08E+03
Concentration factor (pCi/L) per pCi/s	4.40E-06	1.80E-06	6.50E-07	3.00E-07	1.70E-07
Average concentration pCi/l	2.23E-04	7.59E-04	7.13E-04	8.35E-04	1.03E-03
Effective committed Dose, man-rem/year	2.69E-01	7.01E+00	1.86E+01	5.48E+01	1.48E+02
Increased lifetime cancers per year of operation	6.86E-03	1.19E-01	7.43E-02	2.74E-02	7.39E-02

3.7.3 Land Application of Sewage Sludge

Land used for sewage sludge application is generally selected to avoid concurrent use by the public, but access may not generally be as controlled as at a landfill and the public may gain unlimited access sooner than would be the case for a landfill. In some cases sewage is applied on land used for agriculture. Potential risks are direct exposure to the public, effects through the foodchain and contamination of ground water. It is difficult to quantify effects because of the diversity of access to the land and of uses of the land that may develop in the future. In the case of direct exposure, the effect would be similar to that for a landfill after 30 years post-closure, but would likely begin sooner. In land application, the sludge is covered and thus is more accessible, but is likely to be more dispersed. Because of decreased control, the soil on which the sludge was applied might be more likely to be taken by the public for use on gardens, etc. Foodchain effects, again, would be similar to those on a landfill but would begin earlier. The possibility of ground water contamination is probably greater than at a landfill because of the reduced requirements for liners and monitoring. In a study of nitrate contamination, it was predicted that land application of sludge would lead to increasing ground water levels above the standard for 30-40% of the years operated (Kaufman and Haith, 1986).

3.7.4 Sewage Plant Discharge

The effect of increased radiation in sewage plant discharge is highly dependent on the receiving streams flow quantity and downstream usage. Both recreational uses and ingestion of freshwater fish vary greatly by region and stream. The size and flow rates of receiving streams and the degree of exposure varies widely from stream to stream, but it is reasonable to assume that larger communities with bigger sewage outfalls will be located, generally, on larger streams. The following calculation is based on what seems to be reasonable generic assumptions. To use an illustrative value, we make the following assumptions:

- Original concentration of uranium in water (and sewage) is 80 pCi/L.

- Treatment leads to a 60% increase in uranium concentration in the sewage plant influent
- 65% of the increase appears in the effluent (35% removed in sludge).
- Flow of the receiving stream is 20 times that of the sewage plant effluent. That results in an increased concentration in the receiving stream of 0.85 pCi/L.
- Bioconcentration in freshwater fish ranges from 0.3 to 400 (NCRP, 1985). We use 50.
- People eat an average of 0.24 kg of freshwater fish annually.
- Committed effective dose equivalent per unit intake is 6.88×10^{-8} Sv/Bq (2.5×10^{-8} rem/pCi) (Till and Meyer, 1983).
- Total population exposed is the nationwide sum of the populations of the involved water systems, about 2×10^5 .

These assumptions lead to an exposure of 0.26 man-rem per year of operation, posing a nationwide risk of 1.3×10^{-5} lifetime cancers nationwide per year of operation.

4. EXPOSURE AND RISK IN SCENARIO II, DISCHARGE PROHIBITED

This Scenario differs from Scenario I in that no discharges to the sanitary sewer are allowed. All wastes are disposed in a landfill. Thus there are no increased risks associated with the operation of the sewage plant. Indeed, any risks associated with exposure to radionuclides at the sewage plant under pre-rule conditions would be reduced, since radioactive materials would be largely removed at the water treatment plant. Instead, risks of handling wastes at water treatment plants and landfill sites will increase. Only the changes from Scenario I are discussed.

4.1 Accidental Injury in Facility Construction and Operation

Construction at water treatment plants will be the same as in Scenario I with the small addition of holding ponds or other waste handling facilities and additional operations limited to the operation of those facilities.

4.2 Worker Exposure and Risk

Worker exposure and risk associated with Scenario I at water treatment plants and associated with transport and disposal of spent resins will still exist. Since no wastes will be flushed to the sewer, exposure and risk associated with sewage and sewage sludge will not exist in this scenario. Additional exposure in Scenario II is associated with handling waste that was flushed to the sanitary sewer in Scenario I. This involves flushing backwash water or sludges to an evaporation pond. We assume these wastes, once dried in the evaporation ponds, will be transported to a disposal site. This will require handling with mechanical equipment and loading it into trucks. These wastes would ordinarily be kept on site or shipped in bulk to a municipal landfill. It may be necessary, however, to dispose of them in a low-level radioactive waste site or a hazardous waste site, in which case it will be necessary to package them in

containers. These materials will be relatively dry and it should not be necessary to mix them with anything before packaging, although they are likely to maintain sufficient moisture content to avoid generation of airborne dust. Effects on landfill workers will be qualitatively similar to those estimated for sewage sludge in Scenario I.

4.2.1 Worker Exposure and Risk at Water Treatment Plant

Flushing backwash water and sedimentation wastes to evaporation ponds is largely automated and enclosed. No exposures are assumed in this process. Mangelson and Lauch (1990) indicate that backwash was about 10% of water intake to the plant and had a radium concentration of 40 pCi/L and total dissolved solids of 10,000 mg/L. There appears to be an inconsistency in the mass balance for total dissolved solids in the Mangelson and Lauch flow chart, indicating that the estimate of solids, and thus our calculation of total waste mass, may be too high by a factor of 3 or more. The mass balance of radium, however, is reasonable. If the backwash was evaporated to dryness, the residual waste would thus have a concentration of 4 pCi/g. The quantity of this waste would be 1 g per L of plant intake water. Calculation of total quantity for radium-226 is made in Table 19. Assuming similar relationships for uranium containing ground waters treated by ion exchange, with 80 pCi/L in the raw water and backwash water, a similar calculation is made in Table 20. These assumptions, of course, are exemplary only; actual values in any plant will depend on the character of the input water and the configuration and efficiency of the treatment processes.

Table 19. Calculation of evaporation pond residuals for radium-226 removal (dry weight basis).

	Number Systems	Average MGD	Dry waste Mg/y	pCi/y Ra-226
	18	0.02	5.09E+02	2.04E+09
	2	0.29	8.20E+02	3.28E+09
	31	0.88	3.86E+04	1.54E+11
	18	2.43	6.19E+04	2.47E+11
	6	5.29	4.49E+04	1.80E+11
	2	7.9	2.23E+04	8.94E+10
	0	0	0.00E+00	0.00E+00
	1	25.33	3.58E+04	1.43E+11
Totals	78		2.05E+05	8.19E+11

Table 20. Calculation of evaporation pond residuals for uranium removal from ground water (dry weight basis).

	Number Systems	Average MGD	Dry Waste Mg/y	pCi/y Uranium
	1224	0.02	6.92E+04	2.77E+11
	214	0.2	1.21E+05	4.84E+11
	18	0.9	4.58E+04	1.83E+11
	1	3.4	9.62E+03	3.85E+10
Totals	1457		2.46E+05	9.83E+11

Waste from sedimentation and filtration removal of uranium from surface water treatment will be more variable. Removal efficiencies are expected to range from 80-95% (FR 56:33088). Total solids in the raw water might range from about 300 mg/L to above 2,000 mg/L. Assuming a uranium concentration of 80 pCi/L, uranium concentration on a dry-waste basis could range from 4% to 26% and quantities from 0.2 to 2 g solids per L of input water. As an example, we calculate in Table 21 quantities of waste assuming 95% removal efficiency, 2,000 mg/L total solids and 80 pCi/L uranium in the raw water.

Table 21. Calculation of evaporation pond residuals for uranium surface water treatment by sedimentation, coagulation, and filtration (dry weight basis).

	Number Systems	Average MGD	Waste Mg	Uranium pCi
	51	0.06	8.22E+03	3.29E+10
	34	0.5	4.57E+04	1.83E+11
	8	1.3	2.79E+04	1.12E+11
	1	3.3	8.87E+03	3.55E+10
	1	7.2	1.93E+04	7.74E+10
Totals			1.10E+05	4.40E+11

The evaporation ponds themselves will pose some degree of risk from direct exposure and accidental immersion. Exposure levels will vary with the degree of concentration of the waste from evaporation. We estimate that exposure levels 4 feet from the dried

waste will be on the order of 5×10^{-6} rem/hour for radium-226, 3×10^{-6} rem/hour for radium-228, and 4×10^{-13} rem/hour for uranium. If it takes 0.5 hour to load 1 ton of waste, total worker risks associated with loading processes are:

For radium-226: $(2 \times 10^5 \text{ tons/y}) / (2 \text{ tons/hour}) \times 5 \times 10^{-6} \text{ rem/h} \times 2 \text{ men} \times 5 \times 10^{-4} \text{ cancer/rem} = 5 \times 10^{-4} \text{ lifetime cancer risk per year of nationwide operation.}$

For radium-228: Assuming the same conditions as for backwash of radium-226, the 40 water systems required to remove radium-228 averaging 0.61 MGD will produce

$$40 \times 0.61 \text{ MGD} \times 365 \text{ days/y} \times 3.785 \text{ L/gal} \times 1 \text{ g/L} \times 4 \text{ pCi/g} = 1.35 \times 10^{11} \text{ pCi/y}$$

This is 41% of the Curies produced by those systems removing radium-228, and the gamma exposure per Curie is 61%, so exposure is estimated to be

$$5 \times 10^{-4} \text{ lifetime cancer risk/y for radium-226} \times 0.41 \times 0.61 \\ = 1.3 \times 10^{-4} \text{ lifetime cancer risk/y for radium-228}$$

For uranium ion exchange waste: $(2 \times 10^5 \text{ tons/y}) / (2 \text{ tons/hour}) \times 4 \times 10^{-13} \text{ rem/h} \times 2 \text{ men} \times 5 \times 10^{-4} \text{ cancer/rem} = 5 \times 10^{-11} \text{ lifetime cancer risk per year of nationwide operation.}$

For uranium sedimentation and filtration waste: $(1 \times 10^5 \text{ tons/y}) / (2 \text{ tons/hour}) \times 4 \times 10^{-13} \text{ rem/h} \times 2 \text{ men} \times 5 \times 10^{-4} \text{ cancer/rem} \\ = 3 \times 10^{-11} \text{ lifetime cancer risk per year of nationwide operation.}$

4.2.2 Worker Exposure and Risk in Waste Transport

Accident fatality per trip for transporting waste associated with backwash water and sedimentation, coagulation and filtration is the same as tabulated in Table 13. Cancer risks per trip will differ in the ratio of the exposure rates (given above). The number of estimated trips with a dump truck (9.2 Mg) or a 15 cubic yard container (30 Mg) are given in Table 22. These are used to calculate accidental fatality and cancer risk estimates associated with nationwide operations in Table 23 for radium and Table 24 and 25 for uranium.

Table 22. Number of trips by load size for transport of backwash and sedimentation wastes.			
Radionuclide	Radium-226	Uranium from IX backwash	Uranium from Sedimentation
Annual Waste (Mg)	200,000	250,000	110,000
No. trips @ 9.2 Mg/trip	22,000	27,000	12,000
No. trips @ 30 Mg/trip	6,700	8,300	3,700

Table 23. Risk of transporting radium backwash wastes to disposal site for 1 year of nationwide operation.

Distance (miles)	9.6 Mg Load		30 Mg Load	
	Accidents	Cancer	Accidents	Cancer
10	3.08E-03	2.08E-05	9.38E-04	6.32E-06
20	6.16E-03	4.15E-05	1.88E-03	1.26E-05
30	9.24E-03	6.23E-05	2.81E-03	1.90E-05
40	1.23E-02	8.30E-05	3.75E-03	2.53E-05
50	1.54E-02	1.04E-04	4.69E-03	3.16E-05
60	1.85E-02	1.25E-04	5.63E-03	3.79E-05
70	2.16E-02	1.45E-04	6.57E-03	4.42E-05
80	2.46E-02	1.66E-04	7.50E-03	5.06E-05
90	2.77E-02	1.87E-04	8.44E-03	5.69E-05
100	3.08E-02	1.83E-04	9.38E-03	5.58E-05
200	6.16E-02	3.67E-04	1.88E-02	1.12E-04
300	9.24E-02	5.50E-04	2.81E-02	1.68E-04
400	1.23E-01	7.33E-04	3.75E-02	2.23E-04
500	1.54E-01	9.17E-04	4.69E-02	2.79E-04
600	1.85E-01	1.10E-03	5.63E-02	3.35E-04
700	2.16E-01	1.28E-03	6.57E-02	3.91E-04
800	2.46E-01	1.47E-03	7.50E-02	4.47E-04
900	2.77E-01	1.65E-03	8.44E-02	5.03E-04
1000	3.08E-01	1.83E-03	9.38E-02	5.58E-04
1500	4.62E-01	2.75E-03	1.41E-01	8.38E-04
2000	6.16E-01	3.67E-03	1.88E-01	1.12E-03
2500	7.70E-01	4.58E-03	2.35E-01	1.40E-03
3000	9.24E-01	5.50E-03	2.81E-01	1.68E-03

Table 24. Risk of transporting uranium backwash wastes to disposal site for 1 year of nationwide operation.

Distance	9.6 Mg Load		30 Mg Load	
	Accidents	Cancer	Accidents	Cancer
10	3.78E-03	2.04E-12	1.16E-03	6.26E-13
20	7.56E-03	4.08E-12	2.32E-03	1.25E-12
30	1.13E-02	6.11E-12	3.49E-03	1.88E-12
40	1.51E-02	8.15E-12	4.65E-03	2.51E-12
50	1.89E-02	1.02E-11	5.81E-03	3.13E-12
60	2.27E-02	1.22E-11	6.97E-03	3.76E-12
70	2.65E-02	1.43E-11	8.13E-03	4.38E-12
80	3.02E-02	1.63E-11	9.30E-03	5.01E-12
90	3.40E-02	1.83E-11	1.05E-02	5.64E-12
100	3.78E-02	1.80E-11	1.16E-02	5.53E-12
200	7.56E-02	3.60E-11	2.32E-02	1.11E-11
300	1.13E-01	5.40E-11	3.49E-02	1.66E-11
400	1.51E-01	7.20E-11	4.65E-02	2.21E-11
500	1.89E-01	9.00E-11	5.81E-02	2.77E-11
600	2.27E-01	1.08E-10	6.97E-02	3.32E-11
700	2.65E-01	1.26E-10	8.13E-02	3.87E-11
800	3.02E-01	1.44E-10	9.30E-02	4.43E-11
900	3.40E-01	1.62E-10	1.05E-01	4.98E-11
1000	3.78E-01	1.80E-10	1.16E-01	5.53E-11
1500	5.67E-01	2.70E-10	1.74E-01	8.30E-11
2000	7.56E-01	3.60E-10	2.32E-01	1.11E-10
2500	9.45E-01	4.50E-10	2.91E-01	1.38E-10
3000	1.13E+00	5.40E-10	3.49E-01	1.66E-10

Table 25. Risk of transporting uranium sedimentation coagulation, and filtration wastes from surface waters to disposal site for 1 year of operation.

Distance	9.6 Mg Load		30 Mg Load	
	Accidents	Cancer	Accidents	Cancer
10	1.68E-03	9.06E-13	5.18E-04	2.79E-13
20	3.36E-03	1.81E-12	1.04E-03	5.58E-13
30	5.04E-03	2.72E-12	1.55E-03	8.38E-13
40	6.72E-03	3.62E-12	2.07E-03	1.12E-12
50	8.40E-03	4.53E-12	2.59E-03	1.40E-12
60	1.01E-02	5.43E-12	3.11E-03	1.68E-12
70	1.18E-02	6.34E-12	3.63E-03	1.95E-12
80	1.34E-02	7.25E-12	4.14E-03	2.23E-12
90	1.51E-02	8.15E-12	4.66E-03	2.51E-12
100	1.68E-02	8.00E-12	5.18E-03	2.47E-12
200	3.36E-02	1.60E-11	1.04E-02	4.93E-12
300	5.04E-02	2.40E-11	1.55E-02	7.40E-12
400	6.72E-02	3.20E-11	2.07E-02	9.87E-12
500	8.40E-02	4.00E-11	2.59E-02	1.23E-11
600	1.01E-01	4.80E-11	3.11E-02	1.48E-11
700	1.18E-01	5.60E-11	3.63E-02	1.73E-11
800	1.34E-01	6.40E-11	4.14E-02	1.97E-11
900	1.51E-01	7.20E-11	4.66E-02	2.22E-11
1000	1.68E-01	8.00E-11	5.18E-02	2.47E-11
1500	2.52E-01	1.20E-10	7.77E-02	3.70E-11
2000	3.36E-01	1.60E-10	1.04E-01	4.93E-11
2500	4.20E-01	2.00E-10	1.30E-01	6.17E-11
3000	5.04E-01	2.40E-10	1.55E-01	7.40E-11

4.2.3 Worker Exposure and Risk at Landfill.

It was assumed that these wastes will have a high moisture content and will not generate significant airborne dust in dumping or spreading at the landfill.

4.3 Public Exposure and Risk in Scenario II

No additional public exposures at water treatment plants were quantified in Scenario II compared to Scenario I. There is an increased risk to potential intruders from the evaporation ponds. Scenario II poses no increased risks associated with sewage plant effluent and handling and disposal of sewage sludge because no wastes are flushed to the sewer. No significant difference in public risk from land disposal of

water treatment plant wastes was quantified, because the wastes are assumed to be wet enough to avoid airborne emissions. An increased amount of waste will be disposed at land sites, leading to a potentially increased risk of groundwater contamination in the future, but that was not quantified.

5. SCENARIO III: GRANULAR ACTIVATED CARBON REMOVAL OF RADON

Only the risk associated with granulated activated carbon (GAC) removal of radon and disposal of the depleted GAC media are included in this scenario. This risk replaces the risk of packed tower aeration in Scenarios I and II. GAC is offered by EPA as an alternative, but was not rated as a "Best Available Technology" because of questions regarding disposal of the resulting solid waste.

5.1 Facility Construction

Because construction risks are derived from facility capacity factors, risks associated with facility construction are unchanged.

5.2 Worker Exposure and Risk at Water Treatment Plant

5.2.1 Treatment Processes

Granulated activated carbon absorbs radon. The key to effective performance of a GAC in removing radon, however, in contrast to a more typical pattern of declining efficiency and breakthrough characteristic of other materials absorbed on activated carbon, is that the radon quickly decays, freeing the carbon to absorb more radon. Thus, a steady state develops in which the device is able to maintain a constant removal rate over a considerable time. This has the advantage that the carbon need be changed only at relatively long intervals, perhaps decades (Lowry and Brandow, 1986). The ultimate need to replace the carbon will probably be associated with other contaminants in the water and the carbon medium will be contaminated with these, potentially including uranium and radium, as well as ^{210}Pb and other progeny of the radon itself. Backwashing also appears to be counter productive (Lowry and Brandow, 1986). No backwashing is assumed. The disadvantage is that substantial levels of radioactivity can build up in the GAC bed. Radiation levels have been reported as "usually less than 1.0 mR/hr at a distance of three (3) feet from the GAC tank surface" (Kinner et al., 1989). Measurements of smaller systems showed radiation levels up to 4 mrem/hr at 1 foot from the tank and about 1 mrem/hr at 3 feet from the tank (Lowry and Brandow, 1986). Thus, the external gamma radiation levels around the GAC tank are expected to be about 5 times higher than those around the ion-exchange vessel for removal of radium 226 (above). Because it does not require as frequent attention, however, workers may spend less time in its vicinity. If a worker spends 1 hour/week 1 foot from the GAC tank (compared to the previously assumed 3.5 hours/week near the ion exchange vessel), an annual dose of 28 mrem/y would result. The associated lifetime cancer risk for a single worker would be 1.4×10^{-5} . Summing this risk over the 27,657 systems that are expected to require removal of radon would result in 0.4 lifetime cancers per year of operation.

5.2.2 Dewatering and packaging of GAC media.

At some point, the GAC media becomes degraded by other contaminants in the water and must be disposed in an appropriate landfill. We assumed the media must be replaced after 400 million gallons have been treated. This is more frequent than Lowry and Brandow's (1986) estimate, on the basis that early estimates of life time of the media may prove optimistic when the devices are put in widespread use under varying field conditions. Using this assumption, requirements for replacement of GAC media are estimated in Table 26. Worker exposure in making the replacement will depend on the technology used and the operating procedure. We assume that radiation exposure levels during the process are never higher than in process operation, i.e., 4 mrem/hour at 1 foot distant. We assume that during the replacement process and the subsequent handling of the spent media at the water treatment plant, 2 workers are exposed at this level for 1 hour. The individual risk associated with one operation is thus

$$0.004 \text{ rem/h} \times 1 \text{ hr} \times 5 \times 10^{-4} \text{ cancer/man-rem} = 2 \times 10^{-6} \text{ lifetime cancers per exchange.}$$

Summing over 2 workers per plant for the 2780 plants estimated in Table 26 yields an estimated 0.01 cancer per year of operation.

Table 26. Requirements for replacing GAC media in water treatment plants required to remove radon gas.

	Number Systems	Average MGD	GAC Changes/y
	23789	0.02	434
	3284	0.2	599
	584	0.86	458
	204	2.38	443
	85	4.25	330
	20	9.43	172
	5	11.2	51
	12	26.67	292
Total	27983		2780

5.2.3 Worker Exposure and Risk of Waste Transport

Radiation exposure at 3 feet from the GAC container is 1 mrem/hr (Kinner et al., 1989). Making the same assumptions as for transporting the ion-exchange medium for radium-226 removal above, estimates of cancer risk and accident deaths from traffic accidents were calculated in Table 27 for various distances from water treatment plant to landfill ranging from 10 to 3000 miles. This assumed that each container of spent GAC media was transported separately. Man-rem per mile was calculated as

$$\text{Man-rem/mile} = 2 \text{ men} \times 1 \times 10^{-3} \text{ rem/hour} \times \text{mph}^{-1}$$

where mph = 53 mph for distances less than 100 miles and 60 mph for longer distances.

As an example, with a distance of 1000 miles, risk per trip was estimated to be 1.7×10^{-5} lifetime cancers and 2.8×10^{-5} accidental death. Summing over the estimated 2780 annual trips nationwide yields a nationwide risk of 0.05 lifetime cancers and 0.08 accidental deaths for each year of operation.

Table 27. Worker exposure and risk of transporting spent GAC to landfill by distance.

Nation-wide trips/y=		2780		
man-rem/mile, <100 mi = 3.8E-5, 100 mi and over = 3.3E-5				
Distance (miles)	Each trip		Nationwide	
	Cancers	Accidental Deaths	Cancers	Accidental Deaths
10	1.90E-07	2.80E-07	5.28E-04	7.78E-04
20	3.80E-07	5.60E-07	1.06E-03	1.56E-03
30	5.70E-07	8.40E-07	1.58E-03	2.34E-03
40	7.60E-07	1.12E-06	2.11E-03	3.11E-03
50	9.50E-07	1.40E-06	2.64E-03	3.89E-03
60	1.14E-06	1.68E-06	3.17E-03	4.67E-03
70	1.33E-06	1.96E-06	3.70E-03	5.45E-03
80	1.52E-06	2.24E-06	4.23E-03	6.23E-03
90	1.71E-06	2.52E-06	4.75E-03	7.01E-03
100	1.65E-06	2.80E-06	4.59E-03	7.78E-03
200	3.30E-06	5.60E-06	9.17E-03	1.56E-02
300	4.95E-06	8.40E-06	1.38E-02	2.34E-02
400	6.60E-06	1.12E-05	1.83E-02	3.11E-02
500	8.25E-06	1.40E-05	2.29E-02	3.89E-02
600	9.90E-06	1.68E-05	2.75E-02	4.67E-02
700	1.16E-05	1.96E-05	3.21E-02	5.45E-02
800	1.32E-05	2.24E-05	3.67E-02	6.23E-02
900	1.49E-05	2.52E-05	4.13E-02	7.01E-02
1000	1.65E-05	2.80E-05	4.59E-02	7.78E-02
1500	2.48E-05	4.20E-05	6.88E-02	1.17E-01
2000	3.30E-05	5.60E-05	9.17E-02	1.56E-01
2500	4.13E-05	7.00E-05	1.15E-01	1.95E-01
3000	4.95E-05	8.40E-05	1.38E-01	2.34E-01

As was done for radium-226 above, we assume a simple example of consolidation of trips. We assume 5 consolidation points in different regions where waste is collected and stored for re-shipment to a single disposal site at a distance ranging from 500 to 3000 miles. The average distance from water treatment plants to the consolidation point was assumed to be 100 miles. We assumed the waste from each GAC media exchange was transported to the consolidation point separately. Radiation exposure associated with unloading, handling, storage, storage and loading at the consolidation point was assumed to be the same as the exposure associated with exchange and handling at the water treatment plant, i.e., 2×10^{-6} lifetime cancer risk per unit. These effects are tabulated in Table 28. The overall risk is reduced from that in Table 27, but the effects are dominated by the 2,780 100-mile trips to the consolidation points.

Table 28. Nationwide risk of transport of spent GAC media to remote disposal site with regional consolidation. Includes risk associated with radiation exposure associated with handling and storage at consolidation point.

Distance to remote site (miles)	Accidents	Cancers
500	7.80E-03	5.26E-03
1000	7.81E-03	5.26E-03
1500	7.83E-03	5.27E-03
2000	7.84E-03	5.28E-03
2500	7.85E-03	5.29E-03
3000	7.87E-03	5.30E-03

5.2.4 Worker Exposure and Risk at Land Disposal Site

Worker Risk at the disposal site would be limited to direct radiation associated with unloading, handling, and emplacing the containers. If one assumes 1 man-hour per unit at an exposure level of 1 mrem/hr, the exposure per unit would be 0.001 man-rem and for 2870 trips would be 2.9 man-rem or 0.0014 lifetime cancers nationwide for each year of operation.

5.3 Public Exposure and Risk

Public exposure at the water treatment plant would be considerably reduced from that in Scenario I because of the elimination of the emission of radon removed from water to the air. There would be no excess public exposure in comparison to the pre-radon removal situation. Some additional risk to potential intruders would exist, especially if spent GAC media were stored on site (not assumed in this analysis).

Additional public risk would be imposed because of the increased truck traffic hauling spent GAC media to a disposal site. There would be further public risk in the event of a traffic accident in which GAC material was released. It is clear from the accidental fatality estimates for transport that accidents are to be expected and incidents resulting in the release of the radioactive material are likely. A quantitative estimate of risk to the public from such accidents would vary greatly by circumstance. The spent GAC material almost certainly will exceed the DOT minimum level for labeling vehicles.

Public Risk at the disposal site during operation is assumed to be nil. Because the wastes contain uranium and lead-210, the disposal site would pose a potential risk of ground water contamination post-closure long into the future.

6. COMPARISON OF RISKS AMONG THREE SCENARIOS

Worker risks associated with operating the treatment processes for removal of radium and uranium from drinking water for 1 year are summarized in Table 29-31 for the three scenarios. Risks are presented in terms of lifetime risk for one year of operation, assuming all water treatment plants of a given type use the same process.

Table 29. Radium removal: incremental risks to workers associated with one year of operation (Number of deaths).

	Scenario I		Scenario II	
	Radium-226	Radium-228	Radium-226	Radium-228
Facility Construction	6.70E-03	3.30E-03	6.70E-03	3.30E-03
Facility Operation	6.70E-03	3.30E-03	6.70E-03	3.30E-03
Ion-exchange operation	1.09E-02	3.13E-03	1.09E-02	3.13E-03
Replace I-X media	1.00E-04	2.00E-05	1.00E-04	1.00E-05
Handling backwash waste	0.00E+00	0.00E+00	5.00E-04	1.30E-04
Waste transport(1000mi)	2.40E-05	1.10E-04	7.20E-06	1.30E-04
Sewage sludge				
Incineration	0	0	0	0
Landfill	0	0	0	0
Land application	0	0	0	0
Land disposal of wastes	1.00E-04	2.00E-05	1.00E-04	2.00E-05
TOTAL WORKER RISK	2.45E-02	9.88E-03	2.50E-02	1.00E-02

Table 30. Uranium removal: incremental risks to workers associated with 1 year's operation (number of deaths).		
	Scenario I	Scenario II
Facility Construction	0.04	0.04
Facility Operation	0.09	0.09
Ion-exchange operation	1.5E-08	1.5E-08
Replace I-X media	1.0E-10	1.0E-10
Handle backwash	0.0E+00	5.0E-11
Coag & Filtration opn	0	3.00E-11
Waste transport (1000mi)	2.7E-04	4.0E-01
Sewage sludge disposal		
Load at treatment plant	8.00E-08	0
Incineration	3.00E-19	0
Landfill	6.0E-16	0
Land application	nil	0
Land disposal of waste	1.0E-10	1.0E-10
TOTAL WORKER RISK	0.13	0.53

Table 31. Radon removal: incremental risks to workers associated with on year's operation (number of deaths).		
	Scenarios I & II	Scenario III
Accidents, facility Construction	0.04	0.04
Accidents, facility Operation	1.6	1.6
Packed Tower Aeration	0.1	NA
Operation of GAC process	NA	0.4
Handling spent GAC media	NA	0.01
Transport of spent GAC media	NA	0.1
Land disposal of GAC	NA	0.001
TOTAL WORKER RISK	1.74	2.151

Risks to the public are summarized for radium and uranium in Tables 32-33. Like risks to workers, public risks are expressed as lifetime effects per year of operation. We have quantified only risks to people exposed during that year of operation. Several public risks exists that have not been quantified, e.g., the risk to intruders entering the evaporation pond holding backwash. Land disposal of wastes carries with it a continuing long-term future risk of leakage and associated contamination of soils and groundwater and risks associated with some future land use

in which the hazard posed by the waste goes unrecognized. These have not be quantified here.

Risk to the public for radon treatment was estimated to be 24 lifetime cancers per year of operation for packed tower aeration (scenarios I and II). For scenario III, risk to the associated with exposure during operation would be primarily associated with traffic accidents during transport of waste GAC media. These would vary greatly depending of circumstances of the release and clean-up of radioactive material.

Table 32. Radium: incremental risks to the public associated with one year's operation (Numbers of deaths).				
	Scenario I		Scenario II	
	Radium-226	Radium-228	Radium-226	Radium-228
Treatment plt opn	0	0	0	0
Waste transport (1000 mi)	4.00E-04	9.00E-05	4.00E-04	9.00E-05
Sewage sludge	0	0		
Incineration				
Landfill				
Land application				
Sewage discharge	0	0		
Land disposal of waste	0	0		
Total	4.00E-04	9.00E-05	4.00E-04	9.00E-05

Table 33. Uranium: incremental risks to the public associated with 1 year's operation (number of deaths).		
	Scenario I	Scenario II
Treatment plt opn	0	0
Waste transport (1000 m	2.0E-04	2.0E-04
Sewage sludge		
Incineration	0.073	0
Landfill	0	0
Land application		
Sewage discharge	1.3E-05	0
Land disposal of waste		
Total	7.3E-02	2.0E-04

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