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Risk Analyses of Buried Wastes from Electricity Generation*

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1 INTRODUCTION

The straightforward way to do a probabilistic risk analysis (PRA) is to identify all possible sequences of events that can lead to deaths among the public (or to some other targeted endpoint), estimate the probability for each event, multiply these probabilities for all events in a sequence to determine the total probability for that sequence, and finally add up the probabilities for all sequences. Interdependencies of events (common mode failures) must be taken into account, which can add greatly to the complication and uncertainty (1). Such a PRA involves development of a large program run on a digital computer.

This type of PRA has been carried out for systems like aircraft and nuclear power plants where there is complete knowledge of construction details and failure-rate experience with every component. But even in these cases, many approximations must be made, and uncertainties are quite large. If one were to apply this technique to wastes buried in the ground, the difficulties would be enormously greater because knowledge of the system is much less complete and is changing with time. Geochemistry is a much more complex subject than mechanical or electrical design of machines, and it is sensitive to a number of factors on which there is limited information.

An alternative approach is to replace the digital computer with an analog computer. Constructing such an analog computer would be a tremendous project, and it would be enormously expensive. However, that analog computer is now available, and we are all free to use it. It is our earth itself.

This chapter describes how such an approach can be used. For example, to study how waste converted into rock behaves, how ordinary rock behaves will be reviewed. Several cases of this type directed at the analysis of wastes generated by nuclear and by coal-burning power plants are presented.

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2 NUCLEAR HIGH-LEVEL WASTE (HLW)

High-level waste (HLW) is the residue of spent nuclear fuel after it has been removed from reactors. One plan for its disposal is to convert it into a rocklike material and bury it in the natural habitat of rocks, about 600 m underground.

It is generally agreed that the principal concern is that buried HLW will be contacted by groundwater, dissolved, transported to the surface, and thereby get into our food and water supplies. We are therefore concerned with the health effects, principally the cancer risk, of this material entering human stomachs. The potential risks associated with this hazard are presented in Fig. 1; the following discussion reviews its derivation.

2.1 Risk if Material Enters Human Stomachs

Consider the risk of one particular type of cancer, liver cancer, from eating 1 millicurie (mCi)—3.7×10^7 radioactive decays per second—of one particular radioactive isotope, plutonium-239 (^{239}Pu). Ingested plutonium has a 10^{-4} probability for transmission through the walls of the gastrointestinal tract into the bloodstream; once in the blood it has a 45% chance of becoming deposited in the liver (1a). Thus, 0.45×10^{-4} or 45×10^{-6} mCi gets into the liver where it remains for an average of 40 years (1a). The number of alpha (α) particles emitted into the liver is then $(45 \times 10^{-6}\,\text{mCi}) \times (3.7 \times 10^7\,\alpha/\text{s}\cdot\text{mCi}) \times (3 \times 10^7\,\text{s/yr}) \times (40\,\text{yr}) = 2 \times 10^{12}$. Since the alpha particle energy is 4.8 MeV, the radiation energy absorbed by the liver is $(2 \times 10^{12}) \times (4.8) \times (1.6 \times 10^{-13}\,\text{J/MeV}) = 1.5\,\text{J}$.

Radiation dose in rads is defined as 0.01 J of radiation energy deposited per kg of body

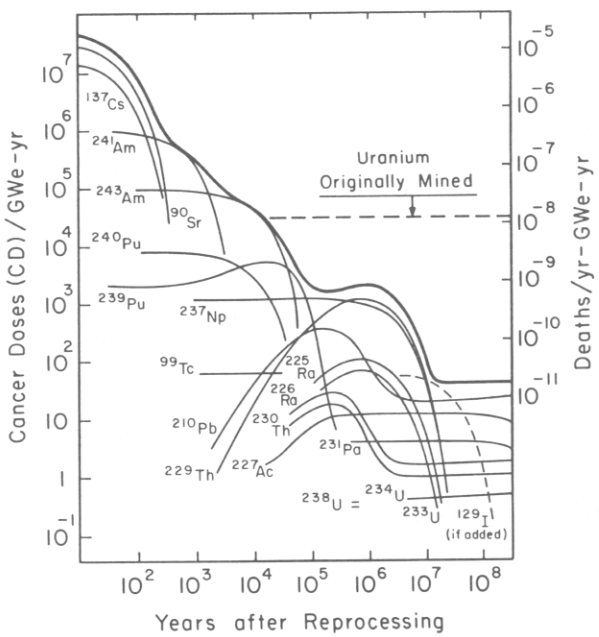

Figure 1. Cancer doses and predicted deaths per year vs. years after reprocessing of high-level radioactive waste.

organ. Since the average weight of a human liver (2) is 1.8 kg, the dose to the liver from eating 1 mCi of ^{239}Pu is (1.5 J/1.8 kg) × (1 rad/0.01 J/kg) = 80 rads.

Estimating the cancer risk from a given radiation dose is principally the realm of the National Academy of Science's Committee on Biological Effects of Ionizing Radiation (BEIR) and the United Nations Scientific Committee on Effects of Atomic Radiation (UNSCEAR).

The latest BEIR (3) and UNSCEAR (4) reports estimate the cancer risk due to alpha particle bombardment of liver principally from studies of German, Japanese, Danish, and Portugese patients injected with colloidal thorium dioxide as an x-ray contrast medium: there were 301 liver cancers versus only six normally expected among them. These studies derive a risk of 300×10^{-6} per rad. Applying this to our calculation, we find the liver cancer risk to be $(80 \times 300 \times 10^{-6}) = 0.024$ per mCi of ^{239}Pu ingested.

However, ingested plutonium may also get into the bone and cause bone cancer or into the bone marrow and cause leukemia; calculations similar to the above give these risks to be 0.011 and 0.0006/mCi, respectively (5). In addition, the 99.99% of the ^{239}Pu that does not get through the gastrointestinal tract spends about 24 h radiating its inner walls before it is excreted, thereby causing (5) an intestinal cancer risk of 0.002. Summing the results for all types of cancer gives the risk as (0.024 + 0.011 + 0.0006 + 0.002) = 0.038/mCi of ^{239}Pu ingested.

If the ingestion of 1 mCi is shared among N people, each will have an average risk of $0.038/N$, making the risk of a cancer within the group $(N \times 0.038/N) = 0.038$. Thus, the number of cancers caused by ingestion of 1 mCi of ^{239}Pu is independent of N, the number of people among whom it is shared. This is a consequence of the linear, no-threshold dose–response relation used for estimating radiation risk.

The radioactive waste produced by a 1-GW (= 1,000,000 kW) output electric power plant in 1 year (1 GWe-yr), after reprocessing, contains 6×10^4 mCi of ^{239}Pu. Thus, if all

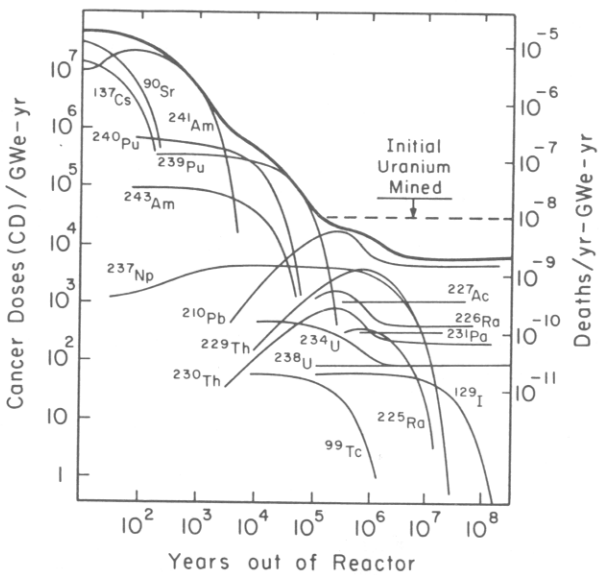

Figure 2. Cancer doses and predicted deaths per year vs. years after the high-level wastes leave the reactor.

the ^{239}Pu in 1 GWe-yr of this HLW were fed to people, we could expect $(6 \times 10^4) \times (0.038) = 2300$ fatal cancers. We refer to this as the number of cancer doses (CD)/GWe-yr.

This result is shown on the curve labeled ^{239}Pu in Fig. 1, which is a plot of CD versus time after removal from the reactor (5). We see that it increases in the first 20,000 years as ^{243}Am decays into ^{239}Pu, but after that time it decreases due to the 24,000-year half-life of ^{239}Pu.

Of course, ^{239}Pu is not the only radioactive species in the HLW. Results of similar treatments for all other important species are also shown in Fig. 1, and they are summed to obtain the heavy curve. Its initial value of 4×10^7 means that if all the waste from 1 GWe-yr were fed to people after 10 years, we could expect 40 million cancers; on the other hand, if this feeding were delayed for 100,000 years, there would be only 1300.

If the waste is not reprocessed, but rather is buried as spent fuel, the corresponding results are shown in Fig. 2. We see that the effects in the early years are little changed, but after 100,000 years the toxicity is 30 times higher.

2.2 Probability/Year of Transfer from Rock into Groundwater

Now that we know the toxicity of the waste, the remaining problem is to estimate the probability versus time for an atom of buried waste to be dissolved in groundwater, carried with the latter back toward the surface, and eventually to reach a human stomach. Since the waste is in the form of a rock, it seems reasonable to assume that it will behave somewhat like a normal rock—we consider the validity of this assumption later. We proceed by calculating the release probability for an atom of average rock 600 m underground, which is a typical burial depth planned for the waste.

From the rate at which rivers carry dissolved and suspended material into the oceans, it is straightforward to calculate that the surface of the United States is eroding away at an average rate of 4.5×10^{-5} m/yr (6), or 1 m/22,000 yr. About 28% of this is in solution (6) (the rest is suspended particles), about 15% of the water flow in rivers is derived from groundwater (7, 8) (the rest is from surface runoff), and groundwater contains about twice the solute concentration of river water (9). Applying these corrections leads to the conclusion that groundwater dissolves and removes a total of $(4.5 \times 10^{-5}) \times (0.28) \times (0.15) \times (2) = 3.8 \times 10^{-6}$ m/yr.

We next seek the fraction of this total that is derived from 1 m of depth 600 m below ground level, that is, from between a depth of 599 and 600 m. If the groundwater flow were equal at all depths down to 600 m and zero below 600 m, this would be 1/600 of the total; but that is obviously an overestimate because there is some flow below 600 m and, more important, the flow between 599 and 600 m is much less than the average for the flow between 0 and 600 m. With an elaborate calculation (9) the flow between 599 and 600 m comes out 1/4000 of the total. Thus, $(3.6 \times 10^{-6}/4000) \approx 10^{-9}$ m/yr of depth is removed from between 599 and 600 m. This means that the average probability per year for an atom at this depth to be removed is 1.0×10^{-9}.

We now present an alternative derivation (10, 11) of this key result which is essentially completely independent of the above derivation.

A typical aquifer reaching to the waste burial depth of 600 m is about 100 km long and its flow velocity is about 100 m/yr. For a typical porosity of 10%, the water discharged from it annually per square meter of cross-sectional area is 10% of a volume 1 m^2 in cross section and 100 m long, or 10 m^3. Chemical analyses of groundwater (6) indicate that it typically contains 30 mg/L of calcium so the 10^4 kg of water in this 10 m^3 would carry 0.3 kg/m^2·yr of calcium into the river it feeds. This is the first entry in column 2 of Table 1,

TABLE 1. Calculation of Fractional Removal per Year of Rock Materials by a Typical Aquifer

Element	Amount of Element ($kg/m^2 \cdot yr$) Discharged with Water	Amount of Element (kg/m^2) in Rock ($\times 10^6$)	Fraction of Element Removed per year ($\times 10^{-8}$)
Ca	0.3	15	2
Mg	0.03	3	1
K	0.02	3	0.7
Fe	0.003	9	0.03
U	3×10^{-6}	8×10^{-4}	0.3
SiO_2	0.2	150	0.13
CO_3	1.5	18	8

and other entries in column 2 are obtained analogously. The volume of rock traversed by the aquifer is $10^5\ m^3/m^2$, which weighs about $3 \times 10^8\ kg/m^2$. Typical rock contains (10, 11) about 5% calcium, so the calcium content in the rock traversed by the aquifer is about (0.5) $\times (3 \times 10^8) = 15 \times 10^6\ kg/m^2$. This is the first entry in column 3 of Table 1, and the other entries in that column are obtained analogously. If the rock contains $15 \times 10^6\ kg/m^2$ of calcium, and $0.3\ kg/m^2$ of calcium are removed each year, the fractional removal of calcium is $(0.3/15 \times 10^6) = 2 \times 10^{-8}$ per year. This is the first entry in column 4 of Table 1 and from this example we see that column 4 is obtained as column 2 ÷ column 3, which then allows us to determine the other entries in column 4.

From the results in column 4 it is apparent that for most elements the fractional removal per year is less than 10^{-8}. The elements most similar chemically to the important components in the waste are uranium and iron for which removal rates are much less than 10^{-8}/yr, and the important waste components bear little chemical similarity to CO_3 ions, the only material removed at a rate appreciably higher than 10^{-8}/yr.

We conclude that average rock, traversed by an aquifer, is removed at a rate somewhat less than 10^{-8}/yr, perhaps something like 4×10^{-9}/yr. But only about one-fourth of all rock is traversed by an aquifer at a given time. Thus, average rock is removed at a rate of about $(\frac{1}{4}) \times (4 \times 10^{-9}) = 1 \times 10^{-9}$ per year. This is the same as the result obtained above by a very different method.

2.3 Probability for Transfer from Groundwater into Human Stomachs

To complete our exposure and risk estimates, we need the probability for an atom dissolved in groundwater, which would eventually discharge into a river, to enter a human stomach. The most important contribution to this probability is from the use of wells for potable water supplies. The total water flow in U.S. rivers (6) is 1.7×10^{15} L/yr and the water flow in shallow aquifers is 16% of this (7), or 2.8×10^{14} L/yr. About 45% of the U.S. population (12), 1.05×10^8 people, ingest an average of 1.95 L/day (2) from these wells, a total of $(1.95) \times (365) \times (1.05 \times 10^8) = 7.4 \times 10^{10}$ L/yr. The probability for a given atom dissolved in this water to be ingested is then $(7.5 \times 10^{10}/2.8 \times 10^{14}) = 2.6 \times 10^{-4}$. There are additional contributions from the use of rivers for potable water supplies, the eating of fish, and the use of rivers for irrigation, which bring the total probability to 4×10^{-4} (13).

2.4 Numbers of Deaths Expected

If the probability for an atom of rock to be transferred from rock at 600-m depth to a river is 1×10^{-9}/yr, and the probability that it will get into a human stomach from this process

is 4×10^{-4}, the total probability per year for transfer from the rock into human stomachs is the product of these, or 4×10^{-13}/yr. An alternative independent derivation of this quantity has been developed (13), and it gives essentially the same result. We then apply this probability per year to the waste. Multiplying the ordinates in Figs. 1 and 2 by 4×10^{-13} only requires a shift in the ordinate scale, as shown by the scales on the right side of each of Figs. 1 and 2. This gives the deaths/yr·GWe-yr. To find the total number of eventual deaths, we must integrate these curves over time.

This raises the question of the time period over which integration is to be performed. For example, if it is the first million years, the result is 0.0026 and 0.021 deaths/GWe-yr from HLW and spent fuel, respectively. Another reasonable upper limit would be the time when the toxicity of the waste reaches that of the uranium originally mined because after that time, more lives would be saved by removal of the uranium (preventing it from entering human stomachs via groundwater release) than would be lost to the waste. That time is 14,000 years for HLW and 120,000 years for spent fuel, up to which the integrations yield 0.0018 and 0.014 deaths/GWe-yr, respectively. To carry the calculation beyond 1 million years requires a more complex treatment, taking into account the reducing depth of the waste as overlying material is eroded away, and assuming that when the material reaches the surface it becomes dissolved in rivers. The final results (9–11), are 0.006 and 0.021 deaths/GWe-yr for HLW and spent fuel, respectively, not including the effects of the uranium in the waste since that material was originally in the ground before it was mined out to produce reactor fuel.

2.5 Differences between Buried High-Level Waste and Average Rock

Before these results can be applied to HLW, however, we must take into account differences between buried waste and the average rock we have been discussing. There are basically three ways in which the waste is less secure than average rock:

1. To bury the waste, it is necessary to dig shafts down from the surface; this raises the possibility that these shafts may serve as entries for water or escape paths for the waste. The resolution of this problem depends on our ability to seal these shafts. The experts in that technology (of which I am not one) seems confident that they can seal the shafts to make them as secure as the original rock (14)—they often say more secure.

2. The radioactivity of the waste generates substantial amounts of heat, and there have been worries that this might crack the rock and thereby compromise the security of the waste. This problem has been extensively studied (15) and all indications are that rock fracture does not become a problem until temperatures reach 350°C. Since current designs for repositories limit temperatures to little more than 150°C, rock cracking seems not to be a problem. If more conservatism is desired, lower temperatures can be achieved either by spreading the waste over larger areas or by delaying burial. Heat generation declines by a factor of 10 after 100 years and by a factor of 100 after 200 years.

There are also concerns that if groundwater should contact the waste, the elevated temperature would accelerate its dissolution. However, present plans are to seal the waste in a leach-resistant casing (16) that virtually guarantees isolation from groundwater during the period of elevated temperatures.

3. Glass is thermodynamically less stable than other rocks and hence is more easily dissolved. Actually, however, the difference in dissolution rates found in tests is relatively small; the rate for waste glass is similar to that for basalt and is only about 10 times higher than that for the most durable rock materials (17). These data correlate well with thermodynamic expectations; the ratios are therefore expected to apply, at least roughly,

in a wide variety of conditions. Research on this matter is continuing, and if it is decided that glass is much less secure than average rock, other waste forms such as synroc, which chemically is effectively identical to average rock, are available.

It thus seems apparent that there are no very important ways in which buried waste is less secure than the average rock with which it is compared. On the other hand, there are several ways in which it is more secure:

1. The waste will be emplaced in a carefully selected rock formation, which is presumably more secure than the "average" location of our average rock.
2. The leach-resistant casing in which waste packages will be sealed gives a complete backup safety system that should prevent escape of the waste even if everything else goes wrong (16). This casing is obviously not available to average rock.
3. If waste should escape, the elevated radioactivity would easily be detected by routine monitoring of river and well water in plenty of time to avert health impacts.

Considering all the differences between the buried waste and the average rock with which it is compared, the former seems to be comparable in security to the latter.

2.6 Conclusions on the Burial of High-Level Wastes

Reprocessing of spent fuel involves removal of 99.5% of the uranium and plutonium for use as fuel in future reactors. It could be argued that it is unfair to consider reprocessed HLW because its burial still leaves the plutonium, which has to be disposed. Our treatment of spent fuel includes this plutonium. If nuclear power has a long-term future, it will be burned up in breeder reactors, making the spent fuel treatment an overestimàe of the danger. Nevertheless, we use the result for spent fuel, 0.02 deaths/GWe-yr, as the effects of HLW in further discussion. Note that even for the case of spent fuel burial, we must still consider the residues from uranium mining and milling and the depleted uranium left at the isotope enrichment plant; these are discussed later.

It is interesting to compare the health impact of HLW—0.02 deaths/GWe-ye—with that of the best known waste from coal-burning power plants—air pollution. Typical estimates for air pollution from coal burning are about 50 deaths/GWe-yr (18) with present practice; even giving credit for improvement due to tightening of environmental regulations would probably not reduce this below 20 deaths/GWe-yr, 1000 times the toll from nuclear HLW. Note that we are not considering the nonfatal illnesses caused by this air pollution, once estimated to be 60,000 cases/GWe-yr (18).

It should be recognized that the calculations presented here (and in the remainder of this chapter) do not address the problem of individual, peculiar, or unusual situations. They average over all possible situations, weighting each with its probability for occurrence.

3 CHEMICAL CARCINOGENS FROM COAL BURNING AND PHOTOVOLTAIC POWER (19)

3.1 Risk if Material Enters Human Stomachs

A consortium of U.S. government agencies—the Environmental Protection Agency (EPA), the Consumer Product Safety Commission, the Department of Health and

TABLE 2. Calculation of Deaths/GWe-yr Due to Carcinogenic Elements Released in Coal Burning, Integrated over 10^5 Years and over $\sim 10^7$ Years

Element	Deaths/g Ingested	Probability Ground → Stomachs	Deaths/t in Ground	Coal Deaths/GWe-yr	
				10^5 yr	10^7 yr
Cd	0.0013	0.013	8	20	5
Be	0.0053	0.0005	2.9	13	16
Cr	0.001	0.00002	0.19	7	39
Ni	0.00008	0.0006	0.047	1.4	8
As	0.0001	0.0024	0.24	10	1.3
			Total	50	70

Source: Reference 19.

Welfare, the Food and Drug Administration, the Department of Agriculture, and the Occupational Safety and Health Administration—have adopted the linear, no-threshold dose–response relation for chemical carcinogens (20). They have also established a Carcinogen Assessment Group within the EPA to determine the "slopes" of these linear relations, the risk/unit exposure. Unfortunately, this group only makes particular assessments when requested to do so by a government agency, so it is sometimes necessary to extend their work. For example, cadmium inhalation has been judged to cause prostate cancer with a risk of 0.0038/g inhaled (21). In order to cause prostate cancer, the cadmium must be transported from the lung to the prostate gland by the bloodstream. Since orally ingested cadmium can also get into the bloodstream, it must also present a cancer risk. Studies show that 15% of inhaled cadmium, versus 5% of orally ingested cadmium, gets into the bloodstream (1a). This implies that the cancer risk of orally ingesting cadmium is 5%/15% = 1/3 that of inhaling it, or 0.0013/g ingested. Risks from the other established everlasting chemical carcinogens—beryllium, arsenic, chromium, and nickel—are similarly available (19) and are listed in Table 2.

3.2 Probability of Transfer from the Ground into Human Stomachs

Consider the fate of a cadmium atom located randomly in the ground. The principal process for its removal is erosion followed by river transport into the oceans. Rivers carry 10^{15} g/yr of U.S. soil into the oceans (6); since cadmium is present in the ground at 0.19 ppm, about $(0.19 \times 10^{-6}) \times (10^{15}) = 1.9 \times 10^8$ g/yr of cadmium is removed from U.S. soil by this process.

From chemical analysis of food, it is estimated (22) that per capita dietary intake of cadmium is 3×10^{-5} g/day in the United States. The quantity of cadmium entering U.S. male stomachs each year—prostate cancer is a male disease—is then $(3 \times 10^{-5}$ g/day) $\times$ (365 days/yr) $\times$ $(1.1 \times 10^8$ men) $= 1.2 \times 10^6$ g/yr. The probability for a cadmium atom randomly located in the ground to enter a male stomach before being washed into the oceans is therefore $(1.2 \times 10^6/1.9 \times 10^8) = 0.6\%$.

3.3 Deaths Expected from Coal Burning

The number of cancers eventually expected per metric tonne (t) of cadmium in the ground can now be calculated to be (0.0013 deaths/g) $\times$ $(10^6$ g/t) $\times$ (0.006) = 8 deaths/t. A coal-

burning power plant consumes 3×10^6 t/GWe-yr of coal, with an average cadmium content of 0.8 ppm. Since this cadmium eventually ends up in the ground, coal burning will cause $(3 \times 10^6) \times (0.8 \times 10^{-6}) \times (8$ deaths/t$) = 20$ deaths/GWe-yr. Similar analyses for other carcinogenic elements are listed in Table 2; we see that their effects total 50 deaths/GWe-yr.

The 1×10^{15} g/yr of U.S. soil carried into the oceans corresponds to an erosion rate of 1 m of depth per 22,000 years. Assuming that the ashes from coal burning are distributed through the top 5 m of soil, the time scale for these 50 deaths/GWe-yr is about 10^5 years. If we extend our consideration to multimillion-year time periods, it could be argued that the coal will eventually reach the surface by erosion of overlying material, so the cadmium in the coal would be released even if the coal were not mined and burned. That nullifies the health effects of the cadmium in the coal but introduces another cadmium source.

Burning coal makes the carbon in it disappear permanently from the ground: the carbon dioxide product becomes distributed largely between the atmosphere and the oceans. When erosion eventually brings the coal near to the surface, the carbon in the coal will be replaced by average rock. However, this carbon contains no cadmium—the cadmium in the coal has already been taken into account—whereas its replacement does contain cadmium as a trace element impurity. This cadmium in the replacement rock represents an additional health hazard. It causes $(3 \times 10^6$ t rock/GWe-yr$) \times (0.19 \times 10^{-6}$ t Cd/t rock$) \times (8$ deaths/t Cd$) = 5$ deaths/Gwe-yr. Effects of other carcinogenic elements are similarly calculated in Table 2, from which we see that the total is 70 deaths/GWe-yr.

3.4 Deaths Expected from Photovoltaic Power

It is interesting to point out that deployment of a photovoltaic array for solar electricity requires a great deal of steel, glass, cement, and perhaps aluminum. Producing these requires about 3% as much coal burning as would be required to generate the same amount of electrical energy by direct coal burning (19). Thus, photovoltaics should be charged with 3% of the health effects of coal burning: $(3\% \times 70) = 2.1$ deaths/GWe-yr. Since coal or nuclear plants require at least an order of magnitude less materials, we ignore this problem for them.

One option for photovoltaics under serious consideration is the use of CdS photovoltaic cells. These require about 10 t/GWe-yr of cadmium; if this material ends up in the ground, it will cause (10 t/GWe-yr) $\times$ (8 deaths/t) = 80 deaths/GWe-yr. Since most cadmium used in the United States is imported, this represents a net increase in health impacts here even in the very long term. Deep burial can postpone these health effects but cannot reduce them because, unlike radioactive waste, the cadmium does not decay away.

4 RADON PROBLEMS

The principal decay chain for uranium-238 (^{238}U) is shown in Fig. 3. When uranium decays into thorium (Th), or when thorium decays into radium (Ra), there is no movement of the material, but when radium decays into radon (Rn) the situation is very different: radon is a noble gas and thus can often diffuse away from its original location. If it is near the surface of the soil, it can percolate up into the atmosphere where it and its short half-life decay daughters can be inhaled by people. The radon itself is rapidly exhaled, but the decay

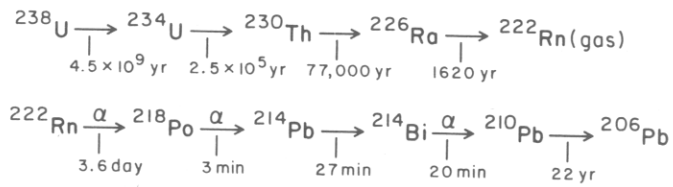

Figure 3. Decay schemes for uranium-238 and radon-222.

daughters—^{218}Po, ^{214}Pb, and ^{214}Bi—stick to the surfaces of our bronchial tubes and bombard the latter with alpha particles, which can cause lung cancer. Since uranium is present in all soil, there is radon in the air everywhere, causing a significant health hazard.

The health effects of radon have been investigated intensively among miners who worked in poorly ventilated mines and were therefore exposed to high levels of radon (3). Among one group of 4000 U.S. uranium miners so exposed, there were 159 fatal lung cancers up to 1974, versus only 25 normally expected. When this problem was first recognized in the late 1960s, ventilation was dramatically improved to the point where exposure to radon is of relatively negligible importance relative to the other hazards in mining.

It is worthwhile to point out that reducing air leakage in buildings in order to conserve fuel traps radon inside and therefore increases human exposure. Using the data from studies of miners coupled with measurements of radon levels in houses, it is straightforward to calculate that radon is now causing about 10,000 deaths/yr in the United States; the tightening of homes now recommended will cause an additional several thousand deaths per year (23). By comparison, the average number of deaths per year expected from nuclear power is generally estimated to be about 10 by government reports, or a few hundred according to the antinuclear activist organization Union of Concerned Scientists (24). Thus energy conservation is by far the most dangerous energy strategy from the standpoint of radiation hazards!

4.1 Uranium Mill Tailings

The best-known radon problem connected with the nuclear industry is the uranium-ore-processing mill tailings. After the ore is mined, it is carried to a nearby mill, where the uranium is chemically separated out and all remaining materials, known as tailings, are left at the site in a form resembling a sandy beach. We see from Fig. 3 that this includes the ^{230}Th and all of its decay products, so it continues producing radon as before, decreasing only with the 77,000-year half-life of ^{230}Th. Since without mining this radon would have been formed underground and therefore would have been much less accessible to the atmosphere, this represents a hazard created by uranium mining. Analysis (25) indicates that this will eventually cause 300 deaths/GWe-yr integrated over the 77,000-year half-life. Fortunately, however, there is an easy solution to this problem: covering the tailings with a few meters of soil reduces the emissions by a factor (26) of 20—diffusion through this layer allows time for decay of the 3.8-day half-life of radon. This is now legally required (27), and it reduces the effects to $300/20 = 15$ deaths/GWe-yr. Note that this is still almost 1000 times higher than our estimated effects from HLW: if we are worried about radioactive waste from the nuclear industry we should worry about mill tailings much more than about HLW.

4.2 Uranium Mining

A more important radon consideration is the lives saved by mining uranium out of the ground to fuel nuclear power plants (28). Most of the health effects of radon are due to uranium (and its daughters) in the top 1 m of the ground. Since the area of the United States is 8×10^{12} m^2, and the ground has a density of 2.7 t m^{-3} and contains 2.7 ppm of uranium, the total quantity of uranium involved is (1 m) $\times$ (8×10^{12} m^2) $\times$ (2.7 t m^{-3}) $\times$ (2.7×10^{-6}) $= 6 \times 10^7$ t. It is this uranium that is causing 10,000 deaths/yr in the United States, and it will continue to do so for 22,000 years, the average time required for 1 m of depth to erode away. Thus, this 6×10^7 t of uranium will eventually cause $10{,}000 \times 22{,}000 = 22 \times 10^7$ deaths, or 3.7 deaths/t. Of course, only a tiny fraction of the uranium mined is from the top 1 m of the ground, but as erosion proceeds, essentially all uranium in the ground will eventually spend its average 22,000-years in the top 1 m, causing 3.7 deaths/t.

Fueling 1 GWe-yr of nuclear power requires mining 160 t of uranium and therefore saves $160 \times 3.7 = 590$ lives. However, this 1 m/22,000 yr erosion rate applied to mill tailings and their covers causes 29% of that number of deaths (28); about 170. Thus, the net effect of the radon from mining and milling uranium is to save $590 - 170 = 420$ lives.

Before we can count these lives as saved, we must trace what happens to the uranium that is mined. Some of it ends up with the HLW, but 80% of it is left as depleted uranium (99.8% ^{238}U, 0.2% ^{235}U) at the enrichment plant. If nuclear power has a long-range future, it will be burned up as fuel for breeder reactors. However, a viable alternative would be to dump it in oceans, where it has an average residence time of about 1 million years before becoming incorporated into the bottom sediments. Essentially all uranium in the ground is destined eventually to be eroded into the oceans and spend its million years therein, so dumping it in the oceans now has no long-term net health effects.

4.3 Radon Exposure from Coal Burning

There is still one other radon problem to be discussed: coal contains an average of 1 ppm of uranium (some coals contain as much as 40 ppm) and after the coal is burned, this uranium and its decay daughters ends up in the top few meters of the ground, serving as a source of radon (28). This causes (3×10^6 t coal/GWe-yr) $\times$ (1×10^{-6} t U/t coal) $\times$ (3.7 deaths/t U) $= 11$ deaths/Gwe-yr. These deaths will occur over the next 10^5 years. Over a multimillion-year period, the coal with its uranium would have reached the surface by erosion of overlying materials, but as for the chemical carcinogens discussed above, the fact that the carbon in the coal will have effectively disappeared and will be replaced by average rock means that the uranium in this replacement rock is an additional source of radon. Since average rock contains 2.7 ppm of uranium, this will cause (3×10^6 t GWe-yr) $\times$ (2.7×10^{-6} t U/t) $\times$ (3.7 deaths/t U) $= 30$ deaths/GWe-yr.

5 LOW-LEVEL WASTE (LLW)

The nuclear industry generates large amounts of low-level radioactive wastes including resins from demineralizers used to clean the reactor cooling water; filter elements from air and water cleaning; reactor components made radioactive by exposure to neutrons; and contaminated gloves, clothing, tools, instruments, and equipment. This material is appropriately packaged and buried in 6-m deep trenches at carefully selected sites governed by extensive regulations (29).

A risk analysis requires estimation of the transfer probability per year, T, from the ground into human stomachs. To do this (30), we assume that the buried waste becomes randomly distributed through the ground between the surface and depth D, and then we assume that an atom of this waste has the same transfer rate T as an atom of the same element in that soil. For example, we assume that T for ^{137}Cs is the same as T for natural Cs in the soil. The latter quantity is readily calculated as $T = q_1/q_2$, where q_1 is the quantity of Cs entering human stomachs each year, known from chemical analyses of food (2), and q_2 is the quantity of Cs in U.S. soil down to depth D, calculated from the measured abundance of Cs in soil (31).

Before proceeding, a value for D must be chosen. The new regulations (29) require that the water table be far below the trench bottoms and that there be good drainage downward from the trench to the water table, so the trenches cannot be flooded. Under these conditions, waste mobilized by water percolating through can only move downward until it reaches the top of the water table. We thus choose D to be the latter depth; a typical value is 20 m. The results are just proportional to D, so they would not be greatly changed by any other reasonable choice of its value.

Average daily dietary intake for cesium is 1×10^{-5} g/day (2); hence, $q_1 = (1 \times 10^{-5}$ g/day) $\times$ (365 days/yr) $\times$ (2.3×10^8 population) $= 8.2 \times 10^5$ g/yr. The density of soil is 2 g/cm^3 and it contains 7 ppm of cesium; hence, q_2, the quantity of cesium in the top 20 m of U.S. soil, is (20 m) $\times$ 8×10^{12} m^2 area) $\times$ (2.0×10^6 g/m^3) $\times$ (7×10^{-6}) $= 2.2 \times 10^{15}$ g. Thus, $T = (8.2 \times 10^5/2.2 \times 10^{15}) = 3.6 \times 10^{-10}$/yr. Since the half-life of ^{137}Cs is 30 years, its average lifetime is 43 years; hence, the total probability for transfer of ^{137}Cs from the soil into human stomachs is $(3.6 \times 10^{-10}) \times (43) = 1.6 \times 10^{-8}$. The LLW from 1 GWe-yr (32) contains 11 Ci of ^{137}Cs; a calculation analogous to that given for ^{239}Pu in our discussion of HLW indicates that we may expect 5.8 cancers/Ci of ^{137}Cs ingested by humans (5). The number of cancers expected from the ^{137}Cs in LLW is therefore (5.8 cancers/Ci) $\times$ (11 Ci) $\times$ (1.6×10^{-8}) $= 1.0 \times 10^{-6}$. Analogous calculations are shown for other components in LLW in Table 3. For transuranics, transfer probabilities T were obtained from systematics versus position in the periodic table. All materials remaining after 10^5 years were assumed to be released into rivers by erosion, with a 10^{-4} probability for human ingestion.

TABLE 3. Calculation of Deaths/GWe-yr from Principal Components of Low-Level Waste (7.5E4 means 7.5×10^4)

Radionuclide	Half-Life (yr)	Ci/GWe-yr	T (Prob. yr)	Total Probability	Deaths/Ci Ingested	Deaths/ GWe-yr
^{59}Ni	7.5E4	1.9	1.8E − 9	2.4E − 4	0.018	6.0E − 6
^{60}Co	5.2E0	1600	7.1E − 9	5.5E − 8	0.67	5.9E − 5
^{129}I	1.7E7	0.0012	4.3E − 8	4.4E − 3	65	3.4E − 4
^{137}Cs	3.0E1	11	3.6E − 10	1.6E − 8	5.8	1.0E − 6
^{238}U	4.4E9	0.0062	2.2E − 10	1.2E − 4	8	6.0E − 6
^{239}Pu	2.4E4	0.016	7.9E − 10	3.4E − 5	38	2.1E − 5
^{240}Pu	6.6E3	0.008	7.9E − 10	7.5E − 6	38	2.2E − 6
^{241}Am	4.6E2	0.078	7.9E − 10	5.1E − 7	180	7.2E − 6
^{243}Am	7.7E3	0.0012	7.9E − 10	8.7E − 6	180	1.9E − 6
					Total	4.0E − 4

Source: Reference 30.

The deaths/GWe-yr for each radionuclide listed in Table 3 sum to a total of 4×10^{-4} deaths/GWe-yr. This is our estimate of the health consequences of LLW burial.

The principal questionable assumption in this risk analysis is that an atom of LLW is no more easily picked up by plant roots than an atom of the same element that is part of the mineral structure of soil. As a test of this assumption, we may refer to experiments in which T is determined by injecting radioactive tracers of various elements into the soil and later measuring the amount of radioactivity in the edible parts of plants grown in that soil. Numerous experiments of this type have been reported (32), and values of T obtained from them average about the same as values of T obtained from the ratio of dietary intake to concentration in soil described above (30). This confirms that atoms artificially inserted into the soil (like LLW) are no more easily taken up by plants than atoms of the same element in the mineral structure of the soil.

6 REVIEW OF THE VARIOUS RISKS ASSOCIATED WITH POWER GENERATION

The results of the risk analyses we have presented and two others are summarized in Table 4 in the column headed Eventual. The analysis for routine emissions of radioactivity from nuclear plants is by the United Nations Scientific Committee (4) and the analysis for transport of HLW is by a group from Sandia Laboratory (33).

We seen from Table 4 that there are three different types of waste from coal burning, *each* of which is at least a thousand times more harmful to human health than the wastes from nuclear power that draw so much public concern. Clearly, this public concern is very much misdirected.

While public discussion of hazards of buried waste usually centers on the very-long-term risks listed in Table 4 as "eventual," there are many good reasons to limit our consideration only to the next few hundred years (34). That is *not* because the lives of people living many thousands of years from now are less valuable than those of people living now, but because of the following considerations:

TABLE 4. Summary of Deaths/GWe-yr from Wastes Produced in Electricity Generation

Source	First 500 yr	Eventual
Nuclear		
High-level waste	0.0005	0.02
Radon problems	−0.065	−420
Routine emissions	0.05	0.3
Transport (radiation only)	0.0001	0.0001
Low-level waste	<0.0001	0.0004
Coal burning		
Air pollution	20	20
Chemical carginogens	0.5	70
Radon	0.11	30
Photovoltaics		
Coal for materials	0.6	4
Add if CdS	0.8	80

1. There is an excellent chance that a cure for cancer will be developed within the next few hundred years, in which case the projected deaths will never materialize.
2. Historically, money has drawn at least 3% real interest (after allowing for inflation) continuously for at least 5000 years. If the money spent to protect our distant progeny from our wastes were invested at even 1% interest, it would make available to them tremendous sums of money that could be spent to save enormous numbers of their lives. Simply not spending this money and thereby reducing the national debt would be essentially equivalent to this, giving our distant progeny more money to spend on themselves.
3. Spending money now on biomedical research is enormously more effective for saving future lives than spending money to protect them from our wastes.

In view of these considerations, Table 4 also contains risk estimates for the various wastes added up over only the next 500 years. In arriving at these estimates, effects of materials deposited in the top layers of soil are taken as 1% of those expected over the next 100,000 years (there is more deposit in the upper layers of soil). Lives saved by uranium mining are those due to surface mining. For high-level radioactive waste, one order of magnitude credit is given for the many important time delays associated with release scenarios.

Clearly, as shown in this 500-year perspective, the three types of waste from coal burning are *each* many times more harmful to human health than the nuclear wastes.

7 USEFULNESS OF PUBLIC EDUCATION REGARDING RELATIVE RISK

In order for the public, regulators, and other scientists to make rational decisions regarding various choices involving technological issues, it might be useful to educate them regarding risks associated with activities which they understand. One approach to the process is to compare activities by assuming that all the eventual deaths occur now, or what is equivalent, that current waste generation rates and disposal practices continue for millions of years, and consider the effects of all accumulated wastes. Certainly, it would be useful to first present the risks posed by the nuclear HLW and other energy or natural radiation related hazards.

For HLW, Table 4 lists 0.02 deaths/GWe-yr, and U.S. electricity generation (assumed to be all nuclear) is 250 GWe-yr/yr; hence, the total annual effect is $0.02 \times 250 = 5$ deaths/yr. If the U.S. population had a stable age distribution, there would be 3×10^6 deaths/yr (actually there are 2×10^6); hence, the average person's risk of dying from HLW is $\frac{5}{3} \times 10^{-6} = 1.7 \times 10^{-6}$. The average person who does die from the HLW loses about 20 years of life; hence, the average loss of life expectancy is $(1.7 \times 10^{-6}) \times (20\,\text{yr}) \times (365\,\text{days/yr}) = 0.012$ days, or 18 min.

The educational process could then proceed into a discussion of relative risks. It was shown during Ruckelshaus' most recent tenure as head of the EPA that by comparing some environmental risks posed by certain activities with risks deemed acceptable by the public, the EPA was able to dismiss demands to regulate trivial hazards so that they could focus on higher-priority issues. One approach that is easily understood is to discuss the risks that reduce life expectancy by about 18 min:

- A regular cigarette smoker smoking $1\frac{1}{2}$ extra cigarettes in his or her lifetime.

- An overweight person eating 120 extra calories in his or her lifetime.
- An overweight person increasing body weight by 0.006 ounces.
- Driving an extra $\frac{1}{2}$ mile every year (35).

Hopefully, such efforts will help educate the public so that they will be able to make more rational decisions as we move into a period when such decisions will be too frequent to address in only an emotional, uninformed manner.

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