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A New Paradigm for High-Level Radioactive Waste Disposal: Intrinsic Radionuclide Properties and Comparative Hazard

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Abstract

Disposal regulations and technologies for high-level radioactive waste have evolved separately from other hazardous wastes, despite shared concerns about long-term risks. This study compares these historical trajectories and reassesses the hazard profile and physical/geochemical properties of key radionuclides. Long-lived radionuclides have low specific activity and emit little or no penetrating radiation; their risks arise primarily from internal exposure, analogous to chemical carcinogens disposed of in the shallow subsurface. Actinides have carcinogenic potency comparable to dioxin but are strongly immobilized under reducing conditions, whereas mobile long-lived radionuclides have lower potency. These findings support a paradigm shift in disposal strategies: (a) shifting the long-term safety basis from heavily engineered systems toward inherent geochemical/physical mechanisms; (b) considering trade-offs between hypothetical future risks and present-day environmental impacts; and (c) harmonizing regulatory requirements for disposal, including long-term stewardship. Such a unified, lifecycle-aware framework will enhance overall environmental protection and improve the sustainability of energy systems.

Significance Statement

High-level radioactive waste (HLW) disposal remains a critical challenge for nuclear energy deployment, yet radiological hazards are often evaluated separately from other environmental risks. This study shows that the long-term HLW hazards are governed primarily by long-lived radionuclides that emit little or no penetrating radiation and pose risks dominated by internal exposure pathways similar to chemical carcinogens. In addition, intrinsic radionuclide properties contribute to long-term safety, with actinides exhibiting limited mobility in the deep subsurface and mobile long-lived radionuclides associated with relatively low cancer risks. These findings underscore the need to incorporate lifecycle considerations and to better align disposal and stewardship expectations across radioactive and chemical wastes to support long-term public health and environmental protection.

Main Text

Introduction

High-level radioactive waste (HLW)— including spent nuclear fuel (SNF) and associated processing wastes—has been recognized as one of the key challenges for deploying and expanding nuclear energy (1). The scientific and technological foundations for deep geological repositories (DGRs) have been established to ensure long-term containment of radionuclides over geologic timescales based on multi-barrier systems, and to minimize potential public health impacts (2, 3). In parallel, repository assessment methodologies—modeling radionuclide transport in the geosphere and exposure pathways—were developed to quantify potential risks to the public and to demonstrate regulatory compliance (4). Beyond these technical advances, substantial effort has also been devoted to institutional designs, governance and public engagement (e.g., 5, 6, 7).

Yet substantial resistance persists for siting DGRs. Even for sited European DGRs, concerns remain about the longevity of engineered barrier systems (8). Furthermore, paradoxically, emphasizing greater depth and increasingly sophisticated—and costly—materials does not necessarily build confidence; instead, it often reinforces the perception that HLW is uniquely

dangerous (9). Often missing from the discussion is a clear account of the actual hazard that HLW poses to public health compared to other hazardous and energy waste.

The hazard posed by radionuclides and low-dose ionizing radiation is primarily associated with DNA damage and the resulting carcinogenic effects. However, over the past five decades, more than 100 chemical substances have been classified as carcinogenic to humans (10), many of which are disposed of in the shallow subsurface or still released into the environment, even though some are persistent and non-degrading. Although Willrich and Lester (11) explicitly called for comparative evaluations of radioactive waste hazards relative to other environmental pollutants in the 1970s, such analyses have been limited and largely ignored in policy-making. In particular, the potential exposure to radionuclides from DGRs is mainly through ingestion via groundwater uses. More fundamentally, long-lived actinides and fission products are primarily internal carcinogens which emit little or no penetrating gamma radiation (12, 13, 14, 15, 16). These radionuclides therefore warrant direct comparison with chronic ingestion–based risks from genotoxic chemical contaminants.

The objective of this paper is to re-evaluate the characteristics and hazards posed by radionuclides in HLW as they evolve over time, so as to inform disposal strategies and system designs. First, we provide a brief historical review of HLW disposal concepts in comparison to other hazardous waste. We then quantify the hazard and characteristics of key radionuclides, focusing on the presence of penetrating gamma radiation and cancer risk from ingestion. Cancer risks are estimated using established regulatory methodologies, applied separately to radiological and chemical carcinogens, and expressed as the lifetime cancer risk associated with ingesting a specified volume of water at a given contaminant concentration (17, 18). In addition, we evaluate their geochemical characteristics that control their mobility in the geosphere. Finally, we discuss how the resulting hazard profile and characteristics can guide disposal strategies.

Beyond the identification of chemical carcinogens, two major developments have emerged over the last several decades that have not been adequately reflected in the discussions of HLW disposal. First, the remediation of large-scale soil and groundwater contamination—mostly associated with past improper waste management—began in the 1980s (20). Experience has shown that complete cleanup is elusive; many large/complex sites are expected to remain under institutional control indefinitely, owing to persistent residual contaminants (21, 22). In parallel, remediation strategies have shifted from heavily engineered systems toward more sustainable approaches, reflecting growing recognition that intrinsic geochemical/hydrological processes provide more robust long-term protection (23, 24, 25).

Second, remediation activities at US nuclear weapon production sites since the 1990s have yielded detailed observations of radionuclide migration across diverse geological and climatic settings. These observations consistently show that environmental impacts are dictated primarily by a small subset of highly mobile radionuclides rather than by a bulk radionuclide inventory (26). Actinides generally exhibit limited mobility except for particulate/soil-bound transport on the surface (27). In addition, chemical substances (e.g., chromium, mercury, organic solvents) often pose greater risks than radionuclides (28, 29).

These empirical observations—the reliance on long-term land stewardship, the transition of safety bases to intrinsic properties, and the central role of contaminant mobility —motivate a reassessment of HLW disposal strategies. This does not diminish the need for robust isolation; rather, the intent is to shift attention from the current focus on radionuclide inventory and half-lives alone to the hazards posed by specific radionuclides in comparison with chemical carcinogens, as well as the geochemical retardation as an inherent safety function. By providing this broader comparative perspective, this study aims to stimulate renewed discussion of disposal strategies

and regulatory frameworks that strengthen environmental protection and improve the sustainability of energy systems.

Overview of Radioactive and Other Hazardous Waste Disposal

Comparative History

The development of HLW disposal concepts predates the regulatory frameworks established for other hazardous wastes. In the US, comprehensive environmental regulations were not implemented until the 1970s. Prior to this period, industrial facilities commonly released waste directly into the environment, leading to severe contamination (e.g., 30). The major environmental legislations, including the Resource Conservation and Recovery Act (RCRA, 1976)—markedly improved air and water quality nationwide (31, 32). In contrast, HLW management planning began almost immediately after commercial nuclear power generation started. In 1957, the National Academy of Sciences (NAS) recommended geologic disposal for HLW (33).

Such early development is attributed partly to the fact that radiation was one of the earliest carcinogens recognized in the early 20th century (34). In addition, radiation is one of the easiest carcinogens to measure *in situ*, even with technologies available by the 1940s (35). Radiation subsequently became the first carcinogenic hazard to receive a formal, systematic regulatory framework such as the International Commission on Radiological Protection (ICRP) established in 1928 (36).

In contrast, while chemical carcinogens were recognized sporadically from the 19th to 20th centuries (37, 38, 39), a modern standardized framework did not begin until the 1970s, when the International Agency for Research on Cancer (IARC) Monographs Programme was established in 1971 (10). Consequently, many chemical carcinogens had been discharged into the environment before health effects were well understood. Since the 1980s, hazardous-waste management practices have been improved significantly, transitioning from routine releases to intentional isolation, although considerable attention was paid balancing the regulatory requirements and financial burdens to industries (40).

In addition, the amount of waste is important; SNF is small compared to other energy wastes, *i.e.*, 24.6 Mg of SNF per 1 GW-year of electricity production as opposed to 500,000 Mg of coal ash and 8.3 million Mg of CO_2 from coal energy (41, 42, 43). The electricity value generated per unit of waste is approximately $48.8M per Mg of SNF for nuclear energy compared to $2,494 per Mg of coal ash (see Methods). Such a high energy density and high profit per unit of waste allowed the nuclear industry to afford the development of proper waste-management solutions. Since then, SNF has been contained in storage pools or dry casks at power plant sites with extensive monitoring programs.

Current Disposal Concepts

Typical HLW repositories are based on a multi-barrier concept that combines engineered and natural barriers (2). Engineered barriers include waste forms (UO_2 fuel matrix for SNF or vitrified glass for waste from reprocessing) designed to limit radionuclide release, metal canisters to provide near-complete containment for 10^3–10^4 years, and bentonite clay buffers to retard radionuclide transport (44, 45). In addition, the natural barrier—such as crystalline, clay/shale, or salt host rock in deep subsurface—is characterized by low permeability (hydraulic conductivity), and reducing geochemical conditions favorable for radionuclide retention. Repository safety assessments follow probabilistic safety analysis methodologies that model radionuclide release, transport, and

exposure pathways to quantify potential consequences and their likelihood under both expected and disruptive scenarios over regulatory timeframes of 10^4–10^6 years (4).

For chemical hazardous wastes, RCRA defines both disposal design requirements and waste acceptance criteria in the US (40 CFR 264). RCRA disposal cells for hazardous waste are near-surface engineered containment systems that incorporate a redundant engineered double liner system at their base—commonly consisting of composite barrier comprised of a compacted clay layer with low hydraulic conductivity overlain by a polymeric geomembrane, a leak detection layer, and an upper polymeric geomembrane. The final cover system is composed of similar composite barrier layer, a drainage layer, and erosion protection. RCRA disposal cells also include a leachate collection and removal system, with the leachate being treated before discharge to the environment. In addition, the disposal cells rely on a combination of active and passive institutional controls, including groundwater monitoring, maintenance, and corrective action programs. The post-closure compliance/monitoring period was initially defined as 30 years, although US EPA subsequently required that management continue until the waste becomes non-toxic, effectively implying indefinite management for persistent contaminants (46).

The RCRA disposal cell designs reflect a prescriptive approach rooted in traditional civil engineering practice, by specifying system designs and barrier properties such as liner thickness and hydraulic performance (47, 48, 49). Substantial research has been devoted to evaluating the longevity and performance of the barrier components. Geomembranes are designed to provide effective containment over time scales on the order of 1000 years (49), while clay liners minimize water flux and promote diffusion-dominated transport conditions (48). However, RCRA does not require site-specific risk assessments as part of regulatory compliance.

Hazards of HLW and Characteristics of Individual Radionuclides

The hazard associated with HLW is governed by the time-dependent inventory of radionuclides and by their physical/chemical characteristics that determine exposure pathways. Although more than 300 radionuclides are produced in reactors, the long-term inventory is dominated by only ten radionuclides contributing more than 0.1% of total activity at 5000 years (Figure 1). Fission products that emit intense high-energy gamma radiation decay out within the first 1000 years, after which radioactivity is dominated by actinides. In addition, a limited number of long-lived fission products (Se-79, I-129, Cs-135) are considered important in safety assessments due to their high mobility despite low inventories (Table 1).

Table 1 illustrates that long-lived radionuclides are weakly radioactive as reflected by their low specific activities, compared with typical short-lived fission products such as Cs-137. This is because radioactivity (decays per unit time) is inversely proportional to half-life. For example, U-238 (half-life 4.46 billion years) poses health risks dominated by chemical nephrotoxicity rather than radiological dose (50). In addition, there are short-lived decay daughters (Np-239, Nb-93m, Pa-233), which are in secular equilibrium (i.e., equal activities) with their long-lived nuclides (Am-243, Zr-93, Np-237, respectively).

Beyond 1000 years, the hazard of HLW is dominated by radionuclides whose potential health impacts arise primarily through internal exposure pathways. This is partly because the principal pathway for radionuclides from DGRs is ingestion through well-water use. More fundamentally, however, it is because these radionuclides exhibit no gamma radiation or only low-energy and/or low-intensity gamma emissions (Table 1). Internal dose from alpha- and beta-emitting radionuclides already dominates total risk throughout this period; what changes at ~1000 years is specifically the external gamma contribution, which declines to negligible levels as short-lived fission products decay out. This reflects fundamental nuclear decay processes: long-lived radionuclides decay predominantly via alpha/beta emissions, whereas intense penetrating gamma radiation is associated with short-lived excited nuclear states that undergo prompt electromagnetic

de-excitation (52). Low-energy gamma radiation has limited penetrating power; for example, the mean free path of <75 keV gamma radiation (*e.g.*, Am-243/241, Np-237, I-129) is a few centimeters in a body and a few millimeters in rock (51). Exceptions are the short-lived ($t_{1/2}$ < 30 d) decay daughters Np-239 and Pa-233, which emit moderate-energy gamma radiation. However, the associated emission intensities are low because these gamma rays arise as secondary emissions following beta decay (53, 54). The resulting activities also fall within limits permitted for low-level radioactive waste disposal (55, 10 CFR 61.55, < 26TBq/ m^3 or 700Ci/m^3). Although the Pu isotopes emit neutrons from spontaneous fissions, the health impact is negligible (16) due to its small decay fraction (e.g., 0.00057% for Pu-240).

Because long-lived radionuclides pose health risks primarily through internal exposure pathways following ingestion, their potential hazards can be evaluated in comparison with chemical carcinogens using pathway-consistent metrics (see Methods). For radionuclides, the internal dose hazard and cancer risk are evaluated using slope-factor constructs—i.e., cancer morbidity coefficients per unit lifetime intake—that integrate radionuclide-specific biokinetics, organ irradiation geometry, and epidemiologically derived risk models (17, 56, 57). For chemicals, cancer slope factors represent the excess lifetime probability of cancer per unit of chronic intake normalized by body mass (18). A common basis for comparison can therefore be established by estimating lifetime cancer risk associated with chronic ingestion of drinking water (2 L/day) containing a specified contaminant concentration (1 ppb). This comparison does not imply equivalence between radiological and chemical risk mechanisms, but rather provides a consistent exposure-based metric for evaluating long-term ingestion hazards.

Using this framework, the calculated cancer risk for long-lived actinides is comparable to or slightly lower than dioxin (Tables 1 and 2). The highest cancer risk is associated with Am-241, owing to its relatively short half-life compared with the other actinides. In contrast, long-lived fission products (I-129, Se-79, and Tc-99) yield cancer risks lower than those of chemical carcinogens such as dioxins and perfluorooctanoic acid (PFOA) and comparable to those of perfluorooctane sulfonate (PFOS) and arsenic.

After disposal, radionuclide transport in groundwater is controlled primarily by chemical properties, especially solubility and sorption. Solubility provides an upper limit on their dissolved concentrations after their release. The dissolved ions can then adsorb onto mineral surfaces, further reducing the concentrations and retarding the transport (58). Solubility is based on thermodynamic solubility values, while sorption is frequently described using empirical distribution coefficients (K_d in Table 1). Under the reducing conditions typical of deep geological environments, actinides and high inventory fission products (Tc-99, Zr-93, Nb-93m) exhibit low solubility and strong sorption (i.e., high K_d), resulting in limited mobility (Table 1; 59). In contrast, I-129, Se-79, and Cs-135 exhibit weak sorption (i.e., low K_d) and therefore higher relative mobility in groundwater systems. These retardation effects can be expressed in terms of diffusive transport times through a 1-m-thick bentonite buffer or comparable clay-rich host rock. Mobile radionuclides (I-129, Se-79, and Cs-135) may traverse the buffer within 500–30,000 years, while actinides and other fission products require more than one million years for comparable transport, allowing substantial radioactive decay within the repository.

HLW Disposal Design Principles

Inherent Safety Mechanisms

The evolution of HLW radionuclide inventories and exposure pathways provides inherent long-term safety through intrinsic radionuclide properties. Within 1000 years, high-intensity/energy gamma emitters decay to negligible levels. This transition marks a shift from external exposure to hazards

dominated by internal exposure pathways. The residual radiological hazard is then dominated by long-lived actinides, which are expected to remain strongly confined under repository conditions.

Certain engineered barrier components can be credited over geological timescales, particularly waste forms and clay buffers, the effectiveness of which have been supported by comparison with natural analogues. Under reducing conditions, the degradation of uranium dioxide—the dominant SNF waste form—is limited by low U(IV) solubility and a stable oxide lattice (61, 62). For vitrified waste forms, archaeological analogues indicate that glass surface alteration remains minimal in ~5000 years (63). Clay mineral alteration is limited when thermal conditions are controlled, consistent with observations in natural bentonite deposits (64, 65, 66).

Highly mobile long-lived fission products—such as I-129, Se-79, and Cs-135—cannot be fully contained within the repository. However, slow waste-form dissolution and low-permeability buffers are expected to substantially limit release rates and promote dilution within the geological barrier, decreasing concentrations and associated health impacts. These radionuclides are weakly radioactive and pose carcinogenic risks comparable to or lower than those of persistent chemical contaminants routinely managed in shallow disposal facilities (Tables 1 and 2). Consequently, deep geological repositories are expected to provide more robust containment and risk reduction for these radionuclides than chemical carcinogens.

The strategies outlined above are largely in line with the European DGR programs, siting repositories in stable and water-saturated host formations that maintain chemically reducing conditions and limit radionuclide mobility. Switzerland, for example, selected a site that ensures diffusive transport and reducing conditions in addition to other key geological criteria, including groundwater age, separation from potable aquifers, and erosion potential (68). Beyond this existing framework, our analysis provides a mechanistic basis for decreasing hazards over time and for emphasizing the role of intrinsic radionuclide properties to ensure long-term safety.

Trade-offs between Future Hypothetical Risk vs Current Actual Environmental Impacts

Comparative metrics for long-lived radionuclides and persistent chemical carcinogens provide a basis for evaluating disposal strategies within a broader environmental-impact context. Discussions of intergenerational burden in HLW management often focus narrowly on the premise that delaying disposal would transfer burdens to future generations, and that future generations should not be responsible for maintaining repositories (69). What is frequently overlooked, however, is the trade-off between *current, actual environmental harms* and *future hypothetical, probabilistic risks*. Repository safety assessments typically rely on highly conservative assumptions and extremely unlikely scenarios, motivating increasingly complex and resource-intensive barrier systems. This approach reduces projected long-term risk but imposes disproportionate resource demands on the present generation and may itself harm the present-day environment that needs to be preserved for future generations.

One illustrative example is the use of critical metal for primary waste packaging. Some countries plan to use copper canisters due to the possibility of crystalline rock fractures intersecting the repository and subsequent advective transport, necessitating strong reliance on engineered barriers (70). However, an analysis assuming complete canister failure indicates that peak far-field doses could still meet regulatory limits, except for mobile long-lived radionuclides, as others are strongly retarded by sorption (71). As shown above, the cancer risks of these mobile radionuclides are comparable to or lower than those of some persistent chemical carcinogens managed in the shallow subsurface. At the same time, copper is a critical material for modern energy systems and electronics, with rapidly growing demand (72, 73). The extraction and processing of copper impose substantial environmental burdens such as ecological damage and heavy metal release (74, 75).

This contrast underscores the need for lifecycle–based evaluations of waste-package materials and repository designs.

Another intergenerational consideration arises within the nuclear fuel cycle. SNF recycling and actinide transmutation are often proposed as means to reduce HLW volume and radiotoxicity. However, reprocessing releases substantial quantities of radionuclides to the present-day environment (14). In addition, transmuting actinides does not decrease radiological risk, since their mobility is limited in the geosphere (76). At the same time, uranium mining-related wastes have been recognized as posing greater environmental risks than HLW due to shallow disposal and long-term exposure pathways (60, 77). These observations raise a question—whether greater resources should be devoted to mitigating current environmental impacts, as well as whether HLW assessments should explicitly include uranium decay daughters and radionuclides released from reprocessing, when substantial inventories of these radionuclides already reside in the surface environment. Integrated assessments should be required not only to consider HLW disposal risks, but also to account for broader environmental impacts across the energy life cycle.

Harmonizing Requirements for Radioactive and Chemical Hazardous Waste Disposal

Expressing radiological and chemical risks with a common metric reveals a marked disparity between the disposal requirements for HLW and chemically hazardous wastes: extensive long-term safety assessments are required only for HLW, despite the two having similar hazard profiles. In addition, HLW strategies have historically pursued the “bury-and-forget” paradigm that assumes no need for long-term surveillance (69). While conceptually appealing, this approach substantially increases the burden of proof for long-term safety. In fact, considerable effort has been devoted to evaluating hypothetical human intrusion scenarios and developing permanent markers to warn future generations of repository locations (78). However, given that federal/state agencies already accept indefinite monitoring and control at more than 1000 hazardous waste and contaminated sites (22), the consideration of monitored/controlled HLW repositories over an extended time may represent a governance choice rather than a fundamental departure from existing environmental practice.

We acknowledge that the comparative methodology used here for radiological and chemical carcinogenic risks is simplistic, focusing solely on ingestion through drinking water. The underlying risk assessment frameworks differ substantially: radiological risk is evaluated in terms of lifetime dose and cumulative intake (17), while chemical carcinogenic risk is assessed based on daily intake normalized by body mass (18). Recent efforts have increasingly focused on internal radiation exposure and on developing more consistent comparative frameworks for radiological and chemical risks (57). Further development of these integrated approaches is needed to support and normalize waste disposal strategies across different industries.

In general, substantial attention and resources are directed toward HLW disposal, while many chemical carcinogens have become widely distributed in the environment before their hazards were fully recognized. Since its inception, the nuclear industry has pursued long-term waste isolation, while comparable requirements have historically been absent in other industries. For example, there is no oversight for solar panels and batteries, despite their potential to release toxic heavy metals (79). Greater alignment of disposal and stewardship expectations across industries would support more coherent environmental protection and also improve the sustainability of energy systems.

Methods

Electricity Value Generation per Waste

For nuclear energy, we use typical reactor parameters for pressurized water reactors selected as a reference case in Kim et al. (43). Producing the 1 GW-year of electricity requires 21.7 Mg (uranium equivalent; 24.6 Mg of UO_2) of nuclear fuel and yields the same mass of SNF, with the typical operational parameters of 34% thermal efficiency, the burn-up of 50 GWd per Mg of U and 4.5% enrichment. Based on the recent average electricity price (residential customers) in the US of 14.23 cents/kWh (80), the 1GWe·y electricity is equivalent to 1.2 billion US dollars. The electricity value per ton of SNF is then 48.8 million dollars.

For coal energy, the 1 GW-year electricity production creates approximately 500,000 Mg of coal ash and 8.3 million Mg of CO_2 (41). The electricity value per Mg of waste is then $2,494 for coal ash and $145 for CO_2.

Cancer Risk Quantification

In this section, we aim to establish a common metric between radiological and chemical carcinogenic risks. Since groundwater represents a dominant long-term exposure pathway from waste disposal facilities, we focus on the ingestion of drinking water. We consider a scenario in which a person is chronically exposed to a low level of contaminants (1 ppb) in drinking water, consuming 2 L per day on average over a lifetime. This scenario is not intended to represent site-specific exposure conditions, but rather to provide a consistent, pathway-aligned basis for cross-hazard comparison under low-dose, chronic exposure assumptions commonly used in regulatory risk assessment. The resulting estimates are intended as comparative indicators under a common exposure scenario and are used here to evaluate relative hazard magnitude across radiological and chemical contaminants, rather than as precise predictions of individual risk. This comparison is specifically scoped to lifetime excess cancer risk from chronic low-dose ingestion and does not extend to other axes on which radiological and chemical hazards may differ, including dose-response threshold behavior and potential synergistic effects between co-located contaminants.

For radionuclides, the US EPA Federal Guidance Report No. 13 provides the cancer risk coefficients for environmental exposure (17). We use the cancer morbidity risk coefficients for tap water ingestion, defined as the excess lifetime cancer morbidity risk per unit activity ingested (Bq). We assume that an individual drinks 2 L of water per day containing a radionuclide concentration of 1 μg/L (~1 ppb) for a 70-year lifetime. The lifetime cancer risk is calculated as:

$$2\ (\text{L/day}) \times 10^{-6}\ (\text{g/L}) \times 365\ (\text{days/year}) \times 70\ (\text{years}) \times (\text{specific activity, Bq/g}) \times (\text{risk coefficient, Bq}^{-1}) \quad (1)$$

Specific activity is used to convert mass concentration to activity intake, which is then integrated over ingestion rate and exposure duration. These coefficients implicitly account for radionuclide-specific decay characteristics and biokinetics, organ uptake and retention, and internal energy deposition through the dose and risk modeling framework of the Federal Guidance Report No. 13 (17). This formulation further assumes linear proportionality between cumulative activity ingested and excess lifetime cancer risk, consistent with the linear no threshold (LNT) framework adopted in Federal Guidance Report No. 13. The resulting risk estimates represent population-averaged lifetime excess cancer risk and are not intended to predict individual-level outcomes.

For chemical carcinogens, the US EPA provides a linear cancer slope factor for each chemical substance to define the excess lifetime cancer risk per unit of chronic daily intake normalized by body weight $(\text{mg/kg-day})^{-1}$ (18). We used the oral slope factor from the US Department of Energy's Risk Assessment Information System (RAIS; rais.ornl.gov). We assume the same consumption rate of drinking water (2 L/day), the same contaminant concentration (1 μg/L), and a

body weight of 70 kg, consistent with regulatory defaults (81). The lifetime cancer risk is calculated as:

2 (L/day) x 10^{-3} (mg/L) / (body mass, kg) x (slope factor, $(mg/kg \cdot day)^{-1}$). (2)

Although the duration of exposure is not explicitly accounted for in this equation, the cancer slope factor assumes a lifetime exposure of 70 years.

Differences in data sources, dose definitions, and levels of conservatism between radiological and chemical cancer risk frameworks are well documented (57). The radiological risk coefficients are derived primarily from dose-based models historically anchored in external exposure epidemiology, whereas chemical slope factors integrate toxicological and epidemiological evidence using intake-based constructs. These differences do not preclude pathway-consistent comparison when lifetime excess cancer risk is used as a common metric for low-level chronic ingestion scenarios, but such comparisons should be interpreted as comparative indicators rather than precise predictions of individual risk (57).

Diffusive-Transport Time

Apted and Ahn (82) proposed the diffusive-transport time as the measure to evaluate the containment or delay functions of engineered or natural barriers. The diffusive-transport time is defined based on Fick's law as:

$$t_{dt} = \frac{b^2(\varepsilon + \rho K_d)}{D} \quad (3)$$

where *b* is the thickness of barriers or transport distance (m), ε is the porosity (interpreted here as the accessible porosity), K_d is the sorption coefficient (m^3/kg), ρ is the dry bulk density (kg/m^3), and *D* is the effective diffusion coefficient (m^2/y). We used bentonite parameters from the Swiss disposal program (83) under reducing conditions (ε = 0.36, ρ = 1760 kg/m^3, and D = 6.31 x 10^{-3} m^2/y). When a radionuclide has K_d=1 m^3/kg, the diffusive-transport time for 1 meter is 279,000 years. This means that a radionuclide takes 279,000 years on average to move the distance of 1 meter. When K_d = 0 m^3/kg (nonsorbing), the diffusive-transport time is 57.1 years. For anionic species (Se-79 and I-129), the Swiss program accounted for anion exclusion by assuming a reduced accessible porosity (ε = 0.05) and a reduced effective diffusion coefficient (9.47 x 10^{-5} m^2/y).

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Figures and Tables

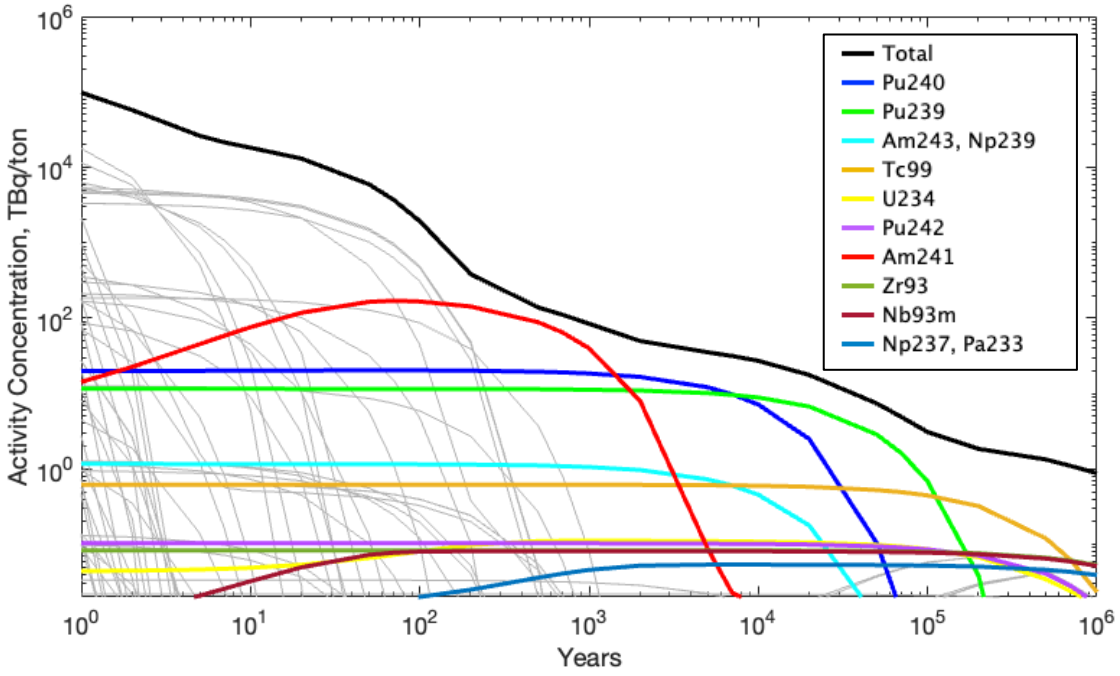

Figure 1. Radioactivity of one metric ton of SNF over time. Neutronics, fuel depletion, transmutation and decay are calculated using the Serpent software (serpent.vtt.fi), assuming a pressurized water reactor (PWR) with the parameters of 50 GWd/MTU burnup and 4.5w% initial enrichment.

690 **Table 1.** Characteristics of key radionuclides after 1000 years (Cs-137 is included as a
691 comparison), with the inventory ranking at 5000 years up to Pa-233. Three radionuclides (I-129,
692 Se-79, and Cs-135) are added given their importance in the previous assessments. The main
693 decay modes are alpha, beta, spontaneous fission (*n*), and isomeric transition (IT). The gamma
694 energy is divided into low (<100 keV), medium (100-500 keV) and high (>500 keV), while the
695 intensity (i.e., emission probability per decay) is divided into negligible (<1%), low (1 – 50%) and
696 high (50–100%) based on the IAEA Nuclide Database (www-nds.iaea.org). The cancer morbidity
697 for drinking water ingestion (per Bq) is from EPA (1999), which is used for the lifetime cancer risk
698 based on the continuous consumption of 2 L of water per day with 1 ppb concentration over the
699 lifetime (70 years; see Methods). The solubility and K_d values are from the Swiss repository
700 assessment under reducing conditions (Nagra, 2002). The diffusive-transport time for 1 m is
701 calculated based on the bentonite properties (see Methods). The short-lived radionuclides (Ba-
702 137m, Np-239, Nb-93m, and Pa-233) are included in their decay parents, since they are at
703 secular equilibrium.
704

Nuclide	Half-life		Specific activity, GBq/g	Decay	Gamma intensity	Gamma energy	Cancer risk slope, /Bq	Cancer risk, ppb-water	Solubility limits, mol/l	Bentonite K_d, m³/kg	1-m transport time, yr
Cs-137, Ba-137m	30.1	y	3.21E+03	β	High	Medium	8.22E-10	1.35E+02	high	0.1	2.79E+04
Pu-240	6561	y	8.40E+00	α, n	Negligible		3.64E-09	1.56E+00	5.00E-08	20	5.58E+06
Pu-239	24110	y	2.30E+00	α	Negligible		3.64E-09	4.27E-01	5.00E-08	20	5.58E+06
Am-243, Np-239	7364	y	7.39E+00	α, β	Low	Medium	2.93E-09	1.11E+00	1.00E-06	20	5.58E+06
Tc-99	2.11E+05	y	6.34E-01	β	Negligible		7.44E-11	2.41E-03	4.00E-09	60	1.67E+07
U-234	2.46E+05	y	2.30E-01	α	Negligible		1.91E-09	2.24E-02	3.00E-09	40	1.12E+07
Pu-242	3.73E+05	y	1.47E-01	α, n	Negligible		3.46E-09	2.59E-02	5.00E-08	20	5.58E+06
Am-241	432	y	1.27E+02	α	Low	Low	2.81E-09	1.82E+01	1.00E-06	20	5.58E+06
Zr-93, Nb-93m	1.61E+06	y	8.85E-02	β, IT	Negligible		5.18E-11	3.32E-04	2.00E-09	80	2.23E+07
Np-237, Pa-233	2.14E+06	y	2.61E-02	α, β	Low	Medium	1.82E-09	2.34E-04	5.00E-09	60	1.67E+07
Cs-135	2.30E+06	y	4.26E-02	β	Negligible		1.23E-10	2.68E-04	high	0.1	2.79E+04
Se-79	3.26E+05	y	5.14E-01	β	Negligible		1.97E-10	5.18E-03	5.00E-09	0	5.28E+02
I-129	1.57E+07	y	6.54E-03	β	Low	Low	3.99E-09	1.33E-03	high	0.0005	9.82E+03

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Table 2. Cancer slope factor and the lifetime cancer risk of chronic ingestion of chemical carcinogens based on the continuous consumption of 2 L per day of 1 ppb concentration over the lifetime (70 years: see Methods). The cancer factors are from the US Department of Energy's Risk Assessment Information System (RAIS; rais.ornl.gov).

	Slope Factor, $(mg/kg\text{-}day)^{-1}$	Cancer risk, ppb-water
Arsenic, Inorganic	32	9.14×10^{-4}
Chromium(VI)	0.16	7.71×10^{-6}
Tetrachlorodibenzo-p-dioxin, 2,3,7,8- (2,3,7,8-TCDD)	130000	3.71
Perfluorooctanoic acid (PFOA)	29300	8.37×10^{-1}
Perfluorooctanesulfonic acid (PFOS)	39.5	1.13×10^{-3}