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

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Abstract

This paper develops a hazard- and pathway-based framework for high-level radioactive waste (HLW) disposal grounded in intrinsic radionuclide decay characteristics, geochemical behavior, and comparative hazard. We examine the physical and geochemical properties of key radionuclides and quantify lifetime cancer risk from chronic ingestion on a per-unit-mass basis using established regulatory models. Long-lived radionuclides are weakly radioactive and emit little or no penetrating gamma radiation; their hazards are therefore dominated by internal exposure pathways, analogous to those of chemical carcinogens commonly disposed of in the shallow subsurface. Actinides exhibit cancer risks comparable to dioxin but are strongly immobilized under reducing deep-geological conditions, while mobile long-lived radionuclides are associated with lower carcinogenic risk than typical persistent chemical contaminants. These findings support a paradigm shift in disposal strategies: (a) from heavily engineered containment systems toward nature-based approaches for ensuring long-term post-closure safety that explicitly leverage intrinsic radionuclide properties, along with slow release from waste forms and diffusion-limited transport, assuming appropriate site selection and geological stability; and (b) toward consideration of lifecycle perspectives and trade-offs between future hypothetical risks and present-day actual environmental impacts, including material use and fuel-cycle emissions. We further highlight asymmetries between radioactive and chemical waste stewardship. Public institutions and regulatory authorities are already responsible for actively managing large inventories of persistent chemical carcinogens in the shallow subsurface indefinitely. Increased efforts are needed to integrate radiological and chemical hazards within a unified environmental risk framework to establish more coherent, lifecycle-aware waste management strategies across industries.

Introduction

High-level radioactive waste (HLW)—defined here in a generic sense to include both spent nuclear fuel (SNF) and associated processing wastes—has been recognized as one of the key challenges for deploying nuclear energy (1). Recently, nuclear energy has re-emerged as a clean and reliable energy source, with numerous advanced reactor designs under development (2). However, while several countries—Finland, Sweden, France, Switzerland, and Canada—have made substantial progress in developing deep geological repositories, many others still lack viable HLW disposal pathways. In the US, six states maintain statutory requirements that a disposal pathway be identified prior to constructing new nuclear reactors (3).

The scientific and technological foundations for HLW disposal were largely established by the 1990s to ensure long-term containment of radionuclides over geologic timescales based on multi-barrier systems, and to minimize potential public health impacts (4, 5, 6). In parallel, repository assessment methodologies—modeling radionuclide transport in the geosphere and exposure pathways—were developed to quantify potential risks and to demonstrate regulatory compliance (7). Beyond these technical advances, substantial effort has also been devoted to institutional designs, governance and public engagement (e.g., 8, 9, 10).

However, HLW disposal discussions often omit explicit treatment of two closely linked questions: (i) how the dominant hazard mechanisms of HLW change over time; and (ii) how radionuclide-specific physical and chemical properties govern exposure pathways and risks. 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 (11), many of which are disposed of in the shallow subsurface or still released into the air and water, even though some are persistent and non-degrading. Although Willrich and Lester (12) explicitly called for comparative evaluations of radioactive waste hazards relative to other environmental pollutants, such analyses have been limited and largely ignored in policy-making.

Regulatory approaches to radioactive waste disposal have historically evolved separately from those governing hazardous chemical waste, despite shared long-term public health objectives and overlapping exposure pathways (13, 14). HLW repository safety has been evaluated based on radiation dose (Sv) resulting from exposure to released and migrated radionuclides from the repository. Although dose provides an aggregate measure for health impacts, it obscures the radiation type, energy, and exposure pathways driving risks. Notably, the dominant dose contributors in the assessments are internal exposure elements (e.g., iodine-129) that emit little or no penetrating gamma radiation (15, 16). These radionuclides therefore warrant direct comparison with chronic ingestion–based risks from genotoxic chemical contaminants.

The objective of this paper is to 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, a brief historical review of nuclear waste disposal concepts is provided in comparison to other hazardous waste. We then quantify the hazard and characteristics of key radionuclides in HLW, focusing on the presence of penetrating gamma radiation (which determines exposure pathways) and cancer risks from ingestion. Cancer risks are estimated following the established regulatory methodologies separately for radiological and chemical carcinogens, using lifetime cancer risk associated with the ingestion of a specified volume of water containing a given contaminant concentration (17, 18). The regulatory risk coefficients for internal emitters reflect radionuclide-specific biokinetics, organ uptake, retention, and clearance (19). In addition, we evaluate the geochemical characteristics of those radionuclides that control their mobility in the geosphere.

Finally, we discuss how the resulting hazard profile and radionuclide characteristics can guide disposal designs and strategies.

Beyond the identification of chemical carcinogens, two major developments have emerged in environmental science since the original HLW disposal design and rulemaking. First, the remediation of large-scale soil and groundwater contamination—mostly associated with past improper waste management—began in the 1980s under the Superfund program (20). Over time, 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, including nature-based solutions and monitored natural attenuation, reflecting growing recognition that intrinsic geochemical/hydrological processes provide robust long-term protection (23, 24, 25).

Second, remediation activities at the 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, 16). Actinides generally exhibit limited mobility except for particulate/soil-bound transport on the surface (27). In addition, chemical substances (such as hexavalent chromium, mercury, organic solvents) often pose greater risks than radionuclides, mainly because of their large plume size (28, 29).

These empirical observations motivate disposal strategies grounded in real-world insights from observing contaminant migration and managing contaminated sites with emphasis on radionuclide-specific mobility and internal dose commitment in comparative hazard evaluation. Building on this perspective, this paper advances a hazard- and pathway-based framework for HLW disposal that aligns repository design, performance expectations, and lifecycle considerations with the time-dependent evolution of radionuclide exposure mechanisms, enabling more realistic and coherent long-term waste management strategies.

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 of air, surface water, and groundwater (e.g., 30, 31, 32). The major environmental legislation—the Clean Air Act (1970), the Clean Water Act (1972), the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA), and the Resource Conservation and Recovery Act (RCRA, 1976)—markedly improved air and water quality nationwide (33, 34). In contrast, HLW management planning began almost immediately after the start of nuclear power generation. In 1957, the National Academy of Sciences (NAS) recommended deep geologic disposal for HLW (35). Since then, an extensive regulatory framework has been developed to govern both high-level and low-level radioactive waste management.

Such early development is attributed partly to the fact that radiation was one of the earliest carcinogens recognized in the early 20th century (36). In addition, radiation is one of the easiest carcinogens to measure *in situ*, even with technologies available by the 1940s, such as ionization chambers and film badges (37). Radiation subsequently became the first carcinogenic hazard to receive a formal, systematic regulatory framework. The International Commission on Radiological

Protection (ICRP) was established in 1928 to develop occupational and public exposure limits (38). The linear no-threshold (LNT) model was established as a regulatory model in the 1950s to quantify health risks associated with ionizing radiation (39). In contrast, while chemical carcinogens were recognized from the 19th to 20th centuries (40, 41, 42), a modern standardized evaluation framework did not begin until the 1970s, when the International Agency for Research on Cancer (IARC) Monographs Programme was established in 1971 (11).

Consequently, many carcinogens had been discharged into the environment before health effects were well understood and a regulatory framework were undertaken. For example, wastewater containing hexavalent chromium (IARC Group 1 carcinogen) was discharged into soil or rivers without much oversight until the 1970s, leading to surface-water pollution (43) and groundwater contamination at more than 300 sites across the US (44). Since then, 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 standards with the additional costs to industries (45).

In contrast, radiation protection standards were established before nuclear technologies became widely available. In addition, the amount of SNF is small compared to other energy wastes, *i.e.*, 24.6 metric tons of SNF per 1 GW-year of electricity production as opposed to 500,000 tons of coal ash and 8.3 million tons of CO_2 from coal energy (46). The electricity value generated per unit of waste is approximately \$48.8M per metric ton of SNF for nuclear energy compared to \$2,494 per ton of coal ash (see Methods). Such a high energy density and high profit per unit of fuel/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 (5). 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 approximately 10^3–10^4 years, and bentonite clay buffers to limit water ingress and retard radionuclide transport following canister degradation (49, 50). Repositories are constructed several hundred meters below ground surface in geologically stable host formations, *i.e.*, the natural barrier—such as crystalline rock, clay/shale, or salt—characterized by low permeability, limited groundwater flow, 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 (7).

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 multilayer engineered liner system at their base—commonly consisting of compacted low-permeability clay and polymeric geomembranes—and a cover system typically composed of similar barrier layers, 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 is 30 years.

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 (51, 52, 53). 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 (53), while clay liners minimize water flux and promote diffusion-dominated transport conditions (52). 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 its constituent radionuclides and by radionuclide-specific nuclear, physical, and chemical characteristics that determine dominant exposure pathways. Although more than 300 radionuclides are produced during reactor operation, the long-term radiological inventory is dominated by a small subset of radionuclides, with approximately ten 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 in groundwater despite low inventories (Table 1).

Table 1 illustrates that long-lived radionuclides (the half-life $t_{1/2}$ > 1000 y) are weakly radioactive 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 (54). 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).

Following the decay of Am-241, the residual hazard of HLW is dominated by radionuclides whose potential health impacts arise primarily through internal exposure pathways, because they exhibit no gamma radiation or only low-energy and/or low-intensity gamma emissions (Table 1). 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 (56). 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 (55). 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 (57, 58). The resulting activities also fall within limits permitted for low-level radioactive waste disposal (59, 10 CFR 61.55, <700 Ci/m^3).

Because long-lived radionuclides pose health risks primarily through internal exposure pathways following ingestion, their potential hazards can be evaluated in a comparative framework alongside 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, 60, 61). 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 ($t_{1/2}$ > 1000 y) is comparable to or slightly lower than dioxin (Tables 1 and 2). The short-lived actinides (Np-239, Nb-93m and Pa-233) exhibit elevated risks due to their high specific activity; however, under disposal conditions, they exist in the same radioactivity as their decay parents due to the secular equilibrium. Their cancer risks are then slightly less than their decay parents (Am-243, Zr-93, Np-237), because their cancer risk coefficients (per radioactivity) are lower than their parents. In contrast, long-lived fission products (I-129, Se-79, and Tc-99) and Np-237 yield cancer risks lower than those of chemical carcinogens such as dioxins and PFOA (perfluorooctanoic acid) and comparable to those of PFOS (perfluorooctane sulfonate) 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 the release from HLW. The dissolved ions can then adsorb onto minerals surfaces and further reduce the aqueous phase concentration and retard the transport (62). 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 concentration 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; 63). 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 the 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.

HLW Disposal Design Principles

Nature-based 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, after which the inventory is dominated by Am-241, a low-energy gamma emitter that decays within ~5000 years. This transition marks a shift from external exposure to hazards dominated by internal exposure pathways. Although gamma-emitting radionuclides remain present beyond this period, their emissions are of relatively low energy and limited penetrating power in geological materials (64). The residual radiological hazard is then dominated by long-lived actinides, primarily Pu-239 and Pu-240, which are expected to remain strongly confined and migrate only over meter-scale distances under repository conditions. In addition, mobile long-lived radionuclides have relatively low carcinogenic risks.

This analysis provides two timescales based on the hazard and radionuclide characteristics, rather than relying on an arbitrary cutoff such as 10000 years (4, 9, 65). On the *human timescale* (<5,000 years, corresponding to the decay of Am-241), engineered systems can reasonably be assumed to provide effective isolation of HLW. Archaeological evidence shows that human-made materials (e.g., glass, metals) can remain minimally altered for thousands of years (66, 67, 68). On the *geological timescale*, safety must rely primarily on intrinsic radionuclide properties, such as low solubility and high sorption, within stable subsurface environments. The limited mobility of radionuclides has also been confirmed by natural analogues, such as uranium ore bodies (69, 70).

Certain engineered barrier components can be credited over geological timescales, particularly waste forms and clay buffers. 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 (71). Natural analogue studies further show that uranium-bearing minerals (*e.g.*, uraninite) have been preserved for millions of years (69). For vitrified waste forms, archaeological analogues indicate that glass surface alteration remains minimal (~a few micrometers) in ~5000 years (68). Clay mineral alteration (*e.g.*, smectite-to-illite conversion which affects sorption) is limited when thermal conditions are controlled, consistent with observations in natural bentonite deposits (72, 73, 74).

Highly mobile long-lived fission products—such as I-129, Se-79, and Cs-135—cannot be fully contained by engineered barriers alone. 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 most existing hazardous waste disposal systems. At the same time, the high mobility assumed for these radionuclides partly reflects limitations in existing experimental and field data. For example, I-129 is commonly modeled as non-sorbing despite recent field observations (75, 76, 77). More research is needed for quantifying their speciation, sorption behavior, and transport.

The strategies outlined above are largely in line with the European HLW disposal programs, which emphasize siting repositories in old, geologically stable host formations that maintain chemically reducing conditions and limit radionuclide mobility. These programs have adopted a safety-case approach that integrates quantitative modeling with complementary qualitative lines of evidence (78). Switzerland, for example, selected a site that ensures geochemically reducing conditions in addition to other key geological criteria, including groundwater age, separation from potable aquifers, and erosion potential (79). 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. In the following sections, we further outline additional considerations for repository design and disposal strategies.

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 framework. 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 and monitoring repositories (80). 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 minerals such as copper for primary waste packaging. Some countries plan to use copper canisters due to the possibility of crystalline rock fractures intersecting the repository and advective transport, which necessitates strong reliance on engineered barriers (81). 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 (82). As shown above, the cancer risks of these mobile radionuclides are comparable to or lower than those of some persistent chemical carcinogens disposed of in shallow disposal cells protected by geomembrane/clay liners. By contrast, copper is a critical material for modern energy systems and electronics, with rapidly growing demand (83, 84). The extraction and processing of copper impose substantial environmental burdens such as ecological damages and heavy metal releases, often disproportionately impacting disadvantaged communities (85, 86). This contrast underscores the need for lifecycle–based evaluations of waste-package materials and repository design choices.

Another intergenerational consideration arises within the nuclear fuel cycle. SNF recycling is often proposed as a means to reduce HLW volume and radiotoxicity. However, this process releases substantial quantities of radionuclides to the present-day environment (15). At the same time, uranium mining and milling wastes have been recognized to pose greater environmental risks than HLW due to shallow disposal and long-term exposure pathways (64, 87). Mill tailing disposal cells are, however, designed only for 1000 years in shallow subsurface. In addition, uranium contamination of surface waters is widespread as a result of phosphate fertilizer applications (88). 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 fuel recycling, 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 on a common metric reveals a marked disparity between disposal requirements for HLW and chemical hazardous wastes. Under RCRA, disposal cells are designed to isolate hazardous waste primarily by geomembrane in composite linear systems, with underlying clay liners serving to retard contaminant migration. These systems have been successful without causing any leakages for the past forty years. Although RCRA initially specified a 30-year post-closure monitoring period for hazardous waste disposal sites, the US EPA subsequently required that such sites remain under institutional control until the waste becomes non-toxic, effectively implying indefinite management (89).

These evolving regulatory practices may motivate re-evaluation of post-closure institutional control for HLW disposal. Historically, HLW strategies have pursued the “bury-and-forget” paradigm that assumes no need for long-term active monitoring and surveillance (80). 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 (4) and developing permanent markers to warn future generations of repository locations (90). However, given that federal/state agencies already accept indefinite monitoring and institutional control at more than 1000 hazardous waste and contaminated sites at the surface (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). In addition, radiological risk coefficients are derived primarily from external exposure data. For internal emitters, cancer risks depend on radionuclide-specific biokinetics, organ uptake, retention, and clearance, as described in contemporary ICRP systemic models (19). Absorbed dose and risk are also governed by parameters such as chemical speciation, oxidation state, particle size distribution, absorption type, and age-dependent physiology, which influence the distribution of energy deposition across target tissues. Recent efforts have increasingly focused on internal radiation exposure and on developing more consistent comparative frameworks for radiological and chemical risks (61). Further development of these integrated approaches is needed to support waste disposal strategies across different industries.

In general, substantial attention and resources are directed toward HLW disposal, while many chemical carcinogens have often become widely distributed in air and water before their hazards are fully recognized. Since its inception, the nuclear industry has been required to plan long-term waste isolation, while comparable requirements have historically been absent or less stringent in other industries. For example, there is no oversight for solar panels and batteries, despite their potential to release toxic heavy metals (91). This contrast underscores that HLW programs have historically pursued the most stringent isolation and containment objectives among industrial waste streams, while persistent chemical hazards are typically managed through long-term stewardship and exposure control rather than permanent isolation. Greater alignment of disposal and stewardship expectations across hazard classes—consistent with exposure pathways and hazard magnitude—would support more coherent long-term protection of public health and the environment.

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. (92). Producing the 1 GW-year of electricity requires 21.7 tons (uranium equivalent; 24.6 tons of UO_2) of nuclear fuel and yields the same mass of SNF, with the typical operational parameters of 34% thermal efficiency, 50 GWd MTU^{-1} burnup and 4.5% enrichment. Based on the recent average electricity price (residential customers) in the US of 14.23 cents/kWh (93), 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 tons of coal ash and 8.3 million tons of CO_2 (46). The electricity value per ton 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 common 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.

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 (L/day) x 10^{-6} (g/L) x 365 (days/year) x 70 (years) x (specific activity, Bq/g) x (risk coefficient, /Bq)

(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 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 (94). The lifetime cancer risk is calculated as:

$$2\ (\text{L/day}) \times 10^{-3}\ (\text{mg/L}) \,/\, (\text{body mass, kg}) \times (\text{slope factor, } (\text{mg/kg·day})^{-1}). \tag{2}$$

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 (61). 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 (61).

Diffusive-Transport Time

Apted and Ahn (95) proposed the diffusive-transport time as the measure to evaluate the containment or delay functions of engineered/natural barriers. The diffusive-transport time is defined based on Fick's law as:

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

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 (96) under reducing conditions ($\varepsilon = 0.36$, $\rho = 1760$ kg/m^3, and $D = 6.31 \times 10^{-3}$ 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 ($\varepsilon = 0.05$) and a reduced effective diffusion coefficient (9.47E-05 m^2/y).

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Figures and Tables

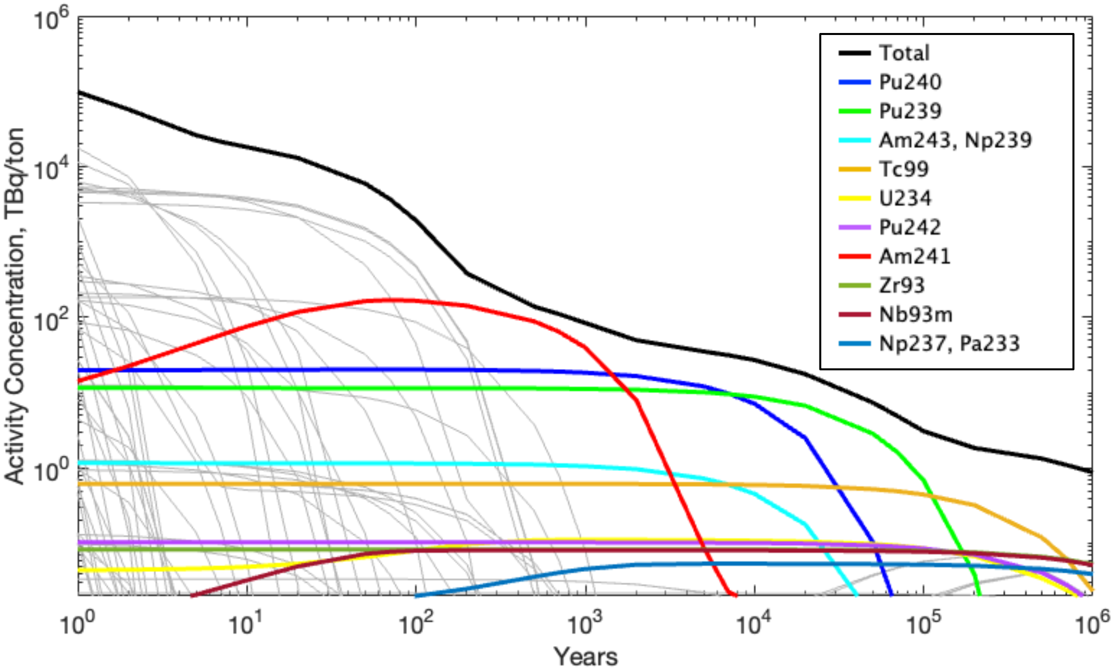

Figure 1. Radioactivity of one metric ton of SNF over time. Neutronics, fuel depletion, transmutation and decay are calculated using the Serpent software (serpent.vtt.fi), assuming a pressurized water reactor (PWR) with the parameters of 50 GWd/MTU burnup and 4.5w% initial enrichment.

Table 1. Characteristics of key radionuclides after 1000 years (Cs-137 is included as a comparison), with the inventory ranking at 5000 years up to Pa-233. Three radionuclides (I-120, Se-79, and Cs-135) are added given their importance in the previous assessments. The gamma energy is divided into low (<100 keV), medium (Med., 100-500 keV) and high (>500keV), while the intensity (i.e., emission probability per decay) is divided into negligible (<1%), low (1 – 50%) and high (50–100%) based on the IAEA Nuclide Database (www-nds.iaea.org). The cancer morbidity for drinking water ingestion (per Bq) is from EPA (1999), which is used for the lifetime cancer risk based on the continuous consumption of 2 L of water per day with 1 ppb concentration over the lifetime (70 years; see Methods). The solubility and Kd values are from the Swiss repository assessment under reducing conditions (Nagra, 2002). The diffusive-transport time for 1 m is calculated based on the bentonite properties (see Methods). *Cs-137 includes the gamma radiation from the decay daughter Ba-137m.

Nuclide	Half-life		Specific activity, GBq/g	Decay mode	Gamma intensity	Gamma energy	Cancer risk slope, /Bq	Cancer risk, ppb-water	Solubility limits, mol/l	K_d, m³/kg	1-m diffusion transport time, yr
Cs-137*	30.1	y	3.21 x10^{3}	beta	High	High	8.22 x10^{-10}	1.35 x10^{2}	high	0.1	2.79x10^{4}
Pu-240	6561	y	8.40	alpha	Neg.		3.64 x10^{-9}	1.56	5.0x10^{-8}	20	5.58x10^{6}
Pu-239	24110	y	2.30	alpha	Neg.		3.64 x10^{-9}	4.27x10^{-1}	5.0x10^{-8}	20	5.58x10^{6}
Am-243	7364	y	7.39	alpha	Low	Low	2.79 x10^{-9}	1.05	1.0x10^{-6}	20	5.58x10^{6}
Np-239	2.36	d	8.56x10^{6}	beta	Low	Med.	1.39 x10^{-10}	6.08x10^{4}	5.0x10^{-9}	60	1.67x10^{7}
Tc-99	2.11x10^{5}	y	6.34x10^{-1}	beta	Neg.		7.44 x10^{-11}	2.41x10^{-3}	4.0x10^{-9}	60	1.67x10^{7}
U-234	2.46x10^{5}	y	2.30x10^{-1}	alpha	Neg.		1.91 x10^{-9}	2.24x10^{-2}	3.0x10^{-9}	40	1.12x10^{7}
Pu-242	3.73x10^{5}	y	1.47x10^{-1}	alpha	Neg.		3.46 x10^{-9}	2.59x10^{-2}	5.0x10^{-8}	20	5.58x10^{6}
Am-241	432	y	1.27x10^{2}	alpha	Low	Low	2.81 x10^{-9}	18.2	1.0 x10^{-6}	20	5.58x10^{6}
Zr-93	1.61x10^{6}	y	8.85x10^{-2}	beta	Neg.		3.01 x10^{-11}	1.36x10^{-4}	2.0x10^{-9}	80	2.23x10^{7}
Nb-93m	16.1	y	8.85x10^{3}	IT	Neg.		2.17 x10^{-11}	9.81	3.0x10^{-5}	30	8.37x10^{6}
Np-237	2.14x10^{6}	y	2.61x10^{-2}	alpha	Low	Low	1.67 x10^{-9}	2.23x10^{-3}	5.0x10^{-9}	60	1.67x10^{7}
Pa-233	27.0	d	7.68x10^{5}	beta	Low	Med.	1.5 x10^{-10}	5.88x10^{3}	1.0x10^{-8}	5	1.39x10^{6}
Cs-135	2.30x10^{6}	y	4.26x10^{-2}	beta	Neg.		1.23 x10^{-10}	2.68x10^{-4}	High	0.1	2.79x10^{4}
Se-79	3.26x10^{5}	y	5.14x10^{-1}	beta	Neg.		1.97 x10^{-10}	5.18x10^{-3}	5.0x10^{-9}	0	5.28x10^{2}
I-129	1.57x10^{7}	y	6.54x10^{-3}	beta	Low	Low	3.99 x10^{-9}	1.33x10^{-3}	high	5x10^{-4}	9.82x10^{3}

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}