



Ionizing radiation potential in life cycle impact assessment: broadening the assessment through inclusion of radioactive decay products

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Abstract

Purpose This work briefly reviews the fate, exposure, and effect modules of established ionizing radiation potential (IRP) impact categories and proposes an IRP methodology with the novel inclusion of radioactive decay products (i.e., progeny), referred to as Clemson University Radiological Protection (CURP).

Methods CURP reports characterization factors (CFs) and midpoint dose factors (MDFs) and is comparable to (1) methods established for other impact categories and (2) those used in radiological risk assessment. Consistent with the previously established UCrad approach, USEtox was used to determine the environmental steady-state concentrations. US EPA tools were used to determine the contribution of radioactive progeny to the collective effective dose rate from air submersion, water immersion, ground irradiation, ingestion, and inhalation. These dose rates were then integrated over time to calculate the total collective effective dose.

Results and discussion The CFs for a majority of the 113 radionuclides characterized in CURP remained constant or decreased as time increased (^{238}U had a maximum increase on the order of 10^{-5}). The percent difference between MDFs with and without progeny inclusion demonstrates that the progeny can have a significant influence on the overall impact. For example, the percent difference in MDFs for ^{222}Rn with and without progeny is 198% by 1000 years for all emissions. Other radionuclides have small changes at lower time horizons that become more pronounced as the time horizon increases; an atmospheric emission of ^{226}Ra and ^{235}U yields a percent difference in the MDFs of ~0.2% and ~0.5%, respectively, at a time horizon of 0.01 years and 155% and 134%, respectively at 10,000 years.

Conclusion Ultimately, the time horizon and progeny inclusion can have drastic implications for the resulting MDF, and consequently the CFs. Thus, interpretation of results needs to be carefully considered due to the heavy dependence of time for IRP, especially when considering releases to the continental and global scale or based on a method using steady-state assumptions.

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1 Introduction

The generation or emission of ionizing radiation is associated with nuclear-related services or products (e.g., nuclear energy production or industrial radiography gauges, respectively), non-nuclear energy production (e.g., coal-burning power plants), and non-nuclear processes like mining or unconventional drilling (“fracking”). These “non-nuclear” processes can release naturally occurring radioactive material (NORM) residues into the environment (e.g., radon from phosphogypsum stacks) that could potentially harm workers, the public, or the environment itself (CFR 2024). Robust LCAs necessitate inclusion of relevant impact categories that are associated with realistic impacts. Thus, IRP should ideally be given more consideration than it has historically. Drawing from existing radiological protection approaches can support further development of the IRP impact category used in LCIA along with providing additional insight in interpretation of the associated results.

The goal of this work is to briefly review previously described methods for evaluating IRP through the lens of radiological protection and propose novel modifications to account for the unique nature of ionizing radiation impacts (e.g., radioactive decay) in the IRP category within LCA. Developing a more robust impact assessment methodology for ionizing radiation will allow for a more comprehensive analysis within LCA and ideally broaden the utility of LCA (e.g., into radiological protection decision-making; Wattier et al. 2023). However, differences in LCA and radiological protection frameworks and assumptions mean that LCA-determined impacts cannot be utilized in radiological protection decision-making for quantitative determination of impacts (e.g., risk), but rather must be used qualitatively for comparative purposes. Namely, radiological protection assesses the impact to a specific individual (e.g., a

maximally exposed individual in the case of compliance calculations) based on a site-specific release whereas LCA is intended to determine the impact to any given individual associated with a generic, site-independent release anywhere in the world. These fundamental differences in methodology lead to some disagreement (e.g., time-dependence (or independence) as discussed in Sect. 2.3) for methodological development.

1.1 Radiological protection and radiation dose

Radiological protection (also referred to as radiation protection or health physics) is the science and practice of preventing or mitigating potentially harmful effects of ionizing radiation (Kathren and Ziemer 1980; ICRP 2007). Given that ionizing radiation has many potential benefits, and that some protection strategies can potentially have detrimental social, economic, or environmental impact, careful consideration is given to the context of a given situation (e.g., risks and benefits) to ensure that protection is appropriate for the prevailing circumstances (ICRP 2007). Risk assessment can be used in radiological protection to evaluate potential radiological impact associated with an environmental release of radioactivity, similar to that for chemical contaminants. Although framed in different ways, radiological (and chemical) risk assessment consists, in general, of release (source term analysis), transport (contaminant movement through the environment), exposure (how and to what extent a receptor is exposed), and consequence (risk or impact characterization) assessments which are conceptually analogous to the steps used to quantify IRP following a release of radioactive material (see Sects. 1.2 and 1.3) (NRC 2009; Fjeld et al. 2024). Of note is that risk assessments, and radiological protection considerations in general, are site specific.

Radionuclides are those nuclides with unstable nuclei that undergo radioactive decay (or often a series of radioactive decays) to become stable. A “parent” nuclide refers to the nuclide at the beginning of a decay chain, and for our purposes is the nuclide directly associated with the production of a product or the provision of a service. A “decay product” (or equivalently “progeny”) refers to any nuclide in the parent’s decay chain, the final member of which is stable. Each decay is associated with a release of energy, i.e., radiation (herein “radiation” refers to “ionizing radiation” specifically). As a result, both external and internal exposure routes should be considered for radiological contaminants. This is a unique difference from chemical contaminants wherein chemical exposure must be internal to observe an effect in nearly all cases. Of note is that many radionuclides can be chemically toxic if taken into the body; as an example, the chemical toxicity of (soluble) natural uranium is higher than its radiotoxicity (Rump et al. 2019).

Historically there have been a variety of quantities, often subtle in difference, that have been used to quantify or otherwise represent exposure to radiation, loosely falling under the umbrella of radiation “dose.” For our purposes, effective dose is perhaps the most useful of these quantities. Effective dose (E , with units of J/kg and the special name sievert, Sv) represents energy deposition in mass weighted for (1) the biological effectiveness of the incident radiation type and (2) the comparative radiosensitivity of the tissue(s) irradiated. It makes use of reference anatomical and physiological parameters and therefore indicates the dose “Reference Person” would receive under the given exposure conditions; it is not truly specific to an individual (ICRP 2007). Effective dose is used to estimate the risk of stochastic effects (hereditary effects and cancer) in a population for practical radiological protection purposes (as opposed to providing personalized, individual risk) (ICRP 2021).

Effective dose can be calculated by multiplying an exposure by an appropriate dose (or dose rate) coefficient, where exposure in this sense could be intake (activity taken into the body per unit time) or activity concentration in the environment (e.g., soil contamination that leads to external exposure). Such coefficients have been tabulated by various organizations (e.g., the International Commission on Radiological Protection [ICRP] and the US Environmental Protection Agency [EPA]) for multiple routes and types of exposure (both external and internal) and are typically radionuclide specific (e.g., ICRP 2012, 2020; EPA 1993, 1999).

Collective dose (person.Sv) is the sum of individual (effective) doses (Sv) over a defined group or population with specified bounds on time and dose (ICRP 2007). This metric is intended to be used for comparative purposes, primarily for occupational exposures. It should not be used to sum very low individual doses over long time horizons or over large geographical regions and is not intended to be used in a risk projection of cancer deaths. In other words, individual effective dose can be used as an approximate indicator of possible human health risk, but collective effective dose (that includes low level exposures) is not appropriate to use for detailed risk estimates due to high levels of uncertainty associated with risk at low doses and dose rates (ICRP 2007, 2021). Interpretation of collective dose requires consideration of number of people exposed as well as stratification by age, sex, range of doses received, dose distribution in time, and geographical distribution (ICRP 2007).

1.2 Human health damages method

Frischknecht et al. (2000) were among the first to propose a method to quantify human health damages (HHD) associated with exposure to ionizing radiation in the context of LCA, drawing heavily from the 1995 ExternE (Externalities of Energy) Report (Dreicer et al. 1995). Prior to this, impacts

associated with the release of radioactivity (and subsequent exposure to ionizing radiation) were excluded in LCAs due to lack of available LCIA methods to quantify the damage resulting from such an emission (Frischknecht et al. 2000). The HHD approach includes the following steps: (1) inventory analysis, (2) fate analysis, (3) exposure analysis, (4) effect analysis, (5) damage analysis, and (6) damage assessment and cultural theory, reviewed in detail by Hauschild et al. (2012) and Paulillo et al. (2018).

Of note is that the data used to build the HHD model is based on a variety of geographical regions including France (fate and exposure analyses), North America (exposure analysis), Japan (effect analysis), and unspecified (effect analysis and damage analysis). Overall, characterization factors [$\text{kBq}_{\text{U-235 air equivalents}} / \text{kBq}_{\text{radionuclide}}$] were reported for 29 unique nuclides (or nuclide groups) with 21 nuclides released to air, 13 nuclides released to freshwater, and 15 nuclides released to saltwater for the egalitarian/hierarchist and individualist scenarios (Frischknecht et al. 2000). Also of note is the use of collective effective dose and disability-adjusted life year (DALY) in the damage assessment, although the use of collective dose in this context weakens the relevance of the impact due to the aggregation of very low individual doses (OECD, 2003; ICRP 2007).

Select impact assessment databases (such as ReCiPe2016) incorporate a slightly modified HHD methodology wherein changes include (1) updating the DALY per fatal cancer incidence, (2) consistently implementing the three cultural perspectives (i.e., individualist, hierarchist, and egalitarian), and (3) specifying dose and dose rate effectiveness factors for each cultural perspective (Huijbregts et al. 2016). Otherwise, to the authors' knowledge, the HHD approach has had no other modifications, although Hauschild et al. (2012) recommend that the midpoint calculation of the HHD approach be modified to consider spatial differentiation and releases from urban environments. They also suggest that the endpoint calculation is likely unreliable due to the high uncertainties associated with the DALY for severe hereditary effects and thus do not recommend its use.

1.3 Critical group method and UCrad

Paulillo et al. (2018) argue that an acceptable LCA method for IRP needs to be site-independent, produce average impacts, quantify impacts for a larger quantity of radionuclides, and be applicable to both direct discharges and releases from a geological repository. While the HHD approach meets some of these qualifications, it has been suggested that it suffers from data limitations (e.g., reliance on the French nuclear fuel cycle) and resulting high levels of uncertainty. To address the gaps, Paulillo et al. (2018, 2020a, b, c) have proposed two new methods, the critical group method (CGM) and UCrad, wherein characterization

factors [$\text{Bq}_{\text{U235 air equivalents}} / \text{Bq}_{\text{radionuclide}}$] are quantified for a larger radionuclide database (i.e., 107 and 115 radionuclides for direct emissions in CGM and UCrad, respectively) and also include geological repository releases (not included in the HHD approach).

The CGM and UCrad methodologies include inventory, fate, exposure, and effects analyses. CGM and UCrad differ only in how they handle radionuclide transport; CGM uses the International Atomic Energy Agency's (IAEA) simplified dispersion models for critical groups 1, 100, 1000, and 10,000 km from the release whereas UCrad uses a multi-compartment model adapted from USEtox, which is an established LCIA model intended to quantify the impact from toxic (chemical) releases (IAEA 2001; Paulillo et al. 2020c; Rosenbaum et al. 2008). To model repository releases, both CGM and UCrad draw from a United Kingdom (UK) safety assessment which was performed to establish how to attain environmental safety for geological disposal in the UK (NDA, 2010). Note that progeny is not included in the impact determination for CGM or UCrad, which is a self-acknowledged limitation of the methodology (Paulillo et al. 2020c).

Exposure and effects are modeled in the same manner for both UCrad and CGM. Usage factors (type and amount of consumed foodstuffs, breathing rate) and exposure factors (time spent in a contaminated environment) were used in conjunction with internal and external dose coefficients to ultimately get to risk (Fantke et al. 2017 in the Excel documentation; IAEA 2001; IAEA 2014; EPA 1993) using the nominal risk coefficient for cancer incidence adjusted for fatality and quality of life (0.055 /Sv; ICRP 2007). The authors did not include risk of hereditary effects, which have much lower probability than cancer occurrence (Paulillo et al. 2020c).

2 Method development

As the models used to assess the fate in both the HHD methodology and CGM rely on site-specific parameters specific to a release from the nuclear fuel cycle, neither methodology is truly generalizable (as is standard in conventional LCA) to determine the impacts associated with the emission of ionizing radiation. Thus, the Clemson University Radiological Protection (CURP) methodology was developed and is based on the ideas presented in UCrad. Note that we were unable to fully reproduce the previously proposed IRP methodologies which is a recognized challenge not just in LCA (Frischknecht 2004; Laurent et al. 2014), but in general for many scientific disciplines (e.g., Joshi and Bhardwaj 2018; Burgelman et al. 2019).

CURP was designed to characterize the midpoint impact resulting from radiological emissions with consideration of

progeny and across variable time horizons. Releases from waste disposal, radionuclides that do not follow standard fate modules (e.g., ^{14}C and ^3H ; Dreicer et al. 1995), and evaluation of endpoint impacts are beyond the scope of this work. In CURP, the steps used are inventory assessment, fate assessment, exposure assessment, and effects assessment.

2.1 Fate

CURP follows the USEtox steady-state methodology to determine the environmental fate of radionuclides emitted to the atmosphere, freshwater, and saltwater, with a modification of the saltwater partition coefficient (detailed in Supplemental 1 Section A.1). USEtox (a) allows emissions to household air, occupational air, urban air, continental rural air, continental freshwater, continental sea water, continental agricultural soil, continental natural soil, and crop residues (Fantke et al. 2017) and (b) reports steady-state concentrations for air (indoor, urban, continental, and global), freshwater (continental and global), continental saltwater, global ocean, agricultural soil (continental and global), and natural soil (continental and global). USEtox is a Microsoft Excel-based application wherein the radionuclide-specific data was inputted, and the steady-state environmental fate was calculated for each of the aforementioned compartments. For consistency with other IRP methods, only emissions to continental air, freshwater, and saltwater were used to determine the steady-state concentrations in all reported compartments. No additional compartments were added for CURP. Details about all radionuclide-specific parameters utilized in USEtox to determine the fate can be found in Supplemental 1 Section A and Supplemental 3.

The concentrations for each foodstuff (i.e., above ground vegetation, below ground vegetation, freshwater fish, saltwater fish, drinking water, milk, beef, poultry, goat/sheep, and pork) were modeled based on the steady-state environmental concentrations reported in USEtox. For clarity, food chain transport was modeled externally from USEtox as discussed in detail in Supplemental 1 Section A.1. After the concentration in each foodstuff was calculated, a hold time (Supplemental 2 Table 2) was applied. The hold time represents the time between harvest, slaughter, etc. until it is consumed. Thus, the activity was decay corrected for the appropriate hold time. Modeling the fate of radionuclides in this way allows the transport of radionuclides and chemicals to be modeled in the environment in the same way and also limits the need for site-specific data.

2.2 Exposure

There are five exposure routes considered in CURP: ingestion, inhalation, air submersion, ground irradiation, and water immersion. Default values regarding breathing rate

and foodstuffs intake rates were pulled from USEtox. Note that exposures were aggregated (i.e., collective exposures) rather than effective doses, so technically collective exposures were multiplied by dose coefficients to get collective effective dose rates as detailed below.

Details about each exposure route can be found in Supplemental 1 Section A.2. Briefly, the internal collective activity rates for inhalation and ingestion $[\text{Bq/year}]_{\text{collective}}$ were determined by multiplying the air or foodstuffs concentration $[\text{Bq/m}^3 \text{ or } \text{Bq/kg}]$ with the intake rate $[\text{m}^3/(\text{person} \cdot \text{day}) \text{ or } \text{kg}/(\text{person} \cdot \text{day})]$ and the population size. The external collective exposure concentration for air submersion $[\text{Bq/m}^3]_{\text{collective}}$, water immersion $[\text{Bq/m}^3]_{\text{collective}}$, and ground irradiation $[\text{Bq/m}^2]_{\text{collective}}$ were determined by multiplying the air, water, or ground concentration $[\text{Bq/m}^3, \text{Bq/m}^3, \text{ or } \text{Bq/m}^2]$ with the fraction of time spent in that contaminated compartment (i.e., exposure factors) and the population size.

2.3 Effects

The collective effective dose rate from internal exposure was calculated by multiplying the activity rate $[\text{Bq/year}]$ taken into the body by the committed effective dose coefficient $(e(\tau))$, where τ is the commitment period, i.e., 50 years). The collective effective dose rate from external exposure was calculated by multiplying environmental activity concentrations $[\text{Bq/m}^3]$ for air or water and $[\text{Bq/m}^2]$ for ground irradiation by the respective effective dose rate coefficient $(\dot{e})$. The effective dose rate coefficient and effective dose coefficient will hereon be referred to as DCs. Additional information about the DCs used can be found in Supplemental 1 Section A.3.1

The time-integrated collective effective dose rate gives the total collective effective dose over the time period of interest (i.e., time horizon). Here, we consider the initial dose rate to be that once steady state in the environment is reached (as in Sect. 2.1). The dose rate is then considered to decline exponentially, consistent with radiological decay. Note that the time required to reach steady state is different for each radionuclide (dependent on its transport properties) and the release compartment. This likely has an influence on the impact assessment; however, as done in previous methods, the impact assessment methodology begins once steady-state equilibrium is achieved. Collective effective dose $[\text{person} \cdot \text{Sv}]$ was determined by integrating the collective effective dose rate over the time interval (i.e., time horizon) of interest. The collective effective dose was calculated for 0.01, 0.1, 1, 10, 100, 1000, and 10,000 years after steady state (with respect to environmental transport) has been reached; this means that CURP combines time-dependent impact calculations (i.e., decay and in-growth) with a time-independent fate methodology. This is an acknowledged limitation of the model; however, the level of complexity

necessary to account precisely for progeny transport was beyond the scope of the current work given the inherent uncertainties associated with generalization and the classically different approach taken in radiological protection compared to LCA (see Sects. 1 and 2.6). Note the specific influence of shared vs independent transport will depend on the specific decay chain (radiological half-lives, chemical form, number of progeny, etc.).

2.4 Characterization factors

As discussed in Sect. 1.1, care needs to be taken when using the collective dose to not include “trivial” doses in the summation. While there is no set value that “trivial” equates to (Smith and Martinez 2017), there are established individual dose limits that should not be exceeded: 1 mSv per year for the public in planned situations (ICRP 2007). Note the dose is allowed to be higher in certain situations such as emergencies (ICRP 2007). In order to give a comprehensive overview of the IRP impacts in CURP, three sets of characterization factors are reported. Characterization factors [$\text{BqU-235 air equivalents} / \text{Bq}_{\text{radionuclide}}$] are used in LCIA to find the relative importance of one release with respect to the reference release. In the case of IRP, the reference radionuclide is ^{235}U released to the atmosphere. In CURP, characterization factors were determined by dividing the midpoint dose factor for each radionuclide [$\text{person.Sv/Bq}_{\text{radionuclide}}$] by the midpoint dose factor of the reference radionuclide, ^{235}U [$\text{person.Sv/Bq}_{\text{U-235, air}}$]. Each midpoint dose factor was calculated using the fate and transport, exposure, and effect (i.e., collective effective dose) assessments.

The three sets of characterization factors reported were very conservative, least conservative, and conservative. Note that qualifiers of “conservative” herein (e.g., most, least, very) are intended to be internal comparisons. The very conservative set includes the collective effective dose to the global and continental populations thus including all impacts, including trivial ones. The least conservative set includes no collective effective doses to the global population and depending on the radionuclide could produce characterization factors that underestimate the impact in the context of LCA. The conservative set of characterization factors (i.e., the ones used in this work) considers the impact on the continental population as well as the impact on the global population for some radionuclides. The global impact was included if the relative difference between the average individual continental and average individual global effective dose rates were less than 25. This number was arbitrarily chosen as discussed in Supplemental 3 to be conservative and include some impacts to the global population while still omitting the global impacts that result in average individual effective dose rates much smaller than the average individual continental dose rates. These three sets of characterization

factors can be found in Supplemental 2 Table 7b–d. Unless specified otherwise, the conservative set of characterization factors is used moving forward.

2.5 Progeny inclusion

Since progeny can also be radioactive (as discussed in Sect. 1.1), progeny can contribute to human health impact resulting from an emission of radioactive material. Consequently, the impacts associated with progeny have been included in the determination of the characterization factor for select radionuclides from the front end of the nuclear fuel cycle (e.g., ^{237}Np , ^{210}Pb , $^{239}\text{Pu}^*$, ^{226}Ra , ^{230}Th , ^{234}U , ^{235}U , and ^{238}U ; NRC 2012), back end of the nuclear fuel cycle (e.g., uranium isotopes, plutonium isotopes, fission products, and transuranics; IPFM 2011), medical applications (e.g., ^{67}Ga , ^{123}I , ^{111}In , $^{99\text{m}}\text{Tc}$, and ^{201}Tl ; NRC and Institute of Medicine Committee on State of the Science of Nuclear Medicine 2007), industrial applications (e.g., ^{241}Am , ^{60}Co , and ^{137}Cs ; Charlton et al. 1975), and while outside the scope of traditional LCAs, weapons testing and accidents (e.g., ^{137}Cs , ^{131}I , ^{90}Sr ; Eisenbud and Gesell 1997). In total, the impact of progeny was included in the characterization factor determination for 21 radionuclides.

All progeny is assumed to stay in the same environmental compartment or foodstuff as the steady-state parent radionuclide in the CURP methodology (this assumption is discussed in greater detail in Sect. 2.6). Consequently, the internal collective intake rates or external collective exposure concentrations (see Supplemental 1 Section A.2) were the starting values to account for the radiological decay of the parent and the in-growth (and potential decay) of subsequent progeny. The Bateman equation–based Radionuclide Decay Chain program created by the EPA (EPA 2020) was used to calculate the decay and in-growth through the decay chain at various points in time. From there, the five exposure route rates or concentrations for each radionuclide in the decay chain were multiplied by the respective DC to get the collective effective dose rate [person.Sv/year] for each exposure route. The total collective effective dose rate was calculated by summing the collective effective dose rates from all five exposure routes. Detailed information about the DCs for each radionuclide for each exposure route can be found in Supplemental 1 Section A.3.1.

Ultimately, the collective effective dose is desired to calculate the damage and characterization factors for each

¹ These radionuclides would only be present in the front end of the nuclear fuel cycle if reprocessing techniques are used. Reprocessing is not done in the USA, but interestingly, these three radionuclides have been emitted from the Paducah gaseous diffusion enrichment plant due to the Paducah diffusion facility enriching uranium in the 1950s to 1970s (NRC 2012).

radionuclide. As was done in Sect. 2.3, the collective effective dose for the parent radionuclide was calculated by integrating the collective effective dose rate over the time horizon of interest. However, the collective effective dose of the progeny could not be calculated this way due to the in-growth and decay of the progeny throughout time. Instead, an averaged time step method was used where the average collective effective dose rate over each time step was multiplied by the duration of the time step. The sum over the time horizon yields the collective effective dose (person.Sv).

2.6 Limitations

Ideally, the fate and transport assessment of progeny would be modeled for each individual radionuclide in a given decay chain (i.e., independent transport vs. shared transport) because transport characteristics are dependent on the particular element and its physiochemical form. However, incorporation of independent transport in the impact assessment calculation is not currently supported by USEtox in the fate module, given that USEtox is designed for chemical transport. A substantial amount of time must pass before reaching steady state wherein decay and in-growth of progeny would be occurring. In this work, it was assumed that the progeny stays in the same environmental compartment as the parent (i.e., shared transport). This assumption means that the environmental fate of the progeny would not be independently considered. For example, if ^{222}Rn was “born” in water, it would likely diffuse to the atmosphere as it is a noble gas. Thus, some of the fate modeling is limited in the current assessment. Alternative approaches to addressing this challenge (although also limited) include (a) modifying the inventory (based on the respective time horizon of interest) to include progeny and have each radionuclide undergo its own fate module [which would be unique to each release scenario and thus not site-independent], or (b) once reaching steady state, model each progeny to undergo an independent transport from the steady-state compartment of the parent radionuclide [practical but likely similar uncertainty as the method employed herein]. To understand the implications of these assumptions, future work could calculate the resulting characterization factors for each of these assumptions and compare them to determine the most appropriate assumption to use. Additionally, assuming that the fate module can be assessed at steady state needs to be evaluated as some radionuclides take a substantial time to reach steady state; for example, atmospheric releases of ^{235}U , ^{212}Bi , ^{90}Sr , and ^{222}Rn take 2.08×10^5 , 1.57×10^5 , 9.52×10^6 , and 9.17×10^4 years, respectively, to reach equilibrium.

The exposure module relies on US-based usage factors for time spent outside, in soil, swimming, bathing, etc. to determine external exposure concentrations coupled with age distributions for the world. While CURP is intended to

assess a generic release anywhere in the world, US data was used due to lack of generalizable data. There was a lack of data for the DCs specific to the progeny. As such, some data was supplemented with that of workers, but most were left blank; these were noted in Supplemental 1 Section A.3.1. An additional limitation comes from the internal effective dose coefficients. This is because the internal effective dose coefficients are defined to be a 50-year committed dose following intake (ICRP 2007) making the result conservative; due to the 50-year committed dose, the largest time horizon is 10,050 years (for internal intakes); however, this is consistent with how doses are calculated and assigned in radiological protection. Of note is that ICRP Publication 103 states that these dose coefficients are specifically for an acute intake.

Finally, for progeny inclusion, modeling the Bateman equations for many small time intervals will allow for a more accurate, integral-like determination of the progeny's impacts. As modeled, the largest percent difference between the integral calculation method and the averaged time step method is 1.28% for ^{240}Pu released to freshwater.

3 Results and discussion

The conservative set of CURP characterization factors (Supplemental 2 Table 7b) was evaluated over 0.01, 0.1, 1, 10, 100, 1000, and 10,000 years to demonstrate the influence of time on the radionuclides due to radiological decay. From there, the midpoint dose factors considering in-growth of progeny were evaluated for select radionuclides at 0.01 years. Finally, the combination of radiological decay and progeny in-growth were considered together. Overall results are presented and discussed below, while a more detailed comparative analysis of the UCrad and CURP methods is discussed in Supplemental 1 Section B.

3.1 Time horizons

While different time horizons do not have meaningful influence on some impact categories, such as water usage, they are critical for calculating ionizing radiation potential due to the radioactive material decaying over time and the potential in-growth of radioactive progeny. The characterization factors herein were evaluated at 0.01, 0.1, 1, 10, 100, 1000, and 10,000 years to allow for a more robust assessment of impacts associated with the additional radiological decay at longer time horizons. Of the 113 radionuclides evaluated, releases to the atmosphere, freshwater, and saltwater at 0.01 years after release resulted in 35, 25, and 26 radionuclides, respectively, that had smaller characterization factors than those presented in UCrad (Supplemental 2 Table 8). In contrast, 10,000 years after the release, 102, 98,

and 96 radionuclides had smaller characterization factors than those calculated in UCrads. This significant increase of radionuclides having smaller characterization factors at the time horizon of 10,000 years is expected as many of the radionuclides have half-lives less than 10,000 years. For radionuclides that have half-lives less than a given time horizon, the midpoint dose factor for the released radionuclide remains constant as the time horizon continues to increase (as little or no additional decay occurs, thus little or no increased effective dose) whereas the midpoint dose factor for ^{235}U increases with increasing time horizon, causing the overall characterization factor to decrease. A further in-depth discussion about the differences in characterization factors for the CURP and UCrads methods is located in Supplemental 1 Section B; note that the specific differences in characterization factors between UCrads and CURP cannot easily be teased out due to the variety of differences in the methodologies. These differences include inclusion (or not) of a hold time, different compartments and exposure routes considered (UCrads added groundwater and marine sediments; CURP considers the water immersion exposure route), calculation methodology for food chain concentrations, different bioconcentration/transfer factors for some radionuclides (to match the original source, but following the same source hierarchy), different dose coefficients, and inclusion (or exclusion) of the global population for impact assessment. Additionally, CURP can include progeny in the impact assessment calculation methodology. At first glance, one may expect CURP to always produce higher

characterization factors than UCrads due to the inclusion of progeny as well as the integration over time. However, due to differences in methodology, this is interestingly not the case. For example, inclusion of a hold time resulted in a large reduction in the collective ingestion rate following an atmospheric emission in CURP (e.g., ^{123}I , ^{131}I , and ^{133}I were all reduced by ~ 6 orders of magnitude with a hold applied and ^{82}Br , ^{197}Hg , $^{197\text{m}}\text{Hg}$, ^{132}I , and ^{135}I were all reduced by ~ 5 orders of magnitude).

A subset of ten of the total 113 radionuclides characterized (Fig. 1) shows how drastically the characterization factor can decrease over time, but also that some long-lived radionuclides, e.g., ^{238}U , are fairly constant through even the 10,000-year time horizon. The characterization factors for all radionuclides can be found in Supplemental 2 Table 7b. The decrease in the characterization factor through time stems from a combination of the radionuclide decaying away (i.e., the overall midpoint dose factor increasing slowly, or not at all, as the time horizon increases) and the midpoint dose factor of the reference radionuclide (i.e., ^{235}U) increasing as time increases due to its long half-life. While the characterization factors are used in LCIA to determine the impact for each impact category, a figure depicting the change in midpoint dose factor with respect to time for each radionuclide can be found in Supplemental 1 Section C.

A large negative change in characterization factor with respect to time (i.e., slope in Fig. 1) indicates that the midpoint dose factor is no longer increasing with time (i.e., little, or no, additional collective effective dose is occurring with

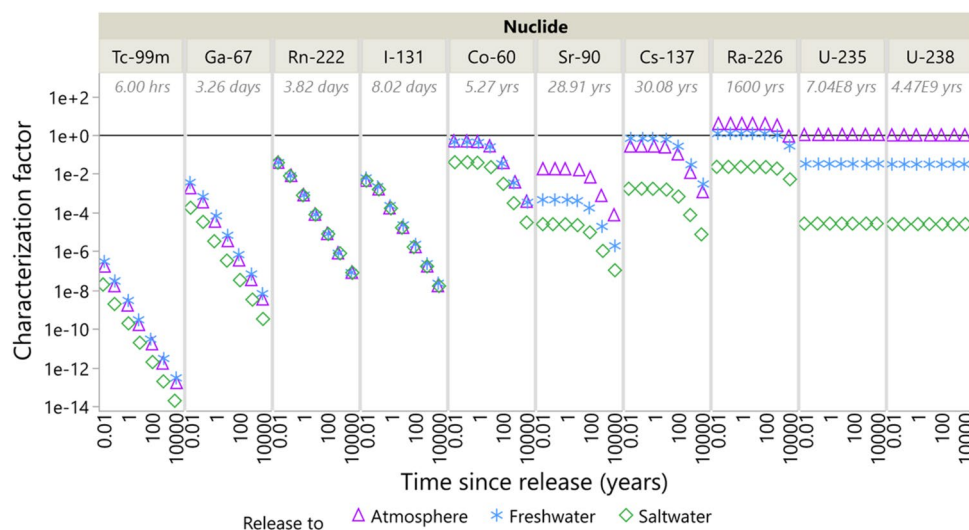


Fig. 1 This figure depicts the characterization factors $[\text{BqU-235 air equivalents} / \text{Bq}_{\text{radionuclide}}]$ for ^{60}Co , ^{137}Cs , ^{67}Ga , ^{131}I , ^{226}Ra , ^{222}Rn , ^{90}Sr , $^{99\text{m}}\text{Tc}$, ^{235}U , and ^{238}U over the seven time horizons analyzed (0.01, 0.1, 1, 10, 100, 1000, and 10,000 years after release). Note that the characterization factors for all time horizons are shown; however, for convenience of the viewer, only every other time hori-

zon is listed. Releases to the atmosphere, freshwater, and saltwater are depicted as purple triangle, blue asterisks, and green diamonds, respectively. The half-life for each radionuclide is listed in gray and italicized under each isotope, and they are ordered by increasing half-life

increasing time, meaning that the radionuclide has decayed away and/or that the collective effective dose of the reference radionuclide is growing faster than the radionuclide of interest), whereas a slight negative slope, or even horizontal line, indicates that the midpoint dose factor is still increasing with time. For short-lived radionuclides (i.e. ^{99m}Tc ($t_{1/2} = 6.0$ h), ^{67}Ga ($t_{1/2} = 3.26$ day), ^{222}Rn ($t_{1/2} = 3.82$ days), and ^{131}I ($t_{1/2} = 8.03$ days)), there is a steep negative slope for even the shortest time step from 0.01 to 0.1 years (Fig. 1). This makes sense because the half-lives of all four of these radionuclides are on the same order of magnitude (or smaller) as the smallest time horizon considered in CURP (i.e., 0.01 years or 3.65 days) and significant decay of the radionuclide has occurred. For ^{60}Co ($t_{1/2} = 5.27$ years), the initial slope is less negative than the slope beginning between the 1-year and 10-year time horizons. By this time step (i.e., between 1 and 10 years), over one full ^{60}Co half-life has occurred signifying that half of the radioactive material that was released has decayed; it is therefore expected that the midpoint dose factor will begin to increase more slowly than at the early time horizons. Characterization factors for ^{90}Sr and ^{137}Cs do not begin to noticeably drop (i.e., large negative slope) until 100 years as they each have a half-life of about 30 years. Radium-226 (^{226}Ra) is not largely affected by the time horizon until the time step between 1000 and 10,000 years, as the half-life is 1600 years. The characterization factor at each time horizon is calculated with respect to the $^{235}\text{U}_{\text{air}}$ calculated for that same time horizon. Thus, the characterization factor is constant for ^{235}U for releases to air, freshwater, and saltwater. Due to the significant half-life, characterization factors for ^{238}U have negligible change over the time horizons considered in CURP. While it is important to assess the characterization factors at different time horizons to understand the influence of radiological decay, this decay leads to an in-growth of other nuclides (some are stable and some are radioactive). The impacts of the radioactive progeny need to be included as well.

3.2 Inclusion of progeny

Including the progeny in the impact assessment is crucial for certain radionuclides, particularly radionuclides wherein the first progeny has a half-life that is shorter than the time horizon. However, inclusion of progeny is insignificant for other radionuclides like those that do not have radioactive progeny or those, like ^{99m}Tc , in which the half-life of the first (and in this case, only) progeny is orders of magnitude larger than the time horizon yielding little additional impact beyond that of the parent.

To understand how the inclusion of progeny influences the ionizing radiation potential, midpoint dose factors need to be considered rather than characterization factors (Supplemental 2 Table 11). If characterization factors were

considered, then the influence of the ^{235}U decay chain could mask the changes in the midpoint dose factor for other radionuclides. For consistency, the same subset of radionuclides from Fig. 1 was considered for progeny inclusion 0.01 years after release (Fig. 2a). A few things to note about these data are that (1) ^{60}Co and ^{67}Ga are not included as neither radionuclide has radioactive progeny and (2) only ^{222}Rn has an impact noticeably above the midpoint dose factor of the parent, albeit slight which is why the percent difference in midpoint dose factors with and without progeny is shown (Fig. 2b).

When the decay chain of ^{222}Rn is included in the midpoint dose factor, the midpoint dose factor is noticeably larger than the midpoint dose factor for just the parent (Fig. 2). In fact, the percent difference (i.e., the percent difference when comparing the midpoint dose factor of the parent with the midpoint dose factor of the parent plus all progeny) for ^{222}Rn is ~53% for atmospheric, freshwater, and saltwater emissions (Fig. 2b). While it may appear that no radionuclides aside from ^{222}Rn have a contribution from their progeny when comparing the absolute difference in midpoint dose factors (Fig. 2a), percent differences in midpoint dose factors reveal that is not the case, particularly for ^{90}Sr (Fig. 2b). Overall, there is an increase in the midpoint dose factor when the progeny is considered for all eight radionuclides ranging from $8.8 \times 10^{-9}\%$ (^{99m}Tc into saltwater) to 53.1% (^{222}Rn into freshwater).

3.3 Time horizons with progeny

The influence of the progeny on the midpoint dose factor depends on the steady-state concentration in each environmental compartment, the half-lives associated with the nuclides in the decay chain, and the time horizon (Supplemental 2 Table 11). To visualize the influence of time horizon on the decay chain, figures analogous to Fig. 2a and b were required (Fig. 3a and b). Three radionuclides (^{226}Ra , ^{222}Rn , and ^{235}U) have a visible contribution from the progeny, especially with increasing time (Fig. 3a). Not only is the contribution of the progeny noticeable for ^{226}Ra , ^{222}Rn , and ^{235}U , but the progeny contribution increases over time. The percent difference between the midpoint dose factors with and without progeny actually increases between at least one time step for all radionuclides except ^{137}Cs (Fig. 3b).

The percent difference in parent and parent plus progeny for ^{99m}Tc increases by approximately an order of magnitude for each increasing time horizon. This is because ^{99m}Tc ($t_{1/2} = 6$ h) decays to ^{99}Tc ($t_{1/2} = 2.11 \times 10^5$ year) so as time increases, the impact of the progeny (^{99}Tc) becomes more pronounced and dominates the impacts for the decay chain after approximately 9.2, 9.2, and 11.5 days for atmospheric, freshwater, and saltwater emissions. While the percent difference increases by an order of magnitude each time step,

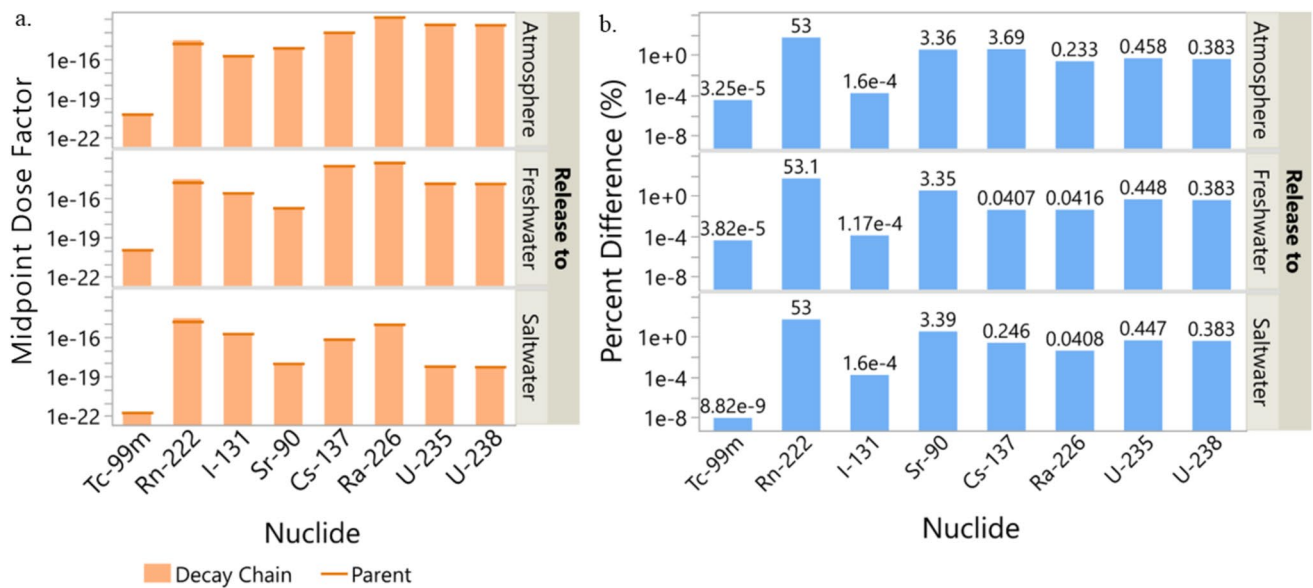


Fig. 2 **a** The midpoint dose factor for the parent plus progeny in the decay chain (light orange bars) and the midpoint dose factor for just the parent radionuclide (dark orange horizontal line) 0.01 years after release and **b** the percent difference between the midpoint dose factors for the parent plus progeny vs. just parent. Note that both figures have the y-axis on a log scale; due to the log axis, **a** does not ade-

quately show the differences between the midpoint dose factors with and without progeny inclusion. Thus, **b** was included to more easily demonstrate the differences, and the quantitative values were included to easily see the implications of progeny inclusion. Note that the radionuclides are ordered in order of increasing half-life

the impacts (i.e., midpoint dose factor) with progeny are still on the order of 10^{-20} to 10^{-23} person.Sv, depending on release compartment.

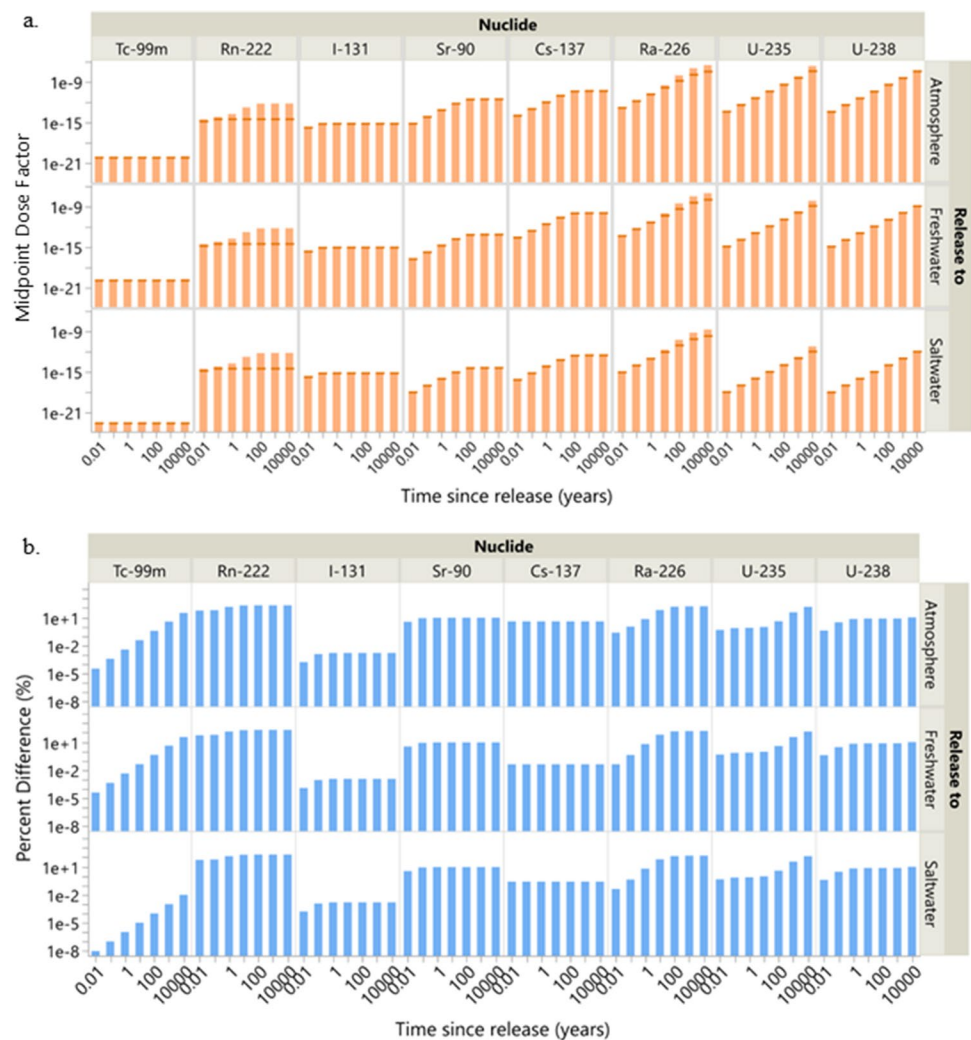
Radon has an increase in the percent difference at shorter time horizons; however, this increase levels out as the time horizon increases. Beginning at 1000 years, the percent difference for all emissions is constant at ~198% for ^{222}Rn because ^{222}Rn and its decay products have all decayed away by that time (the longest half-life in that decay chain is 22.2 years). The three progeny driving the impact between 100 and 1000 years are ^{210}Po ($t_{1/2}=138$ days), ^{210}Pb ($t_{1/2}=22.2$ years), and ^{210}Bi ($t_{1/2}=5$ days), each contributing impacts about an order of magnitude smaller than the previous one. Recall that depending on the specific progeny radionuclide, certain DCs are not published for ingestion, inhalation, or both (Sect. 2.3; Supplemental 1 Section A.3.1), and therefore potential impacts associated with those exposure routes could not be included in the current methodology.

To a lesser extent (i.e., the percent difference is lower), ^{131}I and ^{90}Sr follow ^{222}Rn wherein the earliest time step results in an increase in the percent difference and then that value levels out for remaining time steps (Fig. 3b). The percent difference from ^{131}I is constant at $1.12 \times 10^{-3}\%$ for freshwater emissions and $1.53 \times 10^{-3}\%$ for atmospheric and saltwater emissions starting at the 1-year time horizon. This is unsurprising as the half-life of ^{131}I is 8.02 days and it decays into ^{131m}Xe ($t_{1/2}=11.9$ days) before decaying to stable ^{131}Xe . Xenon-131m (^{131m}Xe) is

present for approximately 120 days (i.e., ~10 half-lives) which is between 0.1 and 1 year. Strontium-90 (^{90}Sr ; $t_{1/2}=28.91$ years) decays to ^{90}Y ($t_{1/2}=64.05$ h) and eventually stable ^{90}Zr . Secular equilibrium (i.e., when every member in the decay chain has the same activity; see Supplemental 1 Section A.4 for more information) is reached in about 19 days and the percent difference between midpoint dose factors increased until 1000 years. The decay chain is only present in appreciable quantities for ~10 half-lives of the longest-lived radionuclide in the chain; in this case, 300 years. The overall percent difference between midpoint dose factors for ^{90}Sr progeny inclusion was 9.23, 9.20, and 9.30% for atmospheric, freshwater, and saltwater emissions, respectively, beginning at the 1000-year time horizon.

The impacts of progeny appear unrelated to time horizon for ^{137}Cs . This is because ^{137m}Ba enters secular equilibrium with ^{137}Cs after approximately 20 min, which occurs before the first time horizon is assessed. The percent difference for the parent plus progeny does not increase with time for ^{137}Cs (Fig. 3b) because for a release to all three environmental compartments, the collective effective dose due to ^{137m}Ba ($t_{1/2}=2.56$ min) is always at least one order of magnitude smaller than the collective effective dose due to ^{137}Cs ($t_{1/2}=30.08$ years). This demonstrates that the addition of impacts from ^{137m}Ba “born” in the environment would not be substantial. However, recall that there were no ingestion or inhalation DCs reported for ^{137m}Ba . Note that ^{137m}Ba “born”

Fig. 3 **a** The midpoint dose factor for the parent plus progeny in the decay chain (light orange bars) and the midpoint dose factor for just the parent radionuclide (dark orange horizontal line) and **b** the percent difference between the midpoint dose factors for the parent plus progeny vs. just parent. All seven time horizons are depicted on the x-axis of both graphs, but only 0.01, 1, 100, and 10,000 are shown for ease of viewing



in the human body is already accounted for as the DCs for ingestion and inhalation are integrated over 50 years.

The percent difference for ^{226}Ra does not reach a plateau during the 10,000 years considered, but the change in percent difference between increasing time horizons slows down at longer time horizons (Fig. 3b). The two progeny driving this impact between 1000 and 10,000 years are ^{210}Po followed by ^{210}Pb . The same note regarding DCs from ^{222}Rn applies here, as well. The ^{210}Po and ^{210}Pb are two of the five progeny with both ingestion and inhalation DCs reported.

The percent difference in midpoint dose factors for ^{235}U continues to increase across all time horizons considered, with more pronounced increases at later time horizons (Fig. 3b). The dominant impact contributors at these later time horizons are ^{227}Ac ($t_{1/2} = 21.77$ years) and ^{231}Pa ($t_{1/2} = 3.2 \times 10^4$ years). The small change in midpoint dose factors with increasing time steps seen between 0.01 and 10 years is because the impact is driven by ^{235}U , which has a long half-life ($t_{1/2} = 7.04 \times 10^8$ years). Around 2510 years (for air, freshwater, and saltwater releases), ^{227}Ac surpasses

as the dominant impact contributor to the decay chain and at around 3980 years after an atmospheric release, ^{231}Pa becomes the radionuclide with the second highest impact contribution for atmospheric releases. Note that this is one of the limitations with the time steps used as there is only one time horizon (3160 years) between 2510 and 3980 years. Thus, the percent difference for the midpoint dose factors begins increasing since the half-lives of ^{227}Ac and ^{231}Pa are shorter than ^{235}U . As discussed in Supplemental 1 Section A, there are select progeny that do not have DCs for some, or all, pathways.

Uranium-238 (^{238}U ; $t_{1/2} = 4.47 \times 10^9$ years) has a larger increase in midpoint dose factors with earlier time steps than later ones (Fig. 3b). In the final time step, the percent difference in midpoint dose factors increases again. To elaborate, the relative change in percent difference is 2.51 between 0.01 and 0.1 years, 0.62 between 1 and 10 years, and 2.49 between 1000 and 10,000 years for an atmospheric release. This is likely a result of the different half-lives associated with the progeny for the ^{238}U

decay chain. The first progeny is ^{234}Th ($t_{1/2} = 24$ days) which then decays into ^{234}Pa ($t_{1/2} = 6.7$ h) which decays to ^{234}U ($t_{1/2} = 2.45 \times 10^5$ years), etc. This suggests that the in-growth and decay of ^{234}Th and ^{234}Pa drive the initial change in the percent difference whereas the long half-life of ^{234}U causes the change in percent difference to slow between later time steps before measurable radioactive decay occurs and the percent difference in midpoint dose factors begins increasing again.

The above discussion regarding the eight radionuclides shows how different each radionuclide behaves with increasing time due to progeny in-growth as well as the half-lives, emission type(s), and emission energies of the parent and progeny radionuclides. During the 10,000 years considered in CURP, ^{222}Rn , ^{131}I , and ^{90}Sr all had increasing percent differences for short time horizons that decreased or ceased for later time horizons. The percent difference for ^{137}Cs remains fairly constant for all time horizons due to the progeny contributing significantly smaller impacts than the parent. Ultimately, ^{222}Rn , ^{131}I , ^{137}Cs , ^{90}Sr and their associated progeny completely decayed during the 10,000 years (Fig. 3a) because the longest-lived radionuclide in each decay chain is 22.2 years, 11.9 days, 30.08 years, 28.91 years, respectively. Additionally, this data demonstrates that the percent difference (Fig. 3b) will stop increasing once 10 times the half-life of the longest-lived progeny is reached. This is 22.2 years, 11.9 days, 2.56 min, and 64.05 h for ^{222}Rn , ^{131}I , ^{137}Cs , and ^{90}Sr , respectively. The midpoint dose factor begins to level out between 100 and 1000 years for ^{222}Rn , between 0.1 and 1 year for ^{131}I , and between 10 and 100 years for ^{90}Sr . The percent difference in midpoint dose factors for ^{137}Cs is constant for all time horizons considered in CURP.

The four radionuclides with long half-lives or long-lived progeny (i.e., $10 \times$ the half-life is longer than the time horizon considered) tell a similar story. The half-life of $^{99\text{m}}\text{Tc}$ is shorter than the smallest time horizon assessed, so the midpoint dose factor does not change with increasing time (Fig. 3a). However, ^{99}Tc has a very long half-life and consequently the percent difference will continue increasing for $\sim 2 \times 10^6$ years (i.e., $10 \times$ the half-life of ^{99}Tc). Radium-226 is the longest-lived radionuclide in its decay chain so Fig. 3a and b will continue increasing for approximately 16,000 years ($10 \times$ the half-life), though the change in percent difference for a consistent time interval will remain the same as the decay chain will be in secular equilibrium by 155.4 years (i.e., 7 half-lives of the longest-lived progeny). The two uranium isotopes have very long half-lives, so like ^{226}Rn , neither the midpoint dose factor nor percent difference in midpoint dose factors with and without progeny are anticipated to stop increasing for many years.

4 Considerations and implications

Transparency in the dissemination of research ensures that the research findings are accessible and usable to scientists across disciplinary bounds (Liu et al. 2023). For example, to clarify the implementation of the ionizing radiation potential in LCIA, CURP utilized specific equations. The intent is that direct reporting of the specific equations used to compute the ionizing radiation potential impacts will enable other researchers to more easily utilize, or even expand, the CURP methodology.

Additionally, a self-identified limitation to previously established methodologies was the exclusion of progeny in the impact assessment. As shown herein, the time horizon is very important, with and without progeny inclusion. Without progeny inclusion, the shorter lived the parent radionuclide is, the more negative the slope (i.e., Fig. 1) will be, resulting in overall smaller characterization factors further in the future. For long-lived progeny, the time horizon is important to the midpoint dose factor; the percent difference in midpoint dose factors will increase as the time horizon increases (up to about $10 \times$ the half-life of the longest lived radionuclide). This means that using a methodology for the fate calculation that assumes steady state is likely unsuitable for the ionizing radiation potential LCIA category as (a) significant radiological decay or (b) progeny in-growth and subsequent decay will take place during the time to reach equilibrium thus changing the environmental and foodstuffs concentrations and potentially the radionuclide needing to be assessed from the inventory (i.e., the radionuclide needing to be assessed might not be the radionuclide in the inventory).

Depending on the radionuclides released by a given product or service, the reasonableness of the steady-state assumption will change and care will need to be taken to identify the radionuclides (i.e., parent or progeny) affecting the systems. For example, when comparing coal combustion electricity production vs. nuclear fission or handling medical waste, the radionuclides needing to be compared have different implications. Radon-222 is emitted to air during mining and milling of uranium and has a smaller midpoint dose factor than ^{238}U , which is released during coal combustion along with its progeny. As discussed above, the midpoint dose factor for ^{222}Rn reaches a plateau for longer time horizons whereas the midpoint dose factor for ^{238}U continues to increase at later time horizons (Fig. 3a). Thus, results should be compared for various time horizons while recognizing that the time of assessment is very important with and without progeny. Technetium-99m is commonly used in medicine for diagnostic imaging and is excreted in urine and feces (Kane et al. 2024). The half-life is short ($t_{1/2} = 6$ h), but its

progeny is long lived (recall, ^{99}Tc , $t_{1/2} = 2.11 \times 10^5$ years). As shown in Fig. 3b, the percent difference in midpoint dose factors continues to increase throughout all time horizons considered in CURP. This illustrates the recognized need for careful disposal for radionuclides that have a short half-life that decay into a nuclide with a very long half-life.

To illustrate how small the IRP is in the grand scheme of things, the entire collective (i.e., entire population) dose is “assigned” to a single person and we consider how large a radioactive release would need to be for that person to reach the individual public dose limit of 1 mSv (ICRP 2007). So, in essence, one person receives a dose that would otherwise be distributed among 10^8 to 10^9 people and still be below the regulatory dose limits. Figure 4 shows the activity that would need to be emitted for various radionuclides that would result in reaching 1 mSv based on the midpoint dose factor. To emphasize, this is a very conservative assumption wherein an individual is assumed to receive the entire collective effective dose for the year rather than their individual effective dose. To calculate the activity needed to reach the public dose limit, 1 mSv was divided by the midpoint dose factor 1 year after release (without progeny inclusion) to determine the necessary initial release needed to result in that effective dose, assuming that only one radionuclide was released. Note that the activity needed to be emitted for each of the 113 radionuclides (including those in Fig. 4) to result in a collective effective dose (assigned to an individual for illustrative purposes) of 1 mSv is listed in Supplemental 2 Table 12.

As seen in Fig. 4, the activity released needs to be higher (or the same) for saltwater releases than for atmospheric or freshwater emissions. This is likely due to dilution and reduced exposure times for saltwater than the other two

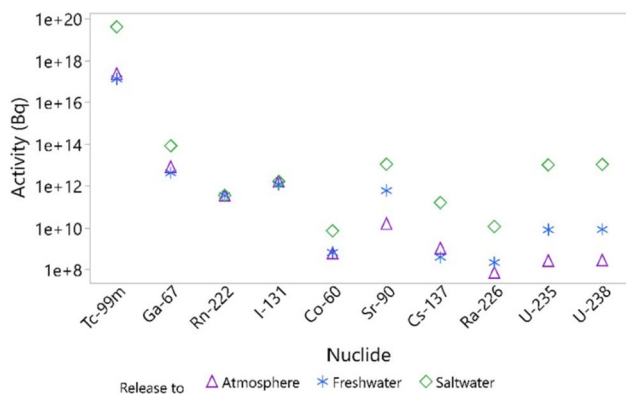


Fig. 4 Activity that would need to be emitted for each radionuclide to cause an individual (very conservatively assumed to take the entire collective dose in one year) to reach their annual dose limit of 1 mSv. The purple triangles represent releases to the atmosphere, the blue asterisks represent releases to freshwater, and the green diamonds represent releases to saltwater

environmental compartments. Of the subset of radionuclides considered in Fig. 4, ^{226}Ra has the smallest amounts of radioactivity that would need to be emitted to reach the 1 mSv public dose limit at 6.74×10^7 and 2.11×10^8 Bq (released via the atmosphere and freshwater, respectively) followed by ^{235}U (atmosphere) at 2.5×10^8 Bq. Overall, of the 113 radionuclides characterized, ^{129}I would need the smallest quantity of radioactivity to be emitted to reach the 1 mSv public dose limit at 2.69×10^4 , 2.69×10^4 , and 2.71×10^4 Bq for atmospheric, freshwater, and saltwater emissions, respectively. The next lowest activity needed was for ^{126}Sn released to saltwater (5.30×10^6 Bq). The radionuclides requiring the largest amount of radioactivity to be emitted to reach the public dose limit were saltwater releases of ^{212}Bi , $^{99\text{m}}\text{Tc}$, and $^{87\text{m}}\text{Sr}$ at 8.63×10^{19} , 4.01×10^{19} , and 3.22×10^{19} Bq, respectively. Recall that this is the activity that one individual would need to be exposed to in order to reach their annual dose limit of 1 mSv with the very conservative assumption that they receive the collective effective dose for the year rather than the individual effective dose.

To put these conservative activity estimates (i.e., from 2.69×10^4 Bq of ^{129}I to air to 8.63×10^{19} of ^{212}Bi to saltwater) in perspective, we offer comparisons with both the activity from coal combustion residuals (CCRs), as well as the activity from annual releases from a nuclear power plant. The equivalent activity from CCRs based on an average ^{210}Pb concentration of 248 Bq/kg (Lauer et al. 2015) is ~ 95 kg and 3.04×10^{17} kg, respectively. Note that additional radionuclides beyond ^{210}Pb contribute to the total activity of the CCRs, so the 95 kg minimum is a conservative estimate. Globally, over 1.22×10^{12} kg of CCRs were generated in 2016 (Harris et al. 2019).

An additional perspective can be to consider the annual releases from a nuclear power plant. During 2023, the three nuclear reactors at Oconee Nuclear Station released 3.70×10^{12} Bq (99.9 Ci) of fission and activation gasses, 2.95×10^{12} Bq (79.8 Ci) of tritium (^3H), and 8.84×10^{11} Bq (23.9 Ci) of ^{14}C to air effluents as well as 1.56×10^8 Bq (0.00421 Ci) of fission and activation products and 4.99×10^{13} Bq (1350 Ci) of ^3H to liquid effluents (Duke Energy 2024). This is a total emission of 7.53×10^{12} and 4.99×10^{13} Bq to atmospheric and liquid effluents, respectively. The maximum total body dose for a child one mile from the Oconee Nuclear Station was calculated to be 2.96×10^{-3} mSv (0.295 mrem; Duke Energy 2024), which is significantly lower than the individual annual dose limit. This is three orders of magnitude smaller than the individual annual dose limit, even with total emissions on the order of 10^{13} Bq. Thus, this very conservative assumption (i.e., that one person takes the entire collective effective dose) illustrates how small the endpoint impact of human health from ionizing radiation emissions could be as calculated from the IRP methodology. To further illustrate this, consider if the

average individual effective dose was used instead of the collective effective dose to determine the activity that would need to be emitted to result in an effective dose of 1 mSv. The midpoint dose factor for ^{129}I (the radionuclide that required the smallest emission of activity to reach 1 mSv) was divided by the global population: 5.76×10^9 people. Note that this is not a “true” value for individual dose as calculations were done on the urban, continental, and global scales. The resulting average individual effective dose would be 6.45×10^{-18} Sv/Bq for ^{129}I . Thus, 1.55×10^{14} Bq would need to be emitted for an individual to reach their annual dose limit of 1 mSv. Due to the large activities that would need to be emitted to exceed regulatory standards (intended to protect human health and the environment) small radioactive emissions are anticipated to have negligible impact in these areas.

Ultimately, assuming steady state for the fate assessment is consistent with other LCIA methodologies; however, moving forward, care should be taken when using an IRP methodology based on fate assessments that do not begin until steady state is reached (as briefly mentioned above). The analysis of time horizons and progeny impact demonstrates how a steady-state assumption may affect the impact assessment results. For example, an atmospheric release of ^{131}I would take 1.30×10^5 years to reach steady state, but the decay chain is completely decayed between 0.1 and 1 years. This means that while this radionuclide is actually decaying, there could be a larger concentration in the release compartment and smaller concentrations in the other environmental compartments than what is modeled. This would be particularly important to consider for an atmospheric release due to the pathways of concern (i.e., inhalation and grass-cow-milk). An atmospheric release of ^{235}U would take 2.08×10^5 years to reach steady state but recall that after about 2500 years, the progeny, ^{227}Ac becomes the largest impact contributor, meaning that with the steady-state assumption, the inventory of radionuclides being assessed is likely inaccurate. Ultimately, the steady-state assumption can lead to an unrealistic ionizing radiation potential calculation; this needs to be kept in mind when assessing radiological releases, and future work should focus on how (if possible) to create a site-independent, non-steady-state fate module. When interpreting the results of an IRP methodology conducted based on a steady-state assumption, we recommend using this LCIA category exclusively for comparative purposes.

To the author’s knowledge, this is the first research to propose a method for including progeny in the IRP impact methodology. In fact, multiple sets of characterization factors were presented to handle the idea of “nontrivial” doses: conservative, most conservative, and least conservative. Due to the use of USEtox steady-state concentrations coupled with radiological risk assessment equations, this

methodology still allows for comparison between chemical and radiological emissions while seeking to support a broader application and expanded influence of LCA studies. CURP seeks to (1) demonstrate a practical method for inclusion of progeny, (2) illustrate that progeny influence can be meaningful, and (3) offer a more transparent method, ideally to support convenient adaptation in the future.

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Data availability Three supplemental documents are included with this work. The first is a Microsoft Word document wherein methods and comparison to UCrad are elaborated on; the second file is an Excel document containing clean tables of data referred to in the article; the third file is an Excel document containing calculations used in this work.

Declarations

Conflict of interest The authors declare no competing interests.

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