Radiological Impacts in Life Cycle Assessment. Part II: Comparison of Methodologies

Abstract

In a complementary article, an overarching framework was proposed to include radiological impacts in Life Cycle Impact Assessment (LCIA). Two methodologies were derived embodying the framework: the Critical Group Methodology (CGM), adapted from the approach commonly used in Human and Environmental Risk Assessment (HERA), and UCrad, based on the compartment modelling approach commonly used in LCIA. In this paper, characterisation factors obtained by the two methodologies are compared in detail to investigate the consequences of the different approaches to fate modelling and the sensitivity of the characterisation factors to the radionuclides' half-life. Characterisation factors from the CGM methodology are strongly affected by radioactive decay at low half-life and by dilution at large distances. Conversely, UCrad factors are not affected by dilution and are affected less than CGM by radioactive decay. It is concluded that UCrad is more appropriate than CGM for LCA because it is consistent with the general approach used in LCIA. However, CGM can be used alongside UCrad to make recommendations on the location and scale of specific processes emitting radionuclides.

Keywords

Life Cycle Impact Assessment; Ionising radiations; Radiological Impacts; Risk assessment; Radionuclides; Nuclear Waste

1 Introduction

Life Cycle Assessment (LCA) aspires to provide a complete analysis of the environmental impacts of delivering products and services. The impacts are usually estimated as contributions to a recognised set of impact categories. However, in the absence of a generally accepted approach to assessment, radiological impacts have usually been omitted. To correct this omission, Paulillo (2018) and Paulillo and colleagues (2019a) proposed a general framework for inclusion of radiological impacts in Life Cycle Impact Assessment (LCIA), leading to two methodologies to model the fate of radioisotopes and the resultant impacts: UCrad and Critical Group Methodology (CGM). The two methodologies respectively represent two different cultures: Life Cycle Assessment (LCA) and Human and Environmental Risk Assessment (HERA). LCA assesses average potential impacts of a very large number of pollutants released over the life cycle delivering a product or service, calculated per functional unit of the product or service. As the life cycle may be spread over many locations around the world, LCIA is usually not site-specific and is concerned with global rather than regional scales. UCrad adopts this approach. By contrast, HERA quantifies the actual, absolute risks to humans and the environment associated with the release of pollutants from a specific process whose location is defined. The assessment is carried out on a regional scale, usually with the focus on a few selected substances of relevance to the process under study. The Critical Group Methodology (CGM) follows the general approach used in HERA. As noted by a number of authors (Nishioka et al., 2002; Saouter and Feijtel., 2000; Sonnemann et al., 2004), LCA and HERA may lead to different results when applied to the same process. Udo de Haes et al. (2006) concluded that, although the two tools differ in a number of respects, the fundamental difference lies in the use of the functional unit, which differentiates the relative impacts of LCIA from the absolute ones of HERA. Further differences, similarities and application of these tools have been discussed by Cowell et al. (2002), Olsen et al. (2001), Owens (1997), Udo de Haes et al. (2006), Wegener Sleeswijk et al. (2003) and Wrisberg et al. (2002) amongst others.

The general framework and the two methodologies are presented in a complementary paper (Paulillo et al., 2019a). The objective of the present paper is to provide a detailed and quantitative comparison between the results of the UCrad and CGM approaches, to relate the differences between them to the underlying assumptions in fate modelling. The consequences of implementing a risk assessment-based methodology within the LCA framework are examined by comparing results from CGM quantitatively with those from the approach commonly used in LCA, represented by UCrad. Finally an approach combining both methodologies is proposed to meet the need, identified by some authors (e.g. Flemström et al., 2004; Udo de Haes et al., 2006), for deeper integration between LCA and HERA.

The article begins with an overview of the main features and differences between the fate models used in UCrad and CGM (Section 2). Section 3 presents a detailed comparison between the characterisation factors obtained from the two methodologies. Section 4 discusses the significance of the comparisons, in particular on the significance of radionuclide half-life, and proposes a way to combine the two approaches. Finally, the main findings of the article are summarised in Section 5.

2 Overview of fate module of CGM and UCrad

Paulillo (2018) and Paulillo et al. (2019a) set out the overarching framework within which the two methodologies for radiological impact assessment have been developed; they differ solely in the transport/dispersion models employed in the fate analysis. The focus here is on how these different approaches and the radionuclides' half-lives affect the resulting characterisation factors. This section gives an overview of the main features and differences between the fate modules of the two methodologies.

The fate module in UCrad follows the widely-used multimedia fugacity approach developed by Mackay (2001) to predict the distribution of emitted species between different environmental media. Mackay models are not new to LCIA: they are commonly used to assess the impact of toxic pollutants, USEtox being a notable example (Rosenbaum et al., 2008). In Mackay models, each medium is represented as

68 a homogenously mixed compartment, while exchange between compartments occurs by advection
69 and diffusion. The fate module in UCrad uses a nested compartmental model comprising two spatial
70 scales: continental and global. The resulting environmental concentrations represent steady-state
71 conditions. In principle, UCrad is able to consider emissions to all compartments at both scales.
72 However, for the purpose of comparing the two methodologies, this article considers direct emissions
73 from routine operations into three compartments: air, fresh and seawater. Results for emissions from
74 a Geological Disposal Facility (GDF) into groundwater are included in the Supporting Information.

75 The Critical Group Methodology (CGM) is conceptually different: it sets out to estimate the impact of
76 emissions on a specific group of humans, referred to as the "Critical Group". The concept of "Critical
77 Group" was developed for Risk Assessment; it represents the group of people expected to receive "the
78 highest dose due to their lifestyle, location and habits" (ICRP, 1990; NRPB, 1993). The critical group
79 does not necessarily represent a worst-case scenario; rather, it allows selection of the most
80 appropriate distance at which to assess radiological impacts. As explained by Paulillo and co-workers
81 (2019a), the fate module of CGM relies on two earlier studies with different purposes: models
82 developed by the International Atomic Energy Agency (IAEA, 2001) are used to describe routine direct
83 discharges from processing operations, whilst the Post-Closure Safety Assessment (PCSA) (NDA, 2010),
84 developed by Radioactive Waste Management Ltd. (RWM), is employed for long-term unplanned
85 emissions from a GDF. For direct discharges, characterisation factors were developed for emissions to
86 air, river and seawater. The behaviour of species discharged is described by analytical models and the
87 resulting predicted concentrations depend on a number of site-dependent parameters such as height
88 of atmospheric emission, width of river, etc. With the purpose of making the methodology generally
89 applicable, Paulillo et al. (2019a) based the CGM model on generic values for all parameters except
90 one: the distance of the critical group from the emission source. The significance of this parameter is
91 explored in detail in the present paper. The transport model used by the PCSA is similar to those used
92 by the IAEA, but is applied to a different medium: groundwater. The model however is not configured

to produce distance-dependent characterisation factors; therefore, comparison of characterisation factors for emissions from a GDF is not pursued in significant detail in this article.

3 Results: Comparison of methodologies

This Section presents a detailed comparison between characterisation factors calculated by the CGM and UCrad methodologies presented by Paulillo and colleagues (2019a), for continuous emissions to three receiving environmental media (air, freshwater and seawater) and for four distances from source to receptor modelled using the CGM approach. The comparison for emissions from a GDF is included in the Supporting Information, but not discussed in detail because the CGM is not set up to produce distance-dependent characterisation factors for these emissions.

The differences between the two sets of characterisation factors are expressed in terms of logarithmic deviations, i.e. the logarithms to base 10 of the ratio between the factors, which indicates order-of-magnitude discrepancies (see Paulillo et.al., 2019a). The Mean Log Deviation (MLD), quantifies the average order-of-magnitude discrepancy between factors: a nil MLD indicates general agreement between the methodologies; a negative MLD indicates that the reference values are generally larger than the set being compared against them; and a positive MLD indicates the converse relationship.

Figure 1 reports Mean Log Deviations for characterisation factors obtained from the CGM model compared to values given by UCrad, as functions of the half-lives of radionuclides. Numerical values of characterisation factors and log deviations are reported in Paulillo et al. (2019b). MLD values for each set of characterisation factors are reported in Table 1 for all values of half-life and for values higher than 7e+07 s (i.e. 2.2 years), approximately equal to a value of the decay constant of 1e-09 s^{-1}. This time is comparable to the slowest environmental transport covered by the model: transport over 10,000 km in seawater.

116 **Table 1 - Mean log deviations of CGM characterisation factors relative to UCrad for different receiving media**
117 **and distances of the critical group.**

DISTANCE	MEAN LOG DEVIATION	
	All values of half-life	Half-life > 7e+07 s
Air emissions		
1 km	5.7E+00	5.2E+00
100 km	2.1E+00	2.0E+00
1 000 km	-2.4E+00	1.8E-01
10 000 km	-1.2E+01	-1.6E+00
Freshwater emissions		
1 km	4.0E+00	2.9E+00
100 km	3.4E+00	2.9E+00
1 000 km	-1.7E+00	2.9E+00
10 000 km	-9.1E+00	2.9E+00
Seawater emissions		
1 km	5.2E+00	3.9E+00
100 km	-2.8E+00	1.7E+00
1 000 km	-9.0E+00	4.8E-01
10 000 km	-2.5E+01	-7.5E-01

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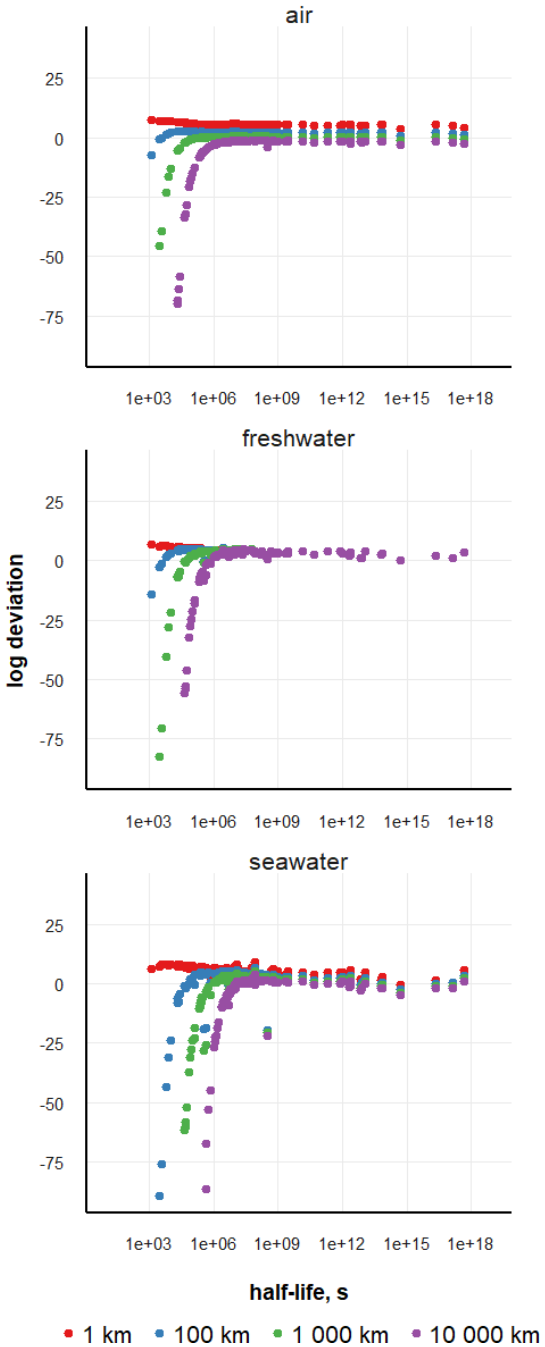
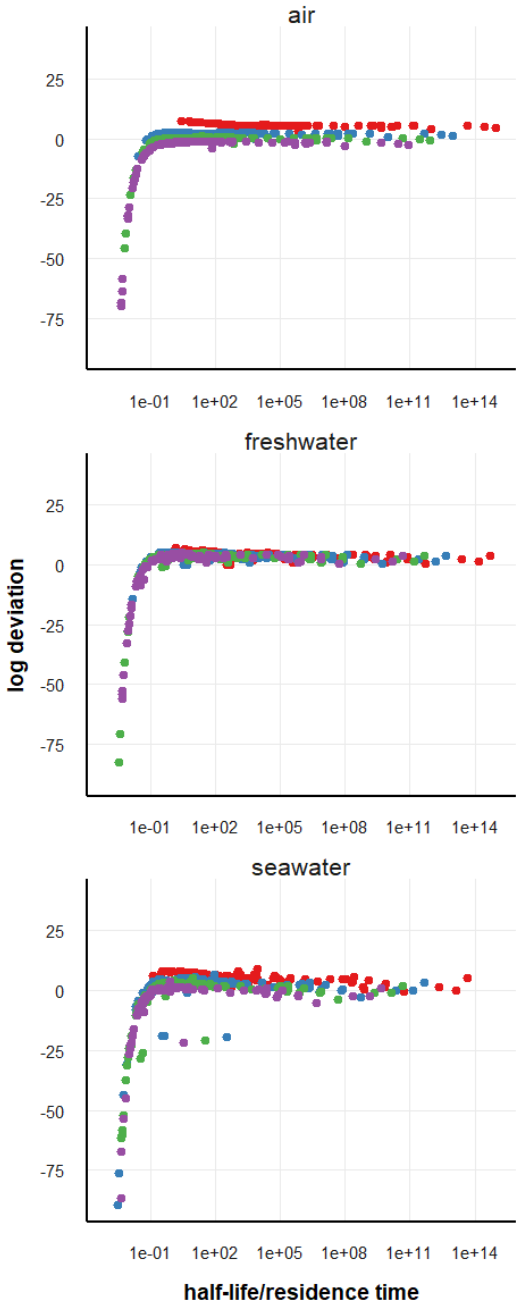


Figure 1 - Log deviations of CGM characterisation factors relative to UCrad as a function of half-life, for air, fresh and seawater emissions and for four distances of the critical group.

Figure 1 reveals two clear regimes that are consistent across all receiving media and distances of the receptor greater than 1 km. For high half-life, log deviations are independent of half-life; at low half-life, log deviations sharply decrease with decreasing half-life. The boundary between these two regimes is given by values of half-life that increase with the distance and differ between environmental media. This is most evident for seawater, where MLD values decrease from ~5 to ~-25 (Table 1). The

trend is weaker for emissions to air and freshwater, with MLD diminishing from ~6 to ~-12 for air and from -4 to ~-9 for freshwater. Thus, the best agreement between the methodologies is found for a distance between 100 and 1,000 km for emissions to air and freshwater, and between 1 and 100 km for emissions to seawater. The two regimes, low and high half-life, are analysed separately in Figure 2 and Figure 3. The results for emissions to seawater in Figure 1 also reveal some outliers; these are most evident in Figure 3 and are discussed below.

Figure 2 shows log deviations between the two models against the ratio of isotope half-life to transit time from source to receptor for each distance from source to receptor and each environmental medium. The transit times are calculated as the ratio of the distance between source and receptor to the speed of the carrier, i.e. wind for emissions to air, river flow for emissions to freshwater and marine current for emissions to seawater; they are reported in Table 2. The curves for low half-life coincide when shown as a function of the ratio of half-life to transit time. The sharp decrease of log deviations starts when half-life is of comparable magnitude to transit time; i.e. the ratio is around unity. This indicates that the distinction between the two regimes depends on whether a significant portion of nuclides decay before reaching the receptor in the CGM model.



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144 **Figure 2 - Log deviations between characterisation factors from CGM and UCrad models as a function of the ratio of half-**
145 **life to transit time, for air, fresh and seawater emissions and for four distances of the critical group.**

146

147 **Table 2 – Transit times for emissions to air, freshwater and seawater in CGM**

	Transit time (s)		
Distance (km)	Air	Freshwater	Seawater
1	5.00E+02	9.26E+02	1.00E+04
100	5.00E+04	9.26E+04	1.00E+06
1 000	5.00E+05	9.26E+05	1.00E+07
10 000	5.00E+06	9.26E+06	1.00E+08

Figure 3 reports log deviations for CGM factors relative to UCrad factors against the distance for values of half-life greater than 7e+07 s; i.e. where nuclide decay between source and receptor is negligible. Log deviations decrease considerably with distance for emissions to air, from an average value of ~5 at 1 km to ~-1 at 10 000 km (Table 1), and for emissions to seawater, from ~4 to ~-0.7, but remain constant for emissions to freshwater, at ~3. For high half-life, the best agreement between UCrad and CGM is found for a distance between 1 000 and 10 000 km for emissions to air and seawater, whilst for emissions to freshwater the methodologies do not converge.

Figure 3 uses colour-coding to rank nuclides from greatest log deviation (red) to least (blue). The ranking does not change with distance for emissions to air and freshwater. For emissions to seawater, the ranking changes between a distance of 1 km and distance greater than 100 km. This behaviour is exemplified by Eu155 and Co60, whose log deviations increase from 1 km (respectively about 3 and 2.4) to 100 km (~4), and by U234 for which the value remains roughly constant at about 0.4. From 100 to 10 000 km, the log deviations of these nuclides decrease in line with those of other nuclides.

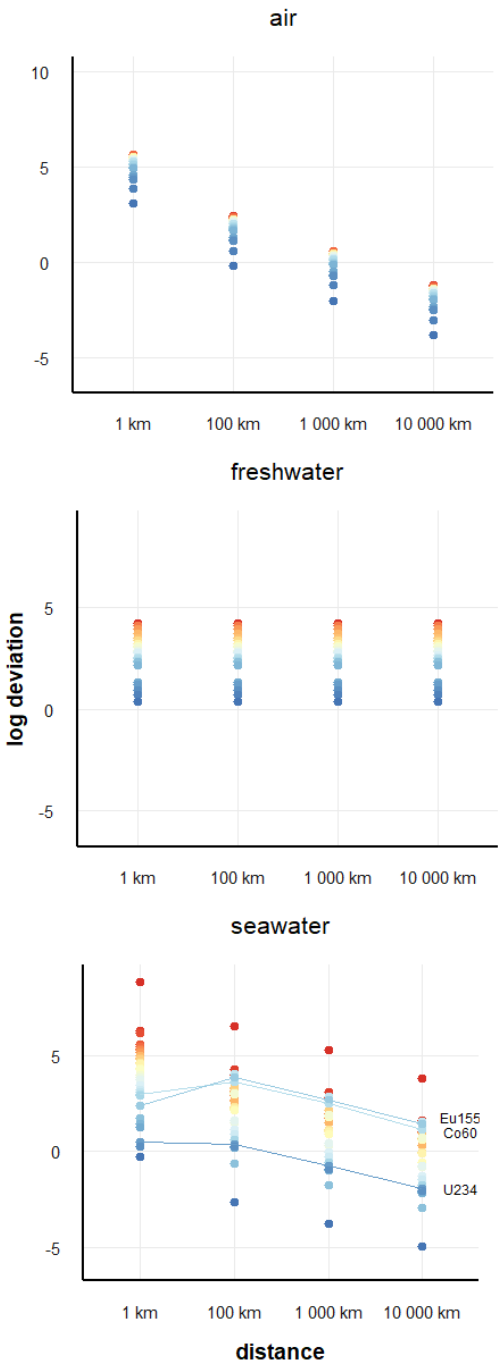

Figure 3 - Log deviations for half-life higher than 7e-07s between characterisation factors of CGM and UCrad as a function of distance for air, fresh and seawater emissions. Colours identify different nuclides, and are ranked from red to blue according to log deviations at 1 km.

4 Discussion

4.1 Comparison between CGM and UCrad

The comparison between the characterisation factors from the CGM and UCrad methodologies for different receptor distance and receiving media gives interesting insight into the two methodologies; notably, it highlights that the deviations between the factors from the two methodologies depend on half-life at low half-life and on the distance from the source of the emission to the receptor at all half-lives.

Figure 1 and Figure 2 highlight the distinct regimes for low and high half-life; the boundary between the regimes does not depend on half-life only (as shown in Figure 1), but rather on the value of half-life relative to transit time (Figure 2). When half-life is lower than the transit time, radioactive decay is faster than dispersion of radionuclides between source and receptor (left side of Figure 2). By contrast, dispersion is faster than decay when half-life is higher than transit time (right side of Figure 2). The transition from one regime to the other occurs when half-life is comparable to the transit time. The ratio between half-life and transit time is analogous to the inverse of the Damköhler number (Fogler, 2006) used in Chemical Reaction Engineering to relate the timescale of the transport phenomena inside a reactor to that of the chemical reaction: when the Damköhler number is less than one, the chemical reaction is slow compared to the transport processes, and thus conversion of reactants into products is low; a higher conversion is achieved when it is greater than one, that is when the chemical reaction is faster than the transport processes.

When half-life is lower than transit time, the CGM factors are strongly dependent on half-life, decreasing sharply with decreasing half-life down to negligible values compared the factors from UCrad. The fate models in CGM use time-dependent analytical models, such as the Gaussian plume for atmospheric emissions, to simulate dispersion from source to receptor. The model allows for the transit time required for nuclides to reach the receptor and therefore includes decay. By contrast,

UCrad employs compartment-type models at steady state conditions. Radioactive decay is independent of time and is modelled as a removal process from each compartment so that the dependence on half-life is less explicit and less strong.

The log deviations for half-life greater than transit time (Figure 3) show that CGM factors are still dependent on the distance between source and receptor for emissions to air and seawater: the higher the distance, the lower are the characterisation factors. This effect is also present in the results for low half-life but is obscured by the stronger dependence on radioactive decay. The analytical models for emissions to air and seawater in CGM account for dispersion and resultant dilution to estimate how the separation distance affects the concentration to which the receptors are exposed; i.e. the radiological doses to the critical group. In contrast to air and seawater, factors for emissions to freshwater remain constant with distance because dilution is not considered for this environmental medium: CGM models freshwater bodies as rivers, so that dispersion is constrained by the river banks.

Because in CGM dilution applies equally to all radionuclides, the ranking of radionuclides with respect to their characterisation factors remains the same at different distances; this is shown in Figure 3 for emissions to air. However, for emissions to seawater the ranking changes from a distance of 1 km to distances greater than 100 km, as highlighted above. The fate model for emissions to seawater in CGM calculates two concentrations: one relevant to predict accumulation of radionuclides in fish, and another to estimate build-up on coastal shorelines. For distances of the receptor that are relatively low compared to the distance between source of emissions and shoreline (e.g. 1 km), the radioactive plume does not reach the shoreline. Changes in the ranking occurs for those nuclides for which the effects of external exposure are more significant than the effects of ingesting fish. This is exemplified by Eu 155, Co60 and U234, for which the characterisation factors increase or remain approximately constant when the distance increases from 1 km to 100 km.

The effect of dilution on the characterisation factors obtained from CGM highlights the most important limitation of the methodology: it assumes that the population affected by emissions is

concentrated at a specific location. For a uniformly distributed population, such as that considered in the Human Health Damages approach (Frischknecht et al., 2000) or in the UNSCEAR methodology (UNSCEAR, 2017), spreading of the plume of pollutant does not affect total collective dose because the reduction in concentration is offset by the increase in the total number of receptors exposed (Dreicer et al., 1995). For a Linear No-threshold dose-response function, the impact is also unaffected: as the plume disperses, the number of receptors affected increases but probability or severity of damage for any individual is proportionately decreased (Spadaro and Rabl, 1999). Therefore, as is routinely assumed in LCIA, the characterisation factors should not be dependent on distance from the point of release. By contrast, it is implicit in UCrad that there is no effect of dilution because a compartment-type fate model already assumes uniform dispersion of radionuclides in each environmental compartment and thus leads to characterisation factors that are not dependent on the location of the target group.

Because the concentrations predicted by CGM depend strongly on half-life and dilution, the distance between source and receptor that gives average agreement (MLD=0) between UCrad and CGM also depends on these parameters. When radioactive decay is negligible (i.e. when half-life is greater than transit time), the methodologies converge. For emissions to air and freshwater (Table 1), this occurs for distance between 100 km and 1 000 km. Because the transit times in seawater are greater than in air and freshwater (Table 2), this convergence is seen for lower distances, between 1 and 100 km. The models do not converge for emissions to freshwater because dilution is not considered (see above).

The comparison between UCrad and CGM for emissions to seawater also highlights three notable outliers - krypton- 85, radon-222, and xenon-133 - whose log deviations different considerably from those of radionuclides with similar half-life (see Figure 2 and Figure 3). Radioactive noble gases differ from other radionuclides because they impart negligible doses through internal pathways like ingestion and inhalation; their main impact pathways are external, for instance through exposure to a radioactive plume (IAEA, 2001; NCRP, 1995). The fate models in UCrad enable radionuclides to

disperse to all environmental media irrespective of the medium to which they are released, whereas in CGM radionuclides primarily remain in the environmental medium to which they are released. By ignoring the transfer of noble gases from seawater to the atmosphere, CGM omits the most significant exposure pathway. This is why factors for CGM are considerably lower than those for UCrad for emissions to seawater.

The analysis described in Section 3 did not include factors for emissions from nuclear waste disposed in a GDF although characterisation factors are available for both methodologies. This is because the CGM methodology is not set up to produce distance-dependent factors for these emissions (Paulillo et al., 2019a). Figure 1 in the Supporting Information compares the UCrad factors with the only set of factors for CGM. MLD for different types of nuclear waste range from ~6 to ~8. They are higher than those for emissions to air, freshwater and seawater at 1 km (Table 1) despite the distance between source and receptor being set at ~3 km for these emissions. This is because the PCSA model, on which CGM relies for calculating factors for emissions from GDF, assumes a worst-case scenario in which, for instance, the critical group's potable water is obtained directly from contaminated groundwater.

4.2 A practical rule for the application of UCrad and CGM

To conclude, we discuss practical applications of these methodologies. We distinguish two distinct types of problem in environmental assessment: selection of technology and selection of site and scale for a plant using a particular technology. LCA is the appropriate tool for comparing technologies, whilst HERA is appropriate for decisions over siting and scale of a plant. Although the two types of problem require different approaches, the methodologies should be compatible and methodologically consistent so that their results can be compared to reveal, for example, the extent to which a specific group receives impacts different from those on the general population, and whether reduction of global impacts is achieved at the expense of damage to a specific group.

It is common practice in LCA to distinguish between the foreground system, comprising specific processes for which primary, site-specific data are available and whose selection or mode of operation

265 is affected by decisions based on the study, and the background system which exchanges materials
266 and energy with the foreground (Clift et al., 2000). UCrad represents the most appropriate
267 methodology for assessing the impacts of ionising radiations in LCA because it is consistent with the
268 general approach used in LCIA; i.e. assessment of global impacts of the whole system, foreground plus
269 background, on a uniformly disperse population. CGM, on the other hand, can be used to describe the
270 geographically specific impacts of the foreground processes. UCrad and CGM describe different
271 impact pathways – generalised and localised – but the comparison in section 4.1 shows that
272 differences between their predictions are explicable and represent real differences between the two
273 types of assessment. Given that the approaches are methodologically consistent and compatible even
274 when their specific predictions are different, they can be used alongside each other to assess the
275 impacts of ionising emissions for the different purposes and treated as equivalent.

276 Thus, combined application of UCrad and CGM can help to resolve one of the concerns in LCA: how to
277 reconcile site-specific with generalised assessment (see Clift et al., 2000). However, it has notable
278 implications for definition of the functional unit, which as noted in Section 1 is one of the fundamental
279 differences between LCA and HERA. Comparison of technologies can be based on any functional unit,
280 be it arbitrarily chosen by the LCA practitioner or representative of actual operational data. However,
281 recommendations on the location and the scale are meaningful only if the functional unit is related to
282 the actual or potential scale of operations of the plant.

283 To take a concrete example: imagine an LCA study that aims to compare two uranium mines, one
284 underground and one superficial (known as open-pit) in different locations and at different distances
285 from inhabited centres. An arbitrary functional unit, as used in conventional LCA, might be 1 kg of
286 uranium mined. However, 7000 tonnes might be a realistic functional unit representing the potential
287 mine[1]. UCrad, used to assess which mine is environmentally preferable by considering emissions

[1] This figure actually represents the approximate annual output of the Cigar Lake underground mine in Canada, the largest uranium mine by volume of production in 2017 (WNA, 2019).

arising both from the foreground mine site and the background processes making up the entire life cycle, can be based on either arbitrary or actual functional units. By contrast, CGM must be based on the actual scale of the mine to estimate its radiological impacts on a chosen critical group, e.g. the closest centre of human population.

The combination of CGM and UCrad represents a first step towards a possible integration of HERA and LCA, an idea that remains to be explored further.

5 Conclusions

Based on the general framework and two specific methodologies for assessing radiological impacts in LCA proposed by Paulillo et al. (2019) - UCrad and the Critical Group Methodology (CGM) - this article presents a detailed quantitative comparison between the characterisation factors obtained from these methodologies. The characterisation factors for CGM are strongly affected by radioactive decay and dilution of the radioisotopes in the pollutant plume. By contrast, UCrad does not account for dilution and is less affected by radioactive decay because it is based on compartment-type models at steady-state conditions to predict the fate of nuclides. Therefore, characterisation factors obtained from CGM are much lower than those from UCrad when a significant proportion of the radionuclide decays during transit from emission source to receptor in a specific environmental medium; i.e. when the half-life of radioactive decay is lower than the transit time. When the half-life is long, radioactive decay is negligible and the factors for CGM are primarily affected by dispersion of the plume: dilution becomes more significant at greater distances, resulting in characterisation factors that decrease with distance for CGM. Factors for emissions to freshwater remain constant at high half-life because transport is assumed to occur by riverine flow and dilution is not considered.

Finally, a practical rule for the applications of these methodologies has been proposed. UCrad represents the appropriate methodology for assessing the impacts of ionising radiations in LCA because it is consistent with the general approach of Life Cycle Impact Assessment (LCIA) and the

312 methodologies for assessing impacts of toxic pollutants. However, the CGM methodology can be
313 applied alongside UCrad to enable recommendations to be made on the site and scale of a plant using
314 a particular technology, for example to comply with regulatory limits.

315 6 Acknowledgments

316 The authors wish to thank the Nuclear Decommissioning Authority (NDA) for the financial support
317 provided through the DISTINCTIVE (Decommissioning, Immobilisation and Storage soluTIons for
318 NuClear wasTe InVEntories) programme (DISTINCTIVE Consortium, 2019).

319 7 References

320 Clift, R., Doig, A., Finnveden, G., 2000. The Application of Life Cycle Assessment to Integrated Solid
321 Waste Management. Part 1 - Methodology. Process Saf. Environ. Prot. 78, 279–287.
322 doi:10.1205/095758200530790

323 Cowell, S.J., Fairman, R., Lofstedt, R.E., 2002. Use of risk assessment and life cycle assessment in
324 decision making: A common policy research agenda. Risk Anal. 22, 879–894. doi:10.1111/1539-
325 6924.00258

326 DISTINCTIVE Consortium, 2019. DISTINCTIVE [WWW Document]. URL
327 http://distinctiveconsortium.org/

328 Dreicer, M., Tort, V., Manen, P., 1995. Externalities of fuel cycles. European Commission, DG XII,
329 Science, Research and Development, JOULE, ExternE Externalities of Energy, Vol. 5, Nuclear.
330 Office for Official Publications of the European Communities, Luxembourg.

331 Flemström, K., Carlson, R., Erixon, M., 2004. Relationships between Life Cycle Assessment and Risk
332 Assessment - Potentials and obstacles, Swedish Environmental Protection Agency.
333 doi:10.1016/j.jempfin.2002.12.003

Fogler, H.S., 2006. Elements of chemical reaction engineering. Pearson Education, Westford, Massachussets. doi:10.1016/0009-2509(87)80130-6

Frischknecht, R., Braunschweig, A., Hofstetter, P., Suter, P., 2000. Human health damages due to ionising radiation in life cycle impact assessment. Environ. Impact Assess. Rev. 20, 159–189.

IAEA, 2001. Generic Models for Use in Assessing the Impact of Discharges of Radioactive Substances to the Environment. Safety Reports Series No. 19. IAEA, Vienna.

ICRP, 1990. 1990 Recommendations of the International Commission on Radiological Protection. ICRP Publication 60. Ann. ICRP 21.

Mackay, D., 2001. Multimedia Environmental Models: The fugacity approach, 2nd ed. ed. Lewis Publisher.

NCRP, 1995. Screening models for releases of radionuclides to atmosphere, surface, water, and ground. NCRP report No. 123 I and II. NCRP, Bethesda, Md.

NDA, 2010. Geological Disposal: Generic post-closure safety assessment.

Nishioka, Y., Levy, J.I., Norris, G.A., Wilson, A., Hofstetter, P., Spengler, J.D., 2002. Integrating risk assessment and life cycle assessment: a case study of insulation. Risk Anal. 22, 1003–1017. doi:10.1111/1539-6924.00266

NRPB, 1993. Occupational, public and medical exposure. Guidance on the 1990 recommendations of ICRP.

Olsen, S.I., Christensen, F.M., Hauschild, M., Pedersen, F., Larsen, H.F., Torslov, J., 2001. Life cycle impact assessment and risk assessment of chemicals -- a methodological comparison. Environ. Impact Assess. Rev. 21, 385–404. doi:10.1016/S0195-9255(01)00075-0

Owens, J.W., 1997. Life-cycle assessment in relation to risk assessment: An evolving perspective. Risk Anal. 17, 359–365. doi:10.1111/j.1539-6924.1997.tb00874.x

Paulillo, A., 2018. Operationalising the use of Life Cycle Assessment to nuclear waste management. PhD Thesis. University College London.

Paulillo, A., Clift, R., Dodds, J.M., Milliken, A., Palethorpe, S.J., Lettieri, P., 2019a. Radiological Impact Assessment in Life Cycle Assessment. Part I: General framework and two practical methodologies. Sci. Total Environ.

Paulillo, A., Clift, R., Dodds, J.M., Milliken, A., Palethorpe, S.J., Lettieri, P., 2019b. Data supporting two novel methodologies for radiological impacts in Life Cycle Assessment: UCrad and the Critical Group Methodology. Data Br.

Rosenbaum, R.K., Bachmann, T.M., Gold, L.S., Huijbregts, M.A.J., Jolliet, O., Juraske, R., Koehler, A., Larsen, H.F., MacLeod, M., Margni, M., McKone, T.E., Payet, J., Schuhmacher, M., Van De Meent, D., Hauschild, M.Z., 2008. USEtox - The UNEP-SETAC toxicity model: Recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. Int. J. Life Cycle Assess. 13, 532–546. doi:10.1007/s11367-008-0038-4

Saouter, E., Feijtel., T.C.J., 2000. Use of life cycle analysis and environmental risk assessment in an integrated product assessment. Environmental Strategies., in: Hauschild, M.Z., Olsen, S., Poll, C. (Eds.), Risk Assessment and Life-Cycle Assessment. Nordic Council of Ministers, Copenhagen, Denmark, pp. 81–97.

Sonnemann, G., Castells, F., Schuhmacher, M., Hauschild, M., 2004. Integrated life-cycle and risk assessment for industrial processes, Lewis Publ. ed. doi:10.1007/BF02994195

Spadaro, J. V., Rabl, A., 1999. Estimates of real damage from air pollution: Site dependence and simple impact indices for LCA. Int. J. Life Cycle Assess. 4, 229–243. doi:10.1007/BF02979503

Udo de Haes, H. a., Sleeswijk, A.W., Heijungs, R., 2006. Similarities, Differences and Synergisms Between HERA and LCA—An Analysis at Three Levels. Hum. Ecol. Risk Assess. An Int. J. 12, 431–449. doi:10.1080/10807030600561659

UNSCEAR, 2017. Sources, Effects and Risks of ionizing Radiation. Annex A: Methodology for estimating public exposures due to radioactive discharges. New York.

Wegener Sleeswijk, A., Heijungs, R., Erler, S.T., 2003. Risk Assessment and Life-cycle Assessment. Greener Manag. Intern. 77–88.

WNA, 2019. World Uranium Mining Production [WWW Document]. URL http://www.world-nuclear.org/information-library/nuclear-fuel-cycle/mining-of-uranium/world-uranium-mining-production.aspx (accessed 5.24.19).

Wrisberg, N., Udo De Haes, H.A., Triebswetter, U., Eder, P., Clift, R., 2002. Analytical tools for environmental design and management in a systems perspective: the combined use of analytical tools. Kluwer Academic Publishers, Dordrecht, Netherlands.