Averting cumulative lifetime attributable risk (LAR) of cancer by decontamination of residential areas affected by a large-scale nuclear power plant fallout: time aspects of radiological benefits for newborns and adults

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Abstract

The averted cumulative lifetime attributable risk (LAR), the residual dose and highest ground deposition of $^{137}$Cs complying with a reference dose level of 20 mSv yr$^{-1}$ to an individual returning after one year to an area contaminated by unfiltered releases of fission products from a nuclear power plant (NPP) were evaluated by applying an existing exposure model designed to compute age- and gender-dependent time-integrated LAR. The model was applied to four types of nuclear fallout scenarios, partly based on data from the Chernobyl and Fukushima releases and from theoretical source terms from Swedish NPPs. For rapid decontamination measures that achieve a 50% relative reduction in external dose rate within 1 year, compliance with the reference level 20 mSv yr$^{-1}$ can be attained for an initial $^{137}$Cs ground deposition of up to 2 MBq m$^{-2}$ with relaxed food restrictions. This compliance can be

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attained at even higher ground deposition (up to 4.5 MBq m\(^{-2}\)) if using the strict food restrictions employed in Japan after 2011. Considering longer than 1 year return times it was also found that the benefit of implementing decontamination decreases rapidly with time (2–3 years half-time), especially if the fallout has a high initial \(^{134}\)Cs to \(^{137}\)Cs activity ratio and if the ecological half-time of the external dose rate is short (<5 years). Depending on fallout scenario the averted cumulative LAR for newborn girls by decontamination that is achieved after 5 years is only between 6% and 11% of that obtained by evacuation alone during the same time, indicating a rather limited radiological benefit of decontamination if delayed more than a few years. We conclude that decision makers and emergency response planners need to consider that protracted decontamination measures may have limited radiological benefit compared with evacuation in terms of averted future cancer cases, albeit it may have other societal benefits.

Keywords: decontamination, life-time attributable risk, residual dose, averted cumulative life-time attributable risk, nuclear power plant release, restoration

(Some figures may appear in colour only in the online journal)

1. Introduction

An interdisciplinary research project, started in 2017, has the goal of providing recommendations for decision-making regarding restoration of urban environments in Sweden following a radiological or nuclear (RN) incident, with special focus on releases from nuclear power plants leading to deposition of fission products. The project aims to produce a tool that allows for easy comparisons between different remediation actions and also considers monetary costs, especially for typical Swedish living conditions. In addition, the tool accounts for expected reactions from the public in its recommendations for protective measures. One part of this project is developing a simple tool for predicting the outcome of decontamination measures in terms of averted detriment to the affected population, considering the age and gender dependence of the radiation sensitivity. A method was developed (Rääf et al 2020) that applies risk coefficients defined in terms of lifetime attributable risk, LAR (U.S. Environmental Protection Agency, Environmental Protection Agency (EPA) 2011), as a measure of detriment from a standard type of nuclear power plant (NPP) fallout based on the wet deposition of Chernobyl fallout in Sweden (Jönsson et al 2017).

Using the LAR concept as a measure of radiation-induced detriment, it was evident that for nuclear fallout scenarios, newborn girls have the highest radiation sensitivity in the population, with up to a factor of 4 higher lifetime cumulative detriment than adults if living in an area that has recently been affected by fallout of fission products from a NPP release (Rääf et al 2020). This phenomenon was also considered by (Yasuda 2018), who furthermore discussed the risks that would be entailed by returning to an area affected by \(^{134}\)Cs and \(^{137}\)Cs deposition. The higher cumulative LAR-values for young individuals, and especially females, result from a combination of longer remaining life expectancy over which late effects can be manifested compared with adults. The gender difference arises from the relatively high radiation risk coefficient for breast, and that females generally exhibit distinctively higher risk coefficients compared with males, in terms of probability of cancer per unit organ radiation dose (Environmental Protection Agency (EPA) 2011). Attempts to repopulate previously restricted
areas after the Fukushima NPP accident in 2011 showed that families with small children were, to a large extent, reluctant to return to decontaminated areas. In order to assure an appropriate remediation and information strategy after a radiological or nuclear incident, the most sensitive age and gender cohorts are of particular interest. Thus, it is essential to investigate these sensitive age cohorts in terms of the benefits that can be achieved by protective measures and restoration of an affected community. Preliminary results from an inquiry made in connection with this multidisciplinary project showed that, among the general population, families with small children may be the most reluctant to resettle in a restored area (Rasmussen et al 2020).

In Japan, areas were designated according to the amount of fallout and the intensity of decontamination measures that were needed to restore an area for returning evacuees (e.g. Yasutaka and Naito 2016). For the so-called special decontamination areas (SDA), the external dose rate in most cases exceeded 20 mSv yr$^{-1}$, but the fallout levels were assumed sufficiently manageable to consider large-scale restoration measures. Experience from the launch and completion of decontamination plans showed that the fastest restoration was achieved in the SDA zone in Tamura municipality (400 affected inhabitants) (Ministry of the Environment (MOE) Japan 2015). A more affected area in the SDA zone, Okuma municipality, which had a $^{137}$Cs deposition $>3$ MBq m$^{-2}$ (Tsubokura et al 2013) and similar numbers of affected inhabitants as Tamura, was restored within 3 y after the fallout. It was one of several areas in the SDA zone being restored with similar pace. In a future fallout scenario in an area governed by an advanced economy, these time frames may be suitable as a reference given the extent of material resources and legal and administrative measures adopted by the Japanese government for restoring the SDA zone. Hence, for this modeling study, the time frame between 1 and 5 y after the fallout is of special interest (Ministry of the Environment (MOE) Japan 2015).

Decontamination measures are costly (e.g. Nisbet et al 2011, Munro 2013), and in the case of extensive fallout that may require restoration of large areas, it is important to be able to predict, even at the planning stage, which radiological circumstances justify decontamination or even the consideration of decontamination. The degree of achieved reduction of the decontamination depends on the physicochemical character of the deposition (such as fraction of wet- or dry deposition, particle size and nuclide composition of the deposit), and on location specific factors (such as type of contaminated surfaces) that may vary substantially even over small areas. An evaluation of the costs and dose reduction effectiveness of decontamination measures undertaken by (Yasutaka and Naito 2016) showed that effectiveness is highly variable and appears to depend on the degree of ground contamination, with greater dose reduction occurring for high deposition areas. Reported values of relative dose reduction for residential areas range from 15% to 35% in areas with ambient dose rate levels of 3 $\mu$Sv h$^{-1}$ and less and from 25% to 70% in highly contaminated areas. N.B. that this way of expressing dose reduction is related to but not identical to the term $DF$ used in e.g. (Nisbet et al 2011), which is defined as the ratio between the initial dose rate and the dose rate after remediation. (Tsubokura et al 2019) also report that the greatest dose rate reduction by decontamination was achieved in areas with a higher dose rate at the decontamination start time, regardless of the timing of the decontamination. A median assumption of a dose rate reduction of 50% will thus represent the approximate median variant of likely achievable reductions in residential areas. (Samuelsson et al 2019) showed that it is theoretically unlikely to attain dose rate reduction more than 90%, and a dose rate reduction of 100% to 90% will therefore be the most optimistic outcome of a large-scale decontamination, although this level may be readily achieved in residential areas under ideal circumstances for single buildings (e.g. Roed et al 1998, Andersson et al 2003).

To date, it is well known in the field of emergency preparedness that in case of an unfiltered atmospheric release from a nuclear reactor, radiocaesium has the main impact on long-term doses, averted doses, and residual doses to affected residents (e.g. USNRC, United States
Nuclear Regulatory Commission 1995, UNSCEAR 2013). $^{137}$Cs (in semi-equilibrium with $^{137}$Ba) is a gamma emitter that is readily mapped by radiometry or by soil sampling. Using simple models based on initial $^{137}$Cs deposition maps alone, decision makers and researchers can predict which areas need to be considered for costly restoration measures. In this study, we have especially focused on residential urban areas, as these involve the most members of the public and hence the largest sensitive age cohorts. Although existing complex decision-support tools, such as (RODOS 2018) and (ARGOS 2018), can be applied to prioritisation on a more detailed level and furthermore also include industrial areas, the simplified model used here computes age- and gender-dependent organ-specific absorbed doses and associated lifetime attributable risks. N.B, that the model used in this study refer to accidental atmospheric releases of fission products from operating NPPs, where $^{137}$Cs can be used as a key nuclide to predict the presence and dose contribution from other released accompanying fission products, such as $^{131}$I, $^{132}$I and $^{132}$Te. The model is, as yet, not adapted for releases from other types of nuclear facilities, such as reprocessing plants and high powered accelerators for research purpose (e.g. European Spallation Source), which potentially can lead to local and regional deposition of radioactive fallout that may warrant large-scale decontamination (e.g. Album Ytre-Eide et al 2009).

Given the extended timeframes for realistically achieving decontamination in large residential areas, together with the range of realistic dose reduction factors achievable, what is the importance of the time delay in justifying decontamination in an area affected by nuclear fallout? The aim of this study was to investigate (i) representative values of $^{137}$Cs deposition that can be used as an a priori indication of whether a specific area can be inhabitable given a fixed time frame and dose reference level and (ii) the time dependence of the radiological benefit of a general decontamination measure compared with evacuation and rigorous food restrictions. The category of newborn girls is of special interest and is mainly used to evaluate the radiological benefit in terms of averted later occurrence of radiation-induced cancers.

2. Materials and methods

2.1. Description of an exposure model for fictive residents in a contaminated area: cumulative effective dose

In previous studies, a simple model for predicting the sum of long-term external and internal exposure from a NPP release was developed based on observations of the Chernobyl fallout in Sweden after 1986 (Isaksson et al 2019). Traditionally in emergency preparedness, the projected effective dose has been used as a measure of radiological detriment to reference populations or persons who are affected by non-planned radiation exposures (ICRP 2009a). However, recently the alternative use of the lifetime attributable risk (LAR), based on risk coefficients presented by the US National Research Council of the National Academies (National Research Council of the National Academies (NRC) 2006) and (Environmental Protection Agency (EPA) 2011), has been advocated as a complement for predicting the potential outcome of non-planned radiation exposures (Yasuda 2018, Rääf et al 2020). The model by (Isaksson et al 2019) was further developed to also express the detriment in terms of time-integrated absorbed dose to specific organs, as well as cumulative lifetime attributable risk (CUMLAR), based on risk coefficients taken from the US EPA (Environmental Protection Agency (EPA) 2011, Rääf et al 2020). The model has mainly focused on radiation exposure pathways associated with long-term exposure and does not include cloudshine and initial inhalation doses to the population. The model also does not include the transfer of short-lived fission products
such as $^{131}$I, $^{132}$I, and $^{132}$Te to dairy milk and the regional distribution to consumers. For planning purposes, it may be assumed that modern emergency preparedness must include initial predetermined protective measures to stable grazing cattle and inhibit the transfer of freshly contaminated crops to the main food distribution chains. A discussion on the iodine-related absorbed doses to the thyroid compared with other contributing exposures with the use of protective measures can be found in, e.g. (Rääf et al 2019). In this study, we therefore focus on the dose contribution from external and internal exposure to gamma emitters ($^{134}$, $^{137}$Cs), as this model is to be used for emergency preparedness planning in cases when the aforementioned protective measures are already prepared to be readily implemented in case of anticipated NPP fallout.

The model is designed so that all components can be related to the initial ground deposition of $^{137}$Cs, $A_{dep}$. The advantage of this approach is that ground deposition is often the first parameter to be determined in an accident scenario. Many resources have also been allocated to the design of radiometry for early car-based and airborne gamma monitoring (e.g. Isaksson and Rääf 2016). A distinction is made in the model between local fallout, $A_{dep,loc}$, and the regional average (areas of $1 \cdot 10^4$–5 $\cdot 10^4$ km$^2$) of the deposition, $A_{dep,reg}$, as the latter relates more to the anticipated radioecological transfer of $^{137}$Cs to main foodstuffs such as dairy milk and meat consumed by urban residents than to the local deposition (Rääf et al 2006a). The radioecological transfer factor can be represented by a factor, $T_{ag,max}$ (presented in table 1 and discussed in more detail in (Isaksson et al 2019), which estimates the anticipated whole-body burden among populations living in the affected area. Typically, a semi-equilibrium between intake of radioaesium contaminated foodstuff and excretion will be obtained among the residents, and body burdens appear to peak within 1–2 years upon fallout, after which the average body burden level exhibit an exponential (or bi-exponential) decrease (e.g. Rääf et al 2006b, Isaksson et al 2019). The time-integrated body burden of $^{134}$Cs and $^{137}$Cs will then determine the corresponding long-term internal dose from the fallout to the population.

The analytical expression for computing the organ absorbed dose is given in equation (1), and a description of the parameters is given in table 1. The corresponding expressions for the cumulative effective dose and CUMLAR for various age and gender cohorts are exhaustively described in (Rääf et al 2020), which also provides a calculation spreadsheet with a downloadable example case of the model.

$$D_{org,sex}(t_{acc}, age, A_{dep,loc}, A_{dep,reg}) = A_{dep,loc} \cdot S_{decont} \cdot d_{Cs} \cdot \Phi_{K/H}$$

$$\left( 600 \text{ keV} \right) \cdot \left( f_{smow} \cdot k_{SEQ,Org,ext} \right) \cdot \int_{0}^{t_{acc}} \left( t(t) \cdot k_{SEQ,K}(age) \cdot \left( f_{out} + (1 - f_{out}) \cdot f_{shield} \right) \right) dt$$

$$+ A_{dep,reg} \cdot T_{ag,max} \cdot S_{aliment} \cdot \int_{0}^{t_{acc}} \left( \left( 1 - e^{-\frac{t}{T_{ag}}} \right) \cdot \left( c_1 \cdot e^{-\frac{t}{T_{ag}}} + c_2 \cdot e^{-\frac{t}{T_{ag}}} \right) \right) dt$$

$$\times f_{sex} \cdot \left( k_{Organ,Int,Cs-137} \cdot e_{Cs-137}(w(age(t)) + k_{Organ,Int,Cs-134} \right)$$

$$\times FR \cdot e \left( \frac{h_{1,34,Cs-137}}{h_{1,34,Cs-137}} - \frac{h_{1,34,Cs-137}}{h_{1,34,Cs-134}} \right)$$

$$\times e_{Cs-134}(w(age(t))) \cdot dt$$

In this continued study, only whole-body exposure was considered, as we aimed to estimate the CUMLAR of all radiation-induced cancers for residents living in an area affected by nuclear fallout. The average value of $k_{SEQ,Org,ext}$ for 11 organs for females (bladder, bone, stomach, colon, liver, lung, red bone marrow, thyroid, uterus, breast, and ovaries) and 9 organs for males (bladder, bone, stomach, colon, liver, lung, red bone marrow, thyroid, and testes) were used
Table 1. Parameter values used for calculation of organ-absorbed dose, $D_{org}$, in equation (1). The parameters are extensively described in (Räaf et al 2020).

| Parameter | Description (unit) |
|-----------|--------------------|
| $A_{dep,loc}(x,y)$ | Local deposition, decay corrected to the time of the fallout event (kBq m$^{-2}$). |
| $A_{dep,reg}$ | Regional average of $A_{dep,loc}$ (kBq m$^{-2}$). An average over an area representing a region defined in terms of administrative or economic relevance for the local population and from which the main part of the ingested local food by the residents in the area originates. Note that in these calculations we have chosen to set $A_{dep} = A_{dep,loc} = A_{dep,reg}$. |
| $d_{Cs}$ | Relation between ambient dose rate (mSv yr$^{-1}$) 1 m above ground and ground deposition (kBq m$^{-2}$) of $^{137}$Cs in fresh fallout. This factor was empirically determined to be 0.636 mSv yr$^{-1}$/kBq m$^{-2}$ for the Chernobyl fallout in Sweden (Jönsson et al 2017, Räaf et al 2020). |
| $\theta_{K/\beta}(600\text{ keV})$ | Ratio between air kerma rate and ambient dose equivalent rate (mGy mSv$^{-1}$). Here, this is specified to 1 m above ground for an infinite uniform surface source consisting of gamma emitters with photon energy 600 keV: 0.83 mGy mSv$^{-1}$ (International Commission on Radiation Units and Measurements (ICRU) 1992). |
| $f_{snow}$ | Factor representing shielding by snow cover. Here, no snow cover is assumed and $f_{snow} = 1$. For a more detailed definition of $f_{snow}$, refer to (Finck 1992). |
| $r(t)$ | Function describing the decrease in external dose rate 1 m above ground, normalised to the maximum initial dose rate following fallout. Apart from $^{134}$Cs and $^{137}$Cs, contributions from short-lived gamma emitters, such as $^{131}$I, $^{132}$Te, $^{136}$Cs, are included (Jönsson et al 2017). The function given by (Jönsson et al 2017) for the Chernobyl fallout in Sweden, with time constants expressed in terms of yr$^{-1}$, is $r(t) = c_0 \cdot e^{-c_1 \cdot t} + c_2 \cdot e^{-c_3 \cdot t} + c_4 \cdot e^{-c_5 \cdot t} + c_6 \cdot e^{-c_7 \cdot t}$, where $c_0 = 0.96$, $c_1 = -36.9$ yr$^{-1}$, $c_2 = 0.108$, $c_3 = -2.45$ yr$^{-1}$, $c_4 = 0.0796$, $c_5 = -0.668$ yr$^{-1}$, $c_6 = 0.0314$, and $c_7 = -0.126$ yr$^{-1}$. In this study, $r(t)$ and the coefficients $c_0$, $c_1$, $c_7$ have been slightly adapted to other types of fresh nuclear fallout (see later section). |
| $f_{out}$ | Outdoor occupancy factor for an individual residing in a temperate climate zone (Almgren et al 2008a, 2008b, Brown 1983, World Health Organization 1999). Here, $f_{out} = 0.2$ was used. |
| $f_{shield}$ | Shielding factor of residents, which ranges between 0.10 and 0.4 for Northern European houses (Finck 1992). A value of $f_{shield} = 0.4$ was used in this work to represent one-story semidetached residential buildings. A conservative value of the shielding factors for a plane surface deposition of gamma emitters with approximate mean energy of 600 keV in the residential buildings of $f_{shield} = 0.4$ has been used. For most types of residential buildings, this value may underestimate the shielding properties of an irradiation geometry. However, in practice, shielding factors depend on the contamination level on the various surfaces and thus on the deposition scenario and the physicochemical properties of the contaminants. Migration of contaminants through weathering processes over time also affects the shielding factor. |
Table 1. Continued.

| Parameter   | Description (unit)                                                                                                                                                                                                                   |
|-------------|-------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|
| $t_{\text{acc}}$ | Integration time for the radiation exposure ($y$).                                                                                                                                                                                  |
| $T_{ag,max}$ | Parameter that determines the time-dependent radioecological transfer function of radionuclides from ground deposition to man ($\text{Bq kg}^{-1}/\text{kBq m}^{-2}$). In our studied scenarios $Tag(t)$ thus relates regional-average ground deposition of $^{134,137}\text{Cs}$ to the anticipated whole-body concentration of $^{134,137}\text{Cs}$ in the human population living in the affected region. For Scandinavian urban residents after the Chernobyl fallout, a value of $T_{ag,max} = 6.7 \text{ Bq kg}^{-1}/\text{kBq m}^{-2}$ best represents empirical data on the long-term body burden of $^{134}\text{Cs}$ and $^{137}\text{Cs}$ (Rääf et al 2006, Rääf et al 2020). The Swedish urban population was subjected to food regulations that complied with the current day (Euratom 2016) directives with action levels set to 300 Bq kg$^{-1}$ fresh weight. For hunters or people with lifestyles involving consumption of game, freshwater fish, berries, and mushrooms, $T_{ag,max}$ was found by Rääf et al (2006b) to be about a factor of 3 higher (18 Bq kg$^{-1}$/kBq m$^{-2}$). The latter categories of food are not extensively distributed on the open market, and were not part of the food restrictions. The radioecological transfer in this population therefore better mimics a situation where no particular food restrictions are undertaken. For rural Russian residents affected by the Chernobyl fallout, values range in between that of Scandinavian urban residents and hunters (Isaksson et al 2019), indicating that the former group had a dietary habit that more mimics the one among people who must rely also on locally produced foodstuff from small scale agriculture and forest products, that are not directly regulated in terms of radionuclides content as the ones on the open market. For Japanese evacuees during the years 2012 to 2015 living in Nihonmatsu city, the corresponding transfer factor between local deposition and $^{137}\text{Cs}$ body burden concentration amounted to less than 10% of that of urban Swedish residents (when comparing data from Hosokawa et al 2017 with Rääf et al 2006a), even in individuals identified as being significantly contaminated. Japan illustrates a case in which this exposure pathway and associated transfer factor virtually becomes zero and is a situation in which comprehensive food countermeasures on locally produced contaminated foodstuff to a large extent have been implemented. |
| $w(\text{age}(t_0 + t))$ | Body mass (kg) as a function of age. A curve fit of data from (Albertsson-Wikland et al 2002) has been used (Rääf et al 2020).                                                                                                                                 |
| $t_1$, $t_2$, and $t_3$ | Time parameters for radioecological transfer depending on population type. The values used here for an urban population are $t_1 = 1.0 \text{ y}$, $t_2 = 0.75 \text{ y}$, and $t_3 = 15 \text{ y}$. Corresponding values for hunters (and a lifestyle associated with similar dietary habits) are $t_1 = 0.9 \text{ y}$, $t_2 = 1.2 \text{ y}$, and $t_3 = 30 \text{ y}$. Values for other types of populations can be found in (Isaksson et al 2019). |
| $c_1$ and $c_2$ | Fit parameters for radioecological transfer depending on type of population. Empirical values based on urban populations in Scandinavia are $c_1 = 1.0$ and $c_2 = 0.10$ and have been used as a reference. Values for other types of populations can be found in (Isaksson et al 2019). |
| $T$ | Time in years.                                                                                                                                                                                                                      |
| $T_{1/2,\text{Cs-137}}$ | Physical half-life of $^{137}\text{Cs}$: 30.2 y                                                                                                                                                                                      |
| $T_{1/2,\text{Cs-134}}$ | Physical half-life of $^{134}\text{Cs}$: 2.06 y                                                                                                                                                                                      |
| $FR$ | Isotopic ratio $^{134}\text{Cs}/^{137}\text{Cs}$ at the time of initial fallout. A value of 0.56 was reported by Rääf et al. (2006a) for the Chernobyl NPP fallow, while reported values for the Fukushima Dai-ichi NPP release range from 1 to 1.1. In this study, source terms of the initial fallout have been taken from (UNSCEAR 2013), reporting $FR = 1$. In a recent Swedish exercise, a nuclear vector representing a release from a Swedish NPP was used with a ratio $FR = 1.47$. |
Table 1. Continued.

| Parameter | Description (unit) |
|-----------|--------------------|
| $f_{sex}$ | Empirical factor accounting for lower observed concentration of radiocaesium per unit body mass in women compared with adult males (Rääf et al. 2006a). $f_{sex} = 0.61$ for females aged > 20 years; $f_{sex} = 1$ for males at all ages and females < 20 years. |
| $e_{Cs-^{137}}$ | Effective dose conversion factor (mSv yr$^{-1}$/Bq kg$^{-1}$) taken from (Falk et al. 1991) based on biokinetic models by (Leggett et al. 1984). $e_{Cs-^{137}}(w) = 0.0014 + w(\text{age}(t))^{0.111}$, where the factor 0.0014 is a curve fit parameter and $w(\text{age})$ (kg) is the mean age-dependent body weight. It is assumed that this quantity is numerically equal to the absorbed dose per unit activity concentration in the body (see also Rääf et al. 2019). |
| $e_{Cs-^{134}}$ | Effective dose conversion factor (mSv yr$^{-1}$/Bq kg$^{-1}$) taken from (Falk et al. 1991) based on biokinetic models by (Leggett et al. 1984). $e_{Cs-^{134}}(w) = 0.00164 + w(\text{age}(t))^{0.188}$, where the factor 0.00164 is a curve fit parameter and $w(\text{age})$ (kg) is the mean age-dependent body weight of an individual at a certain age. It is assumed that this quantity is numerically equal to the absorbed dose per unit activity concentration in the body (see also Rääf et al. 2019). |
| $k_{\text{SEQ},\text{Organ},\text{ext}}$ | Organ-specific absorbed dose rate per unit kerma rate 1 m above ground for an adult of gender female (F) or male (M). Gender-specific values of $k_{\text{SEQ},\text{Organ},\text{ext}}$ (Gy Gy$^{-1}$) for the organs related to cancers specified in (Environmental Protection Agency (EPA) 2011) are given in (Rääf et al. 2020). |
| $k_{\text{SEQ},K}(\text{age}(t))$ | Age-dependent organ-specific absorbed dose rate per unit kerma rate (mGy mGy$^{-1}$), normalised against the corresponding value for an adult (female or male, respectively), described in (Rääf et al. 2020). $k_{\text{SEQ},K}(\text{age}(t)) = (0.0015 \cdot \text{age}(t)^2 - 0.1214 \cdot \text{age}(t)^3 + 3.473 \cdot \text{age}(t)^4 - 40.28 \cdot \text{age}(t)^5 + 136.3 \cdot \text{age}(t) + 1233)/1017$ for $\text{age} < 20$ years and 1 for $\text{age} \geq 20$ years. |
| $k_{\text{Organ},\text{int},Cs-^{134}}$ | Ratio between absorbed dose in a specific organ and the average whole-body absorbed dose incurred by a uniformly distributed internal contamination of $^{134,137}$Cs. Ratios for organs specified in (Environmental Protection Agency (EPA) 2011) are taken from Snyder et al. (1975) and given in (Rääf et al. 2020). |
| $S_{\text{decont}}$ | Factor representing the ratio between the average ambient dose rate in the area after a decontamination procedure and the corresponding dose rate if no such action had been taken. |
| $S_{\text{aliment}}$ | Factor representing the relative decrease in proportion to the standard radioecological transfer factor of foodstuffs brought on by various countermeasures. In this study, the food restriction has not been modeled per se, but instead three typical levels of $T_{ag,max}$ have been assumed, for which $S_{\text{aliment}}$ has been set to unity. |
in equation (1). For internal exposures of radiocaesium ($^{134}$Cs and $^{137}$Cs), the average whole-body dose was retrieved by using a weight dependent conversion factor between whole-body concentration of $^{134}$Cs and $^{137}$Cs and corresponding absorbed dose rate. For a 70 kg person this factor is 16.4 and 14 $\mu$Gy yr$^{-1}$/($\text{Bq kg}^{-1}$) for $^{134}$Cs and $^{137}$Cs, respectively, assuming a homogenous distribution in the body. The values are based on so-called S-factors taken from (Snyder et al 1975), that were further elaborated for different body sizes by e.g. (Falk et al 1991). The expression in equation (1) was then combined with organ-specific risk coefficients taken from the (Environmental Protection Agency (EPA) 2011). The subsequent treatment of data is discussed in more detail in (Rääf et al 2020).

The model for the time-integrated detriment of protracted radiation exposure from NPP fallout can also be used to model the impact of various long-term protective measures. Dose reduction through various decontamination measures is represented as a factor, $S_{\text{decont}}$, that ranges from 0 to unity depending on the relative decrease in external dose rate to affected inhabitants achieved by an unspecified decontamination procedure. A decontamination efficiency of 90% is represented by an $S_{\text{decont}}$ value of $1-0.9=0.1$, and likewise, a relative reduction in external dose rate of 50% is represented by an $S_{\text{decont}}$ value of $1-0.5=0.5$.

2.2. Description of the scenarios

The calculations were based on the reference dose of 20 mSv yr$^{-1}$ applied to a reference person and used to justify resettlement in a contaminated area. This level was used in Japan (e.g. Mccurry 2011). In addition, corresponding values for 1 mSv yr$^{-1}$ were computed, as this represents the dose limit in planned exposure situations for members of the public. It can be mentioned that ICRP suggests that reference levels for actions implemented in public areas should be kept within the range between these two aforementioned values, although levels above 10 mSv yr$^{-1}$ are not generally recommended (ICRP, 2009b). It may also be argued that the lower level of 1 mSv yr$^{-1}$ might be the one actually employed in a future fallout scenario, and it is currently used as a reference level by authorities in Belarus for communications with the public regarding resettlement and economic revitalisation of Chernobyl-contaminated zones (e.g. Ramzaev et al 2020). A number of different settings have been defined to illustrate the impact of the following key parameters: initial activity ratio of $^{134}$Cs and $^{137}$Cs (FR), long-term component in ecological half-time for external dose ($c_6$ in $r(t)$, given in table 1), long-term half-time of external dose ($c_7$ in $r(t)$), and the magnitude of the time-integrated aggregate radioecological transfer factor, $T_{\text{ag, max}}$.

Regarding the internal exposures emanating from contaminated foodstuff produced locally and regionally, it was shown from Swedish data, the urban population exhibited a factor of 3 times lower $T_{\text{ag, max}}$ than for hunters (Rääf et al 2006). Russian data appear to lie somewhere between Swedish urban residents and Swedish hunters (Isaksson et al 2019). In Japan, however, this factor is barely above zero, as the detected internal doses from ingested $^{137}$Cs between resettled residents and a control group of non-returnees was nonsignificant (Tsubokura et al 2013). $T_{\text{ag, max}}$ will depend heavily on (1) the timing of the fallout in relation to the growth season of crops and (2) implemented food restrictions. The assumed values here are just a selection of possible $T_{\text{ag, max}}$ values for the transfer of wet-deposited radiocaesium with deposition occurring prior to the growth season. In the case of direct contamination on crops, it may be argued that such affected food products are unlikely to reach the open market. Instead, the long-term indirect contamination of crops and grazing forage via root uptake will mainly govern how radiocaesium is transferred regionally to the population through dairy milk or meat consumption (e.g. UNSCEAR 1982).

Three types of affected populations were considered:
1. Urban residents with a lifestyle similar to the Swedish urban population after the Chernobyl fallout in Sweden in 1986 (Räaf et al 2006a, 2006b). A value of $T_{ag,max} = 6.7 \text{ Bq kg}^{-1}/\text{kBq m}^{-2}$ was set to represent the radioecological transfer factor of radiocesium from ground deposition to man in this category (table 1).

2. Urban residents subjected to comprehensive protective measures regarding internally contaminated foodstuffs, mimicking the situation in Fukushima prefecture after the Fukushima Dai-ichi fallout in 2011 (Tsubokura et al 2013). A radioecological transfer factor of radiocesium from ground deposition to man of $T_{ag,max} = 0 \text{ Bq kg}^{-1}/\text{kBq m}^{-2}$ was set (table 1).

3. Rural residents with a lifestyle mimicking Swedish hunters after the Chernobyl fallout in 1986, with a higher-than-average consumption rate of game, berries, freshwater fish, and mushrooms (Ågren 1998). A radioecological transfer factor of radiocesium from ground deposition to man of $T_{ag,max} = 18.9 \text{ Bq kg}^{-1}/\text{kBq m}^{-2}$ was set (table 1). For the hunter lifestyle, it is assumed that the time spent outdoors will mostly be in areas that have not undergone decontamination, and thus $S_{decont}$ is applied only to the time fraction spent indoors.

Four types of NPP fallout scenarios were considered:

1. Wet-deposited Chernobyl fallout in Sweden (medium variant of ecological half-time). This scenario was used in (Isaksson et al 2019) and (Räaf et al 2020). An $FR = 0.56$ and a long-term ecological half-time of 5.5 y for the ambient dose rate were assumed based on (Jönsson et al 2017).

2. Swedish NPP release using the medium variant of ecological half-time of ambient dose rate ($=5.5$ years). The fictive source term is taken from the (Swedish Radiation Safety Authority 2020) with an $FR = 1.47$. (Saito and Petoussi-Henss 2014) presented ambient dose rate contributions from various gamma emitters and at various soil relaxation mass depths (g cm$^{-2}$). The relaxation mass depth is a parameter used in radioecology to describe how radionuclides from an atmospheric fallout successively migrate into soil and form an exponential distribution with depth (e.g. Isaksson and Räaf 2016). However, fresh atmospheric fallout associated with wet deposition will result in an initial radionuclide penetration into ground up to a few cm, as was evidenced from a number of soil sample measurements of Chernobyl radiocaesium conducted in Sweden after 1986 (Arntsing et al 1991). Therefore, this initial penetration into soil will attenuate the ambient dose rate above ground, thereby decreasing the proportionality between the initial ambient dose rate and measured deposition density of the gamma emitting fission products in ground. For the scenario of the Swedish NPP release, a relaxation mass depth of 1.0 g cm$^{-2}$ was assumed, and the calculations used ambient dose contribution values at 1 m above ground for the gamma emitters in the release. This relaxation mass depth mimics rather well the corresponding relaxation of the wet deposition of Chernobyl fallout in Sweden.

3. Swedish NPP release using a long ecological half-time of 15 years regarding external dose rate, partly based on observations from Russia with an effective ecological half-time of 17 years for dirt surfaces within rural residential areas (Ramzaev et al 2006). The fictive source term and initial relaxation mass depth are assumed as in the previous scenario.

4. Fukushima Dai-ichi Northern trace release with initial nuclide vector taken from (UNSCEAR 2013) with an $FR = 1$ and with a short variant of ecological half-time of ambient dose rate taken from observations in Japan (Hayes et al 2020).

Note that the long-term ecological half-time is not directly connected to the $^{134}\text{Cs}/^{137}\text{Cs}$ isotopic ratio $FR$ in these scenarios, but instead represents the long-term decay of ambient...
dose rate in different environments for the various scenarios (e.g. types of soil) and of the physicochemical properties of the different radiocontaminants in the nuclide vector for the scenarios. Once deposited on the ground, the fate of the fallout products depends on many different factors, such as the physicochemical properties of the long-lived fission products, the weathering and run-off processes on various types of surfaces, and the soil characteristics of the contaminated ground. In this simple model, these factors are to some extent indirectly accounted for in terms of the empirical normalised dose-rate function, \( r(t) \), based on the Chernobyl and Fukushima fallout.

The complexity of various types of surfaces (in terms of roughness, porosity, and land use) will result in varying half-times of the ambient dose rates (Ramzaev et al. 2020), and for more rural and undisturbed areas, the half-time of this long-term component can be as high as 15–20 years. However, a time series from more inhabited areas in Japan and Sweden showed that this half-time is much shorter (3.2 years for the Fukushima fallout in Japan and 5.5 years for the Chernobyl fallout in Sweden). In Japan, it is also found that a three-component exponential decay adequately fits the observed decline in ambient dose rate. For the Swedish time series (Jönsson et al. 2017), the third component in the \( r(t) \) function has been assigned to the initial relative dose rate contribution from \(^{134}\text{Cs}\), whereas the long-term component (half-time = 5.5 years) has been assigned to the corresponding initial \(^{137}\text{Cs}\) ambient dose rate contribution. For a future fallout scenario in Sweden, a similar \( r(t) \) function is used. Observations from Japan show disparate patterns regarding the long-term decay of ambient dose rate, but the most comprehensive data interpretation so far on ambient dose rate shows that a long-term decay with a relatively short half-time of 3.2 years best fits the data. Since Hayes et al. (2020) treated the sum of \(^{134}\text{Cs}\) and \(^{137}\text{Cs}\) as a single component, we have chosen to use the three-component model in which this long-term component approximately represents the initial dose contribution from both \(^{134}\text{Cs}\) and \(^{137}\text{Cs}\) to the total ambient dose rate at the event of the fallout (on the order of 6% according to UNSCEAR 2013). It may be argued that, in time, an additional long-term component in the ambient dose rate will be visible as all of the \(^{134}\text{Cs}\) has decayed, but the three-component \( r(t) \) function may serve as a case study of the lower extreme compared with the long ecological half-times observed in Russia. Thus, to encompass the possible extreme values of this parameter, the \( r(t) \) function has been adapted to four fallout scenarios (table 2).

In view of the discussion in the introduction, two achievement levels of decontamination measures in terms of relative reduction in external dose rate (\( S_{\text{decont}} \)) are considered here:

- \( S_{\text{decont}} = 0.5 \) (\( DR = 50\% \) reduction on average) of the external dose after decontamination measures have been conducted for a typical one- or two-story detached building. This value represents, by experience (Samuelsson et al. 2019, Tsubokura et al. 2019), the most realistic achievable level of reduction in a situation when large-scale measures, including decontamination of many buildings, need to be employed.
- \( S_{\text{decont}} = 0.1 \) (\( DR = 90\% \) reduction on average) of the external dose after decontamination measures in the near vicinity of the affected building. A similar level has been practically achieved for limited areas by, e.g. (Roed et al. 1998). (Andersson et al. 2003) estimated that topsoil removal of residential areas can, in theory if maximised, reduce the external dose rate by 90%–95%. Monte Carlo simulations of hypothetical ground contamination of \(^{137}\text{Cs}\) on the ground surface in a typical residential area suggest that a significant part of the external dose inside and outside of buildings may arise from remote areas and must be considered a ‘background’ that is virtually infeasible to practically reduce with limited resources (Hinrichsen et al. 2020).
Table 2. Empirical values of ambient dose rate per unit ground deposition of $^{137}$Cs and subsequent time patterns of ambient dose rate normalised to the initial maximum value during the main fallout event.

| NPP fallout source term (FR)                                      | $d_{CS}$ (mSv yr$^{-1}$/kBq m$^{-2}$) | $r(t)$ (t in years)                                                                 |
|-----------------------------------------------------------------|--------------------------------------|-------------------------------------------------------------------------------------|
| Chernobyl wet-deposition fallout in Sweden                       | 0.636                                | $r(t) = 0.96 \cdot e^{-36.9 \cdot t} + 0.10823 \cdot e^{-2.45 \cdot t} + 0.0796 \cdot e^{-0.667 \cdot t} + 0.0314 \cdot e^{-0.138 \cdot t}$ |
| ($FR = 0.56; T_{eco,long} = 5.5$ years)                         |                                       |                                                                                     |
| Fukushima Dai-ichi Northern trace fallout                        | 0.999$^b$                            | $r(t) = 0.758 \cdot e^{-36.9 \cdot t} + 0.205 \cdot e^{-10.1 \cdot t} + 0.0373 \cdot e^{-0.327 \cdot t}$                               |
| ($FR = 1; T_{eco,long} = 3.2$ years)                             |                                       |                                                                                     |
| Swedish NPP fallout (median long-term decay) ($FR = 1.47; T_{eco,long} = 5.5$ years) | 1.13$^c$                             | $r(t) = 0.976 \cdot e^{-36.9 \cdot t} + 0.10823 \cdot e^{-2.45 \cdot t} + 0.0632 \cdot e^{-0.667 \cdot t} + 0.0158 \cdot e^{-0.138 \cdot t}$ |
| Swedish NPP fallout (slow long-term decay) ($FR = 1.47; T_{eco,long} = 15$ years) | 1.13$^c$                             | $r(t) = 0.976 \cdot e^{-36.9 \cdot t} + 0.10823 \cdot e^{-2.45 \cdot t} + 0.0632 \cdot e^{-0.667 \cdot t} + 0.0158 \cdot e^{-0.0462 \cdot t}$ |

$^a$The amplitude of the first component is overcompensated as $r(t)$ is truncated to unity between the time range 0 to 3 d.

$^b$Calculated based on (UNSCEAR 2013) nuclide vector of northern trace and assuming the same penetration depth as for the Chernobyl fallout in Sweden in 1986.

$^c$Calculated with an assumed penetration depth of 1 g cm$^{-1}$ soil, mimicking the penetration depth of the fresh wet-deposited Chernobyl fallout in Sweden in 1986.
Table 3. Averted detriment in terms of CUMULAR, for evacuation alone for 1 y or evacuation + decontamination with 50% efficiency ($S_{\text{decont}} = 0.5$) or 90% efficiency ($S_{\text{decont}} = 0.1$). A full description of the expression of CUMULAR is given in (Rääf et al. 2020).

| Protective action(s)                              | Averted CUMULAR, Averted CUMULAR                                               |
|--------------------------------------------------|--------------------------------------------------------------------------------|
| Evacuation 1 y alone                             | CUMULAR($t = 0$ to 70 y; $S_{\text{decont}} = 1$)—  CUMULAR($t = 1$ to 70 y; $S_{\text{decont}} = 1$) |
| Evacuation 1 y and decontamination with 50% efficiency | CUMULAR($t = 0$ to 70 y; $S_{\text{decont}} = 1$)—  CUMULAR($t = 1$ to 70 y; $S_{\text{decont}} = 0.5$) |
| Evacuation 1 y and decontamination with 90% efficiency | CUMULAR($t = 0$ to 70 y; $S_{\text{decont}} = 1$)—  CUMULAR($t = 1$ to 70 y; $S_{\text{decont}} = 0.1$) |

The highest ground deposition, $A_{\text{dep}}$, that complies with either of the reference levels of 20 mSv yr$^{-1}$ or 1 mSv yr$^{-1}$ for a population evacuated one year after the fallout were computed using the described model. Furthermore, based on a given compliance $A_{\text{dep}}$, averted detriment in terms of CUMULAR was computed by the difference of the unmitigated CUMULAR and the corresponding value for the residual CUMULAR (from time 1 year after the fallout up to $t_{\text{acc}} = 70$ years), as given in table 3. The residual dose, $CED_{\text{res}}$, for each scenario and the combinations of protective measures were computed as well.

3. Results and discussion

3.1. Detriment from unmitigated exposures per unit ground deposition of $^{137}$Cs

The total detriment per unit initial ground deposition of $^{137}$Cs for the four scenarios and three exposure categories of affected populations are given in table 4. Of these scenarios, Swedish NPP release for a population with a hunter lifestyle exhibits the highest detriment in terms of both CUMULAR to the most sensitive cohort (newborn girls at the time of fallout) and cumulative effective dose, $CED(70 \text{ y})$. However, given that the models employ empirical data from the Chernobyl and Fukushima releases and from Swedish NPP source terms, the detriment values per unit $A_{\text{dep}}$ agree within a factor of 2 for a given category. For urban populations, the internal part of the $CED(70 \text{ y})$ will typically constitute 30% of the detriment. Considering that rural Russian residents appear to lie in between urban Swedes and hunters in radioecological transfer, the internal dose contribution would then be about 80–90 mSv, which is almost the same as the external dose contribution (ranging from 67 to 176 mSv per MBq m$^{-2}$ $^{137}$Cs, as given by $CED(70 \text{ y})$ values for exhaustive food restrictions in table 4). This is in agreement with earlier general findings regarding the cumulative doses to populations after the Chernobyl fallout (Howard et al. 2017). The external dose contribution is obviously affected by the different FR ratios, and the highest FR of 1.47 for the Swedish source terms hence exhibits a higher detriment per unit $A_{\text{dep}}$ of $^{137}$Cs compared with the other releases.

3.2. Avertable detriment by means of evacuation and restoration for the different scenarios

Table 5 shows, for the four scenarios, the highest ground deposition of $^{137}$Cs for which a given decontamination efficiency, combined with one-year evacuation to a zone with negligible fallout, can comply with reference dose levels of 20 mSv yr$^{-1}$ and 1 mSv yr$^{-1}$, for
Table 4. Cumulative effective dose and detriment per unit ground deposition of $^{137}$Cs (MBq m$^{-2}$) to the most sensitive cohort at the time of fallout (newborn girls) in terms of CUMLAR of cancer (all organs, i.e. whole-body irradiation) estimated by the model of (Rääf et al 2020) and (Isaksson et al 2019). For reference, values for a fully effective alimentary countermeasure ($S_{\text{aliment}} = 0$) are also given.

| Scenario                                                                 | $S_{\text{aliment}}$ | $\text{CUMLAR}_{\text{WB}}(70\text{ y})$ | $\text{CED}(70\text{ y})$ |
|--------------------------------------------------------------------------|-----------------------|---------------------------------|-----------------|
| Chernobyl fallout in Sweden ($FR = 0.56; T_{\text{eco,long}} = 5.5$ years) |                       |                                 |                 |
| Hunter lifestyle                                                         | 1                     | 8.69                            | 255             |
| Urban residents (relaxed food restrictions)                              | 1                     | 5.61                            | 122             |
| Urban residents (fully effective food restrictions)                      | 0                     | 4.34                            | 87.9            |
| Swedish NPP fallout (median long-term decay) ($FR = 1.47; T_{\text{eco,long}} = 5.5$ years) |                       |                                 |                 |
| Hunter lifestyle                                                         | 1                     | 11.3                            | 302             |
| Urban residents (relaxed food restrictions)                              | 1                     | 7.01                            | 145             |
| Urban residents (fully effective food restrictions)                      | 0                     | 5.63                            | 103             |
| Swedish NPP fallout (slow long-term decay) ($FR = 1.47; T_{\text{eco,long}} = 15$ years) |                       |                                 |                 |
| Hunter lifestyle                                                         | 1                     | 12.5                            | 375             |
| Urban residents (relaxed food restrictions)                              | 1                     | 8.46                            | 218             |
| Urban residents (fully effective food restrictions)                      | 0                     | 7.00                            | 176             |
| Fukushima Dai-ichi Northern trace fallout ($FR = 1; T_{\text{eco,long}} = 3.2$ years) |                       |                                 |                 |
| Hunter lifestyle                                                         | 1                     | 8.61                            | 251             |
| Urban residents (relaxed food restrictions)                              | 1                     | 4.85                            | 105             |
| Urban residents (fully effective food restrictions)                      | 0                     | 3.64                            | 67.3            |

the restoration phase after an accident. Local and regional $^{137}$Cs deposition densities up to 2 MBq m$^{-2}$ can theoretically be restored assuming a median efficiency in decontamination ($S_{\text{decont}} = 0.5$), assuming evacuation of residents for 1 year. For NPP fallout with high FR values, such as the Swedish NPP fallout, the benefit in evacuating per unit $A_{\text{dep}}$ becomes lower than for the other scenarios, resulting in a somewhat lower compliance level of $A_{\text{dep}}$ (here forth referred to as $A_{\text{dep,comp}}$) for the return after 1 year. If strict regulations are also imposed on food distribution to prevent consumption of contaminated foodstuffs, the compliance deposition level, $A_{\text{dep,comp}}$, increases between 50% to 75% compared with a scenario with relaxed food restrictions (represented by a $T_{\text{ag,max}}$ value taken from Scandinavian circumstances after the Chernobyl fallout in 1986).

If decontamination procedures of the affected areas can be made with a high dose reduction efficiency ($S_{\text{decont}} = 0.9$), the $A_{\text{dep,comp}}$ levels become theoretically very much higher, but it is unlikely that residents who hunt, regularly consume game, and/or have a lifestyle with extensive outdoor activities can continue their habits without exceeding the compliance reference dose levels. For people who place a high value on such a lifestyle, this may heavily influence their decision to return to the affected areas, even if extensive measures for reducing external dose have been taken.

Table 6 shows the averted cumulative lifetime attributable risk (CUMLAR) for newborn girls achieved by the protective measures so that the $A_{\text{dep,comp}}$ values in table 5 will comply with reference levels 20 mSv yr$^{-1}$ and 1 mSv yr$^{-1}$, respectively. The largest averted detriment
Table 5. Upper boundary local and regional $^{137}$Cs deposition, $A_{dep,comp}$ (kBq m$^{-2}$) for compliance with reference dose levels 1 mSv y$^{-1}$ and 20 mSv y$^{-1}$, respectively, for new-born girls for four different fallout scenarios. Values refer to one-year evacuation prior to return to the restored area. Affected population: 1: Swedish Hunter; 2: Swedish Urban (relaxed food restrictions); 3: Urban residents (fully effective food restrictions).

| Affected population | Chemobyl fallout ($FR=0.56$; $T_{eco,long}=5.5$ y) | Swedish NPP fallout ($FR=1.47$; $T_{eco,long}=5.5$ y) | Swedish NPP fallout ($FR=1.47$; $T_{eco,long}=15$ y) | Fukushima fallout ($FR=1$; $T_{eco,long}=3.2$ y) |
|---------------------|-----------------------------------------------|------------------------------------------------|------------------------------------------------|------------------------------------------------|
|                     | $A_{dep,comp}$: kBq m$^{-2}$ | $A_{dep,comp}$: kBq m$^{-2}$ | $A_{dep,comp}$: kBq m$^{-2}$ | $A_{dep,comp}$: kBq m$^{-2}$ |
| No decont | 50% | 90% | No decont | 50% | 90% | No decont | 50% | 90% | No decont | 50% | 90% |
| 1 | 719 | 826 | 936 | 494 | 553 | 611 | 488 | 547 | 606 | 675 | 740 | 802 |
| 2 | 1260 | 1993 | 3720 | 969 | 1463 | 2474 | 942 | 1432 | 2455 | 1437 | 2069 | 3193 |
| 3 | 1717 | 3435 | 17160 | 1434 | 2668 | 14340 | 1376 | 2752 | 13760 | 2350 | 4700 | 23500 |

Reference dose level: 20 mSv y$^{-1}$

| Affected population | No decont | 50% | 90% | No decont | 50% | 90% | No decont | 50% | 90% | No decont | 50% | 90% |
|---------------------|-----------|-----|-----|-----------|-----|-----|-----------|-----|-----|-----------|-----|-----|
| 1 | 36.0 | 41.3 | 46.8 | 24.7 | 27.7 | 30.6 | 24.4 | 27.4 | 30.3 | 33.8 | 37 | 40.1 |
| 2 | 63.0 | 99.7 | 186 | 48.5 | 73.2 | 124 | 47.1 | 71.6 | 123 | 71.8 | 103 | 160 |
| 3 | 85.9 | 172 | 858 | 71.7 | 133 | 717 | 68.8 | 138 | 688 | 118 | 235 | 1180 |
is obtained when combining complete depletion of radioecological transfer of fission products into the food chain with highly efficient decontamination.

The residual effective dose, \( CED_{res} \), to a reference person in the four different scenarios is given in table 7, based on upper \( A_{dep,comp} \) levels that comply with reference dose levels of 20 and 1 mSv yr\(^{-1} \), respectively, and an early return after 1 y of evacuation. For the upper \( A_{dep,comp} \) levels, the residual cumulative effective dose, \( CED_{res}(70 \text{ y}) \), will only become less than 100 mSv when food restrictions are imposed. Generally, the values of \( CED_{res}(70 \text{ years}) \) are a factor of 5–9 times higher than the reference dose level for resettlement. Note that, for a given \( A_{dep,comp} \), the \( CED_{res}(70 \text{ years}) \) may be larger for high-efficiency decontamination scenarios, as initial measures are sufficient to suppress the external doses the first year upon return under the reference dose level. However, the cumulative exposures from internal exposures may gradually compensate for this effect by resulting in a higher time-integrated exposure (e.g. for the scenarios with no food restrictions and with relatively short \( T_{eco,long} \), this effect is most evident). Hence, for groups that have a similar lifestyle to the hunters studied in Sweden after the Chernobyl fallout, the residual doses will be significantly higher than 100 mSv in all four model scenarios. To decrease the residual dose, the returnees with such lifestyles must change their habits considerably by, e.g. ceasing to hunt game in the affected region and refraining from consuming berries and mushrooms. This lifestyle-dependent component is also addressed in a study conducted in a related project (Rasmussen et al 2020) and can considerably affect the willingness to return to a restored area.

### 3.3. Time-dependence in avertable detriment by means of evacuation and restoration for the different scenarios

The averted \( CUMULAR \) values for newborn girls and for 30-year-old adults (which is a similar age to the reference persons used for computation of effective dose, ICRP 2007) are plotted in figure 1 for the two long-term protective actions, evacuation and decontamination. Values refer to the unit ground deposition of 1 MBq m\(^{-2} \) \( ^{137}\text{Cs} \) given the four different fallout scenarios described in table 2. In contrast to the upper reference dose compliance levels of ground deposition, \( A_{dep,comp} \), the averted doses by the two protective measures are unrelated to the internal dose contribution. It is evident that the radiological benefit in terms of averted \( CUMULAR_{av} \) decreases rapidly with a delay in implementing decontamination measures. By using nonlinear regression (STATISTICA 6.0\textsuperscript{TM}), one can deduce an effective half-time of this benefit (presented in table 8), ranging from 2.2 years for newborn girls to 3.0 years for adults in the Fukushima-like scenario and 2.4 years for newborn girls and 3.8 years for adults in the Chernobyl-like scenario. For the Swedish NPP nuclide vector, the half-time of the radiological benefit of applying decontamination for girls who were newborn at the deposition event (\( t_{acc} = 0 \)) was as short as 2.1 years for the scenario with a median variant of ecological half-time. However, as expected, significantly longer half-times (4.7 years for newborn girls and 8.4 years for 30-year-old adults) were obtained if instead we assume a value of 15 years for the long-term component of \( r(t) \) (table 2). These radiological benefit half-times will be important for the optimisation and timing of the decontamination measures. This effect is opposite of the so-called discount effect of economic costs in delaying mitigating actions for environmental pollution (e.g. Severa and Bar 1991, Boardman et al 2014), and model outcomes in this study clearly illustrate that the same delay will have an adverse effect in the radiological benefit of decontamination, assuming a fixed ambition level for \( S_{decont} \) (e.g. 50% or 90%). The half-time of radiological benefit, \( T_{1/2,benefit} \), can be translated into a negative annual discount rate of benefit, \( r_{benefit} \), utilising the relationship in equation (2).
Table 6. Averted cumulative lifetime attributable risk over 70 y, \( \text{CUMLAR}_{av}(70\text{ y})(\%) \), for newborn girls for four different fallout scenarios achieved by countermeasures so that exposures one year upon return to the restored area comply with reference dose levels 1 mSv y\(^{-1} \) and 20 mSv y\(^{-1} \), respectively. Values refer to one year of evacuation prior to return.

| Fallout/Scenario | Swedish Hunter (\( \text{FR}=0.56; T_{\text{eco,long}}=5.5 \text{ y} \)) | Swedish NPP fallout (\( \text{FR}=1.47; T_{\text{eco,long}}=5.5 \text{ y} \)) | Swedish NPP fallout (\( \text{FR}=1.47; T_{\text{eco,long}}=15 \text{ y} \)) | Fukushima Dai-ichi (\( \text{FR}=1; T_{\text{eco,long}}=3.2 \text{ y} \)) |
|------------------|--------------------------------------------------|--------------------------------------------------|--------------------------------------------------|--------------------------------------------------|
|                  | \( S_{\text{decont}} \) | \( S_{\text{decont}} \) | \( S_{\text{decont}} \) | \( S_{\text{decont}} \) |
| **No decont**    | 1.80 2.72 3.66 | 1.86 2.53 3.20 | 1.84 2.76 3.67 | 1.47 2.09 2.68 |
| **50\% decont**  | 3.44 7.99 18.71 | 3.14 6.70 13.99 | 3.06 7.61 17.09 | 2.53 5.83 11.7 |
| **90\% decont**  | 3.45 11.28 73.87 | 4.23 11.18 76.95 | 4.08 13.83 91.83 | 3.60 12.2 80.6 |
| **Reference dose level: 20 mSv y\(^{-1} \)** |
| Swedish Hunter   | 0.09 0.14 0.18 | 0.09 0.13 0.16 | 0.09 0.14 0.18 | 0.07 0.10 0.13 |
| Swedish Urban    | 0.17 0.40 0.94 | 0.16 0.34 0.70 | 0.15 0.38 0.85 | 0.13 0.29 0.58 |
| (relaxed food restrictions) |
| Urban residents  | 0.17 0.56 3.69 | 0.21 0.56 3.85 | 0.20 0.69 4.59 | 0.18 0.61 4.03 |
| (fully effective food restrictions) |
Table 7. Residual cumulative effective dose, $CED_{res}(70\ y)$, at the upper boundary level of $^{137}\text{Cs}$ deposition, $A_{dep,comp}$ (kBq m$^{-2}$) for compliance with reference levels 1 mSv y$^{-1}$ and 20 mSv y$^{-1}$, respectively, for four different fallout scenarios. Values refer to one-year evacuation prior to return to the restored area. Affected population: 1: Swedish Hunter; 2: Swedish Urban (relaxed food restrictions); 3: Urban residents (fully effective food restrictions).

| Affected population | Chemobyl fallout in Sweden ($FR=0.56; T_{eco,long}=5.5\ y$) | Swedish NPP fallout ($FR=1.47; T_{eco,long}=5.5\ y$) | Swedish NPP fallout ($FR=1.47; T_{eco,long}=15\ y$) | Fukushima Dai-ichi ($FR=1; T_{eco,long}=3.2\ y$) |
|---------------------|-------------------------------------------------------------|-----------------------------------------------------|-----------------------------------------------------|--------------------------------------------------|
|                     | $CED_{res}(70\ y)$ / mSv                                      | $CED_{res}(70\ y)$ / mSv                             | $CED_{res}(70\ y)$ / mSv                             | $CED_{res}(70\ y)$ / mSv                          |
|                     | Reference dose level: 20 mSv y$^{-1}$                         | Reference dose level: 20 mSv y$^{-1}$                | Reference dose level: 1 mSv y$^{-1}$                | Reference dose level: 1 mSv y$^{-1}$              |
| No decont 50% 90%   | 119 123 127                                                  | 128 149 146                                       | 152 149 146                                        | 135 140 145                                      |
| 1                   | 154 162 171                                                  | 119 123 127                                       | 128 149 146                                       | 135 140 145                                      |
| 2                   | 110 117 134                                                  | 90.8 94.9 103                                     | 156 145 120                                       | 108 110 118                                      |
| 3                   | 97.6 97.6 97.6                                              | 82.7 82.7 82.7                                   | 179 179 179                                       | 103 103 103                                      |
| 1                   | 7.69 8.11 8.53                                              | 5.93 6.14 6.36                                   | 7.62 7.45 7.28                                    | 6.73 6.99 7.26                                   |
| 2                   | 5.50 5.86 6.71                                              | 4.54 4.75 5.17                                   | 7.82 7.23 6.02                                    | 5.39 5.51 5.88                                   |
| 3                   | 4.88 4.88 4.88                                              | 4.13 4.13 4.13                                   | 8.94 8.94 8.94                                    | 5.13 5.13 5.13                                   |
Table 8. Decay time (y) of radiological benefit in terms of $CUMLAR_{av}(70 \text{ years})$ and corresponding annual benefit discount (%) for newborn girls and for 30 y old adults (average over men and women) in four different fallout scenarios.

| Cohort                      | Chernobyl fallout ($FR = 0.56; T_{eco,long} = 5.5 \text{ years}$) | Swedish NPP fallout ($FR = 1.47; T_{eco,long} = 5.5 \text{ years}$) | Swedish NPP fallout ($FR = 1.47; T_{eco,long} = 15 \text{ years}$) | Fukushima fallout ($FR = 1; T_{eco,long} = 3.2 \text{ years}$) |
|-----------------------------|-----------------------------------------------------------------|------------------------------------------------------------------|------------------------------------------------------------------|------------------------------------------------------------|
| Newborn girls               | 2.4 ± 0.11 ($-25\%$)                                            | 2.1 ± 0.12 ($-28\%$)                                            | 4.7 ± 0.53 ($-14\%$)                                            | 2.2 ± 0.02 ($-27\%$)                                       |
| Adults                      | 3.8 ± 0.2 ($-17\%$)                                             | 3.3 ± 0.23 ($-19\%$)                                            | 8.4 ± 0.53 ($-7.9\%$)                                           | 3.0 ± 0.0 ($-21\%$)                                        |
Figure 1. Radiological benefit in terms of averted cumulative lifetime attributable risk (CUMLAR$_{av}$) by evacuation and decontamination (median variant of $S_{decont}$ (50%)) as a function of return time, for urban inhabitants (newborn girls (left) and 30-year-old adults (right) at the time of fallout). Values refer to a fallout nuclide vector containing 1 MBq m$^{-2}$ $^{137}$Cs. Top: Chernobyl fallout in Sweden ($FR = 0.56; T_{eco,long} = 5.5$ years), Next top: Fukushima Dai-ichi Northern trace fallout ($FR = 1; T_{eco,long} = 3.2$ years), Next bottom: Swedish NPP fallout ($FR = 1.47; T_{eco,long} = 5.5$ years) and bottom: Swedish NPP fallout ($FR = 1.47; T_{eco,long} = 15$ years).
The $T_{1/2,\text{benefit}}$ values found here for newborn girls can thus be translated into the annual discount rate of benefit loss of decontamination between 14% and 28%. For adults, this discount rate is somewhat lower, ranging from 8% to 21% annual loss in benefit.

Referring to table 4 and figure 1, for the scenario Swedish NPP fallout ($FR = 1.47$; $T_{\text{eco,long}} = 5.5$ y) as an example, a $CUMLAR(70)$ of about 7% will be incurred to newborn girls at the time of fallout if no protective measures (except for the initial measures of iodine prophylaxis and sheltering) are implemented. Evacuation of newborn girls for two y before resettlement will avert about 4.5% cumulative lifetime attributable risk, and if decontamination with efficiency of 50% is implemented, an additional 0.9% of $CUMLAR$ is averted. Setting $f_{\text{shield}} = 0.2$, representing residential areas that provide better shielding than in the reference cases, will reduce the averted $CUMLAR(70)$ for both evacuation and decontamination but will have little impact on the proportionality between these two. Thus, shielding properties of the houses do not significantly influence the time sensitivity of the countermeasure in terms of relative gain in radiological benefit with decontamination.

Generally, the uncertainties of the estimates for upper compliance ground deposition $A_{\text{dep,comp}}$ of $^{137}$Cs, averted $CUMLAR(70)$, and residual dose $CED$ will be of the same order as described in the uncertainty analysis of the model in equation (1), presented in (Räaff et al 2020), in which the 5th to 95th percentile levels were about a factor of 2 lower and higher, respectively, compared with the central estimates. Given the large uncertainties in these estimates, the $A_{\text{dep,comp}}$ levels in practice must be decreased by 50% to ensure that the 95th percentile annual effective dose of the returnees upon return from evacuation does not exceed 20 mSv yr$^{-1}$.

To estimate the amount of soil and additional waste generated by decontamination measures in order to achieve a given $S_{\text{decont}}$, further generalisations are needed as $S_{\text{decont}}$ highly depends on the specified decontamination measures and the time of its implementation. It must also be noted that $S_{\text{decont}}$ in this model represents a macroscopic level of general decrease in external dose rate (combining occupancies indoors and outdoors), and for specific surfaces, the actual dose rate reduction may vary. Given the aforementioned large variation in the soil migration processes, rough estimates using generic values of the diffusion coefficient and partition coefficient, $K_d$, can be used to give an indication of the waste generation. It is especially important to estimate the time dependence in the volume of waste per unit decrease in $S_{\text{decont}}$, as this will have a bearing not only on the direct costs of handling the waste but also on the environmental impact of waste generation.

4. Conclusions

- Model estimates show that the most sensitive group (newborn girls) will benefit the most from decontamination in terms of averted cumulative lifetime attributable risk ($CUMLAR$), provided that decontamination combined with prompt evacuation can be achieved as early as possible (within a few years). The averted detriment for newborn girls is almost a factor of five times higher than for adults.

- For both infants and adults, the radiological benefit in terms of averted $CUMLAR$ is much higher for evacuation alone than the contributing averted $CUMLAR$ from decontamination, even when using best available techniques ($S_{\text{decont}} = 0.1$). Delaying decontamination (using
best available techniques, assuming a 90% external dose rate reduction) more than 5 years after the fallout will result in avertable \( \text{CUMLAR(70 years)} \) that is less than 30% of that attained with 5 y evacuation alone, regardless of the age of the cohorts. Hence, there is a time window of up to 3–4 years for implementing decontamination measures that provide a significant radiological impact in terms of averted lifetime cumulative radiation detriment in comparison with evacuation.

- The decrease in the averted \( \text{CUMLAR(70 years)} \) with time after fallout translates into an annual loss of benefit of ~25%, which should be considered in the cost-benefit analysis of performing such measures.
- For one-year evacuation, a reference dose level of 20 mSv yr\(^{-1}\) for resettlement, and a median level of achieved dose reduction by decontamination (50%), relatively high ground deposition (up to 2–3 MBq m\(^{-2}\) of \(^{137}\text{Cs}\)) can be manageably restored in several types of potential fallout scenarios. A residual lifetime effective dose on the order of 100 mSv will then be anticipated.
- For residents whose lifestyle involves, e.g. hunting, fishing, and consumption of wildlife food products, residual doses (per 1 MBq m\(^{-2}\) \(^{137}\text{Cs}\) ground deposition) after one-year evacuation and 50% external dose reduction by decontamination will still be significantly higher than 100 mSv. The extent to which such groups are willing to accept to return to a restored area, given the potential restrictions imposed on their lifestyles, should be investigated.

In all, the results may raise concern regarding the justification of large-scale decontamination in some situations, especially when the general long-term decrease in half-time of the ambient dose rate in the affected region is relatively short \( T_{\text{eco.long}} < 10 \text{ years or less} \). Decontamination may be less meaningful if evacuation alone accounts for the largest avertable dose, which may be the case when the ecological half-time is rather short. In some situations, decontamination may perhaps be more justifiable based on psycho-social issues rather than radiological impact in terms of late stochastic radiation effects. Even though this is an issue for decision makers and relevant stakeholders, the radiation safety experts nevertheless need to consider the sometimes very limited radiological benefit of decontamination and, especially, its rapid benefit discount with time for small children.

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