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Radioecological transfer factors for Nordic subpopulations for assessment of internal committed dose from atmospheric fallout of radiocaesium

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Abstract

Measurements of whole body burden of radiocaesium have for a long time been performed in Finland, Norway and Sweden, and used to estimate the impact of the nuclear weapons fallout, as well as the Chernobyl accident, on the populations in the three countries. During this work, it has become evident that the existing data can be analysed to a greater extent than has been possible within the frames of this project.
For example, the issue of modelling uptake of caesium from the food chain, based on ecological half-life or aggragated transfer function, Tag(t), in areas where there are reminiscences of previous fallout occasions of the same extent as the present fallout, has to be further investigated. Modelling is a useful tool to check our state of knowledge and reveal gaps in data, which have to be considered in case of future fallout events. This also emphasizes the need for maintaining continuous time-series of measurements on atmospheric deposition as well as the body-burdens in various populations, especially among those known in advance to be sensitive to radioecological transfer of fission products.

Key words

whole-body measurements, fallout, radiocaesium, aggregated transfer function

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1 INTRODUCTION

Radioecological modelling of transfer of fission products have been carried out since the 1960s at the time when the global nuclear weapons fallout reached its peak (Aarkrog, 1979; Westerlund et al.,1987; Rääf, 2000). UNSCEAR (1977) presented generic models for the transfer of ^{137}Cs from a continuous atmospheric deposition, that could be used for international comparison between the aggregate transfer of the radionuclide in different countries. It was also observed that the radioecological transfer among different subgroups varied widely depending on living conditions, with reindeer herders being identified as a particularly sensitive group in terms of radioecological transfer (Rääf et al., 2006a; Rääf et al., 2006b).

At the event of the Chernobyl accident in 1986, many scientists attempted to use transfer models to predict the long-term body burdens of fission products among various population groups. Due to the difference in the time-pattern of the atmospheric release it was not trivial to translate the transfer models obtained from nuclear weapons fallout to the single fallout event of the Chernobyl release (e.g. Aarkrog, 1988, Rääf, 2000). The deposition data from pre-Chernobyl were scarce and mainly from the few air sampling stations aimed for nuclear weapons fallout monitoring, whereas the Chernobyl fallout was often mapped by aerial surveys, which enabled a high geographical resolution in the deposition data compared with the pre-Chernobyl fallout. Hence, the whole-body counting monitoring programs conducted in many countries consisted of data which potential hitherto has not been fully exploited when adjusting the transfer models to experimental observations.

In connection with a radiation environment database project commission by the Swedish radiation protection authority in the beginning of the 2000s, an attempt to develop a model that could be used both for single and continuous fallout of ^{137}Cs was made (Rääf et al., 2006a; 2006b). However, this model has not been fully validated against data from other countries, such as the Nordic countries or in the affected regions of former USSR, and neither has any difference in the radioecological transfer between the countries been quantitatively assessed over long-term. This work is an extension of the method used by Rääf et al. (2006a), where whole body burden of ^{137}Cs, measured by whole body counting, is related to the county average deposition density. By classifying each measured individual into various subpopulations based on living habits, e.g. farmers, hunters or urban populations, it is possible to assign an ecological transfer factor for each of these subpopulations. The method was successfully applied in a recent publication by Tondel et al. (2017), regarding Swedish hunters in the counties most affected by fallout from the Chernobyl accident.

This work has a clear connection to the NERIS roadmap (see https://www.eu-neris.net/library/sra/157-first-neris-roadmap-november-2017/file.html). For example, regarding challenges and achievements in dose models, the vison is to have “A suite of models … based on all available data…”. It is also stated that these models should take into account the real behaviour of the population, which in this proposal is reflected by the aim to find transfer factors for various subpopulations. The NERIS roadmap also identifies challenges in environmental models, for example identification of regional parameters that characterize the transfer raw-to-product. This proposal could be viewed as an extension to this since it aims at studying transfer raw-to-human in terms of the varying food consumption between subpopulations.

The aim of this project was to consolidate the existing data on whole-body burden and aerial ^{137}Cs fallout mapping in the Nordic countries. The outcome will be a radioecological transfer model that encompasses the observations from several Nordic countries and includes data found in the most recent surveys. This model can then be used to quantitatively assess projected dose commitments to the subgroups and to detect whether common traits in the transfer can be found in some categories over the national borders, such as hunters and reindeer herders. These data can be used as a simple but comprehensive guidance for Nordic decision makers.

2 A MODEL FOR ASSESSMENT OF INTERNAL RADIATION DOSE

Based on the work by Rääf et al. (2006a, 2006b), the effective internal dose rate from the internal sources ^{134}Cs and ^{137}Cs, dominating the long-term dose after fallout, can be described by empirical relations between the effective internal dose rate and the regional-average of the ground deposition density, $A_{esd(county)}$ of ^{137}Cs (kBq m^{-2}). For ^{137}Cs the relation can be written:

$$\dot{E} = \chi \cdot A_{esd(county)} \cdot T_{ag(max)} \cdot \left[\left(1 - e^{-\frac{ln2}{t_1}(t-t_0)}\right) \cdot c_1 \cdot e^{-\frac{ln2}{t_2}(t-t_0)} + c_2 \cdot e^{-\frac{ln2}{t_3}(t-t_0)}\right] \cdot k_s \qquad \text{[Eq. 1]}$$

Since the effective internal dose rate is estimated from the deposition of ^{137}Cs, a similar estimation for ^{134}Cs must take into account the difference in physical half-life, as well as the relative abundance of the two Cs-isotopes in the fallout. For ^{134}Cs, the above expression is then multiplied by a correction factor that can be written as:

$$e^{\frac{ln2(2.06-30.2)}{2.06 \cdot 30.2}} \cdot FR \qquad \text{[Eq. 2]}$$

where *FR* is the initial activity ratio of ^{134}Cs by ^{137}Cs in the fallout.

The product

$$A_{esd(county)} \cdot T_{ag(max)} \cdot \left[\left(1 - e^{-\frac{ln2}{t_1}(t-t_0)}\right) \cdot c_1 \cdot e^{-\frac{ln2}{t_2}(t-t_0)} + c_2 \cdot e^{-\frac{ln2}{t_3}(t-t_0)}\right] \cdot k_s \qquad \text{[Eq. 3]}$$

is the whole-body activity concentration of ^{137}Cs in Bq kg^{-1} at time t after the onset of fallout, where k_s accounts for an observed difference in whole body concentration between men and women. Based on observations from the Swedish whole-body measurement database (Rääf et al., 2006a), the reference value for k_s is 0.61 for adult females and 1.00 for adult males. The transfer of caesium is thus described by the aggregated transfer function $T_{ag}(t)$ [Bq kg^{-1}/(kBq m^{-2})], where

$$T_{ag}(t) = T_{ag(max)} \cdot \left[\left(1 - e^{-\frac{ln2}{t_1}(t-t_0)}\right) \cdot c_1 \cdot e^{-\frac{ln2}{t_2}(t-t_0)} + c_2 \cdot e^{-\frac{ln2}{t_3}(t-t_0)}\right] \cdot k_s \qquad \text{[Eq. 4]}$$

Hence, the aggregated transfer function gives the whole-body activity concentration in Bq kg^{-1} of ^{137}Cs per unit ground deposition density (regional-average, units kBq m^{-2}).

The conversion coefficient, χ, between the whole body concentration and the effective dose rate (mSv y^{-1} per Bq kg^{-1}), has been calculated by Falk et al. (1991), based on a computational model by Leggett et al. (1984). The conversion coefficients depend on body mass, w, and are given by:

$$\chi(^{134}Cs) = 0.00164 \cdot w^{0.188} \qquad \text{[Eq. 5]}$$

and

$$\chi(^{137}Cs) = 0.0014 \cdot w^{0.111} \qquad \text{[Eq. 6]}$$

for ^{134}Cs and ^{137}Cs, respectively.

The parameter $T_{ag(max)}$ (Bq kg^{-1} per kBq m^{-2}) provides an approximate measure of the maximum aggregated transfer to the human body. The parameters c_i, and t_i account for the time dependence of the ecological transfer, and t_0 is the time of onset of fallout. These parameters have been derived by fitting the expression to experimental data (Rääf et al., 2006b).

In summary, the expressions for the two Cs-isotopes read:

$$\dot{E}(^{134}Cs) = 0.00164 \cdot FR \cdot e^{\frac{ln2(2.06-30.2)}{2.06\cdot 30.2}} \cdot A_{esd(county)} \cdot T_{ag(max)} \cdot \left[\left(1 - e^{-\frac{ln2}{t_1}(t-t_0)}\right) \cdot c_1 \cdot e^{-\frac{ln2}{t_2}(t-t_0)} + c_2 \cdot e^{-\frac{ln2}{t_3}(t-t_0)}\right] \cdot w^{0.188} \cdot k_s$$

and

$$\dot{E}(^{137}Cs) = 0.0014 \cdot A_{esd(county)} \cdot T_{ag(max)} \cdot \left[\left(1 - e^{-\frac{ln2}{t_1}(t-t_0)}\right) \cdot c_1 \cdot e^{-\frac{ln2}{t_2}(t-t_0)} + c_2 \cdot e^{-\frac{ln2}{t_3}(t-t_0)}\right] \cdot w^{0.111} \cdot k_s$$

3 METHODS

Data from measurements of whole body activity of ^{137}Cs and deposition density were compiled from Finland, Norway and Sweden. Where possible, the data on whole body burden were partitioned into sub populations representing reindeer herders, hunters, farmers, rural non-farmers and urban residents. The model for internal radiation dose, presented above, was applied to the data from the three Nordic countries.

Measurements of deposition density in Sweden, following the Chernobyl accident, have been reported as surface equivalent deposition (Finck, 1992). This quantity equals the areal deposition density (Bq m^{-2}) of an infinite plane surface source, producing the same primary photon fluence rate at a height of 1 m above the surface, as the actual depth distributed source. The equivalent surface deposition will thus be lower than the actual inventory of the radionuclide, unless the radionuclide is distributed only at the surface. When comparing the values of T_{ag}, the quantity measured to yield $A_{esd(county)}$ therefore has to be taken into account. Based on Swedish observations described in e.g. Jönsson et al. (2017), the relationship between the A_{esd}, defined for the fresh fallout of wet deposition in Sweden and total activity deposition density (Bq m^{-2}) is approximately 1/1.6=0.625.

4 RESULTS AND DISCUSSION

4.1 Swedish data

A database, consisting of data from body burden studies of various Swedish populations, has been compiled (Rääf, 2000; Rääf et al., 2006a; Rääf et al., 2006b), based on measurements conducted by various Swedish authorities and universities (Table 1). The database contains data on age, weight, residence, date for whole body counting, occupation, and body burden of ^{137}Cs and ^{134}Cs (as well as ^{40}K). The database has been anonymized regarding the participating subjects. The place of residence is recorded as dwelling coordinates, as well as by the name of the home municipality and county. Data in the database have been provided by the Swedish Defense Research Agency (FOI, presently the Swedish Defence Research Agency, FOA), the National Radiation Protection Authority (SSI, presently the Swedish

Radiation Safety Authority, SSM), and the Departments of Radiation Physics at the universities in Lund, Malmö, Gothenburg and Umeå.

Occupations that are associated with dietary habits, known to be particularly sensitive to the transfer of radioactive caesium (Ågren, 1998) have been identified and used to categorize the subjects into the following sub-populations: *i*) reindeer herders, *ii*) hunters, *iii*) farmers, *iv*) rural non-farmers and *v*) urban residents (Table 1). Data on these sub-populations have been used to estimate transfer parameters. The parameter values for Swedish sub-population applicable for the expression of $T_{ag}(t)$ (Eq. 4) are found in Table 2.

Table 1. Sub-populations and origin of data for assessment of transfer parameters in Sweden. Details can be found in Rääf et al., 2006a and Rääf et al., 2006b.

Category	Residence	Measurement period
Reindeer herders	Västerbotten county	1986-2001
	Norrbotten county	1992, 1997
	Jämtland county	1992-2001
	Middle of Sweden	1965-1976
Hunters	Counties most affected by the Chernobyl fallout	1994-2001
Farmers	Gävle municipality	1986-1998
Rural non-farmers	Gävle municipality	1986-1998
	Västerbotten county	1991-1996
	Norrbotten county	1991-1996
Urban residents	Stockholm	1959-2001
	Gothenburg	1986-1989
	Lund	1960-1994
	Malmö	1986-1994, 2002

Table 2. Parameter values in aggregated transfer function, applicable for Sweden (Rääf et al., 2006a; Rääf et al., 2006b).

Category	$T_{ag(max)}$ (^{137}Cs) [Bq kg^{-1}/(kBq m^{-2})]	t_1 [y]	t_2 [y]	t_3 [y]	c_1 [-]	c_2 [-]
Reindeer herders	200	2.0	2.0	15	1.0	0.10
Hunters	29.3	1.1	1.2	30	0.9	0.11
Farmers	11.0	1.0	1.0	15	0.9	0.10
Rural non-farmers	9.0	1.1	1.2	30	0.9	0.11
Urban residents	11.0	1.0	0.75	15	1.0	0.10

A scenario-based study of the model (Eq. 1) have been published by Isaksson et al. (2019). In this study, the model was extended to also account for the external exposure from $^{134,137}Cs$ and short-lived radionuclides from fallout caused by release from a nuclear power plant accident. However, the internal dose was found to give a larger contribution to the yearly effective dose than the external exposure at 18 years after the Chernobyl fallout. It is therefore reasonable to assume that the model given here (Eq. 1) can be considered validated by Isaksson et al. (2019), where the extended model was applied to measurements from western Bryansk region in Russia (Thornberg et al., 2005; Bernhardsson et al., 2011). From Fig. 1 it can be seen that the predictions agree fairly well with the effective doses to the studied populations in the Russian villages.

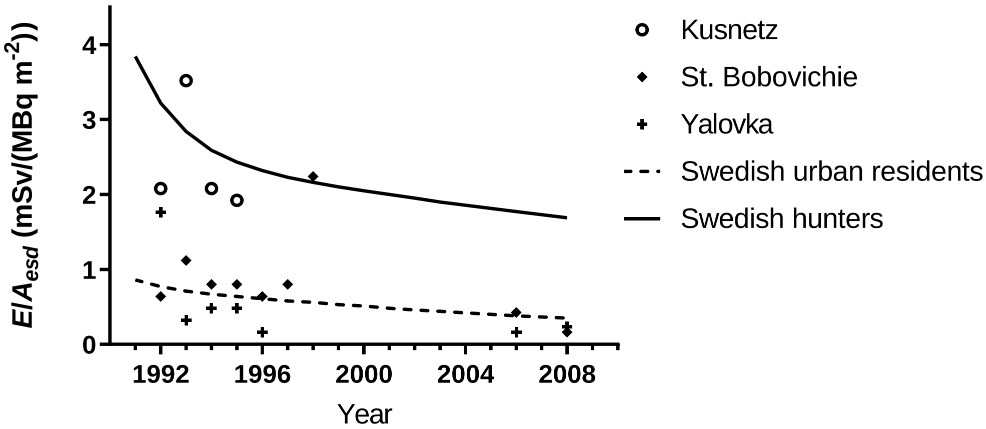

Figure 1. Yearly effective dose per unit ^{137}Cs areal activity deposition density (Data from Isaksson et al., 2019).

4.2 Finnish data

The data from Finland, considered in this report, are measured whole-body activities of ^{134}Cs and ^{137}Cs in Lapland during 1987-2005 (Rahola et al. 1989; Rahola et al., 1991; Rahola et al., 1993; Radiochemistry Unit, University of Helsinki, unpublished). Data are given as whole-body activity in Bq and body mass in kg. Measurements have been carried out in spring and autumn for the years 1987-1989, in spring for the years 1990-1993, 1995, 1997, 2002 and 2005, and in autumn 1994. However, the dates of each individual measurement is not recorded in the above references and the measurement dates are assumed to be 1 April and 1 October for spring and autumn, respectively.

Data on sex, occupation and reindeer herding cooperative are given for each measured subject, and based on the reindeer herding cooperative, each subject has been assigned to one of the two municipalities Utsjoki and Inari (Figure 2). The whole-body activity concentration of ^{137}Cs for all subjects of each sex has been averaged for each of the two municipalities, and $T_{ag}(t)$ calculated by dividing the whole-body activity concentration by the mean deposition density at the respective municipality. For women, each $T_{ag}(t)$ is divided by 0.61 (Eq. 3) to allow a direct comparison with $T_{ag}(t)$ for males.

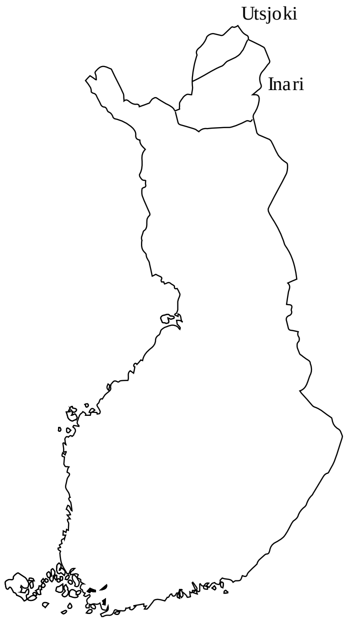

Figure 2. Map of Finland with the municipalities Utsjoki and Inari outlined.

The mean Chernobyl deposition density of ^{137}Cs has been reported to be 1.64 kBq m^{-2} at Utsjoki and 1.63 kBq m^{-2} at Inari. The measurements were made in several campaigns during 1986-1987, and the data have been decay corrected to 1 October 1986 (Arvela et al., 1990). The reported deposition values for ^{137}Cs were based on dose rate measurements and spectrometric measurements of ^{134}Cs, and from these measurements, an empirically determined calibration factor was determined, relating the dose rate to the deposition density of a plane surface source. Since the calculations were based on ^{134}Cs, the deposition density of nuclear weapons fallout ^{137}Cs could be subtracted from the total deposition density of ^{137}Cs.

From Fig. 3 it is seen that the general behaviour of $T_{ag}(t)$ for men and women agree well with the model found for Swedish reindeer herders, although the Finnish data show higher values during the first 5-10 years after the deposition event, assumed to be 27 April 1986, and a possible correction for this is presented below (Fig. 5). The decreasing trend, however is almost the same, which implies that the half-times t_2 and t_3 may be valid also for the Finnish populations.

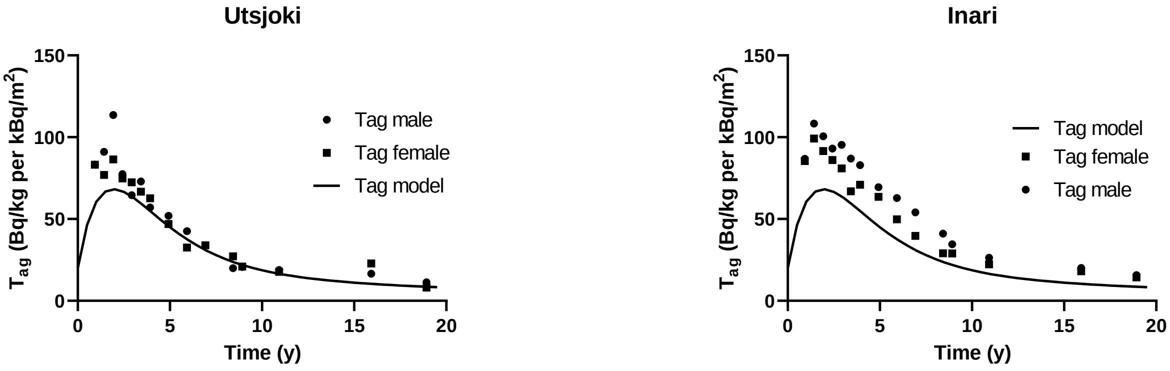

Figure 3. Mean values of $T_{ag}(t)$ for men and women at Utsjoki and Inari municipalities. Values for women are divided by 0.61 (Eq. 3) to allow a direct comparison. In the figures are also shown the corresponding $T_{ag}(t)$ calculated from the model with parameters applied for Swedish reindeer herders.

The data did not allow a fit to Eq.4 and some of the parameters were therefore kept from the Swedish data, given in Tab. 2. The fitted parameters for the Finnish data set were $T_{ag(max)}$ and t_1, and the best fit was provided by the parameter values given in Tab. 3. The data and fitted curves are shown in Fig. 4. Although the value of $T_{ag(max)}$ is lower for the Finnish data, compared to the Swedish, the shorter half-time t_1 will cause the curve to have a higher maximum than for the Swedish reindeer herders.

Table 3. Parameter values, and goodness of fit (expressed as R^2) from fits of Eq.4 to Finnish data, where two parameters were allowed to vary.

Municipality		$T_{ag(max)}$ (^{137}Cs) [Bq kg^{-1}/(kBq m^{-2})]	t_1 **[y]**	R^2
Utsjoki	Males	190	0.9	0.94
	Females	192	1.2	0.96
Inari	Males	334	2.5	0.98
	Females	240	1.6	0.97

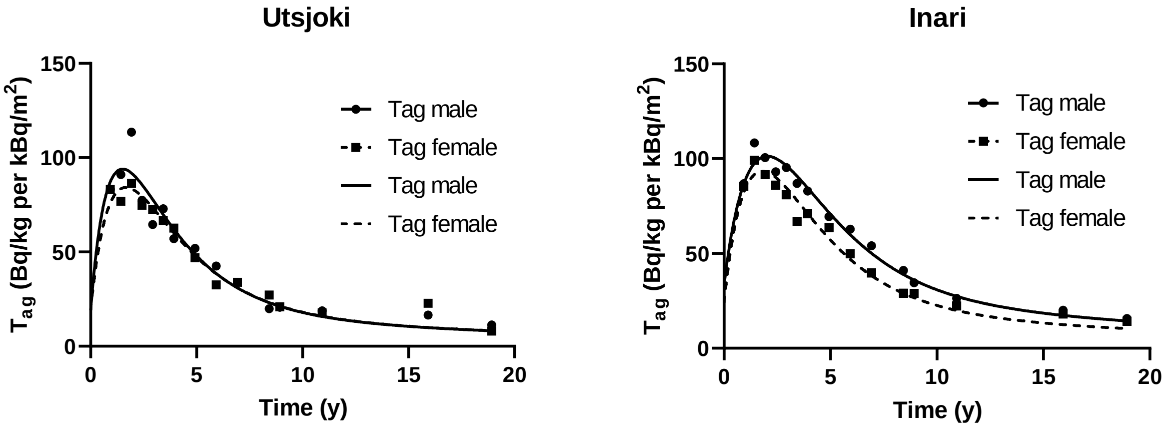

Figure 4. Mean values of $T_{ag}(t)$ for men and women at Utsjoki and Inari municipalities. Values for women are divided by 0.61 (Eq. 3) to allow a direct comparison. In the figures are also shown the corresponding $T_{ag}(t)$ calculated from the model with parameters fitted to the data.

The data on whole body activity concentration, used to determine $T_{ag}(t)$ shown in Figs 3 & 4, are in fact due to both nuclear weapons fallout ^{137}Cs and ^{137}Cs of Chernobyl origin. Since the whole body activity of ^{134}Cs is also included in the Finnish dataset, it was possible to estimate the contribution to the whole body activity from these two sources. The mean activity concentration for Utsjoki males and females, and Inari males and females was decay corrected to 27 April 1987 and the initial activity ratio was assumed to be 0,56. Based on this, it was found that 75% of the body burden of ^{137}Cs in males and females in Utsjoki is of Chernobyl origin, whereas the corresponding fractions in Inari are 65% and 63% for males and females, respectively. Adjusting the data by these fractions, $T_{ag}(t)$ shown in Figs 5 & 6 can be found for the two Finnish municipalities. The fitted parameter values are given in Tab. 4.

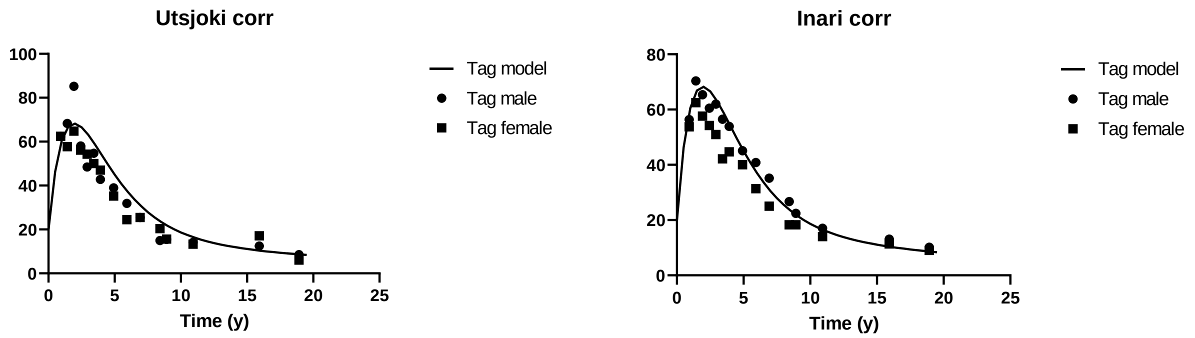

Figure 5. Mean values of $T_{ag}(t)$ for men and women at Utsjoki and Inari municipalities. Values from Fig. 3, recalculated and based on only the fraction of Chernobyl derived ^{137}Cs. In the figures are also shown the corresponding $T_{ag}(t)$ values calculated from the model with parameters applied for Swedish reindeer herders.

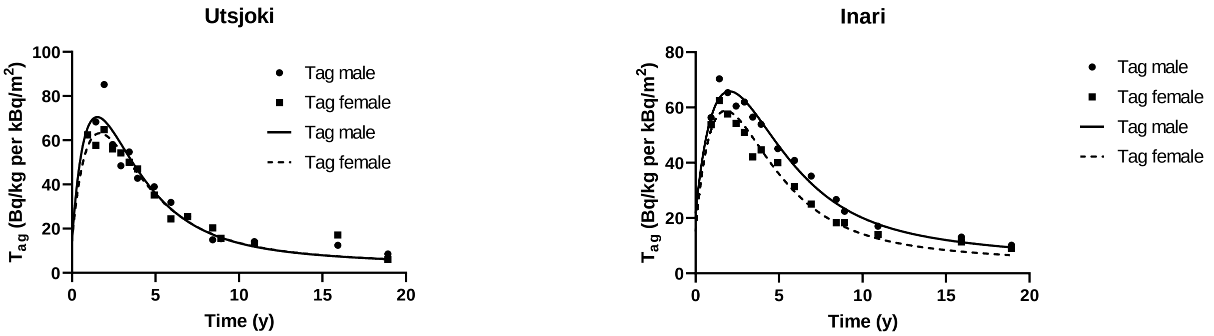

Figure 6. Mean values of $T_{ag}(t)$ for men and women at Utsjoki and Inari municipalities. Values from Fig. 3, recalculated and based on only the fraction of Chernobyl derived ^{137}Cs. In the figures are also shown the corresponding $T_{ag}(t)$ values calculated from the model with parameters fitted to the data.

Table 4. Parameter values, and goodness of fit (expressed as R^2) from fits of Eq.4 to Finnish data, taking only ^{137}Cs of Chernobyl origin into account, where two parameters were allowed to vary.

Municipality		$T_{ag(max)}$ (^{137}Cs) [Bq kg^{-1}/(kBq m^{-2})]	$\boldsymbol{t_1}$ **[y]**	$\boldsymbol{R^2}$
Utsjoki	Males	143	0.9	0.94
	Females	144	1.2	0.96
Inari	Males	217	2.5	0.98
	Females	152	1.6	0.97

Leppänen et al., (2011) found that the ecological half-life of ^{137}Cs in Finnish reindeer herders in northern Finland were 5.5 ± 1.3 y for males and 4.4 ± 0.9 y for females. This is in good agreement with the model fits in this work.

4.3 Norwegian data

The data from Norway consist of measured whole-body activities of ^{137}Cs in various regions of Norway during 1965-2017 (Skuterud and Thørring, 2012; 2015; DSA, unpublished). Data on measured ^{134}Cs are also available, but has not been considered here. Individual data are stored in a database at DSA, but for this study only mean values of whole-body activity in Bq and body mass in kg, for each region and measurement date, was used. The nationwide data on deposition in Norway are based on an average value from 4 soil samples taken per municipality in June 1986 (Backe et al., 1986). Thus, the data have limitations on precision and spatial variability compared to data from Sweden and Finland. $T_{ag}(t)$ was calculated by dividing the whole-body activity concentration by the mean deposition density at the respective area. For women, each $T_{ag}(t)$ is divided by 0.61 (Eq. 3) to allow a direct comparison with $T_{ag}(t)$ for males.

Data are available for reindeer herders from Kautokeino, Røros and Snåsa regions (Figure 7), as well as for a group of rural inhabitants from the municipalities Øystre Slidre, Snåsa and Sel, and the region "Upper Valdres" (Øystre Slidre is one of the municipalities of Upper Valdres) (Figure 7).

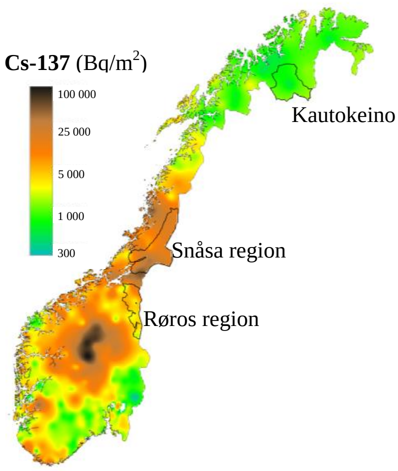

Figure 7. Map of Norway, showing the reindeer herding areas studied and the total ^{137}Cs fallout as of June 1986 (nuclear weapons and Chernobyl fallout). The municipalities Sel and Øystre Slidre are located in the darkest area in southern Norway.

Calculated $T_{ag}(t)$ for the Kautokeino reindeer herders are shown in Figure 8 and it is obvious that the $T_{ag}(t)$ values are much higher than corresponding values for Swedish reindeer herders post-Chernobyl. However, the model for Swedish reindeer herders were parameterized using data from regions where the fallout from nuclear weapons tests made only a minor contribution to the body burden of ^{137}Cs. Kautokeino, however, received minor Chernobyl fallout compared to Central and Southern Norway (Figure 7). Therefore, Figure 8 also shows model calculations using data from reindeer herders in the county Härjedalen in Sweden where the contribution from nuclear weapons fallout is about equal to the fallout from the Chernobyl accident (dashed line). In this case the parameter $T_{ag(max)}$ equals 2000. This example illustrates that the nuclear weapons fallout was still a significant part of the ^{137}Cs contamination in Kautokeino in 1987, as observed in the increase in whole-body concentrations from March 1986 (one month before the Chernobyl accident) to spring 1987 (Skuterud and Thørring, 2015). Another observation reflecting the significance of the remaining nuclear weapons fallout in Kautokeino is the fact that levels here peaked in 1987 (Figure 8) and not in 1988 as in the Swedish and Finnish datasets (see figures above; those cases reflecting the time required for humans to reach maximum values after ingestion of contaminated food commences). The role of already existing contamination must be taken into account when considering a possible future fallout occasion, where the fallout from the Chernobyl accident will contribute to the uptake in a similar manner – especially in the most contaminated areas.

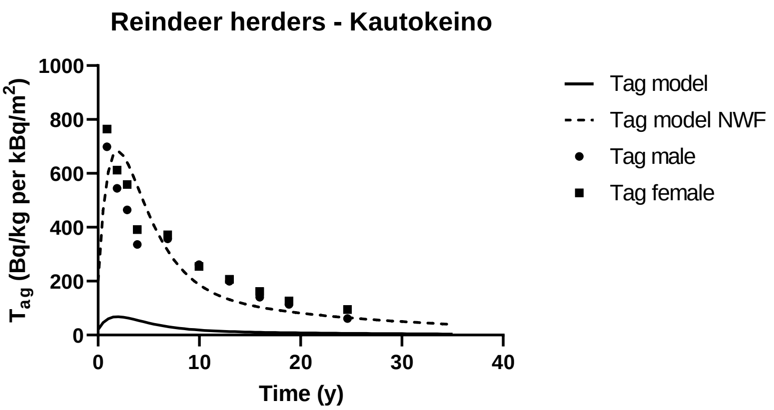

Figure 8. Mean values of $T_{ag}(t)$ for men and women in Kautokeino. Values for women are divided by 0.61 (Eq. 3) to allow a direct comparison. In the figures are also shown the corresponding $T_{ag}(t)$ calculated from the model with parameters applied for Swedish reindeer herders (solid line), as well as for Swedish reindeer herders from the county Härjedalen (dashed line).

Calculated $T_{ag}(t)$ for the reindeer herding regions Røros and Snåsa are shown in Figure 9 and are compared to $T_{ag}(t)$ for the Swedish reindeer herders with model parameters given in Table 2. Initially low values of $T_{ag}(t)$ at the Snåsa region are due to the extensive countermeasures that were found to reduce the body burden in the measured subjects about a factor of ten (Skuterud & Thørring, 2012).

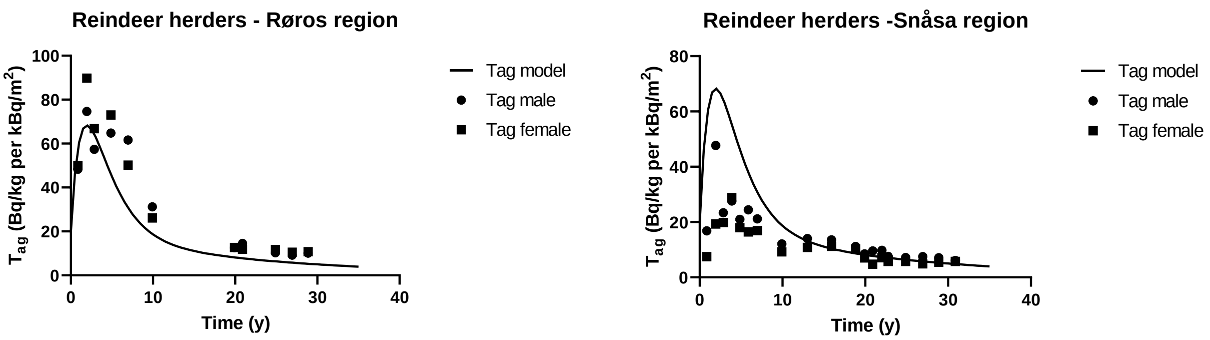

Figure 9. Mean values of $T_{ag}(t)$ for men and women at Røros and Snåsa regions. Values for women are divided by 0.61 (Eq. 3) to allow a direct comparison. In the figures are also shown the corresponding $T_{ag}(t)$ calculated from the model with parameters applied for Swedish reindeer herders.

The calculated $T_{ag}(t)$ values in the Røros region are in between the Kautokeino and Snåsa values, suggesting a combination of different $T_{ag(max)}$. The higher values at the end of the measurement period could be explained by contribution from nuclear weapons fallout. This issue needs to be further discussed.

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Data were also available for a mixed group consisting of rural inhabitants in Øystre Slidre, Snåsa, Sel and Upper Valdres. The calculated $T_{ag}(t)$ for these was compared to the model for rural non-farmers, farmers and hunters, parameterized according to Table 2, and shown in Figure 10. The observed T_{ag}-values appear to be somewhat lower than Swedish model predictions for rural inhabitants, although on the same order of magnitude.

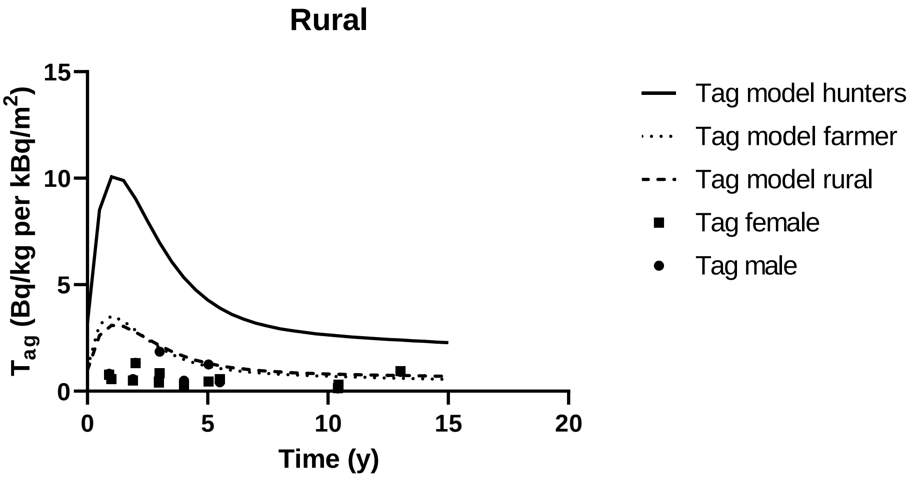

Figure 10. Mean values of $T_{ag}(t)$ for men and women at Røros and Snåsa regions. Values for women are divided by 0.61 (Eq. 3) to allow a direct comparison. In the figures are also shown the corresponding $T_{ag}(t)$ calculated from the model with parameters applied for Swedish hunters (solid line) and rural non-farmers (dashed line).

5 SUMMARY AND CONCLUSIONS

Measurements of whole body burden of radiocaesium have for a long time been performed in Finland, Norway and Sweden. These data have been used to estimate the impact of the nuclear weapons fallout, as well as the Chernobyl accident, on the populations in the three countries. However, during this work, it has become evident that the existing data can be analysed to a greater extent than has been possible within the frames of this project.

For example, comparing the aggregated transfer functions, $T_{ag}(t)$, calculated from Finnish and Norwegian data with models parameterized using Swedish data, the agreement was found to be heavily dependent on the estimated deposition. The issue of modelling uptake of caesium from the food chain, based on ecological half-life or $T_{ag}(t)$, in areas where there are reminiscences of previous fallout occasions of the same extent as the present fallout, has to be further investigated.

Modelling is a useful tool to check our state of knowledge and reveal gaps in data, which have to be considered in case of future fallout events. This also emphasizes the need for maintaining continuous time-series of measurements on atmospheric deposition as well as the body-burdens in various populations, especially among those known in advance to be sensitive to radioecological transfer of fission products.

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Bibliographic Data Sheet **NKS-437**

Title	Radioecological transfer factors for Nordic subpopulations for assessment of internal committed dose from atmospheric fallout of radiocaesium
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Abstract	Measurements of whole body burden of radiocaesium have for a long time been performed in Finland, Norway and Sweden, and used to estimate the impact of the nuclear weapons fallout, as well as the Chernobyl accident, on the populations in the three countries. During this work, it has become evident that the existing data can be analysed to a greater extent than has been possible within the frames of this project. For example, the issue of modelling uptake of caesium from the food chain, based on ecological half-life or aggragated transfer function, $T_{ag}(t)$, in areas where there are reminiscences of previous fallout occasions of the same extent as the present fallout, has to be further investigated. Modelling is a useful tool to check our state of knowledge and reveal gaps in data, which have to be considered in case of future fallout events. This also emphasizes the need for maintaining continuous time-series of measurements on atmospheric deposition as well as the body-burdens in various populations, especially among those known in advance to be sensitive to radioecological transfer of fission products.
Key words	whole-body measurements, fallout, radiocaesium, aggregated transfer function

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