ORGANISATION EUROPEENNE POUR LA RECHERCHE NUCLEAIRE
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European Laboratory for Particle Physics

Safety Commission

CERN-SC-2005-005-IE

Models for environmental impact assessments of releases of radioactive substances from CERN facilities

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ABSTRACT

The document describes generic models for environmental impact assessments of releases of radioactive substances from CERN facilities. Except for few models developed in the Safety Commission, the models are based on the 1997 Swiss directive HSK-R-41 and on the 2001 IAEA Safety Report No. 19. The writing style is descriptive, facilitating the practical implementation of the models at CERN. There are four scenarios assumed for airborne releases: (1) short-term releases for release limit calculations, (2) actual short-term releases, (3) short-term releases during incidents/accidents, and (4) chronic long-term releases during the normal operation of a facility. For water releases, two scenarios are considered: (1) a release into a river, and (2) a release into a water treatment plant. The document shall be understood as a reference for specific environmental studies involving radioactive releases and as a recommendation of the Safety Commission.

Geneva, Switzerland

6 July 2005

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1. Introductory remarks

Modern radiation protection legislation puts regulatory limits on the effective dose to members of the population. The effective dose is a theoretical concept. It cannot be measured directly but only estimated by measuring operational radiation protection quantities or calculated by using appropriate models. This holds in particular for environmental exposure of the population due to radioactive releases. Radio-ecological models, which describe the transport, dispersion and accumulation of radioactivity in various environmental matrices, are combined with radiological models transforming an external exposure or an intake of radioactivity into the effective dose.

The environmental impact of accelerator facilities or other activities, which are likely to release radioactivity into the environment, must be assessed in terms of activity densities found in the environment and in terms of the effective dose to the reference group of the population living in the area of concern. The environment is a very complex system, which is impossible to describe precisely. In addition the true behaviour of the receptor (reference group) is not known and can be predicted only with some uncertainty. Similarly, the time profile of radioactive releases in experimental installations such as accelerator facilities may be variable. In such situation, the so-called screening approach is applied. The screening approach makes use of simplified models concerning release scenarios, behaviour of the released radioactivity in the environment and the habits of the population, which are known to overestimate the activity densities and the effective doses. Often the environmental impact is so unimportant that the screening approach is sufficient to prove that there is no exposure of the population above the regulatory limit. If the limit is exceeded, the calculations shall be refined by entering more accurate site-specific input parameters or by improving the models to be more realistic. If screening models are used for calculation of the effective dose to members of the population due to real radioactive releases (such as annual releases measured at particular facility), their conservatism shall be mentioned when presenting their results.

Despite of the screening approach, the most accurate input parameters or models shall be used if the information is available or if the modern IT technology allows doing so. The degree into which screening techniques and the site-specific information shall be used depends upon the balance between the expected effects of the radioactive releases and the resources needed to use the models. In particular scenarios shall be reasonably predictable, avoiding extreme behaviour and hypothetical situations that are unlikely to be relevant within the period of validity of the assessment. The assessment should be representative of real people living normally in proximity of the facility. Care must be taken in the choice of model parameters to avoid the excessive build-up of multiple conservatisms, especially in the exposure optimization process. Retrospective assessments of doses must be based on *real* scenarios.

Scenarios and assessments should have a robustness and steadiness, which give a firm basis for planning; that is they should not change in short term or be open to manipulation. The need for transparency is principal so that the assessment regime is clear to all stakeholders.

Environmental measurements serve the purpose, amongst other things, of verifying directly that the radiological levels are not incoherent to dose estimates obtained by models. However, in many cases they are not adequate to thoroughly assess doses to members of the public due to the difficulty of measuring very small incremental radiological levels in the environment.

At CERN the safety Commission (SC) has the role of a body that checks the Safety rules, including the environment, to be observed on the territory of the Organization. The SC also has the responsibility to define guidelines for the users in those fields where interpretation of existing rules is requested or where specific rules do not exist. In this respect, this document defines the models to be applied when assessing the environmental impact of radioactive releases. However, it does not exclude a use of different or more complex models provided they guarantee the same or even higher level of Safety. In general, the models presented here are inspired by the Swiss Directive HSK-R-41 [1] and by the IAEA Safety Report No. 19 [2]. In the spirit of the previous paragraphs, some

models are made more realistic by assuming conditions specific to CERN and some recent developments. The terminology and symbols are used as normalized in the ICRU Report No. 65 [3].

Because the sensitivity to the exposure as well as habits may depend on the age of the receptor, exposure of two population groups shall be investigated separately, as extreme cases covering all ages:

a) One-year old infants;

b) Adults (20 years).

For the external exposure, the effective dose integrated during one year shall be calculated. For the internal exposure, the effective dose committed during the rest of the life due to uptake (inhalation, ingestion) of radioactive substances during one year shall be evaluated.

Three main exposure pathways are considered independently:

1. Airborne releases of radioactive air through ventilation outlets;
2. Aqueous releases of radioactive water into watercourses (rivers, streams);
3. Aqueous releases of radioactive water into sewage.

Except for several cases, the models are not explained in detail and the reader shall refer to the original literature.

2. Airborne releases

This Section describes models for an assessment of the environmental impact of radioactive releases from ventilation stacks of accelerator facilities. During an atmospheric release the radioactivity is transported with the wind outside the site and is subject to diffusion. This process depends on weather conditions and it is called dispersion. Further, the radioactivity from the air can be deposited on the ground and enter other environmental matrices, such as soil, plants, animals, etc.

2.1. Scenarios

There are four scenarios, for which different models apply [1]:

1. **Short-term release for release limit calculations**. This scenario corresponds to short-term planned interventions on the facility, when significant amounts of radioactivity may be released at once. The most adverse and unchanging weather conditions for the reference group of the population shall be assumed. This implies that the wind blows in such a direction that the reference group is placed in the plume centreline. The wind speed of $1\ \mathrm{m\cdot s^{-1}}$ shall be supposed. This wind speed is low enough to result in low dilution of the plume, yet the radioactive substances are transported towards the receptor. Rain with a precipitation rate of $2\ \mathrm{mm\cdot h^{-1}}$ shall be considered. It is recommended to check various combinations of the wind speed and atmospheric stability classes to identify the most adverse but realistic weather conditions;
2. **Short-term actual release**. This scenario corresponds to short-term planned interventions on the facility, when significant amounts of radioactivity were released at once, however, the unchanging weather conditions, including the precipitation rate, are known. This is a scenario for retrospective impact assessments. The selection of the reference group of the population depends upon the actual wind direction during the release;
3. **Short-term release during an incident/accident**. This scenario is similar to the actual short-term release but the release was not planned and it was so significant that some countermeasures, such as evacuation, are very likely to take place. The countermeasures and panic may lead to an increased inhalation rate of adults during the first 8 hours of the incident/accident causing a higher effective dose due to inhalation. However, the exposure due to ingestion may be prevented by a ban imposed on contaminated food from the affected area;
4. Chronic **long-term release** corresponding to a normal operation of the facility. The release is assumed to take place uniformly during the last 50 years of the facility operation, including the current year. Weather conditions vary during the 50-year period, however various weather situations occur with well-defined probabilities, which are derived from long-term weather observations.

2.2. Dispersion factor

The dispersion of pollutants in the atmosphere shall be described by the Gaussian plume model. The central quantity of dispersion models is the dispersion factor, $\chi(\mathbf{r})$ ($\mathrm{s\cdot m^{-3}}$), which puts into relation the release rate, $\dot{Q}$ ($\mathrm{Bq\cdot s^{-1}}$), and the average volumetric activity density in the air, $A_v(\mathbf{r})$ ($\mathrm{Bq\cdot m^{-3}}$), at the place $\mathbf{r}$ *during* the release:

$$A_v(\mathbf{r}) = \dot{Q}\chi(\mathbf{r}) \ . \qquad (2.1)$$

The release rate is simply given by the total amount of released radioactivity, Q (Bq), and the duration of the release, t_{rel} (s):

$$\dot{Q} = \frac{Q}{t_{\text{rel}}} \ . \qquad (2.2)$$

Obviously, for the long-term atmospheric release (scenario 4):

$$\dot{Q} = \frac{Q}{\alpha T_1} \ , \qquad (2.3)$$

where

Q = annual radioactive release (Bq),

α = 3.16×10^7 s·a^{-1}, conversion factor from years to seconds,

T_1 = 1 a, time of one year.

2.2.1. Short-term dispersion factor

The dispersion factor for short-term releases, $\chi_S(x,y,z)$ (s·m^{-3}), at a place (x,y,z) in a Cartesian coordinate system is expressed in the following way [1]:

$$\chi_S(x,y,z) = \frac{1}{2\pi\sigma_y\sigma_z\bar{u}} F(h_{\text{eff}},\sigma_z,z)\cdot\exp\left(-\frac{y^2}{2\sigma_y^2}\right)\cdot\exp(-\lambda\frac{x}{\bar{u}}) \ , \qquad (2.4a)$$

$$F(h_{\text{eff}},\sigma_z,z) = (1-E)\left\{\exp\left[-\frac{(h_{\text{eff}}-z)^2}{2\sigma_z^2}\right]+\exp\left[-\frac{(h_{\text{eff}}+z)^2}{2\sigma_z^2}\right]\right\}+2E\exp\left(-\frac{z^2}{2\sigma_z^2}\right) \ , \qquad (2.4b)$$

where

x = downwind distance (m);

y = lateral distance of the receptor point from the plume axis (m),

z = height of the receptor in respect to the stack base (m),

E = entrainment factor,

σ_y = horizontal dispersion coefficient (m),

σ_z = vertical dispersion coefficient (m),

$\bar{u}$ = average wind speed during the release ($m \cdot s^{-1}$),

λ = radioactive decay constant (s^{-1}),

h_{eff} = effective stack height (m).

The equations (2.4a) and (2.4b) account for:

1. Effective stack height, which includes the plume rise and the altitude difference between the stack base and the receptor;
2. Radioactive decay in flight;
3. Plume entrainment due to presence of buildings close to the stack base;
4. Plume reflection from the ground.

The entrainment factor E introduces a fraction of the plume, which is caught by the downwash in the lee of the stack or the building and which is considered as a ground level release. It can be expressed as a function of the wind speed, $\bar{u}$ ($m \cdot s^{-1}$), and the velocity of the exhaust air, W ($m \cdot s^{-1}$). A model given in the IAEA Safety Guide No. 50-SG-S3 [4] shall be used for short stacks (height less than 2–2½ times the height of adjacent solid structures):

$$E = 1.0 \quad \text{for} \quad \frac{W}{\bar{u}} < 1.0 \ , \tag{2.5a}$$

$$E = 2.58 - 1.58\frac{W}{\bar{u}} \quad \text{for} \quad 1.0 \le \frac{W}{\bar{u}} < 1.5 \ , \tag{2.5b}$$

$$E = 0.30 - 0.06\frac{W}{\bar{u}} \quad \text{for} \quad 1.5 \le \frac{W}{\bar{u}} < 5.0 \ , \tag{2.5c}$$

$$E = 0.0 \quad \text{for} \quad 5.0 \ge \frac{W}{\bar{u}} \ . \tag{2.5d}$$

The dependence of the entrainment factor on the ratio $W/\bar{u}$ is shown in Figure 2.1. It increases with the wind speed as the flow is more turbulent and the downwash more pronounced. Note that for the ratio $W/\bar{u} < 1.0$ a complete ground release is conservatively assumed. For $W/\bar{u} > 5.0$ a plume rise is assumed and calculated as for tall stacks.

For tall stacks (height more than 2–2½ times the height of adjacent solid structures),

$$E = 0 \tag{2.6}$$

by definition.

The dispersion coefficients, σ_y and σ_z (m), depend upon the downwind distance, x (m), and the atmospheric stability class. The model uses the standard scheme for assignment of stability classes according to Pasquill and Gifford. There are six stability classes denoted from A (extremely

unstable) to F (moderately stable). It is recommended to use the power functions of Vogt [5] for σ_y and σ_z (m):

$$\sigma_{i,y} = p_{i,y} x^{q_{i,y}} \ , \tag{2.7a}$$

$$\sigma_{i,z} = p_{i,z} x^{q_{i,z}} \ , \tag{2.7b}$$

where the coefficients p and q depend on the atmospheric stability class i and the effective emission height, h_e (m). The values of p and q are given in Table 2.1 for effective emission heights of 50 m, 100 m and 180 m [5].

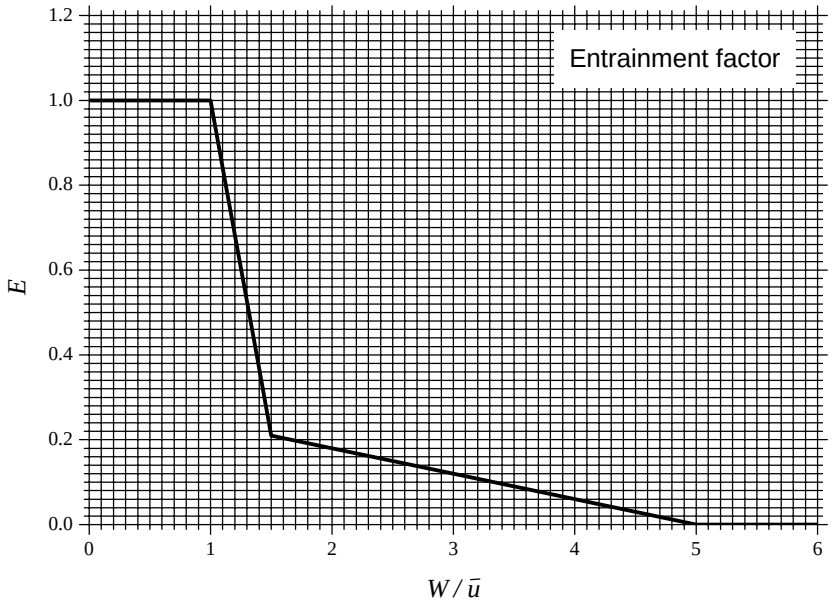

Figure 2.1: Dependence of the entrainment factor E on the exhaust air speed to wind speed ratio $W/\bar{u}$.

For effective emission heights below 50 m, the values for 50 m shall be used. Similarly for the effective emission heights above 180 m, the values for 180 m shall apply. In between, the coefficients p shall be calculated by using geometric interpolation, whereas the coefficients q are given by linear interpolation:

$$p = p_u^{\left(\frac{h-h_l}{h_u-h_l}\right)} p_l^{\left(\frac{h_u-h}{h_u-h_l}\right)} \ , \tag{2.8a}$$

$$q = \frac{(h-h_l)q_u + (h_u-h)q_l}{q_u - q_l} \ , \tag{2.8b}$$

where

h_l = lower interpolation effective emission height (m),

h_u = upper interpolation effective emission height (m),

h = current effective emission height (m),

p_l = lower interpolation coefficient p,

p_u = upper interpolation coefficient p,

p = current interpolation coefficient p,

q_l = lower interpolation coefficient q,

q_u = upper interpolation coefficient q,

p = current interpolation coefficient q.

Table 2.1: Coefficients p and q for the evaluation of the dispersion coefficients σ [5].

Eff. emission height	Stability class	Coefficient			
h_e	A–F (i)	p_y	q_y	p_z	q_z
50 m	A	1.503	0.833	0.151	1.219
	B	0.876	0.823	0.127	1.108
	C	0.659	0.807	0.165	0.996
	D	0.640	0.748	0.215	0.885
	E	0.801	0.754	0.264	0.774
	F	1.294	0.718	0.241	0.662
100 m	A	0.170	1.296	0.051	1.317
	B	0.324	1.025	0.070	1.151
	C	0.466	0.866	0.137	0.985
	D	0.504	0.818	0.265	0.818
	E	0.411	0.882	0.487	0.652
	F	0.253	1.057	0.717	0.486
180 m	A	0.671	0.903	0.0245	1.50
	B	0.415	0.903	0.0330	1.32
	C	0.232	0.903	0.104	0.997
	D	0.208	0.903	0.307	0.734
	E	0.345	0.903	0.546	0.557
	F	0.671	0.903	0.484	0.500

The values of the horizontal dispersion coefficient σ_y for the six atmospheric stability classes and effective emission heights below 50 m are plotted in Figure 2.2. A similar plot for σ_z is shown in Figure 2.3. The case of effective emission heights below 50 m is the most frequent case for stacks at CERN where no facility has a tall stack.

The effective emission height, h_e (m), is given by a sum of the physical height of the stack, h_s (m), and the plume rise, Δh (m):

$$h_e = h_s + \Delta h \ . \qquad (2.9)$$

No thermal buoyancy is assumed for the plume rise, as the ventilation air from accelerator installations is not considerably warmer than the air in the atmosphere. The plume rise, Δh, is expressed according to the atmospheric stability class [4]. For *unstable* or *neutral* conditions (classes A–D):

$$\Delta h = \min\left[1.44D\left(\frac{W}{\bar{u}}\right)^{2/3}\left(\frac{x}{D}\right)^{1/3} - C,\ \ 3\frac{W}{\bar{u}}D\right], \tag{2.10}$$

where D is the internal stack diameter (m) and C is a downwash correction (m) applied for $W < 1.5\bar{u}$:

$$C = 3\left(1.5 - \frac{W}{\bar{u}}\right)D\ . \tag{2.11}$$

A difference between the external and internal stack diameters as assumed in Ref. [4] can be neglected, considering typical stacks at CERN.

For *stable* conditions (classes E and F), the minimum of the expression in Eq. 2.10 and the two following expressions shall be taken [4]:

$$\Delta h = 4\left(\frac{F_{\mathrm{m}}}{S}\right)^{1/4}, \tag{2.12}$$

$$\Delta h = 1.5S^{-1/6}\left(\frac{F_{\mathrm{m}}}{u}\right)^{1/3}\ . \tag{2.13}$$

F_{m} is a momentum-flux parameter and S is a stability parameter, defined as

$$F_{\mathrm{m}} = W^2\left(\frac{D}{2}\right)^2, \tag{2.14}$$

$S = 8.70\times10^{-4}$ for the stability class E, (2.15a)

$S = 1.75\times10^{-3}$ for the stability class F. (2.15b)

The effective height of the release, h_{eff} (m), is the effective emission height corrected for the topological altitude difference between the source and receptor:

$$h_{\mathrm{eff}} = h_{\mathrm{e}} + A_{\mathrm{s}} - A_{\mathrm{r}}\ , \tag{2.16}$$

where

A_s = source topological altitude (m),

A_r = receptor topological altitude (m).

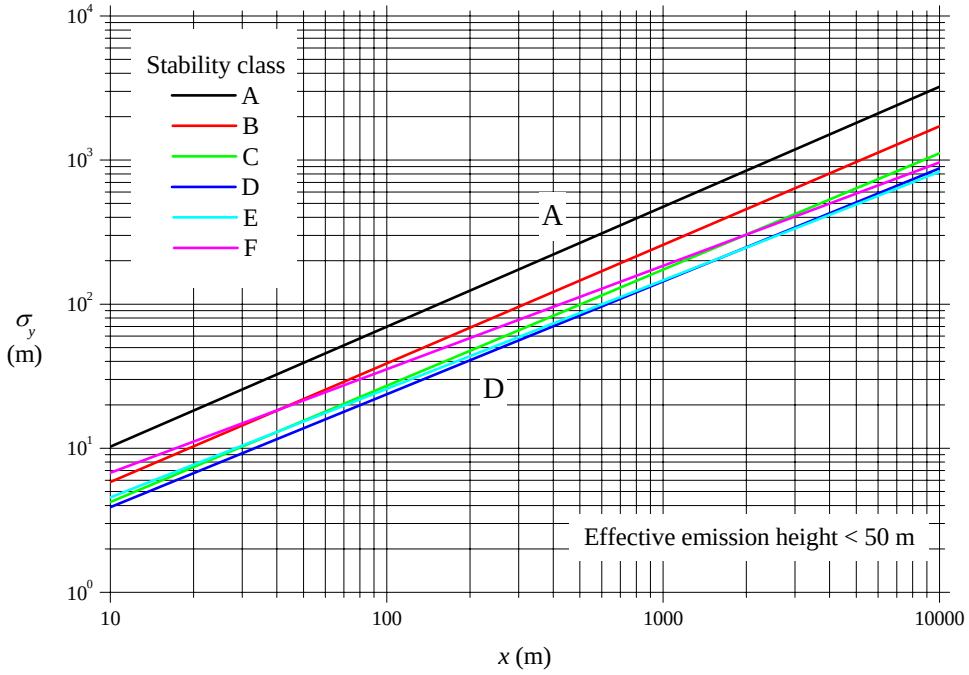

Figure 2.2: Horizontal dispersion coefficient σ_y for effective emission heights below 50 m [5].

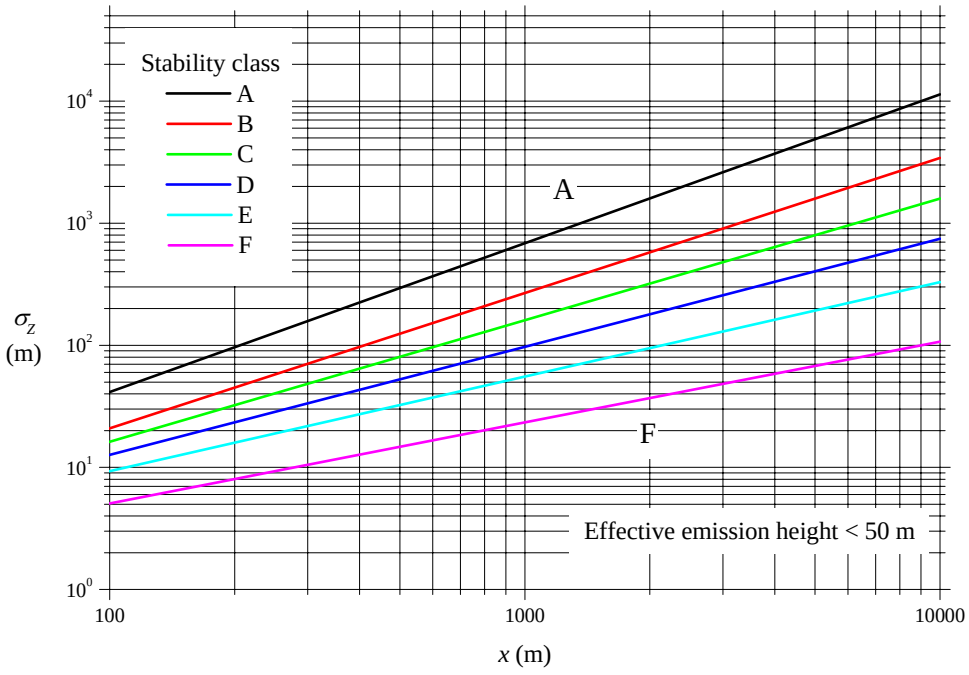

Figure 2.3: Vertical dispersion coefficient σ_z for effective emission heights below 50 m [5].

2.2.2. Long-term dispersion factor

The long-term dispersion factor, χ_{L} (s·m^{-3}), which applies to long-term releases, is an average of the short-term dispersion factors, χ_{S} (s·m^{-3}), weighted with the probability of occurrence of each weather situation $P_{i,j,k}$, which corresponds to that short-term dispersion factor with an atmospheric stability class i, wind blowing into a sector j with an average wind speed from a wind-speed bin k:

$$\chi_{\mathrm{L}}(\mathbf{r}) = \sum_{i,j,k} P_{i,j,k}\,\chi_{\mathrm{S}}(\mathbf{r}, i, j, \bar{u}_k) \ . \tag{2.17}$$

It is recommended to use 72 wind direction sectors, 5 degrees each. If conventional anemometers are used for wind measurements, 20 wind-speed bins, 0–1 m·s^{-1}, each 0.5 m·s^{-1}, and >20 m·s^{-1} are recommended because the data are not reliable below about 1 m·s^{-1}. With ultrasonic anemometers, the wind-speed binning can be finer below 1 m·s^{-1}. The first wind direction sector is positioned around 0 degrees north. Within one 5-degree sector, it is also recommended to average the long-term dispersion factor over 5 directions within the sector, namely by moving the wind direction by 0, ±1 and ±2 degrees from the central direction of the sector.

2.3. Deposition factor

There are two mechanisms of deposition of radioactive substances from the atmosphere. The first is settling out due to gravitational attraction, called fallout or dry deposition, and it affects aerosol and iodine. The second mechanism is scavenging of pollutants by precipitation falling through the plume. This process is called washout or wet deposition and it affects not only aerosol and iodine but also tritium in the form of water vapour.

The parameter describing the deposited fraction of the released radioactive substances is called deposition factor ξ and it is expressed in m^{-2}. It has a different form for short-term and long-term deposition either on the ground (totally) or only on the vegetation (leaves). Obviously, the deposition factor is zero for gases. The total deposition on the surface (Bq·m^{-2}) is a product of the total release and the deposition factor $Q\xi$. The deposition flux of radioactivity (Bq·m^{-2}·s^{-1}) is simply calculated as the product $\dot{Q}\xi$. For iodine the deposition factors shall be multiplied by the coefficient $f_{\mathrm{ei}} = 0.5$, which corrects for the fraction of iodine in the elementary form [1].

2.3.1. Short-term deposition factor

The short-term deposition factor on the ground ξ_{S} is given as follows [1]:

$$\xi_{\mathrm{S}} = \chi_{\mathrm{S}} V_{\mathrm{d}} + \frac{\Lambda}{\sqrt{2\pi}\,\sigma_y \bar{u}} \exp\left(-\frac{y^2}{2\sigma_y^2}\right) \exp\left(-\lambda \frac{x}{\bar{u}}\right) , \tag{2.18}$$

where the first term describes the dry deposition and the second term describes the wet deposition. The factor V_{d} has the unit [m·s^{-1}] and it is called deposition velocity. It is recommended to use values of V_{d} of 1.5×10^{-3} m·s^{-1} for aerosol and 1.0×10^{-2} m·s^{-1} for iodine [1].

The second term describes the wet deposition. It has been derived by integrating the expression for the short-term dispersion factor with respect to z over the interval $(0,\infty)$ – the full plume height – and accounting for imperfect washout by the washout factor Λ (s^{-1}). The washout factor Λ can be determined from the precipitation rate I_N ($mm \cdot h^{-1}$) using the scaling equation:

$$\Lambda = \Lambda_0 \left(\frac{I_N}{I_0} \right)^{\kappa} , \tag{2.19}$$

where

Λ_0 = reference washout factor = 7.0×10^{-5} s^{-1} for aerosol and iodine,
= 3.5×10^{-5} s^{-1} for tritium [1],

κ = correction coefficient = 0.8 for aerosol and iodine,
= 1.0 for tritium [1],

I_0 = 1 $mm \cdot h^{-1}$, reference precipitation rate.

The directive HSK-R-41 assumes the precipitation rate of 2 $mm \cdot h^{-1}$ for calculations of short-term release limits. For actual releases the actual precipitation rate should be used if available [1].

Analogically, the short-term deposition factor on vegetation ξ'_S is given by the formula:

$$\xi'_S = \chi_S V_d + f_d \frac{\Lambda}{\sqrt{2\pi}\sigma_y \bar{u}} \exp\left(-\frac{y^2}{2\sigma_y^2} \right) \exp\left(-\lambda \frac{x}{\bar{u}} \right) , \tag{2.20}$$

where the factor f_d in the second term represents the fraction of direct wet deposition on the plant surface. It has a value of 0.3 for aerosol and a conservative value of 1.0 for iodine [1].

2.3.2. Long-term deposition factor

For calculation of the long-term deposition factors the directive HSK-R-41 recommends the use of a four-dimensional joint probability matrix including also the precipitation rates for various weather situations. Because this method requires very detailed meteorological data and is difficult to handle mathematically, the directive also allows a simplified conservative model using an enlarged deposition velocity $V'_d = 1.7 \times 10^{-2}$ $m \cdot s^{-1}$. This approximation is recommended for use at CERN. The long-term deposition factor on the ground is then simply

$$\xi_L = \chi_L \cdot V'_d , \tag{2.21}$$

whilst the long-term deposition factor on the vegetation is

$$\xi'_L = f_d \cdot \chi_L \cdot V'_d . \tag{2.22}$$

2.3.3. Impact points for food production

Around CERN, agricultural areas can be very close to ventilation outlets, sometimes closer than few tens of metres. In such a situation, deposition on the ground and plants is very inhomogeneous with sharp maxima. Effective doses due to ingestion of food produced in the area may be strongly overestimated if the maximum deposition localized in some hundreds of m^2 is assumed for large areas that are usually needed for long-term fodder and food production. To reduce the conservatism of the model, it is recommended to investigate the deposition profiles (dependence of the deposition factors on the two-dimensional coordinates in the field) in such cases. This holds especially if the regulatory limit put on the effective dose is approached when using the simple deposition model. Representative impact points can be entered into the dose calculation instead of the maximum deposition point. For a representative impact point, the same effective dose is committed with a homogeneous deposition on a large scale as while considering the detailed deposition profile and a suitable food production scenario. The applied method strongly depends upon the particular conditions; however, some general comments can be made:

1. Find the critical ingestion radionuclide and the critical exposure pathway so that during the searching for the representative impact point these parameters can be fixed. For releases from accelerator facilities, the critical radionuclides are often phosphorus isotopes $^{32, 33}$P and the critical exposure pathway is ingestion by infants of milk produced in the area of concern.
2. A typical cow grazes 65 kg of fodder per day [1]. With the typical fodder biomass density of 0.85 $kg \cdot m^{-2}$ [1], a cow can graze an area of 76.5 m^2 per day. It is reasonable to examine the deposition profiles with bins corresponding to this area (8.75×8.75 m^2).
3. Fodder grazed in one plot needs typically 30 days to grow and cannot be grazed the next days again. This fact shall be taken into account in food production scenarios, especially following short-term releases.
4. For long-term emissions, an average deposition factor can be calculated in an area adjacent to the emission site, which is sufficiently large to cover the fodder or food production fields (e.g. 500×500 m^2 – 10 cows grazing from one field).
5. If the ingestion effective dose approaches the regulatory limit, it is recommended to examine the activity densities in single food samples (e.g. milk from one cow) and compare them with the applicable food limits.

2.4. Short-term releases lasting longer than 8 hours

To account for changing weather conditions during longer-lasting short-term releases, a scaling factor can be introduced (Table 2.2, [1]). The dispersion and deposition factors are divided by this factor. However, this approach may lead to wrong external doses from the radioactive plume because the dose depends on the shape and extent of the plume. It is recommended to break a longer-lasting short-term release into a number of shorter-lasting releases with unvarying weather conditions if possible (e.g. day and night).

2.5. Weather data and determination of the atmospheric stability class

To calculate the dispersion and deposition factors, information about the actual weather parameters is required. The parameters needed are the average wind speed, the average wind direction, the atmospheric stability class and, for short-term actual releases and incidents/accidents, the precipitation rate. They can be usually obtained from meteorological services, except the atmospheric stability class, which must be determined. For long-term dispersion factors, these parameters are required over a period lasting at least one year.

Table 2.2: Scaling factors for longer-lasting short-term releases [1].

Release duration (h)	Scaling factor
<8	1
8–24	2
24–72	4
>72	8

2.5.1. Atmospheric stability class

There are various methods for the determination of the atmospheric stability class described in Refs. [4] and [6]. The choice of a particular method depends primarily on the meteorological data available. Hourly averages of the wind speed, wind direction, insolation during the day ($W \cdot m^{-2} \cdot h$), and net radiation during the night ($W \cdot m^{-2} \cdot h$) are available from the weather station at the airport Geneva Cointrin, which is close to the CERN Meyrin site. Furthermore, an overcast observation is available for each third hour. Using these data, a method elaborated by Pasquill and Gifford, with some modifications as recommended in the IAEA Safety Guide No. 50-SG-S3 [4], can be applied for a stability class assignment.

The Pasquill–Gifford method is based on the measurements of the hourly average wind speed at a 10 m level, insolation during the day and observations of the cloud cover during the night.

The assignment of the stability class during the day follows the scheme in Table 2.3 (Table IV from Ref. [4]). For cases of intermediate dispersion conditions marked as for example A–B in Ref. [4], the class with smaller dispersion was taken. The assignment of the stability class during the night, which is based on overcast observations, follows the scheme in Table 2.4 (Table III from Ref. [4]). For the hourly data records during which no overcast value is given, the overcast is interpolated assuming that the cloud cover does not change too rapidly.

The decision whether an hourly weather observation refers to the day or to the night is made according to the value of the insolation. It is assumed that night falls when the insolation drops below 10 $W \cdot m^{-2}$. The Safety Guide No. 50-SG-S3 defines the beginning of the night as one hour before the sunset and the end of the night as one hour after the sunrise. The insolation decision criterion was compared with astronomical sunrise and sunset calculations. It was found that, for the beginning of the day, the agreement is fairly good, whilst daytime ends about one hour sooner than given by the astronomical sunset. The reason is the proximity of the Jura Mountains, which shade the setting sun from the north-west side of the CERN Meyrin site. The direct decision according to insolation has two main advantages: It avoids calculations of seasonally dependent sunrise and sunset hours from astronomical tables, and it takes into account the specific local conditions. Because the driving force of the atmospheric turbulence is the solar-energy flux on the ground surface, the insolation criterion seems to be more appropriate than astronomical calculations and it should cope well with special situations like dark cloudy days during the winter.

An alternative method feasible at CERN is the determination of the atmospheric stability class directly from the turbulence variables, such as the heat flux, measured by ultrasonic anemometers. Some devices provide directly the stability class (e.g. Metek USA-1/T, www.metek.de). The environment for an anemometer of this type must fulfil some specific criteria defined, for example in Ref. [6]. Obviously, there must not be obstacles around, which would change the wind patterns at the 10 m measuring height and the surface below the anemometer must be natural, preferably covered with vegetation. Surfaces strongly affecting the heat balance like roads or roofs of buildings must be avoided.

Table 2.3: Assignment of the stability class during the day [4].

Wind speed $\bar{u}$ (m·s^{-1})	Solar radiation R_D (langleys·h^{-1})			
	$R_D \geq 50$	$50 > R_D \geq 25$	$25 > R_D \geq 12.5$	$12.5 > R_D$
$\bar{u} < 2$	A	A	B	D
$2 \leq \bar{u} < 3$	A	B	C	D
$3 \leq \bar{u} < 4$	B	B	C	D
$4 \leq \bar{u} < 6$	C	C	D	D
$6 \leq \bar{u}$	C	D	D	D

1 langley = 1 cal·cm^{-2} = 4.187 J·cm^{-2}.
1 W·m^{-2} = 0.086 langley·h^{-1}

Table 2.4: Assignment of the stability class during the night [4].

Wind speed $\bar{u}$ (m·s^{-1})	Overcast (octal)	
	$\geq 4/8$ cloud cover	$\leq 3/8$ cloud cover
$\bar{u} < 2$	F	F
$2 \leq \bar{u} < 3$	E	F
$3 \leq \bar{u} < 5$	D	E
$5 \leq \bar{u} < 6$	D	D
$6 \leq \bar{u}$	D	D

2.5.2. Statistics

For long-term releases statistics on the above-mentioned weather parameters shall be available in form of probabilities for each weather situation to occur. The wind speeds and wind directions shall be classified into bins discussed in Section 2.2.2 for every stability class separately. It should be noted that the wind direction is the direction *from* which the wind blows in meteorology, i.e. it is opposite to the wind velocity vector. The calms shall be treated according to the Ref. [4] that is they shall fall into the lowest wind-speed bin with the direction distribution equal to that of the next wind-speed bin for each stability class.

2.6. Activity densities in environmental matrices

The radioecological impact of radioactive releases is expressed in terms of activity densities in various environmental matrices. Although the target quantity is the effective dose to members of the public, knowledge of the activity densities in the environment may be of importance if immission limits or specific activity limits for food shall be observed. The model considers two types of vegetation: (1) vegetables (fruits) – intended for direct consumption by people, and (2) fodder – to be eaten by animals. Similarly all animal food products are divided into two simple categories: (a) meat of cattle and (b) milk of cattle.

2.6.1. Air

The average volumetric activity densities of the given radionuclide in the air at a place $\mathbf{r}$ for short-term and long-term releases, $A_{v,S}(\mathbf{r})$ and $A_{v,L}(\mathbf{r})$, respectively, are given as

$$A_{v,S} = \dot{Q}\chi_S(\mathbf{r}) \,, \tag{2.23a}$$

$$A_{v,\mathrm{L}} = \dot{Q}\chi_{\mathrm{L}}(\mathbf{r}) \ . \qquad (2.23b)$$

For short-term releases Eq. 2.23a gives the average volumetric activity density *during* the release. Obviously, for long-term releases Eq. 2.23b gives the average volumetric activity density during the whole year.

2.6.2. Ground

For *short-term* releases the maximum area activity density on the ground, $A_{a,\mathrm{S}}(0)$ (Bq·m^{-2}) occurs immediately after the release. It is determined by the total release, Q (Bq), and by the short-term deposition factor ξ_{S} (m^{-2}),

$$A_{a,\mathrm{S}}(0) = Q\xi_{\mathrm{S}} \ , \qquad (2.24)$$

provided the duration of the release was considerably shorter than the effective half-life of the radionuclide on the ground surface (see later).

For *long-term* releases one has to consider the build-up of the activity during the operation period of the facility, the radioactive decay of the given radionuclide, and its penetration into deeper layers of soil, which is described by non-radioactive decay constants λ_{F} and λ_{S} [1]. The maximum area activity density, $A_{a,\mathrm{L}}(T_{\mathrm{op}})$ (Bq·m^{-2}), is reached at the end of the operation period T_{op} [1]:

$$A_{a,\mathrm{L}}(T_{\mathrm{op}}) = \dot{Q}\xi_{\mathrm{L}}\left\{0.63\frac{1-\exp\left[-(\lambda+\lambda_{\mathrm{F}})T_{\mathrm{op}}\right]}{\lambda+\lambda_{\mathrm{F}}} + 0.37\frac{1-\exp\left[-(\lambda+\lambda_{\mathrm{S}})T_{\mathrm{op}}\right]}{\lambda+\lambda_{\mathrm{S}}}\right\} \ , \qquad (2.25)$$

where

ξ_{L} = long-term deposition factor on the ground (m^{-2}),

λ = radioactive decay constant (a^{-1}),

λ_{F} = 1.1 a^{-1}, non-radioactive decay constant for the *fast* decay component [1],

λ_{S} = 7.5 × 10^{-3} a^{-1}, non-radioactive decay constant for the *slow* decay component [1],

T_{op} = 50 a, operation period of the facility.

2.6.3. Vegetables and fodder

Plants incorporate airborne radioactivity due to direct deposition on their leaves and through the root uptake from soil. Agricultural land is usually ploughed at the beginning of the year, which leads to periodic mixing of the activity deposited on the ground into the plant root zone. A year is

[1] The most important radionuclides in CERN's airborne releases have half-lives considerably shorter than the assumed operation period of the facility of 50 years. Consequently the calculated area activity densities practically do not depend on the length of this period.

divided into a plant growing period lasting from 16 April to 15 October and the wintertime (rest of the year).

For a *short-term* release, the activity densities in plants essentially depend upon the time when the release takes place. The production time T_{p} (a) plays an important role. It is the time, which elapsed from the release to the next end of the harvest. For an actual release after 15 October but till the end of the current year, it extends to the following year. In the scenario for short-term release limit calculations, the production time has to be set to 8.3×10^{-2} a (1 month) [1].

For a short-term release during the plant growing period ($T_{\mathrm{p}} \le 0.5\,\mathrm{a}$), the average mass activity density in fresh vegetables, $A_{m,\mathrm{S\text{-}veg}}$ (Bq·kg^{-1}), is given as [1]:

$$A_{m,\mathrm{S\text{-}veg}} = A^0_{\mathrm{veg,lv}} \frac{1}{T_1}\left[\frac{1}{\lambda_{\mathrm{e,lv}}} + e^{-\lambda T_{\mathrm{p}}} \cdot \frac{2}{\lambda_{\mathrm{lv}} T_1} \cdot \frac{1-e^{-\lambda T_1/2}}{\lambda}\right] + A^0_{\mathrm{veg,r}} \frac{1}{T_1} \cdot \frac{e^{-\lambda_{\mathrm{e,r}}(T_{\mathrm{p}}+T_1/2)} - e^{-\lambda_{\mathrm{e,r}} T_{\mathrm{inc}}}}{\lambda_{\mathrm{e,r}}}\ , \qquad (2.26)$$

and similarly the average mass activity density in fodder, $A_{m,\mathrm{S\text{-}fod}}$ (Bq·kg^{-1}), can be calculated as [1]:

$$A_{m,\mathrm{S\text{-}fod}} = A^0_{\mathrm{fod,lv}} \frac{1}{T_1}\left[\frac{1}{\lambda_{\mathrm{e,lv}}} + e^{-\lambda T_{\mathrm{p}}} \cdot \frac{2}{\lambda_{\mathrm{lv}} T_1} \cdot \frac{1-e^{-\lambda T_1/2}}{\lambda}\right] + A^0_{\mathrm{fod,r}} \frac{1}{T_1} \cdot \frac{e^{-\lambda_{\mathrm{e,r}}(T_{\mathrm{p}}+T_1/2)} - e^{-\lambda_{\mathrm{e,r}} T_{\mathrm{inc}}}}{\lambda_{\mathrm{e,r}}}\,. \qquad (2.27)$$

All symbols will be explained later in this Section.

For a short-term release in the wintertime, when $T_{\mathrm{p}} > 0.5\,\mathrm{a}$, only the root uptake takes place and the formulae are simpler [1]:

$$A_{m,\mathrm{S\text{-}veg}} = A^0_{\mathrm{veg,r}} \frac{1}{T_1} \cdot \frac{e^{-\lambda_{\mathrm{e,r}}(T_{\mathrm{p}}-T_1/2)} - e^{-\lambda_{\mathrm{e,r}} T_{\mathrm{inc}}}}{\lambda_{\mathrm{e,r}}}\ , \qquad (2.28)$$

$$A_{m,\mathrm{S\text{-}fod}} = A^0_{\mathrm{fod,r}} \frac{1}{T_1} \cdot \frac{e^{-\lambda_{\mathrm{e,r}}(T_{\mathrm{p}}-T_1/2)} - e^{-\lambda_{\mathrm{e,r}} T_{\mathrm{inc}}}}{\lambda_{\mathrm{e,r}}}\ . \qquad (2.29)$$

In the equations above, the four mass activity densities A^0 (Bq·kg^{-1}) are given as follows:

$$A^0_{\mathrm{veg,lv}} = Q \frac{\xi'_{\mathrm{S}}}{B_{a,\mathrm{veg}}}\ , \qquad (2.30)$$

$$A^0_{\mathrm{fod,lv}} = Q \frac{\xi'_{\mathrm{S}}}{B_{a,\mathrm{fod}}}\ , \qquad (2.31)$$

$$A^0_{\mathrm{veg,r}} = Q \frac{\xi_{\mathrm{S}}}{P_{a,\mathrm{veg}}} C_{r,\mathrm{soil\text{-}veg}}\ , \qquad (2.32)$$

$$A^0_{\mathrm{fod,r}} = Q\frac{\xi_{\mathrm{S}}}{P_{a,\mathrm{fod}}}C_{r,\mathrm{soil\text{-}fod}} \ . \qquad (2.33)$$

The symbols used in the Eqs. 2.26–2.33 have the following meanings:

λ = radioactive decay constant (a^{-1}),

λ_{lv} = non-radioactive decay constant on leaves, see later (a^{-1}),

$\lambda_{\mathrm{e,lv}}$ = effective decay constant on leaves, see later (a^{-1}),

$\lambda_{\mathrm{e,r}}$ = effective decay in the root zone, see later (a^{-1}),

T_{p} = plant production time (a),

T_1 = 1 a, time of one year,

T_{inc} = 1 a, incorporation time for plants [1],

Q = released radioactivity (Bq),

ξ'_{S} = short-term deposition factor on leaves (m^{-2}),

$B_{a,\mathrm{veg}}$ = 2.4 kg·m^{-2}, typical biomass density of vegetables [1],

$B_{a,\mathrm{fod}}$ = 0.85 kg·m^{-2}, typical biomass density of fodder [1],

$P_{a,\mathrm{veg}}$ = 280 kg·m^{-2}, typical area density of soil within the reach of roots for vegetables [1],

$P_{a,\mathrm{fod}}$ = 120 kg·m^{-2}, typical area density of soil within the reach of roots for fodder [1],

$C_{r,\mathrm{soil\text{-}veg}}$ = concentration ratio for the uptake from soil to vegetables (element-specific dimensionless factor),

$C_{r,\mathrm{soil\text{-}fod}}$ = concentration ratio for the uptake from soil to fodder (element-specific dimensionless factor).

The effective decay constant on the leaves, $\lambda_{\mathrm{e,lv}}$ (a^{-1}), is a sum of the radioactive decay constant, λ (a^{-1}), and the non-radioactive decay constant on the leaves, λ_{lv} (a^{-1}):

$$\lambda_{\mathrm{e,lv}} = \lambda + \lambda_{\mathrm{lv}} \ , \qquad (2.34)$$

where

λ_{lv} = 18 a^{-1} for aerosol,

= 32 a^{-1} for iodine.

Similarly, the effective decay constant in the root zone, $\lambda_{\mathrm{e,r}}$ (a^{-1}), is a sum of the radioactive decay constant, λ (a^{-1}), and the non-radioactive decay constant in the root zone, λ_{r} (a^{-1}):

$$\lambda_{e,r} = \lambda + \lambda_r \ , \tag{2.35}$$

where the value of λ_r is element-specific and shall be assigned according to Table 2.5 [1]:

Table 2.5: Values of the non-radioactive decay constant in the root zone of plants, λ_r [1].

Element	λ_r (a^{-1})
Tc, Sr, Cs	7.0×10^{-2}
Ca, Br, Ba, Mn, Zn	3.5×10^{-2}
I, Te	1.7×10^{-2}
Others	0

For a *long-term* release, accumulation of the radioactivity in soil during the operation period of the facility, T_{op}, shall be assumed. The mass activity densities in vegetables and fodder, $A_{m,\text{L-veg}}$ and $A_{m,\text{L-fod}}$ ($Bq \cdot kg^{-1}$), respectively, are given as [1]:

$$A_{m,\text{L-veg}} = \frac{1}{T_1}\left\{ A^0_{\text{veg,lv}} \left[\frac{T_1}{2} + \frac{2}{T_1}\left(\frac{1-e^{-\lambda T_1/2}}{\lambda} \right)^2 \right] + A^0_{\text{veg,r}} e^{-\lambda_{e,r} T_h} \cdot \frac{1-e^{-\lambda_{e,r} T_1}}{\lambda_{e,r}} \right\} , \tag{2.36}$$

$$A_{m,\text{L-fod}} = \frac{1}{T_1}\left\{ A^0_{\text{fod,lv}} \left[\frac{T_1}{2} + \frac{2}{T_1}\left(\frac{1-e^{-\lambda T_1/2}}{\lambda} \right)^2 \right] + A^0_{\text{fod,r}} e^{-\lambda_{e,r} T_h} \cdot \frac{1-e^{-\lambda_{e,r} T_1}}{\lambda_{e,r}} \right\} , \tag{2.37}$$

where $T_h = 0.29\,a$ is the time between the beginning of the calendar year (1 January) and the beginning of the plant growing period (16 April). Similarly to the short-term release the mass activity densities A^0 ($Bq \cdot kg^{-1}$) are given as [1]:

$$A^0_{\text{veg,lv}} = \frac{Q}{T_1} \frac{\xi'_L}{B_{a,\text{veg}}} \cdot \frac{1}{\lambda_{e,lv}} \ , \tag{2.38}$$

$$A^0_{\text{fod,lv}} = \frac{Q}{T_1} \frac{\xi'_L}{B_{a,\text{fod}}} \cdot \frac{1}{\lambda_{e,lv}} \ , \tag{2.39}$$

$$A^0_{\text{veg,r}} = \frac{Q}{T_1} \frac{\xi_L}{P_{a,\text{veg}}} \cdot \frac{1-\exp(-\lambda_{e,r} T_{op})}{\lambda_{e,r}} C_{r,\text{soil-veg}} \ , \tag{2.40}$$

$$A^0_{\text{fod,r}} = \frac{Q}{T_1} \frac{\xi_L}{P_{a,\text{fod}}} \cdot \frac{1-\exp(-\lambda_{e,r} T_{op})}{\lambda_{e,r}} C_{r,\text{soil-fod}} \ , \tag{2.41}$$

where

Q = annual release of radioactivity (Bq),

ξ'_{L} = long-term deposition factor on leaves (m^{-2}),

ξ_{L} = long-term deposition factor on the ground (m^{-2}),

T_{op} = 50 a, operation period of the facility,

and the other symbols have the same meaning as above.

In numerical calculations the build-up terms shall be replaced with limit values for very long-lived radionuclides when λ or $\lambda_{\mathrm{e,r}}$ is smaller than10^{-4}, for example:

$$\lim_{\lambda\to 0} \frac{1-e^{-\lambda T_1/2}}{\lambda} = T_1/2 \ , \tag{2.42}$$

$$\lim_{\lambda_{\mathrm{e,r}}\to 0} \frac{e^{-\lambda_{\mathrm{e,r}}(T_{\mathrm{p}}+T_1/2)} - e^{-\lambda_{\mathrm{e,r}}T_{\mathrm{inc}}}}{\lambda_{\mathrm{e,r}}} = T_{\mathrm{inc}} - \frac{T_1}{2} - T_{\mathrm{p}} \ , \tag{2.43}$$

$$\lim_{\lambda_{\mathrm{e,r}}\to 0} \frac{e^{-\lambda_{\mathrm{e,r}}(T_{\mathrm{p}}-T_1/2)} - e^{-\lambda_{\mathrm{e,r}}T_{\mathrm{inc}}}}{\lambda_{\mathrm{e,r}}} = T_{\mathrm{inc}} + \frac{T_1}{2} - T_{\mathrm{p}} \ , \tag{2.44}$$

$$\lim_{\lambda_{\mathrm{e,r}}\to 0} \frac{1-e^{-\lambda_{\mathrm{e,r}}T_1}}{\lambda_{\mathrm{e,r}}} = T_1 \ , \tag{2.45}$$

$$\lim_{\lambda_{\mathrm{e,r}}\to 0} \frac{1-\exp(-\lambda_{\mathrm{e,r}}T_{\mathrm{op}})}{\lambda_{\mathrm{e,r}}} = T_{\mathrm{op}} \ . \tag{2.46}$$

For releases of *tritium* in the form of water vapour as well as for releases of ^{14}C in the form of CO_2, equilibrium and photosynthesis models are used, respectively [1]. Namely for *tritium*:

$$A_{m,\text{S-veg}} = A_{m,\text{S-fod}} = Q\frac{\chi_{\mathrm{S}}}{\phi\alpha T_1} f_{\mathrm{wa}} \ , \tag{2.47}$$

$$A_{m,\text{L-veg}} = A_{m,\text{L-fod}} = Q\frac{\chi_{\mathrm{L}}}{\phi\alpha T_1} f_{\mathrm{wa}} \ , \tag{2.48}$$

where

Q = (annual) release of radioactivity (Bq),

χ_{S} = short-term dispersion factor ($s\cdot m^{-3}$),

χ_{L} = long-term dispersion factor ($s\cdot m^{-3}$),

ϕ = 9×10^{-3} $kg\cdot m^{-3}$, average humidity of air [1],

α = 3.16×10^{7} $s\cdot a^{-1}$, conversion factor from years to seconds,

T_1 = 1 a, time of one year,

f_{wa} = 0.75, average water content of foodstuff [1].

Similarly for ^{14}C:

$$A_{m,\text{S-veg}} = A_{m,\text{S-fod}} = Q\frac{\chi_S}{\Psi\alpha T_1} f_C \ , \tag{2.49}$$

$$A_{m,\text{L-veg}} = A_{m,\text{L-fod}} = Q\frac{\chi_L}{\Psi\alpha T_1} f_C \ , \tag{2.50}$$

where,

Ψ = 1.8×10^{-4} kg·m^{-3}, average carbon content in air [1],

f_C = 0.125, mass fraction of carbon in foodstuff [1],

and the other symbols have the same meanings as above.

2.6.4. Meat and milk

Except for tritium and ^{14}C, the mass activity densities in meat and milk of animals fed with fodder produced in the area of concern is derived from the mass activity density in fodder by using element-specific feed transfer coefficients:

$$A_{m,\text{S-mt}} = A_{m,\text{S-fod}} V_{\text{fod}} C_{ft,\text{fod-mt}} \ , \tag{2.51}$$

$$A_{m,\text{S-mi}} = A_{m,\text{S-fod}} V_{\text{fod}} C_{ft,\text{fod-mi}} \ , \tag{2.52}$$

$$A_{m,\text{L-mt}} = A_{m,\text{L-fod}} V_{\text{fod}} C_{ft,\text{fod-mt}} \ , \tag{2.53}$$

$$A_{m,\text{L-mi}} = A_{m,\text{L-fod}} V_{\text{fod}} C_{ft,\text{fod-mi}} \ , \tag{2.54}$$

where

$A_{m,\text{S-mt}}$ = mass activity density in meat for a short-term release (Bq·kg^{-1}),

$A_{m,\text{S-mi}}$ = mass activity density in milk for a short-term release (Bq·kg^{-1}),

$A_{m,\text{L-mt}}$ = mass activity density in meat for a long-term release (Bq·kg^{-1}),

$A_{m,\text{L-mi}}$ = mass activity density in milk for a long-term release (Bq·kg^{-1}),

V_{fod} = 65 kg·d^{-1}, daily consumption of fodder by cattle [1],

$C_{ft,\text{fod-mt}}$ = feed transfer coefficient from fodder to meat (d·kg^{-1}),

$C_{ft,\text{fod-mi}}$ = feed transfer coefficient from fodder to milk ($\text{d}\cdot\text{kg}^{-1}$).

For *tritium* the equilibrium model applies, however, only a fraction $f_{\text{fod}} = 0.4$ of water in animals originates in food [1]:

$$A_{m,\text{S-mt}} = A_{m,\text{S-mi}} = f_{\text{fod}} A_{m,\text{S-fod}} \quad (2.55)$$

$$A_{m,\text{L-mt}} = A_{m,\text{L-mi}} = f_{\text{fod}} A_{m,\text{L-fod}} \quad (2.56)$$

The two equations above relate to a short-term and a long-term release, respectively.

As all carbon in cattle may originate from fodder, the mass activity densities in meat and milk are equal to that in fodder for ^{14}C:

$$A_{m,\text{S-mt}} = A_{m,\text{S-mi}} = A_{m,\text{S-fod}} \quad (2.57)$$

$$A_{m,\text{L-mt}} = A_{m,\text{L-mi}} = A_{m,\text{L-fod}} \quad (2.58)$$

2.6.5. A remark concerning fodder, food and long-term releases

The models described so far for long-term airborne releases supposed that the total activity was released uniformly in time during one year. However, the accelerator operation at CERN is marked by two distinct periods: a long-term shutdown, usually during winter, and more or less a continuous period with beams. Usually, emissions of radioactive substances take place only during the latter period. The ingestion dose depends on the activity densities in fodder and in food and these depend strongly on the deposition season (summer, winter, plant growing period, etc.). Therefore, it is recommended to examine what contamination and exposure pathways are important for the given problem and when they take place. Very often the deposition during the plant growing period is decisive. In this case it is useful to find a fraction of the year with releases of radioactive substances, which covers the plant growing period, and correct the results with an appropriate factor. If, for example, the accelerator operation period with releases lasts 200 days and covers the whole plant growing period, the activity densities in fodder and food and the effective doses due to ingestion of food produced in the area of concern shall be multiplied by a corrective factor 365/200 = 1.825. Be careful to correct only once in the chain fodder-food-dose.

This remark concerns only long-term releases.

2.7. Effective doses

The following exposure pathways shall be assumed:

1. External exposure from radioactive plume;
2. External exposure from radioactivity deposited on the ground;
3. Internal exposure due to inhalation of radioactive air (direct from the plume and resuspension);
4. Internal exposure due to ingestion of vegetables, meat and milk produced in the area of concern.

Note that for the receptor residing very close to the emission point, as it is usually the case at CERN, the impact place for the exposure pathways 1–3 need not to be the same as for the ingestion exposure pathway and separate calculations may need to be carried out (see also Section 2.3.3).

The total effective dose is a sum of all the contributions listed above and discussed below, while considering the age-specific effective doses to infants and adults separately.

2.7.1. External dose from a radioactive plume

The external dose is calculated by two methods, depending on the characteristics of the emitted radiation. Radionuclides are divided into two categories:

1. Photon emitters;
2. (Almost) pure beta (alpha) emitters.

Photon emitters comprise also pure positron emitters, such as ^{11}C or ^{13}N, because annihilation of the beta-decay positrons leads to emission of 511 keV annihilation photons. The (almost) pure beta (alpha) emitters are radionuclides that emit photons with a yield smaller than 0.1%. Alpha emitters are rare at CERN. External exposure from the radioactive plume is usually negligible compared with other exposure pathways, like inhalation, for this radionuclide category.

For **photon emitters**, the effective dose D_p consists of an effective dose due to photon (gamma) radiation emitted from the plume $D_{p\gamma}$ and the effective dose contribution from the skin irradiation by beta (alpha) particles:

$$D_{\mathrm{S},p}(a,i) = D_{\mathrm{S},p\gamma}(a,i) + 0.01 \times d_{\mathrm{S,ibs}} \ , \tag{2.59}$$

$$D_{\mathrm{L},p}(a,i) = f_{\mathrm{oc}}\left(D_{\mathrm{L},p\gamma}(a,i) + 0.01 \times d_{\mathrm{L,ibs}}\right) , \tag{2.60}$$

where f_{oc} is the long-term occupancy factor (1.0 for the full-time presence in the area of concern), the indices S and L indicate the short-term and long-term dose, and the symbols *a* and *i* indicate adults and infants, respectively. The symbols $d_{\mathrm{S,ibs}}$ and $d_{\mathrm{L,ibs}}$ indicate the organ doses to the skin from beta (alpha) radiation for short-term and long-term releases, respectively. The factor 0.01 [7] is the effective-dose weight for the organ dose in the skin. The contribution of the dose to the skin is usually negligible and does not make more than a few % of the photon dose except some special cases in which only weak photon radiation is emitted.

The effective dose due to photon radiation from the plume is estimated from the dose kernel integral over the plume and converted to the age-dependent effective dose by using photon-energy dependent conversion factors for isotropic irradiation. For the sake of simplicity, the symbol χ means either the short-term dispersion factor χ_{S} or the long-term dispersion factor χ_{L}, according to the relevant case. If the receptor is in the origin of the coordinate system, the effective dose due to photon radiation, $D_{\mathrm{S(L)},p\gamma}(a,i)$ (Sv·a^{-1}), is given as a three dimensional integral

$$D_{\mathrm{S(L)},p\gamma}(a,i) = \frac{Q}{T_1} e_{\mathrm{kerma}}(a,i) \frac{\varepsilon}{\rho_{\mathrm{air}}} k_{\mathrm{s}} \frac{1}{4\pi} \sum_i \mu_{a,i} E_i Y_i \int \frac{B(E_i, \mu_i r) \exp(-\mu_i r) \chi(\mathbf{r})}{r^2} d^3\mathbf{r} \ , \tag{2.61}$$

where

Q = (annual) released radioactivity (Bq),

T_1 = 1 a, time of one year,

$e_{\text{kerma}}(a,i)$ = conversion coefficient for the isotropic external exposure (kerma-in-free-air to effective dose) for adults (a) and infants (i),

ε = 1.602×10^{-13} J·MeV^{-1}, conversion factor from MeV to J,

ρ_{air} = air density (1.293 kg·m^{-3} at STP),

k_s = average indoor shielding factor (0.4 for long-term releases, 1.0 for short-term releases),

i = running number of a photon line,

$\mu_{a,i}$ = linear energy *absorption* coefficient in the air with the density ρ_{air} for the photon line i (m^{-1}),

E_i = energy of the photon line i (MeV),

Y_i = yield of the photon line i (absolute, i.e. 1 for 100%),

μ_i = linear *attenuation* coefficient in the air with the density ρ_{air} for the photon line i (m^{-1}),

r = distance of the volume element $d^3\mathbf{r}$ from the receptor (m),

$\chi(\mathbf{r})$ = dispersion factor at a place with a radius vector $\mathbf{r}$, (s·m^{-3}),

$B(E_i, \mu_i r)$ = dose build-up factor as a function of the energy E_i of the photon line i and the number of mean free paths from the volume element $d^3\mathbf{r}$ to the receptor $\mu_i r$.

The conversion factors e_{kerma} can be taken from the ICRU Report 57 [8] for adults. For one-year old infants, the 1994 data by Yamaguchi [9] can be used. Figure 2.4 compares the data sets for one-year old infants and adults from Ref. [9] with the data set for adults from Ref [8]. The two latter data sets agree reasonably well with each other, giving credibility to Yamaguchi's results for one-year old infants. The conversion factors for infants are higher than those for adults; hence the calculated effective dose is always higher for infants. The conversion factors are highest for neonates. For example, for 511 keV photons, the factor is higher by 7% compared to one-year old infants. This difference can be neglected, however, while strictly following the directive HSK-R-11 that recommends examining effective doses to one-year old infants and adults. A neonate will become a one-year old infant during long-term releases so that the difference in the annual effective doses becomes even smaller for this release scenario.

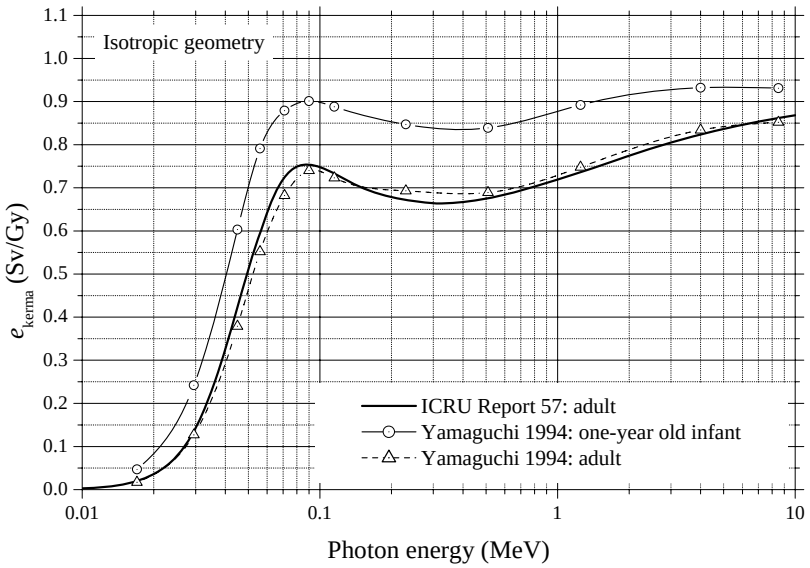

Figure 2.4: Conversion coefficients for the external exposure [8, 9].

The values of mass absorption and attenuation coefficients for air can be downloaded from Ref. [10]. The linear absorption and attenuation coefficients can be derived from the mass coefficients by multiplying with the air density. They are shown for the STP[2] air in Figure 2.5.

The dose build-up factor can be interpolated from values determined by Chilton et al. [11] in a Monte Carlo simulation. A simpler approach is to use a linear build-up factor:

$$B_{\text{lin}}(E,\mu r) = 1 + k\mu r \ , \tag{2.62}$$

where the linear coefficient k depends on the photon energy through the photon attenuation and absorption coefficients, μ and μ_a, respectively:

$$k = \frac{\mu - \mu_a}{\mu} \ . \tag{2.63}$$

The Chilton's build-up factors are about 20% higher for photon energies around 100 keV but very close to the linear build-up factors for other energies.

[2] T = 273.15 K (0°C), P = 1.013 × 10^5 Pa.

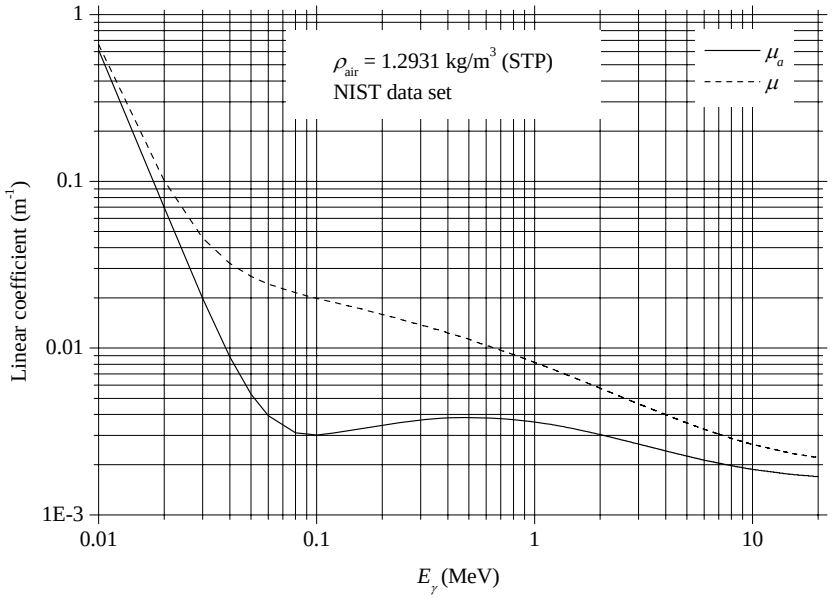

Figure 2.5: Linear gamma absorption and attenuation coefficients for STP air [10].

The integral

$$\int \frac{e^{-\mu r} f(\mathbf{r})}{r^2} d^3\mathbf{r} \tag{2.64}$$

can be transformed into polar coordinates:

$$4\pi \int_0^\infty e^{-\mu r} f(\mathbf{r}) dr \ , \tag{2.65}$$

and integrated by using a Monte Carlo method with biasing the length of the radius vector r. Assuming $\mathbf{r}$ generated isotropically so that the radius vector r follows the probability distribution

$$dP/dr = \mu e^{-\mu r} \ , \tag{2.66}$$

the integral transforms to

$$\frac{4}{\pi}\frac{1}{N_0}\int_1^{N_0} f(\mathbf{r}) dN \ , \tag{2.67}$$

where N is the current number of a Monte Carlo event and N_0 is the total number of Monte Carlo events. The continuous integral is computed as a discrete sum:

$$\frac{4\pi}{\mu}\frac{1}{N_0}\sum_{i=1}^{N_0} f(\mathbf{r}_i) \ , \tag{2.68}$$

where $\mathbf{r}_i$ is the i^{th} generated radius vector. The length of the radius vector r can be easily generated using the formula:

$$r = -\frac{1}{\mu}\ln\left[U(0,1)\right] \ , \tag{2.69}$$

where $U(0,1)$ is a number from a series of random numbers uniformly distributed on the interval (0,1).

The dose $D_{\mathrm{S(L)},p\gamma}(a,i)$ expressed in terms of Monte Carlo events is then given as

$$D_{\mathrm{S(L)},p\gamma}(a,i) = \frac{Q}{T_1} e_{\mathrm{kerma}}(a,i)\frac{\varepsilon}{\rho_{\mathrm{air}}} k_{\mathrm{s}} \sum_i \left(\frac{\mu_{a,i}}{\mu_i} E_i Y_i \cdot \frac{1}{N_0}\sum_{j=1}^{N_0} B\chi\right) \ . \tag{2.70}$$

It is assumed in Eq. 2.65 and all equations derived from it that the radius vectors are sampled from the whole space around the receptor and that the receptor can be above the ground level (above the plume), for example at the top floor of a house. The dispersion factor χ must be set to zero below the ground level. Such radius-vector samples do not contribute to the Monte Carlo sum (2.70), although they would contribute to the total number of Monte Carlo events N_0. If only receptors on the ground level are modeled, the radius-vector sampling from the upper half-space is sufficient. In that case Eq. 2.65 and all equations derived from it must be divided by 2.

For long-term release scenarios the dispersion factor is sampled according to the distribution of weather situations with the probability of a given situation to occur. A weather situation is characterized by the average wind speed, the average wind speed vector and the stability class. For the recommended 72 wind sectors, the wind speed vector shall be uniformly sampled from the 5-degree sector on the interval of ±2.5 degrees. In this way the Monte Carlo integration resembles an analogue Monte Carlo simulation.

The convergence of the Monte Carlo integral depends upon the smoothness of the integrand. The integral converges slower for situations with narrower plumes, offside the plume, for gamma energies around 0.1 MeV, where the build-up factor changes at highest with the number of mean free photon paths, and for receptors close to the source. During the calculation the total number of events to be processed shall be divided into, for example, 100 batches and the values of the integral after each batch shall be stored successively. The last value is assumed to be the best estimate and the maximum relative differences from the last value after the 50^{th} and the 75^{th} batch shall be evaluated. Both values should be smaller than few % and the 75^{th} relative difference should be smaller than the 50^{th} one to assume that the convergence of the Monte Carlo integral is satisfactory. For receptors placed in the plume or in a zone affected by frequent winds, an acceptable

convergence usually occurs after 5×10^5 events for short-term release limit calculations, when the receptor is in the plume centreline, after about 10^6 events for actual or incidental short-term releases, when the receptor may be offside the plume centreline, and after about 10^7 events for long-term releases. These numbers of Monte Carlo events were derived from a series of detailed assessments of the convergence evolution and they shall be used as default ones. The result should be discarded if no reasonable convergence is achieved after more than 10^8 events. This may indicate unsuitable receptor position and the case should be investigated separately assuming a different model, e.g. the sector-averaged model.

Too numerous photon lines can be grouped into energy bins positioned uniformly on the logarithmic energy scale. All lines within a bin are summed up giving the weighted mean bin energy

$$E_b = \frac{\sum_k E_k Y_k}{\sum_k Y_k} \ , \tag{2.71}$$

on which the absorption and attenuation coefficients as well as the dose build-up factor depend, and the total bin yield

$$Y_b = \sum_k Y_k \ . \tag{2.72}$$

The energies dividing the bins can be set equidistant on the logarithmic scale, such as

$$E_m = E_{m0}(1+a)^m \ , \tag{2.73}$$

starting with E_{m0} = 0.01 MeV. The value of $a = 0.2$ provides a sufficiently detailed binning. The interval from 0.01 MeV to 20 MeV (a wide range including radionuclides like ^{16}N) is then covered by 42 bins. Bins with the energy radiated per decay, which falls below some low percentage (e.g. 0.1%) of the total photon energy emitted per decay, can be discarded. Usually no more than 15 energy bins are necessary for a single radionuclide. Numerous X-rays of heavy radionuclides usually fall within few bins. Processing of 10^6 events lasts several seconds on presently available desktop personal computers.

The organ dose from external beta radiation to the skin is derived from the activity density at the receptor's place, which is directly proportional to the dispersion factor. The range of beta radiation in air does not exceed several metres and the spatial variations of the activity density in the plume can be neglected. For short-term and long-term releases it is given as

$$d_{\mathrm{S,ibs}} = \frac{Q}{\alpha T_1} k_{\mathrm{s}} \chi_{\mathrm{S}} e_{\mathrm{ibs}} \ , \tag{2.74}$$

$$d_{\mathrm{L,ibs}} = \frac{Q}{\alpha T_1} f_{\mathrm{oc}} k_{\mathrm{s}} \chi_{\mathrm{L}} e_{\mathrm{ibs}} \ , \tag{2.75}$$

respectively, where

Q = (annual) released radioactivity (Bq),

α = 3.16×10^7 s·a^{-1}, conversion factor from years to seconds,

T_1 = 1 a, time of one year,

k_s = 1.0 for short-term releases and $k_s = 0.4$ for long-term releases, shielding factor accounting for time spent indoors during which the exposure is reduced. It is assumed that the receptor is outdoors during a short-term release.

f_{oc} = long-term occupancy factor (1.0 for the full-time presence in the area of concern),

χ_S = short-term dispersion factor (s·m^{-3}),

χ_L = long-term dispersion factor (s·m^{-3}),

e_{ibs} = dose conversion factor for external organ dose to skin [(Sv·a^{-1})·(Bq·m^{-3})$^{-1}$].

If the dose conversion factor e_{ibs} is not available for a given radionuclide in the literature, an approximating empirical formula can be used instead [12]:

$$e_{ibs} = 1.96 \times 10^{-6} \overline{E}_\beta \ , \qquad (2.76)$$

where $\overline{E}_\beta$ is the mean kinetic energy of beta particles emitted per decay in MeV and e_{ibs} is given in [(Sv·a^{-1})·(Bq·m^{-3})$^{-1}$].

For (almost) **pure beta** (alpha) **emitters**, the semi-infinite cloud model is preferred according to which the annual external effective doses (Sv·a^{-1}) are given by simple formulae for short-term and long-term releases, respectively [1]:

$$D_{S,sic} = \frac{Q}{\alpha T_1} k_s \chi_S e_{imm} \ , \qquad (2.77)$$

$$D_{L,sic} = \frac{Q}{\alpha T_1} f_{oc} k_s \chi_L e_{imm} \ , \qquad (2.78)$$

where

Q = (annual) released radioactivity (Bq),

α = 3.16×10^7 s·a^{-1}, conversion factor from years to seconds,

T_1 = 1 a, time of one year,

k_s = 1.0 for short-term releases and $k_s = 0.4$ for long-term releases, shielding factor accounting for the time spent indoors during which the exposure is reduced. With $k_s = 1.0$ it is assumed that the receptor is outdoors during a short-term release.

f_{oc} = long-term occupancy factor (1.0 for full time presence in the area of concern),

χ_S = short-term dispersion factor ($s \cdot m^{-3}$),

χ_L = long-term dispersion factor ($s \cdot m^{-3}$),

e_{imm} = effective dose conversion factor $[(Sv \cdot a^{-1}) \cdot (Bq \cdot m^{-3})^{-1}]$.

Nevertheless, for these radionuclides, the effective dose is calculated also using Eqs. 2.59 or 2.60 and the higher value is taken. It is recommended to calculate the effective doses by using the semi-infinite cloud model also for photon emitters and to compare the results of the two models.

The semi-infinite cloud model may lead to a sever underestimation for gamma radionuclides at places below and close to tall stacks where the dispersion factor may be too low, as the plume does not reach the ground level but the plume parts with high activity densities just above the receptor may irradiate the receptor efficiently. By contrast, for a receptor in the middle of a narrow plume, the semi-infinite cloud model can severely overestimate because a high activity density, which is localised around the plume centreline, is supposed to occur in the whole half-space above the receptor.

The model for photon emitters (Eqs. 2.59, 2.61 and 2.74) was validated by comparing its results for a constant dispersion factor in the half-space above the receptor with results given by the semi-infinite cloud model (Eq. 2.77) with pre-calculated effective dose conversion factors from the literature. Such comparison is recommended also for testing the Monte Carlo computer code. An extensive data set of immersion dose conversion factors e_{imm} can be found in the US EPA Report 402-R-93-081 [13]. In this work dose conversion factors for adult phantoms and for a large number of radionuclides are computed in a very detailed way. The dosimetric quantity is the effective dose equivalent as defined in the ICRP Publication 26 [14] that, however, agrees well with the effective dose as defined in the ICRP Publication 60 [7] for external exposure. Alternatively, the dose conversion factors for adults and for limited number of radionuclides given in the directive HSK-R-41 [1] can be used. They are based on the organ dose conversion factors calculated by Kocher in 1981 [15] that were weighted by the weighting factors defined in the ICRP Publication 60 [7].

Figure 2.6 shows results of such exercise expressed in terms of the ratio to the reference dose per unit release, which is defined as the dose calculated from the semi-infinite cloud model by using the US EPA dose conversion factors [13]. Radionuclides were selected, which occur in airborne releases from accelerator facilities (^{11}C, ^{13}N, ^{15}O, ^{41}Ar, ^{7}Be, ^{24}Na) and which complete the gamma energy range (^{241}Am, ^{57}Co – lower energies; ^{24}Na – higher energies). Figure 2.6 includes also ratios for the data calculated by using the dose conversion factors from the directive HSK-R-41 [1]. The doses calculated by using the present model agree well with the results of the semi-infinite cloud model, in general within ±10%. It seems that the present model systematically overestimates for radionuclides emitting photons with lower energies (^{57}Co, ^{241}Am), probably due to the factorization of the kerma-in-free-air to effective dose conversion coefficients and the dose build-up factors in Eq. 2.61. Note that the conversion coefficients decrease rapidly with photon energy below 90 keV (Figure 2.4). The dose built up from secondary processes like Compton scattering in the air is due to photons with lower energies than the original photon energy, yet the greater conversion coefficient applicable to the original photon energy is used. Nevertheless, such behaviour is still acceptable because it is (1) conservative and (2) the external exposure is less important for radionuclides emitting low-energy photons.

The present model does not include bremsstrahlung of beta electrons that is taken into account in calculations of the dose conversion factors e_{imm}. Apparently, this contribution can be neglected for radionuclides emitting sufficiently strong photon radiation. By contrast, one should be cautious when using the present model for beta emitters with weak photon radiation.

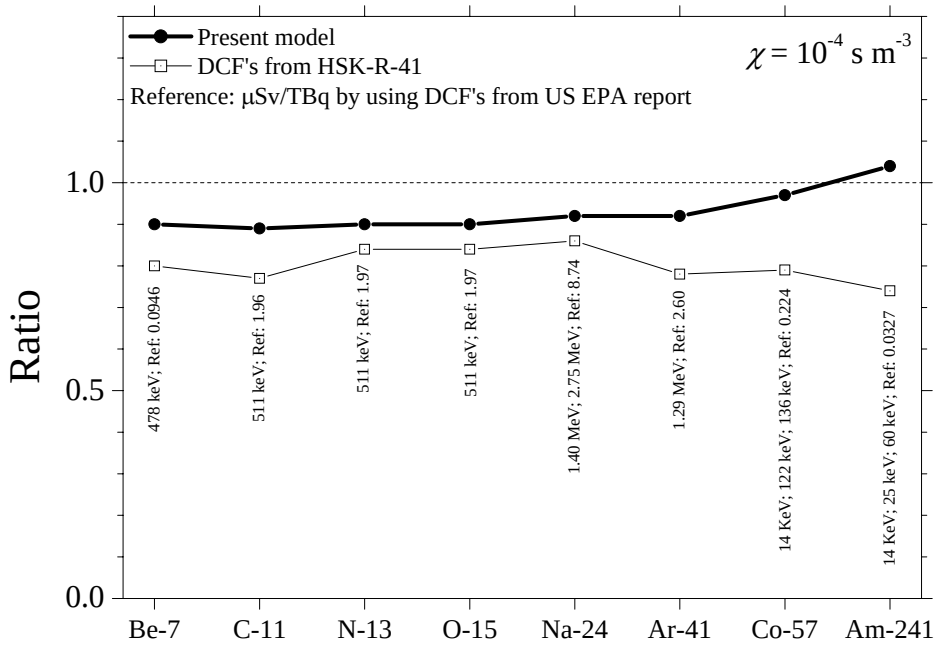

Figure 2.6: Comparison of the present external exposure model (dose kernel integration over the plume) applied to a constant dispersion factor (10^{-4} s·m^{-3}) with the semi-infinite cloud model. The reference values of doses per unit release were calculated by using the dose conversion factors e_{imm} from Ref. [13]. The squares correspond to the semi-infinite cloud model but with dose conversion factors e_{imm} from the directive HSK-R-41 [1]. The data labels list the photon energy lines and give the reference values in µSv/TBq.

2.7.2. External dose from ground deposition

Radionuclides deposited on the ground may contribute to the total effective dose by external exposure. This effective dose is not supposed to be age-specific in the present model. The external dose is integrated over the exposure time $T_{\mathrm{exp}} = 1\,\mathrm{a}$; hence the evolution of the deposited area activity density during the exposure time shall be accounted for. The annual effective dose due to ground deposition from a short-term release, $D_{\mathrm{S,gnd}}$ (Sv·a^{-1}), and the annual effective dose due to ground deposition from a long-term release, $D_{\mathrm{L,gnd}}$ (Sv·a^{-1}), are given as [1]

$$D_{\mathrm{S,gnd}} = Q f_{\mathrm{oc}} k_{\mathrm{s}} \xi_{\mathrm{S}} \left\{ 0.63 \frac{1-\exp\left[-(\lambda+\lambda_{\mathrm{F}})T_{\mathrm{exp}}\right]}{\lambda+\lambda_{\mathrm{F}}} + 0.37 \frac{1-\exp\left[-(\lambda+\lambda_{\mathrm{S}})T_{\mathrm{exp}}\right]}{\lambda+\lambda_{\mathrm{S}}} \right\} e_{\mathrm{gnd}} \,, \tag{2.79}$$

$$D_{\mathrm{L,gnd}} = Q f_{\mathrm{oc}} k_{\mathrm{s}} e_{\mathrm{gnd}} \xi_{\mathrm{L}} \times \tag{2.80}$$

$$\left\{ \left[0.63 \frac{1-\exp\left[-(\lambda+\lambda_{\mathrm{F}})T_{\mathrm{op}}\right]}{\lambda+\lambda_{\mathrm{F}}} + 0.37 \frac{1-\exp\left[-(\lambda+\lambda_{\mathrm{S}})T_{\mathrm{op}}\right]}{\lambda+\lambda_{\mathrm{S}}} \right] \cdot \frac{1-e^{-\lambda T_{\mathrm{exp}}}}{\lambda} + \frac{1}{\lambda}\left(T_{\mathrm{exp}} - \frac{1-e^{-\lambda T_{\mathrm{exp}}}}{\lambda} \right) \right\} ,$$

respectively, where

Q = (annual) released radioactivity (Bq),

f_{oc} = long-term occupancy factor (1.0 for full time presence in the area of concern),

k_s = for long-term releases, shielding factor accounting for the time spent indoors during which the exposure is reduced. $k_s = 0.4$ for long-term releases [1] and it has a special form for short-term releases (see below).

e_{gnd} = dose conversion factor for the external exposure from ground deposition $[(Sv \cdot a^{-1}) \cdot (Bq \cdot m^{-2})^{-1}]$,

and the symbols in the parentheses have the same meanings as in Section 2.6.2.

The directive HSK-R-41 [1] assumes a value of $k_s = 0.4$ also for short-term releases. This may not be sufficiently conservative for very short-lived radionuclides attached to aerosols that can be released from accelerator facilities. The receptor may be outdoors (e.g. in a garden or in picnic) most of the time during which the radionuclide lives deposited on the ground. By contrast, a value of 0.4 is reasonable for longer-lived radionuclides like ^{7}Be ($T_{1/2}$ = 53.3 d) because nobody will spend a number of days necessary for a long-term exposure outdoors. The Safety Commission suggests a model defining k_s for short-term releases as follows:

$$k_s = 1.0 - \frac{0.6}{3.1} T_{1/2} \qquad \text{for } T_{1/2} \le 3.1 \text{ d}, \tag{2.81a}$$

$$k_s = 0.4 \qquad \text{for } T_{1/2} > 3.1 \text{ d}. \tag{2.81b}$$

The radionuclide physical half-life $T_{1/2}$ is given in days in Eq. 2.81a. The breaking point of 3.1 d is derived from the criterion that the dose given during the next day shall not be smaller than 80% of the dose given during the current day.

In numerical calculations, the last two exponential expressions in Eq. 2.80 are replaced with the limit values for very long-lived radionuclides (e.g. $\lambda T_{exp} < 10^{-4}$):

$$\lim_{\lambda \to 0} \frac{1 - e^{-\lambda T_{exp}}}{\lambda} = T_{exp} \ , \tag{2.82}$$

$$\lim_{\lambda \to 0} \frac{1}{\lambda} \left(T_{exp} - \frac{1 - e^{-\lambda T_{exp}}}{\lambda} \right) = \frac{T_{exp}^2}{2} \ . \tag{2.83}$$

Obviously, the dose due to ground deposition is zero for tritium and gases including $^{14}CO_2$.

2.7.3. Inhalation dose

The effective dose due to inhalation of radioactive air is age-specific. The inhalation doses for adults (a) and infants (i) and for short-term and long-term releases, $D_{S,inh}(a,i)$ and $D_{L,inh}(a,i)$ (Sv), shall be calculated according to the following formulae, respectively [1]:

$$D_{\mathrm{S,inh}}(a,i) = f_{\mathrm{inh}} \cdot Q \cdot \chi_{\mathrm{S}} \cdot S_{\mathrm{i}} \cdot U_{\mathrm{inh}}(a,i) \cdot e_{\mathrm{inh}}(a,i) + D_{\mathrm{inh}}^{\mathrm{res}}(a,i) \ , \quad (2.84)$$

$$D_{\mathrm{L,inh}}(a,i) = f_{\mathrm{inh}} \cdot Q \cdot f_{\mathrm{oc}} \cdot \chi_{\mathrm{L}} \cdot U_{\mathrm{inh}}(a,i) \cdot e_{\mathrm{inh}}(a,i) \ , \quad (2.85)$$

where

f_{inh} = inhalable fraction of the released material. It equals to 1 for tritium and gases including $^{14}CO_2$ by definition but may have a value smaller than 1 for aerosol and iodine.

Q = (annual) released radioactivity (Bq),

f_{oc} = occupancy factor in the area of concern,

χ_{S} = short-term dispersion factor at the receptor's height ($s \cdot m^{-3}$),

χ_{L} = long-term dispersion factor at the receptor's height ($s \cdot m^{-3}$),

S_{i} = scaling factor for increased inhalation rate of adults during an incident/accident (see later),

$U_{\mathrm{inh}}(a)$ = 2.3×10^{-4} $m^3 \cdot s^{-1}$, inhalation rate of adults [1],

$U_{\mathrm{inh}}(i)$ = 6.0×10^{-5} $m^3 \cdot s^{-1}$, inhalation rate of infants [1],

$e_{\mathrm{inh}}(a,i)$ = inhalation dose conversion factor for adults (a) or infants (i), respectively, ($Sv \cdot Bq^{-1}$),

$D_{\mathrm{inh}}^{\mathrm{res}}(a,i)$ = inhalation dose due to resuspension to adults (a) or infants (i) (Sv).

During the first 8 hours of an incident/accident, an increased inhalation rate of adults (from 2.3×10^{-4} $m^3 \cdot s^{-1}$ to 3.3×10^{-4} $m^3 \cdot s^{-1}$, that is by a factor of 1.4348) shall be assumed, accounting for an increased physical activity due to panic or emergency measures (e.g. evacuation) [1]. The *adult* inhalation rate is adjusted by multiplying it by the factor S_{i}, which is derived from the release duration t_{rel} according to Table 2.6. The factor S_{i} is different from 1 *only for adults and the incident/accident scenario.*

Table 2.6: Inhalation rate scaling factor for adults during an incident/accident.

t_{rel} (hours)	S_{i}
≤ 8	1.4348
> 8	$\frac{1.4348 \times 8 + 1 \times (t_{\mathrm{rel}} - 8)}{t_{\mathrm{rel}}}$

During long-term releases the prevailing contribution to the inhalation dose originates from the chronic inhalation of the plume air and the resuspension effect is negligible. However, after a short-term release, when the plume has disappeared, there may be a prolonged inhalation exposure from resuspended radioactivity and the effective dose resulting form it can be comparable to the direct

inhalation effective dose from the plume. The inhalation dose due to resuspension of material released during a short-term release is given for aerosol and iodine by the following formula [1]:

$$D_{\mathrm{inh}}^{\mathrm{res}}(a,i) = Q f_{\mathrm{oc}} \xi_{\mathrm{S}} \left(K_0 \frac{1-\exp\left[-(\lambda+L)T_{\mathrm{exp}}\right]}{\lambda+L} + K_{\mathrm{e}} \frac{1-\exp(-\lambda T_{\mathrm{exp}})}{\lambda} \right) \alpha T_1 U_{\mathrm{inh}}(a,i) e_{\mathrm{inh}}(a,i) \ , \qquad (2.86)$$

where

Q = (annual) released radioactivity (Bq),

f_{oc} = occupancy factor in the area,

ξ_{S} = short-term deposition factor (m^{-2}),

K_0 = 10^{-6} m^{-1}, first resuspension factor [1],

K_{e} = 10^{-9} m^{-1}, second resuspension factor [1],

λ = radioactive decay constant (a^{-1}),

L = 3.7 a^{-1}, decay constant for immobilisation on the ground,

T_{exp} = exposure time,

α = 3.16×10^7 $s \cdot a^{-1}$, conversion factor from years to seconds,

T_1 = 1 a, time of one year,

$U_{\mathrm{inh}}(a,i)$ = inhalation rate for adults (a) or infants (i) ($m^3 \cdot s^{-1}$),

$e_{\mathrm{inh}}(a,i)$ = inhalation dose conversion factor for adults (a) or infants (i) ($Sv \cdot Bq^{-1}$).

As most of the resuspension exposure time passes after the incident, normal adult inhalation rates shall be assumed in the model also for incidents/accidents. Because the exposure time is long, one assumes the occupancy factor to account for a partial residence time in the affected area. The exposure time T_{exp} shall be set to 50 years for controlled short-term releases and to 1 year for incidental/accidental releases. After an incident/accident, measurements from the site should be available and a decision about a clean-up taken if the doses to the population were unacceptable. The clean-up work can last one year. It is also recommended to check the exposure due to the resuspension without the clean-up by running the model as for an actual short-term release with the 50-year exposure time. Because an infant will not remain an infant during the 50-year exposure, the dose to a growing infant is between the dose for a 1-year old infant and the one for an adult. To account for this effect, the dose to infant is conservatively set to the dose for an adult if the latter one is greater for the exposure time of 50 years:

$$D_{\mathrm{inh}}^{\mathrm{res}}(i) = \max\left[D_{\mathrm{inh}}^{\mathrm{res}}(i), D_{\mathrm{inh}}^{\mathrm{res}}(a)\right] \qquad \text{for } T_{\mathrm{exp}} = 50\,\mathrm{a}\,. \qquad (2.87)$$

Resuspended iodine is conservatively assumed to be all in the aerosol form. Of course, resuspension does not take place for tritium and gases including $^{14}CO_2$.

2.7.4. Ingestion dose

The ingestion dose due to a *short-term* release, $D_{\mathrm{S,ing}}(a,i)$ (Sv·a^{-1}), to adults (a) or infants (i) is a sum of three components:

$$D_{\mathrm{S,ing}}(a,i) = f_{\mathrm{veg}} D_{\mathrm{S,veg}}(a,i) + f_{\mathrm{mt}} D_{\mathrm{S,mt}}(a,i) + f_{\mathrm{mi}} D_{\mathrm{S,mi}}(a,i) \ , \tag{2.88}$$

where

f_{veg} = fraction of consumed vegetables produced in the area of concern,

f_{mt} = fraction of consumed meat produced in the area of concern,

f_{mi} = fraction of consumed milk produced in the area of concern,

$D_{\mathrm{S,veg}}(a,i)$ = effective dose due to ingestion of vegetables (Sv·a^{-1}),

$D_{\mathrm{S,mt}}(a,i)$ = effective dose due to ingestion of meat (Sv·a^{-1}),

$D_{\mathrm{S,mi}}(a,i)$ = effective dose due to ingestion of milk (Sv·a^{-1}).

For a short-term release *during the plant-growing period* ($T_{\mathrm{p}} \le 0.5$ a), the consumed vegetables and fodder fed to animals include a part directly contaminated during the release, which is consumed during the same year. However, a part resulting from the root uptake will be ingested only in the next year. To account for this delay, the Directive HSK-R-41 assumes for one-year old children the consumption rates and dose conversion factors as for adults in terms describing the ingestion dose due to the root uptake [1]. In this view the effective doses defined above are given as

$$D_{\mathrm{S,veg}}(a,i) = A^0_{\mathrm{veg,lv}} \frac{1}{T_1}\left[\frac{1}{\lambda_{\mathrm{e,lv}}} + e^{-\lambda T_{\mathrm{p}}} \cdot \frac{2}{\lambda_{\mathrm{lv}} T_1} \cdot \frac{1-e^{-\lambda T_1/2}}{\lambda}\right] U_{\mathrm{veg}}(a,i) e_{\mathrm{ing}}(a,i) + A^0_{\mathrm{veg,r}} \frac{1}{T_1} \cdot \frac{e^{-\lambda_{\mathrm{e,r}}(T_{\mathrm{p}}+T_1/2)} - e^{-\lambda_{\mathrm{e,r}} T_{\mathrm{inc}}}}{\lambda_{\mathrm{e,r}}} U_{\mathrm{veg}}(a) e_{\mathrm{ing}}(a) \ , \tag{2.89}$$

$$D_{\mathrm{S,mt}}(a,i) = \left\{ A^0_{\mathrm{veg,lv}} \frac{1}{T_1}\left[\frac{1}{\lambda_{\mathrm{e,lv}}} + e^{-\lambda T_{\mathrm{p}}} \cdot \frac{2}{\lambda_{\mathrm{lv}} T_1} \cdot \frac{1-e^{-\lambda T_1/2}}{\lambda}\right] U_{\mathrm{mt}}(a,i) e_{\mathrm{ing}}(a,i) + A^0_{\mathrm{veg,r}} \frac{1}{T_1} \cdot \frac{e^{-\lambda_{\mathrm{e,r}}(T_{\mathrm{p}}+T_1/2)} - e^{-\lambda_{\mathrm{e,r}} T_{\mathrm{inc}}}}{\lambda_{\mathrm{e,r}}} U_{\mathrm{mt}}(a) e_{\mathrm{ing}}(a) \right\} V_{\mathrm{fod}} C_{r,\mathrm{fod\text{-}mt}} \exp(-\lambda T_{\mathrm{mt}}) \ , \tag{2.90}$$

$$D_{\mathrm{S,mi}}(a,i) = \left\{ A^0_{\mathrm{fod,lv}} \frac{1}{T_1}\left[\frac{1}{\lambda_{\mathrm{e,lv}}} + e^{-\lambda T_{\mathrm{p}}} \cdot \frac{2}{\lambda_{\mathrm{lv}} T_1} \cdot \frac{1-e^{-\lambda T_1/2}}{\lambda}\right] U_{\mathrm{mi}}(a,i) e_{\mathrm{ing}}(a,i) + \right.$$

$$A^0_{\mathrm{fod,r}} \frac{1}{T_1} \cdot \frac{e^{-\lambda_{\mathrm{e,r}}(T_\mathrm{p}+T_1/2)} - e^{-\lambda_{\mathrm{e,r}}T_{\mathrm{inc}}}}{\lambda_{\mathrm{e,r}}} U_{\mathrm{mi}}(a) e_{\mathrm{ing}}(a) \Bigg\} V_{\mathrm{fod}} C_{r,\mathrm{fod\text{-}mi}} \exp(-\lambda T_{\mathrm{mi}}) \ , \qquad (2.91)$$

where

$U_{\mathrm{veg}}(a,i)$ = average annual consumption of vegetables by adults (a) or infants (i) (kg·a^{-1}),

$U_{\mathrm{mt}}(a,i)$ = average annual consumption of meat by adults (a) or infants (i) (kg·a^{-1}),

$U_{\mathrm{mi}}(a,i)$ = average annual consumption of milk by adults (a) or infants (i) (kg·a^{-1}),

$e_{\mathrm{ing}}(a,i)$ = ingestion dose conversion factor for adults (a) or infants (i) (Sv·Bq^{-1}),

T_{mt} = 5.5×10^{-2} a, typical storage time of meat (20 days) [1],

T_{mi} = 2.7×10^{-3} a, typical storage time of milk (1 day) [1],

and the other symbols have the same meanings as in Sections 2.6.3 and 2.6.4. The radioactive decay during the food storage time is assumed as well. The average annual consumptions of vegetables, meat and milk in Switzerland are given in Table 2.7 [1].

Table 2.7: Average consumptions of vegetables, meat and milk in Switzerland [1].

Food product	Adults (kg·a^{-1})	Infants (kg·a^{-1})
Vegetables (U_{veg})	225	60
Meat (U_{mt})	75	20
Milk (U_{mi})	160	200

For a short-term release *during the wintertime* ($T_\mathrm{p} > 0.5$ a) the mass activity densities need not to be written explicitly, as only the radioactivity uptake from the root zone takes place. The adult consumption rates and dose conversion factors are assumed also for infants [1]:

$$D_{\mathrm{S,veg}}(a,i) = A_{m,\mathrm{S\text{-}veg}} U_{\mathrm{veg}}(a) e_{\mathrm{ing}}(a) \ , \qquad (2.92)$$

$$D_{\mathrm{S,mt}}(a,i) = A_{m,\mathrm{S\text{-}mt}} U_{\mathrm{mt}}(a) e_{\mathrm{ing}}(a) \exp(-\lambda T_{\mathrm{mt}}) \ , \qquad (2.93)$$

$$D_{\mathrm{S,mi}}(a,i) = A_{m,\mathrm{S\text{-}mi}} U_{\mathrm{mi}}(a) e_{\mathrm{ing}}(a) \exp(-\lambda T_{\mathrm{mi}}) \ , \qquad (2.94)$$

where $A_{m,\mathrm{S\text{-}veg}}$, $A_{m,\mathrm{S\text{-}mt}}$ and $A_{m,\mathrm{S\text{-}mt}}$ are the corresponding mass activity densities in vegetables, meat and milk as described in Sections 2.6.3 and 2.6.4, respectively, and the other symbols have the same meaning as above.

For a *long-term* release, the ingestion effective doses can be calculated simply from the mass activity densities in vegetables, meat and milk, $A_{m,\text{L-veg}}$, $A_{m,\text{L-mt}}$ and $A_{m,\text{L-mi}}$ (Sections 2.6.3 and 2.6.4), respectively [1]:

$$D_{\text{L,veg}}(a,i) = A_{m,\text{L-veg}} U_{\text{veg}}(a,i) e_{\text{ing}}(a,i) \ , \tag{2.95}$$

$$D_{\text{L,mt}}(a,i) = A_{m,\text{L-mt}} U_{\text{mt}}(a,i) e_{\text{ing}}(a,i) \exp(-\lambda T_{\text{mt}}) \ , \tag{2.96}$$

$$D_{\text{L,mi}}(a,i) = A_{m,\text{L-mi}} U_{\text{mi}}(a,i) e_{\text{ing}}(a,i) \exp(-\lambda T_{\text{mi}}) \ . \tag{2.97}$$

The total ingestion effective dose is then given as

$$D_{\text{L,ing}}(a,i) = f_{\text{veg}} D_{\text{L,veg}}(a,i) + f_{\text{mt}} D_{\text{L,mt}}(a,i) + f_{\text{mi}} D_{\text{L,mi}}(a,i) \ , \tag{2.98}$$

where the fractions of food produced in the area of concern, f_{veg}, f_{mt} and f_{mi} are considered for vegetables, meat and milk, respectively.

2.8 Uncertainties of the models

The models described above are so-called conservative generic models. Conservative means that they shall overestimate in general by making those parameters, which are uncertain, conservative. Generic means that they require the least possible site-specific input parameters. It is clear that such a complex system as the environment cannot be described and modelled to a high accuracy with generic models. This Section gives comments on the expected model accuracy for assessing critically confidence intervals of results. *Although predictions of the models may be quite uncertain, their results as such are relevant to regulatory limits and other legal obligations.*

The accuracy of Gaussian atmospheric dispersion models has been reviewed for example in Ref. [16]. Better agreement is obtained for long-term release scenarios because of averaging. For the long-term dispersion factor at a single point, agreements within a factor of 2 were reported whilst an agreement within one order of magnitude is to be expected for short-term dispersion factors [16]. The prediction of the long-term dispersion factor calculated according to Sections 2.2 and 2.5 with weather data from the Geneva airport was tested in the year 2000 when activity densities of ^{7}Be at an environmental aerosol-sampling station were well distinguishable from the natural background. The releases from the source were known and the experimental long-term dispersion factor could be determined (Eq. 2.1). The calculated long-term dispersion factor for the given situation was higher by a factor of 1.48. This is an excellent agreement taking into account that the model assumes a full reflection of the plume from the ground, which takes place for inert gases, but is only partial for aerosols. The inhalation dose depends directly on the dispersion factors; hence its relative uncertainty shall be at least that of the dispersion factors. The effects of the other parameters entering the relevant expressions shall be reasonably overestimating keeping the result on the safe side. The sensitivity and accuracy of the models for the other exposure pathways will be discussed separately.

2.8.1 External exposure from the radioactive plume

The relative uncertainty of the dose kernel integral is expected to be lower than that for a dispersion factor at a single point because of the averaging effect over the plume volume. A sensitivity test was carried out by replacing the Vogt's dispersion-coefficient data set (Table 2.1) with a NRPB data set with increased horizontal dispersion under standard conditions [17, 18]. The resulting dose kernel integrals differed by negligible 3% for long-term releases. Apparently the result is controlled by the wind statistics whilst the actual distribution of the plume is partially averaged in the dose kernel integral.

The highest uncertainty probably originates in the weather statistics. Conventional cup anemometers do not to provide reliable results with wind speeds below 1 m/s. That is why the weather situations with wind speeds below this limit are put into a single wind speed bin 0–1 m/s with an average wind speed together with pure calms (0 m/s). True calms are modelled as situations with plume transported towards the receptor with the average wind speed in the bin. This approach is conservative. A sensitivity test was carried out, considering the weather situations in the first wind speed bins as calms only. The air was rising in a vertical column to the infinity in this extreme case. The dose kernel integral was smaller by factors ranging from 1.5 to 2.6, depending on the short-lived radioactive gas assumed. For a typical radioactive-gas mixture from accelerator facilities, the uncertainty due to the treatment of calms is probably not higher than a factor of 2. The stability of the model was examined for very low wind speeds (below 1 m/s) under stable atmospheric conditions (class F) – the most problematic situation. The average wind speed assigned to the first wind speed bin was reasonably representative for the whole interval and a typical radioactive gas mixture.

The buildings in which people reside are additional sources of uncertainty. First, they pose obstacles for the air flow and increase the dispersion in the ground level layer. For situations when the receptor is in the plume centreline, the true dose should be smaller than the modelled one; that is without an additional dispersion. If the core of the plume with the highest activity densities is well above the receptor (above the roof), the most contributing, active part of the plume is not affected strongly by the building. Second, a rather uncertain indoor shielding factor of 0.4 [1] is applied for long-term releases. This factor is certainly conservative for the houses made of bricks or concrete and for people residing even on the top floors. For example, a factor of 0.2 is used for such situations in the EU countries [16]. The directive HSK-R-41 was, however, strictly followed because the factor of 0.4 accounts also for situations when people are outside in their gardens or on their balconies.

Concluding from the discussion above, the confidence interval of the effective dose due to external exposure from the radioactive plume may range from -50% to no more than +50% of the calculated value for long-term releases. For short-term releases, the confidence interval can be wider, perhaps a factor of 5 in both directions. The contribution of the external exposure from the radioactive plume is negligible for typical short-term release problems encountered at CERN, however.

2.8.2 External exposure from radioactivity deposited on the ground

The exposure from the radioactivity deposited on the ground is directly proportional to the deposition factors on the ground surface. This quantity may be extremely variable and it depends both on the actual weather situation and on the properties of the aerosol, mainly on the diameter of aerosol particles.

The Swiss directive HSK-R-41 uses so-called augmented deposition velocity for long-term releases, which is assumed to be conservative. Hence all doses derived from it shall be conservative. For short-term releases, the directive recommends a wind speed of 1 m/s and a precipitation rate of 2 mm/h [1]. The precipitation rate corresponds to persistent rain in Switzerland. Although the pollutant concentration increases inversely proportionally with the wind speed in the model, too low wind speeds lead to plume meandering in reality and the real horizontal dispersion coefficients are

greater and consequently there is less deposition per unit area. Similarly, the wet deposition increases with the power of 0.8 of the precipitation rate but very strong rains are accompanied by higher wind speeds and a turbulent air flow leading to stronger pollutant dispersion.

The wet deposition, which dominates the total deposition, depends very much on the aerosol diameter. The washout factors for 10 μm particles can be 10 times greater than those for 1 μm particles [19]. The directive HSK-R-41 assumes particles of 1 μm diameter typical for the aerosols in the environment. If fine aerosol filters are used in the ventilation outlets, particles greater than 0.3 μm in diameter could be retained, for example, leading to a deposition factors smaller, perhaps, by a factor of 3.

Further, for the external exposure, an ideal case is assumed that the radioactivity is distributed uniformly on an infinite plane. This is certainly not true around houses where a part of the radioactive substances may be deposited on vertical surfaces with deposition rates lower by a factor of about 10 [19]. In addition a considerable part of the deposited activity can be washed away by the rain from the asphalted surfaces into drains in urban areas. The uncertainty of the indoor shielding factor, as discussed in Section 2.8.1, adds to the total uncertainty.

Sensitivity tests for the set of dispersion coefficients showed that the deposition can be greater by a factor of 3 for the NRPB set [17, 18] compared with the Vogt's set [5] that is used in this report for impact points at a distance not exceeding 100 m. The difference is smaller (factor 1.3) for receptor distances of several hundreds of metres. The NRPB set is known to give too narrow plumes because it is based on observations of 3-minute averages and this feature was compensated only for the stable atmospheric conditions (classes E and F) [18]. With the Vogt's set, the deposition maximum usually occurs during the stability class D because the horizontal dispersion factor is at smallest for this class and distances below 1 km (see Figure 2.2 and Eq. 2.18).

The discussion above shows that the external dose from the deposited radioactive substances is an extremely uncertain quantity, especially for short-term releases. The model gives results that are conservative within one order of magnitude.

2.8.3 Ingestion of food produced in the area of concern

In the case of the ingestion doses, uncertainties in the biological transfer models and in metabolism of living objects, humans included, and exposure models are added to the dispersion and deposition uncertainties discussed above. As an example, one can cite the range of the feed transfer coefficients into cow milk for Tc: $2.3 \times 10^{-5} - 1.1 \times 10^{-3}$ d·kg^{-1} [20]; that is a factor of 50. The situation is even more complicated for certain elements subject to the metabolic homeostasis (e.g. Na, Mg, P, K, Ca). The level of these elements in the organism is controlled by the metabolism and the intake of a given radioisotope depends on the level of the corresponding element in the fodder or food In general, a surplus of stable isotopes blocks the intake of the radioactive one. Generic coefficients are used so that the results are conservative. The conservatism is further strengthened by the maximum but realistic consumption rates assumed. On the other hand, vertical deposition and washout into drainage do not take place or are limited on fields. Averaging the deposition factors over larger areas for long-term consumption scenarios reduces the uncertainty of the average deposition factor (see Section 2.3.3). The ingestion doses can be overestimated by one order of magnitude, which was confirmed in a recent study carried out in Germany [21].

3. Water releases into a river

This Section describes models for an assessment of the environmental impact of radioactive releases from water outlets of accelerator facilities or sites into surface watercourses such as streams and rivers. During an aqueous release the radioactivity is transported with the flowing water along the river and the effluents from the site are subject to mixing and diffusion. This process depends on the river flow parameters and it is called dispersion. Further, the radioactivity from the water can be deposited in the river bottom sediment. The releases may cause exposure of people externally or through the use of the river water for drinking and/or agriculture.

3.1. Scenario

It is assumed that the radioactivity is released into a river with a constant rate during the whole year. The doses are calculated:

- As received per one year for external exposure;
- Committed (during the rest of the life) due to the radioactivity intake, which took place during the reference year.

The release took place with a constant release rate during the last 50 years ($T_{op} = 50$ a), including the current year. Doses to both adults (*a*) and one-year old infants (*i*) shall be considered.

If the annual release is Q (Bq), the release rate $\dot{Q}$ (Bq·s^{-1}) is given as

$$\dot{Q} = \frac{Q}{\alpha T_1} \tag{3.1}$$

where

α = 3.16 × 10^7 s·a^{-1}, conversion factor from years to seconds,

T_1 = 1 a, time of one year.

3.2. Activity densities in environmental matrices

3.2.1. River water

If the receptor is on the same side as the outlet at a distance $x < 7D$, where D is the river depth (m), then the radioactivity in the river water shall be considered as undiluted and hence the volumetric activity density $A_{v,W}$ (Bq·m^{-3}) in the river water equal to that in the water in the outlet shall be considered [2].

Otherwise the average volumetric activity density in the river water $A_{v,W}$ (Bq·m^{-3}) is given as

$$A_{v,W} = Q\frac{\chi_W}{\alpha T_1}\ , \tag{3.2}$$

where the water dispersion factor, χ_{W} (s·m^{-3}), at the receptor point is given as [2]

$$\chi_{\mathrm{W}}(x) = P_d \frac{1}{J} \exp\left(-\frac{\lambda x}{v}\right), \tag{3.3}$$

where

P_d = partial mixing correction factor,

J = average river flow (m^3·s^{-1}),

λ = radioactive decay constant of the radionuclide (s^{-1}),

x = downstream distance of the receptor from the outlet (m),

v = flow velocity of the receiving river (m·s^{-1}).

For a receptor located on the other side of the river than the water outlet, a complete mixing shall be conservatively assumed with $P_d = 1$. For a receptor on the same side of the river as the water outlet, the partial mixing correction factor, P_d, shall be calculated following the method given in Ref. [2]: The factor P_d depends on the partial mixing index A:

$$A = 1.5\frac{Dx}{B^2}, \tag{3.4}$$

where B is the river width (m), and its value can be obtained by interpolating of values listed in Table IV of Ref. [2] in logarithmic-linear scale. The values of P_d in dependence on the partial mixing index, A, taken from Ref. [2] are plotted in Figure 3.1.

3.2.2. River bottom sediment

Similarly, the average annual mass activity density in the river bottom sediment $A_{m,\mathrm{BS}}$ (Bq·kg^{-1}) is given as

$$A_{m,\mathrm{BS}} = Q\frac{\chi_{\mathrm{BS}}}{\alpha T_1}, \tag{3.5}$$

where χ_{BS} is the bottom sediment dispersion factor (s·kg^{-1}) given as

$$\chi_{\mathrm{BS}} = \frac{(0.1)(0.001)K_d}{1 + 0.001 C_{\mathrm{s}} K_d} \cdot \frac{1 - \exp(-\lambda T_{\mathrm{e}})}{\lambda T_{\mathrm{e}}} \chi_{\mathrm{W}}, \tag{3.6}$$

where

K_d = element-specific distribution coefficient ($l \cdot kg^{-1}$),

C_s = suspended sediment concentration ($kg \cdot m^{-3}$); the default value is 5×10^{-2} $kg \cdot m^{-3}$ [2],

λ = radioactive decay constant (a^{-1}),

T_e = effective sediment accumulation time in years; the default value is 1 a [2].

The factor 0.1 takes into account smaller values of K_d for bigger particles, which form the bottom sediment of the river. For very small decay constants (e.g. $\lambda T_e < 10^{-4}$), the build-up term shall be reduced to

$$\lim_{\lambda \to 0} \frac{1 - \exp(-\lambda T_e)}{\lambda T_e} = 1 \qquad (3.7)$$

in numerical calculations. Default values of the distribution coefficient K_d in freshwater are given for 25 elements in Ref. [2] and reproduced in Table 3.1 for the radionuclides most likely to occur in CERN's effluents.

Table 3.1: Default values of distribution coefficients K_d ($l \cdot kg^{-1}$) in freshwater for the radionuclides most likely to occur in CERN's water effluents [2].

Element (radionuclides)	K_d ($l \cdot kg^{-1}$)
H (HTO)	0
Na (^{22}Na, ^{24}Na)	0
Mn (^{52}Mn, ^{54}Mn)	1×10^3
Fe (^{59}Fe)	5×10^3
Co (^{56}Co, ^{57}Co, ^{58}Co, ^{60}Co)	5×10^3
Zn (^{65}Zn)	5×10^2
Cs (^{134}Cs)	1×10^3
Eu (^{152}Eu)	5×10^2

For *tritium, gases and highly soluble elements* (e.g. Na)

$$\chi_{BS} = 0 \qquad (3.8)$$

by definition, as these substances do not attach to particles.

The average annual area activity density of the bottom sediment, $A_{a,BS}$ ($Bq \cdot m^{-2}$), is given as

$$A_{a,BS} = Q \xi_{BS} \ , \qquad (3.9)$$

where

Q = annual released radioactivity (Bq),

ξ_{BS} = bottom sediment deposition factor (m^{-2}), which is given as

$$\xi_{\mathrm{BS}} = \frac{\rho_a}{\alpha T_1} \chi_{\mathrm{BS}} \ , \tag{3.10}$$

where ρ_a is the area density of the bottom sediment ($kg \cdot m^{-2}$). The default value of ρ_a is 60 $kg \cdot m^{-2}$, assuming a 5 cm thick layer and a bulk bottom sediment density of 1.2×10^3 $kg \cdot m^{-3}$ [2].

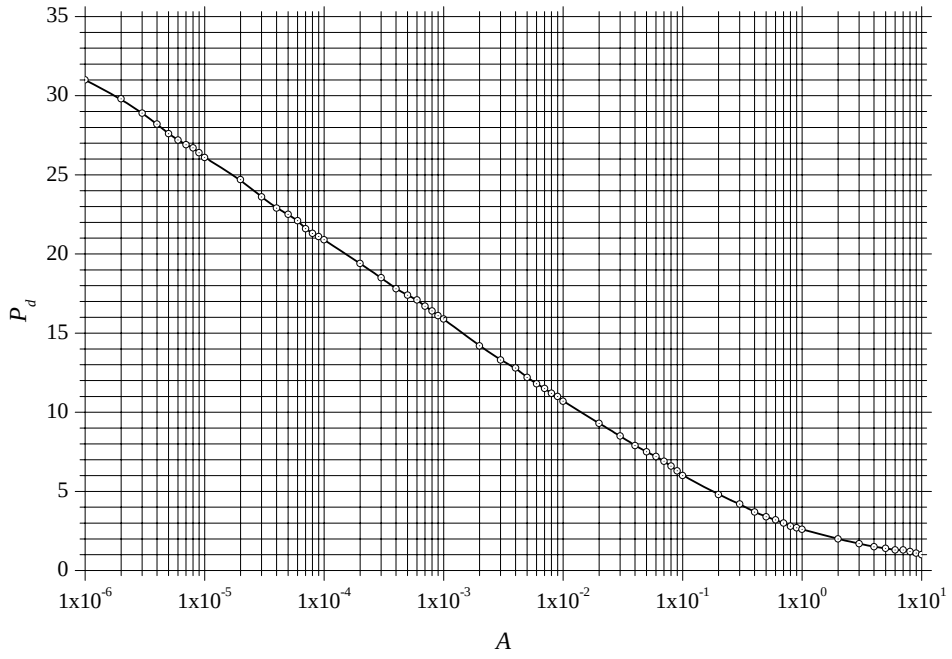

Figure 3.1: Partial mixing correction factor for rivers [2].

3.2.3. Fish

The mass activity density in fish, $A_{m,\mathrm{fi}}$ ($Bq \cdot kg^{-1}$), is given as [1, 2]

$$A_{m,\mathrm{fi}} = Q \frac{1}{P_d} \frac{\chi_{\mathrm{W}}}{\alpha T_1} C_{r,\mathrm{wa\text{-}fi}} \ , \tag{3.11}$$

where $C_{r,\mathrm{wa\text{-}fi}}$ is a concentration ratio for the uptake from water in fish ($m^3 \cdot kg^{-1}$). Fish usually does not stay on one place so the partial mixing correction factor P_d does not apply. As the dispersion

factor χ_{W} is already corrected with P_d, χ_{W} is corrected back to a form with the complete water mixing.

For *tritium* an equilibrium model is used, assuming that all water in fish originates from the river water [1]:

$$A_{m,\mathrm{fi}} = Q\frac{1}{P_d}\frac{\chi_{\mathrm{W}}}{\alpha T_1}\frac{f_{\mathrm{wa}}}{\rho_{\mathrm{wa}}} \tag{3.12}$$

where

f_{wa} = 0.75, average water content of foodstuff [1],

ρ_{wa} = 1000 kg·m^{-3}, water density.

3.2.4. Meat and milk of watered animals

The mass activity densities in meat and milk of animals watered with the river water, $A_{m,\mathrm{W\text{-}mt}}$ and $A_{m,\mathrm{W\text{-}mi}}$ (Bq·kg^{-1}), respectively are given as

$$A_{m,\mathrm{W\text{-}mt}} = Q\frac{\chi_{\mathrm{W}}}{\alpha T_1}V_{\mathrm{W}}C_{ft,\mathrm{fod\text{-}mt}}\ , \tag{3.13}$$

$$A_{m,\mathrm{W\text{-}mi}} = Q\frac{\chi_{\mathrm{W}}}{\alpha T_1}V_{\mathrm{W}}C_{ft,\mathrm{fod\text{-}mi}}\ , \tag{3.14}$$

where

V_{W} = 0.075 m^3·d^{-1}, daily water consumption of animals [1],

$C_{ft,\mathrm{fod\text{-}mt}}$ = feed transfer coefficient from fodder to meat (d·kg^{-1}),

$C_{ft,\mathrm{fod\text{-}mi}}$ = feed transfer coefficient from fodder to milk (d·kg^{-1}).

Similarly to the model for fish, the equilibrium is assumed for *tritium* except that only a fraction $(1-f_{\mathrm{fod}})$ of the water content in animals originates in drinking water if $f_{\mathrm{fod}} = 0.4$ is the fraction of water in animals received from food [1]:

$$A_{m,\mathrm{W\text{-}mt}} = Q\frac{\chi_{\mathrm{W}}}{\alpha T_1}\frac{f_{\mathrm{wa}}}{\rho_{\mathrm{wa}}}(1-f_{\mathrm{fod}})\ , \tag{3.15}$$

$$A_{m,\mathrm{W\text{-}mi}} = Q\frac{\chi_{\mathrm{W}}}{\alpha T_1}\frac{f_{\mathrm{wa}}}{\rho_{\mathrm{wa}}}(1-f_{\mathrm{fod}})\ . \tag{3.16}$$

The symbols have meanings explained above.

3.2.5. Irrigated fields

During irrigation radioactivity is deposited with the river water on fields. The external exposure due to the deposited radioactivity is calculated in the same way as for a long-term atmospheric deposition, assuming the deposition factor ξ_I (m^{-2}) given as

$$\xi_I = \frac{W_I}{\alpha T_1} \chi_W \ , \tag{3.17}$$

where

W_I = total annual amount of irrigation water in m^3 used per m^2 of the irrigated field (m),

χ_W = water dispersion factor χ_W ($s \cdot m^{-3}$) at the point where irrigation water is taken from the river.

For *tritiated water and gases,* $\xi_I = 0$ is assumed by definition due to the high mobility of these substances and taking into account that no external dose can result from tritium deposited on the ground. However, during sprinkle irrigation diluted gases may be liberated from the water and cause exposure as in case of an atmospheric release. This process is difficult to describe in general, as it depends on many factors. Fields and gardens are usually irrigated during sunny days with strong atmospheric dispersion conditions; therefore the liberated gas is quickly removed from the ground-level atmosphere above the field. It is known that for inert gases, except radon with its progenies, the inhalation dose is negligible compared with the external dose. Hence a very conservative estimate of the total dose can be made, assuming that the persons present on the field (in the garden) are exposed to external radiation from a semi-infinite volume of water with dissolved gases. If this approach would lead to a considerable dose, a special study should be conducted for gases. A particular investigation shall be done for radon and its progenies if present in the effluents.

Tritiated water will evaporate from the irrigated field and expose the persons present on the field to tritiated water vapour during the irrigation period. Mixing of the river water with tritium-free water from the precipitation is assumed. During the irrigation period, the average volumetric tritium activity in the air above the field, $A_{v,\text{I-HTO}}$ ($Bq \cdot m^{-3}$), is given as

$$A_{v,\text{I-HTO}} = Q \frac{\chi_W}{\alpha T_1} \cdot \frac{\Phi}{\rho_{wa}} \cdot \frac{1}{1 + \frac{W_p}{W_I} \frac{T_I}{T_1}} \ , \tag{3.18}$$

where

Φ = 9×10^{-3} $kg \cdot m^{-3}$, average air humidity [1],

ρ_{wa} = 1000 $kg \cdot m^{-3}$, water density,

W_p = annual precipitation in m^3 per m^2 (m),

W_{I} = total annual amount of irrigation water in m^3 used per m^2 of the irrigated field (m),

T_{I} = irrigation period in years (a), (fraction of the year during which fields are irrigated).

3.2.6. Irrigated vegetables and fodder

Except for gases, the deposited radioactivity may be incorporated into vegetables and fodder produced on the irrigated fields. Both direct contamination from leaves and incorporation through roots can take place. The mass activity densities at the time of harvest are given as [1, 2]

$$A_{m,\text{I-veg-lv}} = Q\frac{\chi_{\mathrm{W}}}{\alpha T_1}W_{\mathrm{I}}\,f_{\mathrm{d}}\,\frac{1}{B_{a,\text{veg}}}\cdot\frac{1-\exp(-\lambda_{\mathrm{e,lv}}t_{\text{veg}})}{T_{\mathrm{I}}\lambda_{\mathrm{e,lv}}}\ , \tag{3.19}$$

$$A_{m,\text{I-veg-r}} = Q\frac{\chi_{\mathrm{W}}}{\alpha T_1}W_{\mathrm{I}}\,\frac{1}{P_{a,\text{veg}}}\cdot\frac{1-\exp(-\lambda_{\mathrm{e,r}}T_{\text{op}})}{T_{\mathrm{I}}\lambda_{\mathrm{e,r}}}C_{r,\text{soil-veg}}\ , \tag{3.20}$$

$$A_{m,\text{I-fod-lv}} = Q\frac{\chi_{\mathrm{W}}}{\alpha T_1}W_{\mathrm{I}}\,f_{\mathrm{d}}\,\frac{1}{B_{a,\text{fod}}}\cdot\frac{1-\exp(-\lambda_{\mathrm{e,lv}}t_{\text{fod}})}{T_{\mathrm{I}}\lambda_{\mathrm{e,lv}}}\ , \tag{3.21}$$

$$A_{m,\text{I-fod-r}} = Q\frac{\chi_{\mathrm{W}}}{\alpha T_1}W_{\mathrm{I}}\,\frac{1}{P_{a,\text{fod}}}\cdot\frac{1-\exp(-\lambda_{\mathrm{e,r}}T_{\text{op}})}{T_{\mathrm{I}}\lambda_{\mathrm{e,r}}}C_{r,\text{soil-fod}}\ , \tag{3.22}$$

where

$A_{m,\text{I-veg-lv}}$ = mass activity density in vegetables deposited on leaves ($Bq\cdot kg^{-1}$),

$A_{m,\text{I-veg-r}}$ = mass activity density in vegetables incorporated from roots ($Bq\cdot kg^{-1}$),

$A_{m,\text{I-fod-lv}}$ = mass activity density in fodder deposited on leaves ($Bq\cdot kg^{-1}$),

$A_{m,\text{I-fod-r}}$ = mass activity density in fodder incorporated from roots ($Bq\cdot kg^{-1}$),

χ_{W} = water dispersion factor at the point where water for irrigation is taken from the river ($s\cdot m^{-3}$),

W_{I} = total annual amount of irrigation water in m^3 used per m^2 of the irrigated field (m),

f_{d} = 0.3, fraction of deposition on leaves [1],

$B_{a,\text{veg}}$ = 2.4 $kg\cdot m^{-2}$, typical biomass density of vegetables [1],

$P_{a,\text{veg}}$ = 280 $kg\cdot m^{-2}$, typical area density of the soil within the reach of roots for vegetables [1],

$B_{a,\text{fod}}$ = 0.85 $kg\cdot m^{-2}$, typical biomass density of fodder [1],

$P_{a,\text{fod}}$ = 120 $kg\cdot m^{-2}$, typical area density of the soil within the reach of roots for fodder [1],

$\lambda_{e,lv}$ = effective decay constant on leaves (a^{-1}, see Section 2.6.3),

$\lambda_{e,r}$ = effective decay constant in the root zone (soil), (a^{-1}, see Section 2.6.3),

t_{veg} = 0.165 a, typical production time of vegetables (60 days) [2],

t_{fod} = 8.3×10^{-2} a, typical production time of fodder (30 days) [2],

T_I = irrigation period (a), (fraction of the year during which fields are irrigated),

T_{op} = 50 a, operation period of the facility,

$C_{r,soil\text{-}veg}$ = concentration ratio for the uptake from soil in vegetables (dimensionless element-specific),

$C_{r,soil\text{-}fod}$ = concentration ratio for the uptake from soil in fodder (dimensionless element-specific).

For a very small effective decay constant in the root zone (e.g. $\lambda_{e,r} T_{op} < 10^{-4}$), the root zone build-up term reduces to

$$\lim_{\lambda_{e,r} \to 0} \frac{1 - \exp(-\lambda_{e,r} T_{op})}{T_1 \lambda_{e,r}} = \frac{T_{op}}{T_1} \quad . \tag{3.23}$$

The effective decay constant $\lambda_{e,lv}$ has always a value allowing a numerical calculation of the respective build-up factor from Eqs. 3.19 and 3.21.

The mass activity density in vegetables produced on irrigated fields, $A_{m,I\text{-}veg}$ ($Bq \cdot kg^{-1}$), is a sum

$$A_{m,I\text{-}veg} = A_{m,I\text{-}veg\text{-}lv} + A_{m,I\text{-}veg\text{-}r} \quad . \tag{3.24}$$

The mass activity density in fodder produced on irrigated fields, $A_{m,I\text{-}fod}$ ($Bq \cdot kg^{-1}$), is a sum

$$A_{m,I\text{-}fod} = A_{m,I\text{-}fod\text{-}lv} + A_{m,I\text{-}fod\text{-}r} \quad . \tag{3.25}$$

For *tritium* the equilibrium model is assumed. Tritium in plant water is in equilibrium with tritium in water mixed during the plant production period from natural tritium-free precipitation and the irrigation water. One has to consider also situations when the irrigation period is shorter than the plant production period. In this case the contribution of tritium-free water from the precipitation outside the irrigation period further reduces the average tritium mass activity density in plants.

The tritium mass activity densities in vegetables and fodder produced on irrigated fields, $A_{m,I\text{-}veg}$ and $A_{m,I\text{-}fod}$ ($Bq \cdot kg^{-1}$), respectively, are given as

$$A_{m,\text{I-veg}} = Q\frac{\chi_{\text{W}}}{\alpha T_1}\cdot\frac{f_{\text{wa}}}{\rho_{\text{wa}}}\cdot\frac{1}{1+\frac{W_{\text{p}}}{W_{\text{I}}}\frac{t_{m-\text{veg}}}{T_1}}\ , \tag{3.26}$$

$$A_{m,\text{I-fod}} = Q\frac{\chi_{\text{W}}}{\alpha T_1}\cdot\frac{f_{\text{wa}}}{\rho_{\text{wa}}}\cdot\frac{1}{1+\frac{W_{\text{p}}}{W_{\text{I}}}\frac{t_{m-\text{fod}}}{T_1}}\ , \tag{3.27}$$

where

f_{wa} = 0.75, average water content of foodstuff [1],

ρ_{wa} = 1000 kg·m^{-3}, water density,

W_{p} = annual precipitation in m^3 per m^2 (m),

W_{I} = total annual amount of irrigation water in m^3 used per m^2 of the irrigated field (m),

$t_{m,\text{veg}}$ = effective mixing time for vegetables in years (a),

$t_{m,\text{fod}}$ = effective mixing time for fodder in years (a).

The effective mixing times are given as

$$t_{m,\text{veg}} = T_{\text{I}} \quad \text{for } T_{\text{I}} > t_{\text{veg}}\ , \tag{3.28a}$$

$$t_{m,\text{veg}} = t_{\text{veg}} \quad \text{for } T_{\text{I}} \le t_{\text{veg}}\ , \tag{3.28b}$$

and

$$t_{m,\text{fod}} = T_{\text{I}} \quad \text{for } T_{\text{I}} > t_{\text{fod}}\ , \tag{3.29a}$$

$$t_{m,\text{fod}} = t_{\text{fod}} \quad \text{for } T_{\text{I}} \le t_{\text{fod}}\ , \tag{3.29b}$$

where

T_{I} = irrigation period (a), (fraction of the year during which fields are irrigated),

t_{veg} = 0.165 a, typical production time of vegetables (60 days) [2],

t_{fod} = 8.3 × 10^{-2} a, typical production time of fodder (30 days) [2].

For *gases*,

$$A_{m,\text{I-veg-lv}} = A_{m,\text{I-veg-r}} = A_{m,\text{I-fod-lv}} = A_{m,\text{I-fod-r}} = 0 \tag{3.30}$$

by definition. There is no specific model for ^{14}C.

3.2.7. Meat and milk from animals fed with fodder from irrigated fields

The mass activity densities in meat and milk of animals fed with fodder produced on the irrigated field, $A_{m,\text{I-mt}}$ and $A_{m,\text{I-mi}}$, respectively (Bq·kg^{-1}) are given as

$$A_{m,\text{I-mt}} = A_{m,\text{I-fod}} V_{\text{fod}} C_{ft,\text{fod-mt}} \ , \tag{3.31}$$

$$A_{m,\text{I-mi}} = A_{m,\text{I-fod}} V_{\text{fod}} C_{ft,\text{fod-mi}} \ , \tag{3.32}$$

where

V_{fod} = 65 kg·d^{-1}, daily consumption of fodder by cattle [1],

$C_{ft,\text{fod-mt}}$ = feed transfer coefficient from fodder to meat (d·kg^{-1}),

$C_{ft,\text{fod-mi}}$ = feed transfer coefficient from fodder to milk (d·kg^{-1}).

For *tritium*, the mass activity densities in meat and milk of animals fed with fodder produced on the irrigated field are calculated using the equilibrium model:

$$A_{m,\text{I-mt}} = A_{m,\text{I-mi}} = f_{\text{fod}} A_{m,\text{I-fod}} \ , \tag{3.33}$$

where $f_{\text{fod}} = 0.4$ [1] is the fraction of water in animals from food and the other symbols have meanings explained above. This model is conservative because animals are fed a considerable fraction of the year with dried fodder, from which tritiated water has been removed. Similarly, processing of raw meat and milk may lead to loss or replacement of the original water in the food.

There is no specific model for ^{14}C.

3.3. Effective doses

The total annual effective dose due to radioactive releases into a river consists of the following contributions each one of which has a different occupancy or modulation factor:

1. External dose due to immersion in the river water;
2. External dose due to residing on the riverbank;
3. Internal dose due to drinking the river water (age-specific);
4. Internal dose due to ingestion of fish from the river (age-specific);
5. Internal dose due to ingestion of food (meat and milk) produced from animals watered with the river water (age-specific);
6. External dose from radioactivity deposited on a field (garden) irrigated with the river water. In case of tritium internal dose from inhalation of tritiated water vapour (age-specific). In case of gases external dose from dissolved gaseous radionuclides;
7. Internal dose due to ingestion of vegetables produced on the irrigated field (garden), (age-specific);

8. Internal dose due to ingestion of food (meat and milk) produced from animals fed with fodder grown on the irrigated field (age-specific).

3.3.1. Immersion in the river water

The annual external dose due to immersion in the river water, $D_{\mathrm{W,imm}}$ (Sv·a^{-1}), consists of a contribution from the water and a contribution from the bottom sediment:

$$D_{\mathrm{W,imm}} = f_{\mathrm{W,oc}}\left(A_{v,\mathrm{W}} e_{\mathrm{W,imm}} + A_{a,\mathrm{BS}} e_{\mathrm{gnd}}\right), \tag{3.34}$$

where

$f_{\mathrm{W,oc}}$ = occupancy factor for time spent immersed in the river water,

$A_{v,\mathrm{W}}$ = average volumetric activity density in the river water (Bq·m^{-3}),

$e_{\mathrm{W,imm}}$ = dose conversion factor for immersion in water [(Sv·a^{-1})·(Bq·m^{-3})$^{-1}$],

$A_{a,\mathrm{BS}}$ = average annual area activity density in the bottom sediment (Bq·m^{-2}),

e_{gnd} = dose conversion factor for external exposure from the ground deposition [(Sv·a^{-1})·(Bq·m^{-2})$^{-1}$].

The model is conservative, as it assumes a complete immersion in an infinite water body above an infinite plane of contaminated sediment. Moreover, it does not account for the attenuation of the radiation from the sediment passing through the water.

3.3.2. Residing on the riverbank

The annual external dose due to residing on the riverbank, $D_{\mathrm{W,RB}}$ (Sv·a^{-1}), consists of a contribution from the bottom sediment and a contribution from the water:

$$D_{\mathrm{W,RB}} = f_{\mathrm{RB,oc}}\left(f_{\mathrm{B}} A_{a,\mathrm{BS}} e_{\mathrm{gnd}} + \frac{1}{4} A_{v,\mathrm{W}} e_{\mathrm{W,imm}}\right), \tag{3.35}$$

where $f_{\mathrm{RB,oc}}$ is an occupancy factor for residing on the riverbank and f_{B} is the bank geometry factor. For extended areas, like lakes and wide rivers (e.g. the Rhone River), $f_{\mathrm{B}} = 0.5$. For small streams and rivers of a width below few metres, a factor $f_{\mathrm{B}} = 0.2$ shall be still conservative. The other symbols have meanings as above. Compared with the immersion in the river water, only one quarter of an infinite water body and only f_{B}-part of an infinite plane of contaminated sediment are assumed, being still conservative. For leisure activities outdoors (fishing, walking, etc.), the occupancy factor $f_{\mathrm{RB,oc}}$ should be the fraction of the time spent on the riverbank. However, if people live in a house located on the riverbank, a long-term indoor shielding factor $k_{\mathrm{s}} = 0.4$ [1]

should be included in $f_{\mathrm{RB,oc}}$. This means that for a house on the riverbank, in which people reside 100% of their time, $f_{\mathrm{RB,oc}} = 0.4$.

3.3.3. Drinking the river water

The age-specific committed effective dose due to drinking of the river water during one year, $D_{\mathrm{W,dw}}(a,i)$ (Sv·a^{-1}), to adults (a) and one-year old infants (i), is given as [1]

$$D_{\mathrm{W,dw}}(a,i) = f_{\mathrm{W}} A_{v,\mathrm{W}} U_{\mathrm{W}}(a,i) e_{\mathrm{ing}}(a,i) \ , \tag{3.36}$$

where

f_{W} = fraction of drinking water from the river,

$A_{v,\mathrm{W}}$ = average volumetric activity density in the river water (Bq·m^{-3}),

$U_{\mathrm{W}}(a)$ = 0.6 m^3·a^{-1}, average annual water consumption for adults [1],

$U_{\mathrm{W}}(i)$ = 0.25 m^3·a^{-1}, average annual water consumption for one-year old infants [1],

$e_{\mathrm{ing}}(a,i)$ = ingestion dose conversion factor for adults (a) or infants (i) (Sv·Bq^{-1}).

3.3.4. Ingestion of fish from the river

The age-specific committed effective dose due to the ingestion of fish from the river during one year, $D_{\mathrm{W,fi}}(a,i)$ (Sv·a^{-1}), to adults (a) and one-year old infants (i), is given as

$$D_{\mathrm{W,fi}}(a,i) = f_{\mathrm{fi}} A_{m,\mathrm{fi}} U_{\mathrm{fi}}(a,i) \exp(-\lambda T_{\mathrm{fi}}) e_{\mathrm{ing}}(a,i) \ , \tag{3.37}$$

where

f_{fi} = fraction of consumed fish from the river,

$A_{m,\mathrm{fi}}$ = mass activity density in fish $A_{m,\mathrm{fi}}$ (Bq·kg^{-1}),

$U_{\mathrm{fi}}(a)$ = 10 kg·a^{-1}, average annual fish consumption for adults [1],

$U_{\mathrm{fi}}(i)$ = 0 kg·a^{-1}, average annual fish consumption for one-year old infants [1],

λ = radioactive decay constant of the radionuclide in question (a^{-1}),

T_{fi} = 2.7 × 10^{-3} a, typical storage time of fish (1 day) [1],

$e_{\mathrm{ing}}(a,i)$ = ingestion dose conversion factor for adults (a) or infants (i) (Sv·Bq^{-1}).

3.3.5. Ingestion of food produced from animals watered with the river water

The age-specific committed effective dose due to the ingestion of food produced from animals watered with the river during one year, $D_{\mathrm{W,afd}}(a,i)$ (Sv·a^{-1}), to adults (a) and one-year old infants (i), is given as

$$D_{\mathrm{W,afd}}(a,i) = \left[f_{\mathrm{W\text{-}mt}} A_{m,\mathrm{W-mt}} U_{\mathrm{mt}}(a,i) \exp(-\lambda T_{\mathrm{mt}}) + f_{\mathrm{W\text{-}mi}} A_{m,\mathrm{W-mi}} U_{\mathrm{mi}}(a,i) \exp(-\lambda T_{\mathrm{mi}}) \right] e_{\mathrm{ing}}(a,i), \tag{3.38}$$

where

$f_{\mathrm{W\text{-}mt}}$ = fraction of meat from animals watered with the river water,

$f_{\mathrm{W\text{-}mi}}$ = fraction of milk from animals watered with the river water,

$A_{m,\mathrm{W\text{-}mt}}$ = mass activity density in meat of animals watered with the river water (Bq·kg^{-1}),

$A_{m,\mathrm{W\text{-}mi}}$ = mass activity density in milk of animals watered with the river water (Bq·kg^{-1}),

$U_{\mathrm{mt}}(a,i)$ = annual meat consumption for adults (a) or infants (i) (kg·a^{-1}, see Table 2.8),

$U_{\mathrm{mi}}(a,i)$ = annual milk consumption for adults (a) or infants (i) (kg·a^{-1}, see Table 2.8),

λ = radioactive decay constant of the radionuclide in question (a^{-1}),

T_{mt} = 5.5×10^{-2} a, typical storage time of meat (20 days) [1],

T_{mi} = 2.7×10^{-3} a, typical storage time of milk (1 day) [1],

$e_{\mathrm{ing}}(a,i)$ = ingestion dose conversion factor for adults (a) or infants (i) (Sv·Bq^{-1}).

3.3.6. External dose from radioactivity deposited on irrigated fields

The annual effective dose from radioactivity deposited on the field (garden) due to irrigation, $D_{\mathrm{I,gnd}}$ (Sv·a^{-1}), is given as

$$D_{\mathrm{I,gnd}} = Q f_{\mathrm{I,oc}} \xi_{\mathrm{I}} \times \left\{ \left[0.63 \frac{1-\exp\left[-(\lambda+\lambda_{\mathrm{F}})T_{\mathrm{op}}\right]}{\lambda+\lambda_{\mathrm{F}}} + 0.37 \frac{1-\exp\left[-(\lambda+\lambda_{\mathrm{S}})T_{\mathrm{op}}\right]}{\lambda+\lambda_{\mathrm{S}}} \right] \cdot \frac{1-e^{-\lambda T_{\mathrm{exp}}}}{\lambda} + \frac{1}{\lambda}\left(T_{\mathrm{exp}} - \frac{1-e^{-\lambda T_{\mathrm{exp}}}}{\lambda} \right) \right\} e_{\mathrm{gnd}}, \tag{3.39}$$

where

Q = annual radioactivity release (Bq),

$f_{\mathrm{I,oc}}$ = occupancy factor for residing in the irrigated area,

λ = radioactive decay constant (a^{-1}),

λ_{F} = 1.1 a^{-1}, non-radioactive decay constant for the *fast* component [1],

λ_S = 7.5×10^{-3} a^{-1}, non-radioactive decay constant for the *slow* component [1],

T_{op} = 50 a, operation period of the facility [1],

T_{exp} = 1 a, exposure time,

e_{gnd} = dose conversion factor for external exposure from deposition of radioactivity on the ground [$(Sv \cdot a^{-1}) \cdot (Bq \cdot m^{-2})^{-1}$].

The model assumes build-up of the deposited activity during the facility operation period, T_{op}, and the decay due to radioactivity as well as due to non-radioactive processes characterized by the fast and slow decay constants. If a house is built and inhabited in the irrigated area, such as a garden, an additional indoor shielding factor $k_s = 0.4$ shall be assumed within the occupancy factor $f_{I,oc}$.

For very long-lived radionuclides (e.g $\lambda T_{exp} < 10^{-4}$) the terms with λ in denominators reduce to

$$\lim_{\lambda \to 0} \frac{1 - e^{-\lambda T_{exp}}}{\lambda} = T_{exp} \tag{3.40}$$

and

$$\lim_{\lambda \to 0} \frac{1}{\lambda} \left(T_{exp} - \frac{1 - e^{-\lambda T_{exp}}}{\lambda} \right) = \frac{T_{exp}^2}{2} \ . \tag{3.41}$$

For *tritium* the effective dose due to ground deposition is substituted with the age-specific inhalation effective dose committed due to irrigation (and inhalation) during one annual irrigation period to adults (*a*) and infants (*i*), $D_{I,gnd}(a,i)$ ($Sv \cdot a^{-1}$):

$$D_{I,gnd}(a,i) = f_{I,ocT} A_{v,I\text{-}HTO} \frac{T_I}{T_1} \alpha U_{inh}(a,i) e_{inh}(a,i) \ , \tag{3.42}$$

where

$f_{I,ocT}$ = occupancy factor specific to tritium for residing in the irrigated area,

T_I = irrigation period (a), (fraction of the year during which fields are irrigated),

T_1 = 1 a, time of one year,

$A_{v,I\text{-}HTO}$ = average volumetric tritium activity in the air above the field during the irrigation period ($Bq \cdot m^{-3}$),

α = 3.16×10^7 $s \cdot a^{-1}$, conversion factor from years to seconds,

$U_{inh}(a)$ = 2.3×10^{-4} $m^3 \cdot s^{-1}$, average inhalation rate for adults [1],

$U_{inh}(i)$ = 6.0×10^{-5} $m^3 \cdot s^{-1}$, average inhalation rate for infants [1],

$e_{\mathrm{inh}}(a,i)$ = inhalation dose conversion factor for adults (a) or infants (i) (Sv·Bq^{-1}).

Note that the exposure takes place only during the irrigation period T_{I}. No indoor shielding factor should be considered because water vapour may penetrate indoors and hence the HTO volumetric activity density in the indoor air equals to that outdoors.

For *gases* the external exposure from a semi-infinite (factor ½) water body diluted with precipitation water is assumed during the irrigation period. This conservatively estimated dose is believed to cover also the inhalation dose due to gases liberated from the water to the ambient air.

$$D_{\text{I-gnd}} = \frac{1}{2} f_{\mathrm{I,oc}} \frac{T_{\mathrm{I}}}{T_1} A_{v,\mathrm{W}} \frac{1}{1+\frac{W_{\mathrm{p}}}{W_{\mathrm{I}}}\frac{T_{\mathrm{I}}}{T_1}} e_{\mathrm{W,imm}} \ , \tag{3.43}$$

where

$f_{\mathrm{I,oc}}$ = occupancy factor for residing in the irrigated area,

T_{I} = irrigation period (a), (fraction of the year during which fields are irrigated),

T_1 = 1 a, time of one year,

$A_{v,\mathrm{W}}$ = average volumetric activity density in the river water (Bq·m^{-3}),

W_{p} = annual precipitation in m^3 per m^2 (m),

W_{I} = total annual amount of irrigation water in m^3 used per m^2 of the irrigated field (m),

$e_{\mathrm{W,imm}}$ = dose conversion factor for immersion in water [(Sv·a^{-1})·(Bq·m^{-3})$^{-1}$].

3.3.7. Ingestion of vegetables from irrigated fields

The age-specific committed effective dose due to ingestion during one year of vegetables produced on a field or in a garden irrigated with the river water to adults (a) or infants (i), $D_{\mathrm{I,veg}}(a,i)$ (Sv·a^{-1}), is given as

$$D_{\text{I-veg}}(a,i) = f_{\text{I-veg}} A_{m,\text{I-veg}} U_{\mathrm{veg}}(a,i) e_{\mathrm{ing}}(a,i) \ , \tag{3.44}$$

where

$f_{\text{I-veg}}$ = fraction of consumed vegetables produced on the irrigated field (in the irrigated garden),

$A_{m,\text{I-veg}}$ = mass activity density in vegetables produced on the irrigated field (in the irrigated garden) (Bq·kg^{-1}),

$U_{\mathrm{veg}}(a,i)$ = average annual consumption of vegetables by adults (a) or infants (i) (kg·a^{-1}, see Table 2.8),

$e_{\mathrm{ing}}(a,i)$ = ingestion dose conversion factor for adults (a) or infants (i) (Sv·Bq^{-1}).

3.3.8. Ingestion of meat and milk from animals fed with fodder from irrigated fields

The committed effective dose due to the ingestion during one year of food produced from animals fed with fodder, which was grown on fields irrigated with the river water, $D_{\mathrm{I,afd}}(a,i)$ (Sv·a^{-1}), to adults (a) and one-year old infants (i), is given as

$$D_{\mathrm{I,afd}}(a,i) = \left[f_{\mathrm{I\text{-}mt}} A_{m,\mathrm{I-mt}} U_{\mathrm{mt}}(a,i) \exp(-\lambda T_{\mathrm{mt}}) + f_{\mathrm{I\text{-}mi}} A_{m,\mathrm{I-mi}} U_{\mathrm{mi}}(a,i) \exp(-\lambda T_{\mathrm{mi}}) \right] e_{\mathrm{ing}}(a,i) \ , \qquad (3.45)$$

where

$f_{\mathrm{I\text{-}mt}}$ = fraction of meat from animals fed with fodder produced on the irrigated field,

$f_{\mathrm{I\text{-}mi}}$ = fraction of milk from animals fed with fodder produced on the irrigated field,

$A_{m,\mathrm{I\text{-}mt}}$ = mass activity density in that meat (Bq·kg^{-1}),

$A_{m,\mathrm{I\text{-}mi}}$ = mass activity density in that milk (Bq·kg^{-1}),

$U_{\mathrm{mt}}(a,i)$ = average annual meat consumption for adults (a) or infants (i) (kg·a^{-1}, see Table 2.8),

$U_{\mathrm{mi}}(a,i)$ = average annual milk consumption for adults (a) or infants (i) (kg·a^{-1}, see Table 2.8),

λ = radioactive decay constant of the radionuclide in question (a^{-1}),

T_{mt} = 5.5×10^{-2} a, typical storage time of meat (20 days) [1],

T_{mi} = 2.7×10^{-3} a, typical storage time of milk (1 day) [1],

$e_{\mathrm{ing}}(a,i)$ = ingestion dose conversion factor for (a) adults or infants (i) (Sv·Bq^{-1}).

3.4. Parameters of a standard river

If no reliable data on the river flow are available, a generic model shall be used, assuming that the river has standard properties. The basic parameter to be input to the model is the average river width, $\overline{B}$ (m), which can be acquired from observations on the site or from a map.

The average river flow, $\overline{J}$ (m^3·s^{-1}), which corresponds to the average river width, $\overline{B}$ (m), is given as a power function (Figure 3.2), which was obtained by fitting the data from the IAEA Safety Report No. 19 [2]:

$$\overline{J} = a_0 \overline{B}^{a_1} \ , \qquad (3.46)$$

where

a_0 = 6.67×10^{-3},

a_1 = 2.174.

Similarly, a power function for the average river depth, $\overline{D}$ (m), was derived by fitting the data from the Ref. [2] (Figure 3.3):

$$\overline{D} = b_0 \overline{B}^{b_1} \ , \tag{3.47}$$

where

b_0 = 1.74×10^{-2},

b_1 = 0.9714.

The average river flow velocity, $\overline{v}$ ($m \cdot s^{-1}$), is then given as

$$\overline{v} = \frac{\overline{J}}{\overline{B}\overline{D}} \ . \tag{3.48}$$

For screening purposes, a minimum river flow over several decades, J_{min} ($m^3 \cdot s^{-1}$), shall be considered. It is assumed that the minimum river flow is 1/3 of the average river flow [2]:

$$J_{min} = \frac{1}{3}\overline{J} \ . \tag{3.49}$$

Solving Eq. 3.46 gives a power function for the width of the river during the minimum flow conditions B_{min} (m):

$$B_{min} = c_0 \overline{J}^{c_1} \ , \tag{3.50}$$

where

c_0 = 6.03,

c_1 = 0.46,

and the other parameters for the minimum river flow conditions, the river depth, D_{min} (m), and the river flow velocity, v_{min} ($m \cdot s^{-1}$), shall be calculated using Eqs. 3.47 and 3.48, respectively.

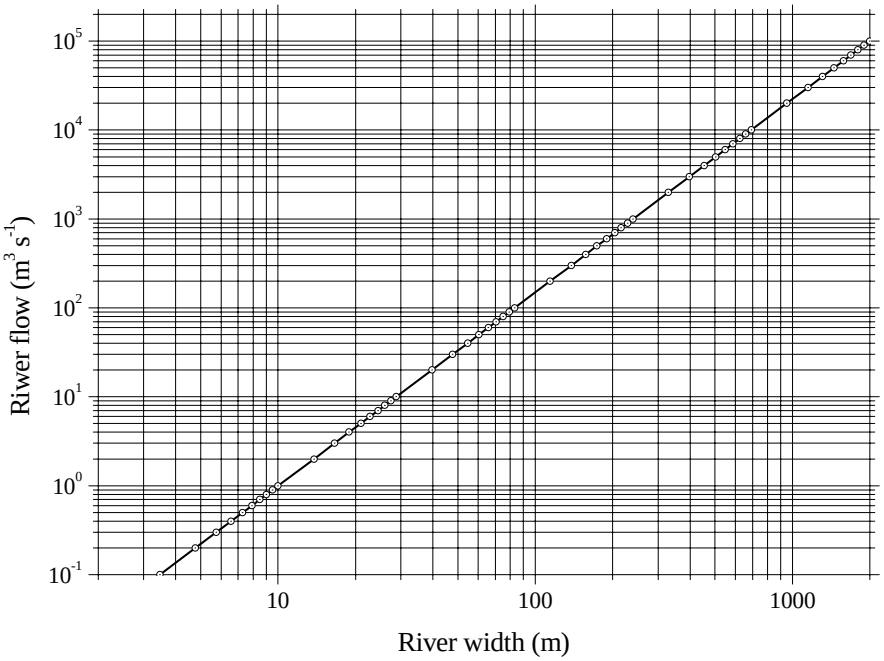

Figure 3.2: Dependence of the river flow on the river width for a standard river [2].

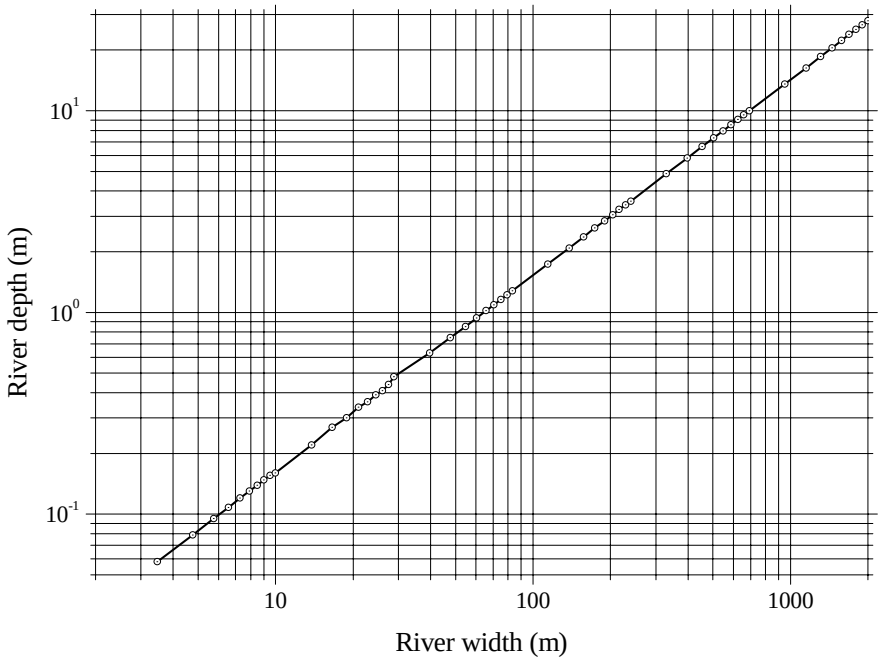

Figure 3.3: Dependence of the river depth on the river width for a standard river [2].

4. Water releases into a water treatment plant

4.1 Scenarios

Radioactive wastewater may also be released into a sewer. In this case the water is treated in a water treatment plant and plant workers may be exposed. It is assumed that the discharge has taken place uniformly during the last 50 years, including the current year. For screening purposes two extreme cases shall be investigated [2]:

1. Radionuclides from the wastewater are not retained in the plant sludge but are released with the cleaned water into the receiving river. In this case the receiving river shall be identified and models from Section 3 used;
2. Radionuclides are completely retained in the sludge, which contributes to the exposure of the plant workers. It depends upon the final destiny of the sludge what further public exposures may take place. For example, the sludge may be incinerated, and in that case a long-term atmospheric release from the stack of the incineration plant shall be investigated (Section 2). Or the sludge can be used for field fertilization. Then models for ground deposition of radioactivity on fields shall be considered.

These two extreme cases cover the uncertainty of the radionuclide retention in the sludge. Default values of the distribution coefficient K_d may not be suitable because various water-purification processes may change the affinity of the given radionuclide to particles substantially. This, however, does not apply to highly soluble radionuclides, which are easily transported with water, such as HTO or in particular Na radioisotopes (^{22}Na, and ^{24}Na at CERN). For such substances, which have $K_d = 0$ by definition, the only mechanism of enrichment in the sludge is drying of the wet sludge with the subsequent increase of the mass activity densities. Of course, HTO is removed with the evaporated water completely.

4.2. Materials processed at the water treatment plant

At the water treatment plant the radioactivity incoming with wastewater from a nuclear facility is diluted with water from other sources. If the volume of radioactive water discharges received from the facility is much smaller than that received from other sources, which is the usual case, the annual average volumetric activity density of a given radionuclide in the water treated at the plant, $A_{v,\mathrm{T\text{-}W}}$ (Bq·m^{-3}), will be

$$A_{v,\mathrm{T\text{-}W}} = \frac{Q}{W_\mathrm{T}} \exp(-\lambda t_\mathrm{T}) \ , \qquad (4.1)$$

where

Q = annual radioactivity release (Bq·a^{-1}),

W_T = annual amount of water treated at the plant (m^3·a^{-1}),

λ = radioactive decay constant (s^{-1}),

t_T = sewage transport time from the facility outlet to the water treatment plant (s).

The decay term due to the sewage transport time, t_T, is included in order to cut off very short-lived radionuclides, typically spallation gases. By default the sewage transport time can be estimated from the length of the pipeline (distance from the facility to the water treatment plant) assuming a sewage water velocity in the pipeline of 2 m·s^{-1}.

Except for tritium, highly soluble elements and gases, the annual average mass activity density of the given radionuclide in the wet sludge, $A_{m,\text{T-S-wet}}$ (Bq kg^{-1}), is given as

$$A_{m,\text{T-S-wet}} = \delta \frac{Q}{S_T} \exp(-\lambda t_T) \ , \tag{4.2}$$

where S_T is the annual sludge production at the plant in dry weight (kg·a^{-1} dry), and the factor δ is the sludge drying factor, which is given as the ratio between the masses of the wet and the dry sludge. The default value of δ is 0.05 [2]. The radioactive decay during the sludge treatment time is conservatively neglected.

For *tritium, highly soluble elements* and *gases* the mass activity density in the wet sludge, $A_{m,\text{T-S-wet}}$ (Bq kg^{-1} wet), is derived directly from the volumetric activity density of the water treated at the plant $A_{v,\text{T-W}}$, assuming equal densities of the pure water and the wet sludge:

$$A_{m,\text{T-S-wet}} = (1-\delta)\frac{A_{v,\text{T-W}}}{\rho_{\text{wa}}} \ , \tag{4.3}$$

where $\rho_{\text{wa}} = 1000\,\text{kg}\cdot\text{m}^{-3}$ is the water density and the factor $(1-\delta)$ accounts for the mass fraction of water in the wet sludge.

Although the dosimetric models described in the following text use mass activity densities in the wet sludge, mass activity densities in the dry sludge, $A_{m,\text{T-S-dry}}$ (Bq·kg^{-1} dry), may be of interest for assessing the radiological hazards related to the dry sludge, which may be disposed off or reused in other way.

$$A_{m,\text{T-S-dry}} = \frac{Q}{S_T} \exp(-\lambda t_T) = \frac{A_{m,\text{T-S-wet}}}{\delta} \ . \tag{4.4}$$

For *tritium* and *gases*, the mass activity density of the dry sludge $A_{m,\text{T-S-dry}}$ is zero by default.

If possible, the site specific values of S_T and W_T shall be used. If these are unknown, at least the number of persons served by the plant shall be identified. For developed European countries like Switzerland and France, a default annual sludge production per served person of $20\,\text{kg}\cdot\text{a}^{-1}$/person dry weight provides a suitable estimate for screening purposes. The recommended default annual amount of water treated at a plant is 200 m^3·a^{-1}/person in the Geneva region and the neighbouring France. These estimates include not only water from households but also water from industry [2].

4.3. Ambient air at the water treatment plant

Except for tritium and gases, the volumetric activity density of the resuspended sludge in the ambient air, $A_{v,\text{T-S-res}}$ (Bq·m^{-3}), is given as

$$A_{v,\text{T-S-res}} = K_{\text{sludge}} A_{m,\text{T-S-wet}} \ , \tag{4.5}$$

where

K_{sludge} = 1×10^{-7} kg·m^{-3}, sludge resuspension factor [2],

$A_{m,\text{T-S-wet}}$ = mass activity density of the given radionuclide in the wet sludge (Bq kg^{-1} wet).

For *tritium*, the equilibrium between the air humidity and the water treated at the plant is assumed, giving the volumetric activity density of HTO vapour in the air at the plant $A_{v,\text{T-HTO}}$ (Bq·m^{-3}):

$$A_{v,\text{T-HTO}} = \frac{\Phi}{\rho_{\text{wa}}} A_{v,\text{T-W}} \ , \tag{4.6}$$

where

Φ = 9×10^{-3} kg·m^{-3}, average air humidity [1],

ρ_{wa} = 1000 kg·m^{-3}, water density,

$A_{v,\text{T-W}}$ = volumetric activity density of tritium (HTO vapour) in the water treated at the plant, (Bq·m^{-3}).

It is difficult to estimate the volumetric activity density of *gases* liberated from the wet sludge to the ambient air. It is known that for inert gases except radon with its progenies the inhalation dose is negligible compared with the external dose so the total dose can be estimated as the external dose. The case of radon with its progenies shall be investigated separately. As no inhalation dose will be further considered for gases, a zero volumetric activity density in the ambient air is assumed for radionuclides in this state by definition.

4.4. Effective doses

Only two exposure pathways are assumed for adult workers of the water treatment plant:

1. External exposure from the sludge;
2. Inhalation dose due to resuspension of the sludge or due to evaporation of tritiated water vapour.

4.4.1. External exposure from the sludge

The annual effective dose due to the external exposure from the sludge, $D_{T,ext}$ (Sv·a^{-1}), can be estimated in the same way as the dose from the ground deposition [2]:

$$D_{T,ext} = f_{T,oc} A_{m,T\text{-}S\text{-}wet} \rho_S d_S e_{gnd} \ , \qquad (4.7)$$

where

$f_{T,oc}$ = 0.21, occupancy factor for the workers of the water treatment plant (40 hours per week of working time),

$A_{m,T\text{-}S\text{-}wet}$ = mass activity density of the given radionuclide in the wet sludge (Bq·kg^{-1} wet),

ρ_S = density of the wet sludge (kg·m^{-3}) = 1000 kg·m^{-3} by default [2],

d_S = depth of the sludge basin (m) = 1 m by default [2],

e_{gnd} = dose conversion factor for external exposure from deposition of radioactivity on the ground [(Sv·a^{-1})·(Bq·m^{-2})$^{-1}$].

For inert *gases* the tabulated dose conversion factors e_{gnd} are usually zero because no ground deposition is considered for radionuclides in this phase. The external dose can be estimated by using the water immersion dose conversion factor, assuming that the plant workers are exposed to a semi-infinite sludge volume (factor ½). Such approach is sufficiently conservative so that the inhalation dose due to gases liberated from the treated sludge can be neglected.

$$D_{T,ext} = \frac{1}{2} f_{T,oc} A_{v,T\text{-}W} e_{W,imm} \ , \qquad (4.8)$$

where

$f_{T,oc}$ = 0.21, occupancy factor for the workers of the water treatment plant (40 hours per week of working time),

$A_{v,T\text{-}W}$ = volumetric activity density of the given gaseous radionuclide in the water treated at the plant (Bq·m^{-3}),

$e_{W,imm}$ = dose conversion factor for immersion in water [(Sv·a^{-1})·(Bq·m^{-3})$^{-1}$].

4.4.2. Inhalation dose

The effective dose committed to the adult plant workers due to inhalation of resuspended sludge or evaporated HTO during one year, $D_{\mathrm{T,inh}}(a)$ (Sv·a^{-1}), is given as [2]

$$D_{\mathrm{T,inh}}(a) = f_{\mathrm{T,oc}} A_{v,\mathrm{T\text{-}S}} \alpha U_{\mathrm{inh}}(a) e_{\mathrm{inh}}(a) \ , \qquad (4.9)$$

where

$f_{\mathrm{T,oc}}$ = 0.21, occupancy factor for the workers of the water treatment plant (40 hours per week of working time),

$A_{v,\mathrm{T\text{-}S}}$ = volumetric activity density of the given radionuclide in the air at the plant (Bq·m^{-3}),

α = 3.16×10^{7} s·a^{-1}, conversion factor from years to seconds,

$U_{\mathrm{inh}}(a)$ = 2.3×10^{-4} m^{3}·s^{-1}, average inhalation rate for adults [1],

$e_{\mathrm{inh}}(a)$ = inhalation dose conversion factor for adults (Sv·Bq^{-1}).

The volumetric activity density $A_{v,\mathrm{T\text{-}S}}$ is either the volumetric activity density of the resuspended sludge in the air $A_{v,\mathrm{T\text{-}S\text{-}res}}$ or the volumetric activity density of tritiated water vapour $A_{v,\mathrm{T\text{-}HTO}}$ in case of tritium. Zero volumetric activity density is assumed for gases.

5. Radionuclide data

The models described above require radionuclide-specific and element-specific data, which can be found in various literature sources. For each radionuclide (element) the following data are needed:

1. Half-life;
2. Energies and yields of gamma lines;
3. Dose conversion factors;
4. Element-specific biological transfer factors;
5. Element-specific distribution coefficients.

5.1. Half-lives and photon emission data

There are numerous sources of radionuclide half-lives, energies and yields of gamma lines. It is recommended to use the IAEA nuclide database NUDAT with a Web interface on the BNL Web site [22].

5.2. Dose conversion factors

Dose conversion factors can be found in several publications. The effective dose conversion factors for immersion in air, e_{imm}, and ground deposition, e_{gnd}, are given for selected radionuclides in the directive HSK-R-41 [1]. More complete tables for a greater number of radionuclides are contained in the EPA Report 402-R-93-081 [13]. This reference includes also dose conversion factors for immersion in water, $e_{\mathrm{W,imm}}$ and dose conversion factors for organ dose to skin, e_{ibs}. The dose conversion factors e_{imm}, e_{gnd} and $e_{\mathrm{W,imm}}$ in Ref. [13] are for the effective dose equivalent, which, however, should not differ very much from the effective dose for external exposure. It shall be also noted that the dose is integrated over 1 second in Ref. [13] and to get the dose conversion factors for dose integrated over 1 year, as used in this report, one should not forget to multiply the factors with the conversion coefficient $\alpha = 3.16 \times 10^{7}$ s·a^{-1}. Another source of the external dose conversion factors is the work of Kocher [23] (effective dose equivalent).

Age-specific inhalation dose conversion factors, e_{inh}, are given in the ICRP Publication 72 [24] for a number of radionuclides and for various ages. The inhalation dose conversion factors are classified according to the type of lung absorption, which depends on the physical and chemical form of the released radionuclide. In case when the physical and chemical forms are unknown or variable, the default lung-absorption type, if indicated in the ICRP Publication 72, shall be assumed. If the default lung-absorption type is not suggested, the greatest dose conversion factor among those listed for different lung-absorption types shall be taken. The inhalation dose conversion factors for inert beta-gamma emitters (Ar, Kr, Xe) shall be set to zero. The ICRP Publication 72 gives only combined values for the external exposure in air and the internal exposure from gas in the respiratory tract. However, the external exposure dominates [24] and the contribution of the inhalation dose can be neglected when compared with the external dose. Carbon-11 is known to occur in two main forms in accelerators: ^{11}CO and ^{11}CO$_2$. Other chemical forms of ^{11}C may have higher inhalation dose conversion factors but their occurrence is so low that they can be neglected from the radiological point of view [25]. All ^{11}C shall be supposed to be in the form of ^{11}CO$_2$, for which the inhalation dose conversion factors are higher than those for ^{11}CO. Although as much as 80% of ^{11}C is in the form of ^{11}CO just after the ^{11}C formation, carbon monoxide can oxidize to carbon dioxide by ionized air during the transport in the accelerator tunnel [25]. Oxygen isotopes, ^{13}N and Ar isotopes shall be assumed in their normal chemical form: O$_2$, N$_2$, Ar. Reactive gases F

and Cl shall be supposed to attach to aerosol during their transport in tunnels and ducts. Inhalation dose conversion factors for aerosols are higher than those for the gaseous forms. Unlike gases, aerosols can also deposit on the ground and enter the food chain as well as expose the receptor from the ground.

The ICRP Publication 72 [24] contains age-specific ingestion dose conversion factors, e_{ing}, as well.

If the released radionuclide has progenies, radioactive series shall be considered either by additional releases of the progenies or by adding their dose conversion factors where applicable. The correct approach depends upon the particular case.

5.3. Biological transfer factors

Numerous element-specific biological transfer factors are given in the directive HSK-R-41 [1] and in the IAEA Technical Report No. 364 [20]. Following the Directive, missing transfer factors shall be substituted with transfer factors for a chemically similar element, generally the one, which is the closest element in the same column of the periodic table of the elements.

5.4. Distribution coefficients

The IAEA Safety Report No. 19 [2] contains default distribution coefficients K_d in freshwater for 25 elements, which are most common in water released from nuclear installations. For elements not included in Ref. [2], site-specific values shall be used or conservative values assumed.

6. Summary and conclusion

The report compiles models for environmental impact assessment of radioactive releases. They are based on widely accepted and up-to-date literature, and adapted for CERN facilities. All models are generic, which means that they require the least site-specific information as their input. At the same time they follow the screening principle and are conservative.

Except for the Monte Carlo integration of the dose kernel for external exposure from the radioactive plume and other minor elements, which were developed in the Safety Commission, the models follow the 1997 Swiss directive HSK-R-41 [1] (mainly for airborne releases) and the 2001 IAEA Safety Report No. 19 [2] (mainly for water releases). Instead of an explicative writing style, the descriptive way of presentation of the models was chosen in order to ease their practical application. In this sense, they can be considered as a recommendation of the Safety Commission. The reader shall refer to the original literature for detailed explanations.

A programme package implementing the described models was written in FORTRAN in the Safety Commission and it is available for CERN users. The Safety Commission possesses also other necessary data, such as the weather statistics for the airport Geneva Cointrin, which can be used for the two CERN main sites and the adjacent SPS and LHC sites, or hydrological information about the rivers receiving effluents from CERN.

For the sake of flexibility, pre-calculated effective doses per unit release and reference values for long-term airborne and water releases, which apply to the existing facilities, as well as a description of the software package, are published in separate reports.

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