Rapid procedure for determining the nuclide-specific contributions of a contamination in soil (in-situ gamma spectrometry)

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1 Scope

In-situ gamma spectrometry is the rapid measurement procedure of choice in the intense operation of IMIS when it comes to determining the nuclide-specific concentration on and in the ground as well as individual contributions to the local dose rate.

Compared to the classical method of determining the radionuclide content on and in the ground, which is the gamma spectrometric analysis of soil samples in a laboratory (according to procedure F-γ-SPEKT-BODEN-01), it offers a number of advantages and disadvantages.

The advantages of this procedure are that

— the comparatively time-consuming steps of collecting, preparing and transporting samples and their subsequent gamma analysis in a laboratory do not apply,

— averaging the activity concentration over a comparatively large area of ground increases the degree of representation compared to one soil sample only,

— averaging the activity concentration over a comparatively large area of ground keeps measurement periods short, and

— the results of the measurements are available immediately after the measurement.

The advantages of the results of the measurement being rapidly available counterbalance the disadvantage of a certain loss in accuracy due to possible systematic uncertainties:

— The parameters required for converting the measured data into activity per unit area of ground will always only be known or will only be quantifiable with some uncertainties attached. This is in spite of the possibility of identifying these parameters upfront and parametrising fixed metering localities and assessing them ad hoc on the basis of, in particular, the meteorological conditions prior to taking measurements (wet or dry depositions). Fixing the parameters of fixed metering points should be done whenever possible. If the required parameters are insufficiently known, the resulting uncertainty may amount to a factor of 2.

— A quantitative analysis of the data supposes that the activity of the ground within the measured area is approximately spatially constant and the area under observation (ca. 30 cm in radius around the detector) is free of higher vegetation and other obstacles.

— The initial nuclide-specific gamma flux in the ground from long-lived caesium nuclides that are a result of atmospheric nuclear tests and the accident at Chernobyl needs to be known and taken into account for the analysis. This is most easily achieved by means of a prior, but recently taken measurement of the "background spectrum" that can then be analysed with the same parameters as the present spectrum.

In in-situ gamma spectrometry, the actually measured quantity is the unscattered gamma flux at the location of the detector. The activity per unit area of ground calculated from it varies substantially with the presumed degree of absorption by the ground. In reverse, the degree of absorption by the ground substantially influences the dose rate resulting from it. The two effects compensate each other save for the contribution of scattered radiation (build-up effect).

With the initial soil contamination usually having an unknown vertical distribution, the results for the "background spectrum" analysed thus will be unsuited for quantifying in absolute figures the actually encountered activity per surface area. In order to prevent an over-interpretation of these data, the activity per area can be converted into a nuclide-specific dose rate by means of predetermined conversion factors. This requires that the processing of these data suppose *the same* vertical distribution as for the analysis of the spectrum. The nuclide-specific dose rate determined thus is robust to errors in the presumed vertical distribution.

In spite of these disadvantages, in-situ gamma spectrometry is the rapid measurement method of choice in all cases where short-term availability of measurement data is of greater importance than high accuracy.

In-situ gamma spectrometry has the following possible application areas:

— Rapid determination of the contamination of soils including vegetation following the release of gamma emitting radionuclides into the environment;
— Mapping of contaminated ground including vegetation;
— Identification of the contributions to the local gamma dose rate by individual radionuclides present on and in the ground;
— Monitoring of dumps and other areas such as parking lots, lawn-covered playgrounds etc.

The quantitative analysis of measurements taken of the ground reaches its limits when there is concurrent airborne activity and the contribution of this airborne activity to the gamma-flux at the location of the detector is unknown. Individual cases may therefore require that in-situ measurements be postponed until the airborne activity has subsided to an insignificant extent and there is no substantial deposition any more. In these cases, the data required for IMIS will be supplied by stations equipped with aerial measuring instruments (e. g., DWD weather stations). By no means may measurements be taken during periods with precipitation.

2 Sampling

not applicable

3 Analysis

not applicable

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4 Measuring the activity

For the applications of in-situ gamma spectrometry envisaged here, semiconductor detectors (high-purity germanium) with high energy-resolution capabilities are required. Small-volume detectors should be given preferences in cases of high-degree contamination of the ground.

For the measurement, the detector is set up in the terrain in such a manner that the crystal is positioned at a level of one metre above the ground (see Figure 1). In the context of the present measuring instruction it is supposed that the detector will not be shielded so that the half-space "ground" and the half-space "air" will be available as volumes to be measured by the detector.

Compared to the uncertainties arising from the conversion of the measurements into units of activity per area of the ground, the uncertainties that may result from the positioning of the detector are comparatively small in this case. A tolerance in the height of 1 m ± 10 cm should nevertheless be adhered to.

Contamination of both the detector and the tripod needs to be avoided. If the air is still contaminated during the measurement, the detector has to be sufficiently protected with a plastic hood or similar cover. Verification measurements in the non-contaminated laboratory are required to quantify the degree, or to exclude the possibility, of contamination.

In order to minimize possible uncertainties and sources of error in the conversion of measured data into activity per unit area of the soil, the measurement should take place in a sufficiently large, unaltered, plain area such as a meadow, pasture or lawn for example. The vegetation at the location of the measurement should be as low as possible, and obstacles resulting from buildings, for example, and heterogeneities, such as intersecting tarred roads etc., have to be avoided.

The "field of view" of the in-situ gamma spectrometer used determines the dimensions of the required measurement location for the practical effectuation of in-situ measurements. It depends, amongst others, on the height above the ground at which measurements are taken, the energy of the unscattered gamma quanta, and the vertical distribution of radionuclides in the ground. As a rule of thumb, a "field of view" of about 30 m in radius may be supposed in the case of a fresh contamination. More details are discussed in section 5.2.

As a result of "fields of view" of this order of magnitude, in-situ gamma-spectrometry will record the contamination of several thousands of square metres of ground. This makes it possible to obtain a statistical measuring accuracy with relatively short measurement periods that are comparable or small compared to the other uncertainties encountered in the procedure (cf. section 5.5). In the case of a contamination by a typical mix of radionuclides amounting to several 10 kilobecquerels per square metre, the respective measurement period will be less than 30 minutes.
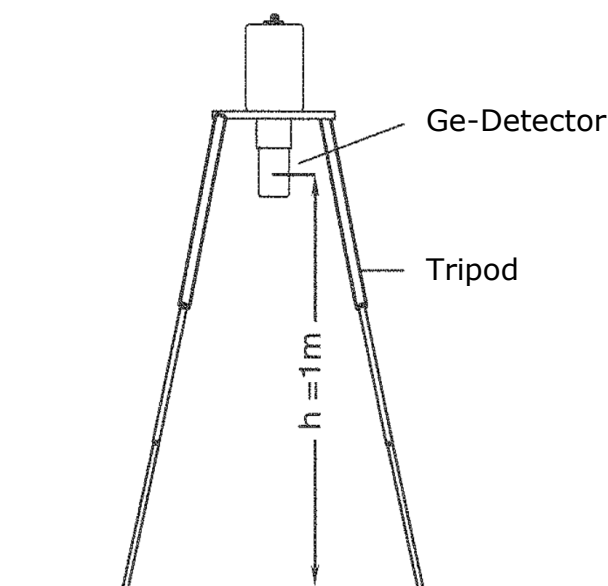
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Fig. 1: In-situ gamma spectrometer in position for measuring (8)

To make optimal use of the short measurement periods a rapid method like this offers, it needs to be ensured that the other steps required for taking measurements, such as assembling and dissembling the spectrometer and the transfer of processed data, can be effected just as quickly. This necessitates, amongst others, the availability of a suitable vehicle, easy handling of the required equipment, rapid yet reliable data processing, and regular training of all processes involved.

The maximum activity per area of the ground that can be measured by an in-situ spectrometer with the measuring geometry as illustrated in Figure 1 is limited by the number of pulses per second that can be processed by the processing electronics, that is pre-amplifier, amplifier, ADC, and multi-channel analyser. The limiting factor here is not the activity of the individual nuclides, but the sum of all individual activities. It needs to be taken into consideration that the number of pulses to be processed per second by the processing electronics does not only consist of the sum of the nuclide-specific peak counting rates, R_r (cf. section 5), but includes the background of the spectrum as well (Compton background of the detector, build-up effect). This contribution also increases with increasing activity per area and is at least of equal magnitude.

Commercially available electronic equipment is typically limited to counting 10 000 to 50 000 pulses per second. Extrapolated to I-131, this corresponds to a maximum limit of about 10 MBq·m^{-2} to 50 MBq·m^{-2}. The dose rate resulting from this is 12 µGy·h^{-1} to 60 µGy·h^{-1}.

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5   Calculation of the results

The analysis of an in-situ spectrum and the calculation of the activity per area are usually performed with the aid of commercially available computer software that converts, largely automatically, a spectrum into the nuclide-specific activities per area (in Bq·m^{-2}). It remains the responsibility of the user, however, to first implement the correct, energy-dependent calibration factors. The analytic software that the BMU has made available together with their in-situ measuring vehicles to the federal states for the aims of the StrVG contains wide-ranging support for the calculation of the calibration factors described below.

The activity per area, $a_{F,r}$ of the ground (in Bq·m^{-2}) is calculated from the net count rate, R, of the gamma line of a nuclide of interest, r, as follows:

$$a_{F,r} = \varphi \cdot R_r \tag{1}$$

Because it is virtually impossible to determine the calibration factor, ϕ, through directly measuring a defined, large-area, standard surface source, it is necessary to approximate it with the aid of mathematical model calculations. The reciprocal value of the calibration factor can be presented as the product of three factors, with each factor covering a different influencing variable:

$$\frac{1}{\varphi} = \frac{R}{\Phi} \cdot \frac{\Phi}{L_\gamma} \cdot p_\gamma = Q_D \cdot G_F \cdot p_\gamma \tag{2}$$

where

$Q_D = R_r/\Phi$ detector response function in m^2, i. e., the ratio of the count rate in the photo peak to the number of gamma quanta per second and square metre present at the measurement location

R_r net count rate, in s^{-1};

φ gamma flux in s^{-1}·m^{-2}, i. e., the number of undispersed gamma-quanta at the measuring location (centre of the detector) that arrive from any direction per area;

$G_F = \Phi/L_\gamma$ geometric function, i. e., the gamma flow density relative to the number of gamma quanta emitted from the ground per square metre and second;

L_γ specific emission density in s^{-1}·m^{-2}, i. e., the number of quanta of a gamma line of the radionuclide, r, per second that is created in a square metre of ground by radioactive decay. It is the product of the activity per area, $a_{F,r}$, and the absolute emission probability, p, of the gamma line.

p_γ absolute emission probability of a gamma line of the radionuclide, r (dimensionless).

The quantities Q_D and G_F as well as their quantification are explained in the following two sections.

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5.1 Detector response function

The detector response function, Q_D, is a detector-specific, energy-dependent quantity. To determine it, a point source of known activity, usually a calibrated source, is mounted at a distance of $d = 1$ m from the detector. It is recommended to use point sources of Eu-152 and Ba-133 for calibration purposes. Since the largest part of the gamma radiation will be incident nearly horizontally during the measurement (see measuring geometry as illustrated in Figure 1), a horizontal direction of irradiation needs to be selected for the determination of the detector response function as well (see Figure 2). The centre of the detector crystal has to be chosen as reference point. The horizontal distancing between the centre axis of the crystal and the source of calibration must be accurate to 5 mm at minimum. In doing so, the error in the detector response function is kept below 1 %. On the other hand, a reference point with a minor vertical misalignment from the centre of the crystal (l < 5 cm) may be accepted without incurring major errors.

As a matter of principle, in-situ measurements should make use of a nearly direction-independent detector. In the case of detectors with an expressed directional sensitivity (> 10 % anisotropy), respective corrections need to be implemented (cf. also (2) through (4)). The isotropy of the crystal therefore needs to be checked when the detector is calibrated for the first time.

Depending on the thickness of the calibration source, overlying lines originating from the natural background radiation may cause inaccurate results. It is therefore recommended that the detector be shielded with lead at least 5 cm thick during the calibration process; it must be ensured, however, that the direct radiation from the standard calibration source can reach the detector without being obstructed (cf. Figure 2).

The measurement period for the calibration process should be selected in a manner as to comprise more than 10 000 pulses each in the net peak areas of all gamma lines of the calibration source used for analysis. In order to still be able to analyse the 411 keV-line ($p_\gamma = 2,3$ %) of a Europium-152 source with an activity of 370 kBq, for example, a measurement on a 10 %-detector needs to be run for at least 12 hours. The aim is to obtain the largest number of gamma lines possible, covering the entire energy spectrum from 80 keV to 2000 keV.

The flux of the individual gamma lines of the calibration source is given by:

$$\Phi = \frac{A_r}{4 \cdot \pi} \cdot \frac{p_\gamma}{d^2} \tag{3}$$

where A_r is the activity of the calibration source (in Bq) and $d = 1$ m the distance of the calibration source from the centre of the detector. The detector response function for each energy is therefore:

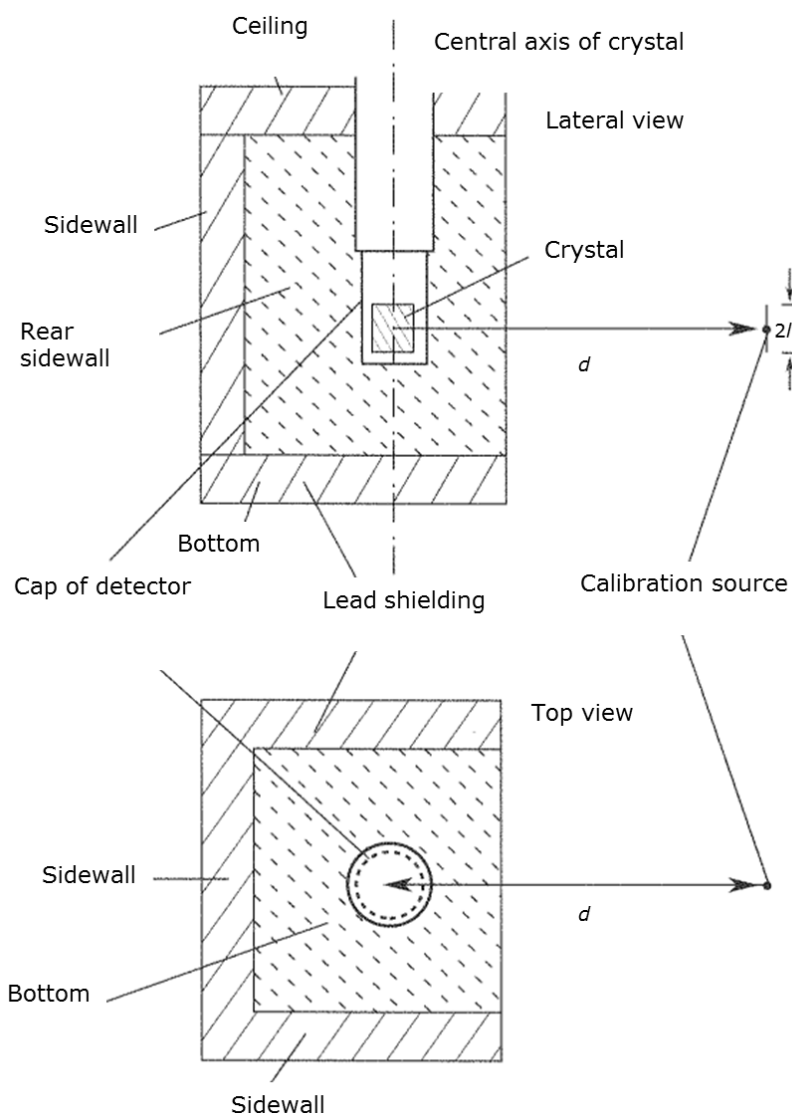$$\Phi = \frac{R_r \cdot 4 \cdot \pi \cdot d^2}{A_r \cdot p_\gamma} \tag{4}$$

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Fig. 2: Positioning for calibration of the detector with lead shielding on 5 sides. The distance, d = 1 m, has to be accurate to at least ± 0,5 cm; the tolerance, l, in vertical alignment of the centre of the crystal may be as much as 5 cm, though.

5.2 Geometry function

The geometry function G_F is a dimensionless quantity that depends on the measuring configuration and the gamma energy. Its calculation incorporates the following parameters:

— The structure of the ground and the density of the ground in particular;
— The vertical distribution of the radionuclides in the ground;
— The roughness of the surface of the ground.

Determining the geometric function with the aid of mathematical models supposes that the activity per area is spatially constant throughout the "field of view" of the spectrometer (cf. (2) through (4)).

The term geometry factor is often used in the literature. It refers to the product of the geometry function, G_F, and the absolute probability of emission, p_γ, and therefore has to be calculated for each nuclide separately.

The most important parameters of the structure of the ground are its density and moisture content. While the soil density may be supposed to be ca. 1,5 g·cm^{-3} for deeper strata, the density of the topmost centimetres, which are relevant here, may decrease to < 0,5 g·cm^{-3}, depending on root penetration, conditioning of the soil etc. For analytic purposes it is therefore presumed that the soil density has a value of 1,0 g·cm^{-3} and its moisture content amounts to 10 %. Deviations from these supposed values may cause systematic errors in the analytic calculation of the contamination of the ground. In the case of the soil density, deviations of max. ±0,5 g·cm^{-3} may be experienced and cause deviations of ca. ± 20 % in $a_{F,r}$. As far as the moisture content of the ground is concerned, the respective deviations amount to less than one percent.

The influence of the vertical distribution of a contamination in the ground on the geometric function is quantified by the relaxation length, $1/\alpha$ (1). This supposes that the activity decreases exponentially with increasing depth. The relaxation length, $1/\alpha$, characterises this decrease: at a depth of $1/\alpha$, the activity will be $1/e$ of the activity at the surface.

Surface roughness of the ground leads to an attenuation of the radiation even if the contamination is only present on the surface. In practice, this effect can be described by an additional, fictitious relaxation length. According to model calculations and experimental data (10, 11), a fictitious relaxation length of 0,1 cm to 0,3 cm may be assumed for an even lawn area.

The geometry function is calculated as follows for an exponential distribution of the radionuclides:

$$G_F = \frac{1}{2} \cdot \left[E_1(\mu_A \cdot h) - \exp\left(\frac{\alpha}{\mu_G} \cdot \mu_L \cdot h\right) \cdot E_1\left\{\left(1 + \frac{\alpha}{\mu_G}\right) \cdot \mu_A \cdot h\right\}\right] \tag{5}$$

Here, h refers to the height of the detector, μ_G and μ_A to the linear attenuation coefficients for ground and air, respectively, and E_1 to the exponential integral function of the first order. The precise calculation can be found in literature (12, 13). The geometric functions for cases that are relevant to the context of this measuring instruction can be found in Table 1 or Figure 3. It becomes clear that the geometry function for a given gamma-energy depends heavily on the relaxation length. The difference in the geometric function between a fresh surface contamination through dry deposition ($1/\alpha = 0,3$ cm) and one through wet deposition ($1/\alpha = 1$ cm) amounts to ca. 30 %. The difference between a situation of a fresh fallout ($1/\alpha$ ca. 0,3 cm to 1 cm) and an older contamination in the ground ($1/\alpha = 10$ cm) may be as much as one order of magnitude. This emphasizes the importance of knowing the current relaxation length for determining the geometric function and thus the conversion of the measurement results.

Tab. 1: Geometry function for various relaxation lengths, $1/\alpha$, and energies (soil density $\rho = 1,0$ g·cm^{-3})

Energy in keV	Geometry function			
	$1/\alpha = 0$ cm	$1/\alpha = 0,3$ cm	$1/\alpha = 1$ cm	$1/\alpha = 10$ cm
100	1,72	1,20	0,83	0,21
200	1,82	1,31	0,94	0,27
300	1,89	1,38	1,00	0,30
400	1,95	1,44	1,05	0,33
500	1,99	1,48	1,10	0,35
600	2,03	1,52	1,13	0,37
700	2,06	1,54	1,16	0,39
800	2,09	1,58	1,19	0,41
900	2,12	1,61	1,22	0,43
1000	2,15	1,63	1,24	0,44
1100	2,17	1,65	1,26	0,46
1200	2,19	1,67	1,28	0,47
1300	2,21	1,70	1,30	0,48
1400	2,23	1,72	1,32	0,50
1500	2,25	1,73	1,34	0,51
1600	2,27	1,75	1,35	0,52
1700	2,28	1,76	1,37	0,53
1800	2,30	1,78	1,38	0,54
1900	2,31	1,79	1,40	0,55
2000	2,32	1,81	1,41	0,56
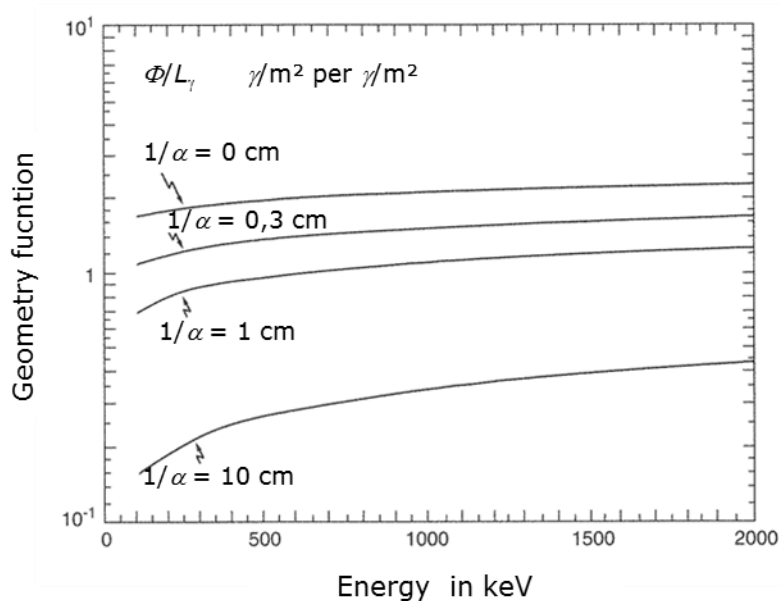
Fig. 3: Flux of unscattered gamma quanta in the energy range from 100 keV to 2000 keV in contamination of the ground with varying penetration depth ($1/\alpha = 0$ cm to 10 cm). Numerical values are given in Table 1.

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Rule of thumb

In cases where the actual relaxation length and the density of the ground are unknown, values of $1/\alpha = 1$ cm and $\rho = 1{,}0$ g·cm^{-3} are supposed for analyses of fresh contamination events.

In cases where it is certain that the deposition has been exclusively dry, a relaxation length of 0,3 cm has to be applied.

Because this selection of parameters will not always accurately describe the factual conditions, the two parameters, 'density of the ground' and 'relaxation length' always need to be submitted together with the nuclide-specific measurement results. This enables a quantification with relative accuracy of both the nuclide-specific gamma flux, $a_{F,r} \cdot G_F \cdot p_\gamma$, and the nuclide-specific local dose rate regardless of the circumstances. For data transferred to IMIS, the density of the ground is fixed at 1 g·cm^{-3} and the relaxation length can be selected only from two values, either 0,3 cm or 1,0 cm.

The differences in the geometric functions for various gamma-energies are slightly less at low penetration depths than at deep penetration depths, which is due to the absorption effect of radiation by radionuclides contained at deeper levels in the ground. In the energy range 200 keV to 2 MeV, they amount to about 30 % for $1/\alpha = 0{,}3$ cm, and a factor of 2 for $1/\alpha = 10$ cm. Below 200 keV, the absorption in the soil becomes increasingly important, and accordingly, the geometric functions decrease ever more rapidly.

In cases of e. g. landfills, where the vertical distribution will typically be unknown and possibly cannot be described with the aid of a relaxation length either, a quantitative interpretation of measurement data by means of in-situ spectrometry will be impossible. In the case of a qualitative identification of artificial radionuclides, determining the measurement result will always require specific additional studies (14).

The "field of view" of the detector configuration according to Figure 1 can be determined mathematically (2, 7). It depends on the relaxation length, the energy of the gamma-emitter of interest, and the height at which measurements are taken. It is defined via the ratio of the flux of a contamination of finite extent to an infinitely extended contaminated area: For a relaxation length of 1 cm, 90 % of the gamma-quanta registered originate from a distance of up to 17 m ($E = 200$ keV) and 34 m ($E = 2000$ keV), respectively. If the depth of penetration is larger (smaller), the respective "fields of view" become smaller (larger).

5.3   Calculation of the energy-dependent detection efficiency

In general, the energies of the calibration source will not exactly match the energies of the radionuclides of interest contained in the soil. It is therefore necessary to interpolate the calibration data from sections 5.1 and 5.2, and/or to fit an analytic function to these as is common practice for the detection efficiency in gamma spectrometry. In the case of in-situ spectrometry, the detection efficiency is determined from:

$$\varepsilon = Q_{\mathrm{D}} \cdot G_F \tag{6}$$

Figure 4 illustrates as an example the detection efficiency for a 10 %-detector and a relaxation length of 1 cm. The individual data points correspond to a calibration with Eu-152 and Ba-133 sources. The illustration follows a log-log pattern, with the interpolating graph consisting of two parables, one for 80 keV to 300 keV and another for the range of 300 keV to 2000 keV. This fitting procedure corresponds to the analytic software for the in-situ spectrometer the BMU has made available to the federal states for the aims of the StrVG. For illustration purposes, the same figure also contains the detector response function. The latter differs from the detection efficiency by the geometric function and would correspond to 4π times the detection efficiency if a point-shaped sample were mounted 1 metre from the detector.
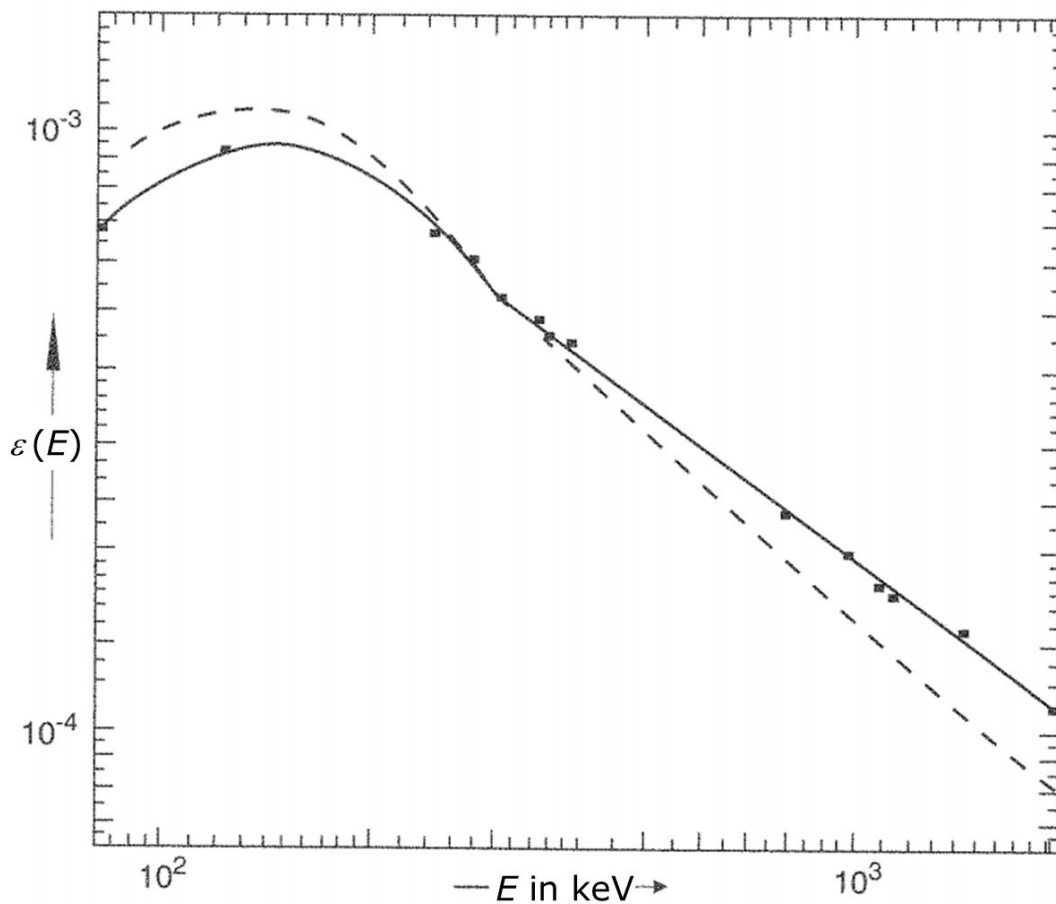

Fig. 4: Example of the detection efficiency, $\varepsilon(E)$, as a function of energy (solid curve) with the corresponding data points for $1/\alpha = 1$ cm. It results from the detector response function (dashed line) and the geometric function from Figure 3.

5.4   Worked example

Assumptions: The soil is contaminated with I-131 and Cs-137 caused by fresh precipitation. Measurements have been taken with a 10 % high-purity germanium detector at 1 metre above a meadow. The relaxation length is assumed to be 1 cm. The measurement period is 30 minutes.

	Nuclide	
	I-131 (364 keV)	Cs-137 (662 keV)
Measurement data		
Net count number	619	1397
Net count rate in s^{-1}	0,344	0,776
Data used for analysis		
Detector response function (from Fig. 4)	$4{,}45\cdot10^{-4}$	$2{,}27\cdot10^{-4}$
Geometry function (from Fig. 3 or Tab. 1)	0,94	1,16
Detection efficiency = detector response function x geometry function	$4{,}18\cdot10^{-4}$	$2{,}63\cdot10^{-4}$
Emission probability, p_γ	0,81	0,85
Calibration factor (from equation 2)	2951	4468
Activity per area calculated according to equation 1, $a_{F,r}$, in Bq·m^{-2}	1015	3467

5.5   Consideration of uncertainties

The accuracy of the results of the analysis depends to a high degree on the knowledge of the parameters of the models that are employed for the conversion of the measurement results data into activity per area of ground and the nuclide-specific contributions to the local dose rate (cf. Table 2). The importance of statistical counting uncertainties, which decrease as the measurement period increases, is usually of less importance.

On the basis of the results obtained up to now, including those following the accident at Chernobyl (5, 8), it can be stated that differences of up 30 % may be experienced between the results of in-situ spectrometry and those obtained in the laboratory in some instances, provided the vertical distribution of radionuclides in the soil is known from the laboratory analyses of soil samples (cf. procedure F-γ-SPEKT-BODEN). A comparison of the measured local dose rates with the values obtained from in-situ spectrometry will typically show a difference of 10 %.

The differences discussed here refer to measurements taken in the knowledge of model parameters and adhering (and being able to adhere) to all specific conditions (cf. (6)) required for the execution of the measurement. In cases where the model parameters are not known with the necessary accuracy, or where the model assumptions are not (or cannot be) adhered to, for example, for reasons of logistic preconditions, greater degrees of uncertainty must be expected (e. g., Figure 3).

It should be noted that the possible differences between the two measurement procedures mentioned here are not necessarily to be interpreted as *uncertainty* of the measurements of in-situ spectrometry. Rather do these differences point out that laboratory analyses may possibly include systematic errors as well, which may be due to inhomogeneities in the activity per area of the ground within the "field of view" of the in-situ spectrometer (cf. (8)), for example. Such inhomogeneities cannot be recorded with just one comparative sample. It is therefore recommendable to collect *more than one* sample of soil from within the "field of view" of the in-situ

spectrometer for the determination of the relaxation length in a contamination of the soil. As to how far the differences mentioned can be attributed to uncertainties in the in-situ spectrometry is a question that is difficult to answer in this case as well.

Tab. 2: Relative error in the activity per area of the ground depending on the variation of model parameters:

Relaxation length	= 1 cm
Density of the ground	= 1 g·cm⁻³
Height above ground at which measurement is taken	= 1 m

Parameter	Assumed variation	Rel. error (%)	
		Cs-137 (662 keV)	Te-132 (228 keV)
Relaxation length	± 1 cm	-20/+80 %	-25/+90 %
Density of the ground	± 0,5 g·cm⁻³	-15/+20 %	-15/+25 %
Moisture content of the soil	± 10 %	< ± 1 %	< ± 1 %
Height above ground at which measurement is taken	± 10 cm	< ± 1 %	± 1 %

6   Characteristic limits of the procedure

The detection limits of in-situ spectrometry depend on a number of quantities, i. e.:

— The properties of the detector, especially its relative efficiency;
— the measurement period;
— the penetration depth of the contamination in the soil;
— the natural background radiation of the ground at the measurement location.

Detection limits are calculated from equation (2.13) contained in chapter IV.5 of this procedures manual.

The data in Table 3 refer to a 10 %-detector and a measurement period of 30 minutes. The natural background radiation in this example corresponds to an activity in the ground of ca. 700 Bq·kg⁻¹ K-40 and 50 Bq·kg⁻¹ each of the nuclides of the uranium/radium and thorium decay chains. They apply to individual nuclides, i. e., in the case that the contamination is caused by only one nuclide. If the contamination involves several nuclides, the background will increase significantly from superposition of numerous Compton continua and weak lines. The detection limits and decision thresholds of Table 3 will not be attained in these cases.

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Tab. 3: Detection limits for a contamination of the soil with a relaxation length of 1 cm (measurement period: 30 minutes; relative detector efficiency: 10 %, and $k_{1-\alpha} + k_{1-\beta} = 4{,}645$)

Nuclide	Energy (keV)	Detection efficiency of the activity per area of the ground (Bq·m^{-2})
Co-60	1332	170
Zr-95	757	230
Nb-95	766	160
Mo-99	739	1100
Tc-99m	141	220
Ru-103	497	160
Ru-106	622	1250
Ag-110m	658	140
Sb-125	428	420
Te-129m	696	4200
Te-129	460	1900
I-131	365	160
I-132	668	140
I-133	530	160
Te-132	228	200
Cs-134	605	140
Cs-136	341	280
Cs-137	662	160
Ba-140	537	620
La-140	1597	190
Ce-141	145	390
Ce-144	133	1900

7 Catalogue of chemicals and equipment

7.1 Chemicals

not applicable

7.2 Equipment

— High purity germanium detector (with 10 % relative detection efficiency) with preamplifier and portable dewar (2 l to 5 l), splash water-resistant;
— Set of cables;
— Tripod for the detector;
— Data acquisition electronics (amplifier, ADC, memory) with high-voltage power supply unit for the detector;
— Computer for data processing;
— Point source for the calibration of the detection efficiency, ca. 400 kBq (e. g., Eu-152 and Ba-133);
— Point source for energy calibration (activity below the exemption level);
— Storage container for liquid nitrogen.

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