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Good Practice Guide for the Measurement of Surface Activity and Mapping the Contamination

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1. Surface contamination

1.1 General

1.1.1 Radioactive decay and radionuclides

Radioactive decay or radioactivity is the process by which an unstable atomic nucleus loses energy by emitting radiation, such as alpha particles, beta particles, gamma rays and electrons. A material containing such unstable nuclei is considered radioactive. Radioactive decay is a nuclear process and is largely independent of the chemical and physical states of the nuclide. Radioactive nuclides decay spontaneously by the following processes: (i) Alpha decay; (ii) Beta decay; (iii) Electron capture; (iv) Internal transitions; (v) Isomeric transitions.

The first radioactive elements – radium and polonium – were discovered by Marie Curie at the end of the nineteenth century. The first artificial radionuclide, ^{30}P , was produced by Frederic and Irene Joliot-Curie in an accelerator by bombarding aluminum with protons. Today, more than two thousand artificial radionuclides have been produced and identified.

In nature there are three types of radionuclides: those belonging to the decay chains of uranium and thorium, single very long-lived radionuclides, and cosmogenic radionuclides. Artificial radionuclides form the largest group of radionuclides. The main sources of artificial radionuclides are: (i) nuclear weapons production and explosions; (ii) nuclear energy production; (iii) radionuclide production by reactors and accelerators.

1.1.2 Quantities associated with radioactive decays

The main quantities associated with radioactive decays are: (i) energy of emitted particles or photons; (ii) maximum energy of the beta spectrum; (iii) transition probability; (iv) emission probability; (v) activity; (vi) decay constant and (vii) half-life.

Transition probability assigned to a specific transition is the probability that this transition occurs when one nucleus of a given radionuclide disintegrates. The emission probability assigned to the particles or photons (specified by type and energy) emitted by a radionuclide is the probability that such a particle or photon will be emitted when one nucleus of this radionuclide disintegrates, i.e. the emission probability is the mean number of particles or photons emitted per disintegration of one nucleus. According to the above definition, the emission probability for alpha, beta and gamma radiation has a value between 0 and 1. For X-rays, Auger electrons and gamma radiation due to positron annihilation, the emission probability may exceed 1. Because the probability is smaller than 1, the notation “number of electrons (or photons or alpha) per disintegration” can be used instead of emission probability.

The activity, A , of an amount of radioactive nuclide in a particular energy state at a given time is the quotient of dN by dt , where dN is the expected value of the number of spontaneous

nuclear transitions from that energy state in the time interval dt . Hence, we have $\Lambda = dN / dt$. The special name for the unit of activity is Becquerel (Bq) and 1 Bq = one disintegration per second. The old unit of activity is Curie (Ci) and 1 Ci = 3.7×10^{10} Bq.

The decay constant, λ , is the probability of disintegration of a given radionuclide. The radioactive decay law represents the decrease of the activity of a radioactive substance with time by spontaneous nuclear transformations. It is expressed by the following equations

$$dN / dt = -\lambda N \quad N = N_0 e^{-\lambda t} \quad \Lambda = \Lambda_0 e^{-\lambda t} \quad (1)$$

where N is the number of atoms of the corresponding radionuclide consisting of nuclei being in the same energy state at time t , N_0 is the number of such atoms at time $t = 0$. The half-life, $T_{1/2}$, is the time required for the initial number of atomic nuclei, in a given state, to decrease by a factor of two. It follows that $T_{1/2} = \ln(2) / \lambda$.

1.2 Mechanism of radioactive contamination

Contamination of materials occurs because of the physical or chemical transfer of radioactive materials onto surfaces. Some contamination may be strongly adhered to the surface, which is called fixed contamination, and the other is relatively easy to remove, referred to as removable or loose contamination. Generally, removable contamination is simply deposited on the surface and has little interaction with the surface. The primary mechanisms by which radioactive contamination is produced include: (i) simple deposition of solid particles on the surface, as in the removable contamination case; (ii) material deposition on and within a barrier layer (e.g. an oxide layer) formed on the substrate surface; (iii) materials deposited via transport into the bulk of the substrate (Demmer et al., 2014).

For smooth non-porous materials, the removal of loose contamination is easy requiring usually a wipe with a damp cloth or washing with water. Loose contamination can become fixed contamination if it remains on surfaces for long periods of time. For rough non-porous materials, the removal of the loose contamination is difficult because it is retained by the surface irregularities.

In water cooled nuclear reactors, the main part of radioactive contamination comes from corrosion products from the primary coolant system which were activated during their passage through the reactor core. Additionally, a small amount of fission product from an accidental fuel cladding breach may also get into the coolant. A part of these corrosion and fission products are trapped by the metal oxide layer of recirculation pipes and other components.

In some circumstances, paint was applied on different contamination surfaces to fix the loose contamination. Also, there may be cases when thick layers of oil, grease, dirt and dust cover

contamination surfaces. In these cases, the evaluation of the surface contamination using the direct method of measurement is difficult especially for soft beta emitters such as ^{14}C and ^{147}Pm .

Porous materials, such as wood and unsealed concrete, have the property of absorbing contaminants beyond their surface. The evaluation of the surface contamination for porous surfaces using the direct method of measurement is difficult and often impossible. It can be achieved provided that the identity of the radionuclide is known, the maximum depth penetration of the contamination is smaller than the maximum range of beta particles and there is some knowledge on the surface condition and the depth distribution of the nuclide.

1.3 Contamination source

Contamination sources are composed of the substrate material and materials of active and inactive layers. The substrate is the supporting material of the source on which the active layer is formed. A contamination source model is schematically shown in Figure 1 (ISO 7503-3, 2016). The substrate of the contamination source may be a non-porous or porous material which may have either a smooth or rough surface. There are many non-porous materials with smooth surfaces such as polished metals, painted/plated metals, glass, hard plastics, linoleum, painted wood, painted wallboard and glazed ceramic tiles. Unpainted/unpolished steel and stainless steel have relatively smooth surfaces. Cast iron and galvanized steel are examples of materials with rough surfaces. The active layer of the source contains the radioactive material, which has usually a non-uniform activity depth distribution. The activity distribution of the radionuclide material in the plane of the source surface is also non-uniform. The inactive layer is usually formed by deposition of different materials (e.g. water and dust). The main parameters of the contamination source are x_M and s , which are the thicknesses of active and inactive layers, respectively. The thickness of the source, $\chi = x_M + s$, varies within a broad range, typically from hundreds of nm to hundreds of μm .

Surface beta contamination measurements are performed by means of beta contamination monitors provided with thin-window large-area detectors. To obtain reliable results, the surface contamination measurement must be performed in a fixed counting geometry keeping the detector face stationary above the contaminated surface (e.g. a distance of 3 mm from the surface). The aim of this measurement is to determine the average value of the activity per unit area, A_S , of the contamination source. Taking into consideration that the contamination source contains the radionuclides from the area of the orthogonal projection of the detector window onto the contaminated surface, A_S is defined by

$$A_S = \frac{A}{S} \quad (2)$$

where S is the area of the detector window and Λ is the activity of the contamination source. Assuming that the source contains a single beta emitter, the efficiency of the contamination source for beta radiation and electrons, ε_b , is defined by

$$\varepsilon_b = \frac{E_b}{\eta_b \Lambda} \quad (3)$$

where E_b is the surface beta emission rate of the contamination source and η_b is the total emission probability for beta particles and electrons.

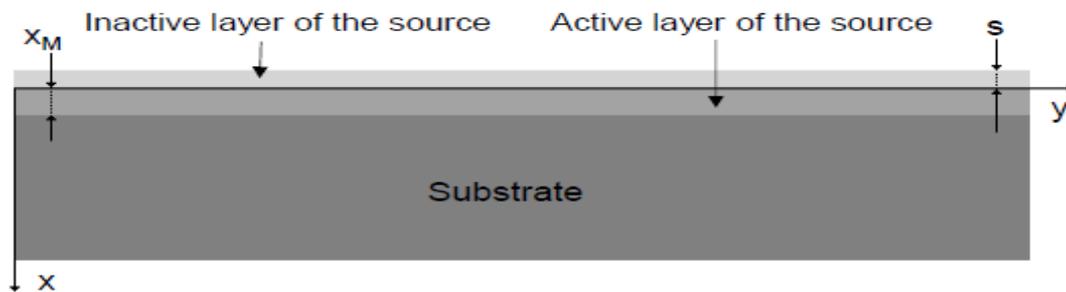


Figure 1. Schematic view of a contamination source

2. Measurement of surface contamination

2.1 General

2.1.1 Detection of ionizing radiation

Ionizing radiation is any type of particle or photons that carries enough energy to ionize or remove electrons from an atom. There are two types of photons that can ionize atoms: gamma-rays and X-rays, and sometimes they have the same energy. Gamma radiation is produced by interactions within the nucleus, while X-rays are produced outside of the nucleus by electrons. There are officially two types of ionizing radiation that are energetic particles emitted during an interaction within the nucleus. The alpha particle is composed of two protons and two neutrons, or a helium nucleus. The beta particle is either a positron or an electron. Neutrons emitted during some nuclear decay processes are often included as ionizing particles but they do not actually ionize an atom directly. Neutrons interact with another nucleus, which may result in a secondary process involving ionizing radiation.

Ionizing radiation is generated through nuclear decay, nuclear reactions, by very high temperature, or via acceleration of charged particles in electromagnetic fields. Natural sources include the sun, lightning and supernova explosions. Artificial sources include nuclear reactors, particle accelerators, and X-ray tubes. Radiation is a form of energy. This energy can be partly or wholly deposited in a suitable medium and thus produce an effect. The detection and

measurement of radiation is based upon the detection and measurement of its effects in a medium, and the history of the emergence of radiation detectors is closely related to the discovery of radiation and radiation effects. The development of radiation detectors could not have taken place without an important development that occurred in parallel: the development of equipment to receive signals from radiation detectors, process them and produce a convenient output.

2.1.2 Measurement of radioactivity

Measurement of radioactivity refers to the measurement of the activity of samples containing radionuclides. Generally, radioactive measurement systems consist of two parts: (1) a radiation detector or an array of detectors that determines the main features of the measurement system and (2) electronic equipment that enable the measurement of the radioactivity.

In practice, most radionuclide measurement systems make use of relative methods of measurement and are based on pulse counting technique. Radiation detectors should have a low and stable radiation background to reduce interference and uncertainty in measurements. Contributors to the radiation background for a detector are the following: (a) Cosmic and terrestrial radiations; (b) Radionuclides in the detector and nearby components of the detector system; (c) Radioactive contamination on the detector, sample holder, and other components; (d) Radionuclides in the counting room walls, floor, and air; (f) The radiation environment near the counting room; (g) Other radionuclides in the source that is being counted.

A detector operated in pulse mode produces a pulse associated with individual events. In many instances, the pulse is proportional to the energy of the incident radiation. Measuring systems using detectors that utilize this proportionality feature are known as spectrometers and are used not only for measuring the activity of radionuclides but also for identifying them.

Other detectors, known as gross radiation counters, measure and count pulses regardless of energy. For measurements using gross (or integral) counting, the count rate, R , provided by the radioactivity measurement system must be corrected for dead time losses, decay and background (B). When the corrections for dead time losses and decay can be neglected, the net count rate is given by $R_{net} = R - B$.

The activity, A , of a given radioactive source containing a single radionuclide and the response of a radioactive measurement system (net count rate) are related by (ICRU, 1994)

$$R_{net} = \varepsilon_c A \quad (4)$$

where ε_c is the overall or the counting efficiency of the measurement system for a given counting geometry. The counting efficiency thus represents the ratio between R_{net} and A . It depends on the detection efficiencies for different particles and photons emissions. For given

radionuclide, measurement system and counting geometry, the detection efficiency, ε_{di} for a given type of emission (e.g. conversion electron emissions) is defined by

$$\varepsilon_{di} = \frac{R_i}{\eta_i \Lambda} \quad (5)$$

where R_i is the net count rate due to “ i ” type emission and η_i is the total number of type “ i ” emissions per decay. Because ε_{di} depends on η_i and energy of particles or photons emissions, it follows that ε_c depends on the parameters of the decay scheme of the radionuclide. As a result, to measure the activity of a given radionuclide, it must firstly be identified. Radionuclides emitting alpha or gamma radiations can simultaneously be identified and measured using spectrometers. Chemical separation can be used to identify isotopes that do not emit gamma or alpha radiations. In case of measurements using gross counting, the radionuclides from a mixture must firstly be identified and their percentage activity must be known. Considering a mixture composed of n nuclides having the activities Λ_i , the counting efficiency is defined by

$$\varepsilon_c = \frac{R_{net}}{\Lambda} = \left(\sum_{i=1}^n R_{net,i} \right) / \Lambda = \sum_{i=1}^n (\Lambda_i / \Lambda) \frac{R_{net,i}}{\Lambda_i} = \sum_{i=1}^n c_i \varepsilon_{c,i} \quad (6)$$

where $\Lambda = \Lambda_1 + \dots + \Lambda_n$, $\varepsilon_{c,i} = R_{net,i} / \Lambda_i$ and $c_i = \Lambda_i / \Lambda$ is the percentage activity. It follows that

$$\Lambda_i = c_i \frac{R_{net}}{\varepsilon_c} \quad (7)$$

Eq. (7) shows that the activity, Λ_i , of the i th radionuclide from a mixture can be determined by gross counting if all radionuclides from the mixture were identified, $\varepsilon_{c,i}$ and c_i are known.

2.1.3 Measurement of the surface contamination using direct and indirect methods

Surface contamination is quantified in terms of activity per unit area, the quantity which is normally used to specify “derived limits”, i.e. maximum limits of surface contamination (ISO, 2016). Surface contamination can be evaluated by direct and indirect methods of measurement. Direct measurements are carried out with surface contamination monitors that respond to both removable and fixed surface contamination (ISO 7503-1, 2016). Indirect evaluation of the surface contamination is used for assessing the removable contamination (ISO 7503-2, 2016).

The direct evaluation of the surface contamination by means of contamination monitors is based on the detection and gross counting of alpha and beta particles, electrons, gamma radiation and X-rays using thin-window large area detectors. The direct method is the best approach whenever possible. In the direct method, the monitor probe is kept stationary above a contaminated surface, with the face of the probe at a minimal distance of approximately 3 mm from the surface. The probe shall be kept for a minimum to obtain sufficient accuracy.

Contamination that is fixed to the surface would not be transferred to the body and is usually of concern only as a source of external exposure. However, it is the removable contamination that is transferrable to humans by contact, inhalation, or ingestion that poses a hazard of internal radiation exposure. The levels of removable contamination that are of concern may be much lower than those for fixed contamination.

The amount of removable contamination is usually determined by obtaining wipe samples. Usually, only a portion of the removable activity is collected on the first wipe, and several wipes are needed to assess the total removable activity. The interpretation of wipe sample results quantitatively is generally difficult because the sampling and measurement uncertainties are often not adequately evaluated.

2.2 Direct measurement of surface contamination

2.2.1 Direct measurement of contamination sources according to ISO 7503

According to ISO 7503-1, when only one radionuclide is responsible for the contamination, the activity per unit area, A_S , of a contamination source, measured by means of a contamination monitor, is given by

$$A_S = \frac{A}{S} = C_A R_{net} = \frac{R_{net}}{I_A} \quad (8)$$

where C_A and $I_A = 1/C_A$ are, respectively, the activity calibration factor of the instrument and the activity instrument response, R_{net} is the observed count rate from the contamination source corrected for dead time losses, background and decay. If the corrections for dead time losses and decay are negligible then $R_{net} = R - B$ where R is the count rate observed from the contamination source and B is the background rate.

ISO 7503-3 provides a complicated formula for calculating I_A (see ISO 7503-3, Eq. (9)), which takes into account the coincidence summing effect. Also, ISO 7503-3 in Annex A shows that the correction due to the coincidence summing effect is small (it does not exceed 7 %) and can consequently be neglected. Under these conditions, considering a radionuclide emitting alpha and beta particles, electrons and photons and taking into account the instrument efficiency and the source efficiency defined by ISO 7503, I_A can be expressed as

$$I_A = \frac{R_{net}}{A} S = \left[\eta_a \left(\frac{R_a}{E_a} \right) \left(\frac{E_a}{\eta_a A} \right) + \eta_b \left(\frac{R_b}{E_b} \right) \left(\frac{E_b}{\eta_b A} \right) + \eta_g \left(\frac{R_g}{E_g} \right) \left(\frac{E_g}{\eta_g A} \right) \right] S \quad (9)$$

where $R_{net} = R_a + R_b + R_g$, R_a , R_b and R_g are, respectively the count rates due to alpha, beta (including electrons) and gamma (including X-rays) radiations, E_a , E_b and E_g are, respectively, surface emission rate of alpha particles, beta particles (including electrons) and photons, η_a , η_b and η_g are, the number of emissions per decay corresponding to alpha particles, beta particles (including electrons) and photons, respectively. Taking into account the instrument efficiency and the source efficiency defined by ISO 7503, Eq. (9) becomes

$$I_A = \eta_a e_a \varepsilon_a + \eta_b e_b \varepsilon_b + \eta_g e_g \varepsilon_g \quad (10)$$

where e_a , e_b and e_g are, respectively, the instrument efficiency for alpha and beta particles (including electrons) and photons, ε_a , ε_b and ε_g are, respectively, the source efficiency for alpha and beta particles (including electrons) and photons.

To calculate I_A by means of Eq. (10), an approximation to the value of I_A for contamination sources can be obtained by taking $e_a = e_a^{ref}$, $e_b = e_b^{ref}$ and $e_g = e_g^{ref}$, where the values of e_a^{ref} , e_b^{ref} and e_g^{ref} are determined during the calibration using reference sources. The values of ε_g are usually equal to 0.5, excepting the case of low energy photons when this value must be corrected for the photon attenuation in the source material. For the values of ε_a and ε_b , the standard ISO 7503-3 provides conservative values. Also, it recommends the empirical estimation of their values, whenever possible.

The relative standard uncertainty of Λ_S can be calculated by means of Eq. (8) using law of propagation of uncertainty (ISO, 1995). Thus, we have

$$\delta^2(\Lambda_S) = \delta^2(R_{net}) + \delta^2(I_A) \quad (11)$$

where $\delta(R_{net})$ and $\delta(I_A)$ are relative standard uncertainties of R_{net} and I_A , respectively. The relative standard uncertainty of I_A can be calculated by means of Eq. (10) and is given by

$$\delta^2(I_A) = \delta^2(S) + \omega_a^2[\delta^2(e_a) + \delta^2(\varepsilon_a)] + \omega_b^2[\delta^2(e_b) + \delta^2(\varepsilon_b)] + \omega_g^2[\delta^2(e_g) + \delta^2(\varepsilon_g)] \quad (12)$$

where $\omega_a = \eta_a e_a \varepsilon_a (S/I_A)$, $\omega_b = \eta_b e_b \varepsilon_b (S/I_A)$, $\omega_g = \eta_g e_g \varepsilon_g (S/I_A)$ and $\delta(I_A)$, $\delta(e_a)$, $\delta(e_b)$, $\delta(e_g)$, $\delta(\varepsilon_a)$, $\delta(\varepsilon_b)$ and $\delta(\varepsilon_g)$ are the relative standard uncertainties of I_A , e_a , e_b , e_g , ε_a , ε_b and ε_g , respectively.

Eq. (10) can be applied when the method of direct calibration of an instrument with respect to a specific radionuclide, using a single reference source made from the same radionuclide, is employed. In the case of the multisource calibration procedure, a basic set of reference sources

are used for determining the instrument efficiency in four different energy regions, which cover the whole energy range of beta and gamma emitters (ISO 7503-3, 2016). Also, it is assumed that the values of the instrument efficiency and the source efficiency are constant over each region. Under these conditions, the activity instrument response, I_A^{ms} , is given by

$$I_A^{ms} = \eta_a e_a \varepsilon_a + \sum_{k=1}^4 (\eta_{b,k} e_{b,k} \varepsilon_{b,k} + \eta_{g,k} e_{g,k} \varepsilon_{g,k}) \quad (12)$$

where $\eta_{b,k}$, $e_{b,k}$ and $\varepsilon_{b,k}$ are, respectively, the summarized emission probability of beta particles and electrons, instrument efficiency and source efficiency for beta radiation and electrons corresponding to the k -th energy region, $\eta_{g,k}$, $e_{g,k}$ and $\varepsilon_{g,k}$ are, respectively, the summarized emission probability of photons, the instrument efficiency and source efficiency for photons corresponding to the k -th energy region.

The relative standard uncertainty of I_A^{ms} , $\delta(I_A^{ms})$, can be calculated by means of Eq. (12).

Thus, we have

$$\delta^2(I_A^{ms}) = \delta^2(S) + \alpha_a^2 [\delta^2(e_a) + \delta^2(\varepsilon_a)] + \sum_{k=1}^4 \left\{ \alpha_{b,k}^2 [\delta^2(e_{b,k}) + \delta^2(\varepsilon_{b,k})] + \alpha_{g,k}^2 [\delta^2(e_{g,k}) + \delta^2(\varepsilon_{g,k})] \right\} \quad (13)$$

where $\alpha_a = \eta_a e_a \varepsilon_a (S / I_A^{ms})$, $\alpha_{b,k} = \eta_{b,k} e_{b,k} \varepsilon_{b,k} (S / I_A^{ms})$, $\alpha_{g,k} = \eta_{g,k} e_{g,k} \varepsilon_{g,k} (S / I_A^{ms})$ and $\delta(e_{b,k})$, $\delta(\varepsilon_{b,k})$, $\delta(e_{g,k})$ and $\delta(\varepsilon_{g,k})$ are uncertainties of $e_{b,k}$, $\varepsilon_{b,k}$, $e_{g,k}$ and $\varepsilon_{g,k}$, respectively.

2.2.2 Calibration of contamination monitors

Instrument efficiency shall be determined under geometrical conditions which shall be as close as practicable to the conditions met during the measurement. The area of the calibration source should be sufficient to cover the active or working area of the detector. Where the detector is larger than the calibration source sequential measurements shall be carried out with the calibration source moved across the surface of the detector. These measurements shall cover the whole of the window area of the probe and give an average value for the instrument efficiency. The calibration source type may be adapted to the expected homogenous or heterogeneous contamination distribution.

According to ISO 7503-3, the instrument calibration should be done by means of reference radiations provided by reference sources of certified surface emission rate in accordance with ISO 8769. It follows that the the calibration of contamination monitors must be performed on the surface emission rate basis. The same standard recommends two calibration procedures:

1. a procedure for the direct calibration of an instrument with respect to a specific radionuclide using a single calibration source made from the same radionuclide.
2. a multisource procedure.

The first procedure can be applied to all radionuclides for which calibration sources are available and all types of radiation emitted can, in principle, be utilized for such a calibration. In this case, reference values of e_b and e_g for beta-gamma emitters can be determined using reference sources with certified values of the surface beta emission rate and activity. Thus, the reference values, e_b^{ref} and e_g^{ref} , are given by

$$e_b^{ref} = \frac{R_{bg}^{ref} - R_g^{ref}}{E_b^{ref}} \quad e_g^{ref} = \frac{R_g^{ref} - B}{E_g^{ref}} \quad (14)$$

where R_{bg}^{ref} is the count rate observed from the reference source, R_g^{ref} is the count rate observed from the reference source when it is covered with adequate filters to suppress beta particles and electrons, E_b^{ref} is the surface beta emission rate and $E_g^{ref} = \eta_g (A_{ref} / 2)$ is the surface photon emission rate (A_{ref} is the activity of the reference source). In case of low energy photons, the value of E_g^{ref} must be corrected for the photon attenuation in the source material. For pure beta emitters, $R_g^{ref} = B$. For alpha-gamma emitters, e_a^{ref} can similarly be determined using reference sources with certified values of the surface emission rate and activity.

For the application of the multisource calibration procedure, calibration can be performed by a set of basic radionuclide reference sources with certified alpha, beta, or photon surface emission rate (see ISO 7503-3, Table A.1). These calibration reference sources are used as emitters of alpha particles, electrons, or photons of a particular energy range. They are not perceived as sources of particular radionuclides. This procedure can be applied in case of radionuclides with complex emission characteristics or of radionuclides for which no calibration sources are available. The instrument efficiency is measured versus radiation energy using sources emitting mono-energetic radiation. Instrument efficiency values for the radionuclides under consideration are then calculated individually, using the energy and emission probability data relating to the mono-energetic components of the radiation. In this case, reference values of e_b and e_g for beta-gamma emitters, corresponding to the k th energy region, can be determined using reference sources corresponding to the same energy region. Thus, the reference values, $e_{b,k}^{ref}$ and $e_{g,k}^{ref}$, are given by

$$e_{b,k}^{ref} = \frac{R_{b,k}^{ref} - B}{E_{b,k}^{ref}} \quad e_{g,k}^{ref} = \frac{R_{g,k}^{ref} - B}{E_{g,k}^{ref}} \quad (15)$$

where $R_{b,k}^{ref}$ and $R_{g,k}^{ref}$ are, respectively, the count rates observed from beta-emitting and gamma-emitting reference sources corresponding to the k th energy region, $E_{b,k}^{ref}$ and $E_{g,k}^{ref}$ are, respectively, the surface beta and gamma emission rate corresponding to the k th energy region and B is the background of the contamination monitor.

2.2.3 Efficiency transfer method applied to surface contamination measurements

The activity per unit area, Λ_S , of contamination sources is defined by $\Lambda_S = \Lambda / S$. The activity, Λ , of a given radioactive source containing a single radionuclide and the response of a radioactive measurement system (net count rate) are related by $R_{net} = \varepsilon_c \Lambda$, where ε_c is the overall or the counting efficiency of the measurement system for a given counting geometry (ICRU, 1994). It follows that

$$\Lambda_S = \frac{R_{net}}{S\varepsilon_c} = C_A R_{net} \quad (16)$$

where $C_A = 1/(S\varepsilon_c)$. Neglecting the coincidence summing effect, ε_c can be expressed as

$$\varepsilon_c = \frac{R_{net}}{\Lambda} = \frac{R_a + R_b + R_g}{\Lambda} = \eta_a \varepsilon_{da} + \eta_b \varepsilon_{db} + \eta_{ph} \varepsilon_{dg} \quad (17)$$

where $R_{net} = R_a + R_b + R_g$ is the count rate recorded by the monitor and corrected for background, dead time losses and decay, R_a , R_b and R_g are, respectively, the count rates due to alpha, beta particles (including electrons) and photons, ε_{da} , ε_{db} and ε_{dg} are, respectively, the detection efficiencies for alpha, beta radiation (including electrons) and photons.

Taking account of Eq. (17) and applying the efficiency transfer method, we get

$$C = \frac{1}{S\varepsilon_c} = \frac{1}{S(\eta_a T_a \varepsilon_{da}^{ref} + \eta_b T_b \varepsilon_{db}^{ref} + \eta_g T_g \varepsilon_{dg}^{ref})} \quad (18)$$

where $T_a = \varepsilon_{da} / \varepsilon_{da}^{ref}$, $T_b = \varepsilon_{db} / \varepsilon_{db}^{ref}$ and $T_g = \varepsilon_{dg} / \varepsilon_{dg}^{ref}$ are efficiency transfer factors from the reference measurement to the measurement of the actual contamination source, T_a , T_b and T_g are efficiency transfer factors for alpha particles, beta radiation (including electrons) and photons. The quantities ε_{da} , ε_{db} and ε_{dg} are determined during the calibration using reference sources with certified values of the activity. In this case, the procedure for the direct calibration of an instrument with respect to a specific radionuclide using a single calibration source made from the same radionuclide must be applied.

The reference values of ε_{db} and ε_{dg} for beta-gamma emitters, ε_{db}^{ref} and ε_{dg}^{ref} , are given by

$$\varepsilon_{db}^{ref} = (R_{bg}^{ref} - R_g^{ref}) / (\eta_b A_{ref}) \quad \varepsilon_{dg}^{ref} = (R_g^{ref} - B) / (\eta_g A_{ref}) \quad (19)$$

where R_{bg}^{ref} is the count rate observed from the reference source due to beta and gamma radiations, R_g^{ref} is the count rate observed from the reference source due gamma radiation (beta radiation and electrons emitted by the reference source were suppressed using filters according to ISO 8769) and A_{ref} is the activity of the reference source. For alpha-gamma emitters, ε_{da}^{ref} can similarly be determined using reference sources with certified values of the activity.

To apply in practice the efficiency transfer method, the efficiency transfer factors T_a , T_b and T_g must be calculated and/or determined experimentally.

2.3 Indirect method for evaluating surface contamination

2.3.1 Introduction

Indirect evaluation of surface contamination is more generally applicable when the surfaces are not readily accessible because of difficult location or configuration, where interfering radiation fields adversely affect contamination monitors or when methods of direct measurement with standard instrumentation are not available (e.g. tritium).

The amount of removable contamination is usually determined by obtaining wipe samples. The removable surface contamination transferred to a wipe sample will vary according to the: (i) type of wipe material; (ii) method used; (iii) physical and chemical nature of the contaminated material; (iv) surface roughness of the material wiped, and (v) physical and chemical nature of the radionuclide contaminants. In order to determine the extent of surface contamination of materials and the effectiveness of the decontamination processes reliably, the wipe removal factor for the contaminant must be determined for the various materials wiped. The removal factor is the ratio of the activity of the radionuclides removed from the surface by one initial swipe sample to the total removable activity. This wipe removal factor must then be applied when evaluating the measurement results of the wipe samples. Among the many types of wipe materials used are dry wipes that use various dry absorbent materials such as glass and cellulose fibres, and wet wipes used by application of various solvents to the dry wipe to enhance the amount of material removed from the surface.

2.3.2 Guidelines on wipe test sampling

According to ISO 7503-2, when taking wipe samples from large areas, the following points shall be taken into consideration:

- a) Outline the required area to wipe using string or tape, for example, 100 cm² for surface contamination guideline values or 300 cm² for surface contamination limits.

- b) If a wetting agent is used for moistening the wipe material, this wetting agent should not exude from the material.
- c) The radioactive material should be dissolved or suspended in the wetting agent. The wetting agent water should be used for tritium, dust and water-soluble compounds.
- d) The wipe should be pressed moderately against the surface to be checked, using fingertips or, preferably, by means of a holder.
- e) The entire area required by regulations shall be wiped.
- f) After sampling, if using other methods than liquid scintillation counting, the wipe material shall be carefully dried in such a way that loss of activity is prevented. In case of contaminations of tritium, iodine isotopes or other volatile isotopes, drying of the wipe material shall be omitted.
- g) Use extreme caution when handling potentially contaminated wipe materials in order to prevent cross-contamination.

Detection and evaluation of removable tritium contamination or other volatile radionuclides should be carried out using wet wipe samples and the following points shall be taken into consideration when taking wipe samples:

- a) If possible, the area to be wiped should measure 100 cm².
- b) Where regulations permit larger areas to be wiped, such areas shall be used for sampling and their actual size shall be included in the calculation of the result; wiping of very large areas with a single wipe should be avoided since the removal factor decreases with increasing area.
- c) The wipe shall be pressed moderately against the surface to be wiped, preferably by means of a holder which is designed to ensure uniform and constant pressure.
- d) The entire area of 100 cm² or larger shall be wiped.
- e) Unused wipe materials should be stored in a tritium free atmosphere.
- f) The use of disposable gloves is recommended during wipe testing of highly contaminated surfaces; changing of gloves may be necessary to avoid cross-contamination of samples.

2.3.3 Instrumentation

Measurements of wipe samples are often carried out using well shielded, pulse-counting, installed laboratory instruments. Portable surface contamination meters or monitors can also be used, but not in preference to well shielded installed laboratory instruments. If a portable instrument is calibrated for the direct method of surface contamination evaluation, it is of value to calibrate it also for the indirect method (see ISO 7503-3). In the absence of interfering radiation fields, portable instruments may be used to evaluate surface contamination when preferred laboratory instruments are not available. If wipe tests with high activities are

expected, a pre-measurement using an adequate portable measuring instrument is recommended to avoid contamination of installed instruments.

Liquid scintillation counting is primarily used to count low energy betas particles and to a lesser extent alpha particles. Counting efficiencies range from 30 % up to 100 %. The liquid scintillation method is most often used to measure ^3H , ^{63}Ni , and ^{14}C . Emission can be detected by liquid scintillation counting with varying degrees of efficiency depending on the following: (i) the type and energy of emission (X-rays and photons may escape from the scintillator without interacting); (ii) whether the sample is dissolved in the scintillator; in the form of particles; or retained in the wipe material; (iii) the effect of chemicals on the scintillator; (iv) absorption of the emitted light by colour or opacity of the sample before it reaches the photomultiplier(s); (v) the orientation of the wipe in the sample vial.

2.3.4 Measurement procedure

According to ISO 7503-2, the operating instructions for the instruments used for measuring the wipe samples shall be followed. The background radiation level in the area where the wipes are counted should be low and constant. The following requirements shall be followed:

- a) Before making a wipe measurement, the background count rate shall be determined at the place of measurement.
- b) The background count rate shall be checked from time to time to account for changes in background levels.
- c) The instrument shall have a current calibration status and shall be verified using a suitable check source. Frequency: daily for instruments in frequent use, otherwise before each use. Deviations higher than 25 % from the agreed value impose the recalibration of the instrument.
- d) Instrument calibration factors suitable for the nuclides to be measured must be available.
- e) The geometry conditions during a measurement should be identical to those used during instrument calibration. For portable instruments, they should be as close as practicable to those used during instrument calibration; removable spacers may be used for this purpose.
- f) For methods other than liquid scintillation counting, the sensitive area of the detector shall be larger than the wipe sample.
- g) If a wipe sample with alpha or beta emitters should be measured, alpha-beta pulse discrimination is required.

For the measurement of wipe samples by liquid scintillation counting, the following requirements shall be complied with:

- a) For watery wipe samples, appropriate scintillation cocktails are required.

- b) Tritium wipe samples should be placed into the counting vial already containing an appropriate amount of scintillation cocktail.
- c) Wet tritium wipe samples shall be placed directly in the scintillation cocktail for about 20 min prior to counting to approach equilibrium distribution of the tritium activity and dark adaption.
- d) Care should be taken that the introduction of the wipe sample into the scintillation cocktail does not cause excessive losses of light due to the mass of the material or to quenching effects, and does not introduce spurious scintillations due to brightening agents.

2.3.5 Evaluation of measurement data

Using installed instruments, the activity per unit area of the removable contamination of the wiped surface can be derived from the activity measurement result of the wipe material. According to ISO 7503-2, for installed instruments, the activity per unit area $A_{w,S}$ (Bq/cm²) of the removable contamination of the wiped surface, in relation to the activity A_w (Bq) of the wipe sample, is given by

$$A_{w,S} = \frac{A_w}{S_w \epsilon_w} \quad (20)$$

where S_w (cm²) is the wiped surface area and ϵ_w is the wiping efficiency. Determining the wiping efficiency experimentally involves time and effort. Usually, a conservative value of 0.1 is used which assumes the wipe material only picks up 10 % of the available removable contamination.

Portable instruments calibrated according to ISO 7503-3, Annex D can be used to determine the activity per unit area of the removable contamination of the wiped surface. Thus, we have

$$A_{r,S} = C_r (R_w - B) \quad (21)$$

where $A_{r,S}$ (Bq/cm²) is the activity per unit area of the removable contamination of the wiped surface, C_r (Bq.cm⁻²s) is the activity calibration factor, R_w (s⁻¹) is the observed count rate from the wipe and B (s⁻¹) is the background count rate of the instrument.

The uncertainties in assessing the levels of contamination from wipe tests are great and are affected by many factors including, but not limited to the following: (i) the type of wipe material used; (ii) the pressure applied by the person when making the wipe; (iii) the area wiped; (iv) the contamination distribution; (v) the porosity, chemical composition, texture and cleanliness of the surface.

Wiping efficiency can vary enormously and is almost impossible to assess accurately. The uncertainties in the wiping efficiency are an order of magnitude larger than other uncertainties in the measurement. Consequently, it has been common practice to allocate a value of conservative 10 % to the wiping efficiency, as essentially a “best guess”.

2.3.6 A new promising wiping technique

Strippable gels such as Decon Gel 1101, which are relatively new materials, have a high removal factor (up to 90–100%), even on porous surfaces such as concrete. The narrower range in removal factors should reduce the uncertainty associated with this factor. However, repeated sampling should be performed to verify this as both a quality control (QC) and validation measure. The removal factor for the strippable gel does not seem to depend on the chemical form of the contaminant. This could minimize the uncertainty due to the combination of material and solvent chosen for traditional swipes. Strippable gels can be applied over an area larger than the desired sample area. The exact area needed can be cut from the stripped gel, provided it can be removed as a single large piece. This eliminates uncertainties in the area due to freehand wiping. Pressure applied would not appear to enter as a factor. The gel can be counted directly if it has been previously calibrated for the radionuclide of concern. Variations in thickness may contribute to the uncertainty due to self-absorption. These gels may be rehydrated and the encapsulated radionuclides separated for analysis, for example, by liquid scintillation counting.

This new material has the potential for making wipe measurements a more realistically quantitative determination of removable surface activity with detection limits that may be lower than current practice can achieve (EPA, 2011).

3. Mapping the contamination

3.1 General

Radiological characterization plays an important role in decommissioning of nuclear facilities. It is the basis for planning, identification of the extent and nature of contamination, assessing potential risk impacts, cost estimation, implementation of decommissioning framework, radiation protection, protection of the environment, and management of material arising from decommissioning, as well as supporting decisions for release of buildings and site. At all stages of a decommissioning project, adequate radiological characterization is of crucial importance. A large number of measurement techniques are available for successful application of radiological characterization, allowing rapid and comprehensive determination of the activities of most relevant radionuclides. For other radionuclides that are hard to detect, scaling factors can be established that relate their activities to key nuclides.

3.2 Radiological surveys for decommissioning of nuclear facilities

Radiological surveys deals with measurements of radiation levels and radioactivity associated with a site, area or objects together with appropriate documentation and data evaluation. The strategy of the survey depends on the survey goals. Thus, routine surveys are usually performed

in areas where radiological conditions are well known and generally consistent. Routine surveys can be associated with:

- Measurements within and at the boundary of areas to ensure they are correctly designated;
- Day-to-day duties to prevent the spread of contamination in an area or to personnel.

Non-routine surveys are usually associated with circumstances where the radiological conditions could be unknown, difficult to predict, or highly variable. In the decommissioning of nuclear facilities are performed many types of radiological surveys such as Scoping, Characterization, Remedial Action Support and Final Status Survey. In addition, Clearance Surveys are performed for the demonstration of compliance with clearance levels of materials and equipment resulted from the decommissioning process.

The characterization survey is the most comprehensive of all the survey types and generates the most data. It needs careful planning and a dedicated infrastructure as well as knowledgeable staff for its implementation. A comprehensive radiological characterization program normally comprises the following steps:

- An initiating step where the targets of the campaign are defined and, if necessary, where consent from the competent authority is gained.
- A planning step where historical information from the facility is evaluated and where the strategy and the plan for sampling and measurements are developed.
- An implementation step where sampling and measurements are carried out, if necessary aided by calculation methods, e.g. for determination of activation.
- A step for data assessment and evaluation, in which the various results are interpreted and reviewed, statistical evaluation of measurement results is carried out, etc., and where calculated results and measured data are compared.
- A finalization step where the results are documented and (if necessary) reported to the competent authority and are used for the various purposes and objectives for which they have been derived.

Contamination surveys deal with the measurement of the surface contamination and are usually associated with the radiological characterization of nuclear sites, areas or objects from nuclear sites. Mapping the surface contamination is usually performed by means of contamination surveys measuring the activity per unit area at different locations in a given area. The statistical *sampling design* provides the number of locations and their coordinates.

The contamination surveys may combine a number of direct measurements with scanning to identify areas of elevated activity. The scans must be performed for 100 % of the investigated surface if the probability of areas of elevated activity is high. For direct measurements, the investigated surface may be considered as being overlaid by a grid (rectangular or otherwise),

and measurement locations are at gridline intersections at fixed distance apart in each of the two directions. Both the extent of the grid and the spacing between locations are important. The measurement grid should span the investigated surface so that any part of the surface could become a measurement location. If the goal of the study is to describe spatial correlations, the spacing between measurement locations should be shorter than the range of the correlation. A systematic grid may also include additional points at short distances from some of the grid points that are located into the areas of elevated activity. These points can provide additional information about small-scale spatial correlations.

In case of surface with low potential for areas of elevated activity, the measurement locations can be determined randomly. If there is knowledge of areas with different levels of contamination over the investigated surface, random measurement locations would be selected within each area. The number of measurement locations in each area would be defined on the basis of its relative value.

Each point from the data set obtained by means of the contamination survey contains the spatial coordinates of each measurement location and the measured value of the activity per unit area at that location.

3.3 Mapping the surface contamination

3.3.1 Statistical sampling

Characterization surveys may have multiple objectives that must be clearly specified before the sampling program is designed, because different purposes require different sampling strategies and sampling intensities in order to be efficient, and to permit general inferences.

Statistical sampling is the process of gaining information about a *population* from a portion of that population called a *sample*. For characterization surveys, sampling refers to collection of data taking:

- (1) a discrete number of measurements at different data points;
- (2) a discrete number of laboratory measurements using samples of material.

Statistical sampling is pertinent and necessary in radiological surveys because of the natural stochastic variation that occurs in all environmental media, and the fact that this variation is usually much larger than variations associated with measurement uncertainties. The use of valid statistical - sampling techniques increases the chance that the sample is collected in a way that is representative of the population. If limited in number, samples should contain only typical members, or with adequately large sample numbers, sampling should cover the whole population, including the less-typical members. Representativeness of the sample is difficult to

demonstrate and is usually considered justified or achieved by the procedure used to select the samples (ICRU, 2006).

A five-step protocol for designing a sampling program includes:

- Define the objectives and questions to be answered.
- Summarize the environmental context for the quantities being sampled and measured.
- Identify the target population, including spatial and temporal extent.
- Select an appropriate sampling design.
- Document the sampling design and its rationale.

There are many different sampling designs or schemes, which vary in terms of their practicality and applicability. The main sampling schemes are:

- (i) Simple random sampling;
- (ii) Stratified sampling;
- (iii) Systematic sampling.
- (iv) Judgmental sampling

Simple random sampling involves enumeration of each member in a defined target population and a random selection of members (samples) to be collected for a measurement. For stratified-sampling designs, the population is divided into strata that are more homogeneous than the entire population. Usually, the proportion of sample observations in each stratum is similar to the stratum proportion in the population, but this is not a requirement. Systematic sampling usually involves sampling at regular spatial and/or temporal intervals along a line or at nodes of a defined grid and is probably the most commonly used method for field sampling. It is generally unbiased as long as the starting point is randomly selected and the systematic rules are followed with care. A systematic sample, using either a triangular or square grid, is the preferred sampling design to detect a single hot spot. The probability of detection depends on the size and shape of the hot spot and the spacing between locations on the grid.

Judgmental sampling is based purely on judgment. Judgmental sampling can be problematic in that the sample may not be unbiased, precision cannot be quantified, and representativeness is unknown. However, judgmental sampling, if applied with expert knowledge, can have important benefits.

Other sampling schemes include two-stage sampling, in which primary sampling units are selected randomly, then the primary unit is sub-divided and the sub-units might be selected randomly. Cluster sampling is frequently applied in situations in which members of the population are found in clusters. Adaptive sampling is most commonly used when the sampling units aggregate or cluster together. Double sampling is used in a situation in which one

characteristic may be difficult or expensive to measure but another related characteristic is simple or easy to measure.

Relatively simple and straightforward formulas for estimating population averages, totals, proportions, percentiles, and ratios are available for data obtained from nearly all the sampling schemes described above. Likewise, statistical formulas for estimating the variance associated with such numerical estimates are readily available.

3.3.2 Spatial sampling

The classical sampling designs and analyses described above can be used in a spatial context. When it is necessary to sample a large area with closely spaced sampling locations, a systematic grid can require a very large number of sampling locations. Transect sampling can provide almost as much information with many fewer sampling locations. A transect sample is a random or, more commonly, a systematic sample along a line. Customarily the lines are straight lines.

Spatial sampling deals with random spatially correlated processes defined in a spatial domain. The process specifies values of an attribute across the spatial domain. The locations at which the attribute of interest is measured are defined by a series of spatial coordinates. Consider a spatial random process, $Z(u)$, defined in a two-dimensional domain. Here, u is a two-dimensional coordinate vector of a point in the spatial domain. Realizations of the process, $z(u_i)$, are observed at sampling points u_i , where $i = 1, \dots, n$.

The values of the attribute of interest of the spatial random process at two locations are not generally independent. Values at two places nearby are likely to be more similar than values at two places further apart. Thus one can define the spatial covariance Cov of the random process in general as a function of lag distance (h) separating the sampling locations, by (ICRU, 2006)

$$Cov(z(u), z(u+h))=2C(h) \tag{22}$$

where $C(h)$ is the spatial covariance between points separated by the lag distance h . Generally, the lag is a vector quantity and the covariance might depend both on length and direction. However, in many practical situations the covariance function does not display dependence on direction and the lag can be considered as a scalar value.

The spatial correlation, $\rho(h)$, between attributes measured at two locations separated by the distance h can be written in terms of the spatial covariance, $C(h)$, or the semivariogram, $\gamma(h)$.

$$\rho(h)=C(h)/C(0)=1-\gamma(h)/\gamma(\infty) \tag{23}$$

where $C(0)=\gamma(\infty)$ is the variance of the attribute over the spatial region. It is the semi-variogram of pairs of locations separated by sufficiently large a distance to eliminate spatial correlation. The spatial correlation defined by the semivariogram is preferred because it can be estimated

more accurately and is slightly more general mathematically. The most commonly used of several variogram estimators is the classical estimator

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} [z(u_i + h) - z(u_i)]^2 \quad (24)$$

Summation in Eq. (24) is done over all points separated by the lag vector h . The distance values are grouped into arbitrary defined bins, therefore Eq. (24) represents a variogram in binned form. An alternative is to work directly with variogram cloud, i.e., with the cloud of points

$$\gamma(h_{ij}) = \gamma(u_i - u_j) = 0.5 \times [z(u_i) - z(u_j)]^2 \quad \text{for } i, j = 1, \dots, n. \quad (25)$$

Three quantities from the variogram are commonly used to summarize the magnitude and extent of spatial correlation. They are illustrated by Figure 2. The nugget is the limiting value of the semivariance as the distance approaches zero. The nugget captures spatial variability at very small spatial scales (those less than the separation between observations) and also measurement error. The sill is the horizontal asymptote of the variogram, if it exists, and represents the overall variance of the random process. The range is the lag value at which the semi-variance value reaches the sill.

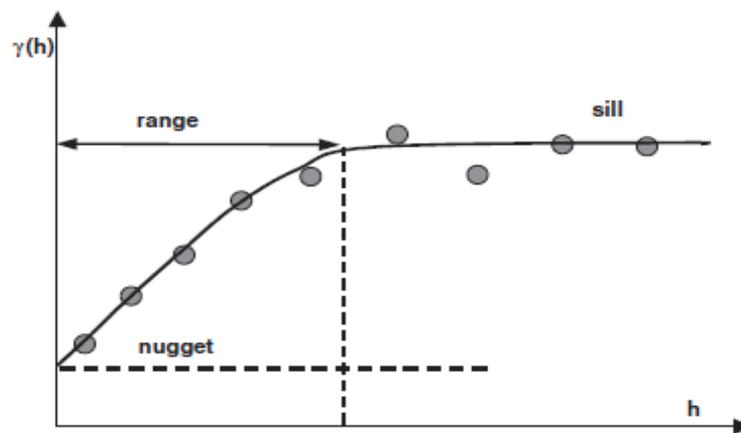


Figure 2. Parameters of a variogram: the range, the sill, and the nugget.

With input data and the spatial structure identified through the variogram, geostatistical techniques estimate the studied variable by a method similar to regression analysis called kriging. This always includes a quantification of the associated uncertainty.

3.3.3 Mapping the surface contamination

A key goal of statistical *sampling design* is to specify the sample size (number of samples) and sampling locations that will provide reliable information for a specific objective. The number of samples required depends on the quantity to be estimated (e.g., a mean, a variance, or something else), on the required precision, and on the natural variation inherent in the population being sampled. The sampling locations depend on both the sampling design and the number of

samples. Specific guidance on the use of statistics for sampling design can be found in the bibliography (ICRU, 2006). Software tools for sampling design are also available (Matzke et al., 2014).

The sampling design based on geostatistics make use of the variogram to design a sampling strategy. The variogram provides information about the spatial correlation between locations. This information can be used in two different ways, depending on the goal of the sampling strategy. If the sampling goal is to produce a map, or to estimate the proportion of an area where the attribute of interest exceeds some critical value, it is important to detect small-scale fluctuations in the quantity of interest. Prediction of the values at individual locations is most precise when those locations are highly correlated with observed locations. This can be achieved by ensuring that no place on the map is too far from an observed location. One design to achieve this is a systematic sample with a grid spacing less than the range of the variogram. A rectangular grid is slightly inferior to a triangular grid, but the rectangular grid is more practical.

Contamination mapping is the process of recording the level of contamination at numerous points throughout an area and displaying the data in terms of position and activity per unit area in a graphical form. The contamination map provides the spatial structure (pattern) of the contamination and is provided with a coordinate-based local system to uniquely determine the position of all measuring locations. Presenting the collected data as part of a map generally consists overlaying a colour scaled intensity representation onto a two dimensional map/spatial representation (drawing, picture) of the mapped area. In Figure 3, a simple and illustrative example of contamination map is shown.

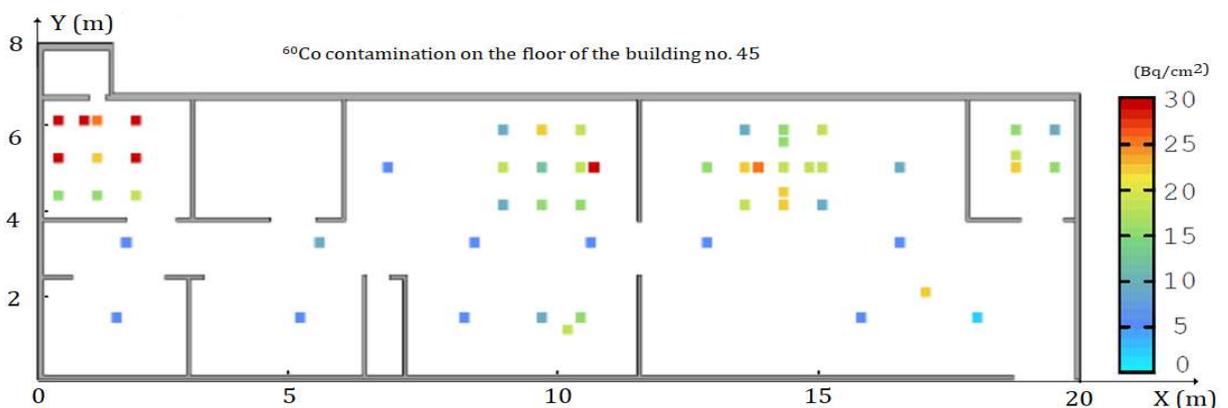


Figure 3. A simple and illustrative example of contamination map

There are many methods for spatial interpolation and prediction, given a set of spatially registered measurement data. A widely used class of models includes polygon tessellation and triangulation based on a spatial sample. Others include global and local polynomial fitting, natural neighbour and inverse distance weighting interpolation. One of the more sophisticated

methods of interpolation is called kriging. Kriging makes the assumption that points closer to each other are more similar than point far away from each other. Instead of using a deterministic model for interpolation, kriging uses a stochastic model based on the dataset itself. There are several types of kriging techniques. The more general and the most common type is called ordinary kriging. It assumes an unknown but constant mean of the dataset. The interpolation for a single point is given by the kriging prediction equation

$$\bar{z}(x_0) = \sum_{i=1}^n \lambda_i z(x_i) \tag{26}$$

where $\bar{z}(x_0)$ is the estimated value at point x_0 , λ_i is the weight of point x_i with respect to x_0 and $z(x_i)$ is the value at point x_i . The sum of the weights is always equal to one. For a data set, we have

$$A = \begin{bmatrix} \gamma(x_1, x_1) & \gamma(x_1, x_2) & \dots & \gamma(x_1, x_n) & 1 \\ \gamma(x_2, x_1) & \gamma(x_2, x_2) & \dots & \gamma(x_2, x_n) & 1 \\ \dots & \dots & \dots & \dots & \dots \\ \gamma(x_n, x_1) & \gamma(x_n, x_2) & \dots & \gamma(x_n, x_n) & 1 \\ 1 & 1 & 1 & 1 & 0 \end{bmatrix}, \quad b = \begin{bmatrix} \gamma(x_1, x_0) \\ \gamma(x_2, x_0) \\ \dots \\ \gamma(x_n, x_0) \\ 1 \end{bmatrix}, \quad \lambda = \begin{bmatrix} \lambda_1 \\ \lambda_2 \\ \dots \\ \lambda_n \\ \mu \end{bmatrix}$$

where A is the semivariances of all the point-pairs, b is the semivariance of all the sampled points from the estimation point x_0 and λ is the weights in the kriging equation. The following equations in matrix form can be set up.

$$A\lambda = b \quad \text{and} \quad \lambda = A^{-1}b \tag{27}$$

where b can be obtained using a variogram. With the semivariance and weights known, the kriging variance can be calculated using

$$\sigma^2(z_0) = 2 \sum_{i=1}^n \lambda_i \gamma(z_0 - z_i) - \sum_{i=1}^n \sum_{j=1}^n \lambda_i \lambda_j \gamma(z_i - z_j) \tag{28}$$

To construct a map, the kriging prediction equation is used to predict values for a finely spaced grid of observations. These predicted values are then used to construct the map, using contour lines, shading, or colouring as appropriate.

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