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National and Regional Surveys of Radon Concentration in Dwellings

Review of Methodology and Measurement Techniques

IAEA

International Atomic Energy Agency

NATIONAL AND REGIONAL SURVEYS OF RADON CONCENTRATION IN DWELLINGS

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INTERNATIONAL ATOMIC ENERGY AGENCY
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FOREWORD

Reliable, comparable and 'fit for purpose' results are essential requirements for any decision based on analytical measurements. For the analyst, the availability of tested and validated sampling and analytical procedures is an extremely important tool for carrying out such measurements. For maximum utility, such procedures should be comprehensive, clearly formulated and readily available to both the analyst and the customer for reference. In the specific case of radon surveys, it is very important to design a survey in such a way as to obtain results that can reasonably be considered representative of a population.

Since 2004, the Environment Programme of the IAEA has included activities aimed at the development of a set of procedures for the measurement of radionuclides in terrestrial environmental samples. The development of radon measurement procedures for national and regional surveys started with the collection and review of more than 160 relevant scientific papers. On the basis of this review, this publication summarizes the methodology and the measurement techniques suitable for a population representative national or regional survey on radon concentration in the indoor air of dwellings. The main elements of the survey design are described and discussed, such as the sampling scheme, the protocols, the questionnaire and the data analysis, with particular attention to the potential biases that can affect the representativeness of the results. Moreover, the main measurement techniques suitable for national surveys on indoor radon are reviewed, with particular attention to the elements that can affect the precision and accuracy of the results.

The IAEA wishes to thank all the participants for their valuable contributions to developing the procedure and reviewing the report, especially F. Bochicchio from the Italian National Institute of Health. The IAEA officer responsible for this publication was A. Ceccatelli of the IAEA Environment Laboratories.

CONTENTS

1. INTRODUCTION AND RATIONALE

Prolonged exposure for several years to radon and its decay products in homes has been recently proven – through many epidemiological studies – to increase the risk of lung cancer [1−5]. However, indoor radon concentrations can be reduced. Therefore international and national regulations/recommendations, as well as comprehensive national radon programmes have been set up – and recently updated on the basis of epidemiological results – with the aim to reduce population exposure to indoor radon and the corresponding risks [6−12]. Such regulations/recommendations or programmes include some *reference levels* or *action levels*[*], which are levels above which some action to reduce radon concentration is recommended or required. In order to optimize the choice of a national reference level, it is important to know the impact of different possible values of reference level, i.e. the estimated number of dwellings with indoor radon concentration above such levels.

Therefore, an important component of any national radon programme is the evaluation of the frequency distribution of radon concentration levels in dwellings. Such evaluation should be obtained with nationwide radon surveys designed and conducted in such a way as to be representative of population exposure in homes. A population representative nationwide survey is actually one of the first stages of a national radon programme aimed to reduce risks from radon exposure, as also recently recommended by WHO [7].

Although the dose to the lungs arising from radon exposure in homes is due mainly to radon decay products rather than radon gas itself, radon concentration is more related to dose than to radon decay product concentration, and radon concentration measurements are appropriate to evaluate the lung cancer risk. In fact:

- The dose to the lungs depends not only on the concentration of radon decay products, but especially on the size distribution of the aerosols to which radon decay products attach themselves, as well as on the fraction of radon decay products which are not attached to aerosols;

- Adequate and convenient techniques for measuring concentration of radon decay products – and especially aerosol size distribution – are not well established yet (although some interesting new methods for determination of long-term radon decay product concentration have been recently developed [14]), whereas simple and inexpensive techniques for measuring long-term radon concentration are available for many years;

- Model calculations show that the dose is roughly proportional to radon concentration level for the typical range of aerosol concentration that can be found in a dwelling [15, 16].

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[*] In the most recent general recommendations of the International Commission of Radiological Protection [13], the concept of *action level* has been replaced with the concept of *reference level*. The reference level is the level of dose or risk above which it is judged to be inappropriate to allow exposures to occur, and below which optimization of protection should be implemented, whereas with the previous concept of action level, actions had to be considered only in case of levels higher than action level.

Therefore, for the purposes of a survey aiming to evaluate the risk, it is adequate to measure long-term radon concentration.

1.1. PURPOSES AND CONTENT OF THIS REPORT

The purpose of this report is to review methodologies and techniques to design and carry out a nationwide survey aimed to estimate the distribution of annual average radon concentrations in dwellings that can be considered representative for population exposure.

The intent is to provide a quite comprehensive review of the main relevant methodologies and techniques, in a concise way and at a technical level that should allow a wide audience. Anyway, an extensive bibliography is also provided for expanding on the issues partly covered in this report.

The main elements of the survey design are described and discussed, such as the sampling scheme, protocols and questionnaire, and data analysis, with particular attention to the potential biases that can affect the results.

It is worth noting that methods presented here can be applied also to large surveys other than a nationwide one, such as surveys carried out in one or more regions within a country. Regional surveys have been used to conduct a nationwide survey in steps, each one consisting of a regional survey, thus making a large nationwide survey more feasible. Moreover, regional surveys are suitable for federal countries.

Geographical or mapping surveys (i.e. surveys aimed to evaluate in detail the geographical distribution of radon concentration in dwellings) generally represent a following stage of a radon programme, requiring a quite higher number of sampled dwellings. These surveys are aimed to optimize the process of finding those dwellings with a high radon concentration, whose total number has been estimated with representative nationwide surveys. Even if geographical surveys are not covered by this report, several methodological issues of population representative nationwide (or regional) surveys are however useful for geographical surveys as well. An appropriately designed geographically-based survey could also give adequate information on the population representative nationwide distribution of indoor radon, as well as a properly designed representative nationwide survey can also produce some useful information on the geographical distribution of radon concentration.

A review of techniques suitable for long-term measurements of radon concentration used in nationwide and regional radon surveys is also included in this report, with particular attention to the elements that can affect the accuracy of the measurement results. Measurement techniques for radon decay products and for thoron and its decay products are not covered as they are outside the scope of this report. However, the possible impact of the presence of these radionuclides on the radon concentration measurements is discussed.

2. RADON: CHARACTERISTICS, SOURCES, INDOOR CONCENTRATIONS AND THEIR VARIATIONS

2.1. PHYSICAL CHARACTERISTICS AND SOURCES

Radon is a radioactive noble gas, and it was acknowledged in 1900 by Friedrich Ernst Dorn, who called it radium emanation. The main isotopes of radon are ^{222}Rn (usually and hereafter

referred to as *radon*, which belongs to the radioactive decay series starting with ^{238}U and ending with stable ^{206}Pb), ^{220}Rn (usually and hereafter referred to as *thoron*, which belongs to the radioactive decay series starting from ^{232}Th and ending with stable ^{208}Pb), and ^{219}Rn (usually and hereafter referred to as *actinon*, which belongs to the radioactive decay series starting from ^{235}U and ending with stable ^{207}Pb). Series of ^{238}U and ^{232}Th are shown in Figure 1a and Figure 1b, respectively. Radon is the only gaseous element of these radioactive decay series. The indoor concentration of radon, due to its half-life quite longer than that of the other two isotopes (3.8 days, compared with 55.8 s and 3.96 s for thoron and actinon, respectively), is generally the highest one, followed by that of thoron, whereas actinon is not found in normal environments. The main source of indoor radon is soil and rocks underlying the building, especially for high levels of indoor radon concentration, with a minor contribution from building materials and well water, contributions which can be relatively more important in case of low indoor radon levels [17-20]. However, since the 1980s some building materials have been recognized as possible sources of high radon concentrations, such as building materials containing by-product gypsum [20, 21], concrete containing alum shale [22], and bricks made with soil and rocks with high levels of natural radioactivity as volcanic tuffs and pozzolana [23]. The main sources of thoron are generally the building materials of dwelling walls [21, 24]. The radon concentration in a room is quite homogeneous [25], whereas thoron concentration strongly decreases with the distance from walls [26]. Many data and references on sources and levels of radon and thoron can be found in UNSCEAR and other reports [20, 21, 27-30]. Some physical characteristics of radon, thoron and their decay products, including those of the emitted alpha particles, are reported in Table 1.

FIG. 1a. Decay scheme of the ^{238}U series (adapted from [18]). Under the symbol of each radionuclide the half-life value is reported, taken from www.nucleide.org/DDEP_WG/DDEPdata.htm (except half-lives of ^{218}Po, ^{214}Bi and ^{214}Pb, which are taken from [31] and [32]). The historical name of the short lived radon decay products is also reported, above the nuclide symbol.
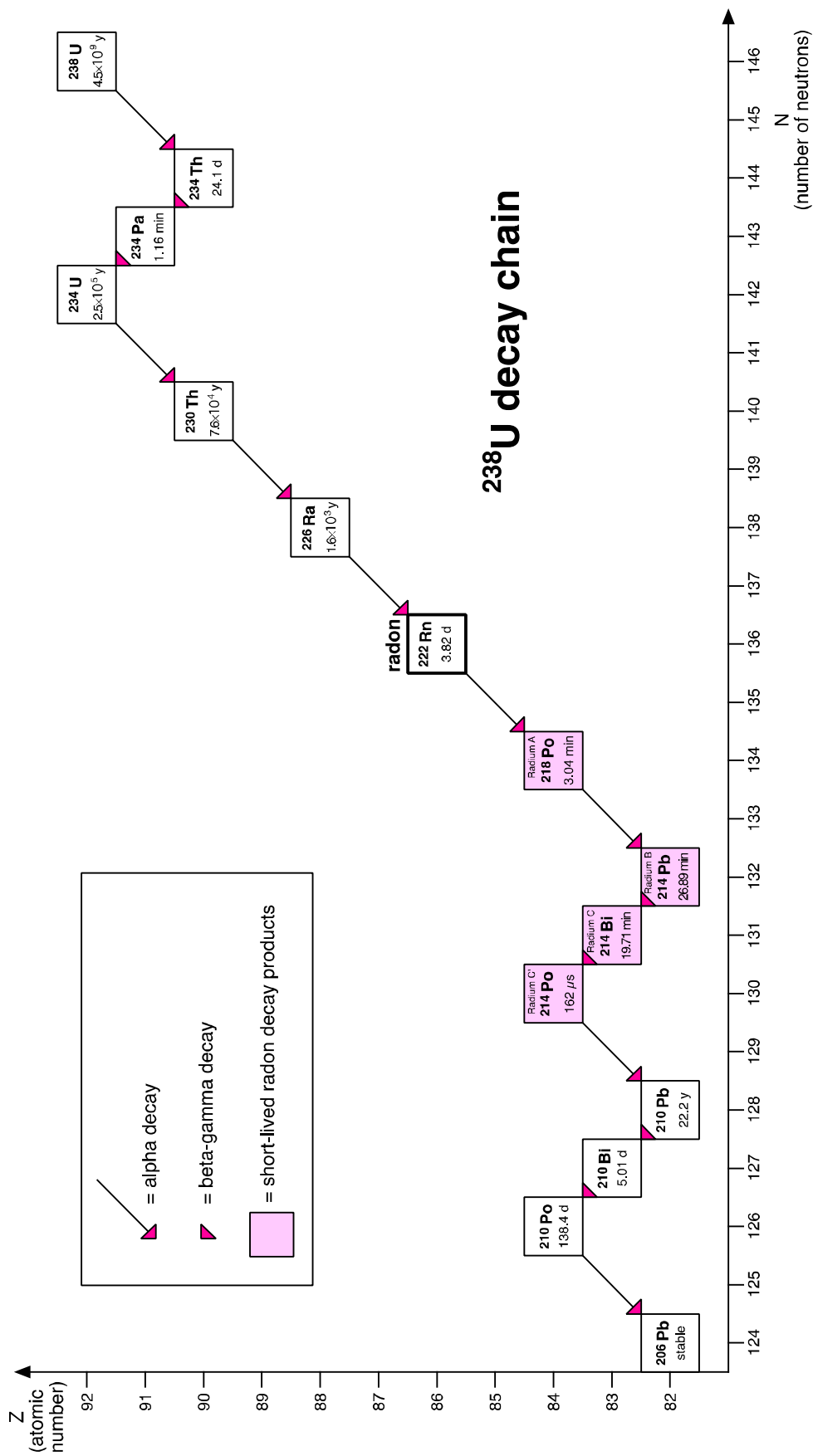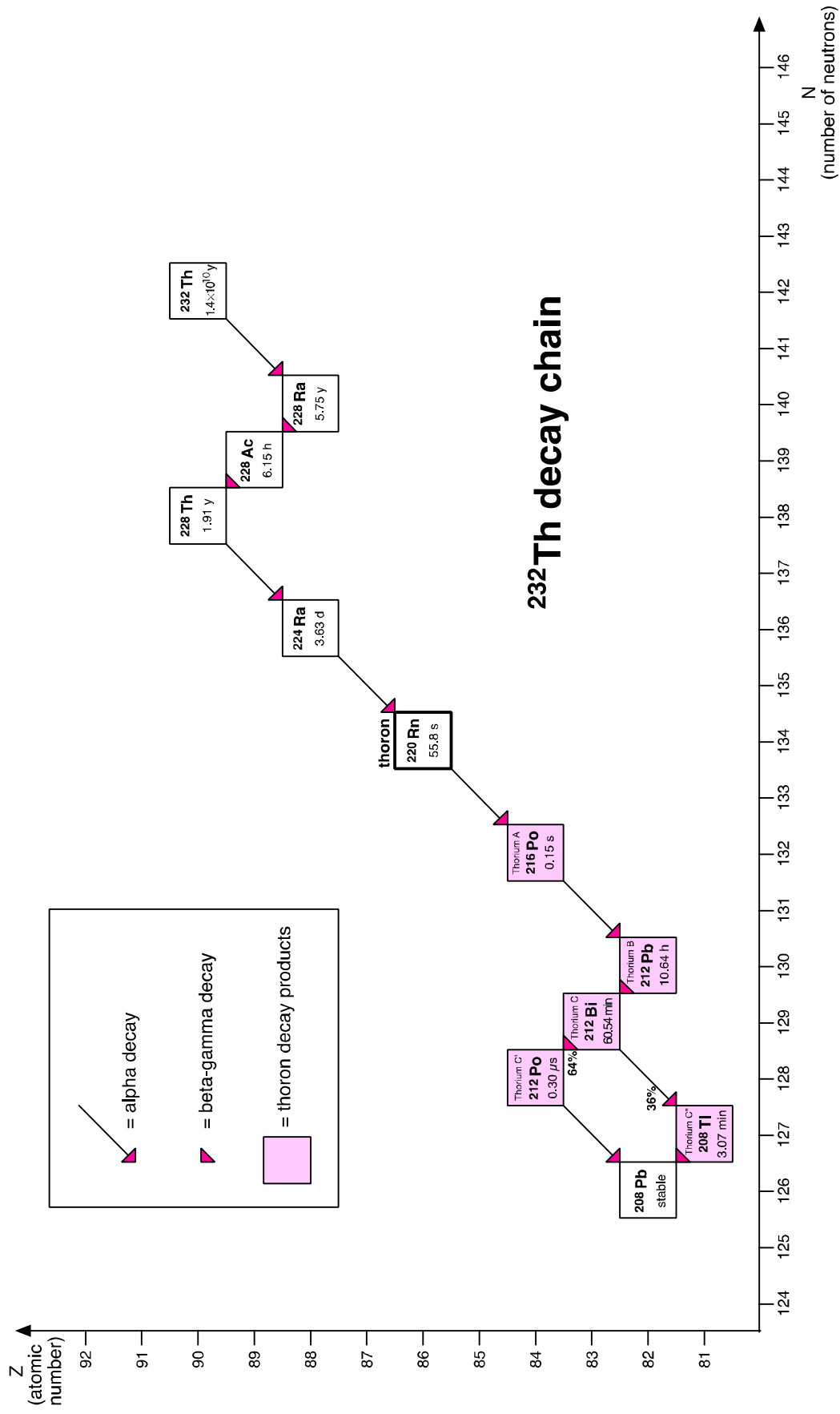
4

232Th decay chain

FIG. 1b. *Decay scheme of the* ^{238}U *series (adapted from [18]). Under the symbol of each radionuclide the half-life value is reported, taken from www.nucleide.org/DDEP_WG/DDEPdata.htm. The historical name of the thoron decay products is also reported, above the nuclide symbol.*

TABLE 1. SOME PHYSICAL CHARACTERISTICS OF RADON, THORON AND THEIR (SHORT LIVED) DECAY PRODUCTS, AND OF THE MAIN EMITTED ALPHA PARTICLES.

Radio-nuclide	Half-life	Decay Mode	Principal alpha energy					
			E_α (MeV)	Intensity	Range in air (cm)	Range in water (μm)	Range in CR-39 (μm)	Range in Makrofol (μm)
^{222}Rn	3.8232 d	A	5.49	99.9 %	4.0	39	35.5	38.7
^{218}Po	3.04 min	A	6.00	100 %	4.6	44	40.7	44.5
^{214}Pb	26.89 min	β, γ						
^{214}Bi	19.71 min	β, γ						
^{214}Po	162.3 μs	A	7.69	100 %	6.9	66	59.8	65.6
^{210}Pb	22.23 y	B						
^{210}Bi	5.012 d	B						
^{210}Po	138.4 d	A	5.30	100 %				
^{206}Pb	Stable							
^{220}Rn	55.8 s	A	6.29	99.9 %	4.9	48	43.6	47.7
^{216}Po	0.15 s	A	6.78	100 %	5.6	54	48.9	53.6
^{212}Pb	10.64 h	β, γ						
^{212}Bi	60.54 min	α (36 %) β (64 %)	6.05		4.7	45		
^{212}Po	0.30 μs	A	8.79	100 %	8.5	83	75.8	83.3
^{208}Tl	3.07 min	β, γ						
^{208}Pb	Stable							

2.2. CONCENTRATION IN INDOOR AIR AND ITS SPATIAL DISTRIBUTION AND TEMPORAL VARIATIONS

Radon concentration in indoor air of dwellings can range from a few Bq/m^3 to several thousands of Bq/m^3 [20]. This large spatial variation of concentration values is due to the combined effects of: (i) a large variation of radon exhalation fluxes from its sources, regarding both the exhalation from soil underlying the building (which is generally the main source of indoor radon) and the exhalation fluxes of building materials; (ii) a large variation of building characteristics affecting radon entry and ventilation.

Radon concentration in indoor air is not constant but varies strongly with time. It is generally higher during the night and early morning than during daylight hours, and higher in winter than in summer. The latter ones, generally called *seasonal variations*, have been extensively studied [e.g. 33, 34, 62], also in order to derive *seasonal correction factors* to apply to measurements carried out few months in a given season, thus obtaining an estimate of the annual average. However, besides these typical or average trends, large variability occurs due to variations of weather, occupancy habits and other factors. Generally, the shorter is the period, the larger is the radon concentration variability, e.g. variability for weekly and monthly concentrations is larger than that for 3-month concentrations [35−37]. Therefore long term measurements (i.e. measurements with a sampling period from a few months to about 1

year) are less sensitive to such temporal variations, compared with short-term measurements (i.e. measurements with a sampling period up to few weeks). Some year-to-year variations occur also for annual averages [38−41]. The annual average, i.e. an average over a total period of 12 consecutive months, is generally considered as the optimal estimate of the time averaged radon concentration [7], although shorter exposure periods (typically 3 months) are widely used, often with results corrected by means of seasonal correction factors.

3. DESIGN OF A POPULATION REPRESENTATIVE NATIONWIDE (OR REGIONAL) RADON SURVEY

3.1. INTRODUCTION ON RADON SURVEYS AND REPRESENTATIVENESS

The annual average of radon concentration in a specific dwelling cannot be estimated in a reliable way and a measurement is necessary to obtain it. Moreover, although soil is the main radon source, the distribution of indoor radon concentrations in a region or in a whole country cannot be reliably estimated on the basis of information on soil characteristics only, because both the soil characteristics affecting radon exhalation and the impact of the building characteristics are not generally known. In fact, it is not uncommon that no or poor correlation is observed between soil characteristics and average indoor radon concentration [42]. Therefore, surveys to directly measure radon concentration in a significant and representative sample of dwellings are generally used to estimate the distribution of indoor radon concentration. Such a population representative nationwide survey is one of the first steps for a national radon programme. Several collections and reviews of radon survey results are available [7, 20, 29, 43−45].

Some reports collecting and reviewing the main results of radon surveys include not only surveys carried out in a representative sample of the population but also non representative surveys. Unfortunately, adequate information to describe and qualify survey representativeness is often not reported in these reports. Examples of non-representative surveys are those carried out in some radon-prone areas, which clearly produce an overestimate of the national radon concentrations. Non random samples can also produce some unintended biases: for example the use of volunteers could result in an overestimate of radon concentration levels because people suspecting that their home could be affected by high radon concentration are more prone to be volunteers [46, 47]. Similar biases can result by using radon concentration measurement data collected in databases without paying attention to their sampling scheme and representativeness characteristics [6, 48]. Therefore, it is very important to carefully design a survey in such a way as to obtain results that can reasonably be considered representative for population exposure.

Adequate information on population representative of national distribution of radon concentration levels in a country can be obtained with a survey of moderate size, i.e. several hundreds to a few thousands of sampled dwellings. Of course, such a survey cannot give detailed information on the geographical distribution of indoor radon concentration over the whole territory, which would require a much higher number of sampled dwellings. A detailed knowledge of the geographical distribution of indoor radon concentration will allow the identification of radon-prone areas, i.e. areas where the distribution of indoor radon levels is significantly shifted towards levels higher than those of the national distribution. In recent years many surveys aimed to geographically describe indoor radon distribution have been carried out, dividing the territory into geographical units (e.g. by a regular grid of rectangular

areas or following the administrative boundaries), and several specific and complex statistical methods have been applied to obtain detailed radon maps [49]. However, it is important to underline that also for these geographically based radon surveys (as well as for the nationwide population representative survey) it is important that the sampled dwellings in each geographical unit are selected in such a way as to be representative for the population of that geographical unit. Moreover, considering the relatively low number of sampled dwellings for each geographical unit, the possible bias deriving from a non-random selection of the sampled population could have a high impact on the results and their representativeness [6, 7].

In this section the main elements for the design of a nationwide survey – aimed to obtain a population representative estimate of the distribution of radon concentration levels in dwellings – are briefly reviewed and discussed on the basis of knowledge derived from many surveys. More details can be found in the papers regarding methodology of specific surveys and in some reports [7, 50–59].

3.2. SAMPLING SCHEME

Dwellings and householders. The radon concentration in a dwelling is more related to dwelling characteristics rather than householders' ones, and it can reasonably be assumed that all the householders are exposed to the same radon concentration. Therefore it is equivalent to consider the housing units or the householders as the target population of the survey [50]. The choice depends on the availability of complete lists from which the sample units could be randomly selected. It is often easier to have a complete list of inhabitants rather than of dwellings.

Randomness and completeness. In principle, in order to obtain a sample that is representative of the population exposure, it is sufficient to select a *random* sample of dwellings (or of inhabitants) from a *complete* list of all the dwellings (or inhabitants) in a given country. This is called a *simple* random sampling. Randomness of dwelling (inhabitant) selection and completeness of dwelling (person) list are the key elements to select a representative sample. Deviations from a pure random sample can produce biases, as in case of a sample of volunteers [46, 47], or in case of large numbers of refusals to participate the survey (see Sections 3.3 and 3.4). For practical reasons a surrogate of a complete list of dwellings (or inhabitants) could be used (e.g. the list of telephone numbers, or the list of dwellings of workers of a national public or private company distributed all over the country [60]). In such cases a careful evaluation of such surrogates is needed because deviations from the completeness of the list of dwellings (or of persons) from which the selection is carried out can result in biases [61]. The sample size is also important for representativeness: in fact, a sample of adequate size would reduce the probability that, by chance, the selected sample could contain a non-representative selection of dwellings (persons).

Simple and stratified sample. A simple random sampling is in principle the easiest way to obtain a representative sample. However, a rather complete list of all the dwellings in a country may not be easily available. Moreover, a simple random sample would imply that the selected dwellings/persons are probably located in many different towns. For example, a random sample of 1000 persons in a country with 30 million inhabitants living in 5000 towns would probably be distributed in many hundreds of towns, each of which would include a few selected persons. In such cases the selected persons could be effectively contacted by mail [51], whereas a direct contact with trained personnel probably would not be feasible or convenient. The target population (in our case all the dwellings or all the inhabitants) can also be partitioned into separated groups or *strata (*e.g. the strata could be all the states or the

administrative regions of a country), sampling within each stratum and allocating the sample to the strata proportionally to the strata size, in order to control the distribution of sampled units. Moreover, a multi stage sampling is often used (e.g. sampling of towns in the first stage, and sampling of families/dwellings within the sampled towns in the second stage), especially when a door-to-door contact with householders is chosen [50, 56, 57].

Sample size. The required sample size depends on several factors, such as the precision target of the specific objectives (e.g. the precision target of the national average), the number of strata, the country population, and other factors [50, 58]. In any case, the sample size used for nationwide surveys ranges typically from a few to several thousands of dwellings (see Table 2). For example, about 2000 in the UK [51], about 3500 in Australia [52], and about 5000–6000 in Italy and the USA [56, 57, 62], although some smaller studies were also carried out, as in Croatia, with about 800 dwellings [63].

Monitored rooms. In order to obtain results representative of population exposure, radon detectors should be collocated in inhabited rooms, such as living rooms and bedrooms. If the dwelling has more than one floor level, at least one room per floor level should be monitored in order to obtain information on radon concentration variation between floors. However, if the lowest floor level is a basement with a very low occupancy, basement radon measurement results should not be used to estimate population exposure, because radon concentration in basements can be quite different (generally higher) than in the rest of the house [56]. If the dwelling has one floor level only, one or two rooms are usually monitored, the latter case allowing information to be obtained on radon concentration variation between rooms. If two rooms are measured, usually one living room and one bedroom are selected in order to obtain measurements representative of the whole period spent at home. Measuring two rooms per dwelling will also reduce the impact of detector losses.

TABLE 2. EXAMPLES OF INFORMATIVE SUMMARIZED DATA ON CHARACTERISTICS AND RESULTS FOR SOME NATIONWIDE RADON SURVEYS

Country	Sampling	No. of dwellings	Monitored rooms per dwelling	Exposure time length	Detector type	Device configuration (Closed or Open)	AM (Bq/m³)	GM (Bq/m³)	GSD	%>200 Bq/m³	%>400 Bq/m³	Reference
Australia	Random (stratified)	3413	1	12 mo.	CR-39	Closed	11	8	2.1	<0.1%	<0.1%	[52]
Italy	Random (stratified, two-stages)	5631	1	6+6 mo.	LR 115	Closed	70	52	2.1	4.1%	0.9%	[57, 62]
Republic of Korea	Random (stratified)	2190	1	3+3+3+3 mo.	CR-39	Closed	53	43	1.8	1.7%	–	[59]
UK	Random (simple)	2093	2	6+6 mo.	CR-39	Closed	20	15	2.2	0.5%	0.2%	[51]
USA	Random (stratified, three stages)	5694	≥2	12 mo.	CR-39	Closed	46	25	3.1	~3.5%	~0.6%	[56]

Point of measurement within the room. Radon atoms are quite homogeneously distributed in a room [25]. Therefore the position of radon detectors can be chosen in many convenient ways, e.g. over a wardrobe or a bookcase, provided that it is exposed to indoor air. However, due to the heat effects on many detectors [64], the point of measurement should not be close to heat sources. Some attention to the detector position should be paid also in case the device to measure radon concentration has a non-negligible sensitivity to thoron. In fact some devices show sensitivity to thoron that is not much lower than that to radon (see Table 3 and [73]). In such cases, it is important that the position of the radon measuring device in the room is not too close to the wall, in order to reduce the impact of possible high thoron concentration on the radon measurements [59].

TABLE 3. SENSITIVITY TO RADON, THORON, AND RELATIVE SENSITIVITY (I.E. SENSITIVITY TO THORON OVER SENSITIVITY TO RADON) OF SOME PASSIVE RADON MEASURING DEVICES

Measuring device	Sensitivity (tracks/cm^2 per kBq/m^3×h)		Relative sensitivity	Reference
	to radon	to thoron		
KfK device	0.9	0.7	0.78	[65]
Radtrak	2.8	1.9	0.68	[65]
NRPB/SSI	2.2	0.1	0.05	[66]
Radopot[*]	2.2	0.1	0.05	[67]
Radopot[†]	2.2	1.3	0.59	[68]
Raduet[*]	2.3	0.04	0.02	[69]
Raduet[†]	2.1	1.9	0.90	[69]
Winnipeg device	1.2	0.024	0.02	[70]
ENEA/ISS device	0.8	0.003	0.004	[71]
ENEA	4.7	0.16	0.03	[72]

[*]Thicker diffusion barrier (low air-exchange rate).

[†]Thinner diffusion barrier (high air-exchange rate).

Measurement period. Considering the time variations of radon concentration, the duration of radon concentration measurements should be preferably of 12 months in total in order to obtain an unbiased estimate of the annual average. A single twelve-month period of exposure can be used [52, 56], but shorter consecutive periods have also been adopted, e.g. two consecutive six-month periods [51, 54, 57], or four consecutive three month periods [59]. Radon detectors used for shorter periods of exposure are less sensitive to the effects of fading and ageing (see section 4.2 for more details).

Seasonal variations. The use of consecutive periods of six or three months is more expensive and labor consuming compared with the use of a single 12-month period, but it can enable an estimate of average seasonal variations [51, 59, 62, 74−76]. This would allow the use of

shorter periods (3 or 6 months) for routine measurements, whose results can be used to estimate the annual average by means of seasonal correction factors, evaluated on the basis of the observed average seasonal variations. Seasonal correction factors can be expressed as a function of measurement period and starting month [74, 77], or of external temperature [78]. However, it has to be pointed out that, for any given dwelling, the annual average radon concentration estimated with a three month measurement and a seasonal correction factor has a higher uncertainty compared with results obtained with a 12-month measurement, because seasonal variation can be different for different dwellings and the correction factors are based on average seasonal variations [62, 79, 80]. Moreover, some reverse seasonal variation can occur [62, 81]. Finally, seasonal variations can be quite different for different countries, as well as for different regions within a country (e.g. colder ones compared with warmer ones). Therefore, seasonal correction factors should be evaluated in the region where they will be applied, avoiding to use values obtained in different regions or countries [34].

3.3. OPERATIVE ISSUES, PROTOCOLS AND QUESTIONNAIRE

Protocols for contacting householders. Protocols to effectively contact householders are needed to be developed aiming to obtain their acceptance. In case of a refuse, some basic information should be collected to verify if refuses belong to specific categories of dwellings or inhabitants that are correlated with radon levels, which could result in biased results of the survey; however, for practical reasons, this verification is generally conducted on the sample that accepted to participate the survey, see below. Such protocols should take into account specific national characteristics. If the contact with householders proceeds by mail, sending repeated letters increases acceptance [51], whereas if the contact occurs through telephone or door-to-door it is important to try the contact also in late afternoon and evening and on week-ends, in order to succeed to contact householders working all the day out of home. In case of a single 12-month period of radon detector exposure, a periodical contact with householders (e.g. by phone) could help to identify detector losses, allowing replacement of detectors [56]. This is particular important in case of using a single detector per dwelling. Despite any reasonable efforts, the percentage of refusals can be quite high, even greater than 50%. Therefore it is important to compare the main characteristics of the final selected sample with the corresponding ones of the general population, in order to check for the representativeness of the sample.

Protocols for reducing unintended detector exposures. Alpha track detectors used in radon measuring devices start to be sensitive to alpha particles emitted from the decay of radon and its decay products just after their production and end their sensitivity only when etched. Therefore, a careful protocol should be set up in order to avoid or reduce unintended detector exposures, both before and after the intended exposure period, especially if the exposure period is relatively short [82, 83]. After preparation of radon measuring devices, they are generally put in a heat-sealed radon-proof bag (or, for a better protection, in two radon-proof bags, one within the other one, because sealed bags leak in some cases [84]) to keep them protected till the moment they are collocated in the dwelling. After the exposure radon devices should be returned to the laboratory as soon as possible in order to dismount the detectors and etch them, making the tracks stable and suitable for counting. However, especially in nationwide surveys, the time needed to return radon devices to the laboratory can be quite long, and in such cases it is important that the exposed devices are not temporarily stored in rooms with high radon concentrations, or that they are heat-sealed in radon-proof bags. Radon proof bags should be heat-sealed preferably in a room with a low radon concentration, in order to reduce the radon concentration enclosed in the radon proof-bag, which would produce an unintended exposure.

Questionnaire. One of the relevant purposes of a nationwide representative radon survey is to try to identify dwelling characteristics and other parameters (such as living habits) that could explain (part of) the variability of the observed radon concentration levels. For this purpose a questionnaire is generally set up in order to collect information on such parameters [51]. This questionnaire can be more or less detailed and comprehensive, depending on who fills it, i.e. trained personnel interviewing householders or householders themselves. However, most of the information that can be collected is actually a proxy of the real parameters affecting radon concentration levels, and therefore it is quite common that such parameters explain only a small fraction of the total observed variability [85]. Moreover, some studies suggest that building design has a more pronounced influence on the radon levels than the habits and preferences of householders [86]. Another important purpose of the questionnaire is to collect information on the sampled units (both dwellings and householders) that can be compared with corresponding values for the whole target population in order to verify the representativeness of the actual sample in the common case of a high percentage of refusals.

3.4. DATA ANALYSIS

Checks of representativeness. As already discussed in section 3.3, despite all the efforts, the fraction of randomly selected dwellers that refuses to participate a survey is generally non negligible, so that a check of the actual representativeness of the final sample should be carried out by comparing sample characteristics (especially those that could be correlated with radon concentration level) with those of the general population, usually collected through a census. This check is necessary in case of the use of a proxy or surrogate of a representative survey (60).

Sampling scheme and data analysis. The analysis of the measurement results has to take into account the sampling scheme in order to obtain representative estimates of the parameters of the indoor radon concentration distribution. In particular, the probability for each dwelling to be included in the sample and the strata size has to be properly considered. The easiest case is the simple random sampling, where all the results can be analysed together with no weighting, because the probability to be included in the sample is the same for all the sampled dwellings and there is a single stratum. In this case, for example, the representative value of the average radon concentration is just the sum of average radon concentration for each sampled dwelling divided for the number of sampled dwellings. In case of more complex sampling schemes, e.g. a stratified one, the parameters have to be calculated separately for each stratum and combined in a proper way to take into account the strata size [57, 87].

Log-normality of radon concentration frequency distribution: parameters and related formulas. The frequency distribution of radon concentration tends to have a log-normal shape, i.e. logarithm of measured radon concentrations tends to be normally distributed. This is due to the multiplicative combination of the many parameters affecting indoor radon concentration [88]. Therefore it is useful to estimate parameters describing the log-normal distribution. For a pure log-normal distribution (see Eq. 1), the parameters fully describing the distribution are the *geometric mean* (*GM*) and the *geometric standard deviation* (*GSD*), which are defined by:

$$GM = e^{\mu_y} \qquad GSD = e^{\sigma_y} \tag{1}$$

where $y = \ln(x)$ is the natural logarithm of the radon concentration x, μ_y and σ_y are the arithmetic mean and the standard deviation of y values, respectively. In a normal (Gaussian) distribution, 68.3 %, 95.4 %, and 99.7 % of values are within the arithmetic mean ± 1, 2, 3

standard deviations. In a log-normal distribution, the same percentage of values are within *GM* divided and multiplied by GSD^k, with $k = 1, 2, 3$. For example, 95.4 % of values are within GM/GSD^2 and $GM \times GSD^2$.

The relation between *GM, GSD* and arithmetic mean (*AM*) is given by the following formula (Eq. 2):

$$AM = e^{\ln(GM) + \frac{\ln(GSD)^2}{2}} = e^{\mu_y + \frac{\sigma_y^2}{2}} \tag{2}$$

Knowing GM and GSD, the percentage of dwellings exceeding any reference level (RL) can be easily estimated by using statistical tables of the area under the standardized normal curve, with *k* given by the following formula (Eq.3) [89]:

$$k = \frac{\ln(RL) - \ln(GM)}{\ln(GSD)} \tag{3}$$

Providing that the distribution is well approximated by a log-normal function and that the GM and *GSD* are correctly evaluated (see following paragraphs), the estimates obtained with the above formula are very useful because, due to the log-normal shape, the number of dwellings exceeding high reference values that can be directly observed in a nationwide survey is generally quite low and therefore its direct estimate would be quite uncertain. This argument is more and more valid the smaller the sample size is. Robust estimates of *GM* and *GSD*, or alternative distributions, can be used in order to reduce the effect of outliers and measurement uncertainty [90-93].

Log-normality of radon concentration frequency distribution: checks and corrections. It is important to perform some check of log-normality (in practice some test of normality of the logarithms) of measured concentration distribution. The best way is to plot measured values on a *normal probability plot,* where perfectly log-normally distributed data would appear as a straight line. Normal probability plots can generally be produced by statistical software packages, however they can also be obtained with Excel by following the instructions reported on www.stat.unc.edu/teach/rosu/Stat31/E1_104.html. Significant deviations from log-normal distribution often occur on the high value tail of the frequency distribution of measured radon concentrations. In particular, observed percentage above some reference values are often higher than those estimated on the basis of the log-normal approximation, which therefore would produce an underestimate of such percentages [54, 62, 85, 94]. These deviations are probably (at least in part) due to an additive component of the measured radon concentration, related to the outdoor radon concentration or to other sources. This additive component can be estimated on the basis of representative outdoor radon concentration values [85], or numerically estimated by subtracting different constant values from each measured radon concentration, so that, after correcting for such additive component, the frequency distribution corresponds more closely to a log-normal distribution [62, 89]. An example of normal plots with uncorrected and corrected radon concentrations is shown in Figure 2.

Correction for year-to-year variations of radon concentration. The measured frequency distribution of indoor radon concentration, as obtained with a nationwide survey with a 12-month measurement period, has a larger width than the true one, due to the impact of the year-to-year variations of radon concentration. Therefore, in order to obtain an unbiased estimate of the radon concentration distribution width, the variance of the year-to-year variations (in the log scale) should be estimated and subtracted from the variance (in the log scale) of the observed radon distribution [95]. This correction reduces the *GSD,* whereas the *GM* remains

unchanged. The observed arithmetic mean (*AM*) can be corrected by the following formula [96]:

$$AM_{corr} = AM_{obs} \times \exp\left\{-\frac{\sigma_{yty}^2}{2}\right\}$$

(4)

where AM_{corr} is the corrected average, AM_{obs} is the observed average, and σ_{yty}^2 is the estimated variance of the year-to-year variations, in the log scale.

Main analytical results to be reported. On the basis of the previous paragraphs, the main data analysis results to be reported include: (i) a frequency distribution plot; (ii) a normal probability plot of the logarithm of measured concentrations in order to evaluate possible deviations from a pure log-normal distribution; (iii) evaluations of the log-normal distribution parameters *GM* and *GSD*, for example specifying whether they are obtained from the uncorrected distribution or from the distribution corrected to better follow a log-normal shape; (iv) the arithmetic mean (and its standard error), which is proportional to the total population risk; (v) percentages of dwellings exceeding several reference levels, possibly specifying if they have been directly obtained from the measured distribution or if they have been estimated on the basis of *GM* and *GSD* (ideally both values could be reported). Last, but not least, some summary information on the sampling scheme – as well as of the representativeness check described above – should be reported in order to enable evaluation of the survey representativeness.
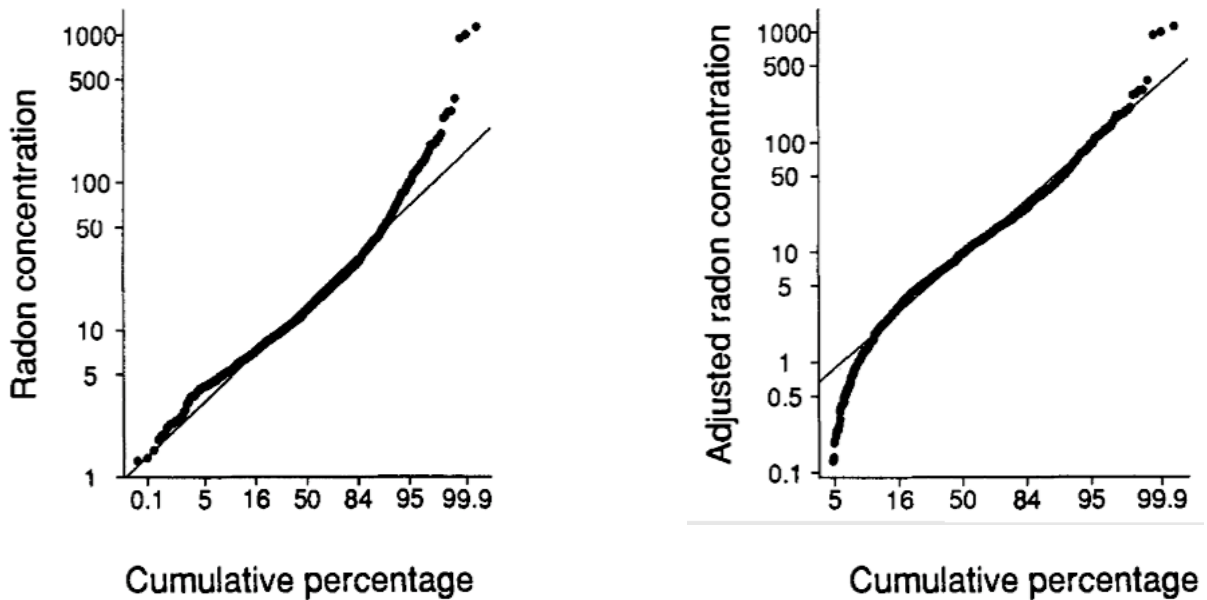
FIG. 2. Frequency distribution of logs of indoor radon concentration in UK (taken from [85]). Left: in the observed (uncorrected) distribution, the plotted points have GM = 14.9 Bq m^{-3} and GSD = 2.2 (solid line indicates best fitting straight line). Right: in the distribution corrected for a constant additive component, representing outdoor radon concentration (i.e. concentrations less than or equal to 4.1 Bq/m^3 are set to 4.1 Bq/m^3, and 4 Bq/m^3 is then

subtracted to all concentrations), the plotted points have GM = 8.8 Bq/m³ and GSD = 3.6 (solid line indicates best fitting straight line).

4. MEASUREMENT TECHNIQUES

4.1. MEASUREMENT REQUIREMENTS FOR LARGE RADON SURVEYS AND SUITABLE TECHNIQUES

Measurement requirements for large radon surveys. In order to measure the annual average radon concentration – either with a single period of 12 months or with consecutive shorter periods of 3 or 6 months with a total duration of 12 months – in a large number of dwellings within a nationwide (or regional) survey, a radon concentration measurement technique is needed with the following main requirements: (i) sampling periods of 3 to 12 months, (ii) low cost, and (iii) low impact, i.e. small size of the measuring device.

Suitable techniques. Passive measuring devices[1] based on *alpha track detectors* [97] fulfil all the above requirements, and therefore are reviewed in this section. It is worth noting that similar requirements to those above are valid also for epidemiological studies aimed to estimate lung cancer risk from prolonged exposure to radon in dwellings, so that also these studies were carried out by using alpha track detectors [98]. Alpha track detectors (ATD) are also called *nuclear track detectors* (NTD*)*, or *etched track detectors,* or *solid state nuclear track detectors* (SSNTD). The possible use of these detectors for radon monitoring was first suggested by [99]. A detailed description of the track formation in these detectors is beyond the scopes of this review, but it can be found in several books and review papers [100, 101]. The lower limit of detection (LLD) for this technique for a 3-month exposure is 5–10 Bq/m³, depending on the size of the scanned detector area [102].

In principle, the electret ionization chamber (EIC) is a suitable technique as regards its capability of long sampling periods and the small size of the device [103]; however, the cost for measurement is quite higher than that of etched track detectors. Moreover, contrary to alpha track detectors, electrets are sensitive to gamma radiation too, thus requiring a separate determination of gamma dose rate for an accurate measurement. For these reasons, EICs in long-term measurement configuration are often used for single measurements, for small surveys, or for outdoor measurements [80], but they are rarely used for large surveys on long-term radon concentration in dwellings [104]. Moreover, EICs in short-term measurement configuration are also used for screening purposes, and they are described in section 4.4.

Non-suitable techniques. Short-term measuring techniques are in principle not suitable to measure annual average radon concentration. However, these techniques (e.g. charcoal canisters and, more recently, electret ionization chambers and alpha track detectors) have been largely used in some countries (e.g. in the USA) for screening purposes, i.e. trying to quickly identify dwellings with potential high radon concentrations [105, 106], and are shortly reviewed in section 4.4. In any case, measurement protocols generally recommend the confirmation of such screening measurements with a long-term measurement [107], due to

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[1] Radon concentration measuring devices are often improperly referred to as *radon dosimeters*. In this report they are referred to as *radon devices,* or *radon measuring devices*, or *radon concentration measuring devices*.

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high uncertainty associated with short-term radon concentration measurements used to estimate annual average radon concentration [108, 109].

Active instruments (which require a power source) can continuously monitor radon concentration over long periods, but they do not fulfil the other requirements for a nationwide survey, due to their high cost and large size. Active instruments were used only in one early nationwide survey, when etched track detectors were not common, yet [110]. However, active instruments are needed to monitor radon concentration during calibration exposures and for other quality assurance operations. These instruments are not reviewed here, but some information can be found in several review papers and reports [102, 111, 112, 153].

4.2. PASSIVE DEVICES BASED ON ALPHA TRACK DETECTORS

Alpha track detector materials. The main materials used for ATDs are the three following polymeric plastics: poly-allyl-diglycol-carbonate (PADC) (generally known by its commercial name *CR-39*), cellulose nitrate (CN) film (commercial names *LR 115* and *CN 85*, although the latter one has been recently discontinued), and polycarbonate (PC) (commercial names *Makrofol* and *Lexan*). The passage of an alpha particle through an ATD produces a narrow primary damage trail or latent track along the length of its path in the material (typically 20 to 70 μm) (see Table 1). The diameter of latent tracks is in the order of tens of nm, whereas the diameter of etched tracks can reach some μm thus being visible under an optical microscope [100]. The thickness of PADC and PC detectors range from about 0.1 to 1 mm, whereas CN detectors consist of a 12–13 μm thin film on a 100-μm thick polyester support.

Radon concentration measuring devices: 'closed' and 'open/bare' configurations. Several different devices for measuring radon concentration have been used for indoor radon surveys [18, 97]. These can be categorized in two main groups: 'closed' configuration devices and 'open' (or 'bare') configuration devices.

In the *closed* configuration devices, some type of filter or barrier is used to prevent radon decay products from entering the sensitive volume of the devices, so that the signal on the detector is related only to radon which entered the sensitive volume and to the decay products formed in the sensitive volume. Therefore, the measurement result of radon devices in closed configuration is not affected by the presence of radon decay products in indoor air. Filters and barriers against radon decay products are of course effective also against thoron decay products, whereas their effectiveness against entry of thoron gas can be quite different for different devices. Sensitivity of radon measuring devices to thoron is discussed in more detail in section 4.3.

In the *open* (or *bare*) devices, alpha track detectors are open faced (or bare) and exposed to the alpha particles emitted by all radionuclides present in indoor air, with no filter or barrier for the radon decay products. Therefore the measurement result is not related to radon concentration only, but both to radon concentration and to radon decay product concentrations, so that, for a given level of radon concentration to be measured, the measurement result will be different for different equilibrium factors (i.e. for different ratios of radon decay product concentration over radon concentration), as shown in intercomparison exercises [113]. Moreover, devices in open/bare configuration are also very sensitive to radon decay products plated out on the detector surface [114, 115], although this effect is not relevant for CN detectors, which are not sensitive to radon decay products plated out on the

detector surface, due to their small thickness. Therefore devices with open configuration are also difficult to calibrate [116, 117].

Most of the nationwide surveys were carried out with radon concentration measuring devices in closed configuration, which is generally considered the preferred configuration on the basis of the above arguments. Therefore these devices are further reviewed in this section.

Several radon measuring devices in *closed* configuration have been developed and applied for nationwide surveys. They are characterized by different systems to avoid the entrance of radon decay products in the sensitive volume. In *filter permeation devices*, a permeable filter closes the open end of the device. The filter is made of a porous material (such as glass fiber, microporous paper, polyethylene or PVC) with area and thickness designed to prevent the entry of radon decay products and of thoron, with a small reduction of radon entry [118]. Examples of such devices used in nationwide surveys are the TRACK ETCH (by Terradex Co. USA), containing a PADC detector [119], the KfK (Kernforschungszentrum Karlsruhe, Germany) Type A device and the STUK's own design one [120], both containing a PC detector. In *plastic bag permeation devices*, the device is fully enclosed in a heat sealed polyethylene bag, which protects the device from dust and humidity and prevent the entry of radon and thoron decay products and reduce the entry of thoron [121, 122]. An example of these devices is that used in the Italian national survey on radon in dwellings and in an epidemiological study carried out in Italy, containing two CN detectors [57, 123]. The thickness of the low density polyethylene bag is about 35 μm, although quite thicker plastic bags have been used to prevent entrance of moisture in particular environments with very high humidity [124]. The *diffusion devices* are more and more used and are made by a small diffusion chamber composed by a base and a lid, where diffusion of radon take place around a snap-fit lid. In the small container, the temperature gradient is small, which reduces the effect of convection inside the chamber and hence the effects of electrification and plate-out [125]. Examples of radon measuring diffusion devices utilized or suitable for nationwide surveys are: the NRPB/SSI device, designed by the National Radiological Protection Board (UK, presently Health Protection Agency) and modified by the Swedish Radiation Protection Institute by using a conductive material, the TASTRAK device by TASL (UK), the Radopot device by Radosys (Hungary) and the RadOut device by MI.AM (Italy), all containing a PADC detector.

Track etching and counting methods. After exposure, latent tracks in the detectors can be made visible by a chemical etching with aqueous alkaline solutions of potassium hydroxide (KOH) or sodium hydroxide (NaOH), thus obtaining enlarged pits that can be easily counted [126]. The main etching parameters are temperature, etchant concentration and etching time. Stirring can also affect etching results. Tracks in CN detectors are usually etched with a 10% NaOH solution (2.5 N) at about 60ºC for about 90 to 110 (or a few more) minutes. For PADC detectors, 25% NaOH solutions (6.25 N) are generally used with different combinations of temperature and etching times, but KOH solutions are also applied to obtain a faster etching than with NaOH solutions. For PC detectors, a chemical etching (usually with KOH and other solutions) followed by an electrochemical etching are generally used. Several examples of etching conditions can be found in inter-laboratory comparison reports e.g. [113, 127].

Detector track density can be measured with different systems, depending also on the detector type [126]. Tracks in CN detectors can be viewed with an optical microscope and are manually or automatically counted, or (in the 'strippable' version, the 100-μm thick polyester support of which can be removed after etching) they can be counted with a non-optical system, such as a spark-counter [128, 129]. This technique is quite simple, although a

significantly non-linear response occurs at high track density (with few thousands of tracks/cm^2 as upper limit of measurable track density) due to the size of the typical evaporated aluminium area, which is quite larger than the track areas in order to avoid repeated counts of the same track. Tracks in PADC detectors can be counted by automatic systems based on an optical microscope or on a scanner [130] acquiring images of detector surface and a computer programme analysing such images to identify and count tracks. Tracks in PC are electrochemically etched, so that the etched track diameter is so large that tracks can be easily counted by using a simple microfiche reader. However, for the same reason of the large size of etched track as with the spark-counter, non-linear effects of the response occur at relatively low exposures.

4.3. QUALITY ASSURANCE FOR DEVICES BASED ON ALPHA TRACK DETECTORS

In addition to the choice of an appropriate technique, a comprehensive quality assurance (QA) programme has to be set-up in order to obtain radon concentration measurements of good quality, as the results of several inter-laboratory comparison exercises showed that precision and trueness of passive radon devices can be quite different, also for similar or identical devices [113, 131]. A comprehensive QA programme should include: (i) the identification and control of possible sources of errors affecting measurement trueness and precision, including detector background and ageing/fading effects; (ii) calibrations, including tests of response linearity; (iii) quality control procedures, including duplicate measurements and participation in inter-laboratory comparison exercises [82, 83, 132].

(i) Detector background, ageing/fading, and other possible sources of error. Detector background depends on production and storage conditions, and can be quite different for detectors from different manufacturing lots, so it should be routinely evaluated by measuring track density in unexposed detectors. A strong reduction of detector track density background can be obtained on PC and PADC detectors by pre-etching, although this technique is not widely used [133, 134]. Ageing and fading effects can be observed for detectors exposed for long periods, resulting in an underestimation of the true exposure. These effects depend on detector types, storage conditions, track counting techniques, and are particularly important for PADC detectors counted by automatic image analyser systems [135, 136]. The stability of ATD response is also dependent on the environmental conditions, such as ultraviolet radiation, oxygen, humidity and temperature [138–141].

(ii) Calibration and linearity tests. Response of a radon measuring device depends on its own characteristics, but also on detector characteristics – which can be different for different manufacturing lots due to varying production conditions – and on etching and counting methods and procedures. Therefore a proper calibration should be carried out for each laboratory and for each manufactured lot of detectors.

The main goals of a proper calibration of radon concentration measuring devices are to obtain traceability to primary standards of radon [142–146] and to test linearity of the device response over the whole range of interest of exposure values. This can be obtained by exposing samples of devices in specially constructed calibration chambers, where radon concentration is monitored with active measurement equipment which has itself been compared with (and referred to) a standard of radon [82].

Typical exposure levels for calibrating devices used for measuring radon concentration in dwellings range from a few hundreds of kBq/m^3 × h (e.g. 220 kBq/m^3 × h, corresponding to about 100, 50, and 25 Bq/m^3 for exposure periods of 3, 6, and 12 months, respectively) to

several thousands of kBq/m³ × h (e.g. 6500 kBq/m³ × h, corresponding to about 3000, 1500, and 750 Bq/m³ for exposure periods of 3, 6, and 12 months, respectively).

Non-linear effects due to overlapping of tracks at high track density can be accounted for by using a calibration curve instead of a single calibration factor, which assumes a perfectly linear response. Significant deviations from linear response can be observed at different exposure levels, depending on the detector type, the etching conditions, and the track counting system. For example, PC detectors saturate at relatively low exposure values due to the large size of electrochemically etched tracks. Various approaches can be used to extend the linearity range, e.g. a measurement of the total dark area instead of the identified tracks can be used when a significant fraction of tracks are overlapping [130].

The response of some radon devices is sensitive to atmospheric pressure [147], so a careful calibration over a range of air pressures should be performed for radon devices chosen for nationwide surveys to be carried out at very different altitudes, or radon devices with low sensitivity to atmospheric pressure changes should be chosen [57].

(iii) Inter-laboratory comparisons and other quality controls. A very important way to control the quality of radon concentration measuring devices is to periodically participate in international inter-laboratory comparisons [82]. These inter-laboratory comparisons have been very effective in highlighting very different precision and trueness for similar or identical radon devices, presumably due to inadequate control of sources of errors in the whole measurement procedure, which include also etching conditions and track density measurement procedures. Periodical national and international inter-laboratory comparisons have been organized in different regions, i.e. in Europe, in Asia, etc. [113, 131, 148, 149]. However, inter-laboratory comparisons in radon chambers are generally conducted for short periods (up to few weeks) so that not all factors affecting measurement quality can be analysed, e.g. the effects due to ageing and fading. Therefore some inter-laboratory comparisons of more real conditions have also been organized, although it is much more difficult to control exposure in such situations [150].

Finally, it is worth noting that several studies have demonstrated that adequate quality control can improve precision and trueness of radon concentration measurements [82–84, 132].

Sensitivity of passive radon devices to thoron. Some radon devices are designed to measure radon concentration without considering the possible co-presence of thoron. Therefore, depending on the sensitivity to thoron and the device position, the measurement results of some radon devices can actually be affected by the presence of thoron [151]. In such cases, radon concentration will be overestimated [152]. The sensitivity to thoron (expressed as relative sensitivity, i.e. as sensitivity to thoron over sensitivity to radon) of some radon devices based on alpha track detectors is reported in Table 3. Relative sensitivities range from about 0.4 % for a device enclosed in a polyethylene bag [71], to about 90 % for a device designed to be highly sensitive to thoron [66]. If radon devices which are somewhat sensitive to thoron are used for a nationwide survey, special attention should be paid to the placement of devices, trying to collocate them not too close to walls suspected to have a considerable exhalation of thoron [59, 65].

4.4. SHORT TERM MEASUREMENT TECHNIQUES SUITABLE FOR SCREENING PURPOSES

Electret ionization chambers (EIC)

An electret is an electrostatically charged Teflon disc placed within a small container (ion chamber), where radon can diffuse through a filter-covered opening. EICs are often known under the commercial name of Rad Elec E-PERM, although EICs are also manufactured by other factories [153]. During the measurement period, the ionization resulting from the decay of radon and its progeny within the chamber generates ions which are collected by the charged electret, thus decreasing its voltage proportionally to the integrated radon concentration. The electret ionization chamber can be deployed for exposure periods of a few days to 12 months [103, 154], depending upon the thickness of the electret and the volume of the ion chamber chosen for use.

Contrary to alpha track detectors, electrets are also sensitive to gamma radiation, thus requiring a separate determination of gamma dose rate.

For indoor radon measurements, the electret voltage must be measured with a special reader before and after exposure. In general, all electrets should be analysed in the field or laboratory as soon as possible after removal from dwellings, in order to avoid further electret discharge due to gamma radiation background and to radon in air. The following formula (Eq.5) is used to determine radon concentration [154]:

$$C_{Rn} = \frac{V_i - V_f}{t_{exp} \times CF} - BG \tag{5}$$

where C_{Rn} is the radon concentration (expressed in Bq/m^3), t_{exp} is the exposure period (d), V_i and V_f are the measured initial and final electret voltages (V), respectively, CF is the conversion factor (V per Bq/m^3×d) and BG is the radon concentration equivalent of the gamma background.

The advantages of these devices are that the results can be obtained in the field and the devices are reusable. The disadvantages are the somewhat higher cost in comparison with SSNTDs, and the sensitivity to gamma radiation. In addition, if the temperature at the time of analysis is significantly different (more than 6°C) from that at the time when the pre-exposure voltage was determined, some corrections may be necessary. It is therefore advisable to measure voltages after the temperatures of the reader and the detector have stabilized to the room temperature in which both pre- and post-exposure voltages have been measured. In addition, Sorimachi et al. [155] reported that there is little influence of humidity, ambient aerosols and thoron on the detection response of EICs for radon concentration measurements. Then, an attention must be paid when using electret monitors in extremely high thoron concentrations.

The LLD of this technique depends on the thickness of the electret used. The short-term exposure of the more sensitive E-PERM yields a voltage drop of 0.054 V per Bq/m^3/d or about 7 Bq/m^3 in 3 days. The long-term E-PERM, which is less sensitive, yields about 0.0045 V per Bq/m^3/d [103]. A detailed evaluation of the uncertainties associated with radon concentration measurements carried out with EICs is available in literature [156].

Activated charcoal canisters (ACC)

The activated charcoal collector (or canister) has been one of the most popular passive devices for short-term measurements of indoor radon concentration [154, 157, 158]. These collectors allow continual adsorption and desorption of radon on the active sites of the carbon beds. During the measurement period (typically 2–7 days because the half-life of radon is

only 3.8 days), the adsorbed radon undergoes radioactive decay, so that the detector response gives high weight to the final days of exposure. Therefore activated charcoal devices cannot be considered good integrating devices, especially if the radon concentration is rapidly changing during the measurement period [159]. In particular, charcoal detector response gives a high weight to the final 1–2 days of exposure and is affected by temperature and humidity [129]. Therefore, protocols for radon concentration measurements with activated charcoal canisters generally recommend using them in rooms kept closed in order to both maximize radon concentration levels for screening purposes, and to reduce radon concentration variations [107, 160]. The most useful configuration has a diffusion barrier to separate the charcoal from the air and improves the uniformity of response to variations of radon concentration with time [161].

Charcoal is also a very good absorber of water vapor, which can reduce adsorption efficiency for radon. Therefore, a moisture correction is needed for this method. In some cases, desiccant (silica gel) is also incorporated in containers to reduce interference from moisture adsorption during longer exposures [162]. Before measurement of radon, the charcoal must be heated at 110°C for several hours (or at 120°C in an oven for 3 hours) to remove the moisture from the charcoal. After exposure, the device is tightly sealed to maintain maximum sensitivity and quickly returned to a laboratory for determination of the quantity of radon adsorbed by using gamma spectrometry or liquid scintillation [163].

Gamma spectrometry analysis of the exposed devices is generally performed using a NaI(Tl) gamma scintillation detector, due to its high counting efficiency, to count the gamma rays emitted by the radon decay products on the charcoal. The detector may be used in conjunction with a multi-channel gamma spectrometer or with a single-channel analyzer with the window set to include the appropriate gamma energy window. Gamma photons with energies of 352 keV from ^{214}Pb and 609 keV from ^{214}Bi are used for analysis. The background of the system must be determined by measuring an unexposed canister. The detection efficiency of the detection system can be calibrated by measuring a standard canister. The standard canister is prepared by adding a known activity of ^{226}Ra to a normal canister, which is then sealed permanently. To determine the average radon concentration during the exposure period, the difference between exposure and the background counting is divided by (i) the detection efficiency, (ii) the number of exposure days, and (iii) the decay factor. In addition, if the device does not have a moisture barrier, the detector should be weighed, and, if necessary, a correction should be applied for the increase in weight due to moisture adsorbed. A description of the procedure used to derive the moisture correction factor can be found elsewhere [158].

For liquid scintillation analysis, the devices are prepared for analysis by radon desorption techniques. This technique transfers a major fraction of the radon adsorbed on the charcoal into a vial of liquid scintillation fluid [164]. The vials of liquid scintillation fluid containing the dissolved radon are placed in a liquid scintillation counter and counted for a specified number of minutes (e.g. 10 minutes) or until the standard deviation of the count is acceptable (e.g. less than 10 %) [165]. Finally, radon concentrations are calculated by the observed count rate of radon and its decay products and by applying an empirically determined calibration factor, correction factor for radon decay, adsorption time, influencing factors for the transfer of radon from the charcoal to the scintillation fluid under rigorously controlled conditions, and the counting efficiency, which must be achieved with the specified scintillation mixture and liquid scintillation counting system [162].

The advantages of all activated carbon collectors are their simplicity and their recyclability many times after regeneration to drive off radon and other organic gases (in case of gamma analysis) and the low additional cost (if a gamma spectrometric system is already available). On the other hand, the disadvantages are that they require prompt analysis usually within 7 (preferably less) days after the end of exposure, and that water vapor and temperature affect the collection efficiency of all types of activated carbon. The LLD for collectors analyzed by gamma counting ranges from 4–8 Bq/m^3 of radon in a measurement interval of 4 days, depending on type and size. The LLD of collectors using liquid scintillation counting is about 5 Bq/m^3 in a two-day measurement interval [102, 166].

Alpha track detectors

Alpha track detectors, although less sensitive than EICs and ACCs, have been recently proposed to be used for short-term radon concentration measurements as well [106]. Compared with EICs and activated charcoal canisters, the response of radon measuring devices based on ATDs is not sensitive to gamma radiation (contrary to EICs) and does not give high weight to the final days of exposure (contrary to ACCs). The proposed protocol recommends an exposure of 14 days. In order to take into account the higher uncertainties related to such a short period, compared with a long term measurement, a precautionary uncertainty of a factor 2 is applied. For example, if the action level is 200 Bq/m^3, short term measurement results above 100 Bq/m^3 should be followed by a long-term measurement to verify that the long term result is below or over the action level. However, the precautionary uncertainty for short-term measurements based on ATDs is lower than that for measurements based on ACCs [106].

5. CONCLUSIONS

A proper survey design and data analysis are very important to obtain, by a nationwide or regional survey, an evaluation of the frequency distribution of radon concentration in indoor air of dwellings which could be representative of population exposure in the whole country or in a region within the country. Representative survey results can be compared with those of other countries and should be the basis for any national radon programme aimed to reduce the risks from radon exposure. An important role is played by the correct use of suitable measurement techniques, such as the passive radon concentration measuring devices based on alpha track detectors. A quality assurance programme is required to guarantee measurement results with adequate trueness and precision.

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