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Dehkordi, Ava Nourian

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Assessment of radiation doses to the West Rand public due to inhalation of ^{222}Rn and its daughter products

Ava Nourian Dehkordi

A dissertation submitted to the Faculty of Science,
University of Witwatersrand, Johannesburg, in fulfilment of the
requirements for degree of Master of Science

Johannesburg, 2012

Declaration

I declare that this dissertation is my own work. It is being submitted in fulfilment of the requirements for the degree of Master of Science at the University of the Witwatersrand. It has not been submitted before for any degree or examination at any other university.

Signature: 

Student's name: Ava NourianDehkordi

Date: 19-09-2012

Abstract

Radon is a radioactive gas which contributes significantly to the natural radiation background. For this reason, it is important to estimate doses to representative member of the public especially in areas with an elevated concentration of naturally occurring radionuclides e.g. the Witwatersrand Basin. This study presents the results of radon monitoring carried out in selected houses close to the gold and uranium mining operations in South Africa. Monitoring of radon and its progeny was performed inside selected houses that were built about 100 years ago, using a calibrated AlphaGuard PQ2000PRO active instrument. Radon monitoring the interpretation of the results took into consideration meteorological parameters, which are critical for evaluating radon concentrations. The results obtained by AlphaGuard were compared with long-term measurements performed in the same locations using the passive RGM (Radiation Gas Monitor) caps manufactured in South Africa. Indoor monitoring of radon and its progeny was supplemented by additional monitoring outdoors, around the selected houses, the slimes dams and in several selected locations close to mining operations. Assessment of doses due to inhalation of air contaminated by radon and its progeny was based on the obtained results of radon monitoring and the doses were compared to internationally accepted intervention and action levels established by International Commission of Radiological Protection 65 (ICRP) and various regulatory authorities throughout the world.

The values of the indoor radon concentrations were compared with the international recommendations, and since the meteorological parameters are critical for analysing the radon concentration, an effort has been made to find a possible correlation between them and the indoor radon concentration.

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Chapter 1

Introduction

1.1 Historical Background

Radon is a radioactive element discovered by the English physicist Ernest Rutherford in 1899. The discovery is also credited to a German physicist, Friedrich Ernst Dorn, in 1900. More specifically, Rutherford discovered radon's alpha-particle emission and Dorn discovered that radon was released from radium as a gas. About 90% of the radiation exposure to humans comes from natural sources such as cosmic radiation and terrestrial radiation [CA09]. However, ^{222}Rn alone can contribute about half of the average annual effective dose to humans from nature. As early as 1907, Ernest Rutherford pointed out the possibility of a health hazard due to inhalation of decayed daughters from radium and thorium (^{226}Ra,^{224}Ra,^{228}Ra). The experience of radium watch dial painters in the 1920's led to measures to decrease the danger involved in handling radioactive materials [HA79].

Long after the discovery of radon it was found that it may likely be responsible for causing lung cancer. It had been known for centuries that men who worked in the mines of Schneeberg in Germany and Joachimstahl in the Czech Republic had a high mortality rate. A significant difference in mortality rates of men and women has been observed since the 14th century, when these mines were used for mining of silver. The high mortality rate of miners working in uranium mines in the 1950's was ascribed to the incidence of so-called "Joachimstal disease" (later diagnosed as lung cancer), which appeared frequently among miners in these mines [HA79]. The high radon concentration evidently occurred underground because radon (^{222}Rn) is the immediate daughter radionuclide of ^{226}Ra, and also there are low ventilation rates underground. When it was realized that cancer could also result from internal exposure to radiation, the high mortality rate of

uranium miners was attributed to ^{222}Rn. This initiated a number of studies in the 1950's which provided not only information on health effects, but also some understanding of the mechanism of ^{222}Rn-induced lung cancer.

This discovery was related to a lung decease survey, which was carried out on an international basis and reported to the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) The UNSCEAR Report [TS97], has confirmed that in some countries people are exposed to high radon levels that are similar to those in mines. Based on the studies of underground miners, it has been estimated that indoor radon may account for 6000-36000 lung cancer deaths each year in the United States [DU05]. This estimate has significant uncertainties because there is a substantial difference between working underground and working and living in houses.

There are many case-studies which assessed the lung cancer risk from indoor radon. According to the World Health Organization (WHO), radon may increase the risk of lung cancer from 3% to 14% due to smoking [WH09].

Based on these findings, occupational limits for exposure to ^{222}Rn and its daughters in mines were formulated and substantial effort was devoted to reducing this exposure below the newly introduced limits [NE88]. While the health risks associated with high radon exposures in underground mines have been known for a long time, relatively little attention was paid to environmental radon exposures until the 1970's, when some scientists began to realize that indoor radon exposures could be quite high and in some case comparable to the exposures experienced by miners [NE88]. Since then, the flood of information on radon continues unabated. Many of the recent publications on this subject are improving our understanding of the environmental processes that affect radon exposure, but there are still many problems associated with the accurate assessment of exposures and doses to individuals and populations.

Indoor radon may specifically be hazardous in houses built in mining areas, firstly because of elevated ^{226}Ra concentration in the soil, on which the houses were built, and secondly because waste rocks, gravel and sand from the mining area

might have been used as building materials. This was, therefore, the main reason why this study has been conducted.

To provide more information on the risk of lung cancer due to indoor radon, an analysis has been conducted, which included at least 200 case studies involving long-term indoor radon measurements [DU05].

Mapping of the radon levels in a specific are a is a task that is rather difficult, since radon levels between neighbouring houses can differ quite significantly, depending on the construction material of the building and the living habits of the inhabitants. It has been found that almost every country adopted different mapping techniques and strategies because mapping involved some averaging of data collected, therefore hiding are as exhibiting higher concentration levels [ZB11]. Figure 1.1 shows an example of the worldwide map of mean atmospheric radon concentration [ZB11].

1.2 Problem definition

An important theme in radiation protection is the protection of the public against ionising radiation. A significant source of human exposure is radon (^{222}Rn). For this reason it is important to study the indoor behaviour of ^{222}Rn. Radon concentrations vary during the day, as well as with seasons due to the influence of metrological conditions [CH09].

The present study has been focused on the measurement of the concentration of indoor radon in houses built in the area of the First Uranium Ezulwini Mine (West Rand, Johannesburg, South Africa) and on the corresponding assessment of dose to the inhabitants of these houses. Indoor radon measurements were carried out in six houses built on First Uranium Ezulwini Mine land and the correlation of indoor radon concentration with meteorological parameters has also been studied.

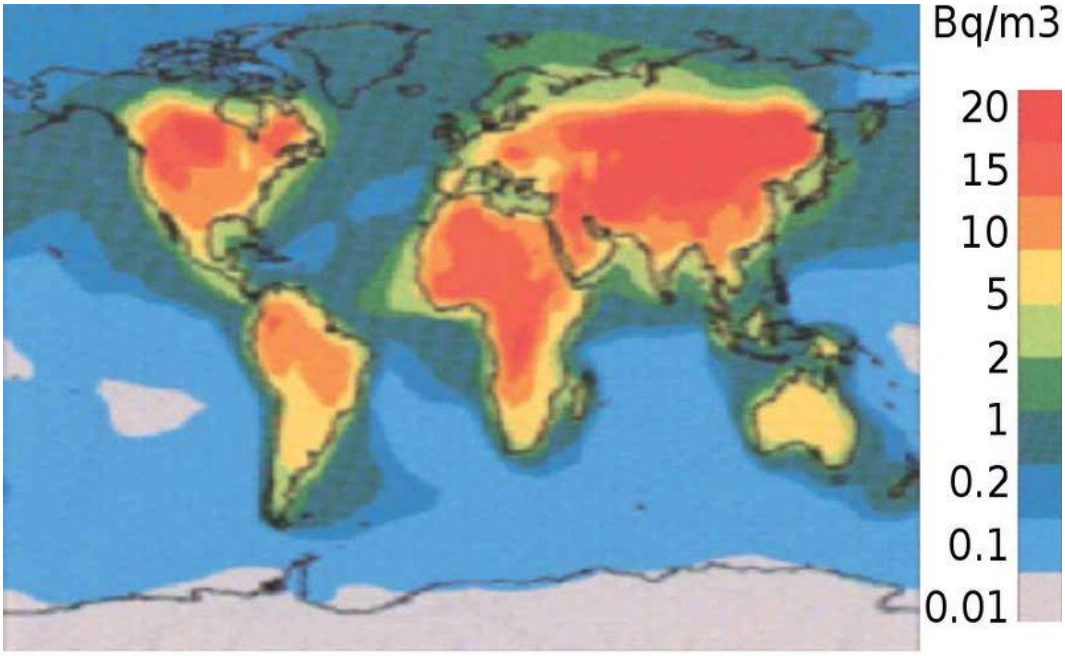


Figure 1.1 Map of the mean atmospheric radon distribution [ZB11].

This research project should be seen as a pilot investigation which will undoubtedly lead to follow-up investigations.

1.3 Dissertation arrangement

The arrangement of this dissertation is as follows:

- Chapters 2 and 3 present the necessary theory and descriptions associated with radon. Chapter 3 presents a brief discussion of the relevant measuring techniques and methods of radon concentration determination.

- Chapter 4 presents the measurements of radon concentrations indoors and outdoors during winter and summer seasons and shows the correlation of indoor concentrations with meteorological parameters.

- Chapter 5 presents the evaluation of the measured data and assessment of doses due to indoor and outdoor concentration of radon. The calculation of doses is discussed in details in this chapter. Crystal Ball [OR11] is a statistical scientific software package which forecasts the entire range of results for a given situation. All calculations and the corresponding results in Chapter 5 have been performed using this software. Accordingly, to our knowledge, this method of evaluation of doses due to indoor radon concentration has been done for the first time in the analysis of such results in South Africa.

- Chapter 6 presents a summary and recommendations for the follow-up studies. This chapter basically summarises the main conclusions of the dissertation and draws several recommendations that should be considered in the future to protect members of the public living in the measurement area.

Chapter 2

Theoretical considerations

2.1 Characteristics and properties of radon

Radon is an important source of natural radiation and it is directly produced from radium in the soil. The past studies have determined that about 90% of the radiation exposure to humans comes from natural sources such as cosmic radiation and terrestrial radiation and exposure due to radon may contribute up to 50% of the natural sources [RE05].
It is necessary to study the physical properties together with the processes of emanation and exhalation of radon to have a better understanding of radon transport modelling and estimate the general risk which can be caused by radon.

2.1.1 Physical properties of radon

Radon is a radioactive colourless, odourless, tasteless, non-flammable noble gas at room temperature which decays by alpha particle emission with a half-life of 3.82 days. As a solid with a density of 9.96 kgm^{-3} radon has a melting point of -71°C and a boiling point of -61.8°C. It is a noble (inert) gas and has three important isotopes [RE05]:

- ^{222}Rn which is called Radon and is a member of the ^{238}U decay series,
- ^{220}Rn which is called Thoron and is a member of the ^{232}Th decay series,
- ^{219}Rn which is called Actinon and is a member of the ^{235}U decay series.

2.1.2 Emanation and exhalation of radon gas

Radon is continuously being formed in radium containing soil, rocks, and minerals. It is capable of being released from the mineral grains (emanation) during its short lifetime and travelling significant distances from its site of generation [TS97]. When radon migrates from a solid surface to the atmosphere it is defined as exhalation.

Recoil could be described as a factor that facilitates or influences the emanation of an alpha particle by the parent ^{226}Ra atom. Generally, the radium isotopes are relatively heavy and immobile in the material and, therefore, they cannot easily migrate [PR79]. ^{222}Rn gas, which is produced inside a mineral grain through the radioactive decay of radium (^{226}Ra), is able to move out of the mineral into the pore from which is finally an escape into the atmosphere. The amount of ^{222}Rn in the grain which escapes is known as the Emanation Coefficient, or Emanation Fraction, or Emanation Power. Another parameter which characterises the emanation process besides the emanation coefficient is the Emanation Rate, it is the amount of ^{222}Rn leaving a grain in Bqs^{-1} [TS97].

A fundamental understanding of the processes which influence the exhalation of ^{222}Rn and the concentration of its unattached and aerosol-attached short-lived daughters is essential to understanding the operation of devices that measure ^{222}Rn exhalation or concentration [PR79]. The ^{222}Rn exhaled from soil or wall surfaces becomes distributed in the air volume and, being chemically inert, migrates by diffusion and convection without major interaction with trace gases, vapour, aerosols or other constituents of the air [PR79].

The exhalation rate E (t) of ^{222}Rn from a porous piece of material is defined as the flow of ^{222}Rn to the surrounding air per unit area of the body surface per unit time. Exhalation is a two- step process [PR79]:

a) transport of ^{222}Rn from the solid matrix to the pore space and
b) transport of ^{222}Rn through the pores (or cracks) to the surface of the body.

Both for the science and also for choosing proper remedial measures for controlling the ambient level of ^{222}Rn, it is crucial to know the exact nature of the mechanism of the liberation of ^{222}Rn from crystal line minerals [TS97].

2.2 Indoor radon gas

In 1956, high levels of indoor ^{222}Rn gas concentration were found in some houses in Sweden [HE86]. These concentrations were first ascribed to a large amount of ^{226}Ra in building materials made of alum-shale concrete and also low ventilation rates of the houses. Later studies [HE86] have revealed that the high level of indoor ^{222}Rn emanated from soil under the houses. In 1984 it was established that a high indoor concentration of ^{222}Rnrepresented a considerable risk to people's health [HE86].

Figure 2.1 indicates various routes of ^{222}Rn gas entry into a typical house (although the houses investigated in study do not have a basement the principle routs for radon ingress remain as shown in Fig. 2.1). Radon may enter directly from the soil or rock underneath the building or from the building materials. It can also be carried into the building by water, natural gas and air from other areas or from outside. Radon can enter a building from the ground through small cracks in floors and through gaps around pipes or cables. It tends to be sucked from the ground into a building because the indoor air pressure is usually slightly lower than that in the soil underneath and outside air. This pressure difference occurs because warm indoor air is less dense than outdoor air. The permeability near the ground and also the construction style and type of building materials used, especially those ones which are in direct contact with the ground, can influence the ^{222}Rn concentration in the houses [NA88].

Radon concentrations inside houses also depend on the rate of removal by either decay or ventilation. It has been observed that higher ventilation rates during summer decreases the level of ^{222}Rn concentration compared to winter [NA88].

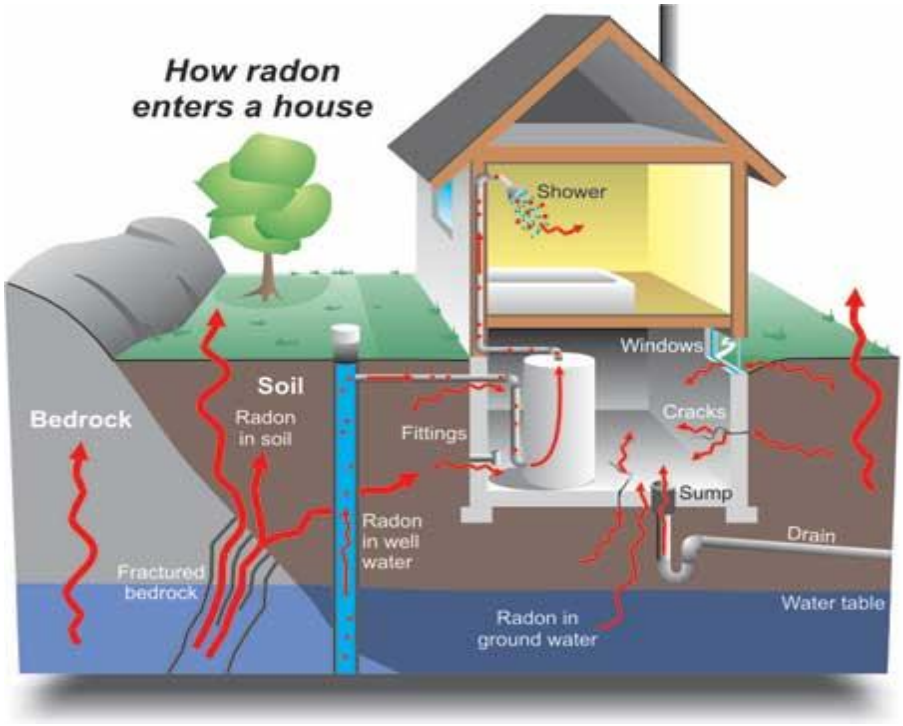


Figure 2.1 Schematic representations of indoor ^{222} Rn gas pathways [RA08].

The variety of specific circumstances in a dwelling makes it difficult to establish a generic theory for the control of ^{222}Rn concentrations in houses.

2.3 Radon family

Alpha decay is a radioactive process in which a particle with two neutrons and two protons is ejected from the nucleus of a radioactive atom. The particle is identical to the nucleus of a helium atom. Alpha decay only occurs in very heavy elements such as uranium, thorium and radium. There are two reason for this event, first is that in the heavy nuclei the electrostatic repulsive forces increase much more rapidly than the cohesive nuclear forces, and depending on the size of the electrostatic forces, may very nearly approach or even exceed that size of the nuclear force, the second reason is that the emitted particle must have sufficient energy to overcome the high potential barrier at the surface of nucleus resulting from the presence of positively charged nucleons. To see the alpha emission from the high atomic numbered naturally occurring elements theoretical considerations

require that an alpha particle have a kinetic energy greater than 3.8 MeV [CE96]. After an atom ejects an alpha particle, a new parent atom is formed which has two fewer neutrons and two fewer protons. Because alpha particles contain two protons, they have a positive charge of +2e. Further, alpha particles are heavy and very energetic compared to other common types of radiation. These characteristics allow alpha particles to interact easily with the material they are travelling through, including air, causing a lot of ionization in a very short distance. Such an interaction is by a high Linear Energy Transfer (LET). Typically, alpha particles will travel not more than a few centimetres in air and are stopped by a sheet of paper [PR79]. As a result, alpha radiation from the source outside of the body can damage skin and eyes.

Generally for alpha-particle emission to occur the following Q-value equation must be defined [CE96]:

$$M_p = M_d + m_\alpha + 2m_e + Q \qquad (2.1)$$

where,
M_p = neutral atomic mass of the parent,
M_d = neutral atomic mass of the daughter,
m_α = mass of the emitted alpha-particle,
m_e = mass of the electron and
Q = total energy released by the radioactive transformation.

The kinetic energy is the total energy released and is divided between the alpha particle and the recoil daughter after the alpha-particle is emitted. The exact energy division between the alpha and recoil nucleus depends on the mass of the daughter and may be calculated by the application of the laws of conservation of energy and momentum. Alpha particles are essentially mono energetic. However, there are separate energy groupings in alpha-particle energy spectrogram, with small energy differences among the different groups. These small differences are because of the differences in the energy levels of the excited daughter nucleus.

The residual nucleus after emission of the highest energy alpha-particle is usually in its ground state. A nucleus left in an excited state emits its energy of excitation in the form of a gamma ray. It should be noted that most of the alpha particles are usually emitted with the maximum energy. Very few nuclei are left in excited state and gamma radiation, therefore, accompanies only a small fraction of the alpha rays. Radium is an example of an alpha emitter with a complex energy spectrum (see Fig. 2.2 [CE96]).

As can be seen in Fig. 2.2 in the great majority of transformations of ^{226}Ra, 94.3%, alpha-particles are emitted with kinetic energy of 4.777 MeV. The balance of the alpha particles is 5.7%, having kinetic energies of only 4.591 MeV. In this case where a lower energy alpha is emitted, the daughter nucleus is left in an excited state, and looses of its energy of excitation by emitting a gamma ray photon whose energy is equal to the difference between the energy of the excited state and the next lowest available excited state or the ground state of the daughter nucleus [CE96].

As a starting point, ^{238}U forms a naturally occurring decay series starting with the alpha decay of the parent nucleus to the corresponding daughter nucleus [FO09]:

$$^{238}\mathrm{U} \rightarrow {}^{234}\mathrm{Th} + \alpha \ (E_\alpha = 4.267\mathrm{MeV}), \tag{2.2}$$

Where E_α was defined as the energy of the α-particle emitted populating the ground state of the daughter nucleus. Again, should be noted that sometimes the daughter nucleus is left in an excited state.

The daughter isotopes of ^{238}U are also unstable and the decay series ends with the stable isotope ^{206}Pb. The pre-formed α-particle is released from the unstable parent nucleus by a process of quantum mechanical tunnelling through the Coulomb barrier. Low energy α-particles ($E_\alpha \approx 4$ MeV) are associated with very long half-lives ($T_{1/2} \approx 10^9$ y) while higher energy α particles ($E_\alpha \approx 9$ MeV) are associated with very short half-lives ($T_{1/2} \approx 1$ ms). In naturally occurring uranium,

these decay chains are generally in secular equilibrium (defined later). This means that in 1 g of natural uranium each nuclide of the ^{238} U series has an activity of 12356 Bq (see Fig. 2.3) [FO09].

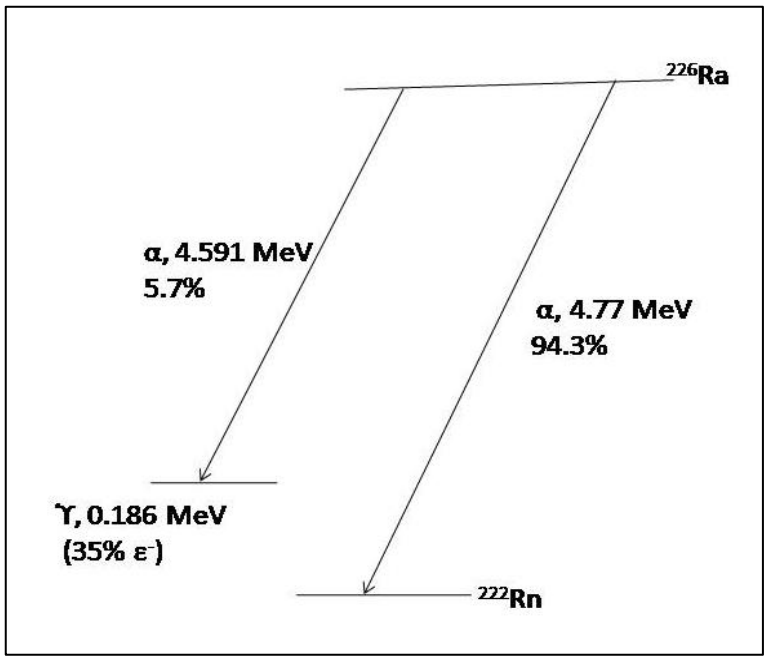


Figure 2.2 ^{226}Ra transformations (decay) by alpha-particle emission [CE83].

Certain nuclei which have an excess of neutrons may attempt to reach stability by converting a neutron into a proton with the emission of an electron and an anti-neutrino, $\bar{\upsilon}_e$ [FO09]:

$$n^0 \rightarrow p^+ + e^- + \bar{\upsilon}_e, \tag{2.3}$$

The decay constant λ quantifies the probability of radioactive decay per second for a radionuclide. For a large number of N atoms of a radionuclide, $N\lambda$ atoms will decay per second. This is the radioactivity of a radionuclide and its unit is Becquerel (Bq). The half-life $T_{1/2}$ is the time it takes statistically until half of the N atoms have decayed and the decay constant can be determined from the expression [PR79]:

$$\lambda = \frac{\ln(2)}{T_{1/2}}. \tag{2.4}$$

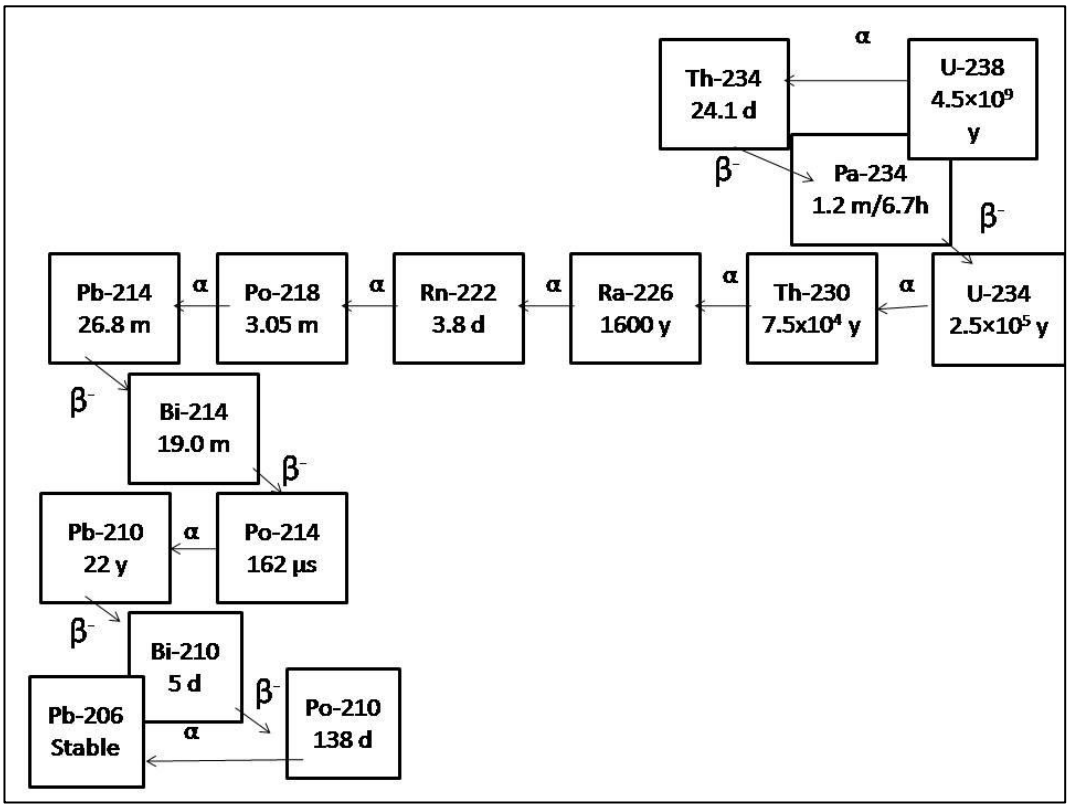


Figure 2.3 ^{238}U decay series (time shown indicate the half-life of each daughter nucleus) [FO09].

- The half-life of ^{222}Rn is 3.8 days with polonium (^{218}Po, ^{214}Po), bismuth (^{214}Bi) and lead (^{214}Pb) isotopes exhibiting half-lives of less than 30 minutes. The ^{238}U decay scheme eventually leads to the relatively long-lived radionuclide ^{210}Pb with the half-life of 22 years. It is well known that when a sample of a radionuclide is in a closed volume and the half-lives of daughters are much shorter than the half-life of the parent radionuclide, the activity of the decay products will eventually become equal to the activity of the parent radionuclide. This set of conditions is defined as secular equilibrium. Secular equilibrium is a steady-state condition of equal activities between a long-lived parent radionuclide and its short-lived daughter. The important criteria upon which the secular equilibrium depends are:

- The parent must be long-lived and
- The daughter must have relatively short half-life, so that $\lambda A << \lambda B$, where λA and λB are the respective decay constants of the parent and the daughter.

However, in case of the uranium series, the fourth and fifth member of this series (i.e. ^{234}U and ^{230}Th) exhibit a very long half-life and the secular equilibrium may be disturbed, if there is different separation between U and Th elements, e.g. different leaching rate in nature. The same applies to ^{226}Ra, which has the half-life of about 1600 years, but can (and often is) separated in nature at a different rate than e.g. uranium. Therefore, in nature, due to the specific temperature due to and leakage of some of the radionuclides in soil, the concentration of ^{226}Ra is not necessarily equal to that of ^{238}U. In addition, because rock is slightly porous and ^{222}Rn is a gas, it will emanate out of any rock or soil surface and escape into the atmosphere. These processes may disturb the radioactive equilibrium in the rock or in the soil. The same holds for ^{222}Rn and its decay products in the air of a room. The ^{222}Rn decay products are chemically active and are often electro-statically charged and, when they are produced in the ambient air, they will attach to aerosols soon after being produced, leaving only a small fraction of the decay products unattached. Both the attached and unattached fractions may plate-out on surfaces and, therefore also in the human respiratory tract. ^{222}Rn, as well as its decay products, may be removed from indoor air by ventilation. Therefore it can be expected that in general there will be no radioactive equilibrium between ^{222}Rn and its decay products in both, indoor as well as outdoor air [PR79].

Working Level (WL) is a measure of the atmospheric concentration of radon and its progeny. One WL is defined as any combination of the short lived radon daughters in one litre of air that will result in the ultimate emission of 1.3×10^5 Jm^{-3} of alpha energy. This unit was defined during the 1950's with the introduction of safety standards for ^{222}Rn concentrations in uranium mines [IC93].

The Potential Alpha Energy Concentration (PAEC) of the ^{222}Rn daughters in radioactive equilibrium with the ^{222}Rn concentration of 3750 Bqm^{-3} (100 pCil^{-1}) is approximately a PAEC of 1 WL [VA85]:

$$1\text{WL} \approx E_{\alpha\,\text{tot}} C_0\,, \tag{2.5}$$

where C_0 = 3750 Bqm^{-3}. In addition as it has been shown in Table 2.1 the total α-energy can be expressed as [VA85]:

$$E_{\alpha\text{tot}} = \sum_i E_{\alpha i}\ (\text{JBq}^{-1}), \tag{2.6}$$

where $E_{\alpha i}$ (JBq^{-1}) is the potential α-energy due to the decay of an individual decay product and the subscript i = 1, 2, 3 denotes ^{218}Po, ^{214}Pb and ^{214}Bi. On the other hand, $E_{\alpha i}$ (JBq^{-1}) is the total energy released per Bq of the isotope as it decays through the intermediate daughters to ^{210}Pb.In other words it is the result of performing an integration of a rate with respect to time.

Table 2.1 Potential α-energy released ($E_{\alpha i}$) per *one* Bq of each of the decay products of ^{222}Rn. Note that 1 MeV= 1.6×10^{-13} J [VA85].

Nuclide	**$E_{\alpha i}$ (JBq^{-1})**
^{222}Rn	Excluded
^{218}Po	0.58×10^{-9}
^{214}Pb	2.85×10^{-9}
^{214}Bi	2.11×10^{-9}
^{214}Po	2.93×10^{-16}
Total $E_{\alpha\text{tot}} = \sum_i E_{\alpha i}$ (JBq^{-1}) (at equilibrium) per Bq of ^{222}Rn	5.56 ×10^{-9}

It then follows that PAEC (Jm^{-3}) can then be expressed as [VA85]:

$$\text{PAEC} = \sum_i E_{\alpha i} C_i, \tag{2.7}$$

where C_i is the individual decay product concentration (Bqm^{-3}).

Equilibrium-Equivalent Decay Concentration (EEDC) is further defined as the concentration of the mixture of the short lived decay products in air. One EEDC (Bqm^{-3}) is the activity concentration of ^{222}Rn gas in radioactive equilibrium with its short-lived progeny that exhibits the same PAEC as the non-equilibrium mixture of the progeny actually present [VA85].

The relation between PAEC and EEDC is given by:

$$\text{EEDC} = \text{PAEC}\frac{C_0}{1\ \text{WL}}, \tag{2.8}$$

where $C_0 = 3750\ Bqm^{-3}$ and $1WL = 2.08 \times 10^{-5}\ Jm^{-3}$.

The Equilibrium Factor F has been introduced to describe the activity concentration of ^{222}Rn daughters in terms of ^{222}Rn concentration for non-equilibrium situations. This factor is defined as the ratio of Equilibrium-Equivalent Decay-product Concentration and activity concentration [VA85]:

$$F = \frac{\text{EEDC}}{C_{\text{Rn}}}, \tag{2.9}$$

where C_{Rn} is the activity of ^{222}Rn concentration.

If ^{222}Rn and all its decay products are in radioactive equilibrium ($C_i = C_{\text{Rn}}$), then F=1. For indoor air F can vary from 0.2 to 0.6, with the nominal value of 0.4 [IC93]. The concentration of decay products may change, depending on ventilation systems and deposition of radon daughters on surfaces.

ICRP also introduced the principle of justification of exposure as a consequence of cost benefit analysis, which is based on the ALARA principle (As Low As Reasonable Achievable). This principle requires that all radiation exposures have

to be kept "as low as reasonable achievable". ICRP recommended in 1985 that the total dose to the general population should not exceed 5 $mSva^{-1}$ for a limited number of years [IC07][KE89].

The Commission reaffirms that "*radon exposure in dwellings due to unmodified concentrations of ^{226}Ra in the earth's crust, or from past practices not conducted within the Commission's system of protection, is an existing exposure situation. Furthermore, the Commission's protection policy for these situations continues to be based on setting a level of annual dose of around 10 mSv from radon where action would almost certainly be warranted to reduce exposure"* [IC09].

2.4 Risks of radon exposure

Because the half-life of radon is significantly longer than the stay time of air in the lungs, most of the radon atoms inhaled are exhaled without decaying. A negligible number of the ^{222}Rn atoms decay while inside the lungs, and usually can be ignored. On the other hand, short-lived radon daughters (^{218}Po, ^{214}Pb, ^{214}Bi and ^{214}Po) are chemically more reactive and attach to the cells of the respiratory tract [RE05]. The dominant mechanism for radiation damage of the respiratory airways is the dissipation of α-particle energy by dense ionisation and excitation of the bronchial and lung cells. Alpha particles emitted have a short range (tens of μm) in this tissue [WI00]. The interaction of α-particles with bronchial and lung tissue can modify the cell structure and may finally lead to the development of cancer [EU09]. Radon gas has, therefore, negligible immediate health effects, but its short–lived solid-particulate daughter products are responsible for the main health damage.

Radon gas enters dwellings from the underlying soil. It builds up to high levels and may cause cancer of the respiratory tract many years after the exposure has occurred. The inhalation of the short-lived ^{222}Rn decay products, and to a lesser extent the decay products of ^{220}Rn (thoron), are the main pathways for radiation exposure of the bronchial tree and lungs [RE05].

The irradiation of basal cells of the upper lung airways by α-particles is responsible for the lung cancer experienced mainly by miners in uranium and other mines where the deposit is contaminated with uranium [WI00]. A quantity widely used for assessing the dose due to radon inhalation is the concentration of ^{222}Rn in air measured or calculated in Bqm^{-3}. As an alternative to a dosimetric approach, ICRP has derived a conversion convention for radon exposures based on the equality of detriments from epidemiological determinations. Radiation deteriment is the total harm that would eventually be experienced by an exposed group and its descendants as a result of the group 'exposure to radiation from a source [IC93], and risk is the probability of a specified health effect occurring in a person or group as a result of exposure to radiation [IC93]. The nominal mortality probability coefficient for radon for males and females was taken to be 8×10^{-5} per mJhm^{-3}. This value was determined from occupational studies of miners. Although the exposure conditions in mines are different from those in homes, the differences are compensating, e.g. lower unattached fractions and higher breathing rate in mines than in homes. The above coefficient was related to the detriment per unit effective dose, chosen as 5.6×10^{-5} per mSv for workers and 7.3×10^{-5} per mSv for the public [IC91]. The values of the conversion convention are thus $8\times10^{-5}\times5.6\times10^{-5}=1.43$ mSv per mJhm^{-3} for workers and $8\times10^{-5}\times7.3\times10^{-5}=1.10$ mSv per mJhm^{-3} for a representative member of the public. The rounded values adopted by the ICRP are 1.4 and 1.1 mSv per mJhm^{-3}, respectively [IC93]. An often used relationship between a unit radon concentration in air (i.e. 1Bqm^{-3}) and an annual dose due to radon inhalation has been formulated by the International Commission on Radiological Protection (ICRP) is given by [IC93]:

$$D_{\mathrm{Rn}}(\mathrm{mSva}^{-1}) = 53.576 \times C_{\mathrm{Rn}} \times F \times P \times B \times S, \qquad (2.10)$$

where,

53.576 = the decay alpha-energy of radon daughters in unit (mSvm^{3}Bq^{-1}a^{-1}) is evaluated as (5.56×10^{-9} JBq^{-1}) × the number of hours in a year (8760 hours) × 1.1 (mSv per mJhm^{-3}), which is the ICRP conversion convention for representative persons of the public [IC93],

C_{Rn} = the measured radon concentration in (Bqm^{-3}). The indoor concentrations were obtained using AlphaGuard and using a chi-square fit. The fitted concentrations from Figs. 4.10 to 4.15 have been used as an input for the assessment of the indoor dose. Outdoor concentrations were assumed to belong-normally distributed with the mean value equal to the average outdoor concentration, measured by RGMs and the standard deviation is a factor of 0.4 of this average. These assumed outdoor concentrations from Figs. 4.21 to 4.26 were used as inputs for the assessment of the outdoor dose,

F = the equilibrium factor calculated as the ratio of EEC (Bqm^{-3}) /C_{Rn} (Bqm^{-3}),

P = the relative occupancy indoors per annum,

B = the relative breathing rate and,

S = the fractional duration of the winter period. The fractional duration of the summer period is then expressed by $(1 - S)$.

More than 10 years of surveys has shown that the arithmetic mean of the world-wide indoor radon concentration is about 40 Bqm^{-3} [IC87]. Although higher ventilation rates could be expected at equatorial latitudes, which in general reduce the indoor concentration of radon, the results obtained indicate that other factors were apparently not considered. The average concentration of indoor radon in various countries as a function of latitude has been shown in Fig. 2.4 [IC87]. It can be seen that there is very little correlation with latitude.

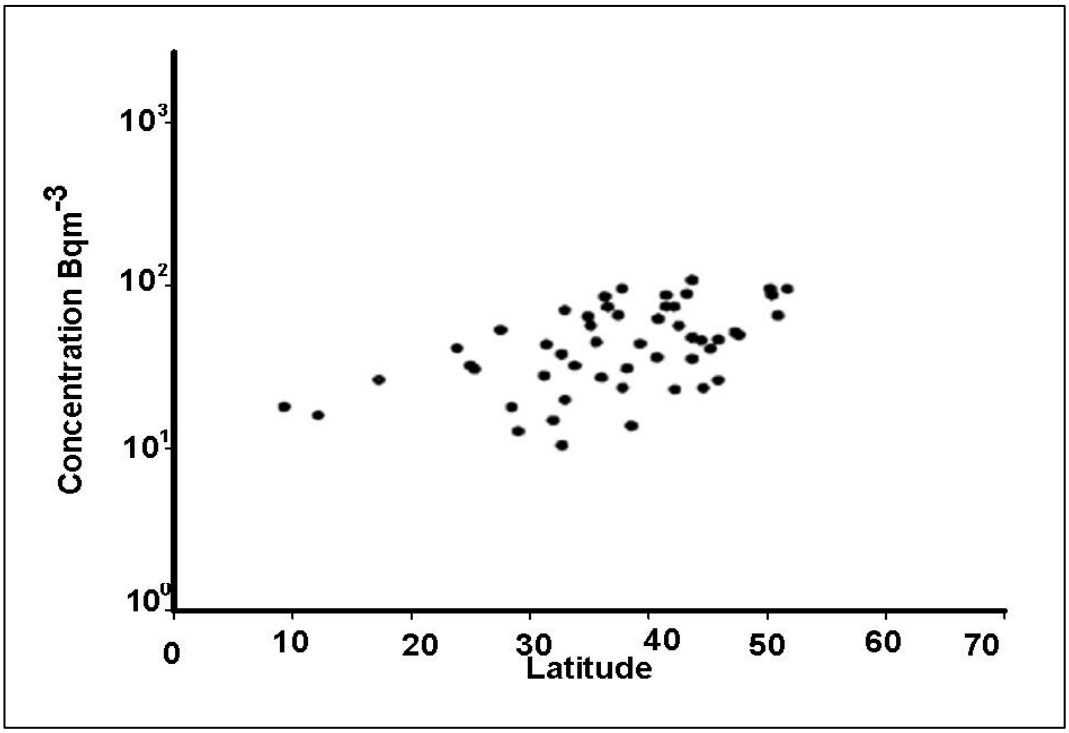


Figure 2.4 Average concentration of indoor radon in various countries in which measurements were made as a function of latitude [IC87].

The ICRP reaffirms that "*radon exposure in dwellings is due mainly to unmodified concentrations of ^{226}Ra in the earth's crust*". In addition, ICRP's protection policy for these situations is "*based on setting a level of annual dose of around 10 mSv from radon where action would almost certainly be warranted to reduce this exposure. As such the ICRP has reduced the upper value for the action level for radon gas in dwellings from the 2007 recommendations of 600 Bqm^{-3} to 300 Bqm^{-3}*" [IC09]. It is instructive to mention in this connection that in the UK the National Radiation Protection Board (NRPB) action level for indoor radon in houses occupied by members of the public is 200 Bqm^{-3} while in Germany this level is increased 250 Bqm^{-3}. National authorities should consider setting lower Bqm^{-3} reference levels according to local circumstances. All reasonable efforts should be made to reduce radon exposures to below the national reference level [WH09]. It is noted that the World Health Organization now recommends an approach similar to the ICRP [WH09]. It should be noted that, action levels related to the annual mean concentration of radon in a building and it is important to take actions which reduce the radon exposures inside the buildings [IC93]. The

maximum indoor concentrations of ^{222}Rn in a number of countries are given in Table 2.2.

In some countries new houses are required to be ^{222}Rn leak proof as far as possible. This will require that builders employ the latest technologies for keeping the homes safe from high ^{222}Rn concentrations and thereby ensuring that the ^{222}Rn level is as low as reasonably achievable.

Mining activities in South Africa have recently become a matter of great public debate and study. Uranium in South Africa is usually a by-product of gold, platinum or copper mining. The Nuclear Fuels Corporation of South Africa (Pty) Limited (NUFCOR) was established to process the uranium-rich slimes dams (wet tailing (debris) from the processing of gold-bearing ore) from gold mining. After political changes in 1994 more access to information has been allowed compared to the past and simultaneously the growth of public environmental awareness has risen enormously in South Africa [RA08]. A number of mining areas have high natural background levels of uranium, which is the source of the radium isotope ^{226}Ra. There is, however, a problem of data reliability, and hence there is a need for direct measurement of radon concentrations and assessment of doses due to radon [WA08]. For this reason, more fundamental investigations aimed at radon monitoring and identifying the mechanisms that determine the entry of ^{222}Rn into dwellings are needed.

Table 2.2 Comparison of the maximum indoor concentrations of ^{222}Rn decay products in houses occupied by members of the public.

Country/Organization	**Upper limit ^{222}Rn existing houses (Bqm^{-3})**	**Upper limit ^{222}Rn new houses (Bqm^{-3})**	**Reference**
Germany	200	-	[RI87]
Sweden	400	70	[RI87]
U.K(NRPB)	200	50	[RI87]
Canada	370	-	[RI87]
U.S.A.	200	-	[NC84]
ICRP	200	100	[IC93]
WHO	400	100	[WH87]

2.6 Dose assessment tools

The effect of radiation on organisms depends on the amount of energy absorbed from the radiation. Here, the absorbed energy per unit mass is termed the absorbed dose, *D*, with the corresponding unit the gray (Gy) and is defined as [NI83]:

$$1Gy = 1\ J\ kg^{-1}. \qquad (2.11)$$

In general, the effects of exposure to radiation are divided into deterministic and stochastic effects. When a significant part of the body is exposed to high doses (>1Gy) in a short time, deterministic effects will occur. These effects manifest very complex symptoms (radiation syndrome), which may result in death. Except for rare accidents, exposures in practices are restricted to low doses at a low-dose rate of radiation of natural or artificial origin. The most important detriment of these so called stochastic effects is inducing cancer. Stochastic effects have a latency period which can last several decades and, in general, a linear dose-response relationship with no threshold has been adopted as a pre-cautionary

measure. This means that even low doses of ionizing radiation exhibit a finite probability of producing an effect [IC07].

There has been a lot of research devoted to the effects of radiation dose and a large portion of this research has been carried out on animals. However, the relationship between the dose and radiation damage obtained is not directly transferable to humans. Indeed, epidemiological studies such as for example, the effect of atomic bomb explosions on Japanese people or patients who were exposed to radiation for therapeutic reasons or miners who were exposed to high levels of ^{222}Rn daughter concentrations in air would better reflect the relationship between the dose and its effect [BE80].

It has been established from radiobiological studies that different types of radiation, for the same absorbed dose (in Gy), have different biological effects. For low levels of radiation exposure, these differences are taken into account by a radiation weighting factor W_R, which is related to the ionization density of the radiation: for γ and β-radiation $W_R = 1$, and for α-radiation $W_R = 20$ [IC07]. A quantity which gives a direct relation to the probability or radiation damage by stochastic effect is the dose-equivalent, quantified in sieverts (Sv):

$$H = D \times W_R. \tag{2.12}$$

ICRP calls the value of radiation weighted dose the equivalent dose, denoted by letter *H*. The unit of the equivalent dose is again the sievert(Sv). All recommendations for dose limits are given in units of sievert. The equivalent dose H_T (Sv), in tissue or an organ *T*, is defined by the ICRP as [CE83]:

$$H_T = \sum_R W_R \times D_{T,R}\,, \tag{2.13}$$

where $D_{T,R}$ is the absorbed dose, expressed in grays (Gy), averaged over the tissue or organ *T* due to radiation *R*, and W_R is defined as the Radiation Weighting Factor (see Table 2.3).

Because the sensitivity of the organs and tissues to radiation also varies, and because the irradiation of organs is usually not uniform, a tissue weighting factor (W_T) has been introduced. This factor is defined as the fraction of the total stochastic risk due to the exposure of organ T (see Table 2.4) [IC07]. Stochastic risk is the probability of occurrence of a radiation induced health effect, which is greater for a higher radiation dose and the severity of which (if it occurs), is independent of dose. Stochastic effects may be somatic effects or hereditary effects, and generally occur without a threshold level of dose. Examples include cancer and leukaemia [IC07].

Tissue weighting factors have been established from several human epidemiology sources, for example, from the studies of the atomic bomb victims in Japan. The effective dose D_{eff} in (Sv) is defined as the absorbed dose weighted by radiation type and tissue or organ sensitivity [CE83]:

$$D_{\text{eff}} = \sum_i H_T W_T, \tag{2.14}$$

where,

W_T = weighting factor for tissue T and

H_T = dose equivalent to tissue T.

The sievert (Sv) is, therefore, a special name for the SI unit of equivalent effective dose, and operational dose quantities in radiation protection. Its physical unit is Joules per kilogram (Jkg^{-1}). Doses in Gy are multiplied by a quality factor which depends on the type of radiation and particular detriment in affected tissue, to obtain sieverts [IC93].

Lungs may be divided into the bronchial and alveolar regions. The bronchial epithelium is largely at risk due to exposure by ^{222}Rn daughters. Half of the weighting factor for lungs (W_T = 0.12 see Table 2.4) has been recognized for each region of the lungs [JA88]. The effective dose due to irradiation of the bronchial region is therefore obtained by using:

$$W_\alpha \times W_{\text{lung}} / 2 = 1.2, \tag{2.15}$$

where W_α is the radiation weighting factor for α radiation and has a value $W_\alpha = 20$ (see Table 2.3).

Table 2.3 Comparison of Radiation Weighting Factors in ICRP 103 and 60 [IC90][IC07].

W_R	**ICRP 60**	**ICRP 103**
Photons (all energies)	1	1
Electrons and muons (all energies)	1	1
Neutrons (all energies)	Step function	Continuous function
< 10 keV	5	2.5
10 to 100 keV	10	2.5 to 10
100keV to 2 MeV	20	10 to 20
2 to 20 MeV	10	7 to 17.5
> 20 MeV	5	5 to 7
Proton energy **>** 2 MeV	5	2
Alpha particles, fission fragments	20	20

Table 2.4 Comparison of Tissue Weighting Factors in ICRP 103 and 60 [IC90] [IC07].

Organ or tissue (*T*)	**Weight factor (W_T) ICRP 60**	**Weight factor (W_T) ICRP 103**
Gonads	0.20	0.08
Breast	0.05	0.12
Red bone marrow	0.12	0.12
Lung	0.12	0.12
Thyroid	0.05	0.04
Bone surface	0.01	0.01
Colon	0.12	0.12
Stomach	0.12	0.12
Bladder	0.05	0.04
Oesophagus	0.05	0.04
Liver	0.05	0.04
Brain	-	0.01
Kidney	-	-
Salivary Glands	-	0.01
Skin	0.01	0.01

Chapter 3

Measurement of radon and monitoring instrumentation

3.1 Overview of instruments used for the measurement

In order to take continuous measurements of radon concentration in air, a sensitive instrument is required which also should be suitable for installation in a dwelling without interfering with the daily life of the residents. Such detectors can be active e.g. AlphaGuard or passive like the CR-39(RGM), which is also used in mining applications.

3.1.1 Active radon monitors

Continuous radon monitoring includes instruments that record, in real-time, the concentrations of radon. Air is either pumped or diffuses into a counting chamber. The counting chamber is typically a scintillation cell or an ionisation chamber. Scintillation counts are processed by electronics, and radon concentrations for programmed intervals can be stored in the instrument's memory or transmitted directly to a printer.

Some advantages of active radon monitors are [GM11]:

- The continuous radon monitor has the ability to measure short term variations as well as to integrate radon measurement in time. Most continuous radon monitors, as a minimum, integrate over an hour, but some models can be set to integrate anywhere from 1 minute to 1 week.

- some continuous radon monitors have a digital readout, which can be set to display the current radon level or the long term average radon level.
- many models of continuous radon monitors are equipped with other environmental sensors to simultaneously measure and record other parameters like ambient temperature, barometric pressure and relative humidity.
- most active radon monitors have the ability to collect and store the measured data. These data can then be downloaded and used for the interpretation of the radon measurements.

Some disadvantages of active continuous radon monitor are [HR11]:

- many of these radon monitors require a trained, skilled operator and
- active continuous radon monitors need to be regularly calibrated.

3.1.2 Passive radon monitors

A passive radon monitor is a small piece of special plastic or film inside a small container. The air being tested moves through a filter, covering the entrance in the container. The purpose of the filter is to remove the dust and particles on which radon daughters might have attached. When alpha particles from radon and its decay products strike the plastic they cause damage tracks. The container is sealed after measurement and returned to the laboratory for reading [DU03].

Some advantages and disadvantages of passive radon monitors are [HR11]:

- they are convenient and economical,
- can be used for long-term measurements,
- are unobtrusive and make no noise,
- do not require any power,
- can be obtained through the mail and
- one of the disadvantages of passive radon monitoring is that long measurement periods are necessary.

3.2 Description of instruments used for the monitoring of radon

As indicated in Section 3.1, two techniques are available for measuring radon concentration. The techniques are integrative, continuous active radon sampling (AlphaGuard) and passive radon sampling (RGMs). AlphaGuard is a radon concentration monitor that uses an ionisation chamber inside and detects the alpha particles emitted from radon gas and RGMs are based on Solid State Nuclear Track Detection (SSNTD) technique. The details of the both instruments are discussed on Sections 3.2.1 and 3.2.2.

3.2.1 AlphaGuard

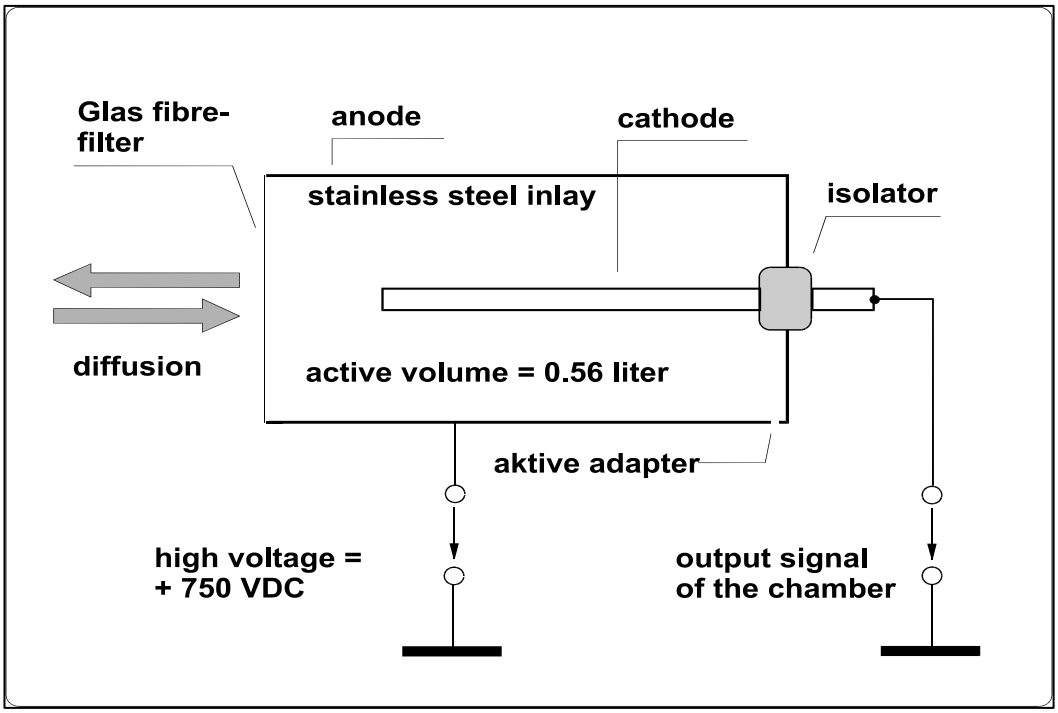
The AlphaGuard was designed specifically for monitoring the radon concentration in air and is an active radon monitor. AlphaGuard is a portable, battery- or mains-operated radon monitor with high storage capacity. In addition to the radon concentration in air, AlphaGuard also simultaneously measures and records ambient temperature, relative humidity and atmospheric pressure with its integrated sensors. Furthermore, tampering is also recorded (acceleration sensor) and up to two external analogue signals can be processed [GM11].

By combining the monitoring of radon with these associated environmental parameters it is possible to draw valid conclusions regarding the time and spatial distribution of the radon gas. This is of significant benefit for mitigation actions of radon entry into houses. AlphaGuard incorporates a pulse-counting ionisation chamber (alpha spectroscopy) [GM11].

Figure 3.1 shows the alpha spectroscopy system based on an ionisation chamber for quantifying and digitalizing the chamber output signal inside the AlphaGuard. Different field strengths between the charge-collecting electrodes affect the gas ionisation detectors characteristics. There is a relationship between the pulse size produced and the potential applied across the electrodes of gas ionisation chamber

exposed to alpha, beta and gamma radiation. The field strength and also the type of radiation which enters the detector volume and produces ions can affect the pulse size. When there is low field strength, there is still opportunity for the many slowly migrating ion pairs to recombine. This recombination condition is not used for radioactivity detectors. Ionising radiation creates more ions and electrons at the electrode when there is higher voltage. Ultimately, the required field strength is reached at the point that rapidly migrating ions do not have a chance to recombine. Therefore, a diffusion region is reached so that all the ions formed move straight from radiation event, and are collected at the electrodes. An additional increase of field strength cannot attract more ions since all of them have already been collected. Ion chambers operate in this region. The ionisation effects of the incident radiation can be shown by the number of charges collected at the electrodes. The type of radiation and also information about radiation energies can be modified and provided by ion chambers. Ion chambers are very useful in radiation dosimetry because the output signal is directly related to the ionisation effect. Alpha-particles produce a great amount of ions along a short path length of travel (high linear energy transfer). They are easily detected because they provide a high output signal [EL98].

Through optimal geometry of the chamber and intelligent signal processing, this radon monitor is suitable for continuous monitoring of radon concentrations between 2 to 2,000,000Bqm^{-3}. The AlphaGuard has high detection efficiency and a large measurement range, fast response and is insensitive to both high humidity and vibrations. AlphaGuard can be used for both short or long-term measurements indoors as well as outdoors. It is a very versatile instrument can be use for the measurement of radon in air, soil, water and building material as it is pointed in Fig. 3.2 [GM11]. As stated above, AlphaGuard has high detection efficiency for ^{222}Rn, it is maintenance free for measurements extending up to more than 3 months, and it is able to respond quickly to variations in radon concentration. Figure 3.2, top left illustrates the use of AlphaGuard for measurements in the air, which can be inside or outside a home [CA09]. Figure 3.3 also shows a photo of an AlphaGuard used.



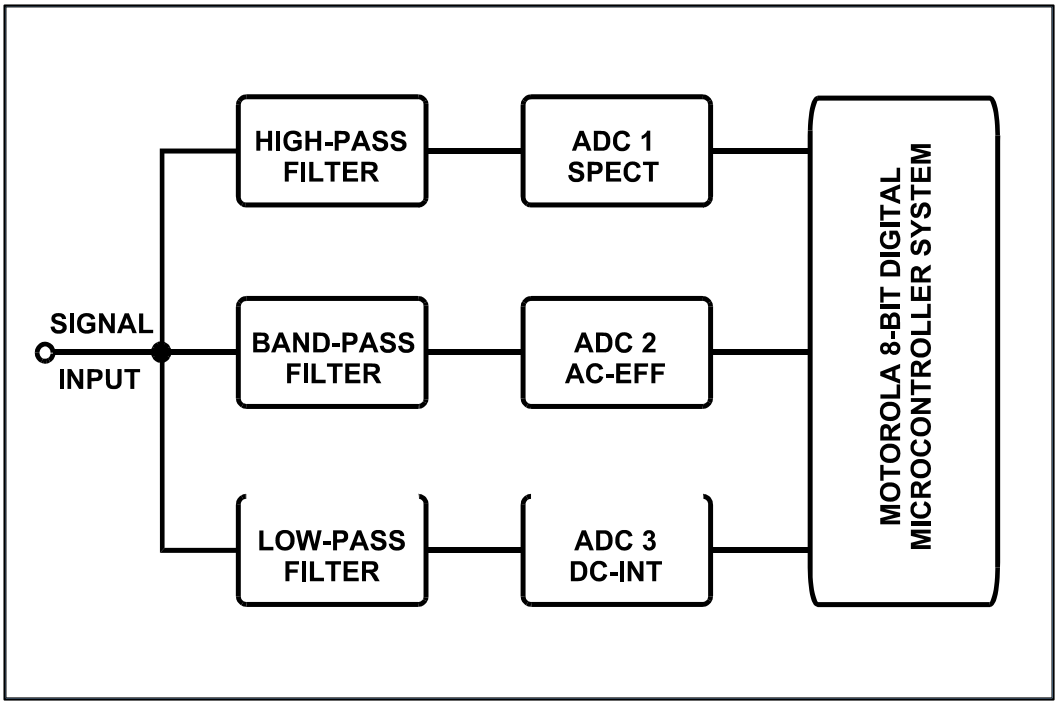


Figure 3.1 Alpha spectroscopy based on ionization chamber (upper) and quantifying and digitalizing the chamber output signal (lower) [GM11].

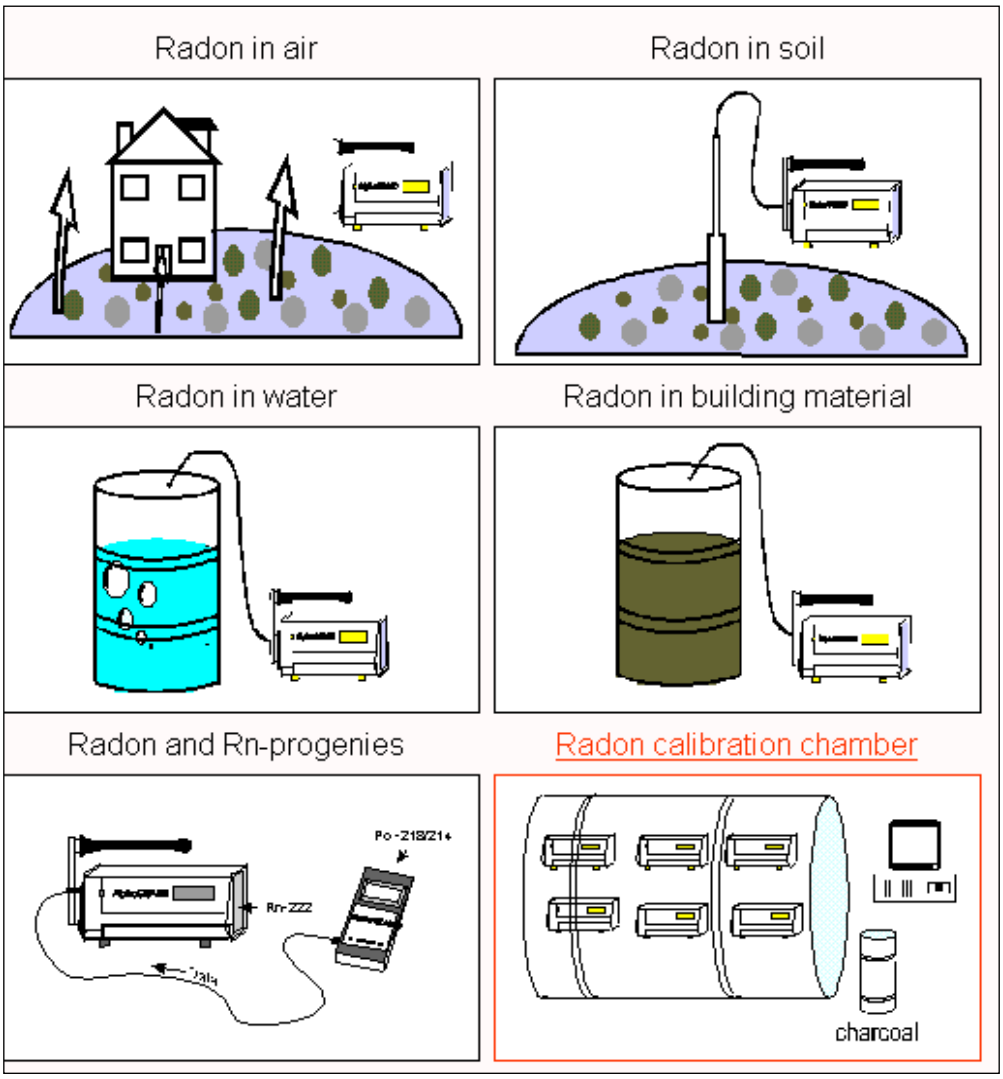


Figure 3.2 AlphaGuard can be used for radon measurement in air, soil, water and building material [CA09].



Figure 3.3 Photograph of an Alpha Guard radon monitor used in the present work.

3.2.2 Passive radon monitor (RGM cap)

There are different ways to detect radon concentrations in the air by passive monitors such as:

- solid state nuclear track detection
- absorption on activated carbon

one of these methods which has been approved by the NNR (National Nuclear Regulator) of South Africa, is Solid State Nuclear Track Analysis. Nuclear track analysis utilizes specific polymer materials sensitive to α-particles. CR-39 is such a material (plastic) and it is the same polymer which is used in plastic lenses.

Nuclear tracks are made visible by etching in concentrated hot NaOH(l) (see Figs. 3.4 to 3.6 and the number of tracks is directly proportional to radon concentration. When alpha particles come into contact with the plastic CR-39 material, the interaction is basically “burning” of the CR-39 material along the particle track, which develops visible spots in the plastic [DU03].

Each RGM is prepared by cutting the CR-39 material into blocks and applying a unique tracking number to each block (see Fig. 3.7). These are then placed in plastic holders with the same unique number on the outside, and are then ready for deployment (see Fig. 3.8).

The Radon Gas Monitor (RGM) is designed for underground conditions and is light weight and compact in size. It has a high sensitivity, its lower limit of detection for one month is 22 Bqm^{-3} or is $11Bqm^{-3}$ for 2 months. This monitor is not sensitive to gamma rays and is not influenced by high temperature and moisture conditions. It is suitable for long-term monitoring and low-cost radon concentration measurement in dwellings. As can be seen in Fig 3.8 the colour of the cover is black because the outer shell is made of carbon-loaded plastic, which reduces uncertainties caused by the build-up of an electrostatic charge. Once the RGM have been exposed to the atmosphere, it will interact with α-particles and the interactions will be made visible by etching.

To ensure consistent evaluation of monitoring, the plastic is treated by hot NaOH (l) and allowed to dry. The number of tracks is counted by using a computerized program, which scans the microscopic picture of tracks (see Fig. 3.9). As mentioned previously, each RGM monitor has its own unique number and the radon concentrations are given in Bqm^{-3}.



Figure 3.4 CR-39 detectors are chemically etched in NaOH solution [PA08].

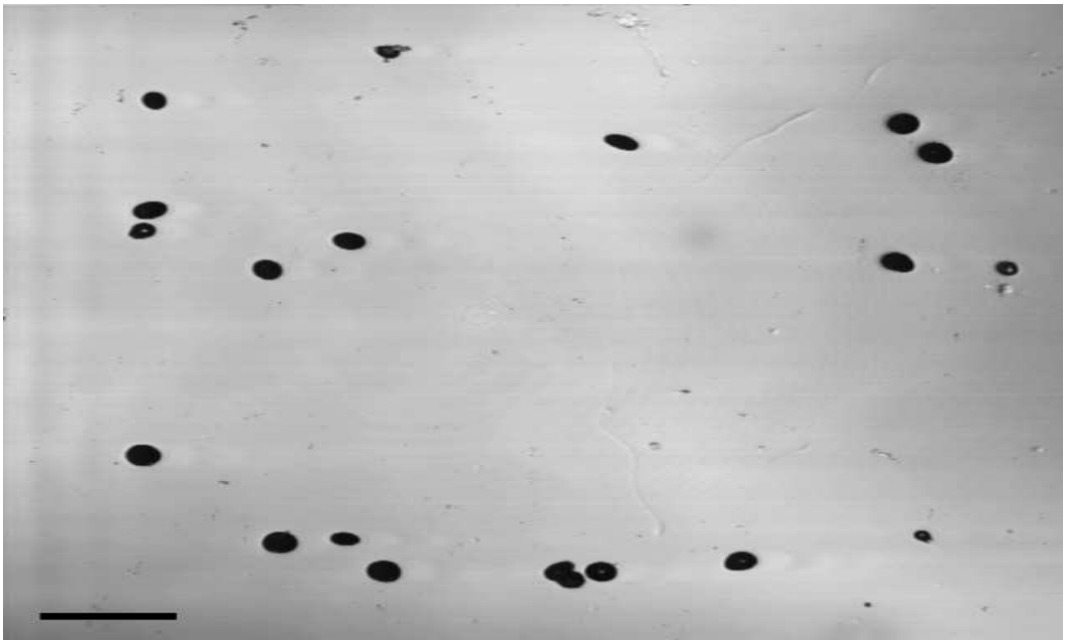


Figure 3.5 Nuclear tracks are made visible by etching in concentrated hot NaOH [PA08].

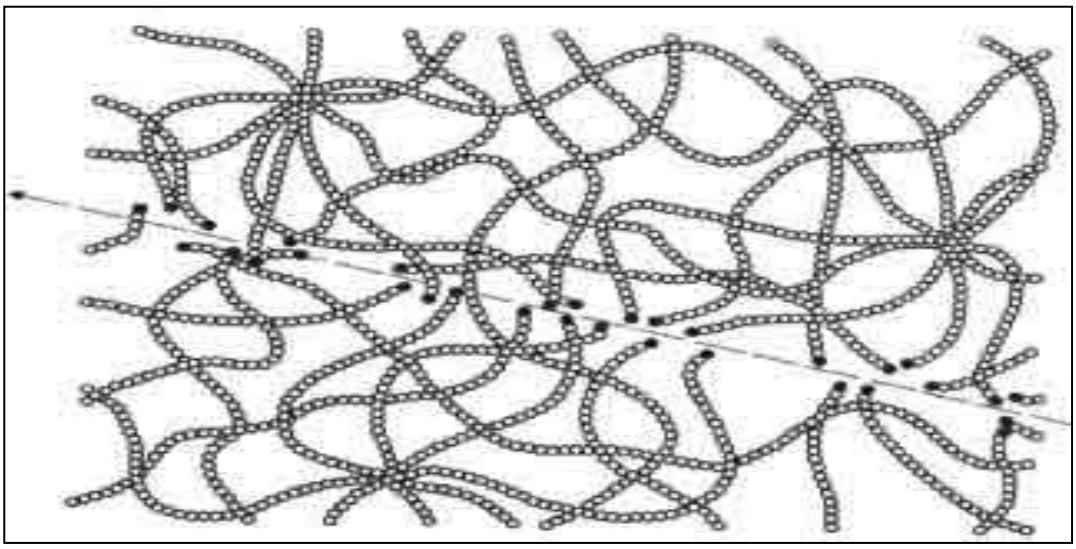


Figure 3.6 Path of an alpha particle from right to left breaking the CR-39 polymer chains [PA08].

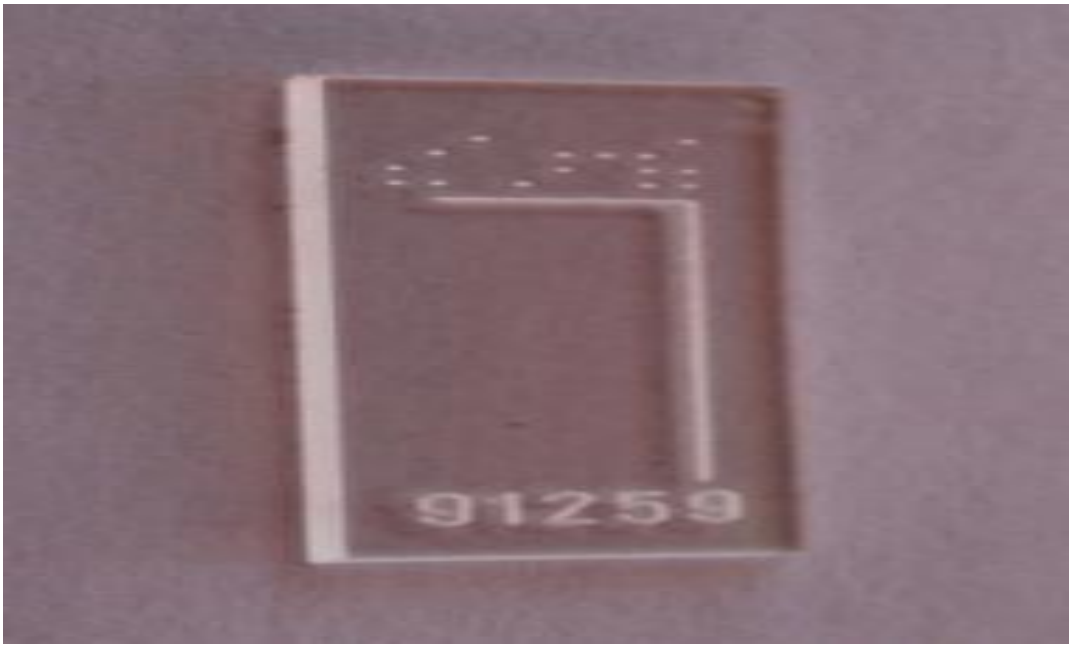


Figure 3.7 CR-39 material is cut into blocks and a unique tracking number is added [PA08].



Figure 3.8 Illustration of a radon cap (RGM).

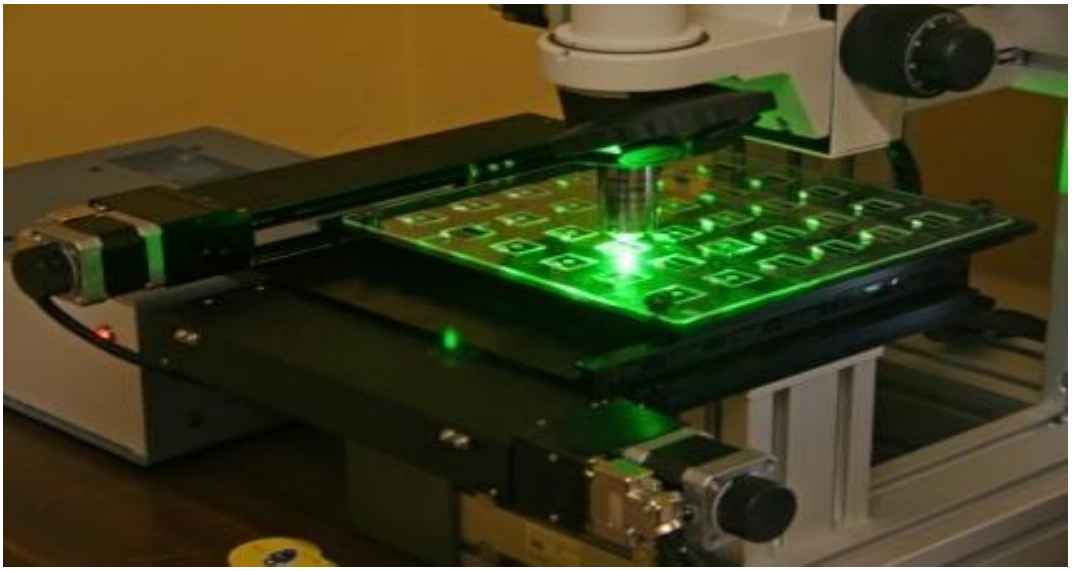


Figure 3.9 Reading device for the etched CR-39 plastic [PA08].

Chapter 4

Results of the measurements of radon concentrations

4.1 Introduction

Six sets of measurements were carried out from May to November 2011, around the vicinity of the First Uranium Ezulwini mine in the West Rand area of Johannesburg using an active radon monitor AlphaGuard (for details see Section 3.2.1) and passive radon monitors RGMs (for details see Section 3.2.2), for location see Fig. 4.1 at the end of Section 4.2.1).Because previous data were available from the Mine management it was decided that additional measurements should be obtained inside the selected houses, which were occupied by mine workers and their families. These selected houses are single family dwellings, built in the 1960's inside the mine area.

4.2 Measurements of radon concentration inside the selected houses using AlphaGuard

The AlphaGuard monitor was employed both for short-term as well as long-term measurements. The short-term measurements took 2 hours for outdoor measurements of radon. Long-term measurements of up to 3 weeks were used to determine the diurnal variations of radon concentration inside the selected houses. In addition, by using AlphaGuardthe variation of radon concentration and its correlation with meteorological parameters (e.g. temperature and humidity) would be investigated.

The houses have a rectangular floor design; they all have open kitchens, which are covered by the roof. The ground floor consists of the entrance hall and bedrooms with an adjacent bathroom. The bedrooms, storage room and bathroom are all on one floor. Ventilation can occur through the windows and through the front door. The walls of the houses are made of concrete bricks (breeze blocks in South Africa). A photograph of a typical house is shown in Fig. 4.2 at the end of Section 4.2.1.

An aerial photograph of the Ezulwini mine area is shown in the upper part of Fig 4.3 at the end of Section 4.2.1. Corresponding features are identified in the lower part of Fig 4.3.

4.2.1 Overview of the data collected indoors by AlphaGuard

The meteorological data (temperature, humidity and pressure) for the location of the selected houses along with the variation of radon concentration were recorded every hour by Alpha Guard. Measurements were taken in winter (May, June, July) and summer (October, November) 2011, inside the bedrooms, store rooms and living rooms of the selected houses. The living room was used in House # 1 and the bedroom was used in Houses # 4, 5 and 6. The store room was used in House # 3 (Golf-Club house) and House # 2. The radon concentration was recorded every hour in each house, over a seven day period. The raw data are presented in Figs. 4.4 to 4.9 at the end of this section. For the sake of comparison all of the figures for this section are presented at the end.

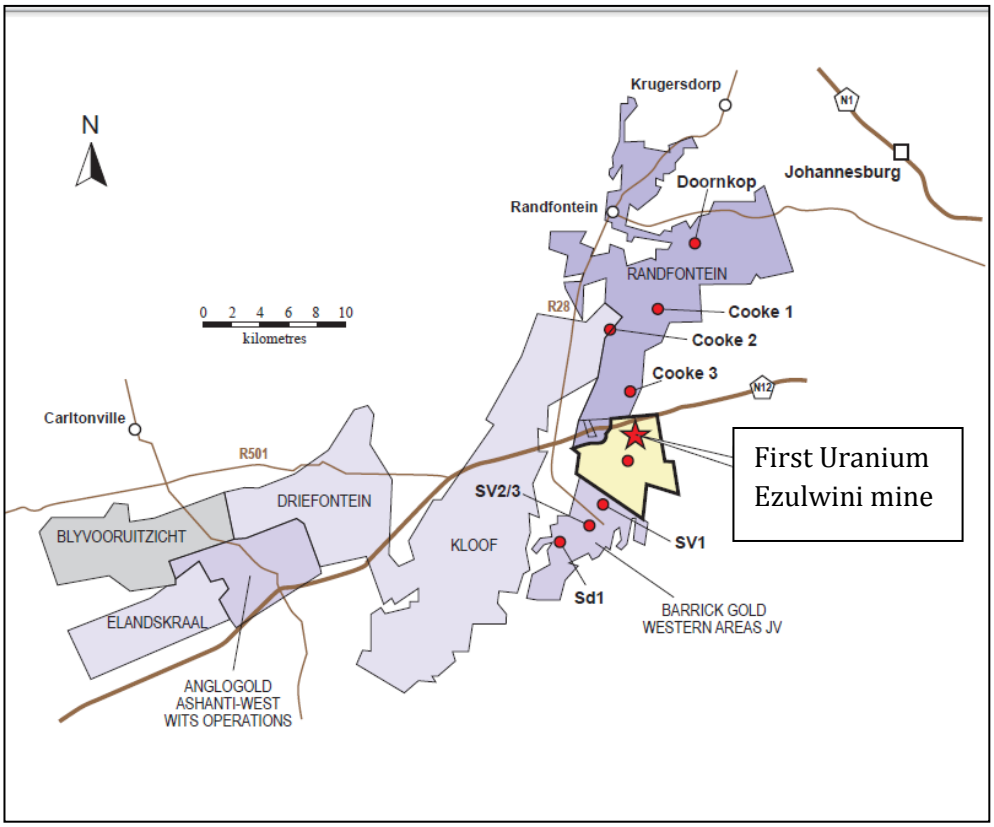


Figure 4.1 Map showing the positions of various mines South-West of Johannesburg including First Uranium Ezulwini mine.



Figure 4.2 Typical selected house around the mine.

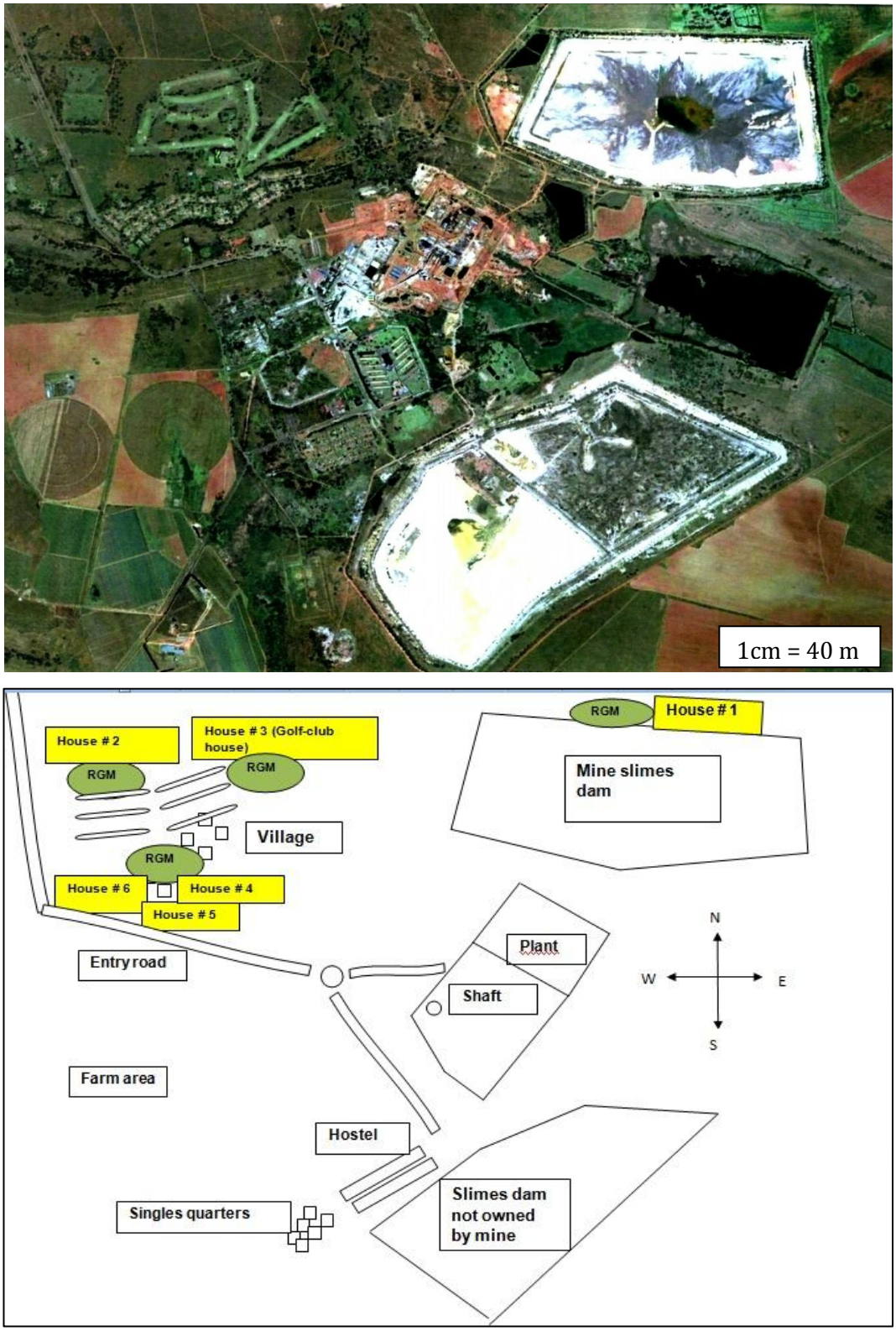


Figure 4.3 (Upper) arial photograph of the West Rand Elzuwini Mine (Lower) layoutof the mine with features as indicated.

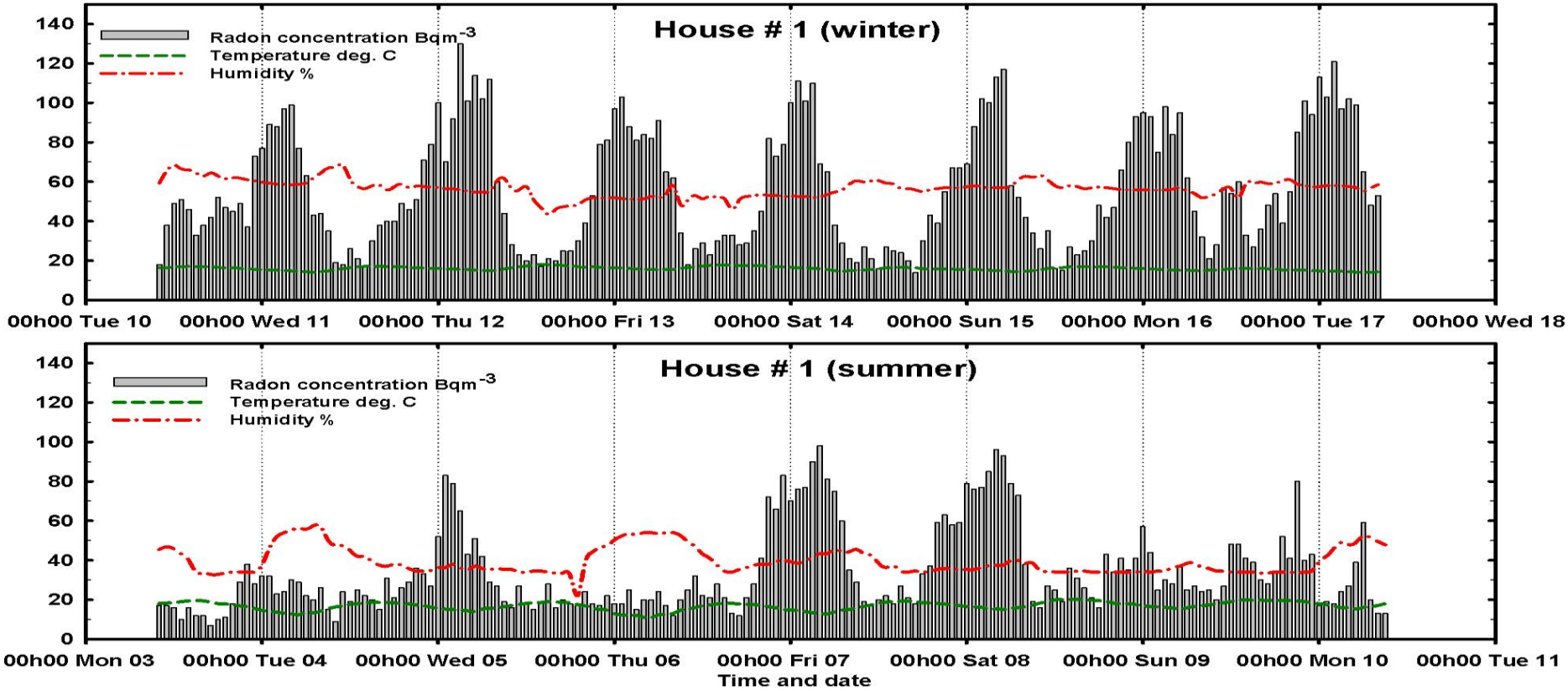


Figure 4.4 Histogram of the measured ^{222}Rn concentration together with the corresponding temperature and humidity (see legend for units) for winter (upper) and summer (lower) 2011 inside House # 1.

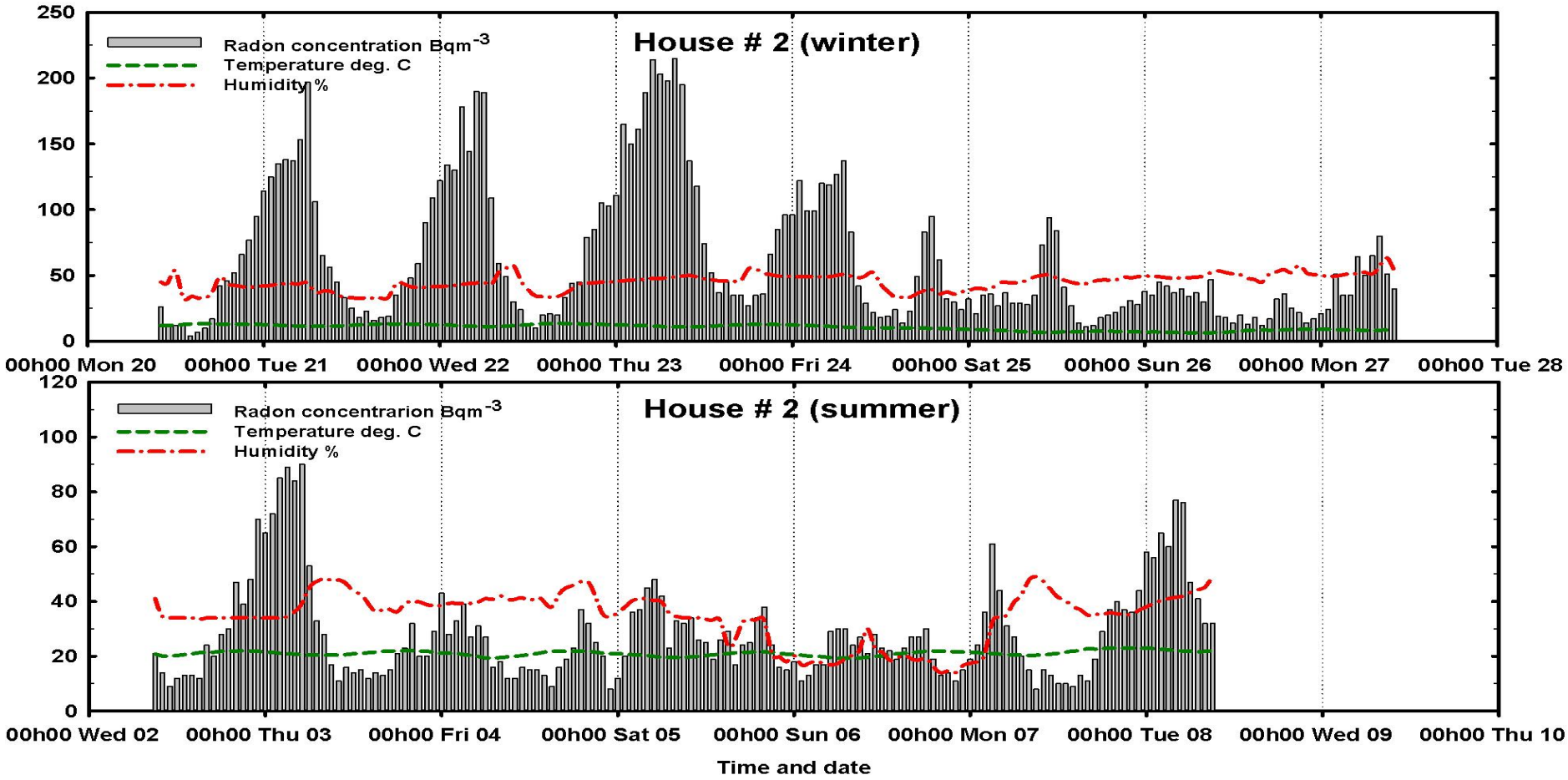


Figure 4.5 Histogram of the measured ^{222}Rn concentration together with the corresponding temperature and humidity (see legend for units) for winter (upper) and summer (lower) 2011inside the House # 2.

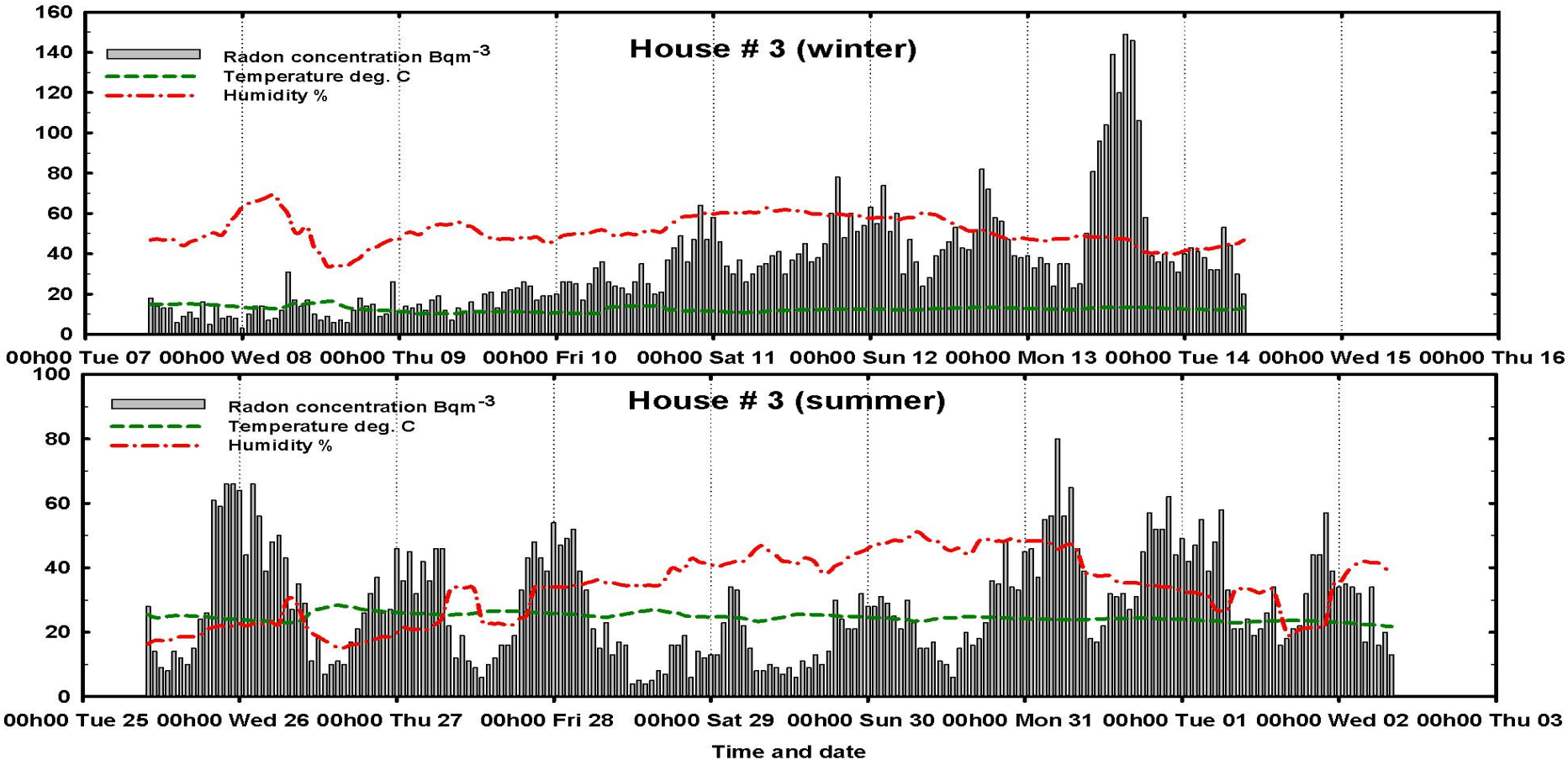


Figure 4.6 Histogram of the measured ^{222}Rn concentration together with the corresponding temperature and humidity (see legend for units) for winter (upper) and summer (lower) 2011inside the House # 3 (Golf-Club house).

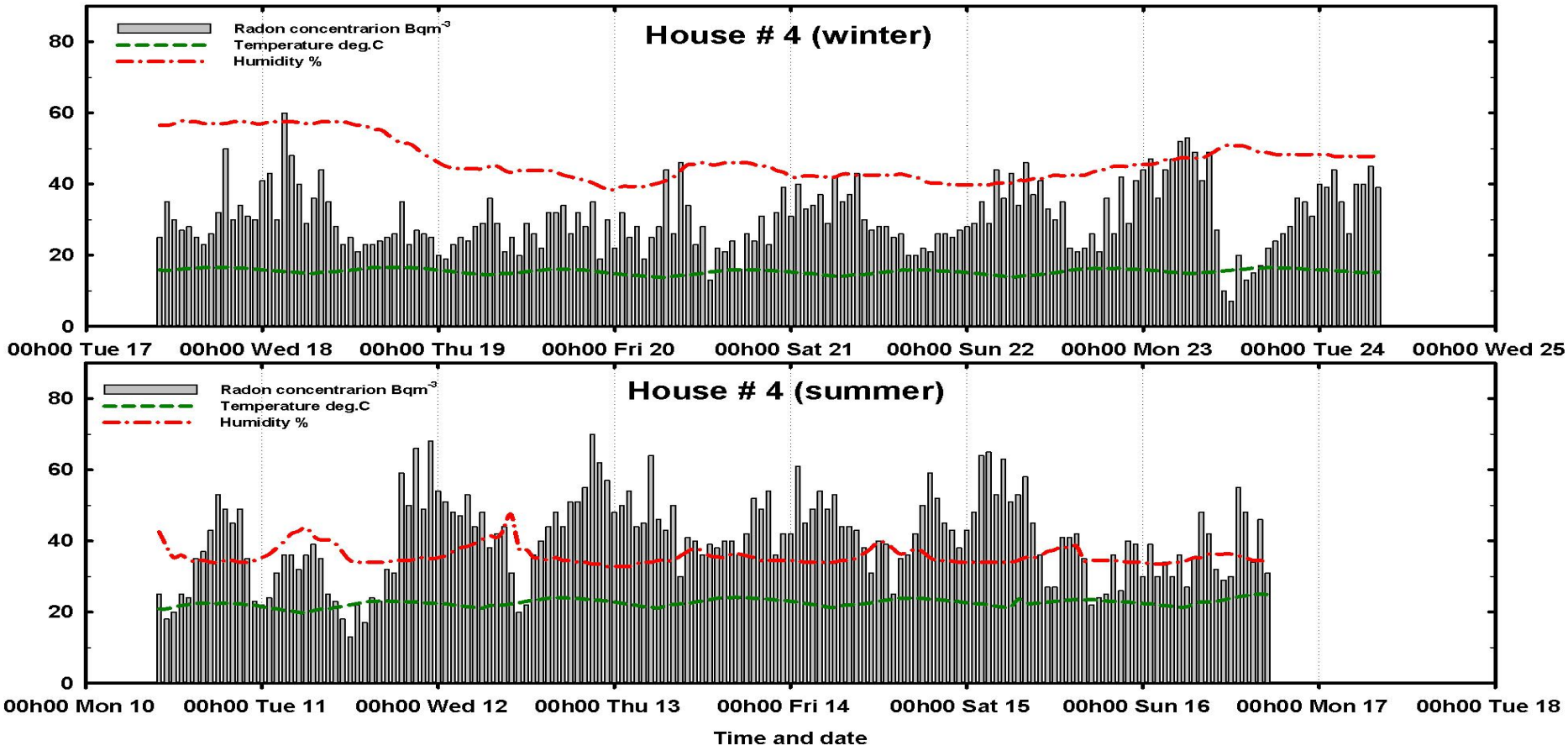


Figure 4.7 Histogram of the measured ^{222}Rn concentration together with the corresponding temperature and humidity (see legend for units) for winter (upper) and summer (lower) 2011inside House # 4.

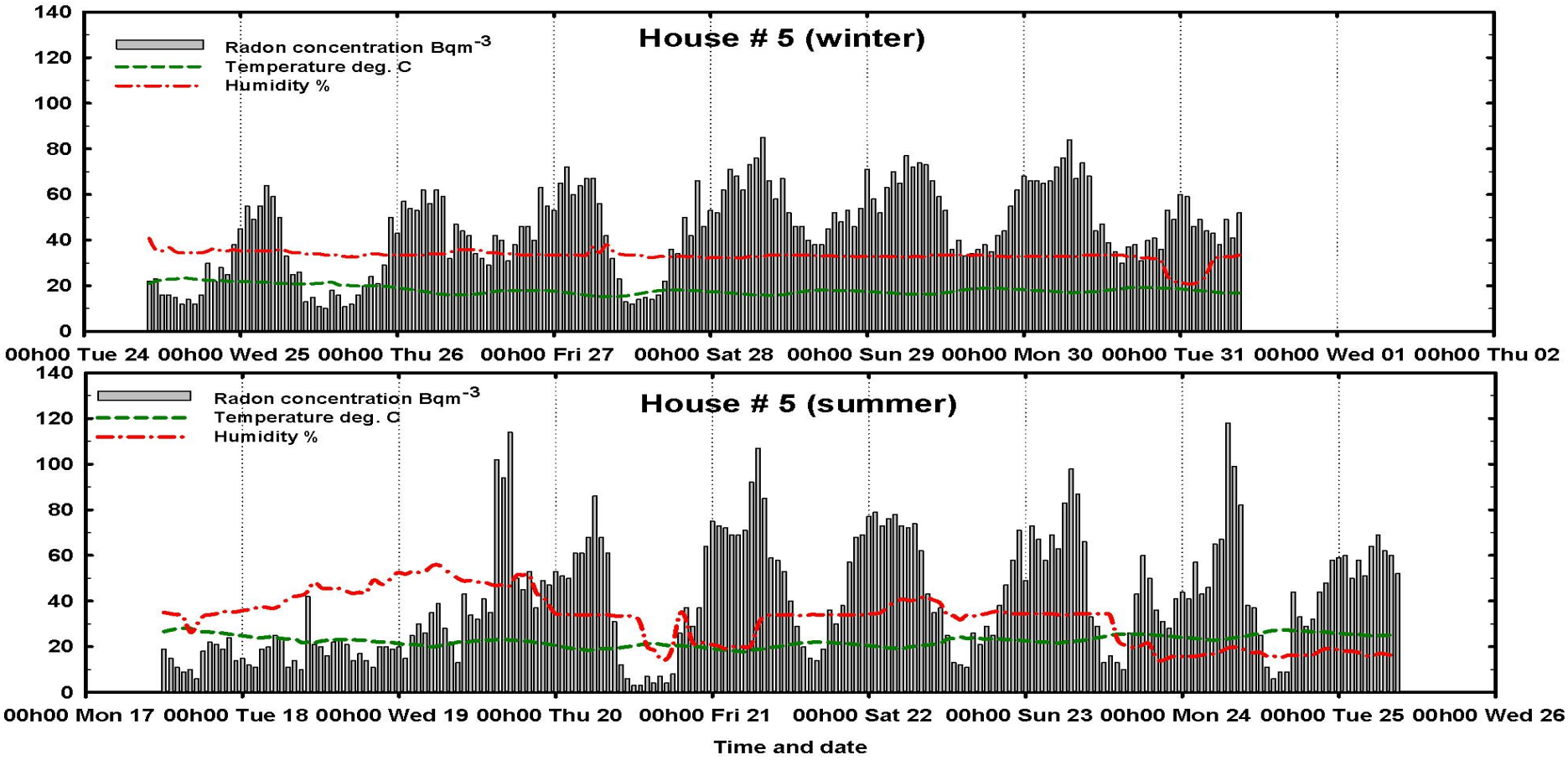


Figure 4.8 Histogram of the measured ^{222}Rn concentration together with the temperature and humidity (see legend for units) for winter (upper) and summer (lower) 2011 inside the House # 5.

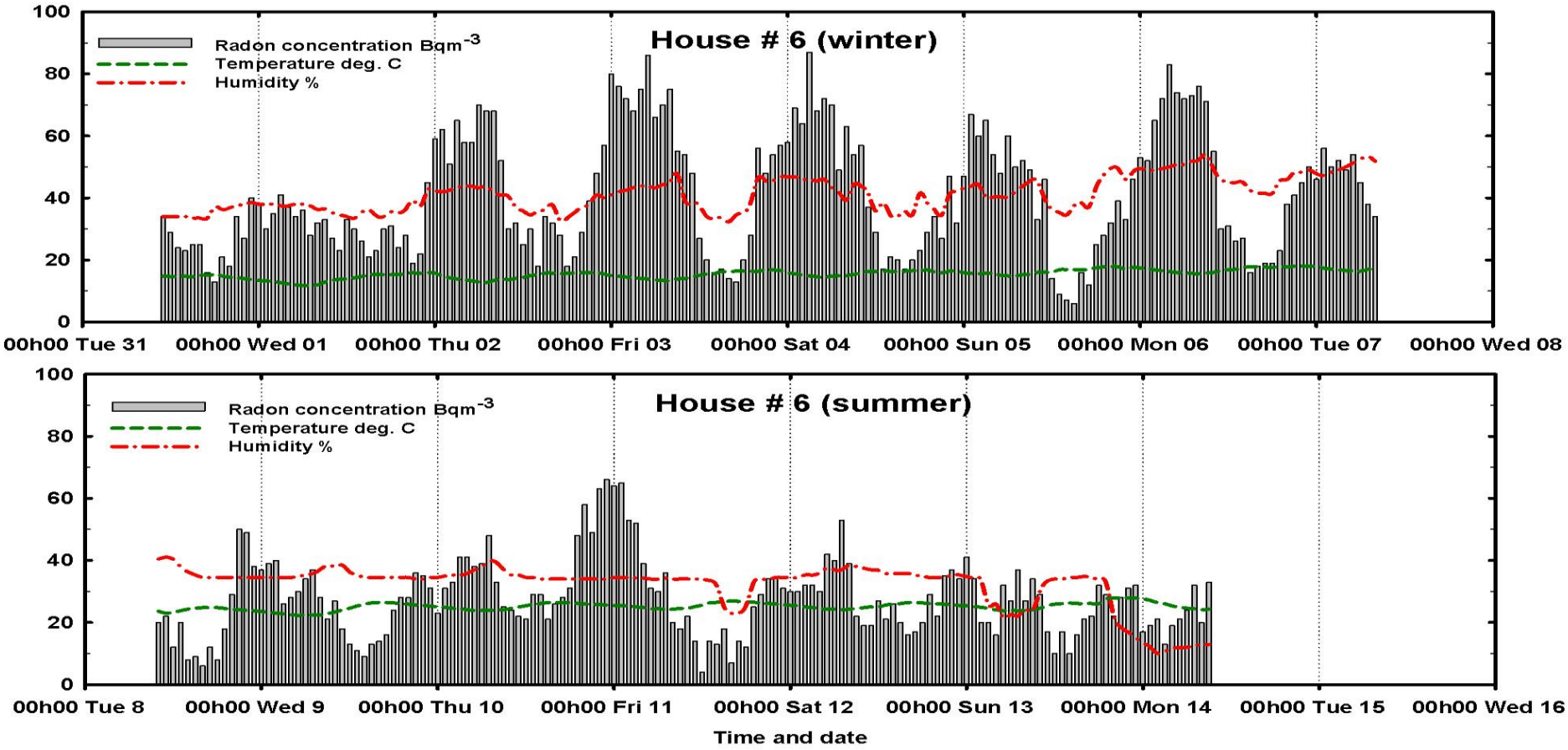


Figure 4.9 Histogram of the measured ^{222}Rn concentration together with the corresponding temperature and humidity (see legend for units) for winter (upper) and summer (lower) 2011 inside the House # 6.

4.2.2 Interpretation and discussion of the AlphaGuard results from inside the selected houses

Radon concentration in dwellings varies in different locations, because of different climate and geology, different building materials and also different domestic habits [IC93].

By way of example referring to the Fig. 4.3 one can notice that the first selected house (House # 1) is located in close proximity to the slimes dam where radon concentrations were measured in the living room. Also this house is near a lake with some farming taking place nearby. The measurements taken are presented in Fig. 4.4. Similarly, all of other houses selected can be located in the plan view of the area (Fig. 4.3, lower) and placed in relation to the environment diagram in the aerial photograph (Fig. 4.3, upper).

As a starting point, some general comments can be made about the recorded radon concentrations. As one can observe from the graphs in Figs. 4.4 to 4.9 the radon concentrations are generally lower in summer than winter time, most likely due to significantly reduced ventilation rate during winter compared to summer. It can also be seen from the figures how the meteorological parameters, such as temperature (and consequent change in humidity), influence radon concentration. Increase of temperature after sunrise results in a decrease of radon concentration and its progeny at the ground level, since the vertical mixing has increased [EU09]. A higher concentration at night and the early morning hours at the ground level, is most likely caused by lower ventilation compared to daytime, can also be seen from the figures. Also, solar heating during the daytime tends to induce some turbulence, so that radon is more readily transported upwards and away from the ground. During the night, the atmosphere is quite calm with low turbulence and little convection [EU09]. Also a higher temperature is related to the decrease of radon concentration. Temperature is an important factor, since the ground floor is usually heated because of the sunshine during the day, while at night the temperature decreases. As a consequence, there is a temperature difference

between the soil at a few centimetres depth and the floor surface. This leads to significant difference in radon emanation into the atmosphere [SC02].

It is worth noting that the variations in temperature and pressure, the nature of building materials and ventilation conditions can cause considerable irregular variation of the indoor radon concentration and its progeny [EU09, CH09]. It has been observed in this study that the atmospheric pressure has little or no effect on the indoor concentration of radon.

Ventilation is the process that involves the flow of air from one compartment to another and the exchange of indoor and outdoor air. Radon concentrations in the air in dwelling compartments cannot therefore be described without taking the ventilation pattern into account [EU09, CH09].

Turning now to specific observations, the average radon concentration in the House # 2 (in Tables 4.1 and 4.2) is higher than in the other selected houses. This house is positioned farthest from the slimes dam, so this high radon concentration can most likely be ascribed to the ventilation inside the store room or to the building material with which the house was built. In addition, House # 3 (Golf-Club house) has an elevated radon concentration during the weekend when the storeroom was not ventilated. As it can be seen in Fig 4.6, the highest concentration of radon in the House # 3 (Golf-Club house) histogram occurred in winter time on a Monday. The reason for this high concentration could be that the Golf-Club house is closed over the weekend, so from Saturday to Monday none of the doors or windows are opened, and there is almost no ventilation inside the house. Houses # 4, 5 and 6 exhibited similar concentrations (but different from House # 2 and 3).

Tables 4.1 and 4.2 give the summary of the average radon concentrations together with temperature and humidity during winter and summer time in the six houses during one week. The radon concentration ranged from 30 to 62 Bqm^{-3} in winter and 28 to 41Bqm^{-3} in summer over the measuring time of one week for each house. All results are well below the ICRP65 recommended minimum action level of 200 to 600 Bqm^{-3} [IC93].

Table 4.1 Average of radon concentration measured by AlphaGuard together with temperature and humidity during winter.

Location(Winter)	**Average radon indoor concentration (Bqm^{-3})**	**Temperature (˚C)**	**Humidity (%)**
House # 1	56	16	57
House # 2	62	10	45
House # 3 (Golf-Club house)	34	12	51
House # 4	30	15.5	47
House # 5	45	18	33
House # 6	41	15.5	41

Table 4.2 Average of radon concentration together with temperature and humidity during summer.

Location (Summer)	**Average radon indoor concentration (Bqm^{-3})**	**Temperature (˚C)**	**Humidity (%)**
House # 1	34	17	39
House # 2	28	21	34.5
House # 3(Golf-Club house)	28	25	34
House # 4	40	23	36
House # 5	41	23	32
House # 6	28	25	32

By way of comparison, a similar study was undertaken in Ghana, where high concentrations (up to 580 Bqm^{-3}) were found [CH09]. Elevated radon concentrations were also observed in Kenya, indicating an average of 170 ± 40 Bqm^{-3} [CH09].

It should be noted that radon levels are on the whole low which is same of surprising since the village in near the uranium mine near the slimes dam. As such the results should be seeing as a positive finding.

4.2.3 Frequency distribution of the radon concentrations measured by AlphaGuard inside the selected houses

The hourly records of the radon concentrations measured by AlphaGuard were approximated by a log-normal distribution using Chi-square fit. The obtained frequency distributions of the measured radon concentration inside the houses are shown in Figs. 4.10 to 4.15. The reason for fitting the measured radon concentrations was to simplify the quantification of various percentiles of radon concentrations. The best fit was obtained for the relative standard deviation of 0.4 of the mean.

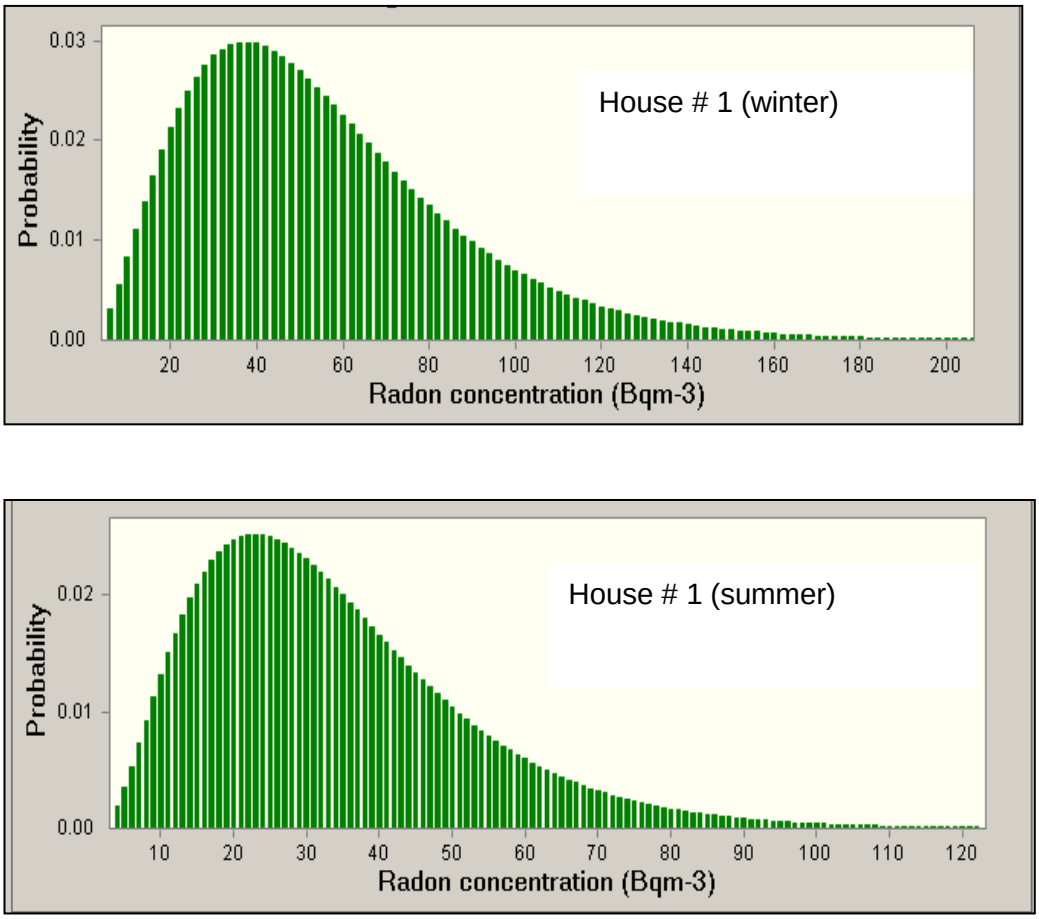


Figure 4.10 The frequency distributions of radon concentration inside House # 1 during the winter (upper) and summer (lower) periods.

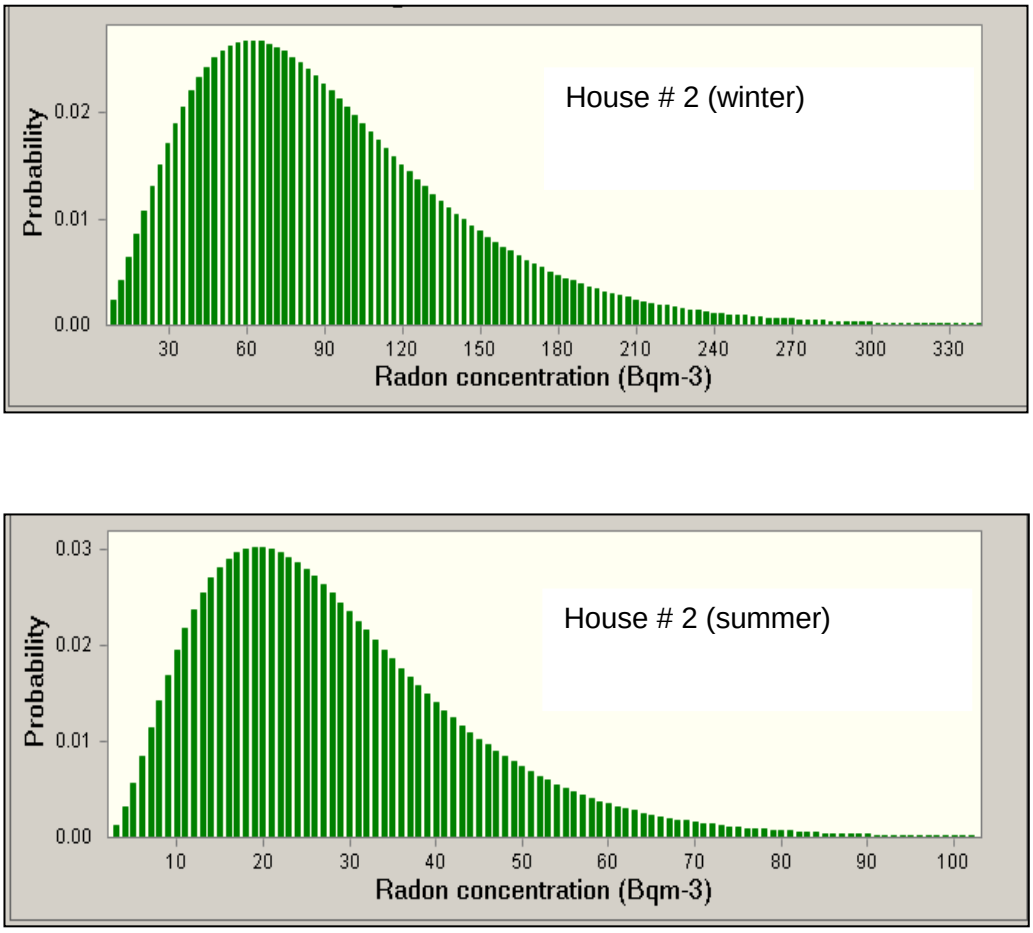
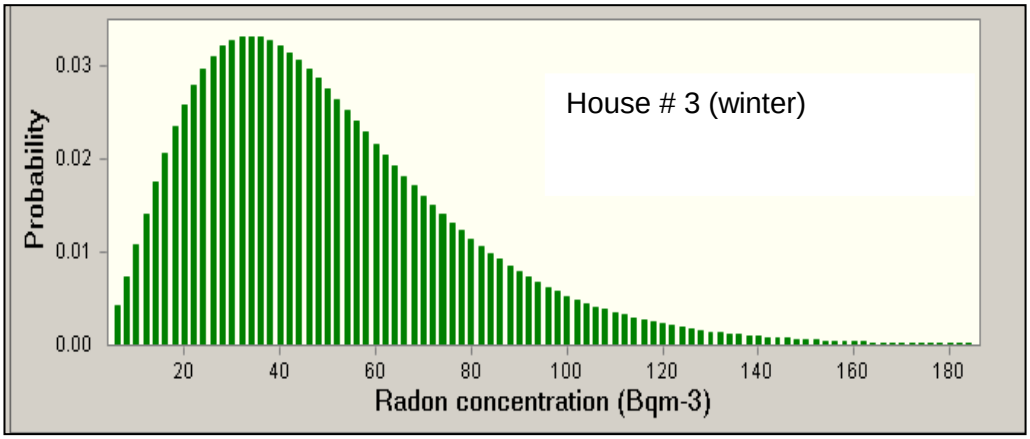


Figure 4.11 The frequency distributions of radon concentration inside House # 2 during the winter (upper) and summer (lower) periods.



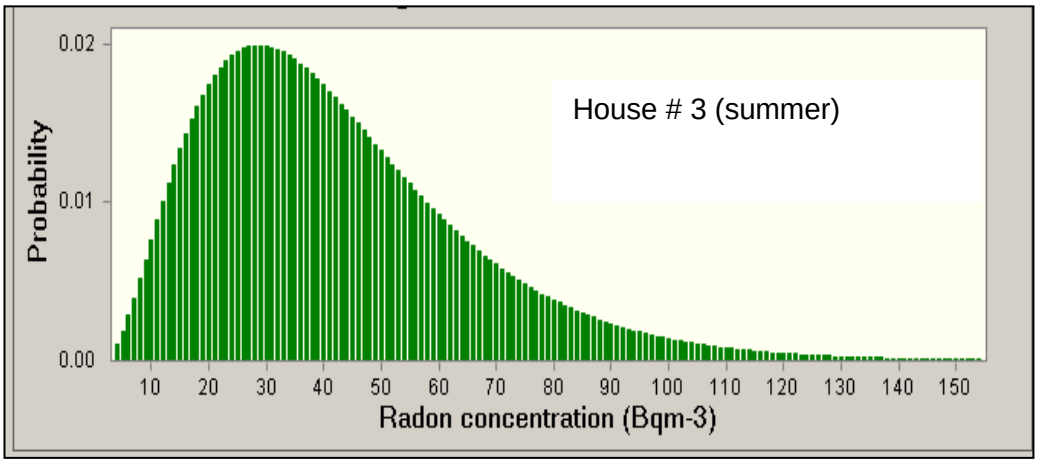
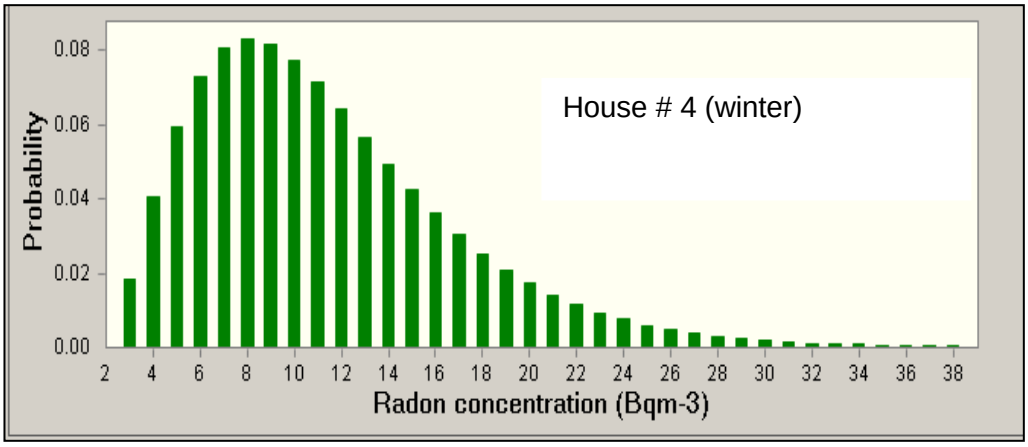


Figure 4.12 The frequency distributions of radon concentration inside House # 3(Golf-Club house) during the winter (upper) and summer (lower) periods.



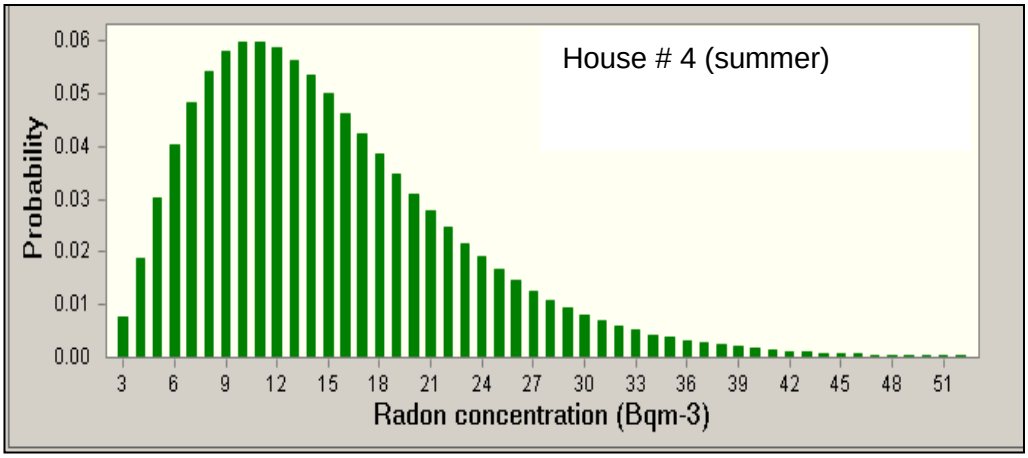
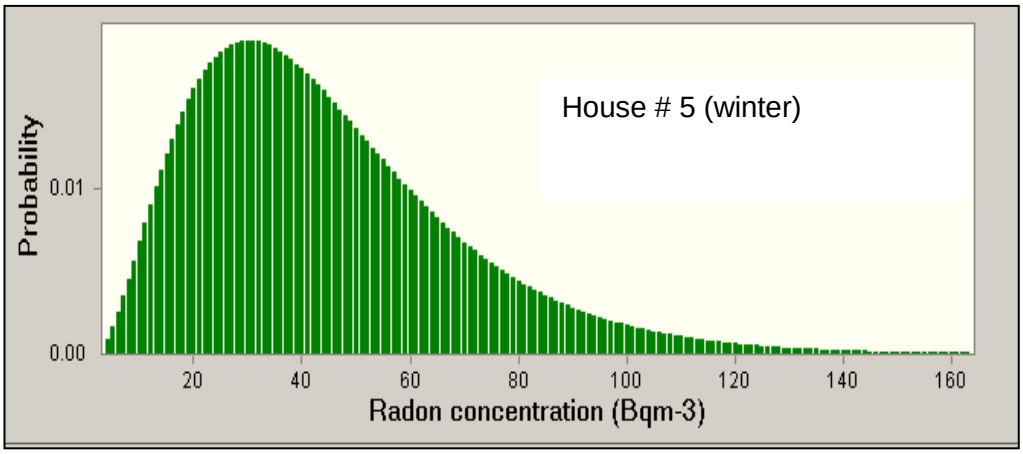


Figure 4.13 The frequency distributions of radon concentration inside House # 4 during the winter (upper) and summer (lower) periods.



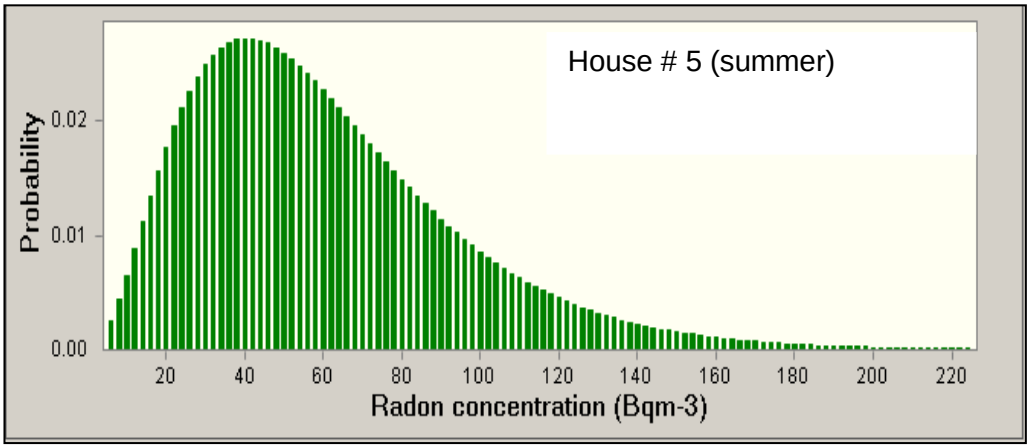


Figure 4.14 The frequency distributions of radon concentration inside House # 5 during the winter (upper) and summer (lower) periods.

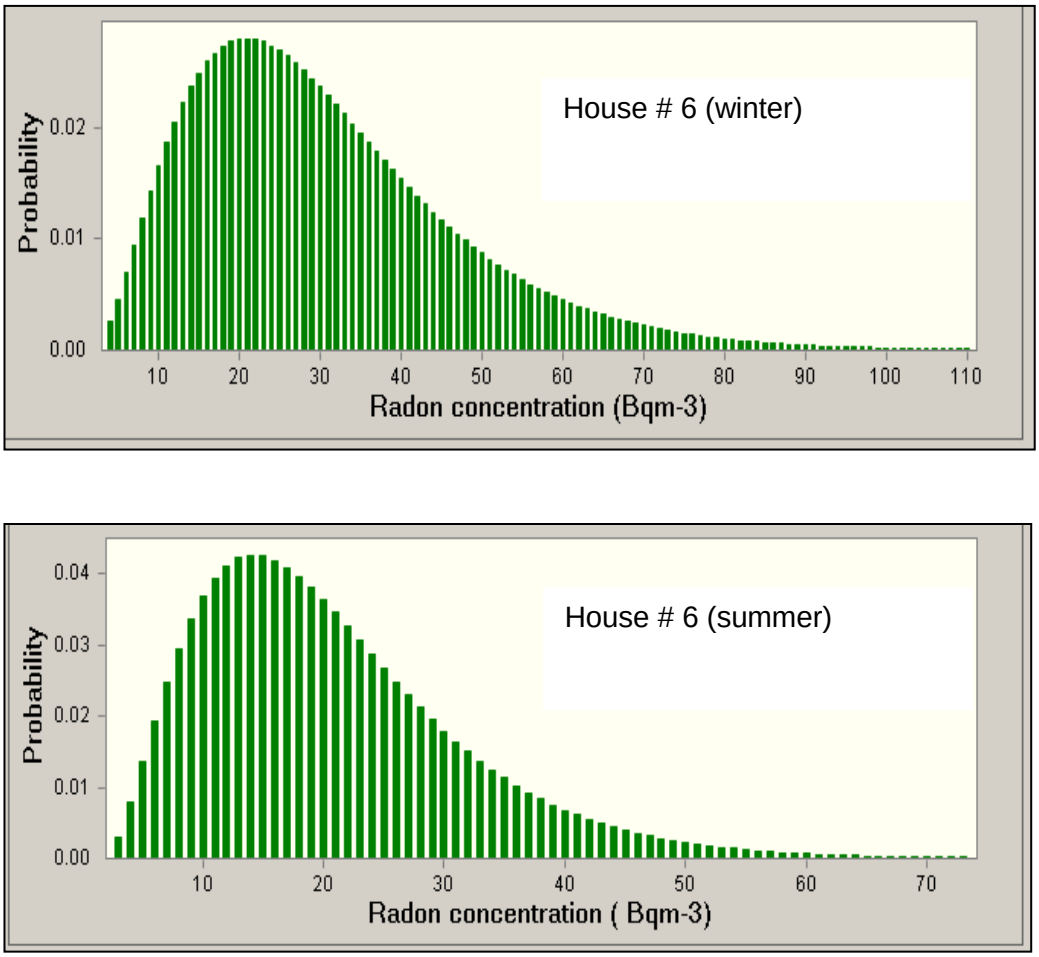


Figure 4.15 The frequency distributions of radon concentration inside House # 6 during the winter (upper) and summer (lower) periods.

4.3 Measurements of radon concentration indoors and outdoors in the selected houses using RGMs

Passive radon monitoring using CR-39 plastics in RGMs for the measurement of radon concentration in dwellings and soil has been used since in early 1990's [NI83]. They do not need to be connect to electricity and can be deployed under field conditions and in remote locations; they can be inaccessible for long period of time without the need for monitoring or any backup, except for initial placement and final recovery [DU03].

4.3.1 Overview of the data collected indoors and outdoors by the RGMs

Six RGM caps were installed for two months around the points where the Alpha Guard has been used. The results of the measurements using RGMs indoor during the winter and summer periods are shown in Tables 4.3 and 4.4. The results from RGMs were obtained from the laboratory of Parc RGM Company in South Africa, which has been accredited by the National Nuclear Regulator (NNR). For the sake of comparison all the tables and figures have been shown at the end of this section.

Both active (AlphaGuard) and passive (RGM) radon monitors were also used for monitoring of the outdoor radon. This monitoring lasted for two months for RGMs outside each of the selected house but only 2 hours for the AlphaGuard (see Table 4.5) in order to protect the instrument from theft.

Although the data are limited it is also useful to compare indoor concentrations measured using RGMs and AlphaGuard (see Tables 4.6 and 4.7 and also Figs. 4.16 and 4.17). As a comparison of the equivalent measurement techniques the results of a study which was done in Krakow, Poland [DM99] is shown in Fig 4.18.

In addition to the data obtained using the RGM technique indoors and outdoors, it is possible also to make a comparison between the use of AlphaGuard both indoors and outdoors albeit for a short measurement period of two hours outdoors. Because of the short time interval of two hours for measurements outside, the concentrations for the corresponding time interval at the same time of the day was extracted from the continuous seven days measurements interval as shown in Figs. 4.4 to 4.9 and the average for each house was determined. The results are given in Table 4.8 and are shown in Fig. 4.19.

In addition, Fig. 4.20 illustrates the consistency between outdoor and indoor radon concentrations measured by RGM in the winter (upper) and summer (lower) periods. As can be seen, clustering in these graphs is very close together, showing generally that the indoor concentration is larger than the outdoor concentration of radon (dashed line indicates a one to one correlation).

Table 4.3 Indoor average radon concentrations measured by the RGMs in winter period (April-June-Aug) and the following year in summer (Feb –April).

Location, indoors (Two months)	**RGM April-June (Bqm^{-3})**	**RGM June-Aug (Bqm^{-3})**	**RGM Feb-April (Bqm^{-3})**
House #1	36	44	30
House # 2	48	72	52
House # 3(Golf-Club house)	32	48	26
House # 4	26	49	21
House # 5	21	44	26
House # 6	27	43	30

Table 4.4 Outdoor average radon concentrations measured by the RGMs in winter period (April-June-Aug) and the following year in summer (Feb –April).

Location, outdoors (Two months)	**RGM April-June (Bqm^{-3})**	**RGM June-Aug (Bqm^{-3})**	**RGM Feb-April (Bqm^{-3})**
House # 1	24	41	16
House # 2	37	48	36
House # 3(Golf-Club house)	29	42	17
House # 4	20	37	22
House # 5	44	48	30
House # 6	45	39	18

Table 4.5 Short time monitoring of radon concentrations outdoors during winter time using AlphaGuard (error value from manufacture calibration data).

Location, outdoors	**AlphaGuard measurement time (hours)**	**AlphaGuard results (Bqm^{-3})**
House # 1	2	15 ± %5
House # 2	2	46 ± %5
House # 3 (Golf-Club house)	2	12 ± %5
House # 4	2	8 ± %5
House # 5	2	9± %5
House # 6	2	34 ± %5

Table 4.6 Comparison of the average of the radon concentrations measured by RGMs (error value from Parc RGM Company) and AlphaGuard (error value from manufacture calibration data) during winter.

Location, indoor(winter)	**RGM (winter) two months (Bqm^{-3})**	**AlphaGuard (winter) seven days (Bqm^{-3})**
House # 1	44 ± %25	56 ± %5
House # 2	72 ± %25	62 ± %5
House # 3 (Golf-Club house)	48 ± %25	34 ± %5
House # 4	49 ± %25	30 ± %5
House # 5	44 ± %25	45 ± %5
House # 6	43 ± %25	41 ± %5

Table 4.7 Comparison of the average of the radon concentrations measured by RGMs (error value from Parc RGM Company) and AlphaGuard (error value from manufacture calibration data) during summer.

Location, indoor (summer)	**RGM (summer) two months (Bqm^{-3})**	**AlphaGuard (summer) seven days (Bqm^{-3})**
House # 1	30 ± %25	34 ± %5
House # 2	52 ± %25	28 ± %5
House # 3 (Golf-Club house)	26 ± %25	28 ± %5
House # 4	21 ± %25	40 ± %5
House # 5	26 ± %25	41 ± %5
House # 6	30 ± %25	28 ± %5

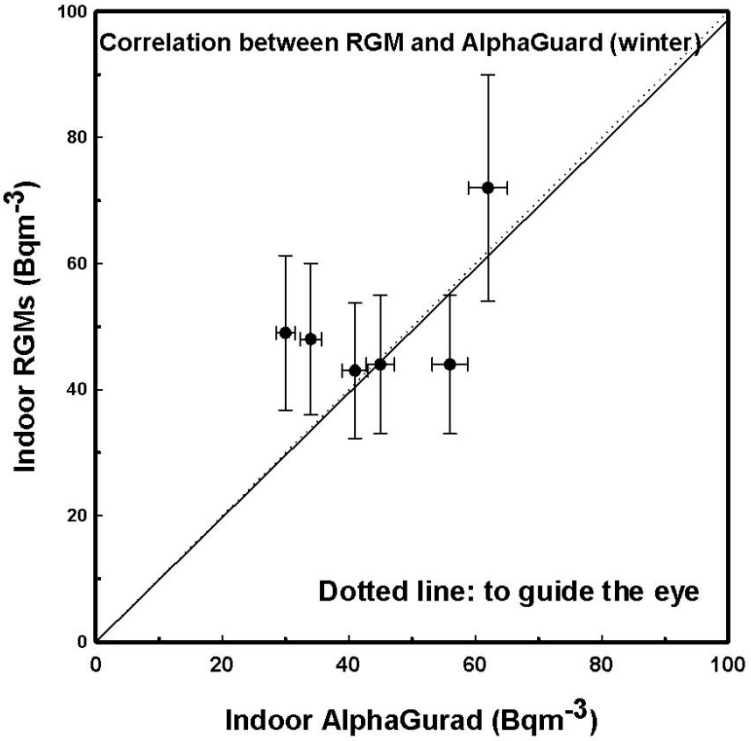


Figure 4.16 Correlation between RGM's and AlphaGuard results during the winter time.

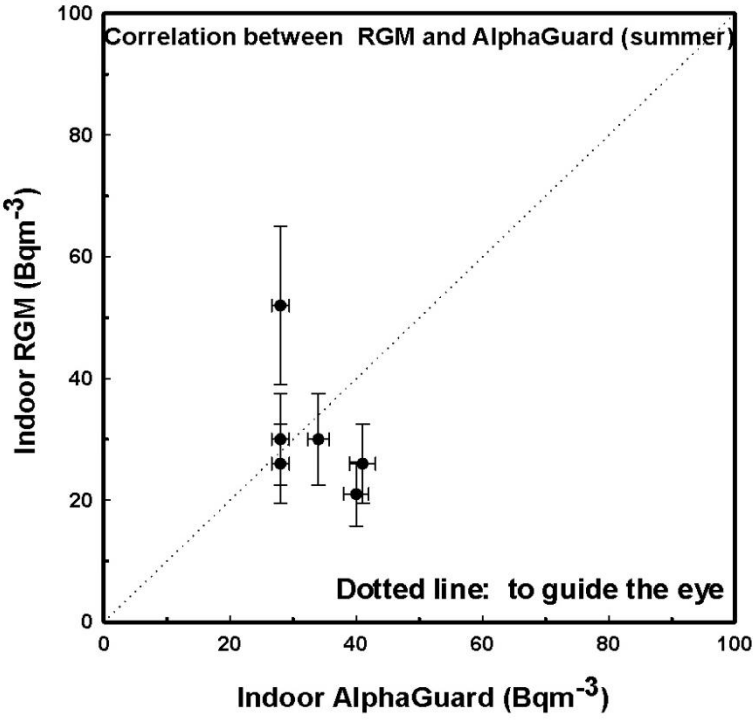


Figure 4.17 Correlation between RGM's and AlphaGuard results during the summer time.

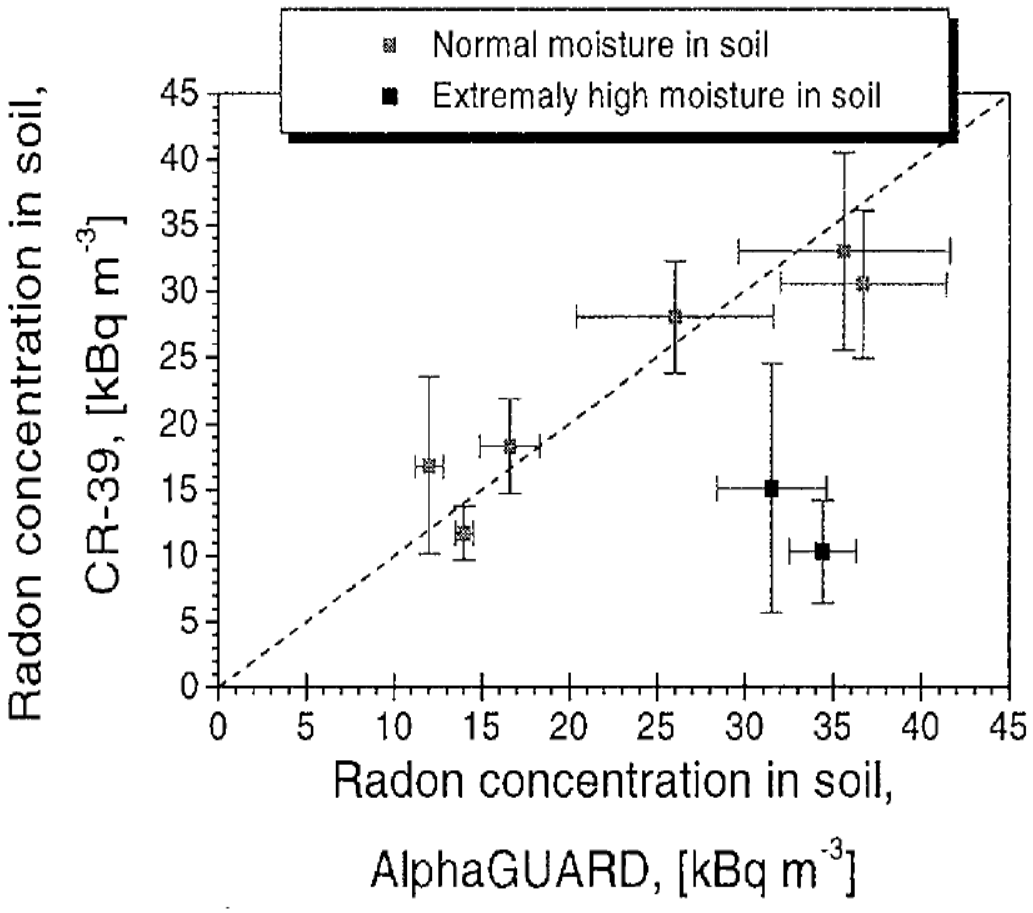


Figure 4.18 Comparison measurements of radon concentrations in soil using CR-39 and AlphaGuard (taken from Ref [MA99]).

Table 4.8 Comparison of outdoor and indoor results by AlphaGuard (error value from manufacture calibration data) during winter time for the six houses selected.

Location, winter	**AlphaGuard indoors two hours (Bqm^{-3})**	**AlphaGuard outdoors two hours (Bqm^{-3})**
House # 1	17.5 ± %5	15 ± %5
House # 2	60 ± %5	46 ± %5
House # 3 (Golf-Club house)	40.5 ± %5	12 ± %5
House # 4	22 ± %5	8 ± %5
House # 5	34 ± %5	9 ± %5
House # 6	53 ± %5	34 ± %5

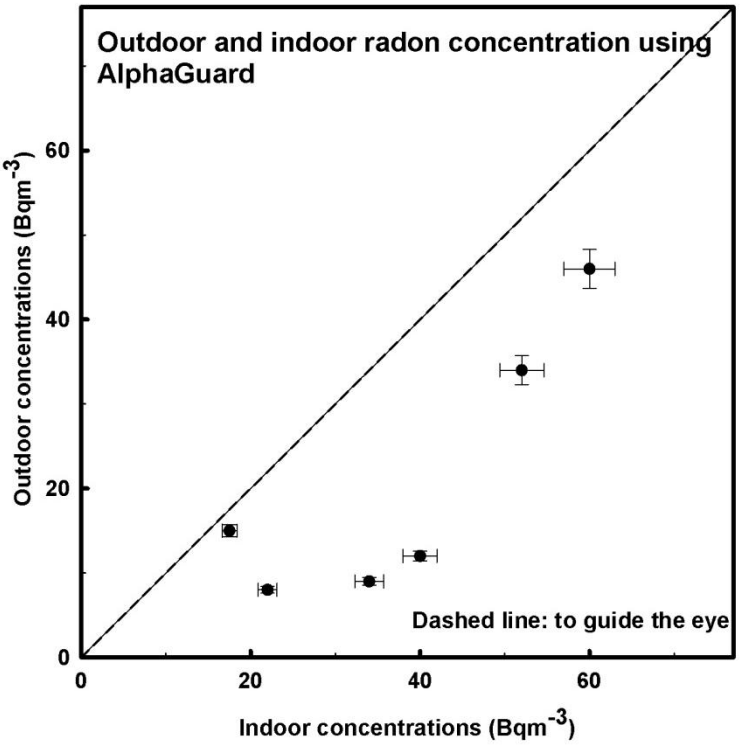
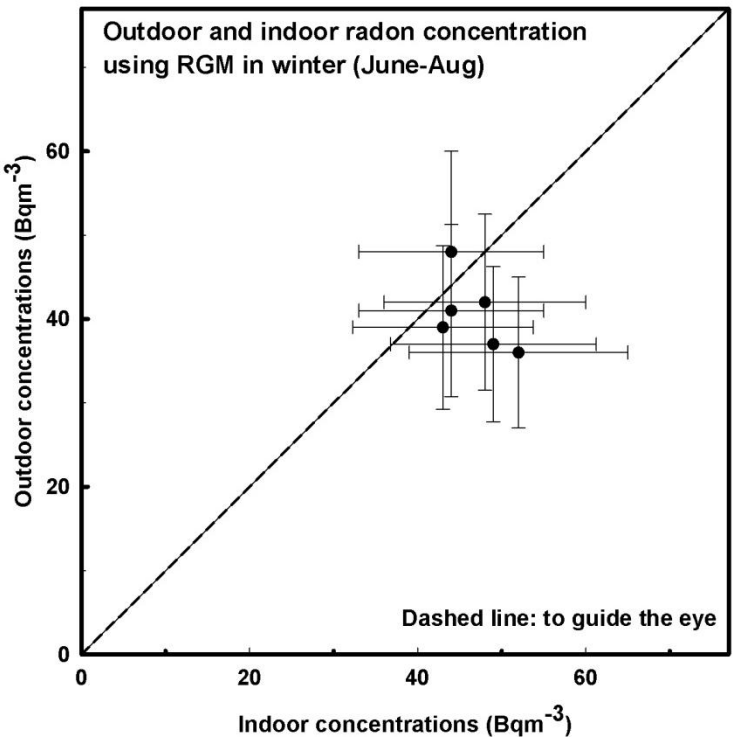


Figure 4.19 Comparison between outdoor and indoor radon concentration results measured by AlphaGuard for the six houses selected.



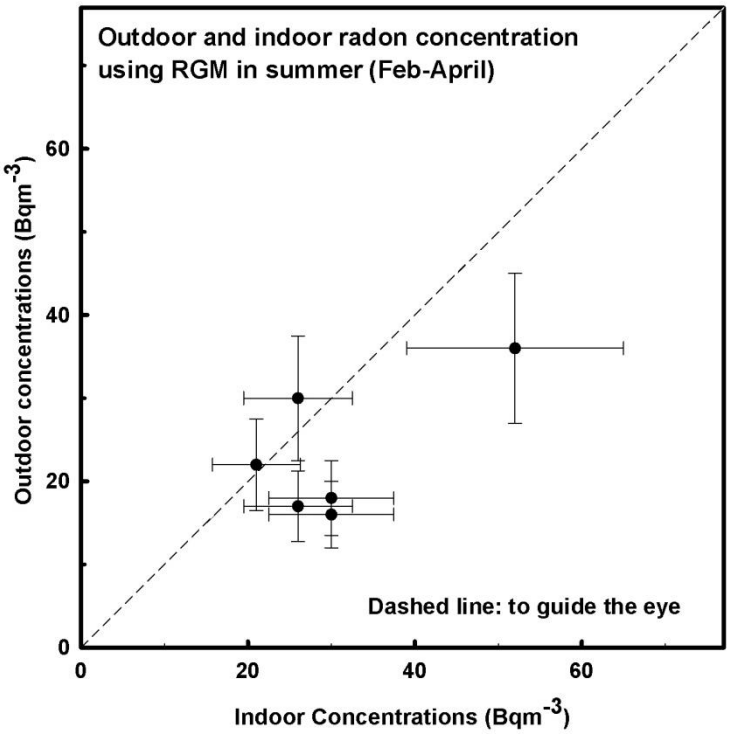
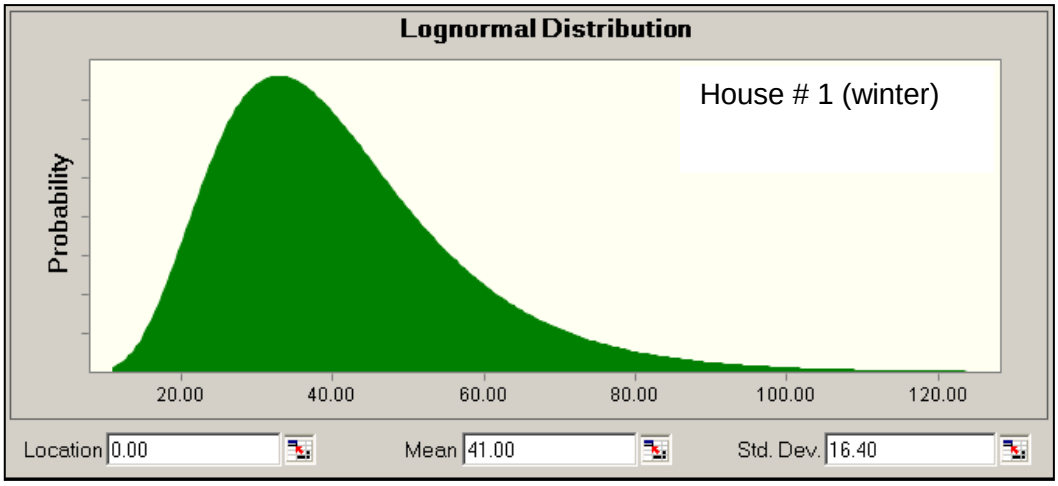


Figure 4.20 Comparison between indoor and outdoor radon concentrations results measuring by RGM during winter (upper) and summer (lower) periods for the six houses selected.

4.2.6 Lognormal distribution of the outdoor radon concentration

Since the RGMs revealed only average concentrations outdoor around each house over the period of two months, no data fitting was possible. Although this study was focused on the assessment of the indoor radon concentrations and the corresponding doses to the public due to dwelling inside the houses, an attempt has been made to estimate also the outdoor radon concentrations and then to estimate the annual doses to the public in the summer and winter periods. As it was mentioned before, the AlphaGuard was not installed outdoors for an extended period of time for obvious reasons. Similarly to the indoor concentration, it has also been assumed that the outdoor concentrations are log-normally distributed with the mean value equal to the measured average radon concentrations by RGMs and the relative standard deviation being 0.4 of the mean. Figures 4.21 to 4.26 show the estimated lognormal distribution of the outdoor radon based on the RGMs results.



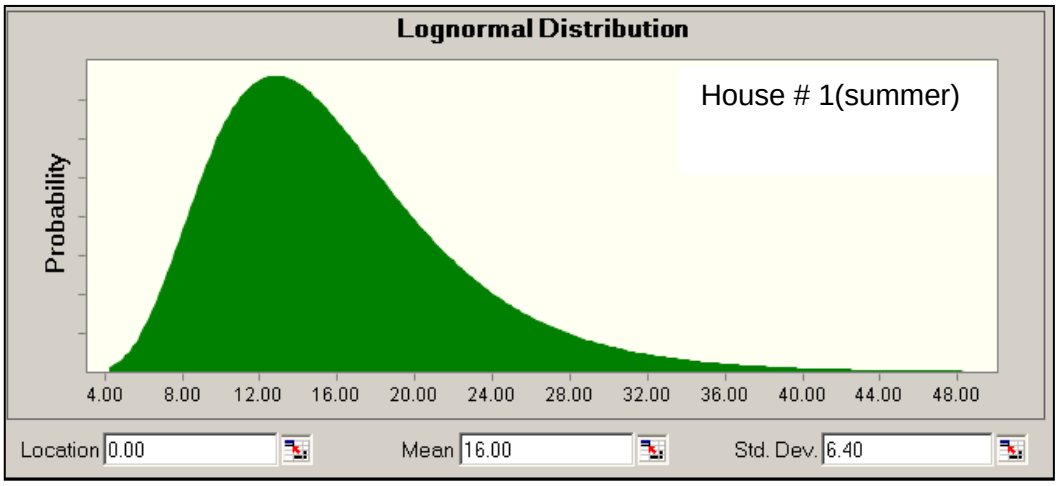
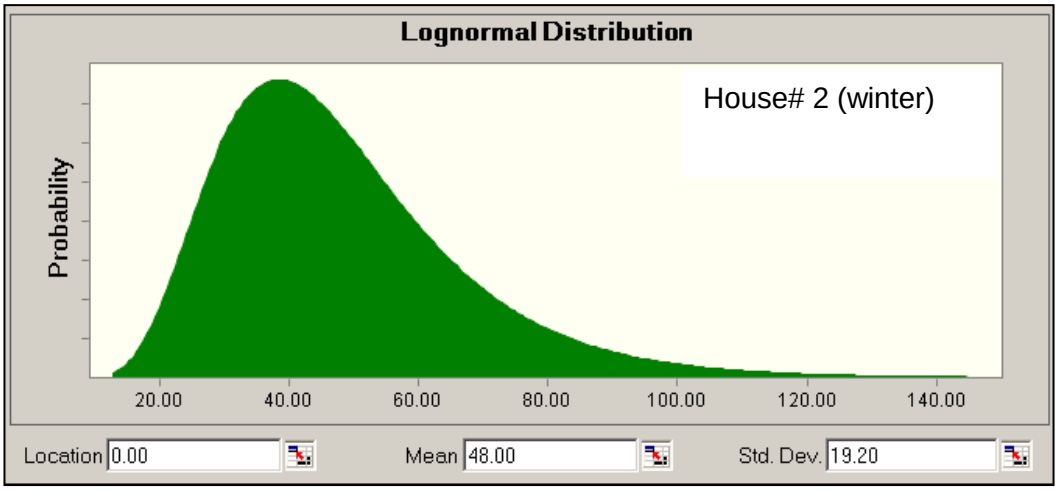


Figure 4.21 The assumed lognormal distribution of the outdoor radon concentration around the House # 1 during the winter (upper) and summer (lower) periods.



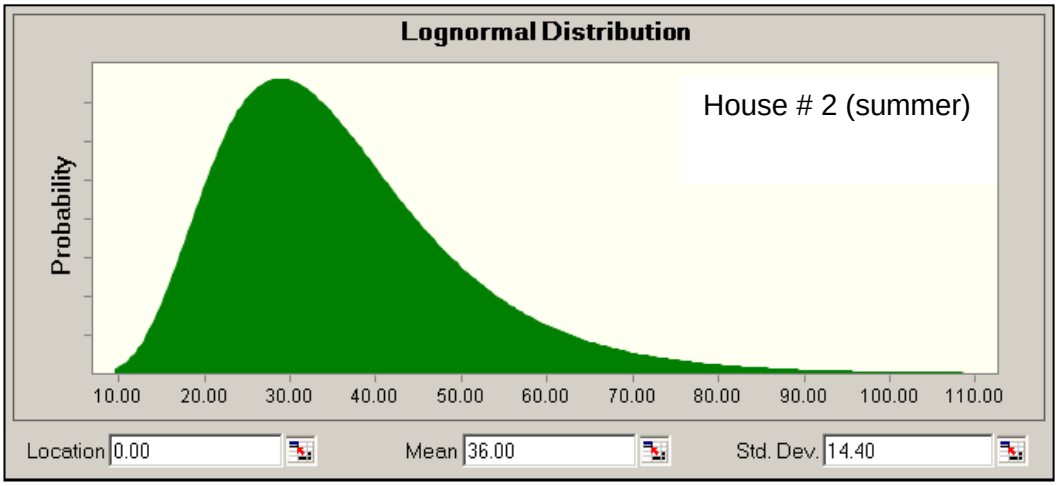


Figure 4.22 The assumed lognormal distribution of the outdoor radon concentration around the House # 2 during the winter (upper) and summer (lower) periods.

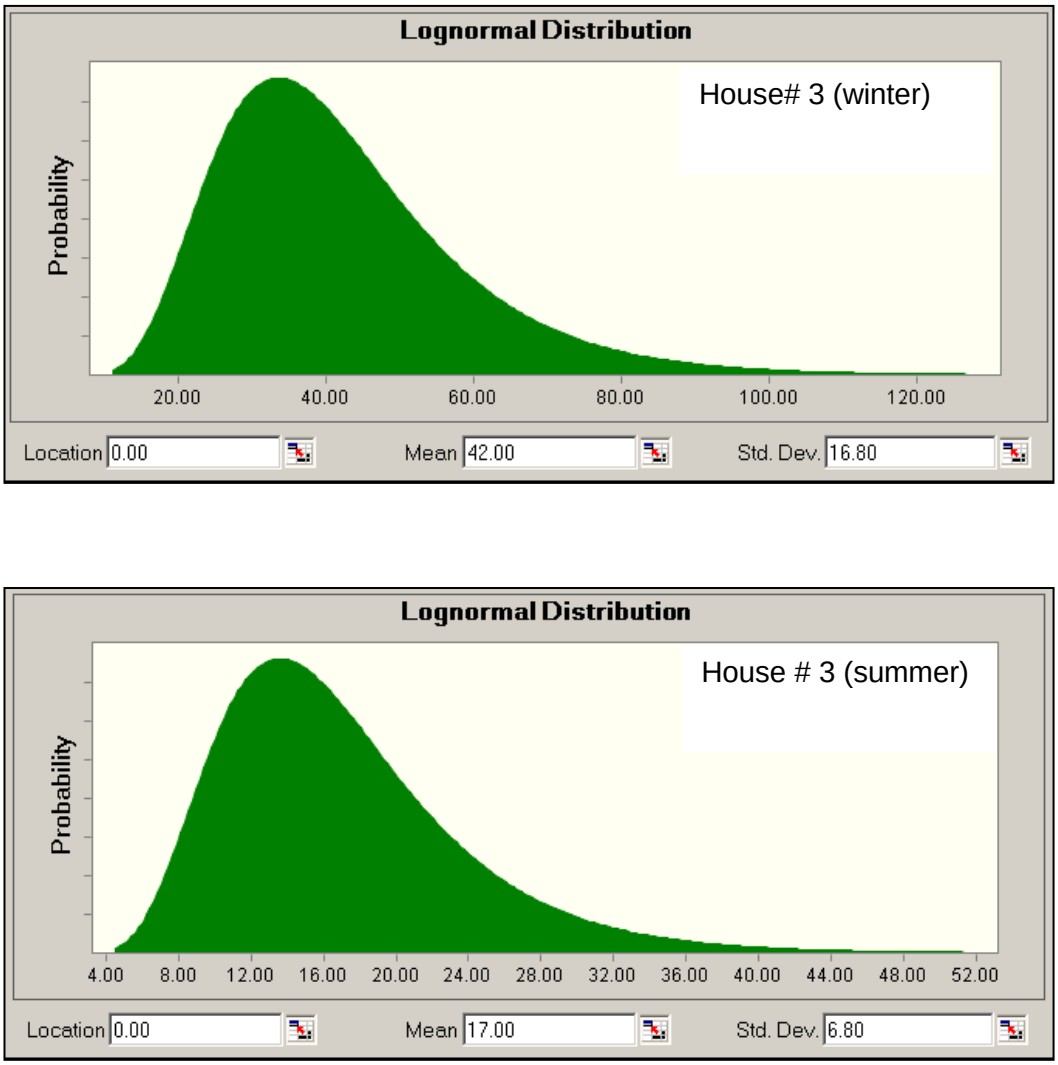


Figure 4.23 The assumed lognormal distribution of the outdoor radon concentration around the House # 3 (Golf-Club house) during the winter (upper) and summer (lower) periods.

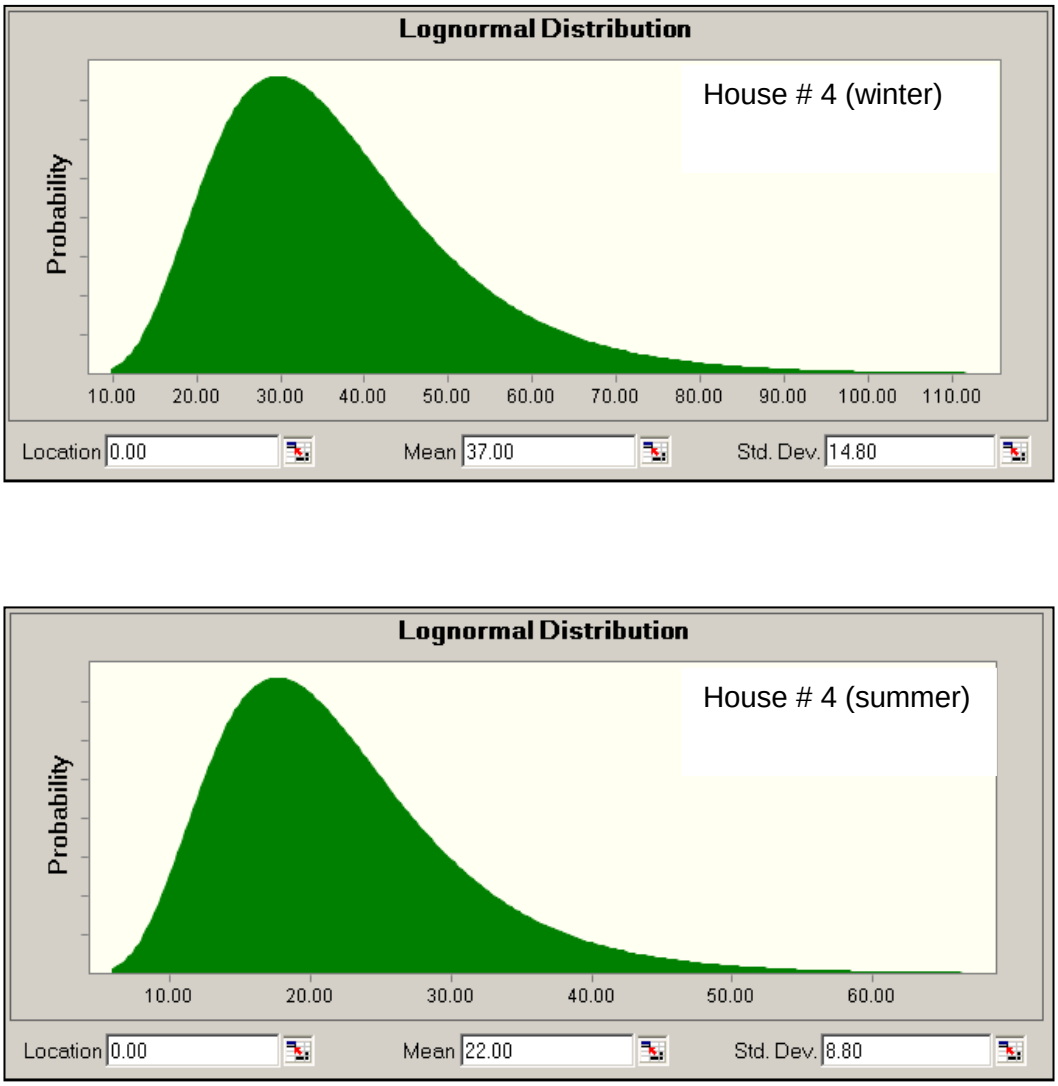


Figure 4.24 The assumed lognormal distribution of the outdoor radon concentration around the House # 4 during the winter (upper) and summer (lower) periods.

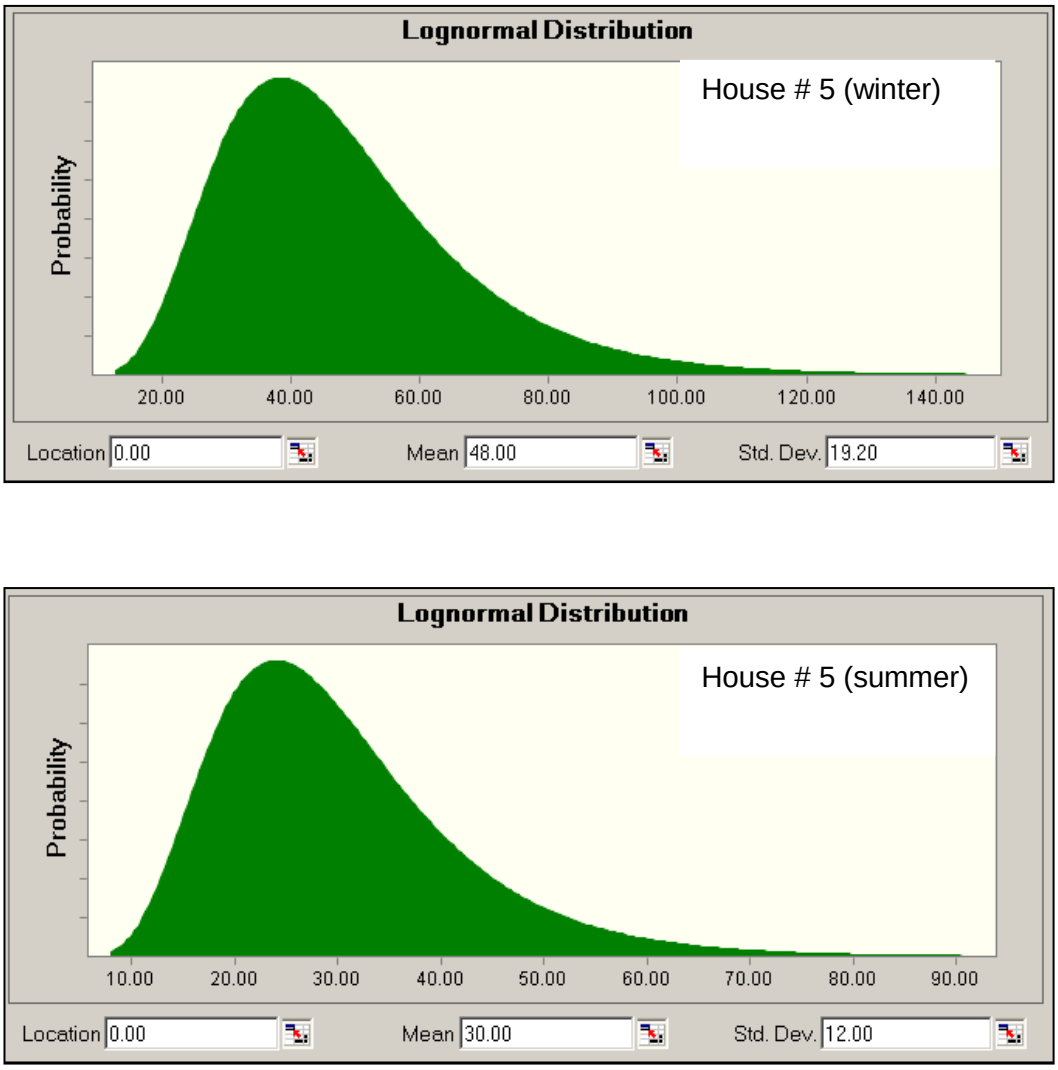


Figure 4.25 The assumed lognormal distribution of the outdoor radon concentration around the House# 5 during the winter (upper) and summer (lower) periods.

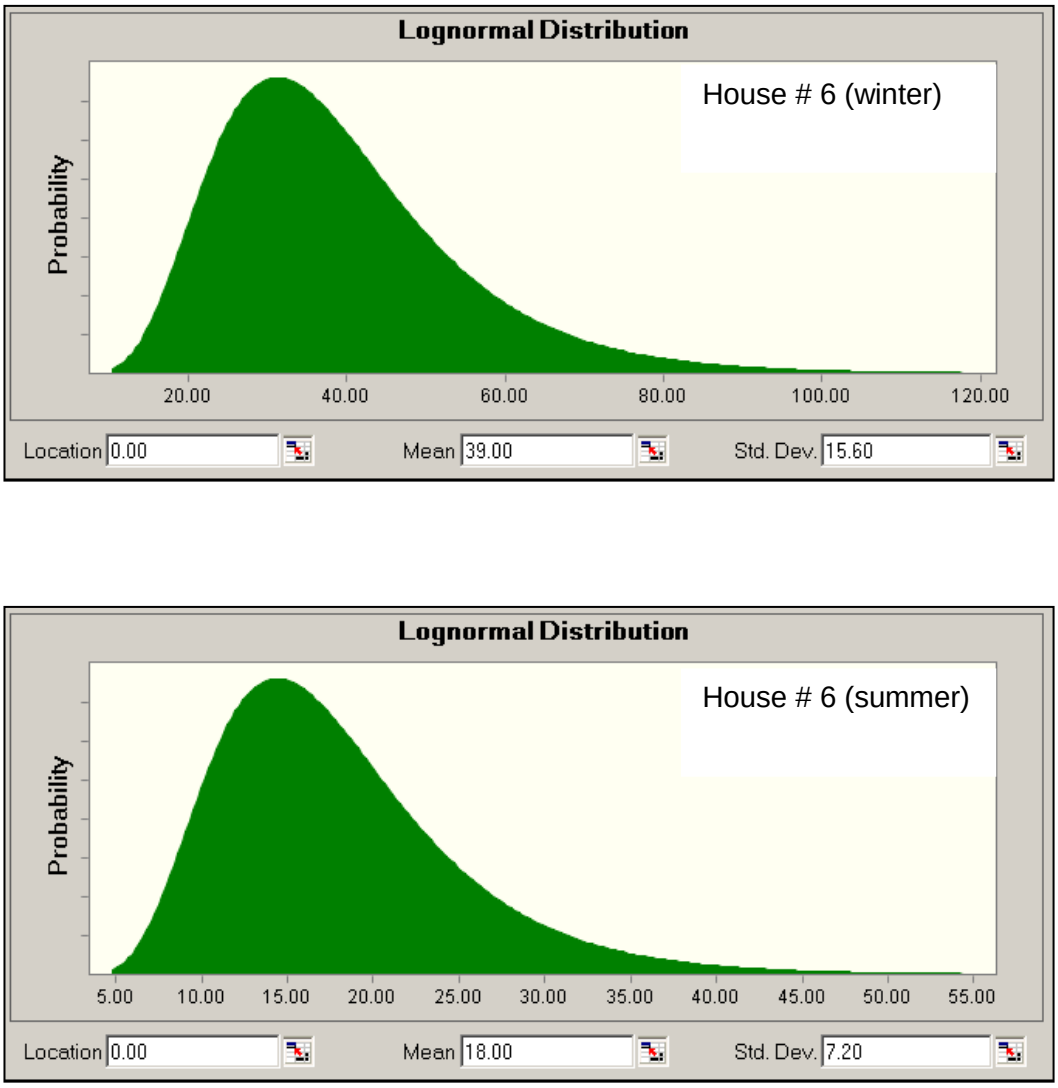


Figure 4.26 The assumed lognormal distribution of the outdoor radon concentration around the House # 6 during the winter (upper) and summer (lower) periods.

Chapter 5

Interpretation of radon concentrations: dose results and discussion

5.1 Introduction

The results of the measurements were described in Chapter 4 and these measurements and the related distribution graphs were used as inputs for the assessment of doses due to indoors and outdoors concentration of radon. The assessment of doses is described in this chapter.

5.2Overview of the dose assessment due to radon concentrations indoors and outdoors

To obtain an estimate of the annual doses from radon for the potential residents of the selected houses in winter and summer, Eq. (2.9) in Chapter 2was used. The dose ($D_{\mathrm{Rn}}(\mathrm{mSva}^{-1})$) due to inhalation of unit radon concentration (i.e. 1 in Bqm^{-3}) can be calculated from the expression [IC93]:

$$D_{\mathrm{Rn}} = 53.576 \times C_{\mathrm{Rn}} \times F \times P, \qquad (5.1)$$

where,

53.576 = the decay alpha-energy of radon daughters in unit ($\mathrm{mSvm}^{3}\mathrm{Bq}^{-1}\mathrm{a}^{-1}$) is evaluated as ($5.56\times10^{-9}\ \mathrm{JBq}^{-1}$) × the number of hours in a year (8760

hours) × 1.1(mSv per mJhm^{-3}), which is the ICRP conversion convention for representative persons of the public [IC93],

C_{Rn} = the measured radon concentration in (Bqm^{-3}). The indoor concentrations were obtained using an AlphaGuard and using a chi-square fit. The fitted concentrations from Figs. 4.10 to 4.15 have been used as an input for the assessment of the indoor dose. Outdoor concentrations were assumed to be log-normally distributed with the mean value equal to the average outdoor concentration, measured by RGMs and the standard deviation is a factor of 0.4 of this average. These assumed outdoor concentrations from Figs. 4.21 to 4.26 were used as inputs for the assessment of the outdoor dose,

F = the equilibrium factor calculated as the ratio of EEC (Bqm^{-3}) / C_{Rn} (Bqm^{-3}),

P = the relative occupancy indoors per annum.

The above formula assumes an average breathing rate of 1 m^{3}h^{-1}[IC93] over the entire year, while seasonal variations of the indoor occupancy and variations of breathing rate indoors and outdoors are disregarded. The present study attempts to take these variations into consideration by modifying the above equation as follows, this Equation has been introduced previously in Section 2.4 as Eq. (2.10):

$$D_{Rn}(\text{mSva}^{-1}) = 53.576 \times C_{Rn} \times F \times P \times B \times S, \quad (5.2)$$

where,

B = the relative breathing rate, which differs for indoor and outdoor occupancy, as explained below and

S = the fractional duration of the winter period. The fractional duration of the summer period is then expressed by (1 – S) and the other symbols remain as before.

The following assumptions also have been made in this study:

- the mean value of the equilibrium factor (F_{ind}) for indoors was assumed to

be 0.4. It has further been assumed that this factor varies usually between approximately 0.35 and 0.62.

- the mean value of the equilibrium factor (F_{out}) for outdoor was conservatively assumed to be 0.8 (due to the vicinity of the slimes dam and elevated concentration of ^{226}Ra in the mining area). It has further been assumed that this factor varies normally between approximately 0.7 and 0.9.

- the mean value of the indoor fractional occupancy in winter ($P_{ind,win}$) was assumed to be 0.85 in South Africa. It has further been assumed that this occupancy varies normally between approximately 0.76 and 0.84.

- the mean value of the indoor fractional occupancy in summer ($P_{ind,sum}$) was assumed to be 0.75. It has further been assumed that this occupancy varies normally between approximately 0.6 and 0.9.

- the fractional outdoor occupancies for winter and summer have been assumed to be $P_{out,win} = (1-P_{ind,win})$ and $P_{out,sum} = (1-P_{ind,sum})$, respectively,

- the mean value of the fractional duration of winter period (S_{win}) per annum has been assumed to be 0.4167 (i.e. 5 months). It has further been assumed that this fractional duration varies normally between approximately 0.21 and 0.63.

- the breathing rate varies with the degree of activity of an individual and is not easily measured. The breathing rate of an adult male was estimated to be 0.45 m^3h^{-1} resting (6 to 8 hours a day) and 1.2 m^3h^{-1} in light activity (16 to 18 hours a day). The values for an adult female are 20% less resting and 5% less in light activity [IC75]. The breathing rates were re-evaluated in ICRP24 [IC94] and somewhat lower average values were derived, namely 22.2 $lmin^{-1}$for the adult male and 17.7 $lmin^{-1}$for the adult female. The

mean value of the indoor breathing rate (B_{ind}) has conservatively been assumed to be $1m^3h^{-1}$ in this study and the mean value of the relative indoor breathing rate was assumed to be $1m^3h^{-1}$. It has further been assumed that this relative breathing rate varies log-normally between approximately 0.81 m^3h^{-1} and 1.23 m^3h^{-1}.

- the mean value of the relative outdoor breathing rate (B_{out}) has been assumed to be $1.1m^3h^{-1}$. It has further been assumed that this relative breathing rate varies log-normally between approximately 0.90 m^3h^{-1} and 1.32 m^3h^{-1}.

The following calculations for the annual dose rate indoors and outdoors in the winter and summer periods have been performed using the Crystal Ball statistical software [OC11] in this study:

(i) The estimated dose due to radon inhalation indoors in winter has been assessed using:

$$D_{\text{ind,win}} = 53.576 \times C_{\text{Rn,ind,win}} \times F_{\text{ind}} \times P_{\text{ind,win}} \times B_{\text{ind}} \times S_{\text{win}}. \tag{5.3}$$

(ii) The estimated dose due to radon inhalation indoors in summer has similarly been calculated using:

$$D_{\text{ind,sum}} = 53.576 \times C_{\text{Rn,ind,sum}} \times F_{\text{ind}} \times P_{\text{ind,sum}} \times B_{\text{ind}} \times (1 - S_{\text{win}}). \tag{5.4}$$

*(iii)*The estimated dose due to radon inhalation outdoors during the winter period has been assessed using:

$$D_{\text{out,win}} = 53.576 \times C_{\text{Rn,out,win}} \times F_{\text{out}} \times (1 - P_{ind,win}) \times B_{\text{out}} \times S_{\text{win}}, \tag{5.5}$$

and for the summer period using:

$$D_{\text{out,sum}} = 53.576 \times C_{\text{Rn,out,sum}} \times F_{\text{out}} \times (1 - P_{\text{ind,sum}}) \times B_{\text{out}} \times (1 - S_{\text{win}}). \quad (5.6)$$

(iv) Thus, the total dose due to radon inhalation indoors for the whole year is obtained as:

$$D_{\text{ind}} = D_{\text{ind,win}} + D_{\text{ind,sum}}. \quad (5.7)$$

(v) Finally, the estimated total annual dose due to radon inhalation has been assessed as the sum of

$$D_{\text{ann}} = D_{\text{ind}} + D_{\text{out,win}} + D_{\text{out,sum}}. \quad (5.8)$$

Here it should be noted that the estimated annual dose assumes that the potential residents spent their outdoors time in the vicinity of the selected house.

5.3 Resulting doses due to radon inhalation in the selected houses

Using Eqs. (5.2) to (5.8) the annual dose rates were determined and the corresponding distributions are shown in Figs. 5.1 to 5.7. These assessments have been based on defining statistical input variables in the Eqs. (5.1) to (5.8) and calculating the resulting dose by the Monte-Carlo method, which is incorporated in the Crystal Ball software. Although the input variables assumed were realistic, the assumptions could be arbitrarily changed to reflect specific situations and the resulting dose could be obtained in the form of statistical distributions. This approach allows an estimate to be made of not only the mean dose, but also various percentiles, as may be needed.

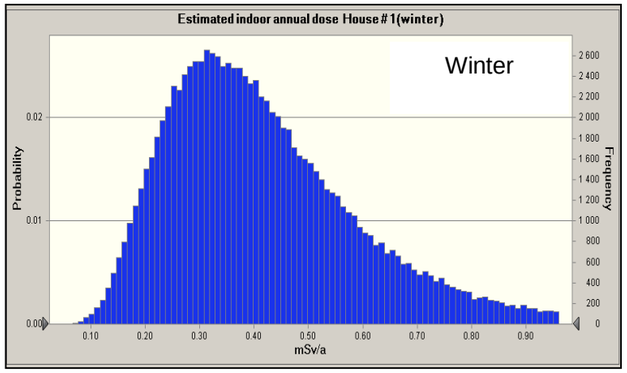
A summary of the resulting dose due to radon is presented in Table 5.1. The doses given in Table5.1 were assessed using the inputs of breathing rate, house occupancy, duration of season and equilibrium factor in a form of the assumed frequency distributions. Due to this approach, the resulting dose, calculated by the Crystal Ball software using Eqs. (5.3) to (5.8), were also obtained in a form of frequency distributions. These distributions of dose can be characterised by a mean value of the distribution, corresponding to the most frequently occurring input situations and by an "upper" dose value, characterised for simplicity by the 90^{th} percentile of the dose distribution. This "upper" dose values corresponds to the situations with conservative (or "upper") input values, which simultaneously occur much less frequently. The "upper" dose values, given in Table 5.1, can therefore be interpreted as conservative doses, which could be reached under quite rare situations when all input values (characterised by the assumptions made) were conservative.

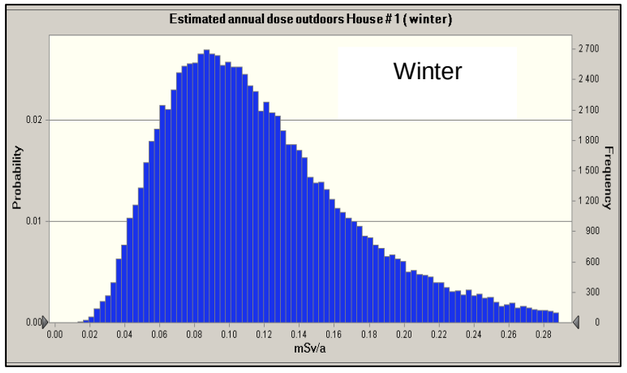
It should be noted that on average these indoors dose (assessed from the calculations measured by the AlphaGuard) in winter time are generally higher than summer, as has been already noted in Section 4.2.2.In addition, results for the outdoors dose (measured by RGMs) show larger values for summer than winter due to temperature differences. (See Section 4.2.2, where variation due to temperature different is explained in detailed).

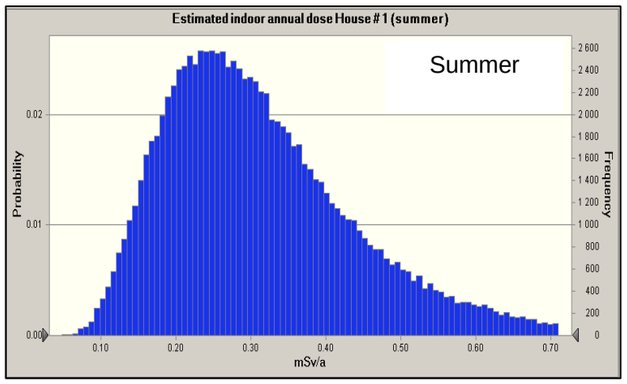
House # 1

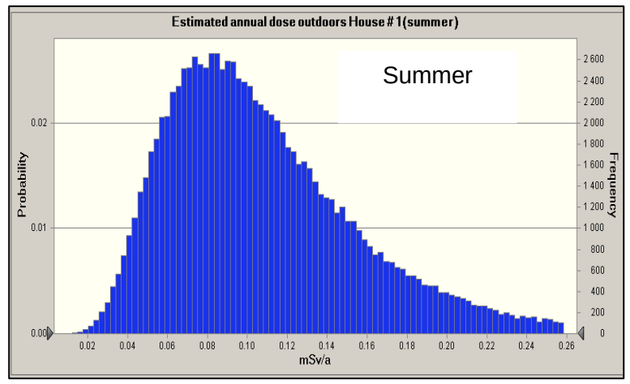
Indoors

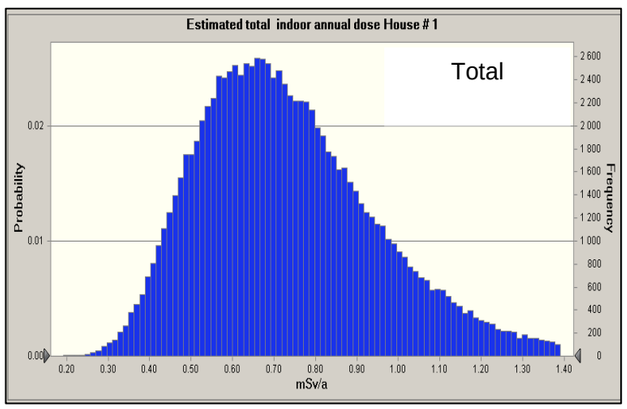
Outdoors

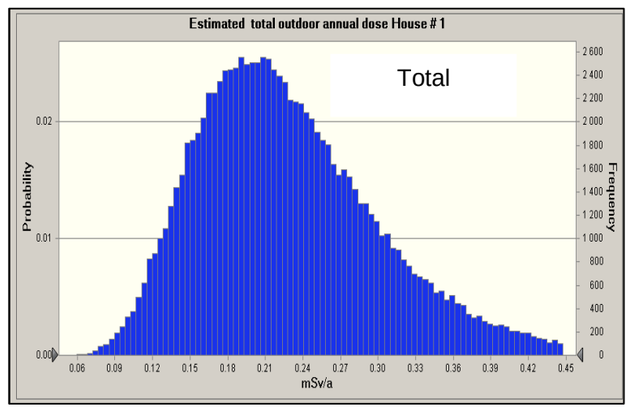












House # 1estimated total annual dose

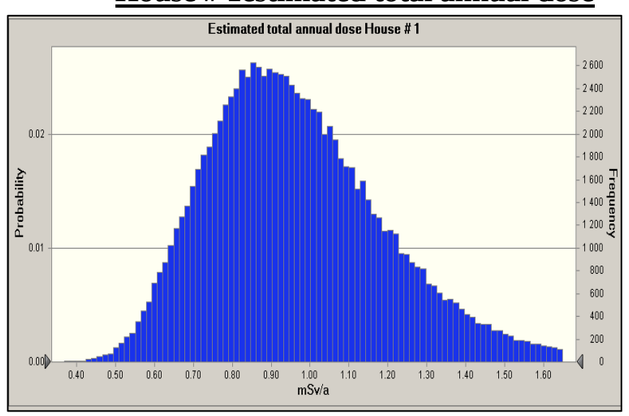
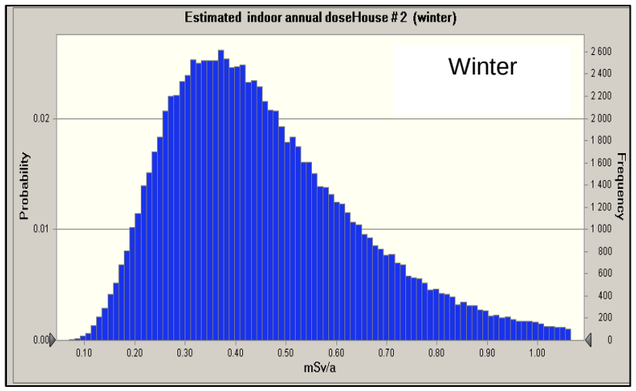
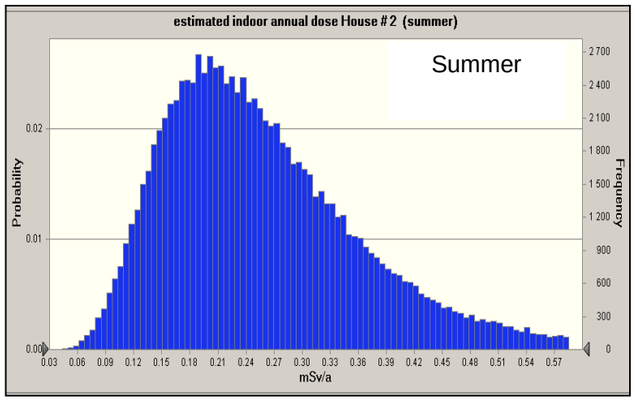


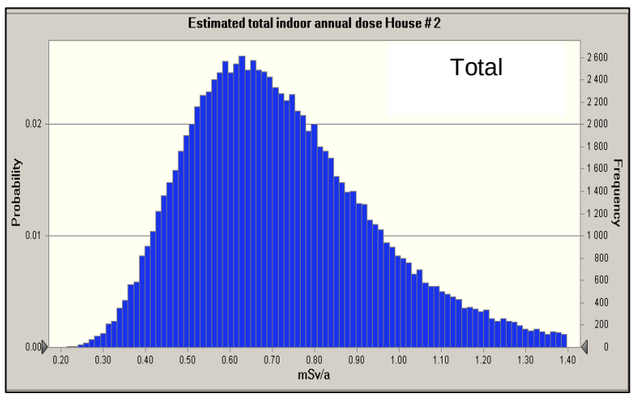
Figure 5.1 Estimated annual doses during winter and summer time, indoors (column 1) and outdoors (column 2) and the total estimated annual dose (column 3) for House # 1.

House # 2

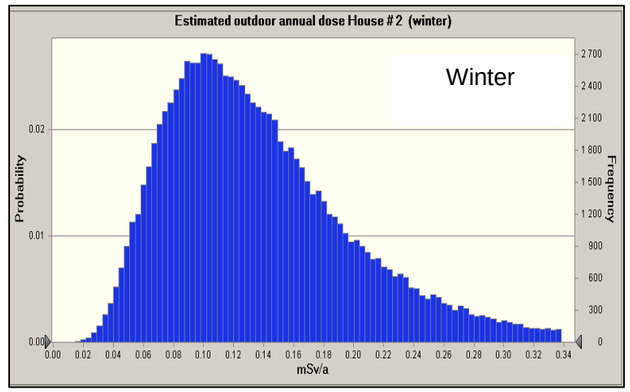
Indoors

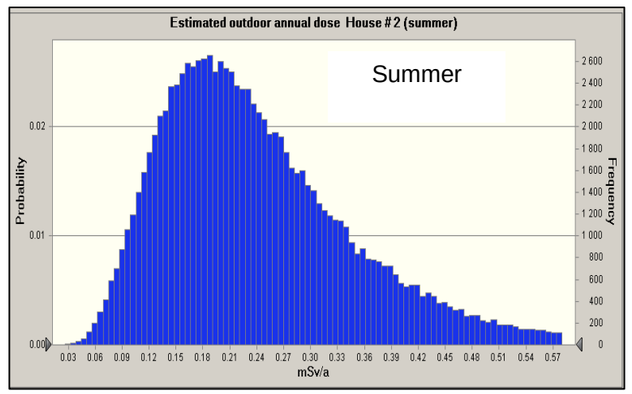


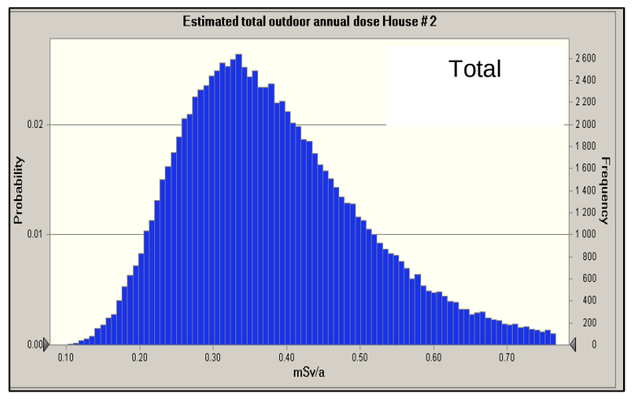




Outdoors







House # 2 estimated total annual dose

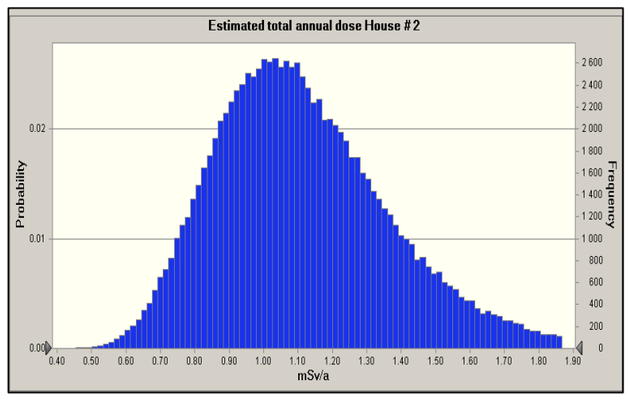
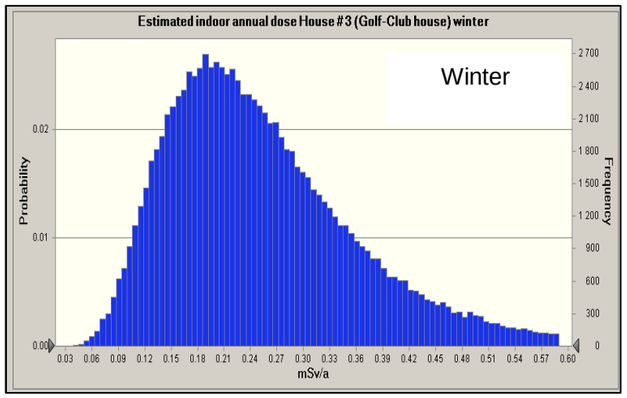


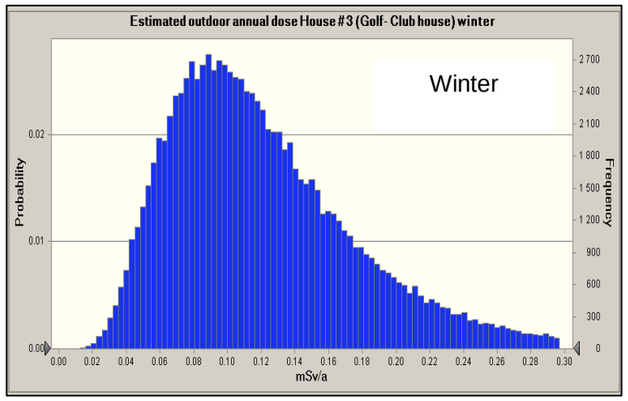
Figure 5.2 Estimated annual doses during winter and summer time, indoors (column 1) and outdoors (column 2) and the total estimated annual dose (column 3) for House # 2.

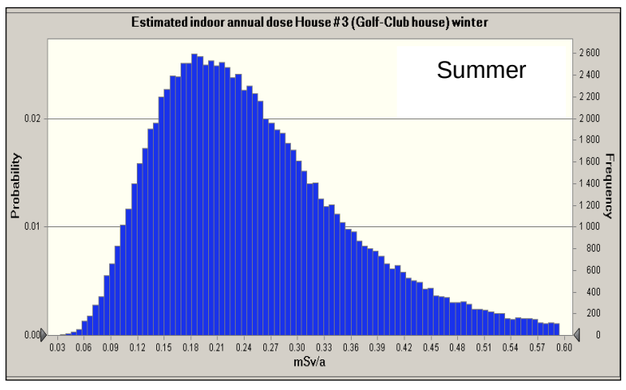
House # 3(Golf-Club house)

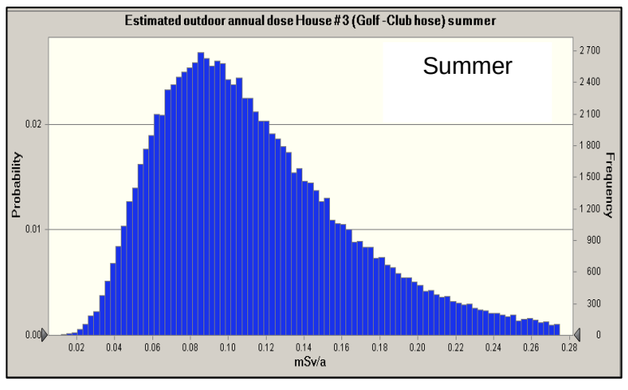
Indoors

Outdoors

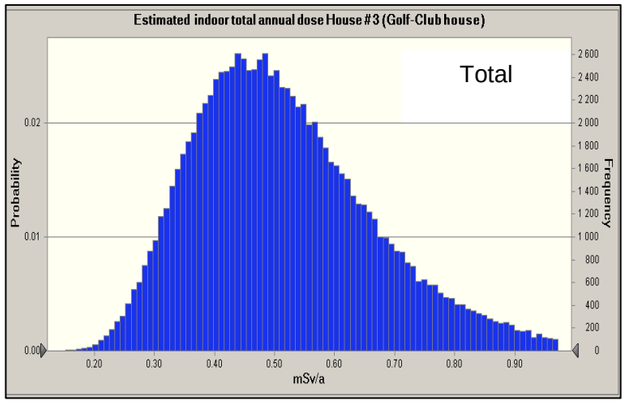


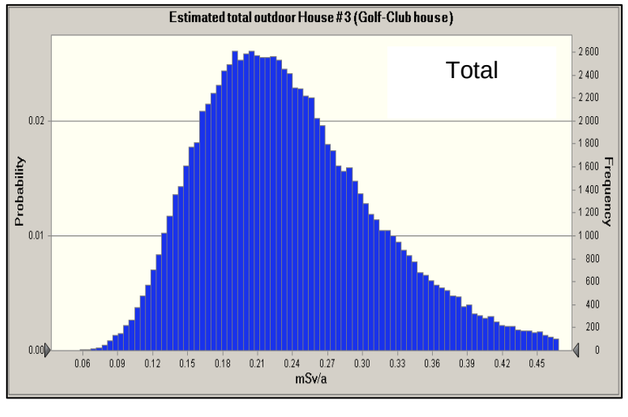






House # 3 estimated total annual dose





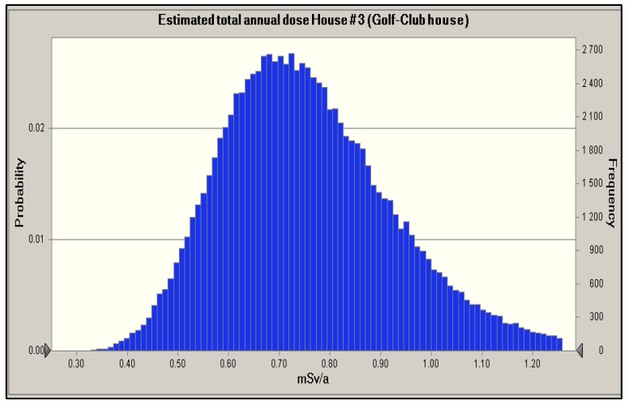
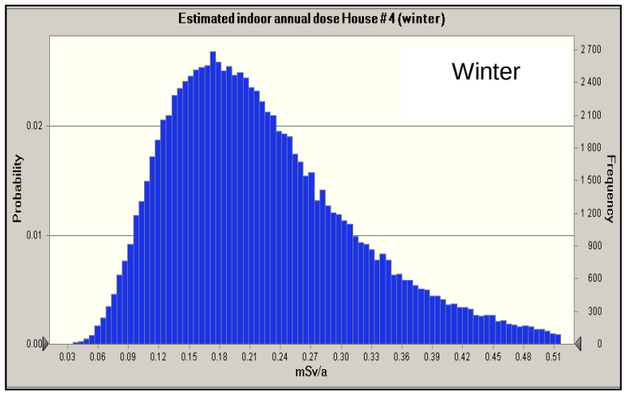
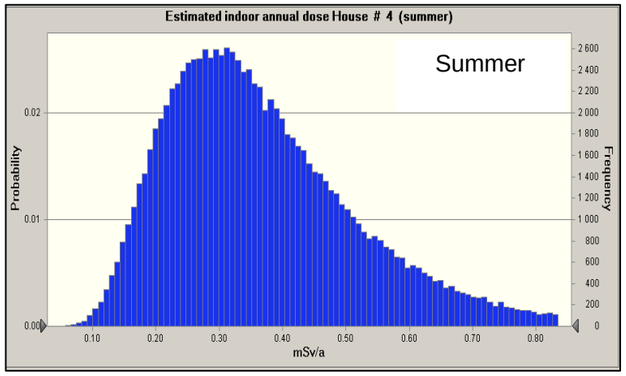


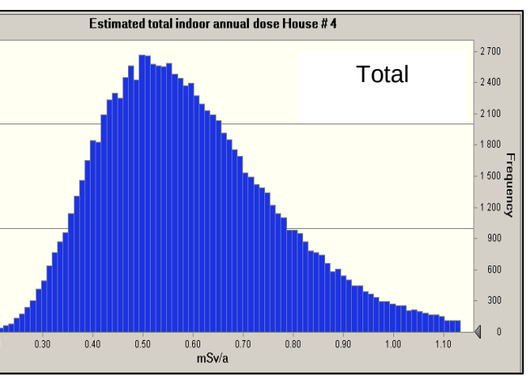
Figure 5.3 Estimated annual doses during winter and summer time, indoors (column 1) and outdoors (column 2) and the total estimated annual dose (column 3) House # 3 (Golf-Club house).

House # 4

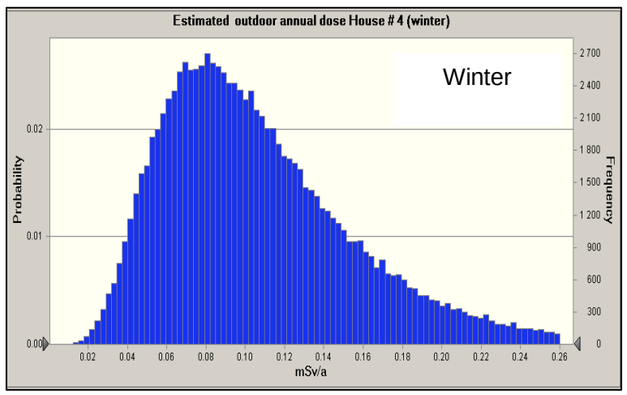
Indoors

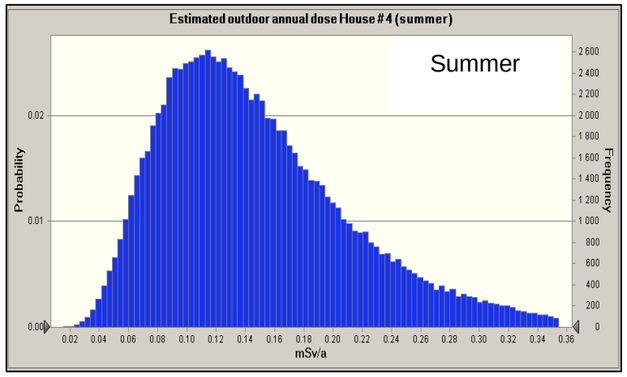


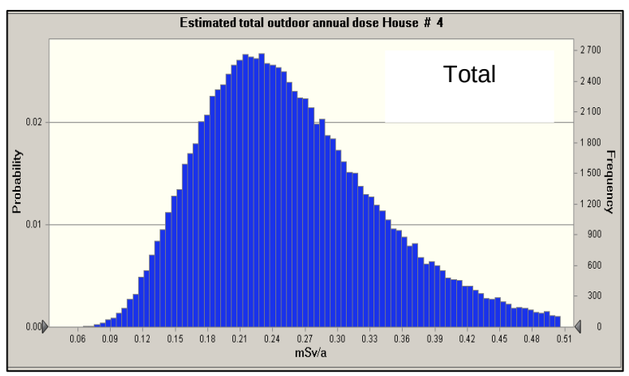




Outdoors







House # 4 estimated total annual dose

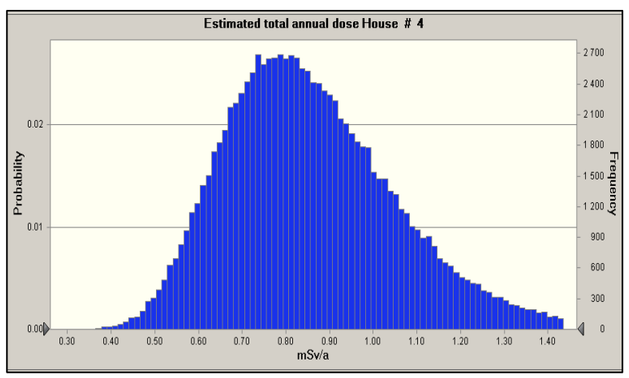
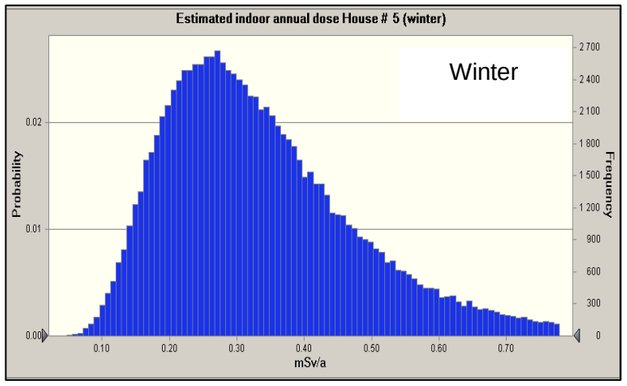
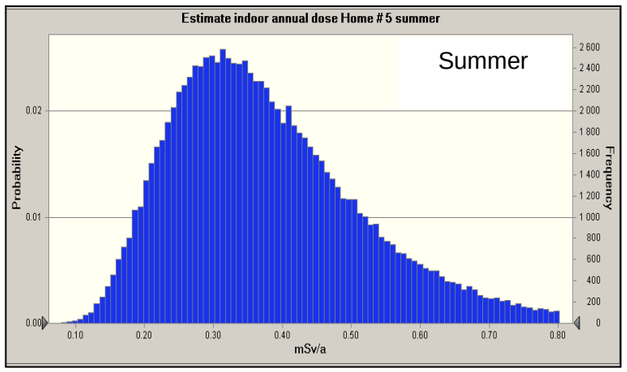


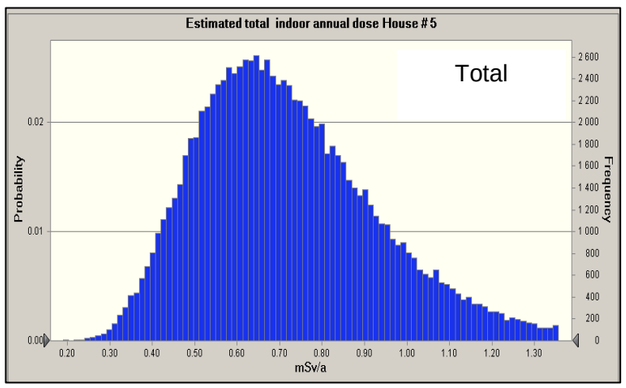
Figure 5.4 Estimated annual doses during winter and summer time, indoors (column 1) and outdoors (column 2) and the total estimated annual dose (column 3) House # 4.

House # 5

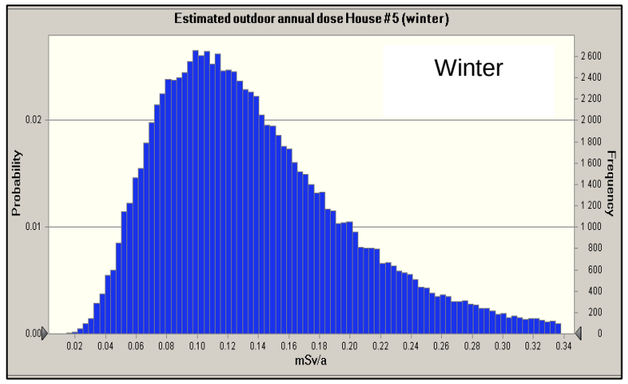
Indoors

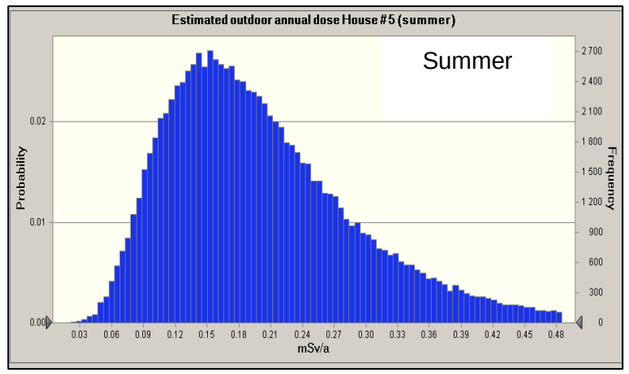


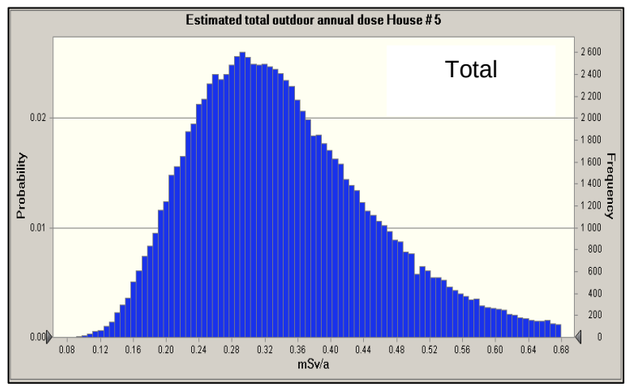




Outdoors







House # 5 estimated total annual dose

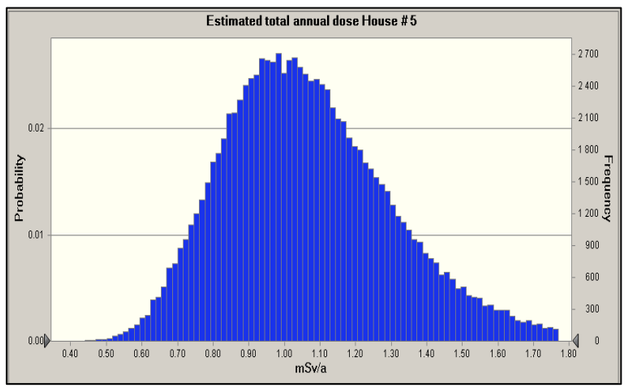
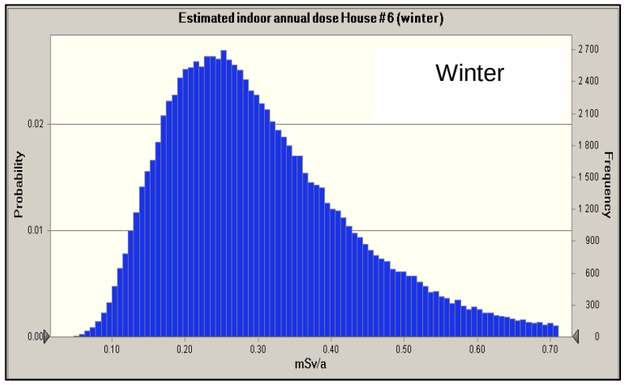
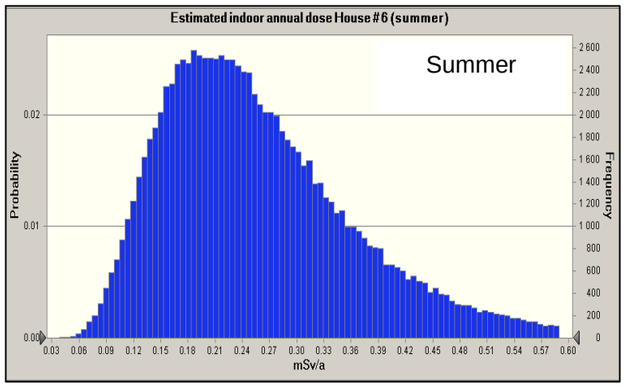


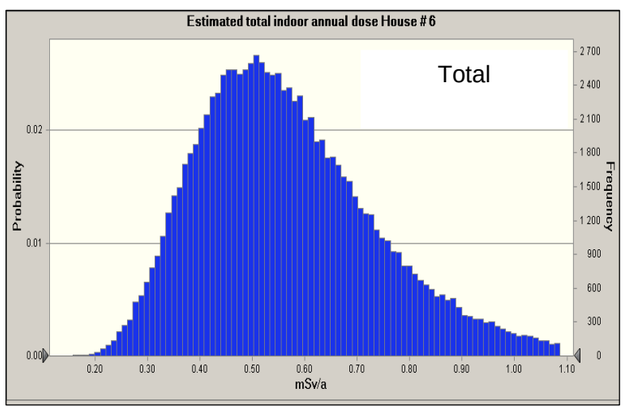
Figure 5.5 Estimated annual doses during winter and summer time, indoors (column 1) and outdoors (column 2) and the total estimated annual dose (column 3) House # 5.

House # 6

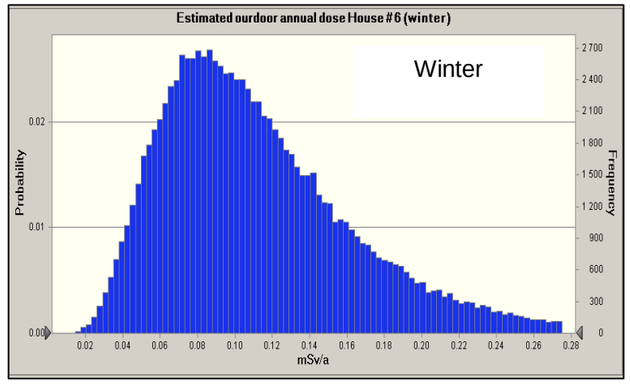
Indoors

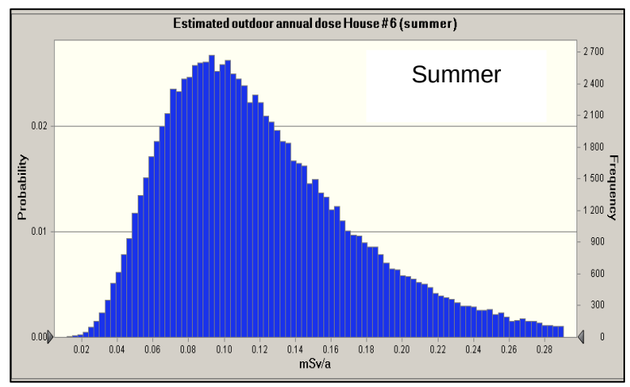


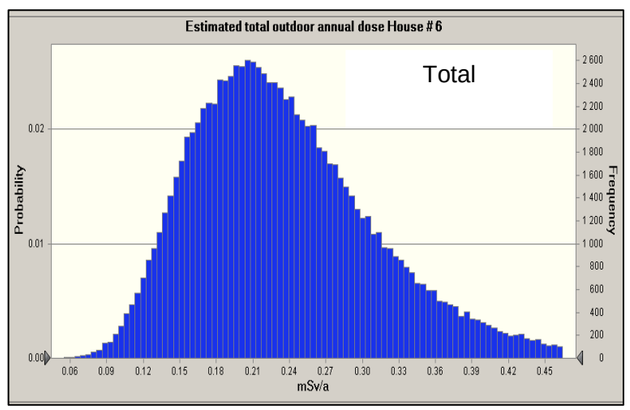




Outdoors







House # 6 estimated total annual dose

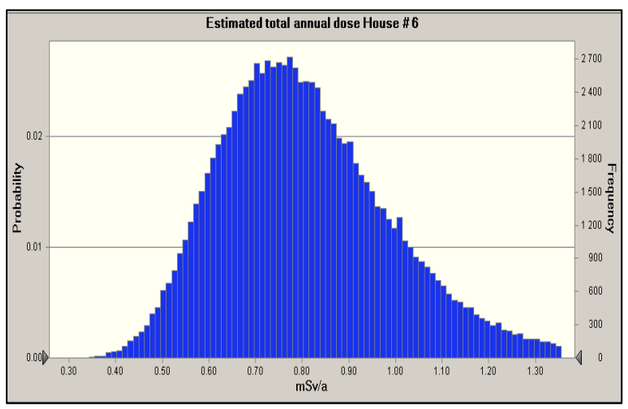
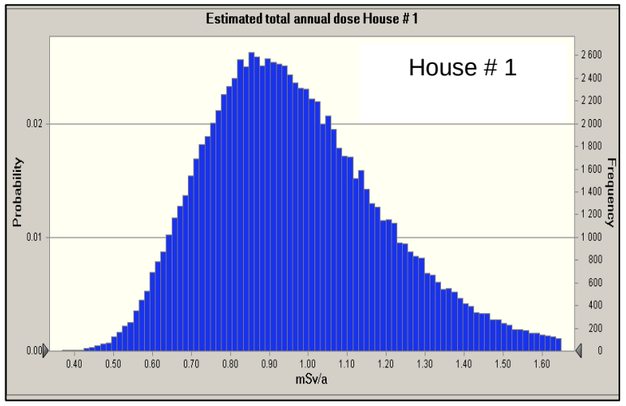
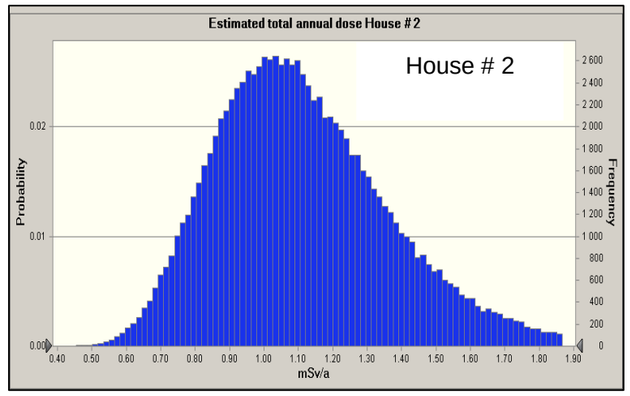
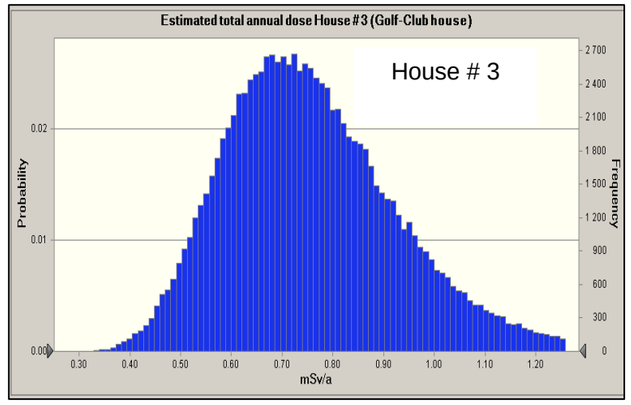


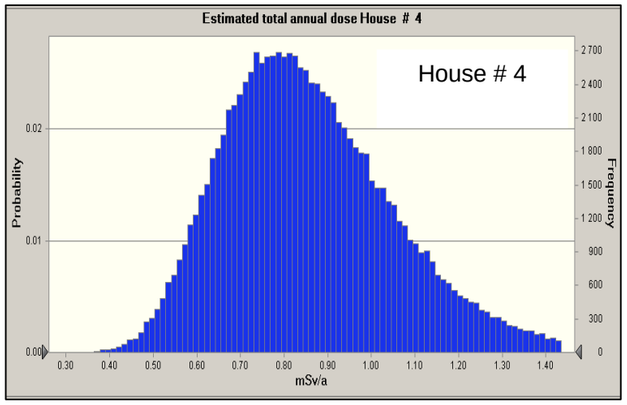
Figure 5.6 Estimated annual doses during winter and summer time, indoors (column 1) and outdoors (column 2) and the total estimated annual dose (column 3) House # 6.

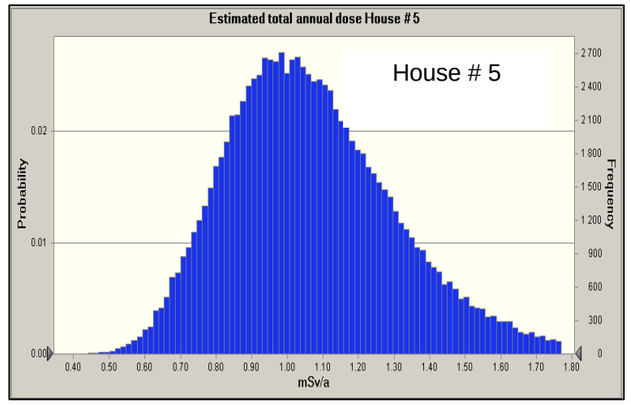
Estimated total annual dose











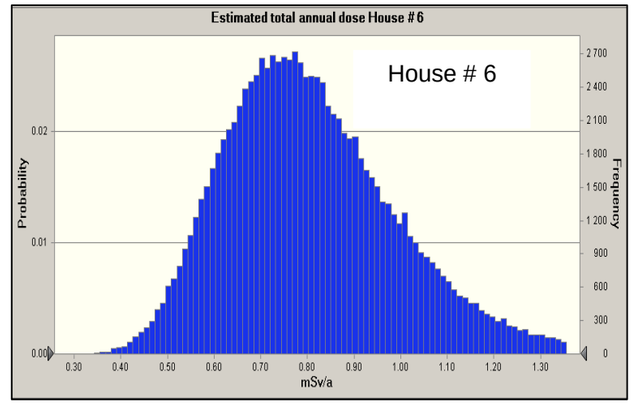


Figure 5.7 Total estimated annual dose during winter and summer time for the six houses selected.

Table 5.1 The summery of the assessed dose in the individual houses for indoor and outdoor occupancy in winter and summer periods and the total estimated annual dose to the potential residents of these houses.

Location	Indoorwinter (mSv/a)	90th percentile Indoorwinter	Indoor summer (mSv/a)	90th percentile Indoorsummer	Total indoor (mSv/a)	90th percentile Total Indoor	Outdoor winter (mSv/a)	90th percentile Outdoor winter	Outdoor summer (mSv/a)	90th percentile Outdoorsummer	Total outdoor (mSv/a)	90th percentile Totaloutdoor	Estimated Total annual dose (mSv/a)	90th percentile of the total annual dose
House # 1	0.42	0.67	0.31	0.50	0.74	1.05	0.12	0.20	0.11	0.18	0.23	0.33	**0.97**	1.29
House # 2	0.47	0.75	0.26	0.41	0.73	1.04	0.14	0.23	0.24	0.40	0.38	0.56	**1.12**	1.47
House # 3 (Golf-Club house)	0.25	0.41	0.26	0.41	0.52	0.73	0.12	0.20	0.11	0.19	0.24	0.35	**0.76**	0.99
House # 4	0.22	0.36	0.37	0.59	0.60	0.85	0.10	0.18	0.15	0.25	0.26	0.38	**0.86**	1.13
House # 5	0.34	0.54	0.38	0.60	0.72	1.02	0.14	0.23	0.20	0.33	0.34	0.50	**1.07**	1.40
House # 6	0.31	0.49	0.26	0.41	0.57	0.81	0.11	0.19	0.12	0.36	0.23	0.34	**0.81**	1.07

Chapter 6

Summary and recommendations

After thorough analysis and interpretation of the results presented in Chapters 4 and 5, this chapter will summarise the results of the study and present recommendations.

It should first be noted that the objective of this work was to assess the indoor doses in the selected houses built on the mining area. These doses were assessed in the form of frequency distributions as described in Section 5.2. In addition, outdoor doses and the total annual doses to the potential resident of these houses were also estimated based on the assumptions given in Section 5.2. It should also be noted that one of the important outcomes of this study was the development of the statistical assessment of doses due to radon inhalation as was explained in Section 5.2.

6.1 Summary of the findings

The outcomes of the present study may be summarised as follows:

1) The AlphaGuard active radon monitor which was used for about week-long measurements of indoor radon concentrations in the selected houses was tested and calibrated by the manufacture in Germany prior to taking measurements. Radon concentrations were recorded every hour and data obtained were converted into frequency distributions (see Figs. 4.10 to 4.15). Using these distributions of indoor radon concentrations and assumptions with regard to occupancy of these houses, variation of the breathing rates of potential residents, variation of the duration of winter and summer seasons (see Section 5.2) the doses were assessed (see Figs.

5.1 to 5.7). The total estimated annual dose due to indoor and outdoor radon exposure was based on measurements undertaken with active and passive monitors. The results differ from house to house, which could have been expected, because of the indoor and outdoor metrological parameters and also different ventilation system. Mean doses and their 90^{th} percentiles are given in Table 5.1. This has resulted in the determination of annual doses within the mine compound which generally did not exceed the accepted radiological protection maxima as determined and promulgated by IAEA except for two houses (House # 2 and # 5).The results obtained using RGMs for April–June are closer to the AlphaGuard results than those for June–July. The difference can be related to the seasonal changes and the ventilation inside the houses during different months, as well as to the variations of humidity and temperature, as it was explained in Section 4.2.2.

2) Measurements performed by the mining staff using the RGMs were compared with the results obtained by the AlphaGuard Tables 4.6 and 4.7.Also the correlation between the AlphaGuard and RGM measurements were shown in Fig. 4.17.

3) It can be concluded that the mean values of the total annual doses, which would be incurred by potential residents, are generally close to the annual limit to the public of $1mSa^{-1}$. Although the estimate of the total annual doses is affected by several uncertainties (mainly the contribution of the outdoor doses) the potential residents would most likely receive annual doses higher than 1 mSv, especially if contributions due to external exposure are taken into consideration.

4) It can also be concluded that the indoor radon concentration in any of the selected houses has not reached the lower level of the action level range 200 to $600Bqm^{-3}$ recommended by the ICRP [IC93]. Nevertheless, the European pooled residential case-control study has shown that there is a

risk of lung cancer even at levels of long-term average radon concentrations below 200 Bqm^{-3} [IC10].

5) In view of the results obtained in this study, the following ICRP statement merits attention :

"*Dose coefficients for radon and progeny will replace the Publication 65 dose conversion convention which is based on nominal values of radiation detriment derived from epidemiological studies comparing risks from radon and external radiation. The current dose conversion values may continue to be used until dose coefficients are available. The commission advises that the change is likely to result in an increase in effective dose per unit exposure of around a factor of two* [IC09]."

6) In general it should be emphasised that the values obtained for the doses are low. For example, according to the Wikipedia article on radon dose: http://en.wikipedia.org/wiki/Health_effects_of_radon, the average dose to the German population from radon is 1.1 mSv/a. In this study the estimated average dose to occupants in all the houses, except for one, is less than the German value and in the remaining house it was 1.12 mSv/a. In other words the average annual dose in these dwellings is pretty low. This is surprising but reassuring and this fact should be emphasised. Of course methods of mitigation should still be considered in the spirit of ALARA, but fortunately it turns out that radon exposure is not a serious problem in this case.

6.2 Recommendations

In view of the dose results obtained and the new ICRP limits contained in [IC10] and [IC93] the following recommend actions are made:

- More detailed investigations should be carried out in the selected houses built in the mining area.

- More effective ventilation systems should be adopted and in particular during winter time heated outside air should be blown into the house, with a turnover of one room volume per hour.

- Attention should be paid to potential radon entry pathways into the selected houses and these pathways should be sealed.

- The present work should be seen as an initial study and it is suggested that more extensive measurements of radon concentration should be taken before and after measures have been implemented in order to reduce the flow of radon into the selected houses. It should be noted that one of the important outcomes of this study was the development of the statistical assessment of doses due to radon inhalation as was explained in Section 5.2.

- It should be borne in mind that monitoring of radon concentration is always subjected to uncertainties, sometimes quite significant, due to both natural processes (diurnal and seasonal fluctuations of radon emanation), as well as changes in house ventilation. In order to improve the accuracy of the assessment of outdoor doses and total annual doses to potential residents it is recommended that outdoor radon concentrations in the vicinity of the selected houses be measured by the AlphaGuard for an extended time period and that the assumptions related to the variations of the house occupancy, breathing rates and seasonal variations be verified.

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