EUROPEAN COLLABORATIVE ACTION

INDOOR AIR QUALITY & ITS IMPACT ON MAN

Environment and Quality of Life

Report No 15

Radon in indoor air

European Commission
Directorate-General for Science, Research and Development
Joint Research Centre-Environment Institute

1995

EUR 16123 EN

EUROPEAN COLLABORATIVE ACTION

INDOOR AIR QUALITY & ITS IMPACT ON MAN

(formerly COST Project 613)

Environment and Quality of Life

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Radon in indoor air

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European Commission
Directorate-General for Science, Research and Development
Joint Research Centre-Environment Institute

1995

Report EUR 16123 EN

Published by the
EUROPEAN COMMISSION
Directorate-General XIII
Telecommunications, Information Market and Exploitation of Research
L-2920 Luxembourg

Cataloguing data can be found at the end of this publication

Luxembourg: Office for Official Publications of the European Communities, 1995

ISBN 92-827-0119-0

Printed in Italy

In this series the following reports have already been published.

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Abstract

ECA (European Collaborative Action "Indoor Air Quality and its Impact on Man"), 1995.
Radon in indoor air. Report No 15, EUR 16123 EN.
Luxembourg: Office for Official Publications of the European Community.

In an attempt to overcome the increasing difficulty of having concise essential information on important indoor pollutants at hand, the European Collaborative Action "Indoor Air Quality and Its Impact on Man" has published several reports amongst which a short report on radon in indoor air was published in 1988. Since then, considerable new information has become available. This report summarizes and discusses the actual state of knowledge on sources of radon in indoor air, typical concentrations, health effects and radon risk estimates, indoor radon measuring methodology, instrumentation, remedial and preventive measures to reduce indoor radon, and recommended and regulatory radon levels. The report concludes that the cost of remedial actions and the lack of public interest are major obstacles in the reduction of population exposure. Appendices which have a decay scheme of radon-222 and its short-lived progeny and which explain special quantities and units for radon and radon decay products complement the report.

Preface

In November 1988 the European Concerted Action "Indoor Air Quality and its Impact on Man" (COST 613) produced the report "Radon in Indoor Air", which was the first in its series of reports dealing with indoor air pollutants and other aspects of indoor air quality. Since then, substantial changes have taken place in our understanding of the public health significance of indoor radon.

Exposure to indoor radon is now considered by most public health agencies as being, after smoking, the second most common cause of lung cancer. In most EU Member States national and regional surveys to determine the distribution of radon levels in the indoor environment have been completed or are underway. In the period since 1988, as is clear from recent publications of the International Commission on Radiological Protection (ICRP), there have been major reassessments of the risk factors due to radiation exposure and specifically from exposure to radon and its decay products. In this period also various national and international agencies have issued recommendations on radon reference and action levels in indoor environments.

Principally for these reasons it was decided by the Steering Committee of the European Collaborative Action "Indoor Air Quality and its Impact on Man" to produce this revised and expanded report on "Radon in Indoor Air". This report deals with a number of aspects of indoor radon, including: sources, measurement methodology and techniques, indoor surveys, risk estimates, recommended/regulatory levels and remedial actions.

CONTENTS

INTRODUCTION

Natural radiation has always been part of the human environment. Its main components are cosmic and cosmogenic radiation, terrestrial gamma radiation from natural radionuclides in rocks and soil, and natural radioactive substances in our diet and in the air we breathe. Until the end of the 1970s the doses received by the vast majority of the general population from natural radiation were considered to be "background" phenomena of little significance, and the average annual dose was estimated to be about 1 mSv. Here and throughout this account the term dose refers to effective dose (see Appendix 1 and 2 for definition of radiation units and radon decay scheme).

Special situations were, of course, known to exist. For example, uranium and other underground miners received large doses, and consequently were subject to elevated cancer risks, due to prolonged exposure to high concentrations in air of the radioactive decay products of the natural radioactive gas radon (NRC 1988). Radon (Rn) is a naturally occurring radioactive noble gas which exists in several isotopic forms. Only two of these isotopes occur in significant concentration in the general environment: radon-222 (usually referred to as "radon"), a member of the radioactive decay chain of uranium-238, and radon-220 (often referred to as "thoron"), a member of the decay chain of thorium-232 (see decay schemes in Appendix 1). Radon is the first and only gaseous and inert element of the radioactive chains, so that it can easily leave the place of production (soil, rock and building material) and enter the indoor air. The contribution made by thoron to the human exposures in indoor environments is usually small compared with that due to radon, due to the much shorter half-life (55 seconds vs 3.82 days), and it will only occasionally be referred to here.

It should be noted that exposure to radon is not a new phenomenon and documentary evidence from as far back as the 16th century indicates that elevated radon exposure was probably responsible for excess lung cancer mortality of miners in some Central European mines, such as the silver mines in Bohemia (see Jacobi 1993 for historical notes).

The view that natural radiation was of little radiological health significance for the general population in most countries changed dramatically with the discovery in the 1970s and 1980s that some homes in a number of countries had indoor radon levels present at concentrations of many hundreds up to some thousands of Bq/m^3 (Gunning and Scott 1982). For people in such houses the doses are in the range of some tens of mSv per year and the associated risk of lung cancer is estimated to be substantially greater than the general population risk for this disease. Surveys in European countries (see below) also reveal that even at typical average indoor radon levels of 50 Bq/m^3 the dose is comparable to the dose of approximately 1 mSv per year from all other natural sources (ICRP 1993, UNSCEAR 1993). It should also be noted that in EU countries the average total dose to the population from artificial sources (excluding medical exposure) amounts to about 0.01 mSv per year, or about 0.5 % of the average total dose from natural sources of which radon is the major contributor to dose. Even the average committed dose to the EU population resulting from the Chernobyl accident (i.e. the total dose that is delivered from the accident beginning to the end of its effects) is about equal to the dose from only 1 year of exposure to radon (UNSCEAR 1988, 1993), so that it represents, on lifetime average, a small percentage.

1 SOURCES

The main source of indoor radon is its immediate parent radium-226 in the ground of the site and in the building materials (Nero 1988, 1989). The outdoor air also contributes to the radon concentration indoors, via the ventilation air. Tap-water and the domestic gas supply are usually radon sources of minor importance, with a few exceptions.

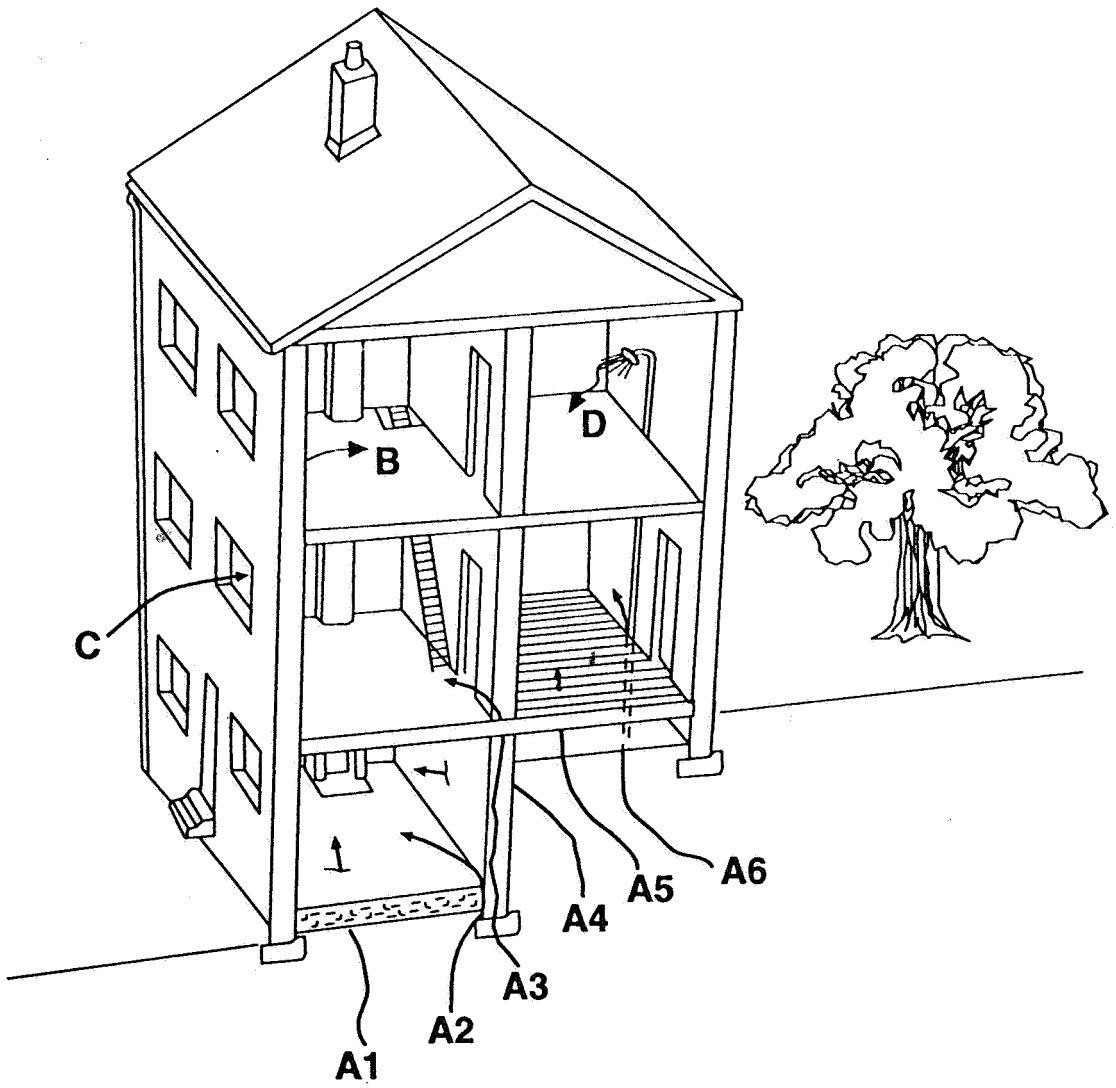
In most situations it appears that elevated indoor radon levels originate from radon in the underlying rocks and soils (e.g. Castrén et al. 1985). This radon may enter living spaces in dwellings by diffusion or pressure driven flow if suitable pathways between the soil and living spaces are present (see Figure 1). It should be noted, however, that in a minority of cases elevated indoor radon levels may arise due to the use of building materials containing high levels of radium-226. Examples of such materials, used in some buildings, are by-product gypsum, alum shale and volcanic tuffs.

The United Nation Scientific Committee on the Effects of Atomic Radiations (UNSCEAR) has made a very simple model to try to estimate the relative contribution of these sources: for a "typical" house, with a radon concentration of 50 Bq/m^3 at ground floor, the contributions of soil, building materials and outdoor air are, respectively, ~60%, ~20% and ~20%, while for the upper floors in high rise buildings, where the radon concentration is estimated to be "typically" 20 Bq/m^3, these values become ~0%, ~50% and ~50% (UNSCEAR 1993).

1.1 Soil

For those who live close to the ground, e.g. in detached houses or on the ground floor of apartment buildings without cellars, the most important radon source is radium in the ground. The radium concentration in soil usually lies in the range 10 Bq/kg to 50 Bq/kg, but it can reach values of hundreds Bq/kg, with an estimated average of 40 Bq/kg (UNSCEAR 1993). Typical radon concentrations in soil gas range from 10000 Bq/m^3 to 50000 Bq/m^3. The potential for radon entry from the ground depends mainly on the activity level of radium-226 in the subsoil and its permeability with regard to air flow. Example of terrains with a high radon potential are alum shales, some granites and volcanic rocks, due to high concentrations of radium-226 and the presence of eskers (gravel, sand and rounded stone deposited from subglacial streams during the ice ages), all these being characterised by high permeability. The ground could also be contaminated with waste tailings from uranium or phosphate mining operations with enhanced activity levels (e.g. Tyson et al. 1993).

The ingress of radon from the soil is predominantly one of pressure-driven flow, with diffusion playing a minor role (de Meijer et al. 1992). The magnitude of the inflow varies with several parameters, the most important being the air pressure difference between soil air and indoor air, the tightness of the surfaces in contact with the soil on the site, and the radon exhalation rate of the underlying soil. If there is no airtight layer between the basement and the ground, the underpressure indoors causes radon to be drawn in from the ground under the building. Underpressure occurs in most houses if either the adjustment of inlet and outlet of air in forced ventilation systems or the outdoor air supply for vented combustion appliances is inappropriate. The underpressure may be considerable for all types of ventilation systems when the inlet air is restricted too much. The tightness of the structures in contact with the ground varies with different building regulations and techniques, and is very dependent on cracks,



A Entry of radon from soil through:

A1 - Cracks in solid floors

A2 - Construction joints

A3 - Cracks and cavities in walls

A4 - Cracks in walls below ground level

A5 - Gaps in suspended floors

A6 - Gaps around service pipes

B Radon exhalation from building materials

C Entry of radon with outdoor air

D Radon released from water

Figure 1. Typical radon sources and entry routes

openings and joints. Structures are hardly ever so airtight that radon inflow is completely prevented. For example, to get a radon daughter concentration of less than 100 Bq/m^3 EER (see Appendix 2) in a house with a volume of 500 m^3 and a ventilation rate of 0.5 air changes per hour, not more than 1 m^3 per hour must be allowed to leak into the house if the radon gas concentration in soil air is about 50000 Bq/m^3. These values are quite typical.

1.2 Building materials

Building materials are generally the second main source of radon indoors, while in the seventies they were considered the principal one (UNSCEAR 1977). Radon exhalation from building materials depends not only on the radium concentration, but also on factors such as the fraction of radon produced which is released from the material, the porosity of the material and the surface preparation and finish of the walls. In general, no action needs to be taken concerning traditional building material. Typical values for radium and thorium content in building materials are 50 Bq/kg or less (NEA/OECD 1979). Building materials containing by-product gypsum (UNSCEAR 1982) and concrete containing alum shale (Swedjemark and Mjönes 1984) may have much higher radium concentration. The activity concentrations in brick and concrete may also be high if the raw materials were taken from locations with high levels of natural radioactivity. Examples of such natural materials, used in some buildings, are volcanic tuffs and pozzolana (Sciocchetti et al. 1983, Campos Venuti et al. 1984, Battaglia et al. 1990), where radium and thorium content can reach some hundreds of Bq/kg. Other measurements of radioactivity content and exhalation of building materials are reported in Ingersoll (1983) and NEA/OECD (1979).

Building materials are the main sources of radon-220 (also called "thoron") in indoor air. Due to its short half life (55 s), thoron originating in soil in effect is usually prevented from entering buildings and therefore makes negligible contribution to indoor thoron levels. For this reason and due to the greater difficulties of measurement, thoron or thoron progeny concentration measurements are very much fewer than those for radon. Although the indoor thoron concentrations are usually low (e.g. Cliff et al. 1992, UNSCEAR 1993), in some cases the doses due to this isotope and its daughters are significant and comparable to those due to radon-222 (Sciocchetti et al. 1983, 1992, Guo et al. 1992, Bochicchio et al. 1993b, Doi and Kobayashi 1994).

1.3 Outdoor air

Outdoor air usually acts as a diluting factor, due to its normally low radon concentration, but in some cases, as in high rise apartments built with materials having very low radium content, it can act as a real source. The radon concentration in outdoor air is mainly related to atmospheric pressure, and (in case of non-perturbative weather) it shows a typical oscillating time pattern, with higher values during night.

Until a few years ago the average level of radon gas concentration in the atmosphere at ground level was usually assumed to be of the order of few Bq/m^3 - e.g. in the range of 4 to 15 Bq/m^3 in USA (Gesell 1983), but more recent measurements seems to indicate higher values, reaching some tens of Bq/m^3 (Hopper et al. 1991, Robé et al. 1992, Bochicchio et al. 1993b, Deyuan 1993, Grasty 1994, Price et al. 1994). Quite high radon concentrations in the outdoor air

have been reported near substantial radon sources, such as mine tailings (e.g. Tyson et al. 1993), or in the case of particular weather conditions, such as thermal inversion or very low precipitation (Grasty 1994).

Ambient air over oceans has very low values (~ 0.1 Bq/m^3) of radon concentration, due to the minimum presence of radium in the sea water and the high solubility of radon in water at low temperatures. Therefore radon concentration in outdoor air of islands and coastal regions is generally lower than in continental countries, e.g. United Kingdom and Japan have an average outdoor air value of ~4 Bq/m^3.

Taking into account recent measurements, the mean value of outdoor radon concentration adopted by UNSCEAR in its last report has been changed from 5 to 10 Bq/m^3 for continental areas and somewhat less in coastal regions (UNSCEAR 1993).

1.4 Tap-water

In wells drilled in rock the radon concentration of the water may be high. When such water is used in the household, radon will be partially released into the indoor air, causing an increase in the average radon concentration. In a few regions, such as Finland and Maine (USA), the tap-water from wells drilled in rock has been shown to contribute significantly to radon concentrations indoors. Radon concentrations in tap-water from deep wells can range from 100 kBq/m^3 to 100 MBq/m^3 (UNSCEAR 1988). The indoor radon concentration in these regions may already be high due to high rates of radon entry from the ground. The world average radon concentration in all types of water supplies is assumed to be 10 kBq/m^3 (UNSCEAR 1993).

1.5 Domestic gas

In some regions, natural gas used for cooking and heating contains elevated concentrations of radon, which is released on combustion. Normally this source is not significant, and it can be monitored at transmission and distribution points. Typically the radon level in natural gas is about 1000 Bq/m^3. Natural gas as supplied usually contains gas from a number of wells and fields and thus can vary over time, depending on the proportions supplied by different sources (UNSCEAR 1993).

2 TYPICAL CONCENTRATIONS

2.1 Dwellings

During the 1980s in many countries surveys of indoor radon levels have been carried out (e.g. Hildingson 1982, McGregor et al. 1980, Put and de Meijer 1984, Schmier and Wicke 1985, Sørensen et al. 1985, Swedjemark and Mjönes 1984, see McLaughlin 1987 for other references). These surveys range in type from small localised short term screening surveys to national surveys in which year long average indoor radon concentrations have been determined in randomly chosen population weighted *representative* samples of national housing stock, which is the recommended methodology (UNSCEAR 1993). National surveys which approximate in character to this latter description have been carried out or are in progress in many European countries - UK (Wrixon et al. 1988), Ireland (McLaughlin and Wasiolek 1988), Italy (Bochicchio et al. 1993), Finland (Castrén 1993), Sweden (Swedjemark et al. 1993) - and non European countries, e.g. the United States of America (Marcinowski 1992) and Australia (Langroo et al. 1991). The preferred method of measurement in such long term surveys is to use passive alpha track detectors which record the alpha activity from radon and its decay products.

Table 1 gives a summary of the principal results from a number of national and regional surveys carried out in recent years in EU Member States, other European countries, North America, Japan and Australia (see also a similar table in UNSCEAR 1993). Unless otherwise stated, the measuring technique in most surveys was based on the use of some form of passive alpha track detector. The summary results presented here are not always directly comparable for a number of reasons. In some surveys the dwellings were chosen in a random and representative fashion, while in others they were chosen from a specially selected group of dwellings. For a number of countries the surveys are still in preliminary stage and the number oh houses surveyed per million inhabitants is rather small. In other countries such as Sweden and the UK the up-to-date total radon dwelling data that has been acquired is much more extensive than the data of the representative surveys presented in Table 1.

National surveys to date have shown that the average indoor radon concentration is in the 10 to 140 Bq/m^3 range. Regional average values above this range have been found in some countries. A good example of this is the UK which has a national average value of 21 Bq/m^3 while Cornwall in south-west England has an average value of about 170 Bq/m^3.

As far as the maximum indoor radon concentration likely to be present in any country is concerned it is impossible to estimate its value. Concentrations greater than 100 000 Bq/m^3 have already been detected in individual dwellings in some countries. In most situations it appears that elevated indoor radon levels originate from radon in the underlying rocks and soils.

While national average indoor radon levels are in the 10 to 140 Bq/m^3 range a small percentage are considerably above this range. In keeping with the current Recommendation of the Commission of the European Communities (CEC 1990), if 400 Bq/m^3 is taken as an "action level" for an existing dwelling (see chapter 8 of this report) then in many surveys the percentage of dwellings in excess of this level ranges from about 0.5 to 3%. Indoor radon levels in most surveys appear to be approximately log-normally distributed; a number of surveys have shown, however, that a log-normal approximation approach may significantly underestimate the percentage of dwellings at the highest radon levels (e.g. Goble and Socolow 1990, Bochicchio et al. 1993, Castrén 1993).

Table 1. Summary of radon surveys in dwellings

Country / Region (population size in millions)	Number of houses sampled	Period and duration of exposure	Sample characteristics	Radon conc. [Bq/m^3]		Geom. Std. Dev.	Percent over 200 Bq/m^3	Percent over 400 Bq/m^3	References
				Average	Geom. Mean				
Australia (17.3)	3413	1989–1990 1 year	stratified random	11	8	2.1	< 0.1 %	< 0.1 %	Langroo et al. 1991
Belgium (10.0)	300	1984–1990 3 months to 1 year	popul. based selected acquaintances	48	37	1.9	1.7 %	0.3 %	Poffijn 1993
Canada (study pop.=7.8)	13457	summer 1977–1980 grab sampling	restricted to 19 maior urban areas, excluding apartments	33	15	3.6	2.3 %	0.6 %	McGregor et al. 1980 Letourneau et al 1984
Czechoslovakia (15.6)	1200	1982 RnD grab sampling	–	140	–	–	–	–	UNSCEAR 1993
Denmark (5.2)	496	1985–1986 6 mo (1/2 houses in winter, 1/2 in summer)	random	47 (dwelling aver.) 53 (population aver.)	29	2.2	2.2 %	< 0.4 %	Ulbak et al.1988
Finland (5.0)	3074	1990–1991 1 year	random	123	84	2.1	12.3 %	3.6 %	Castrén 1993
France (56.9)	1548 (ongoing)	1982–1991 ~ 3 months (using open alpha track detectors)	biased (not stratified)	85	52	2.3	7.1 %	2.3 %	Rannou et al. 1992
Germany (77.4)	a. 6000 b. 1500	a. 1978-1983 3 months b. 1991-1993 1 year	random	50	40	–	1.5–2.5 %	0.5–1.0 %	a. Schmier and Wicke 1985 b. Lehmann 1993
Greece (10.2)	571	1987–1994 6 months	acquaintances	92	68	2.9	3.3 %	1.4 %	Proukakis 1994
Hungary (10.6)	122	1985–1987 ~2.5 years	preliminary survey	55	42 (median)	–	–	–	Sztanyik and Nikl 1993
Ireland (3.5)	1259	1985–1989 6 months	random	60	34	2.5	3.8 %	1.6 %	McLaughlin and Wasiolek1988
Italy (56.8)	4800	1989–1993 1 year	stratified random	77	–	–	5 %	1 %	Bochicchio et al. 1993,1994
Japan (123.9)	6300	1985–1991 1 year	random in a selected group (high school teachers)	29	23 (median)	1.6	<0.4 %	–	Kobayashi et al. 1991
Luxembourg (0.4)	2500	1991	–	–	65	–	–	–	UNSCEAR 1993
Netherlands (15.1)	~1000	1982–1984 1 year	–	29	24 (median)	1.6	–	–	Put et al 1984,1985 UNSCEAR 1993
Norway (4.2)	7525	1987–1989 6 months (spreaded over all seasons)	random	51 (uncorr.) 60 (corr. for exposure period)	26 (uncorr.) 32 (corr.)	–	3.7 % (uncorr.) 5 % (corr.)	1.6 % (corr.)	Strand et al1992 Strand 1993
Portugal (10.3)	4200	1989–1990 1-3 months	volunteers in a selected group (high school students)	81	37	–	8.6 %	2.6 %	Faísca et al.1992
Spain (39.0)	1555–2000	winter of 1988–1989 grab sampling	random	86	41–43	2.6–3.7	–	4 %	Quindos et al. 1991.b UNSCEAR 1993
Sweden (8.4)	1360	1991–1992 3 months in heating season	random	108	56	–	14–26 %	4.8–11 %	Swedjemark et al. 1993
Switzerland (6.6)	1540 (ongoing)	1982–1990 ~ 3 month (mainly in winter)	biased (not stratified)	80 (corr. for sample bias) 70 (corr. for expo. period)	–	–	5.0 %	–	Surbeck et al.1991
UK (57.0)	2093	1986–1987 1 year	random	20.5 (corrected for the pop. housing stock)	15	2.2	0.5 %	0.2 %	Wrixon et al. 1988
USA (249.0)	5694	1989–1990 1 year	stratified random	46	25 (median)	3.1	~ 3.5 %	~ 0.6 %	EPA 1992c Marcinowski et al. 1994

N.B. The duration of exposure in "grab sampling" measurements is usually few minutes.

The observed radon daughter concentrations are usually not in equilibrium (i.e. the radon daughters have a lower concentration than radon), owing to various removal mechanisms which act on them. Radon and radon daughter concentration values are connected by the "equilibrium factor F" (see Appendix 2). Average values reported for the equilibrium factor normally are in the range of 0.3-0.5 for residential buildings in different areas (UNSCEAR 1988).

2.2 Workplaces

All previous radon concentration values refer to dwellings, because people usually spend most of their time there. However in the last years, in some countries, there has been an increase in the number of measurements being carried out in normal workplaces such as schools, offices, etc. (e.g. Strand and Kolstad 1991, Gooding and Dixon 1992, Poffijn et al. 1992, SSI 1993). As discussed in a later chapter on recommended and regulatory radon levels the International Commission on Radiological Protection (ICRP) has recommended that the action level for intervention in the workplace where the occupancy of members of the public is low - e.g. in offices, libraries and theatres - should be in the range 500 to 1500 Bq/m^3 (ICRP 1993).

3 HEALTH EFFECTS AND RADON RISK ESTIMATES

3.1 Introduction

Radon is one of a very small number of substances which have been established to be human carcinogen on the basis of human studies. As such it is a Group 1 and Group A carcinogen, according to the classification used by the World Health Organisation (WHO/IARC 1988) and by the US Environmental Protection Agency (EPA 1987), respectively. The principal adverse health effect arising from the inhalation of radon and mainly its decay products is lung cancer. Recent suggestions (Henshaw et al. 1990, 1992) that exposure to elevated levels of indoors radon may be implicated in the occurrence of other cancers such as childhood leukemia have not yet been scientifically verified.

As shown in the decay schemes in Appendix 1, the gas radon decays to produce a series of decay or daughter products. From a health perspective the daughter products of most significance are the four short-lived ones polonium-218 to polonium-214 inclusive, which are referred to in various ways: radon daughters, radon progeny, radon decay products. These elements, unlike radon, shortly after their formation attach themselves to aerosol particles; only a small fraction of them remain in unattached form, depending on aerosol size and concentration and on ventilation (Nazaroff and Nero, 1988).

When radon and its short-lived decay products are inhaled the radiation dose to lung tissue is dominated by the alpha particles emitted by the deposited decay products, which cause, especially those ones attached to small size aerosols or in unattached form, damage to sensitive lung cells, thereby increasing the probability of cancer developing. The dose to the lung due to the beta and gamma radiation emitted by the decay products is comparatively negligible for two reasons: a) the energy of these radiations are much smaller than the energy of alpha particles; b) these type of radiation interact with matter less strongly than alpha radiation and therefore they release only a small part of their energy to the lung.

It has to be underlined that the contribution to lung dose arising from the radon gas itself is small in comparison, as very little radon is absorbed by lung tissue because it is an inert gas and for the same reason unlike its decay products it cannot be adsorbed onto lung airway surfaces. The principal role of radon is to carry itself and therefore its short-lived decay products from soil and building material to the indoor air and finally to the lung. The dose to the lung due to all the decay products following polonium-214 is comparatively negligible because of the very long decay time of lead-210 (22.3 years).

For the deposited radon decay products - which are short-lived and therefore mainly decay in the lung tissue - the concept of Potential Alpha Energy (PAE) is used (see Appendix 2). PAE is the total alpha energy emitted by the decay of each decay product along the decay chain down to lead-210. For example, an atom of deposited polonium-218 has a PAE of 13.7 MeV (i.e. 6.00 + 7.69) while even beta emitters such as lead-214 and bismuth-214 each have a PAE of 7.69 MeV as they ultimately give rise to the alpha decay of polonium-214.

Currently there are different approaches used to estimate the lung cancer risk arising from exposure to radon decay products in indoor air. These are: a) the dosimetric approach, in which the radiation doses to lung tissues is estimated and from this estimated dose the associated risk is evaluated using currently accepted dose/risk factors for ionising radiations; b) the miner

epidemiology approach, in which risk estimates for underground miners are modified and applied to the general population; c) the residential epidemiology approach, in which case-control studies of the general population are used to estimate risk factors. The risk factors obtained using these three approaches seem to be reasonably well in agreement (see below in this chapter).

Applying these risk/exposure factors to the typical average indoor radon concentrations in North American and European countries, a not insignificant fraction (typically of the order of 10%) of total lung cancers can be attributed to radon and its decay products exposure. However, it should be strongly emphasised that the majority total lung cancers are due to smoking. For example, in a country of 50 millions with a lung cancer lifetime risk of 3% (which is the value assumed by ICRP for its "reference" population), we can estimate that 3 persons per 1000, that is about 2000 each year, may die because of lung cancer due to radon exposure. It has to be underlined that these figures are not precise, and there is an associated uncertainty that will be discussed later in this chapter.

A synergistic effect seems to occur, in a greater or lesser degree, between radon and cigarette smoking both in mines and dwellings, so that smokers exposed to radon have probably a higher risk (6-10 times) than non-smokers (ICRP 1991, Pershagen et al. 1994). However the numerical estimates of this synergism are still very uncertain.

There are three different approaches (see Figure 2) used to estimate the risk due to the exposure of general population to radon progeny in indoor air. In each case the first step consists in evaluating the exposure.

3.2 Exposure evaluation for risk assessment

The indoor exposure E in the time interval t is evaluated by the following formula:

$$E = C_{Rn} \cdot IOF \cdot t$$

where C_{Rn} is the average indoor radon concentration and IOF is the indoor occupancy factor, i.e. the fraction of time spent indoors.

To assess the risk, a reliable estimation of the indoor C_{Rn} is needed (see also chapters 4,5,6). Therefore it is advisable to carry out measurements of long duration (at least one year), so that fluctuations due to climate, meteorological situation and lifestyle in the dwelling can be duly averaged. Short time (i.e. for a few days) measurements, which are usually performed with closed windows and doors, should be avoided as they could give rise to significant overestimates of C_{Rn} (Sextro 1990). Moreover the measurements have to be carried out with adequately calibrated and reliable instruments, and intercomparison exercises should also be regularly performed.

The occupancy factor cannot in practice be measured, because it would require to keep a rigorous diary of activity for a too long period, but it may be estimated by the responses given to a questionnaire during radon surveys (e.g. Benassai et al. 1990, EPA 1992d), or by following special inquiries (Francis 1987).

In radon epidemiological studies a major problem with exposure evaluation exists because radon levels in the past, when the subjects received most of their exposure, may differ significantly from those at present in their dwellings. In Sweden, for example, it has been shown that the average indoor radon levels seem to have increased in recent decades probably because

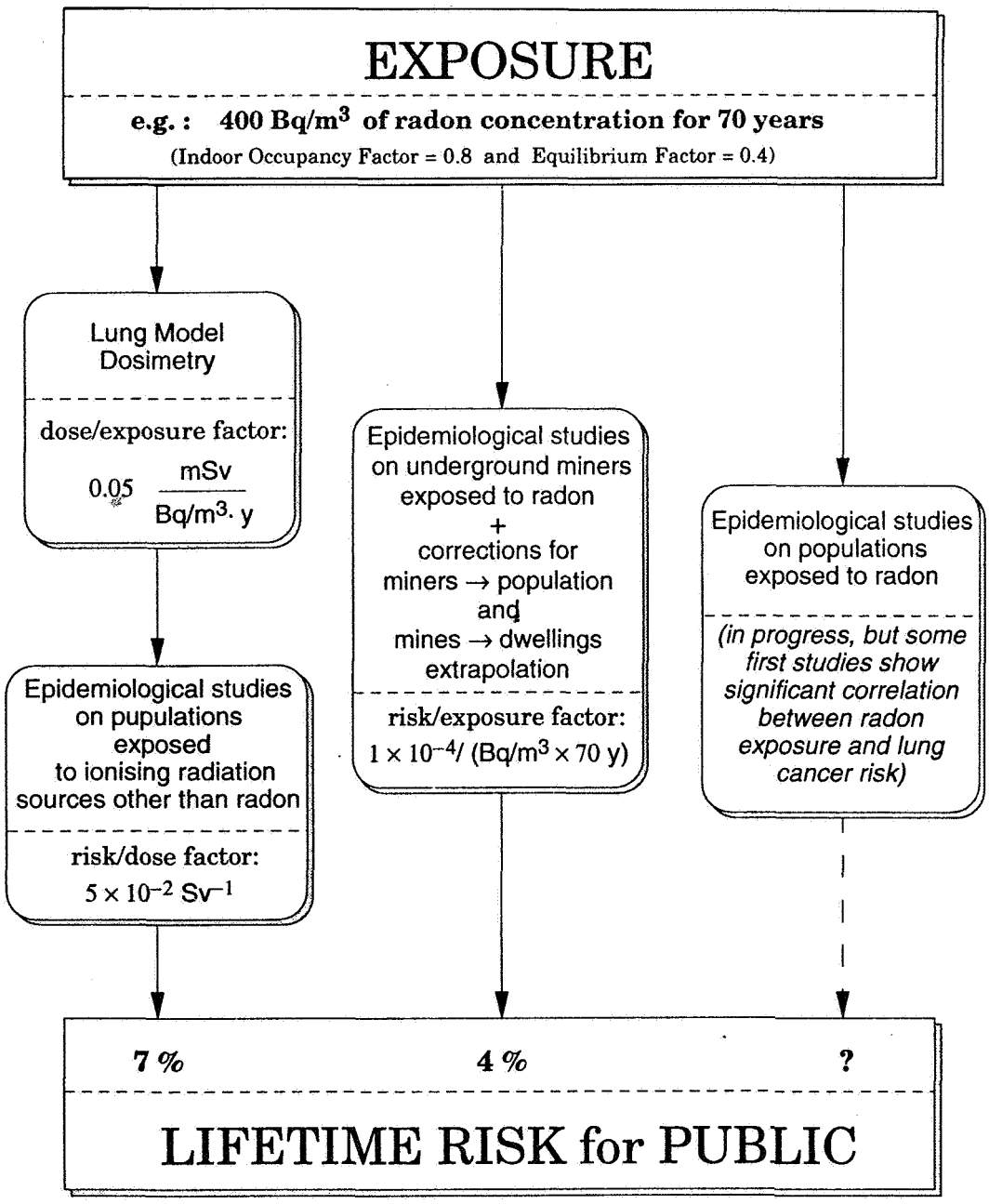


Figure 2. Example of lifetime risk assessment for a chronic exposure (70 years) of general public to radon indoors

of energy conservation practices (Swedjemark and Hubbard 1993). To address this problem techniques are currently under development, with the support of the CEC Radiation Research Programme, to make retrospective assessment of radon exposure in dwellings, based on the measurement of polonium-210 levels in the surface of glass (Samuelsson 1988).

As an example of risk assessment calculation (see Figure 2), it will be considered a lifetime (70 years) exposure to 400 Bq/m^3 - the CEC action level (see chapter 8 of this report) for existing dwellings (CEC 1990) - with an indoor occupancy factor of 0.8, which is the value adopted by UNSCEAR (UNSCEAR 1988, 1993) and ICRP (ICRP 1993).

3.3 Risk estimation (a): the dosimetric approach

One of the three ways to estimate the risk due to the exposure of a population to radon progeny in indoor air is carried out in two stages. In the first one the absorbed dose to the lung is calculated through very complex models, which take into account both physical parameters (radon and radon progeny concentration, fraction of progeny attached to aerosols, unattached fraction, aerosol size distribution, ...) and physiological parameters (characteristics of the respiratory tract, respiratory rate, thickness of bronchial epitelium, location of target cells, ...). These models (e.g. NEA/OECD 1983) are continuously evolving. An improved model of the respiratory tract for use in a wide range of circumstances has been recently adopted by ICRP (ICRP 1994).

It has to be emphasised that although the dose is mostly due to inhaled radon progeny (and its characteristics) rather than to radon, some studies (James 1987, Vanmarke et al. 1989, Postendörfer and Reineking 1992) show that, in domestic ambient air, changes in ventilation rate produce opposite variations in the equilibrium factor F (i.e. the ratio between radon progeny and radon gas concentration) and in the unattached fraction, so that the absorbed dose (defined in Appendix 2) to the lung remains relatively constant at a given radon concentration. After absorbed dose is calculated by models, weighting factors (related to the specific effects of alpha particles and to the lung sensitivity to radiation, as explained in Appendix 2) have to be applied in order to obtain the "effective dose", which is the quantity considered to be proportional to the health effects (ICRP 1991).

Different values of the effective-dose/exposure factor have been proposed (see UNSCEAR 1988 and 1993 for a review). Here, for example purposes only, the value adopted by CEC (CEC 1990) and NRPB (Wrixon et al. 1988) will be used, i.e. 0.05 mSv/y per Bq/m^3 of mean annual radon concentration, with an indoor occupancy factor equal to 0.8 (see Figure 2). Therefore, for 400 Bq/m^3 the effective dose is 20 mSv/y. Another widely used dose factor is 0.025 mSv/y per Bq/m^3, that is adopted by UNSCEAR (1993).

In the second stage of the dosimetric approach the risk connected with the effective dose to the lung is evaluated. This evaluation is made using the results of the epidemiological studies on persons exposed to ionising radiation, mainly studies on survivors of Hiroshima and Nagasaki, although in this case the exposure conditions were very different from those due to residential radon. In particular H. and N. survivors were exposed to gamma and neutron radiation, while the main radiation emitted by radon progeny is alpha radiation. In the last Recommendations of ICRP (ICRP 1991), a review of the most recent studies is made, adopting a risk/dose factor for general public of 5×10^{-5} probability of fatal cancer per mSv of effective dose. Using this figure, 20 mSv/y would imply a 1×10^{-3} annual risk, corresponding to a 7% lifetime risk for a chronic

exposure of 70 years.

The main problems of the dosimetric approach are the appropriate choice of the weighting factors to obtain the effective dose (Birchall and James 1994) and the use risk/dose factors derived from epidemiological data for persons (such as the Japanese survivors) exposed to radiation other than that from radon progeny. For these and other reasons the present position of ICRP (ICRP 1993) is to use only the miner epidemiology approach (see below) to estimate the risk. Nevertheless, ICRP introduces a "dose factor convention", which is derived so as to obtain the same risk from the dosimetric approach as from the miner epidemiology approach.

3.4 Risk estimation (b): the miner epidemiology approach

Another approach for the evaluation of risk is the analysis of epidemiological data of persons exposed to radon progeny. However for historical and other reasons, epidemiological studies on underground miners mainly have been available till now, except for some recent results of epidemiological studies on general population. In fact, as described in the introduction, it is only in the last 10–15 years that an increasing attention has been devoted to radon exposure at home, while the knowledge of radon effects on underground miners dates back many years. Moreover, in underground mines, radon concentrations were usually quite high(*) - due to the fact that the main radon source is the ground itself - so that the related effects, in terms of excess lung cancers, are more evident. For the principal miner cohorts studied more than 700 excess deaths from lung cancer have been recorded (ICRP 1993). It is worth noting that this excess is considerably higher than the deaths from all cancers that have been attributed to radiation in the lifespan study of the Japanese atom bomb survivors (ICRP 1991).

Several cohorts of miners have been and continue to be analysed over the years. In order to apply the results to radon exposure of the general public in dwellings, the miner data have to be corrected to take into account the differences between miners (strong adult males) and the general public, and between indoor air characteristics in mines and in dwellings. In these studies the risk/exposure factor is usually reported per unit of cumulated exposure to radon progeny, the WLM unit, because radon progeny concentration is the quantity measured in mines. Hereafter this factor will be converted and expressed per unit of radon gas concentration (as described in the Appendix 2), using an equilibrium factor of 0.4 and an indoor occupancy factor of 0.8, which are the values adopted by UNSCEAR (1988) and ICRP (1993).

The results of the three major and more recent studies, performed by ICRP and the U.S. Committee on Biological Effects of Ionizing Radiation (BEIR), can therefore be expressed as 0.72×10^{-4} (ICRP 1987), 1.1×10^{-4} (NRC, 1988) and 0.87×10^{-4} (ICRP 1993) lifetime risk for a chronic exposure of general population during 70 years to 1 Bq/m^3 of radon gas concentration. These studies, whose results can be considered in good agreement, will not be analysed in depth. It should be noted, however, that different baseline cancer rates are used to convert the results from *relative* risk to *absolute* risk: ICRP (1987) refers to a reference population with a baseline cancer rate of 400 cases per year / 10^6 persons, NRC (1988) refers to the USA population only, while ICRP (1993) refers to an "average population" composed by populations of United States, Puerto Rico, Japan, the United Kingdom and China. Moreover ICRP (1987) uses a correction factor of 0.8 to extrapolate the miner epidemiology results to general public in dwellings, while in

(*) Of course in recent years improved ventilation, etc, has been installed in mines in order to reduce the risk.

the other two studies this correction factor is assumed to be equal to one (although different decisions have later been taken, as outlined in a subsequent paragraph). In the example under consideration (see Figure 2), an approximate lifetime–risk/exposure factor of 1×10^{-4} per Bq/m^3 during 70 years will be used, obtaining a lifetime–risk of 4% for a chronic exposure to 400 Bq/m^3 of radon concentration.

It is of interest to note that a possible new miner epidemiological data set for radiation risk estimation of radon exposure has recently become available in Germany (Martignoni et al. 1991). As one of the results of the reunification of Germany, access has now become available to the exposure data on uranium miners who worked at various times during the period 1946 to 1989 in the Soviet controlled "Wismut" uranium mining and milling consortium in the former German Democratic Republic. During the peak production years (early fifties) the work force, including forced labour, amounted to between 100000 and 150000 persons. During this period it is roughly estimated that annual exposures of 30 to 300 WLM were being received by the workers. The exposures reduced in subsequent years. Initial studies indicate that these "new" German uranium miner data are of a similar or probably higher quality than the data from other epidemiological studies concerned with the risk of bronchial cancer in miners. The size of the German miner population is much higher than all other known miner cohorts as is their known number of bronchial cancer cases.

3.5 Risk estimation (c): the residential epidemiology approach

The difficulties and the uncertainties in extrapolating the epidemiological results from miners to the general public and the growing interest in the radon problem in dwellings have recently induced many countries to undertake epidemiological studies on the general population exposed to radon progeny in dwellings. Many of these studies are coordinated, e.g. the same protocols are used, in order to facilitate the pooling of the results (DOE/CEC 1989, 1991). These studies take many years to complete and are difficult, thus for most of them, which are underway at present, the final results will not be available in the immediate future. These kinds of studies are expected to produce the most appropriate risk estimation.

A good example of this approach is a study carried out in Sweden (Pershagen et al. 1992). This study included 210 women with lung cancer and 191 hospital and 209 population controls. For the study subjects the radon concentrations measured in their various lifetime residences showed an approximately log-normal distribution with arithmetic and geometric means of 122.2 and 96.0 Bq/m^3 respectively. Even at these slightly elevated indoor radon levels the risk estimates appeared to be within the range expected on the basis of the miner studies. In a more extended study (Pershagen et al. 1994), which includes 1360 (586 female and 774 male) lung cancer cases and 2847 controls, the risk estimate is lower, but still statistically significant and consistent with the previous study.

3.6 Uncertainties in radon risk assessment

Uncertainties in risk assessment are connected in a number of ways as described above. In the following a brief analysis of the various sources of uncertainty is presented.

Exposure uncertainties

The uncertainties in exposure are due not only to measurement techniques, but also to the procedure used to estimate personal exposure. Actually no "personal dosemeters" are used, but "environment measurements" and occupancy factors are utilised to obtain person exposure, especially for long periods of time. A tentative to compare "personal monitoring" with "environmental monitoring" has been recently published (Litt et al. 1990), but it refers to an exposure of few days. The uncertainties can be summarised as follows:

Measurement technique. Passive track dosemeters - in which alpha particles emitted by radon and radon progeny produce tracks that are subsequently counted - are usually utilised for long term measurements. The overall measurement uncertainty due both to calibration and reproducibility usually ranges from 10% to 30% (one standard deviation), depending on the actual radon concentration and other factors.

Measurement period. As previously underlined, one-year integrated measurements are usually performed to get good estimates of the mean radon concentration. However the results could differ from year to year, for instance due to strong climate changes. For example, measurements carried out for 5 year period in 40 residences near the DOE Radon Laboratory of Colorado show a mean coefficient of variation approximately 22% (Martz et al. 1991). Moreover dosemeters continue to measure even when persons are not at home, and this could introduce a bias in case of significant difference in radon concentration with respect to the period of the day when persons are at home or at work.

Retrospective assessment. As previously mentioned, retrospective assessment of exposure is required in case-control epidemiological studies. Usually this assessment is made measuring at present the radon concentration in all dwellings used in the period under study. This procedure could introduce a high bias, that can be tentatively limited if a strict protocol for case and control selection is used, i.e. excluding all cases and controls who lived in houses that have had any significant structure change that could effect radon concentration. An alternative experimental technique for retrospective assessment of radon exposure based on the build-up of polonium-210 on glass surfaces in dwellings is under development (Samuelsson 1988). A similar approach is also now under development in which the build up of Po-210 in porous materials (volume trap) in dwellings is measured as an aid to retrospective assessment of radon exposure.

Sampling location. Radon detectors are usually positioned in one room of the house, preferably in bedrooms where most time is spent. This introduces a bias in those cases where the radon concentration varies appreciably from room to room, as it could happen in multistory dwellings, where significant variations could exist between the ground and upper floors.

Occupancy factor. As outlined above, the occupancy factor is very difficult to measure in practice. Personal judgment is often the only way for its estimation, especially in case of long periods. This factor is widely variable among persons, as it is strictly linked with age, occupation, state of health, etc. Moreover, it could differ significantly during weekends and holidays. However, when averaged over the general population, it is relatively constant. It still depends on climate, being usually higher in cold climate countries. As mentioned above most authorities for a first approximation assume an occupancy factor of 0.8, being made up of about 0.6 at home and 0.2 in other indoor situations (ICRP 1993, UNSCEAR 1993).

Dosimetric approach uncertainties

a) dose/exposure factor

As mentioned above, lung models are continuously evolving, as well our knowledge of the relevant physical and biological-physiological parameters, with an impact on the dose/exposure factor value; in particular a great uncertainty is related to the choice of target cells to be considered (James 1987, ICRP 1993, UNSCEAR 1993) and of the weighting factors used to obtain effective dose from the calculated absorbed dose (Birchall and James 1994).

Often the most recent models predict higher values than the older ones. For example the factor used by UNSCEAR in its report of 1982 (confirmed also in 1988 and, with some warning, in 1993) is about 50% lower than that one adopted by CEC in 1990. Moreover, the model results are usually referred to adults only, whereas the dose factor for the age group from 0 to 10 years could be up to 2 times higher. The range of the dose/exposure factor values can be estimated to be within a factor of about 3 (UNSCEAR 1988).

b) risk/dose factor

Radiation risk/dose factors (e.g. ICRP 1991) generally have been obtained mainly from the epidemiological data of Hiroshima and Nagasaki survivors, whose exposure conditions were very different from those related to radon in dwellings: high dose rates for a relatively short time in contrast to chronic exposure at low dose rates, and neutron and gamma radiation in contrast to alpha radiation. These differences of exposure introduce uncertainties that are difficult to quantify.

Moreover, some uncertainties are related to estimation of the dose received by the exposed persons, and to the confounding factors, such as smoking habits and so on. Another source of uncertainty is due to the fact that not all of the exposed persons are dead, so that the risk value, calculated on the basis of the number of cancers appeared to date, has to be extrapolated, or projected, to take into account the cancers that will appear in the future. Different risk-projection models produce different lifetime risk factors from the same data. The most recent studies show that the "relative risk" models - where the cancer risk due to radiation exposure in related to the baseline cancer rate of non exposed persons - fit the data better than the "absolute risk" ones - where there is no relation with the baseline cancer rate.

On the basis of the above considerations and of new epidemiologic and dosimetric data, the risk/dose factor adopted by the International Commission on Radiological Protection has changed from 1.3×10^{-2} Sv^{-1} (ICRP 1978) to 5×10^{-2} Sv^{-1} (ICRP 1991).

Epidemiological approach uncertainties

Risk factors obtained from the epidemiological studies based on occupationally exposed individuals in underground mines, as well as for the general population, also have many uncertainties. Large uncertainties exist, for example, in estimated cumulated exposure to radon progeny for miners, especially for the period ~1950-1960. Moreover, the presence inside the mines of other co-carcinogenic agents – like metal dust, diesel engine fumes, long-lived radionuclides, gamma radiation and, above all, tobacco smoking – make it more difficult to quantitatively correlate lung diseases to radon occurrence.

For epidemiological studies of the general population contemporary radon measurements in their homes or previous homes may be different from those present in the preceding decades of exposure. Some recent work on retrospective assessment of radon exposure is attempting to address this question (Samuelsson 1992).

Another major source of uncertainty is the extrapolation of the risk from adult male miners in underground mines to the general public in dwellings. This extrapolation is not easy, due to difficulties in quantifying the effects of the differences of the two environments. Over the last decade many different values of an appropriate correction factor were estimated, ranging from 0.65 to 1.4 (for a summary see ICRP 1993). For example, ICRP adopted at first a correction factor of 0.8 (ICRP 1987), while now it assumes a value of unity; on the other hand, the U.S. National Research Council assumed at first a default value of unity (NRC 1988), while later a 0.7 value was adopted (NRC 1991).

Some studies concerning systematic examinations of the full range of the epidemiological data (for a review see Burkart 1989, ICRP 1991, ICRP 1993) give rise to risk estimates which are different within a factor ~3, attributable not only to the above reasons, but mainly to different risk-projection models and, in case of relative risk models, to different baseline lung cancer rates that are used. The most recent studies, all using the relative risk-projection models, agree to within a factor less than 2 (ICRP 1987, 1993, NRC 1988).

3.7 Overall uncertainties and conclusions

We tentatively estimate that currently the overall uncertainty of the risk factor of general population due to radon exposure is probably less than a factor 3. It should be noted that the uncertainty of the radon risk factor is higher for specific groups such as smokers or non-smokers, due to a not yet well quantified synergism between radon and smoking. Uncertainties of this magnitude are not uncommon in estimates of the risk due to radiation or other causes, as chemical substances and so on, where often the uncertainties are much higher. It has to be underlined that the risk assessment is made on human data at exposure values in the range of values found in dwellings.

The results of the dosimetric and miner epidemiology approaches could be considered to be reasonably well in agreement. However, ICRP has decided, in 1993, to use only the miner epidemiology approach to estimate the risk, at present. It is anticipated that the epidemiological studies on general populations, the first of which are just concluded but most are now in progress, will yield more appropriate risk estimates. To date only a small number of such general population epidemiological studies have been completed (e.g. Pershagen et al. 1992, 1994) and while no major conflict with the miner epidemiology has emerged, it is prudent to await the outcome of the ongoing studies.

4 INDOOR RADON MEASURING METHODOLOGY

4.1 Introduction

As explained in earlier chapters of this report radon gas is usually present in indoor air, coming mainly from soil through small cracks in building foundations and driven by pressure differences, or from building materials, or from domestic use of deep well water through its emanation, or from outdoor air.

Radon atoms decay through a chain, the first four short-lived elements of which, are referred to in various ways: radon daughters, radon progeny, radon decay products. These elements, unlike radon, are not inert gases and shortly after their formation attach themselves to aerosol particles; only a small fraction of them remain in unattached form, depending on aerosol size and concentration and on ventilation (Nazaroff and Nero 1988). These daughters, when inhaled together with air, decay in the lung emitting radiations, in particular alpha particles, and cause, especially those ones in unattached form, damage to sensitive lung cells, thereby increasing the probability of cancer developing. However although the dose is mostly due to inhaled radon progeny (and its characteristics) more than to radon, some studies (James 1987) show that, in domestic ambient air, changes in ventilation rate produce opposite variations in the equilibrium factor F (i.e. the ratio between radon progeny and radon gas concentration) and in the unattached fraction, so that the dose to the lung remains relatively constant at a given radon concentration.

In IAQ (Indoor Air Quality) investigations, radon and/or radon daughter concentrations in indoor air are usually measured(*), depending on the aim and the available equipment. In any case the radon concentration is related to those of the radon progeny through the equilibrium factor F, whose values are usually in the range 0.2-0.7 for domestic air. Usually F=0.5 was used, in which case the radon concentration is the double of the radon progeny concentrations (see also Appendix 2). More recently F=0.4 is preferably assumed (UNSCEAR 1988, 1993, ICRP 1993).

In this chapter the general methodology of radon (and radon daughter) concentration measurements for a single dwelling or building is outlined, while in the next chapter some main measurement methods, instruments and detectors will be illustrated. A correct methodology is of great importance in measuring radon concentration in indoor air. For a more detailed analysis, including some measuring protocols, specialised publications should be consulted (e.g. NEA/OECD 1985, NCRP 1988, EPA 1987, EPA 1989, DOE 1990).

4.2 Radon concentration temporal variations and measurement duration

For a correct approach to IAQ investigation, concentration variations of radon and its progeny have to be taken into account. These variations occur on many time scales, from hourly to annually, and depends on seasonal factors and weather conditions, building characteristics, operation of dwelling heating and refrigerating systems, living habits, source relative strength, etc. Usually concentrations are higher in the evening and night than in the late morning and first afternoon, higher in the winter than in the summer; however it is nearly impossible to predict these variations with enough accuracy.

(*) In some cases, in order to better identify radon sources and entry points, other kind of measurements could be done, such as radon or radium concentration in water and radon flux from walls and floors.

The longer the duration of a measurement the lower in general is its variability: individual very short-time measurements (~minutes or few hours) can show variations higher than a factor of 10. Measurements lasting a few days generally show variations less than a factor of 10 (e.g. Sextro 1990), while the seasonal variations found in measurements of six-months duration are usually well within a factor of 5 (e.g. Hess et al. 1985, Bochicchio et al. 1992), while annual variations are well within a factor of 2, e.g. measurements carried out for a 5 year period in 40 residences in Grand Junction (Colorado, USA) show a mean coefficient of variation approximately 22% (Martz et al. 1991). Because of the magnitude of these variations, one-year measurements are considered the best compromise to estimate the "average" value.

4.3 Length of sampling time

Radon and/or its progeny concentration measurements of four types of duration can be used, depending on the purpose: long-term, short-term, very short-term and continuous monitoring.

a) ***long-term*** (typically one year or some months): these measurements give the best estimates of the average value. In particular one-year ones are the most appropriate, except the cases in which the dwelling is not lived in for a long period of the year. If for some purposes a quicker estimation is needed, e.g. for screening purposes, a measurement period shorter than one year could be used. This requires, however, a careful seasonal correction factor in order to estimate the average value, thus introducing a potentially high uncertainty that depends on the way the correction factor is obtained. This procedure can be considered reliable only in cases where many previous determinations of seasonal variations in the same area and in the same type of dwellings were made with results in a narrow range. Long-term measurement instruments are usually available only for radon concentration, because the long-term integrated measurement of radon progeny is very difficult due to technical and scientific problems.

b) ***short-term*** (typically from 1 to 10 days): these measurements can be used for screening purposes, i.e. to quickly discriminate between very low, medium and very high concentrations. Another use of screening measurements is to quickly select in a group of dwellings or rooms those one with higher radon concentration. To minimise underestimation, protocols usually suggest measuring conditions tending to maximise radon concentration (EPA 1987). The results of short-term measurements cannot be used to accurately estimate the long-term average value, and a follow-up long-term measurement is needed (EPA 1987b, 1992, White et al. 1994).

c) ***very short-term or grab-sampling*** (typically some minutes or tens of minutes): these are generally used only for detailed building diagnostic and/or research purposes.

d) ***continuous monitoring:*** the main purposes of this type of measurement are to control particular occupational environments, e.g. mines, and/or research activity. It could also be used for some of the purposes of short-term and grab-sampling measurements.

4.4 Number and location of measuring points

The choice of the type of rooms and points where measurements have to be performed is related to the objective of the investigation.

For *human exposure assessment*, only rooms where people live should be selected, in

particular bedrooms, where most time indoors is usually spent. Kitchen and bathrooms should not be selected, also because the particular air conditions present in these rooms can affect the results of some detectors. The measuring points should also be representative of the air inhaled by the people. Moreover the detector location should be undisturbed and far enough from heat and draft sources.

For *building / dwelling investigations,* instead, non-living rooms and other closed spaces, as basements and cellars, can be also selected for measuring radon concentration. Moreover the measuring points can be selected in order to find-out radon inlet routes.

The number of of rooms and measuring points depends on the type and size of the building/dwelling.

For *residential buildings* one or more rooms (bedroom, ...) are needed in case of one-storey dwellings, where the radon concentration is usually homogeneous, while two or more rooms (bedroom, living room, ...) are suggested in case of two-storey detached dwellings, where the radon concentration is usually higher on the ground floor (where there is the living room) than on the first floor (where there are the bedrooms). Similar considerations are valid for small offices too.

For *non-residential large buildings*, such as schools, large offices, large stores, hotels, etc., more points and rooms have to be selected, depending on the size of the buildings and the size and number of rooms, the type of building construction, the type of ventilation/heating systems and so on. For example the U.S. EPA recommend for schools that all rooms on or below ground level have to be measured (EPA 1989b). For other large buildings some preliminary protocol has recently been proposed (Wilson et al. 1993).

Usually one detector per point is used, but the use of two (or eventually more) detectors produce a more reliable measurement (see also below), and could be suggested depending on the objective, the resources and the detector type.

4.5 Measurement quality assurance

A quality assurance program is needed to assess and guarantee the measurement validity, in particular its precision and accuracy (e.g. NCRP 1988, EPA 1989).

As regards measurement *accuracy*, detectors/instruments have to be first of all calibrated, then checked frequently. Periodic participation in measurement intercomparisons is also to be recommended. Calibration measurements are usually performed in a "radon chamber", in which a known quantity of radon is introduced. In general the calibration of radon daughter detectors is more difficult than the calibration of those for radon, due to many reasons, one of which is that there is no primary standard for radon daughters (NCRP 1988, Nazaroff 1988).

As regards measurement *precision*, it is assessed by replicate measurements, i.e. exposing many detectors contemporaneously to the same concentration. This allows an estimate to be made of the coefficient of variation of the detectors at that value of concentration (Goldin 1984). A periodical check of the precision should also be done.

5 CATEGORIES OF MEASUREMENT

Most methods of measuring radon (Rn) and its decay products (RnD) are based on the detection of the alpha particles emitted by these radionuclides during their radioactive decay. A small number of methods are based on the detection of emitted gamma rays and some techniques exist which detect beta ray decays. Useful reviews of these techniques are those of NCRP (1988) and Nazaroff (1988).

It is important to distinguish between methods which measure the concentration of the gas radon and those which measure the concentrations or other characteristics of airborne radon decay products. In Rn and RnD measurement the techniques may be classified as being active or passive. ACTIVE techniques are those which require electric power and/or the use of air pumps to collect activity from the air. PASSIVE techniques are those where the detector while installed at the sampling location does not require electric power. The air containing Rn and RnD usually enters such passive detectors by free diffusion and the radiation detecting medium itself does not require any power supply (i.e. alpha track plastics or activated charcoal). Passive techniques are usually simple, cost effective and easy to use. They are admirably suited for survey work and for long term measurements. Some instruments are available where the detector requires a low level of power supplied by a battery but no pump is used. These fall into a class of detection technique intermediate between passive and active.

It is also important to distinguish between the different sampling techniques in terms of their temporal characteristics. These may be divided into three principal types:

a) ***Grab sampling.*** Here the activity of Rn or RnD in a discrete sample of air taken at a single location in a short period of time (from about 1 sec to about 20 min) is measured. This approach is at best useful for initial screening purposes or for spot-checking of the efficacy of remedial actions. It is of very limited use for determinations of average indoor air radon concentrations which are more appropriately determined on the basis of long-term measurements.

b) ***Continuous sampling.*** Here air is drawn either continuously (or semi-continuously) for long periods of time through a Rn or RnD detecting instrument. This type of approach gives information on the time dependence of the airborne activities in a building. Information which may be obtained using continuous sampling include the ratio between day-time and night-time concentrations in a building, diurnal variations etc. Such information is quite useful in deciding on strategies used to reduce occupational exposures where occupancy factors are much less than in dwellings.

c) ***Time integrating sampling.*** Techniques using time integration consist of using a device which will yield a single determination of airborne activity averaged over some chosen period from a few days to a year or longer. As time integrating sampling is usually but not exclusively carried out with inexpensive passive detectors it is the preferred approach in survey work. In reaching a decision on the necessity of remedial action it is generally considered in European Community countries that integrating measurements of minimum duration 3 months should be made. Grab sampling or short-term integrating sampling of a few days are considered inadequate for making accurate estimates of long term exposure of occupants of a building to radon. The recommendations of the Commission of the European

Communities (CEC) of February 1990 stress the need and desirability of making long-term (generally one year) integrating measurements in order to determine if indoor air is above or below the appropriate radon reference or actions levels (CEC 1990). In countries such as the U.S. where real-estate transactions may require urgent evidence of the indoor radon concentration in a building short-term integrating techniques of a few days duration may be recommended providing they are carried out according to a recognised measurement protocol (White et al. 1990).

6 INSTRUMENTATION

The following are brief descriptions of the principal characteristics of the instruments or devices most commonly used to measure concentrations of indoor Rn and RnD.

6.1 Scintillation cells

This is one of the oldest and most reliable type of device for measuring the concentration of radon gas. It exists in a number of forms and can be used for grab-sampling or for continuous long term measurements. A radon scintillation cell typically is a small metal cylinder equipped with one or two vacuum-tight inlet/outlet valves mounted on one end. The opposing end of the cylinder is a clear glass or plastic window. The internal volume of such a cell is typically about 100 cm^3. The inner surfaces of the cylinder, excepting the window, is uniformly coated with ZnS(Ag) powder which is a very efficient scintillator for alpha particles. The most common form of such a scintillation cell is called a Lucas cell (Lucas 1957). The scintillation cell filled with an air sample is placed in optical contact with a photomultiplier tube (PMT). The scintillations or flashes of light caused by the alpha particles from radon, Po-218 and Po-214 which strike the ZnS(Ag) are recorded by the PMT and its associated electronics. Using appropriate calibration and decay scheme factors the radon gas concentration may be determined from the rate at which the pulses are recorded.

In grab sampling mode an air sample is taken into the cell which is then sealed and after a delay of 3 hours, to allow for approximate radioactive equilibrium between radon and its short-lived decay products to be reached, a count rate is taken. Active versions using flow through scintillation cells are commercially available and can be used for continuous monitoring of radon concentrations in a building. They are usually equipped with data storage and printout facilities and a range of cycle times.

6.2 Alpha track detectors

A number of plastic or polymeric materials are available which have the property that the primary damage caused in them by the passage of alpha particles remains fixed in them and may be made visible as tracks by means of a suitable etching procedure (Cartwright et al. 1978). These materials are often called Solid State Nuclear Track Detectors (SSNTDs). The commonest of those in use for radon detection are the cellulose nitrate film (LR-115), the thermoset polymer plastic (CR-39) and the polycarbonate plastic (Makrofol). The passage of an alpha particle through a SSNTD produces a narrow primary damage trail or latent track along the length of its path in the material (typically 20 to 70 μm). The production of a visible track, by chemical or by electrochemical etching, is made possible because the damaged material etches faster than the undamaged or bulk material. The tracks are very characteristic and for chemical etching of CR–39 the tracks appear under transmission optical microscopy as dark three dimensional conically shaped features. These may be identified and counted by a human microscopist or by means of an automatic computerised image analysis system. For LR-115 detectors, etched tracks can be rapidly counted by a "spark-counter" (Cross and Tommasino 1970).

The use of such alpha track detectors for passive long term integrating measurement of indoor radon is very popular both for large scale surveys and for radon measurements in single

buildings (Alter and Oswald 1983). Most radon alpha track detectors consist of a few cm^2 of the SSNTD material mounted inside a small, almost airtight, closed container. The air containing radon enters the inner volume of the container by diffusion. Alpha particles from radon and its in-grown decay products which strike the SSNTD produce the alpha tracks from which, by means of calibration factors, the mean radon concentration may be determined. These closed radon alpha track detectors typically have dimensions giving them a volume range of about 20-75 cm^3. In some versions of this detector type the SSNTD material is mounted open-faced or bare on a wall in a building to record directly alpha particles from the air. For a variety of operational and other reasons there is a preference for the closed type alpha track detector although open faced types of excellent characteristics using LR-115 are also used extensively (Rannou et al. 1986).

Alpha track detectors are inexpensive, reliable and easy to use. They compare very well in sensitivity and accuracy with other types of radon detectors. A minor disadvantage, however, is that they do not give a measurement in the field and must be sent to a processing laboratory if the user is not equipped to process them.

6.3 Charcoal detectors

The adsorption of radon by activated charcoal has been used for many years as a detection method (Cohen and Nason 1986). The method is very simple. The gamma radiation emitted by radon and its ingrown decay products in the charcoal is measured by means of a gamma ray detector such as Sodium Iodide (NaI(Tl)). A charcoal detector suitable for indoor radon measurements usually consists of a pocket sized flat metal cylinder containing the charcoal. The best versions of these include a diffusion barrier between the charcoal and the air. In this way the rate of radon adsorption is proportional to its concentration in the air. The charcoal method is a short-term integrating method. It is passive, inexpensive and of sufficient sensitivity that the radon concentration at typical indoor levels can be measured using an integrating time of a few days. It has been used extensively for large scale radon surveys, principally in the U.S., and is also very popular for making determinations of indoor radon concentrations in the space of a few days as may be required for real estate transaction purposes. The principal disadvantage of the charcoal detector method is that its useful integrating time is limited to a little over one week. This arises because radon has a radioactive half-life of 3.8 days. Consequently the gamma activity from radon adsorbed at a time in the past more than about two half-lives previously will be decayed to a small fraction of that from the most recently adsorbed radon. Another similar disadvantage, arising again from the half-life of radon, is that following the completion of its exposure in a building the charcoal detector containing the adsorbed radon must be sent quite promptly to the measuring laboratory for analysis before the gamma activity has decayed to an insignificant level. Moreover, the accuracy of the method is sensible to significant variations of radon concentration during exposure period. Some modified versions of charcoal detectors have been developed to overcome these disadvantages. In one of them the continual gamma emission in the charcoal is recorded by a TLD (Thermoluminescent Detector) chip inserted into the charcoal (Stranden et al. 1983). The mean radon concentration can be determined subsequently using a TLD reader. This approach is not in general use.

Unlike alpha track detectors, charcoal detectors are not actually true detectors of radon or its decay products. They are essentially absorbers which collect radon for subsequent gamma counting. The alpha track detectors on the other hand record continually and permanently store

the damage caused by alpha particles during the full exposure period. This damage is subsequently revealed by processing as permanent tracks.

6.4 Electret detectors

An electret is a material such as some types of aluminised Teflon which when charged will generally retain the charge and associated electric potential for a period of a year or longer. Such charged electrets have been used as electrostatic collectors of charged radon decay products from the air which can then be measured by a scintillation detector, surface barrier detector or other alpha particle detector (Chittaporn et al. 1981). This approach has not been used extensively.

A very reliable application of electrets as radon detectors is, however, commercially available (Kotrappa 1988). Essentially it consists of a small hollow plastic container inside of which is mounted a charged electret. When air containing radon is admitted to the inner volume of the container the ionisation produced, mainly by alpha particles from radon, Po-218 and Po-214, will cause the electret to be discharged at a rate proportional to the radon concentration. From measurements of the potential of the electret before and after exposure to the air containing radon the mean radon concentration may be determined by means of calibration information for the system. This type of electret based radon detector has the advantage of being relatively inexpensive, is rechargeable and may be used a number of times between recharges. It also has the advantage that the user of the system can make the radon determination thus obviating the need to send the electret to a processing laboratory. Some disadvantages do, however, exist associated with the presence of humidity and of ionising sources other than radon. These factors may influence the performance of electret based detectors.

6.5 Electronic monitors

A range of electronic detectors are available in which the common feature is the detection of alpha particles from radon and its decay products by surface barrier or similar solid state detectors and associated electronics (Simon and Schell 1990). These devices are either mains and/or battery operated. In their simplest form radon gas enters a small detecting volume within the instrument by free diffusion through a porous barrier and the mean radon concentration during a measuring period of some hours may be read directly by the user from a panel display. Variable measuring cycle periods can be set for such devices and usually range from 4 to 24 hours with the facility to store the data for up to 100 such intervals. Thus many of these type of devices can be used as continuous monitors for periods as long as three months when set on a 24 cycle and can give more detailed information on the radon temporal variation in a building if set on shorter cycles. In some versions sensitivity is increased by the use of electrostatic deposition of charged radon decay products either directly onto the detector or onto adjacent collecting surfaces. More sophisticated versions of electronic detectors are available in which air is drawn by a pump through a filter mounted in front of the detector. The concentrations of the individual radon decay products are determined by gross alpha counting procedures or even by alpha spectroscopy. Direct printout of data and the facility to transfer data to a computer are also available. These devices are expensive and therefore more useful for detailed investigations in a small number of buildings rather than for use in large surveys. Simpler hand held electronic monitors of radon may be obtained for use in radon diagnostic studies of a building before and after remediation. They are usually equipped with a small battery operated pump and can be used to identify the routes of radon entry into a building.

6.6 Concluding comments

There is a wide and somewhat confusing array of radon detectors which are commercially available. Each has its advantages and disadvantages in terms of technical performance, price etc. Some of these points have been indicated above. For indoor air quality investigations involving radon surveys or assessment of long term average radon levels in a building passive detectors such as alpha track or electret versions are best suited. Where detailed indoor radon diagnostics of a building are required for pre or post remedial work passive detection methods should be complimented by continuous type devices which ideally should have readout facilities. Table 2 summarises the most common modes of operation of the various types of radon detectors described above.

In terms of cost effectiveness of the various techniques presented here it is not possible to make realistic comparisons as each application will be unique. It should be noted that an individual measurement of radon using a commercial passive detector, such as alpha track or charcoal types may be purchased at a cost of between US$ 10 and US$ 20. On the other hand a state-of-the-art electronic radon detector complete with readout facilities, memory and the ability to make alpha spectroscopic measurements of radon decay products may cost US$ 6000. This high capital cost may in some applications, such as remediation work, be acceptable. In large scale survey work the use of passive detectors will usually be more cost effective.

It cannot be overemphasised that accurate calibration of radon detectors is essential for good quality investigations. Experience, unfortunately, has shown that radon sensitivities quoted by manufacturers for some instruments are not always applicable under field conditions. Both in the European Communities (European Community) and in North America there are facilities at some national laboratories where independent calibration of instruments may take place. Standard radium sources for producing precise amounts of radon are also available and recently in the UK standard amounts of radon gas in glass vials have become available (NPL 1992). With a half-life of 3.8 days these recently available radon standards can only be used for a short period of time. It is worth noting that the Commission of the European Communities (CEC) has for, a number of years, at regular intervals organised intercalibrations of radon detectors as part of its ongoing Radiation Protection Research Programme (Miles et al. 1984, Miles and Olast 1990).

Table 2. Most common modes of operation of radon detectors

DETECTOR	GRAB	INTEGRATING	CONTINUOUS	ACTIVE	PASSIVE
Scintillation Cell	Yes	-	Yes	Yes	Yes
Alpha Track	-	Yes	-	-	Yes
Charcoal Detectors	-	Yes	-	-	Yes
Electret	-	Yes	-	-	Yes
Electronic	Yes	-	Yes	Yes	-

7 REMEDIAL AND PREVENTIVE MEASURES TO REDUCE INDOOR RADON

7.1 Introduction

The main sources of indoor radon are: soil, building materials and, in some cases, water from deep wells. Experimental work carried out at international level has shown that radon from soil represents generally the most important source of indoor radon (e.g. Asikainen and Kahlos 1980, Bruno 1983, Damkjaer and Korsbech 1985, Kristiansson and Malmqvist 1984, Nazaroff et al. 1987b, NEA/OECD 1979, Nero 1989, Tanner 1978).

Radon occurs in high concentrations in soil gas with large variations due to the characteristics of local geology. Typical radon concentration in soil gas range from 10000 to 50000 Bq/m^3. In areas of elevated indoor radon levels soil gas concentrations in excess of 500000 Bq/m^3 may occur. In such situations dilution of the incoming soil gas by indoor air will generally be insufficient to maintain indoor radon concentrations below most reference or action levels. In Sweden a categorisation of soils has been used as part of a system of classification of risk to occupants of houses from radon. In this system a low risk radon ground is a soil with a radon soil gas concentration less than 10000 Bq/m^3, while high radon ground would have a concentration greater than 50000 Bq/m^3 (Åkerblom et al. 1984, Snihs 1992). Normal radon ground would lie between these two categories.

The actions to reduce indoor radon concentration are mainly oriented to limit the ingress of radon from soil. This goal can be reached by removal of the source, diverting the radon before entering the building and/or using barriers between the soil and the living space.

The techniques available at present can be grouped in two types: passive methods (which do not require further intervention after installation) and active methods (requiring extraction fans, air cleaning devices, etc.).

7.2 General approach to control radon indoors

The problem of remedial action has been faced from the seventies in Canada (McLaren 1979, Leung 1979, Scott 1979, DSMA 1979), Nordic countries (Ericson 1980, Ericson et al. 1984) and in the following years also in the United States, mainly from EPA, and in the United Kingdom (EPA 1987c, 1988, 1989c, 1991, 1992b, Cliff 1980, Green at al. 1992). In these countries experience has been gained, although restricted mainly to single family dwellings and some schools (e.g. EPA 1987d). As far as our present knowledge is concerned, a complete reference picture of all the different methodologies available to reduce indoor radon and their effectiveness and durability is still not available (CEC/DOE/EPA 1993).

The main methods available to reduce the ingress and the concentration of indoor radon can be summarised as follows:

- reduction of radon entry from soil through *depressurisation* of subfloor spaces;
- increasing the building *ventilation* rate with a consequent increase of radon removal;
- increasing the resistance of building to radon entry by *sealing* the floor (or the walls, in case of building materials with high radon exhalation rate);
- removing the radon source: applicable only to water supply and not discussed here.

7.3 Depressurisation

Normally the pressure in a building is less than in the soil gas and this causes soil gas to enter the building. If, however, the pressure differential between the soil and the building is reduced the radon entry is decreased. In order to obtain soil depressurisation, a zone of subjacent soil is maintained at a lower pressure difference than the building, by means of a small fan or by a vented stack.

Different methods are available to obtain such soil depressurisation (DOE 1990b, NCRP 1989, EPA 1991). If the building has a concrete floor, a fan can be used to suck air from beneath the floor and vent it to the atmosphere. Usually a radon sump is built which consists in a hole in the ground with a suitable fan connected to it sucking from the hole and generating a negative pressure in the hole.

The more permeable the material below the slab, the more efficient is the sump. The pipe which connects the sump to the outdoor air can go under the floor to an external wall or to an internal wall and then up to the roof. According to the Building Research Establishment of the United Kingdom, an effective exhaust fan for a single family house can have a power rating around 60 watt and a pipe 110 mm diameter. This should achieve good results and has been found to remove in some cases 90% of radon (Wolliscroft 1992). In some circumstances, mainly when a new building is going to be constructed, a plastic membrane could be placed over the soil within the foundation, which will act as a barrier to radon (EPA 1991).

An effect similar to the radon entry preventation obtained by the subfloor space depressurisation can be achieved also through the overpressurisation of a dwelling by a fan mounted in the attic of a dwelling.

7.4 Ventilation

Increasing the ventilation rate to reduce the radon concentration is one of the easiest methods available. Moreover, a knowledge of the radon source is not required. There are, however, practical difficulties in determining how much the ventilation rate should be increased and how it can be supplied in order to be acceptable by the occupants.

Mechanical ventilation with fans which blow fresh air into the house may have other effects other than to increase air flow. The air pressure inside the house can be increased to exclude ingress of radon. The efficiency of the system is strictly connected to the tightness of the building (DOE 1990, Holub et al. 1985, Nazaroff et al. 1981, NCRP 1989, Wolliscroft 1992). Exhaust fans can be used to increase the ventilation rate in the subfloor space, in this way, significant reduction can be achieved in the living space (Renken et al. 1992). To compensate for the heating cost during the winter season due to increased ventilation a balanced mechanical ventilation system with heat recovery can be used. Quantitative data on the effectiveness of such systems are rather poor (Renken et al. 1992).

Another possible use of ventilation is the "natural" ventilation of basements: in a basement (or cellar) with open windows radon concentrations become lower not only because of dilution, but also because it reduces basement depressurisation (Cavallo et al. 1992).

In conclusion it should be cautioned that while ventilation techniques may be effective, in some cases changes in building ventilation patterns may exacerbate the radon situation by increasing the underpressure in the building.

7.5 Sealing

In the foundations of a dwelling, usually many openings exist through which soil gas can penetrate. These openings include junctions between walls and floor, gaps between floor and slab sections, openings left for service entries, etc.

The remedies which can be applied are to fill the gaps and openings by using sealing materials like epoxy resin. A rigid sealant is not convenient since the cracks are in constant motion due to climate changes, shrink-swell cycle, etc. As an alternative a plastic cover can be applied on the floor having care to seal accurately the junctions between wall-floor, etc. (EPA 1991). This solution can be easily applied in the case of new dwellings built in high radon prone areas as suggested by the UK Building Research Establishment (Wolliscroft 1992). The achievable reduction factor is at maximum around 50%; the durability of this remedy is not well established. A drawback of the system could also be linked to possible emission of VOC from the sealing materials.

7.6 Air cleaning devices

As far as the use of air cleaning devices are concerning most of experience gained in removing radon decay products derives from the mining industry. Although the principles on which the devices to remove radon daughters are based - mechanical and electrostatic filters - are independent of their use whether in mines or in home, nevertheless the volumes of air involved and environments are completely different, so that this technique has been considered not applicable in houses (EPA 1987c). Some years ago, this has been considered as a developing field and laboratory tests were carried out. The data available to date do not suggest that these techniques are very effective, but it could be useful perhaps in cases where the building materials are the principal radon source (McLaughlin 1989).

7.7 Conclusions

The preventive and remedial measures described above have been experienced in single family houses of some countries. It is very difficult to give general rules applicable to every situation. Each house seems to represent its own unique problem. It has often been said that it is almost impossible to have two houses which behave in the same way with respect to radon.

A large amount of work remains to be done in order to establish general criteria for old and for new dwellings, to collect data on effectiveness and durability of remedies, to explore the best predictors to identify radon prone areas, etc. The approaches to be followed are obviously strongly dependent on building styles, constructions regulatory regime and on the use of buildings.

It has to be stressed that till now emphasis has been given to housing rather than to commercial buildings since the greatest need arises in housing stock where people usually spend most of their time. Studies are being undertaken in recent years to model the ingress of radon in large commercial buildings, offices, schools etc., to single out their peculiarities with respect to single families houses and to ascertain the kinds of remedies or preventive measures which are likely to reduce indoor radon (e.g. EPA 1991b , 1992b).

8 RECOMMENDED AND REGULATORY RADON LEVELS

8.1 Introduction

Regulation of radon in the domestic environment is an issue which has been the subject of much debate by national and international bodies, due to its complexity and peculiarities.

Radon is a natural gas, ubiquitous and affects homes and the private sphere of individuals. In contrast to what happens with many artificial pollutants whose use could be forbidden if considered necessary, radon can be reduced but not totally eliminated. Moreover, due to the stochastic characteristic of radon health effects and to the hypothesis of a linear relationship between radon exposure and excess lung cancer probability, it is considered that there is no level with zero risk.

Therefore it is difficult to choose an "action level" above which remedial actions are recommended, some of which were outlined in the previous chapter. This choice is not totally based on scientific aspects only, as health effect risk, but must take into account other aspects of the problem. The ICRP recommends that "the best choice of an action level may be that level which defines a significant, but not unmanageable, number of houses in need of remedial work. It is then not to be expected that the same action level will be appropriate in all countries" (ICRP 1991). Moreover "any action affecting the whole housing stock of a country would be extremely costly, although it might still be cost-effective in terms of the reduction in the national collective dose. It is for national authorities to decide whether the necessary funds would be available and best spent on general radon reduction or other aspects of housing improvement" (ICRP 1993).

On the other hand the radon problem is increasingly being considered as part of the overall indoor air pollution and as such should be seen as a public health issue. Solutions to this problem should therefore be approached on a broader and more integrated front than that of traditional radiation protection.

There is now, from both occupational and domestic epidemiology, consolidated knowledge that radon is probably the single major agent responsible for lung cancer in the population - after tobacco smoking. Even a modest reduction of radon exposure is expected to provide a decrease in occurrence of lung cancer and thus radon exposure levels are of importance from the public health point of view. When selecting an "action level" and planning the necessary action to conform to that level, the community should be made aware of the relative importance of radon risk in comparison to risk factors from other substances which are, at the same time and nevertheless, a concern to public health.

As the current recommendations concerning exposure to most carcinogenic agents generally require that exposure be contained at a level which will guarantee an "excess risk" of no more than 10^{-5} or 10^{-6} cases per year, it is clear that if radon had to be treated in the same way, the resulting recommended level would be very low. These levels might be generally unachievable in existing buildings, but could perhaps be reached as targets for new buildings if appropriate preventive measures are incorporated in the building codes and norms.

In many countries one or more surveys have been carried out before their action levels were chosen, in order to be able to estimate the impact of the chosen value on the basis of radon concentration distribution.

Moreover some national and international authorities decided to use different values for

existing and future (also referred to as "new") dwellings and workplaces, as suggested by ICRP in 1984, because of the greater difficulties in applying remedial action in existing buildings. Both levels, the action level for existing buildings and "upper bounds" for future ones, are often referred to as "reference levels". Generally all recommended reference values are to be considered annual radon concentration averages. Please note that in other fields the term "reference level" may have a different meaning.

Up to the present in most countries the approach to dealing with indoor radon has been limited to recommendations. It has proven to be very difficult to transform these into regulatory requirements.

8.2 Application to dwellings

The actual recommendations are summarised in Table 3 and show a wide spectrum. Due to radon concentration variability, reference levels are generally to be intended as annual average values. Some of the presented values were originally expressed in terms of radon progeny concentration, and are here converted to radon gas values assuming an equilibrium factor of 0.5 (see Appendix 2).

Historically ICRP suggested in 1984 - in its publication No.39 (ICRP 1984) - an action level of 400 Bq/m^3 for existing dwellings as a reference level above which remedial action should be considered, the urgency and severity of the remedial action being related to the amount by which the level is exceeded. Moreover an upper bound of 200 Bq/m^3 for new dwellings was also proposed.

The Commission of European Communities issued in 1990 a recommendation (CEC 1990) to limit the radon exposure in buildings which follows strictly the ICRP Publication No.39.

In the United States the EPA in 1986 recommended 150 Bq/m^3 as a reference level above which remedial action is suggested - the urgency and severity of the remedial action being related to the amount by which the level is exceeded. The same upper bound level was indicated for new dwellings. In the Indoor Radon Abatement Act of 1988, the U.S. Congress has set the long-term goal to reduce indoor radon concentration as low as the ambient air. The EPA considers that while this goal is not yet technologically achievable in all cases, it is possible to reduce radon levels to 75 Bq/m^3 in most cases (EPA 1992).

In Canada the level suggested by the Federal and Provincial Ministers of Health in 1988 (Létourneau 1985) for both old and new dwellings is 800 Bq/m^3 (McGregor 1993), but it is recognised that "there is some risk at any level, so that home owners may wish to reduce levels of radon as low as practicable" (Eaton 1990).

In the Nordic Countries (1986) the radiation protection authorities have collectively adopted in 1986 the limit value of 800 Bq/m^3 for existing dwellings and the design level of 200 Bq/m^3 for future dwellings. The recommended value for existing dwellings in case of simple remedial actions is 200 Bq/m^3. Individual Nordic Countries have since then adopted national reference values.

In Sweden a regulatory action level of 800 Bq/m^3 was established in 1980, and homeowners are recommended to decrease radon concentrations higher than 200 Bq/m^3 if this is possible with reasonable efforts. In 1990 the limit has been decreased to 400 Bq/m^3 and the recommended action level for simple measures has been decreased to 140 Bq/m^3 (Swedjemark 1990, Snihs

1992). The design level for new dwellings is 200 Bq/m^3 (Swedjemark 1994).

In Finland the reference levels are 400 Bq/m^3 for existing dwellings and 200 Bq/m^3 for future ones (Castrén 1993b).

In Norway both the action level for existing dwellings and the upper bound for future ones are 200 Bq/m^3 of radon concentration annual average (Strand 1993).

In the United Kingdom, in 1990 following a nationwide radon survey and using the revision of radiation risk made by ICRP (1991), the action level has been decreased from 400 Bq/m^3 to 200 Bq/m^3, above which householders should reduce radon concentration before the occupants receive a further time-integrated concentration of 1500 Bq· y/m^3 (NRPB 1990). In future dwellings the radon concentration levels should be as low as reasonably practical, and at least below the action level of 200 Bq/m^3.

A recommended reference level for both existing and new dwellings in Ireland of 200 Bq/m^3 was adopted in 1990.

In Germany, in 1994 the German Commission on Radiological Protection recommended no action for radon concentrations below 250 Bq/m^3, considered "normal", and simple actions to reduce radon concentration for levels to 1000 Bq/m^3, while stronger actions are recommended only for higher levels. The upper bound for future dwellings is 250 Bq/m^3.

In Belgium a single reference level of 250 Bq/m^3 has been adopted.

The WHO includes radon among the agents that are carcinogenic to human (WHO/IARC 1988) and recommends simple remedial action for buildings with annual average radon concentration of 200 Bq/m^3, without delay if the radon concentration is higher than 800 Bq/m^3; the same upper bound of 200 Bq/m^3 is recommended for new dwellings (WHO 1987).

The ICRP has recently adopted a new recommendation on "protection against radon at home and at work" (ICRP 1993), in which a range of action level values is proposed for existing dwellings: from 200 Bq/m^3 to 600 Bq/m^3. The upper bound for future dwellings is now considered unhelpful and is substituted with issuing guidance and codes on construction practices, especially for areas where many high radon concentration values have been measured or estimated.

Table 3. Recommended reference levels for radon gas in dwellings (Bq/m^3)
(See text for details)

Country/Organisation	Existing Dwellings	Future Dwellings
Belgium	250	250
Canada	800	800
CEC	400	200
EPA (USA)	150	150
Finland	400	200
Germany	250	250
ICRP (1984)	400	200
ICRP (1993)	200 to 600	–
Ireland	200	200
Norway	200	200
Sweden	140 and 400	200
United Kingdom	200	200
WHO	200	200

8.3 Application to "normal" workplaces

In the new ICRP recommendation on radon, a range of reference levels are proposed also for workplaces where the occupancy of members of the public is low, e.g. in offices, libraries and theatres: from 500 Bq/m^3 to 1500 Bq/m^3 (ICRP 1993). These values differ from those for dwellings because of the shorter period of exposure in workplaces, where 2000 hours per year are assumed, and because of the assumed different radiosensitivity between the general population and workers.

In UK the law requires employers to limit the exposure of workers to radon daughters, and the action level is 400 Bq/m^3 of radon gas (Gooding and Dixon 1992).

8.4 Limitation of radioactivity concentration in building materials

Up to now, limitations on the use of building materials are generally related to the *gamma* radiations emitted by the radionuclides that are incorporated into the building materials (NEA/OECD 1979). However building materials are considered also as radon sources and ICRP recommends to identify building materials with high Ra-226 content, and to prevent or limit their use (ICRP 1993). Some countries have already adopted recommendations on radionuclide content in building materials, e.g. Netherlands and Nordic Countries (McLaughlin 1987).

8.5 Conclusions

The spectrum is rather wide. The levels suggested are linked to the special situation of each country: geology of soils, its characteristics, etc. and to general social and economic status.

A basic problem is the cost of remedial actions and who has to sustain the economical burden: the individual citizen or the society as a whole. In Sweden, for example, since 1988 owners of single-family houses with radon levels exceeding the action level have been able to get partial financial assistance from the government (Swedjemark et al. 1993).

As outlined above the setting up of reference levels for indoor radon in dwellings is a political problem more than a technical one. National authorities can offer a careful analysis of the situation (levels detected, number of houses with high concentrations, etc.) and could establish what is the economical and social cost of remediation. International bodies can only give general guidelines but it is up to the governments to make the final decision.

A major obstacle in reducing the doses from radon to the population by a non regulatory approach is the general apathy of the public. In recognition of this a number of organisations such as the World Health Organisation recommend that effective radon risk communication programmes should be an essential part of a comprehensive radon risk management policy (WHO 1993).

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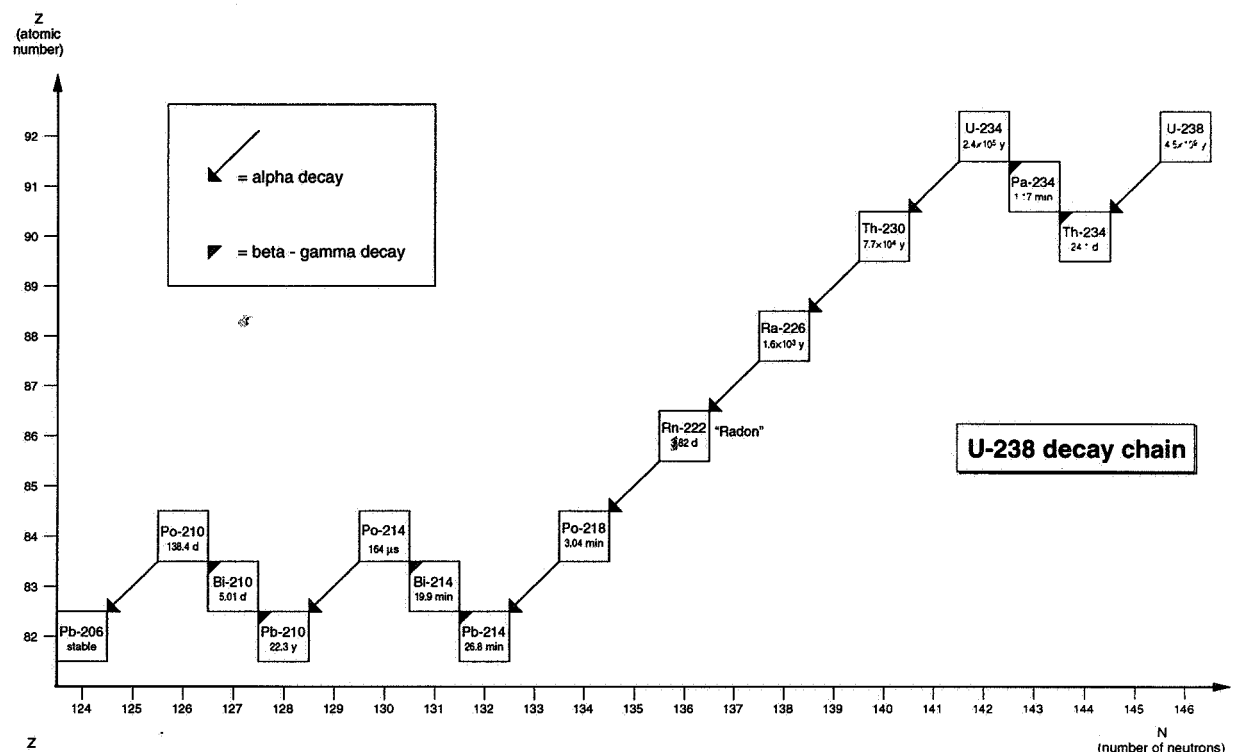
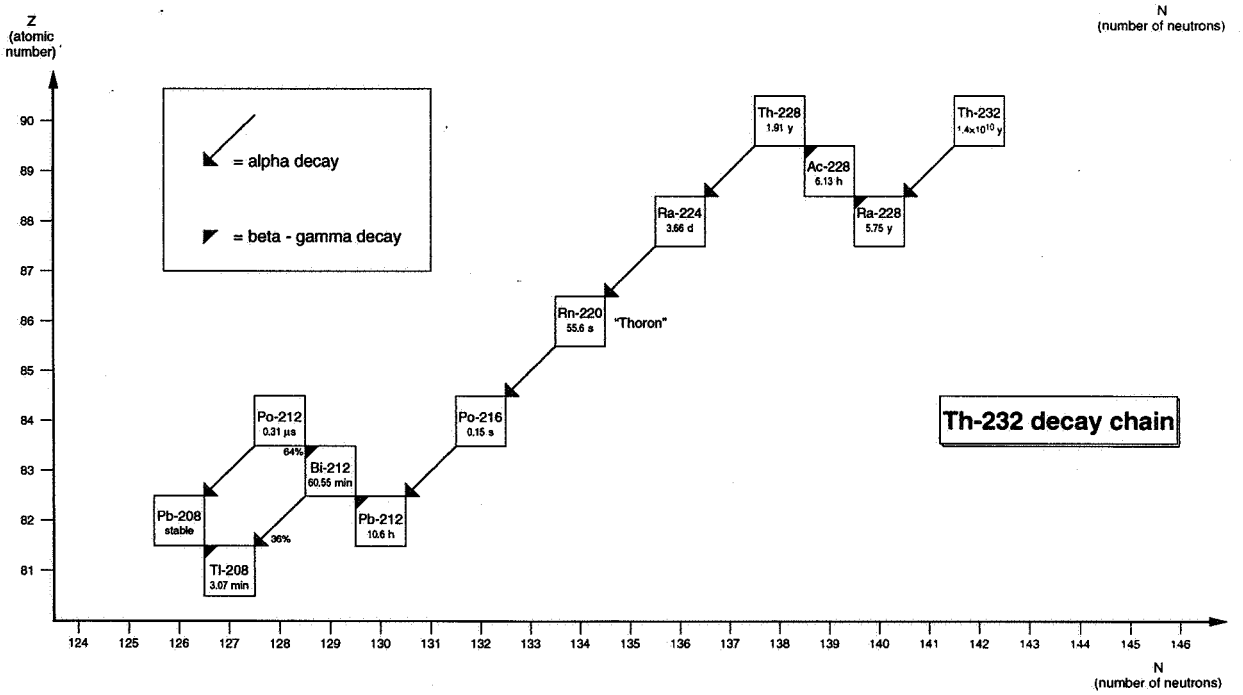
Appendix 1

Radioactive decay chains of uranium-238 and thorium-232





Under the symbol of each isotope the half-life time is reported.

Decay scheme of radon-222 and its short-lived progeny

Radionuclide	Historical name	Half Life	Energies (expressed in MeV) and intensities (%) of the principal radiation emitted		
			Alpha	Beta	Gamma
Rn-222	Emanation Radon	3.82 d	5.490 (100%)	–	0.51 (0.07%)
↓ α					
Po-218	Radium A	3.04 * min	**6.003** (~100%)	–	–
↓ α					
Pb-214	Radium B	26.8 min	–	0.65 (50%) 0.71 (40%) 0.98 (6%)	0.295 (19%) 0.352 (36%)
↓ β,γ					
Bi-214	Radium C	19.9 min	–	1.0 (23%) 1.51 (40%) 3.26 (19%)	0.609 (47%) 1.12 (17%) 1.76 (17%)
↓ β,γ					
Po-214	Radium C'	164 µs	**7.687** (100%)	–	–
↓ α					

* The reported updated value is taken from Martz et al. (1989)

Appendix 2

Special quantities and units for radon and radon decay products

Absorbed dose

It is defined by $D = d\varepsilon/dm$, where $d\varepsilon$ is the mean energy imparted to a mass dm of matter by ionising radiation. For radiation protection purposes, a tissue - or organ - average absorbed dose D_T is defined by $D_T = \varepsilon_T / m_T$, where ε_T is the total energy imparted to a tissue - or organ - of mass m_T.

The SI unit of absorbed dose is the Gray (Gy), equal to one Joule per kilogram (J/kg).

1 Gy = 100 rad (old unit).

Equivalent dose

The equivalent dose is the product of the absorbed dose and the weighting factor for a specific type of radiation. The weighting factor w_R accounts for the different ability of the different type of radiation to cause biological damage. For beta particles, gamma rays and X rays w_R is taken as unity, for alpha particles as 20, for neutrons as from 5 to 20, depending on energy (ICRP 1991). The value of w_R for alpha particles, such as those emitted by radon daughters, is still a matter of discussion, with some evidences which suggest that it may be below 20 (Birchall and James 1994).

The SI unit of equivalent dose is the Sievert (Sv), equal to one Joule per kilogram.

1 Sv = 100 rem (old unit).

Effective dose

This is the sum of the products obtained by multiplying the equivalents doses to various organs and tissues by the appropriate risk weighting factor w_T for each. This quantity is considered to be proportional to the total probability of stochastic effects.

Its SI unit is the Sievert (Sv), as for equivalent dose.

1 Sv = 100 rem (old unit).

The most recent estimation of the risk of stochastic effects for general population, expressed as probability of fatal cancer per unit of effective dose, is 5×10^{-2} Sv^{-1} (ICRP 1991).

Activity

The activity of a radioactive source is defined by $A = dN/dt$, where dN is the number of decays in time dt.

The SI unit of activity is the Becquerel (Bq), equal to one nuclear transformation per second.

1 Curie (Ci) (old unit) = 3.7×10^{10} Bq.

Activity concentration

It is the activity per unit of volume or mass, whose SI unit is the Becquerel per cubic meter (Bq/m^3) or the Becquerel per kilogram (Bq/kg).

1 pCi/ℓ (old unit) = 37 Bq/m^3.

Radon progeny concentration

The concentrations of the four short-lived radon daughters are usually expressed in terms of Potential Alpha Energy Concentration (PAEC) or Equilibrium Equivalent Radon Concentration (EERC or EER). These two quantities combine the four concentrations in a single value, using weighting factors that take into account the relative importance for the health effects to the lung, which are due to the alpha particles emitted by the radon daughters along the decay chain down to lead-210.

The *PAEC* of any mixture of radon daughters is the sum of the potential alpha energy (i.e. the total energy of all alpha particles emitted by a radon daughter atom in its chain decay to the Pb–210) of all the short-lived radon daughter atoms present per unit of air volume. Its SI unit is the Joules per cubic meter (J/m^3), but the old Working Level (WL) unit is still widely used, representing a PAEC of 1.3×10^5 MeV per litre.

1 WL = 2.083×10^{-5} J/m^3, therefore:

PAEC [nJ/m^3] = 20.83 PAEC [mWL], and PAEC [mWL] = 48.01 PAEC [$\mu J/m^3$].

The *EERC* is the fictitious activity concentration of radon in equilibrium with its daughters (i.e. all daughters having the same concentration as radon; actually there is always a certain degree of disequilibrium) giving the same PAEC of the actual non-equilibrium mixture of radon daughters. Its SI unit is the Becquerel per cubic meter (Bq/m^3).

A PAEC of 1 WL corresponds to 3749 Bq/m^3 (= 101.3 pCi/ℓ) of EERC.

Equilibrium factor

The ratio of EERC to actual radon concentration is called "equilibrium factor". It is usually represented by the symbol "F". Due to the actual non-equilibrium conditions its value in indoor air is usually in the range 0.2–0.7.

Cumulated exposure to radon progeny

Its practical measurement unit is the Working Level Month (WLM). It corresponds to an exposure to ≅ 3700 Bq/m^3 of radon progeny concentration in equilibrium with radon gas (i.e. all progeny having the same concentration as radon; actually there is always a certain degree of disequilibrium), during the working period of one month (170 hours).

This unit is used historically for occupational exposure in mines, while for indoor exposure of the general public in dwellings it is preferable to express the cumulative exposure in terms of mean radon gas concentration during a one-year period, because this is the quantity that is usually measured in dwellings. The following conversion formula can be used:

$$1 \text{ WLM} \cong 72 / (F \times IOF) \quad \text{Bq y/m}^3$$

where F is the equilibrium factor between radon and its progeny, and IOF is the indoor occupancy factor, i.e. the fraction of time spent indoors. Using F=0.5 and IOF=0.8, a cumulated exposure of 1 WLM corresponds to an exposure to ≅ 180 Bq/m^3 of radon gas concentration during the period of 1 year, or 90 Bq/m^3 for a period of 2 years, and so on. If F=0.4 is assumed, 1 WLM ≅ 220 Bq y/m^3.

Appendix 3

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EUR 16123 – European collaborative action 'Indoor air quality and its impact on man': Radon in indoor air

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Luxembourg: Office for Official Publications of the European Community

1995 – 60 pp., 2 fig., 3 tab. – 21.0 x 29.7 cm

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In an attempt to overcome the increasing difficulty of having concise essential information on important indoor pollutants at hand, the European Collaborative Action 'Indoor Air Quality and Its Impact on Man' has published several reports amongst which a short report on radon in indoor air was published in 1988. Since then, considerable new information has become available. This report summarizes and discusses the actual state of knowledge on sources of radon in indoor air, typical concentrations, health effects and radon risk estimates, indoor radon measuring methodology, instrumentation, remedial and preventive measures to reduce indoor radon, and recommended and regulatory radon levels. The report concludes that the cost of remedial actions and the lack of public interest are major obstacles in the reduction of population exposure. Appendices which have a decay scheme of radon-222 and its short-lived progeny and which explain special quantities and units for radon and radon decay products complement the report.

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