



# **PROCEEDINGS**



**Nordic Society for Radiation Protection  
12th ordinary meeting  
23-27 August 1999  
Skagen, Denmark**



**Nordic Society  
for  
Radiation Protection**

**Proceedings of the  
12th ordinary meeting  
Skagen, Denmark  
23-27 August 1999**

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ISBN 87-550-2617-6  
Printed by the Information Service Department  
Risø National Laboratory  
P. O. Box 49  
DK-4000 Roskilde  
Denmark  
November 1999

# Contents

## Preface

Asker Aarkrog .....	ix
---------------------	----

## Session I - Invited Lectures, Radiation in a General Environmental Perspective with Special Emphasis on Information

Chairman: Ulf Bäverstam

### **The Bo Lindell lecture**

*Environmental Radiation Protection and Information Expectations  
of the Public - Past and Future*

Anneli Salo .....	3
-------------------	---

*The Balance between Scientific and Public Influence on  
Marine Environmental Protection*

J. Michael Bowers .....	15
-------------------------	----

*Radiation Protection: A Shining Example or a Dazzling Image?*

Richard John Pentreath .....	17
------------------------------	----

*The Role of Risk Perception for Risk Management*

Ortwin Renn .....	19
-------------------	----

*Communicating with Stakeholders and the Public at Large – the Brent  
Spar Case*

Margrethe M. Skov .....	43
-------------------------	----

*What can be said can be said clearly*

Björn Wahlström .....	45
-----------------------	----

## Session II – Information, Education and Regulatory Aspects

Chairman: Sigurður Emil Pálsson

*Lunds Universitets Strålskyddsinformation på Intranet*

Christer Samuelsson .....	57
---------------------------	----

*The Norwegian Information Campaign on Radon*

Georg Thommesen and Terje Strand .....	61
--	----

*University Courses on Radiation Protection in Estonia*

Enn Realo .....	65
-----------------	----



<i>A Compartment Model for Analysis of Routine Releases of Radionuclides to a Fjord</i>	
Tord P. Walderhaug .....	155

<i>Direct Monitoring of Radiocaesium in Live Reindeer and Reindeer Carcasses</i>	
Birgitta Åhman .....	159

## **Session VII – Dosimetry, Instrumentation and Quality Assurance**

### **a) Diagnostic Radiology b) Dosimetry**

Chairman: Enn Realo

<i>Generelle Principper og Problemer ved Bestemmelse af den Kollektive Dosis til Befolkningen fra Røntgendiagnostik</i>	
Peter Grøn .....	165

<i>A Practical Approach to Radiation Protection Information in Diagnostic Radiology</i>	
Åke Cederblad, Elisabeth Bjurklint, Susanne Kheddache, Lars Gunnar Månsson and Patrik Sund .....	169

<i>Exposure of Patients and Creation of a System of Quality Assurance in Conventional X-ray Radiology in Lithuania</i>	
G. Morkunas and J. Ziliukas .....	173

<i>Radiation Doses from Contaminant Aerosol Deposition to the Human Body</i>	
Kasper G. Andersson, Christian L. Fogh, M.A. Byrne, Jørgen Roed, and A. J. H. Goddard .....	177

<i>Mikrokjerner og Apoptose i Humane Lymfocytter</i>	
Kirsti Bredholt, Alicja Jaworska, Gøril Olsen and Jon B. Reitan .....	181

<i>Uncertainties involved in the Calculation of Intakes and Committed Effective Doses from Whole-Body Measurements</i>	
Bente Lauridsen and Jens Søgaard-Hansen .....	185

<i>AlN Ceramics as a Detector for UV Exposure</i>	
Laima Trinkler, Lars Bøtter-Jensen, Poul Christensen, B. Berzina and E. Palcevskis .....	189

<i>New Way of Demonstrating the Competence of a Laboratory Measuring Radionuclides - The International Draft Standard ISO/IEC DIS 17025</i>	
Sigurður Emil Pálsson .....	193

## Posters

<i>The National Radiation Protection Infrastructure</i> Albinas Mastauskas .....	199
<i>Swedish Radiation Protection Institute: Information Activities</i> Lars Persson .....	203
<i>Ändringar i Reglerna för Transport av Radioaktiva Ämnen</i> Birgitta Svahn .....	207
<i>Kombinert Lager og Deponi for Lav-og Middels Radioaktivt Avfall i Norge (KLDRA-Himdalen)</i> Gordon C. Christensen, Steinar Backe og Elisabeth Strålberg .....	211
<i>Erfaringer fra Transport af Radioaktivt Avfall fra IFE -Kjeller til KLDRA Himdalen</i> Aud Raalum, Berit Juvik, Osmund Mykland, Trygve Gravdahl, Tore Ramsøy og Steinar Backe .....	217
<i>Radiologisk Kartlegging i Himdalen</i> Elisabeth Strålberg, Per Varskog, Tone D.S. Bergan .....	221
<i>Nuclear Threats in the Vicinity of the Nordic Countries</i> Inger M. H. Eikermann and B. Møller .....	225
<i><math>^{210}\text{Pb}</math> and <math>^{238}\text{U}</math> in Estonian Fuel Products and Ashes</i> Küllike Realo and Enn Realo .....	229
<i>Afskærmning med Olivinsand - Om Opbygningen af Lavstrålingsrummet ved SIS</i> Anita Schiøler og Klaus Ennow .....	233
<i>Mapping of the Radiation Fields at a Research Reactor</i> Jens Søgaard-Hansen and Lisbeth Warming .....	237
<i>Kartlegging av Radioaktivt Nedfall ved hjelp af Ull fra Sau</i> Åse Berg .....	241
<i>The Inhabitants of Lund and Gävle and the Legacy of Chernobyl a preliminary Study</i> Bengt Erlandsson .....	245
<i><math>^{137}\text{Cs}</math> Contamination of Trees in the Briansk Region</i> Christian Lange Fogh, Kasper Grann Andersson and Jørn Roed .....	249

<i>Low Emissions of Radionuclides from the Studsvik Site</i> Robert Hedvall .....	253
<i>A Long-term Study of <math>^{137}\text{Cs}</math> Distribution in Soil</i> Mats Isaksson, Bengt Erlandsson and Sören Matsson .....	257
<i>Environmental Monitoring around the Swedish Nuclear Facilities</i> Anita Bondesson, Maria Lüning, Lena Wallberg and Heléne Wijk .....	261
<i>Tsjernobylnedfall i Midt-Norge: Undersøkelser i 1996</i> Hanne Mehli, Lavrans Skuterud og Annhild Mosdøl .....	265
<i>The Role of Bioaccumulation in Migration of Technogenic Radionuclides in Freshwater Ecosystem</i> Davia Petkeviciute .....	269
<i>Level of <math>^{90}\text{Sr}</math> in the Urine of a Small Group of Finnish People</i> Marketta Puhakainen and Matti Suomela .....	273
<i>Uptake and Activity Concentration of <math>^{137}\text{Cs}</math> and <math>^{90}\text{Sr}</math> and <math>^{40}\text{K}</math> in <i>Salix Viminalis</i></i> Klas Rosén, Youhua von Fircks and Lisa Senneby-Forsse .....	277
<i>Differences in the Ecological Half-Time of Radiocaesium for the Chernobyl Accident and from Nuclear Weapons Fallout as measured in a South Swedish Population.</i> Christopher L. Rääf, Bengt Hemdal and Sören Mattsson .....	281
<i><math>^{137}\text{Cs}</math> in Saamis and Reindeer in Kautokeino, Norway, 1965 - 1999</i> Lavrans Skuterud, Hanne Mehli and Lise Flø .....	285
<i>Fordeling og Retensjon av Radioaktivt Cs i Naturlig Jordsmonn i Norge</i> Per Varskog, Trygve O. Bjerk og Eiliv Steinnes .....	289
<i>Risk for Haematological Malignancies after Radiation Treatment of Painful Benign Conditions in the Skeleton</i> Lennart Johansson, L-G. Larsson and L. Damber .....	293
<i>Ländryggsundersökningar - varför varierar Stråldosen så mycket?</i> Wolfram Leitz och Birgitta Hansson .....	297

<i>Apoptosis and other Indicators of Ionizing Radiation at Doses 0.1 - 2.0 Gy in five Cell Culture Systems</i> Alicja Jaworska, Paula De Angelis, Gøril Olsen, Gunnar Kinn, Fredrik Liland and Jon Reitan .....	301
<i>Molecular Dosimetry based on Radiation Induced Degradation of Polyisobutylene</i> Kristina Jørkov Thomsen, Anders Damkjær and Walther Batsberg Pedersen .....	305
<i>Long-term Decrease in the External Radiation Exposure, due to Chernobyl Fallout, to Inhabitants in the Bryansk Area in Russia</i> Charlotte Thornberg, J. Börjesson, Sören Mattson, Elisabeth Wallström, Raine Vesanen, M. Alpsten, I. Zvonova, T. Jesko, V. Kovalenko, and M. Balonov .....	309
<b>List of participants .....</b>	<b>313</b>
<b>List of exhibitors .....</b>	<b>317</b>



## **Preface**

The key themes of the 12th ordinary general meeting of the Nordic Society for Radiation Protection were: RADIATION – ENVIRONMENT – INFORMATION.

It was a great pleasure for our society that a number of outstanding international experts accepted to contribute on the meetings first day with invited presentations, which focussed on these themes. In all 38 oral presentations and 28 posters are included in the present Proceedings, which furthermore contains a resumé of discussions from the special session on "Controllable Dose".

The 12th ordinary meeting was the first in the history of NSFS, which could welcome participants from Estonia, Latvia and Lithuania, the new associated member states of our Society. Six members from the Baltic countries attended the meeting and contributed all with presentations.

The title of our meeting in Skagen was "Radiation in a general environmental pollution perspective with special emphasis on the information of the public". The present Proceedings reflect how well the participants succeeded to cope with this perhaps both ambiguous and ambitious title.

I thank all participants for their contributions to the successful meeting in Skagen.

Asker Aarkrog  
Past president of NSFS  
Risø, 1st November 1999



# **SESSION I**

## **INVITED LECTURES**

### **RADIATION IN A GENERAL ENVIRONMENTAL POLLUTION PERSPECTIVE WITH SPECIAL EMPHASIS ON INFORMATION**

**Chairman: Ulf Bäverstam**





# **NORDIC RADIATION PROTECTION SOCIETY**

## **12<sup>th</sup> ORDINARY MEETING**

AUGUST 23-27, 1999; SKAGEN, DENMARK

### **THE BO LINDELL LECTURE**

# **ENVIRONMENTAL RADIATION PROTECTION AND INFORMATION EXPECTATIONS OF THE PUBLIC - past and future**

Anneli Salo

## **ABSTRACT**

The development of “environmental radiation protection” and changes in information expectations and needs during the past half a century is approached from three angles, namely changes in the importance of the sources of radionuclides, including the targets of the studies; in protection criteria; and in the society, including the expectations and needs of information.

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I want to thank the Nordic Society for Radiation Protection for inviting me to give the Bo Lindell lecture. It is a great honour and privilege for me to present the Bo Lindell lecture at the Nordic Society’s ordinary meeting. Bo Lindell’s central role in the international regime of radiation protection, ICRP, UNSCEAR and other organisations and in his homeland never prevented him to share his knowledge and ideas with his Nordic colleagues. In those days when information exchange was in its baby shoes compared with to-days situation Bo made a bridge to international information sources and co-operation for many of us.

The development of “environmental radiation protection” and related information can be looked at from many perspectives. I intend to discuss it using three angles of vision:

1. changes in the importance of the sources of radionuclides in the environment including the targets of the studies,
2. changes in the protection criteria
3. changes in the society, including the expectations and needs for information

## SOURCES OF RADIONUCLIDES IN THE ENVIRONMENT AND RELATED STUDIES

### *Nuclear weapons tests*

Each decade can be characterised by some distinct source of radionuclides causing environmental concerns and invoking research and monitoring since the 1940s. Monitoring of artificial radionuclides in the environment was introduced in many countries as a consequence of atmospheric nuclear weapons tests in the 1950s. The main motivation for environmental research was the fear that nuclear weapons could be used. This encouraged to learn about environmental behaviour of biologically important fission products and the possible protective measures.

During the pioneer era of 10-20 years the monitoring and research programmes were often chosen rather on the basis what can be measured than on in depth consideration of the use of the results for radiation protection purposes. Very little was known about the chemical and biological behaviour of such elements that the fission products represented. Depending on their background the scientists could be categorised as "analysts" and "metabolists", the former interested in analytical techniques and the latter having studied previously the metabolism of trace elements or other chemical compounds.

Radionuclide contamination of air, water, soil and biota, inclusive man, was studied, with the emphases on important food chains leading to man /UNSCEAR, 1988/. Some scientists used the results for radiation dose calculations for humans but many stopped by environmental concentrations and their trends, which was customary with other contaminants.

It should be noted that widespread concern of the effects of ionising radiation from man made sources on plants and animals existed since 1945. The relative ease of laboratory studies brought about work on various species in the ensuing decades, but much of this work was directed to acute exposures of individual organisms. In the late 1950s increased effort was focused on more ecologically relevant research. Experiments were carried out with longer term exposures at much lower dose rates and attention was paid to responses other than mortality /UNSCEAR 1996/.

A different approach was the use of radionuclide injections, from nuclear weapons tests, into the environment for geophysical or radioecological research. Some satellite and nuclear weapons transport accidents could also be efficiently used for research purposes.

From the environmental radiation protection point of view perhaps the most important step internationally in the early days was that the General Assembly of the United Nations established the Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) in 1955. The Committee has provided estimates of doses received by the world's population in the past and expected to receive in the future from various natural and manmade sources of radiation, and of risks of induction of various types of harm by radiation.

UNSCEAR has introduced several important concepts for environmental radiation protection purposes. These concepts have had an important role in practical radiation protection. Nuclear test explosions in the atmosphere introduced time elements that made this source different from other radiation sources handled so far, in the sense that the period of practice and the period of exposure were different. In other words contamination by long lived radionuclides, spread from past explosions was expected to last for a long time, thus committing mankind to exposures in future years. The concept of *dose commitment* was introduced in 1962.

In 1977 UNSCEAR introduced the concept of collective dose and collective dose commitment, important in the source related assessments for justification and optimisation purposes.

#### *Operational releases from nuclear fuel cycle*

The main emphases moved to environmental consequences of nuclear power production towards the 1970s. Environmental programmes for predicting and corroborating the behaviour and consequences of planned (authorised) releases from nuclear facilities superseded studies using nuclear weapons fall-out as tracers. Modelling of the release – dose to man – relationships before starting the operation, and monitoring in the environment of nuclear facilities during the operation, became vital parts of the authorisation process. Unfortunately it took almost two decades to get the "modellers" and the "environmentalists" to work together for better understanding of the processes in the environment and consequently describing them.

The concept of using indicator organisms, which strongly accumulated important radionuclides was popular at that time. This was partly aimed at getting an early warning of leakage and of accumulation in the environment because the measuring techniques were not yet very sensitive compared with the prescribed release limitations. Indicator organisms were not used to reveal any biological effects, like in the context of some toxic substances, nor to determine radiation doses to potentially most exposed organisms.

Releases from nuclear facilities have also been used for research on physical transport processes in the recipients. Well known examples are the major releases of  $^{137}\text{Cs}$ , as the dominant radionuclide, from Sellafield in the mid- to late 1970s and  $^{106}\text{Ru}$  releases from La Hague.

#### *Natural radioactivity*

Throughout the years natural radionuclides in the environment were studied for assessing the natural radiation burden to man and biota, and to provide a comparison for artificial radioactivity.

In the 1970s and 1980s the emphases was in studying the radiation protection problems caused by radon in dwellings. This seemed to be the major environmental radiation protection problem before the Chernobyl accident in some countries, in particular in Finland and Sweden /CEC 1990s/ as well as Norway of the Nordic countries /Strand 1999/. It continues to be important from the radiation protection point of view, but less so in the public perception, as it is experienced as a natural phenomenon.

A special area of study is the research on "natural analogues" to find solutions and information basis to long term problems with final disposal of high level nuclear waste. The expectations in this area were probably too ambitious concerning parameter values to models. Instead they have fulfilled the competence building expectations and are useful in information activities.

#### *Nuclear accidents/UNSCEAR 1993/*

Two serious accidents in nuclear weapons production, releasing radionuclides into the environment, happened already in 1957, one in Kyshtym in the former Soviet Union and one at the Windscale plant at Sellafield in UK. Only the latter accident was known at that time in the Western world.

In the Three Mile Island reactor accident in 1979 serious damage to the core of the reactor was caused but almost all the fission products were retained in the containment system. The

accident affected considerably the reactor safety community but first the Chernobyl accident in 1986 really brought the nuclear accidents into the spotlight from the environmental point of view. After the urgent studies necessary to protect the population from the consequences of the accident a broad spectrum of research programmes was started in all European countries and elsewhere. The Chernobyl accident actually revived the dormant radioecological studies after a period with hardly any measurable concentrations of artificial radionuclides in the environment. The possibility of harm to plants and animals was also demonstrated and extensive studies initiated.

#### *Nuclear waste*

Concerns about radioactive wastes became louder in the 1990s and exceeded in many countries the concerns about accidents. The primary concern is the final disposal of spent nuclear fuel and other high level waste.

With the deep bedrock disposal options the environmental research interests got a new dimension. Geosciences took a central role but even biological life was encountered deeper in the bedrock than previously believed.

Also other waste related concerns surfaced in the 1990s for various reasons. One initiating event was when it was reported in 1992 that the former USSR had, for over three decades, dumped radioactive wastes in the shallow waters of the Arctic Seas. Part of these dumpings violated the globally applicable legal instrument (London Convention 1972) for the control of radioactive waste dumping at sea, /IASAP, 1998/.

Many other concerns were expressed about chronic exposure and potential future exposures at various contaminated sites of past military and peaceful activities. At various requests a number of international assessments was carried out under the auspices of the IAEA, such as Radiological Conditions of the Western Kara Sea about the above mentioned dumping, Radiological Conditions at Bikini Atoll, /IAEA, 1998a/. The Radiological Situation at the Atolls of Mururoa and Fangataufa, /IAEA, 1998b/ and Radiological Conditions at the Semipalatinsk Test Site, Kazakhstan, /IAEA, 1999/. Another assessment having a broader scope than only radioactivity was carried out as part of the Arctic Monitoring and Assessment Programme (AMAP) on Arctic Pollution Issues, /AMAP, 1998/.

## **DEVELOPMENT OF RADIATION PROTECTION CRITERIA AND GUIDES**

#### *Past mileposts from the environmental viewpoint*

Although "environmental radiation protection" has been exercised half a century, the protection criteria were solely based on minimising the risks to humans, not demonstrating how the environment was protected. Still the protection has been effective also from the viewpoint of the environment as a whole in most cases. This in my view is due to the efficient nuclear safety, limiting the likelihood of accidents and the systematic dual approach in radiation protection: the source related and the individual related assessments to limit the releases into the environment. Also the research efforts were directed all the time to both radiation doses/effects to man and accumulation processes of radionuclides in the environment, with some research on the effects on other species than man. Thus it would be likely that significant harm to populations of other species would have been detected.

The ICRP extended its radiation protection recommendations from occupational protection to cover members of the public in the mid 1950s. ICRP also stated that "recommendations in

quantitative terms are needed in the design of power plants and other radiation installations and particularly in making plans for disposal of radioactive waste products"/ICRP, 1954; ICRP, 1956/. This was stated after the 1955 Geneva conference on the Peaceful Uses of Atomic Energy, which was a turning point in many respects.

In 1958 (ICRP 1) a provisional limit of 5 rem (50 mSv) per generation for the genetic dose (other than background and medical) to the whole population was given, /ICRP, 1959/

Also growing public concern was expressed about the possibility of undesirable contamination of the sea from disposal of radioactive waste in the mid-1950s. In 1956 UNSCEAR initiated a limited study on this matter and reported it in its 1958- report, /UNSCEAR, 1958/

ICRP recommendations were the basis of practical guidance given by other international organisations such as IAEA concerning the releases of radionuclides from practices into the environment. IAEA started its long series of guides on environmental matters in 1961 with guidance on Radioactive Waste Disposal into the Sea, /IAEA 1961/, followed by Disposal of Radioactive Waste into Fresh Water, /IAEA, 1963/ and Radioactive Waste Disposal into the Ground, /IAEA, 1965/.

The initiating event for the guide on sea disposal was the UN Conference on the Law of the Sea 1958, adopting the Convention on the High Seas, which provided that "Every state shall take measures to prevent pollution of the seas from the dumping of radioactive wastes, taking into account any standards and regulations which may be formulated by the competent international organisations". The conference also adopted a resolution recommending that the IAEA pursue whatever studies and take whatever action is necessary to assist States in controlling the discharge or release of radioactive materials to the sea, in promulgating standards, and in drawing up internationally acceptable regulations to prevent pollution of the sea by radioactive materials in amounts which would adversely affect man and his marine resources".

The fresh water guide refers to the effects on both the "human and other forms of life being of first order importance", but assumes that the need to "keep human exposure below acceptable limits probably precludes the possibility of damage to other organisms". This was based on the view that man is one of the most radiosensitive of all organisms, but no reference is given to support this statement. The rest of the document then deals with human exposure only.

In 1965 in ICRP 9 Dose Limits were given for planned exposures of individual members of the public. The previous provisional genetic dose limit (of 50 mSv) for the whole population was restated. It was mentioned that dose limit for the public is somewhat theoretical concept, intended for planning. It was also emphasised that no single type of population exposures should take up a disproportionate share of the total. Environmental contamination should be limited by exposure of members of the public and the necessity in some cases of radioactivity surveys is mentioned, /ICRP, 1965/.

In 1966 (ICRP 7), /ICRP, 1966/, guidance on the principles of environmental monitoring was provided on the basis of ICRP 9. ICRP 7 summarises the broad objectives of environmental monitoring programmes.

First in 1977 in Publication 26, /ICRP, 1977/, other species are mentioned as follows:

"Although the principal objective of radiation protection is the achievement and maintenance of appropriately safe conditions for activities involving human exposure, the level of safety required for the protection of all human individuals is thought likely to be adequate to protect other species, although not necessarily individual members of those species. The commission therefore believes that if man is adequately protected then other living things are also likely to

be sufficiently protected". It should be noted that already a substantial number of studies on biological effects on other species existed according to the references in UNSCEAR 1996.

The above assumption in ICRP 26 on the protection of other species had not been formally defended nor the "sufficient protection" quantified. However, at specific sites the assumption had been shown /e.g. Kaye, 1969; Woodhead 1973, 1974/ to be tenable. It was seriously challenged first after a decade /Thompson, 1988; IAEA, 1988/.

In its 1990 recommendations (ICRP 60) a more cautious formulation was given as follows: "The Commission believes that the standard of environmental control needed to protect man to the degree currently thought desirable will ensure that other species are not put at risk. Occasionally, individual members of non-human species might be harmed, but not to the extent of endangering whole species or creating imbalance between species. At the present time, the Commission concerns itself with mankind's environment only with regard to the transfer of radionuclides through the environment, since this directly affects the radiological protection of man", /ICRP, 1990/. From ICRP 26 to ICRP 60 also the criteria for man's protection had been changed. The emphases of justification of the practice and constrained optimisation of the protection is stronger and the Dose Limit lowered to 1 mSv/a. The Dose Limit presently indicates the boundary between unacceptable and acceptable in normal circumstances. This means that the choice of dose limits involve social judgements and are not based on human health consideration alone. The main aim of constrained optimisation in public exposure was said to be to develop practical restrictions on the sources of the exposure, e.g. restrictions on the release of radioactive waste to the environment.

In 1997 (ICRP 77) the ICRP still sticks to the above policy in its recommendations regarding disposal of radioactive waste, /ICRP, 1997/.

Because of the increasing interest around the world in nurturing the environment and concerns about possible detrimental effects of radiation, the UNSCEAR published a summary of the effects of radiation on the environment, drawing from reported observations in natural and contaminated environments, from experimental studies and from its own assessments of radiobiological effects, /UNSCEAR, 1996/.

#### *Future trends and development*

Since the Earth Summit in Rio de Janeiro in 1992 on the environment and development a change in focus in the environmental protection is evident. Presently the activities which may give rise to radiation impacts on the environment are becoming subject to requirements for both traditional human radiation protection and achieving separate protection of the environment and demonstrating it.

Several countries e.g. USA, Canada, Sweden, and a number of international conventions (e.g. promulgated by IAEA) have adopted principles or are in the process of developing regulatory standards for protection of the environment from ionising radiation. There are several possible approaches which could be adopted but the environmental criteria can hardly be developed any more in isolation from criteria for other potentially polluting activities or activities exploiting natural resources. How could different policies be explained to the public and their confidence obtained?

To co-ordinate within the radiological regime the IAEA has prepared a discussion document on protection of the environment, considering how the present knowledge can best be used for regulatory purposes, /IAEA, 1999/. ICRP is planning to reconsider its position on the matter,

possibly together with other changes in its protection policy for the next millennium, /Clarke, 1998, 1999/.

## CHANGES IN THE SOCIETY AND IN THE INFORMATION EXPECTATIONS

Many societal and technological changes have taken place during the past decades. This chapter tries to summarise some of them, which, in my view, have also changed or may change the needs, contents and communication of information concerning issues related to environmental radiation protection.

### *Changes of concerns in the society*

During the past fifty years the concerns of the people have clearly changed; but how, it depends on the country. Historical and geographical differences between the Nordic countries had a more pronounced impact in the post-war societies than can be observed to day. In the so called welfare societies there is more space for concerns about the quality of life than in societies where the main concern is to get the daily bread or where external threats to life exist. In the 1950s and 1960s the nuclear weapons certainly made such an external threat, which was of joint concern to the Nordic populations.

The Danish tradition of extending scientific and philosophical discussions to the general public in those early days, led to different decision concerning the use of nuclear power than in the neighbouring Sweden, which introduced it in 1972. The nuclear debate in Sweden really started first during the energy crisis 1974-1975, /Djerf, 1988/. At the end of 1980s the environmental questions started getting stronger attention among the politicians, mass media and the public in Sweden, /Hedberg, 1991/.

In the 1960s the Finnish environmentalists still considered that the nuclear power would rescue the rapids of the country and conserve its nature unspoiled. The main environmental concern was hydropower. They changed their attitude against nuclear power first in mid 1970s /Myllyntaus, 1992/. It could still happen in 1967 that the power company IVO had prepared the plans constructing a nuclear power plant, up to the final decision before starting construction, without consulting the owners (the state) and the political decision makers. However, as a consequence the government decided to abandon the acquisition of nuclear power on the basis of the received tenders for some time.

Now, in the 1990s the planning practice is totally different in the Nordic countries. The political decision makers have to give clear signals for the acceptability of such major projects well in advance and the planning practice has also changed due to the Environmental Impact Assessment (EIA) process. The projects have to be brought to public knowledge at an early stage in order to open the interaction between various interested groups; the developers, authorities, the public and interest groups.

In 1970s, at least in countries introducing nuclear power for energy production, may be in neighbouring countries as well, concerns rose in particular as regards nuclear waste and reactor accidents; the latter increased in 1980s after the Three Mile Island and the Chernobyl accidents and the former is presently and presumably some years in the next millennium the main concern in many countries producing electricity with nuclear energy.

In 1990s the concerns of our environment in general, including environmental contamination by radionuclides and radiation effects on human and non-human species, have become important factors in decision making. Environmental awareness has also led to the use of



environmental factors for marketing of products, e.g. energy, or speeding up abandoning of some chemical products.

#### *Improved information/communication technology*

The largest change is probably in information/communication technology, both regarding the types of data processing and communication means and their properties, such as the speed, range, capacity etc.. Just to give some examples: in early 1950s, in my country one computer was provided for scientific purposes and now over 40% of households have PC. In the beginning of 1999 internet connections per 1000 inhabitants were in Finland 107, Iceland 81, Norway 73, Denmark 54 and Sweden 49 /RIPE DNS hostcount, 1999/.

People can easily be reached. The number of cellular phones passed this spring that of ordinary ones, being now 3.1 million in Finland, which means 60% of the population; in Sweden 53, Norway 49, Denmark 40 and Iceland 37% respectively have cellular phones, /Tilastokeskus, 1999/. Development in the technology continues to be rapid but all the expectations of the "information society" have not yet become true, the possibilities are still predominantly used for entertainment and plays by the public. Video conferencing is increasingly used for meetings and for teaching.

#### *Openness in information*

As is well known there are differences between the European and even between the Nordic countries in the information culture. Never the less changes into the same direction have taken place in the past half a century, although not with the same speed or extent in all countries

Presently the information expectations by the public, at least in the Nordic countries, are the high publicity of documents and high transparency of decisions. The requirements of media, authorities and the public to be informed on observations, events etc. immediately has even become sometimes a controversy in particular between the media on one hand and the scientists and authorities on the other hand, the latter two wanting to be certain of the correctness of the information before releasing it. Media want to publish rapidly, rather as first and even uncertain information. They want to select the topics that they consider important on other grounds than the scientists /Pini, 1995; Saxén, 1999/.

#### *Information needs for democratic decision making*

We all apply in our countries representative democratic decision making processes. The important recipients of substance information from the experts in various fields are in this case the elected political decision makers. However, the trends in our region have been to move towards more direct democracy e.g. by using national or local referenda, as examples some EU matters, continuation of the use of nuclear power, siting of nuclear waste repositories.

There are also negative examples of direct democracy e.g. if there are weak leaders and strong media influence it can lead to so called Gallup-democracy, in which the well or badly done polls actually determine the course of action for the politicians. In this case media practically make the decisions.

The most recent development is the "Internet democracy", in which the politicians have a direct contact with their supporters via Internet without media mediation. From the communication viewpoint this is interesting and gives an opportunity to dialogue. However, it still involves information acquisition either from mass media or other sources.

Presently preferred form of direct democracy is continuous participation in the planning process of environmentally important projects; the Environmental Impact Assessment (EIA) process aims at that. Since the beginning of the 70's besides the original EIA also other impact assessments on social matters (SVA) and risks (RIA) have developed and been included in the EIA process.. In recent years such assessments have changed from pure expert assessments to methods with public participation (meaning not only to be heard but to be able to influence the outcome). This offers a procedure for interaction between the public and other parties before decision making and shows quite clearly the importance of psycho-social factors, besides the technical, in decision making. The EIA process is presently applied in more than 100 countries.

One could say that in nuclear projects the EIA has "forced" into simplification of matters difficult to understand. This has been and still is a challenging task for experts in the field. However, a proper balance between producing new scientific information and popularising it, is vital, and must not be forgotten.

Experience so far has shown that fairly small percentage of the population is actively participating in the two way communication process and further more that in areas where several EIA procedures have been followed through in succession, a kind of EIA fatigue is perceived when the novelty value of the process is wearing off. Thus continuous efforts are needed to meet the purpose of the process.

#### *Information needs for individual's well-being*

Democratic decision making is not the only reason for the need of improved communication in environmental matters. There is a psychological need to know about one's environment and how it affects one's well-being. Quantitative risk estimates and expert information are not sufficient to the public. People organise received information in relation to social, ethical and political properties of the project or situation in question, when assessing the acceptability of the risks involved. Acceptability is the key element in risk perception, /Paavola, 1999; Eränen, 1993/.

Nuclear energy and radionuclides in the environment can arouse fear and stress reactions in people for various reasons, which should be analysed to form the basis of communication.

People have different strategies to cope with stressors. The coping strategies can vary depending on the case. The most rational coping strategy is such that his/her interpretation of the situation covers both the facts and the emotional levels and combines them to a decision on action.

In the case of normal releases from nuclear installations and concerning waste repositories a clear decision making process and the opportunity to influence the decision making can affect the experience of threat. In the event of an accident people can not influence the stressor, but they can mobilise their mental resources to action resources. This has an important impact in coping with the situation. Besides individual's coping strategies also the communities have strategies to jointly cope with stress situations, /Eränen, 1993/.

## **IMPROVED COMMUNICATION**

It has never been easy to communicate on environmental radioactivity and related risks to people. We have an advanced source-related protection philosophy based on human protection. We need to complement it to cover other species. New challenges as regards other species will be even more difficult, if a simple and practical protection philosophy is not

established, which will also cover other substances and uses of the environment. In environmental protection it is difficult, and even confusing, to treat the various things affecting the environment in isolation. Joint approach may also be one way to demystify radionuclides.

Fear and anxiety can be reduced by increasing communication between the concerned parties in order to make the issues more familiar. Fear for unknown increases suspicions. Peoples estimate on the reliability of the received information depends on

- the contents of the information and how contradictory it is,
- their interpretation of and confidence in the information source
- belief if the source tends to hide important facts

Communication processes, such as EIA, provide good framework for information exchange, because they are connected to some concrete project, motivating participation. Such information exchange provides people with resources to modify their fears and threat pictures collectively to more rational action. Post accident situations require their own planned communication processes to assist people to cope with the situation both on physical and emotional levels. Confidence building in advance, and uncontradictory timely information and advise in the accident situation are vital.

It is important that the science-technical experts concentrate on communicating the scientific facts to the public in an understandable manner and keep in mind that people react to various issues within the framework of interaction process in which they together with other members of the society formulate perceptions and definitions of the phenomena in their environment..

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THE BALANCE BETWEEN SCIENTIFIC AND PUBLIC INFLUENCE  
ON  
MARINE ENVIRONMENTAL PROTECTION

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ABSTRACT

This review examines the basis for the development of international marine environmental protection agreements since the 1972 Stockholm Conference on the Human Environment. It is illustrated primarily by developments within the London Convention 1972 (*the Convention for the Prevention of Marine Pollution from the Dumping at Sea of Wastes and Other Matter, London 1972*). These developments are traced through a series of limitations on sea disposal options culminating in the adoption of a new Protocol in 1996.

Particular emphasis is given to concepts underlying new initiatives for marine environmental protection. Some of these have been revised in response to changing public perceptions and have led the way to more restrictive regimes with questionable improvements in overall environmental benefits. In this context, the development and widespread adoption of the so-called "precautionary" approach to environmental protection has had a profound influence on marine environmental protection initiatives particularly during the last decade. This appears to reflect societal sympathies towards the adoption of simplistic and radical solutions to problems that are frequently amenable to resolution through the application of sound scientific and management principles.

The primary purpose of this critical assessment is to illustrate the declining influence of science and the increased importance of public and political perception to marine environmental management during the last quarter of a century.



## **Radiation Protection : A Shining Example or a Dazzling Image?**

Dr R J Pentreath

The system developed for the protection of man from the effects of radiation is both elegant and comprehensive. It has taken many decades to evolve and has, quite rightly, been seen as an example of sound science, logic, impeccable peer review, and uncompromising leadership in setting standards and criteria which have been internationally accepted. Its evolution has also been accompanied by outstanding technological developments, particularly in relation to radiometrology and its associated analytical chemistry. It is therefore surprising, to some, that it attracts any form of critical comment. And it is also surprising, to some, that such a system has not been more widely adopted for other forms of pollution control. So what lessons, if any, can be learned?

There are many comments which could be made, in including the problems of translating such a theoretically sophisticated set of concepts, criteria, and - continually changing - definitions and ideas into practical effect. The sheer complexity has made matters difficult for regulators, the judiciary, and operators; and rendered it incomprehensible to the public. Much of this could still be rectified. Another problem is that of re-dressing the balance of protecting man **and** the environment in an explicit way.

Other problems have arisen because of the basis of describing the environmental distribution and effects of radionuclides in purely radiometric terms. Some useful lessons can be drawn here for a pragmatic interpretation of environmental protection in general.

Because of the need to develop an equitable approach to all environmental problems, particularly in relation to the sustainable management of energy and natural resources, it is useful to ensure that a clear and sensible perspective emerges over the next few years.





# The Role of Risk Perception for Risk Management

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## 1. Introduction

Talking about risks faces the immediate danger that everybody talks about something different (see for example: Fischhoff *et al.* 1984; Short 1984; Renn 1992a; Vlek 1996). There is no commonly accepted definition for the term risk -neither in the sciences nor in public understanding. All risk concepts have one element in common, however: the distinction between reality and possibility (Markowitz 1991; Evers and Nowotny 1987). If the future were either predetermined or independent of present human activities, the term "risk" would make no sense. If the distinction between reality and possibility is acknowledged, the term "risk" is often associated with the possibility that an undesirable state of reality (adverse effects) may occur as a result of natural events or human activities (similar in Kasperson and Kasperson 1987: viii; Starr and Whipple 1991: 53; Hillgartner 1992: 40; the same idea is expressed in more technical terms in: National Research Council 1983; ICE 1993). It may be difficult to determine, however, what characteristics are necessary to label an outcome as "adverse" rather than "desirable" or "tolerable" (cf. discussion in: Renn 1990; HMSO 1988). In particular, if one intends to include the common use of the term "risk" in economic theory, both gains and losses need to be subsumed under "risk".

In addition, Machlis and Rosa pointed out that there is a phenomenon such as "desired risk" (for example in sports activities) that people aspire to reach for experiencing a special thrill (Machlis and Rosa 1990). Rosa hence recommended to use the term "risk" for uncertain outcomes regardless whether they are positive or negative (Rosa 1996: 19). Inspired by these arguments, I have chosen the following definition. Risks refer to the *possibility that human actions or events lead to consequences that affect aspects of what humans value.*

This definition implies that humans can and will make causal connections between actions (or events). Consequences are perceived from a non-fatalistic viewpoint. They can be altered either by modifying the initiating activity or event or by mitigating the impacts (Appelbaum 1977). Risk is therefore both a descriptive and a normative concept. It includes the analysis of cause-effect relationships, which may be scientific, anecdotal, religious or magic (Douglas 1966; Wiedemann 1993), but it also carries the implicit message to reduce undesirable effects through appropriate modification of the causes or, though less desirable, mitigation of the consequences.

*Risk assessment* is the scientific process of defining the components of risk in precise, usually quantitative terms. In technical risk assessments, this means specifying what is at stake, calculating the probabilities for (un)wanted consequences, and aggregating both components by multiplying the probabilities by the magnitude of the effects (Kolluru and Brooks 1995: 2.3f). Technical analysis provides society with a narrow definition of undesirable effects and confines possibilities to numerical probabilities based on relative frequencies. However, this narrowness is a virtue as much as it is a shortcoming. Focused on

"real" health effects or ecological damage, technical analyses are based on a societal consensus of undesirability and a (positivistic) methodology that assures equal treatment for all risks under consideration. *Risk management* refers to the process of reducing the risks to a level deemed tolerable by society and to assure control, monitoring, and public communication (Morgan 1990, Kolluru 1995; Fischhoff 1996). Since risk refers to a potential of "real" consequences, it is both a social construction and a representation of reality (Renn 1992a; Renn *et al.* 1992, cf. Short 1989, p. 405).

The social science perspectives on risk broaden the scope of undesirable effects, include other ways to express possibilities and likelihood, and expand the understanding of reality to include the interpretations of undesirable events and "socially constructed" realities (Bradbury 1989; Renn 1992a). Within the social science framework, *risk perception* denotes the process by which people receive and process physical signals (such as witnessing an explosion) as well as information about possible outcomes of human actions or natural events (Renn 1991). People tend to form an opinion or an attitude in response to the perceived signals or information about risks.

This paper attempts to review the present knowledge of risk perception and to explain the psychological and social factors that shape the experience of risk. The main objective of this paper is to integrate the results of psychological, sociological, and cultural studies and present the major findings of the social sciences as they seem relevant for a deeper understanding of risk and the role of risk perception for risk management.

## **2. How Do Individuals Perceive Risks?**

### **2.1 Attention and Selection Filters**

Most risks that modern society faces are not experienced by human senses but learned through communication. Rarely do we face disasters personally; however, the media provide us ample information about hazardous events wherever they take place. The dangers of technologies or nature, the risks of food additives or chemicals in drinking water, the threat of nuclear disaster or a chemical explosion would probably never reach public attention unless society communicates about these adverse possibilities. Risk perception is less a product of experience or personal evidence than a result of social communication (Luhmann 1986).

This observation has major consequences: Today's society provides an abundance of information, much more than any individual can digest. It is assumed that the average person is exposed to 7,000 bits of information each day of which s/he perceives around 700, acknowledges 70, stores seven in the short term memory and may remember less than one in the longer term. Most information to which the average person is exposed will be ignored. This is not a malicious act but a sheer necessity in order to reduce the amount of information a person can process in a given time. Human evolution has provided us with an almost automated and often subconscious tool of selecting the important information from the abundance of information supplies.

In order to economize information processing, individuals are likely to evaluate whether it is necessary to study the content of the information in detail or to make a fast judgment according to some salient cues in the message received. The first strategy refers to the central route of information processing, the second to the peripheral route (Petty and Cacioppo 1986;

Renn and Levine 1991). In the central mode, the receiver performs two types of evaluations: first, an assessment of the probability that each argument is true; and second, an assignment of weight to each argument according to the personal salience of the argument's content. The credibility of each argument can be tested by referring to personal experience, plausibility, and perceived motives of the communicator. The major incentives for changing an attitude in the central mode are the proximity with and the affinity to one's own interests, values, and world views.

In the peripheral mode, receivers do not bother to deal with each argument separately, but look for easily accessible clues to make their judgment on the whole package. Examples of such cues are the length of a message, the number of arguments, the package (color, paper, graphic appeal, and others), and the presence of symbolic signals that trigger immediate emotional responses (cf. Kasperson *et al.* 1988).

## 2.2 *Intuitive heuristics*

Once information has been received, common sense mechanisms process the information and help the receiver to draw inferences. These processes are called intuitive heuristics. They are particularly important for risk perception since they relate to the mechanisms of processing probabilistic information. Early psychological studies focused on personal preferences for probabilities and attempted to explain why individuals do not base their risk judgments on expected values, i.e. the product of probability and magnitude of an adverse effect (Pollatsek and Tversky 1970; Lopes 1983). One of the interesting results of these investigations was the discovery of systematic patterns of probabilistic reasoning that are well suited for most everyday situations. People are risk averse if the stakes of losses are high and risk prone if the stakes for gains are high (Kahneman and Tversky 1979). Many people balance their risk taking behavior by pursuing an optimal risk strategy which does not maximize their benefits but assures a satisfactory payoff and the avoidance of major disasters (Luce and Weber 1986).

Second, more specific studies on the perception of probabilities in decision making identified several biases in people's ability to draw inferences from probabilistic information (Festinger 1957; Kahneman and Tversky 1979; Ross 1977; Renn 1990). These biases are summarized in Table 1

Although these biases constitute clear violations of logical rules, they might have been overrated in the literature (Fischhoff *et al.* 1981). Many laboratory situations provide insufficient contextual information to provide enough cues for people on which they can base their judgments (Lopes 1983). Relying on predominantly numerical information and being unfamiliar with the subject, many subjects in these experiments retrieve to "rules of thumb" in drawing inferences. In many real life situations, experience of and familiarity with the context provide additional information to calibrate individual judgments, particularly for nontrivial decisions (cf. Heimer 1988). Nevertheless, risk managers should be aware of these biases because they are found in public perception and may be one of the underlying causes for the observed public response.

**TABLE 1: INTUITIVE BIASES OF RISK PERCEPTION**

BIASES	DESCRIPTION
<i>Availability:</i>	Events that come to people's mind immediately are rated as more probable than events that are less mentally available.
<i>Anchoring effect:</i>	Probabilities are adjusted to the information available or the perceived significance of the information.
<i>Representativeness:</i>	Singular events experienced in person or associated with properties of an event are regarded as more typical than information based on frequencies.
<i>Avoidance of cognitive dissonance</i>	Information that challenges perceived probabilities that are already part of a belief system will either be ignored or downplayed.

### 2.3 *Semantic Images*

Psychological research has revealed different meanings of risk depending on the context in which the term is used. With respect to technological risk four distinct semantic images can be identified (Renn 1990). These four semantic models of risk are summarized in Table 2. In addition to these models, additional images of risk exist for natural disasters and lifestyle risks.

*Risk as a pending danger (Damokles sword):* Risk are seen as a random threat that can trigger a disaster without prior notice and without sufficient time to cope with the hazard involved. This image is linked to artificial risk sources with large catastrophic potential. The magnitude of the probability is not considered. It is rather the randomness itself that evokes fear and leads to avoidance responses. The image of pending danger is particularly prevalent in the perception of large-scale technologies

*Slow killers (Pandorra's Box):* Risk is seen as an invisible threat to one's health or well-being. Effects are usually delayed and affect only few people at the same time. Knowledge about these risks is based on information by others rather than on personal experience. These risks pose a major demand for trustworthiness in those institutions that provide information and manage the hazard. If trust is lost, people demand immediate actions and assign blame to these institutions even if risks are very small. Typical examples of this risk class are food additives, pesticides, and radioactive substances. The recent concern about the so called "Mad Cow Disease" (BSE) is a typical example for a risk that belongs in the Pandorra's box. Due to the importance of trust in monitoring and managing slow killers, risk managers should place a major effort to improve their trustworthiness and credibility in the community.

**TABLE 2: THE FOUR SEMANTIC IMAGES OF RISK IN PUBLIC PERCEPTION**

1.	<i>Pending Danger (Damokles Sword)</i>
	<ul style="list-style-type: none"> <li>• artificial risk source</li> <li>• large catastrophic potential</li> <li>• inequitable risk-benefit distribution</li> <li>• perception of randomness as a threat</li> </ul>
2.	<i>Slow Killers (Pandorra's Box)</i>
	<ul style="list-style-type: none"> <li>• (artificial) ingredient in food, water, or air</li> <li>• delayed effects; non-catastrophic</li> <li>• contingent on information rather than experience</li> <li>• quest for deterministic risk management</li> <li>• strong incentive for blame</li> </ul>
3.	<i>Cost-benefit Ratio (Athena's Scale)</i>
	<ul style="list-style-type: none"> <li>• confined to monetary gains and losses</li> <li>• orientation towards variance of distribution rather than expected value</li> <li>• asymmetry between risks and gains</li> <li>• dominance of probabilistic thinking</li> </ul>
4.	<i>Avocational Thrill (Hercules Image)</i>
	<ul style="list-style-type: none"> <li>• personal control over degree of risk</li> <li>• personal skills necessary to master danger</li> <li>• voluntary activity</li> <li>• non-catastrophic consequences</li> </ul>

*Cost-benefit ratio (Athena's Scale):* Risks are perceived as a balancing of gains and losses. This concept of risk comes closest to the technical understanding of risk. However, this image is only used in peoples' perceptions of monetary gains and losses. Typical examples are betting and gambling both of which require sophisticated probabilistic reasoning. People are normally able to perform such probabilistic reasoning but only in the context of gambling, lotteries, financial investment, and insurance. Laboratory experiments show that people orient their judgment about lotteries more towards the variance of losses and gains than towards the expected value.

*Avocational thrill (Hercules theme):* Often risks are actively explored and desired (Machlis and Rosa 1990). These risks include all activities for which personal skills are necessary to master the dangerous situation. The thrill is derived from the enjoyment of having control over one's environment or oneself. Such risks are always voluntary and allow personal control over the degree of riskiness.

The semantic images allow individuals to order risks and risk sources on the basis of a few salient characteristics. Reducing complexity by creating classes of similar phenomena is certainly a major strategy for coping with information overload and uncertainty. The four

semantic images are powerful guides to help individuals to navigate through an abundance of often contradicting information and provide a rather efficient method to balance the time for collecting and processing information with the personal need for orientation and attitude formation.

## 2.4 Qualitative risk characteristics

In addition to the images that are linked to different risk contexts, the type of risk involved and its situational characteristics shape individual risk estimations and evaluations (Slovic 1987; Renn 1990). Psychometric methods have been employed to explore these qualitative characteristics of risks. The following contextual variables of risk have been found to affect the perceived seriousness of risks (Slovic *et al.* 1981; Vlek and Stallen 1981; Covello 1983; Gould *et al.* 1988; Renn 1990; Jungermann and Slovic 1993):

- *the expected number of fatalities or losses:* Although the perceived average number of fatalities correlates with the perceived risk of a technology or activity, the relationship is weak and generally explains less than 20 percent of the declared variance.
- *the catastrophic potential:* Most people show distinctive preferences among choices with identical expected values (average risk). Low-probability high-consequence risks are usually perceived as more threatening than more probable risks with low or medium consequences.
- *situational characteristics:* Surveys and experiments have revealed that perception of risks is influenced by a series of perceived properties of the risk source or the risk situation. Among the most influential factors are: the perception of dread with respect to the possible consequences; the conviction of having personal control over the magnitude or probability of the risk; the familiarity with the risk; the perception of equitable sharing of both benefits and risks; and the potential to blame a person or institution responsible for the creation of a risky situation. In addition, equity issues play a major role in risk perception. The more risks are seen as unfair for the exposed population, the more they are judged as severe and unacceptable (Kasperson and Kasperson 1983; Short 1984). The most influential factors are listed in Table 3.

**TABLE 3: LIST OF IMPORTANT QUALITATIVE RISK CHARACTERISTICS**

<i>Qualitative Characteristics</i>		<i>Direction of Influence</i>
1.	Personal control	increases risk tolerance
2.	Institutional control	depends on confidence in respective institution
3.	Voluntariness	increases risk tolerance
4.	Familiarity	increases risk tolerance
5.	Dread	decreases risk tolerance
6.	Inequitable distribution of risks and benefits	depends on individual utility, strong social incentive for rejecting risks
7.	Artificiality of risk source	amplifies attention to risk, often decreases risk tolerance
8.	Blame	increases quest for social and political responses

- *the beliefs associated with the cause of risk:* The perception of risk is often part of an attitude that a person holds about the cause of the risk, i.e. a technology, human activity, or natural event. Attitudes encompass a series of beliefs about the nature, consequences, history, and justifiability of a risk cause (Thomas *et al.* 1980; Otway and Thomas 1982). Due to the tendency to avoid cognitive dissonance, i.e. emotional stress caused by conflicting beliefs (Festinger 1957), most people are inclined to perceive risks as more serious and threatening if the other beliefs contain negative connotations and vice versa. Often risk perception is a product of these underlying beliefs rather than the cause for these beliefs (Clarke 1989).

It should be noted that the estimation of seriousness and the judgment about acceptability are closely related in risk perception. Most people integrate information about the magnitude of the risk, the fairness of the risk situation, and other qualitative factors into their overall judgment about the (perceived) seriousness of the respective risk.

### **3. How Do Social Factors Influence Risk Perception?**

#### **3.1 *Institutional Trust and Confidence***

Among the most influential social factors in shaping risk perception and responses social networks and reference group judgments are particularly influential since most information about risk is not learned through personal experience and senses but through "second-hand" learning. With the advent of ever more complex technologies and the progression of scientific methods to detect even smallest quantities of harmful substances, personal experience of risk has been more and more replaced by information about risks and individual control over risk by institutional risk management. As a consequence, people rely more than ever on the credibility and sincerity of those from whom they receive information about risk. Thus, trust in institutional performance has been a major key for risk responses. Trust in control institutions are able to compensate for even a negative risk perception and distrust may lead people to oppose risks even when they are perceived as small.

Trust on a personal level is a subjective expectation that a person will refrain from behavioral options that may harm the trusting person. Trust necessarily entails risk-taking, but, in contrast to the scientific endeavor of predicting the probability of potential outcomes, trust implies that the selection of options is left to the entrusted person or institution. Due to the perceived competency and honesty of the entrusted entity, one does not need to bother with assessing the outcomes of actions and with controlling the decision making process of that entity (Luhmann 1980, 1973). This saves time and effort.

On a more aggregate level, trust denotes a generalized medium of social differentiation and division of labor (Parsons 1960). The performance of specialized institutions in economy and government relies on a prior investment of trust by those who are served by this institution or finance its functioning. Total control would imply that the control agencies would need the same expertise and the same time allocation as the performing institution. Such an arrangement would neutralize the desired effect of social differentiation and ultimately lead to a society of intimate clans performing all necessary social, economic, and political functions simultaneously. By shortcutting normal control mechanisms, trust can be a powerful agent for efficient and economical performance of social tasks (Durkheim 1933; Luhmann 1973).



It is obvious that modern societies face difficulties in providing sufficient trust for reaching consensus on its complex and differentiated activities. All public institutions have lost trust and credibility over the last two decades except for the news media (Lipset and Schneider 1983). Trust and credibility losses are high for industry, the political system, and many government agencies. Science still has a high degree of credibility although much less than two decades ago. Most sociologists believe that the decline of confidence in public institutions is partially a function of better education and the increase of public aspirations with respect to their share of public resources and welfare (Lipset and Schneider 1983). In addition, the complexity of social issues and the pluralization of values and lifestyles may have contributed to a growing dissatisfaction with the actual performance of institutions (Renn 1995). But at the same time, most people are confident in the governmental and economic system and do not support fundamental changes in the organizational structure of society. Therefore, the confidence crisis is less a systems than a performance or competence crisis.

Lack of trust does not indicate, however, a declining relevance of trust for governing modern societies and managing technological risks. The contrary is true. The reliance of the technological society on trustful relationships between and among its subsystems has never been stronger than today. However, such a need for trust makes people more and more sensitive towards situations in which their investment of trust has been factually or allegedly misguided. The more trust is needed for implementing cooperative efforts or for coping with external effects of social actions, the more cautious are people in assigning credibility to those whom they are supposed to trust.

This is particularly relevant for risk issues. Since the notion of risk implies that random events may trigger accidents or losses, risk management institutions are always forced to legitimate their action or inaction when faced with an accident. On one hand they can cover up mismanagement by referring to the alleged randomness of the event (labeling it as unpredictable outcome or an act of God), on the other hand they may be blamed for events for which they could not possibly provide protective actions in advance (Luhmann 1986). The stochastic nature of risk demands trustful relationships between risk managers and risk bearers, since single events do not prove nor disprove management failures; at the same time they provoke suspicion and doubt.

In risk debates issues of trust evolve around institutions and their representatives. People's responses to risk depend among others on their confidence that they have in risk initiating and controlling institutions. The handling of risk by private corporations and governmental agencies has been crucial for explaining the mobilization rate of individuals for taking actions. The more individuals believe that risks are not properly handled (in addition to being perceived as serious threats) the higher is the likelihood that people will be politically active. Negative attitudes are one necessary but by far not a sufficient reason for behavioral responses. Public confidence in institutional performance is another and even more important element in triggering behavioral responses.

### **3.2    *Value Commitments***

Positions towards risks of technologies or human activities differ as a result of divergent views about the goal(s) and values that are to be accomplished by providing risk-related technologies for production, consumption or distribution. In the social sciences, values

are placed in clusters that seem to belong together although most people are characterized by mixed values systems (Fiorino 1989). These clusters are summarized in Table 4:

**TABLE 4: VALUE CLUSTERS AND THEIR SOCIAL AND CULTURAL FUNCTIONS**

<i>Cluster Name</i>	<i>Examples</i>	<i>Function</i>
<i>traditional values</i>	patriotism, regional or ethnic identity, social status, family stability	group and cultural identity
<i>work ethics</i>	diligence, punctuality, efficiency, discipline, deferred gratification	functionality, efficiency
<i>hedonistic values</i>	consumption, enjoyment, fun, immediate gratification	incentive, motivation
<i>postmaterialistic values</i>	harmony, social responsibility, environmental quality, decentralization, quality of life	moral legitimization, cultural commitment

In contrast to many popular views, there is not a universal shift towards postmaterialistic values throughout the western world (Klages 1984). It is true that these values have become more important and can be found on the value priority list of almost every individual, but the claim of a new postmaterialistic personality is total fiction. Most people demonstrate a mix of all value clusters depending on context and social relations. A vast majority of people is still interested in gaining additional personal income (even if they rate it low on the scale of personal aspirations). Even unfashionable virtues such as discipline and efficiency have their place in most people's value portfolio. However, many of the traditional and work-related values are withdrawn from situations in which they used to be the dominating orientations. This has been the case with many technologies: they were perceived as manifestations of work ethics and hedonistic values (production and consumption), but are increasingly related to postmaterialistic concerns. This shift in value application is partially responsible for the perception of ambivalence, which is so typical for modern attitudes towards technologies.

Since values have an impact on technology evaluation, responses to risk rely on the requirement that they load high on each value cluster except for the traditional values. Traditional values are normally disassociated with the use of technologies. Positive responses to risk-related technologies or activities depend on the possibility of the evaluator to forge links between the three value clusters and the development or use of the technologies under consideration. That is why technology design and development should acknowledge and incorporate the likely implications of new products or processes for the three major value clusters. If the perceived implications meet all three clusters, acceptance problems are unlikely to play a major role in the diffusion process.

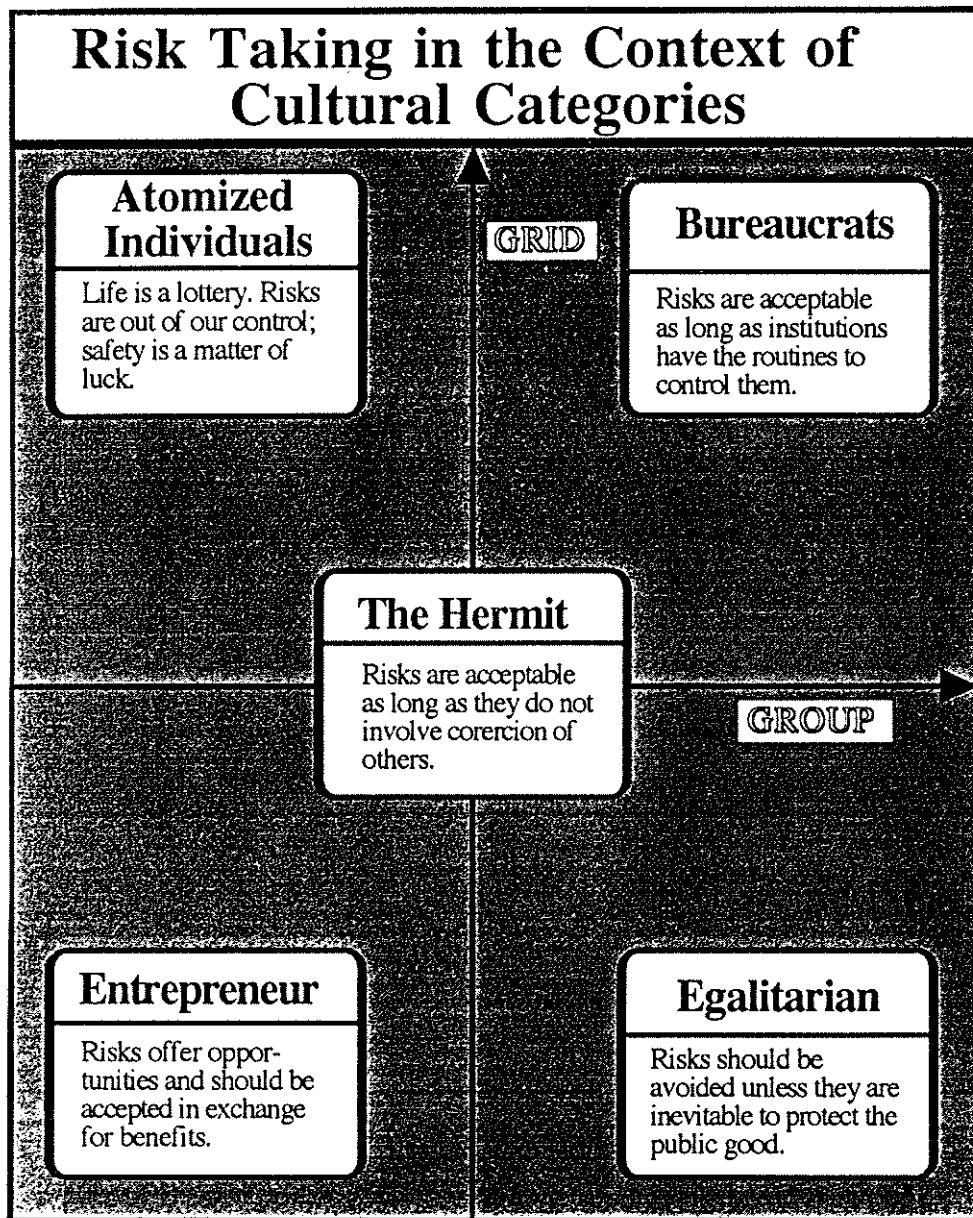
### 3.3 Cultural Group Affiliations

In recent years, anthropologists and cultural sociologists have investigated the social response to risk and have identified four or five patterns of value clusters that separate different cultural groups from each other (Thompson 1980; Douglas and Wildavsky 1982; Rayner and Cantor 1987; Schwarz and Thompson 1990; Thompson, Ellis and Wildavsky 1990). These different groups have formed specific positions on risk topics and have developed corresponding attitudes and strategies. They differ in the degree of *group cohesiveness* (the extent to which someone finds identity in a social group), and the degree of *grid* (the extent to which someone accepts and respects a formal system of hierarchy and procedural rules).

There are four major groups in modern society that are likely to enter the risk arena: the entrepreneurs, the egalitarians, the bureaucrats, and the stratified individuals. They can be localized within the group-grid space as shown in fig.1. Organizations or social groups belonging to the *entrepreneurial* prototype perceive risk taking as an opportunity to succeed in a competitive market and to pursue their personal goals (Rayner 1987). They are characterized by a low degree of hierarchy and a low degree of cohesion. This group contrasts most with organizations or groups belonging to the *egalitarian* prototype, which emphasizes cooperation and equality rather than competition and freedom. Egalitarians are also characterized by low hierarchy, but have developed a strong sense of group cohesiveness and solidarity. When facing risks they tend to focus on long term effects of human activities and are more likely to abandon an activity (even if they perceive it as beneficial to them) than to take chances. The third prototype, i.e. the *bureaucrats*, relies on rules and procedures to cope with uncertainty. Bureaucrats are both, hierarchical and cohesive in their group relations. As long as risks are managed by a capable institution and coping strategies have been provided for all eventualities, there is no need to worry about risks. The fourth prototype, the group of *atomized or stratified individuals*, principally believe in hierarchy, but they do not identify with the hierarchy to which they belong. These people trust only themselves, are often confused about risk issues, and are likely to take high risks for themselves, but oppose any risk that they feel is imposed on them (Thompson 1980).

Finally, the last group is the group of *autonomous individuals* in the center of the group-grid coordinates. Thompson describes autonomous individuals as self-centered hermits and short-term risk evaluators. I like to refer to them as potential mediators in risk conflicts, since they build multiple alliances to the four other groups and believe in hierarchy only if they can relate the authority to superior performance or knowledge (Renn 1992b).

Cultural theory has been criticized on several grounds (Nelkin 1982, Johnson 1987, Renn 1992a). First, most authors within the cultural theory emphasize that cultural prototypes do not characterize individuals but social aggregates. Second, the relationship between cultural prototype and organizational interest is unclear and problematic. If cultural affiliation precedes interest, then what determines to which cultural prototype groups or organizations belong? Third, the selection of the five prototypes as the only relevant cultural patterns in modern society needs more evidence than the reference to tribal organizations (Douglas 1985) or generic models of human interactions (Wildavsky and Dake 1990). Furthermore, if prototypes are mixed in organizations, then the perspective (similar to many sociological concepts) is not falsifiable. Any observed behavior is compatible with some mix of prototypes. Lastly, the cultural perspective has not provided sufficient empirical evidence of its validity.



**Figure 1: Cultural prototypes and their risk profile:** Organizations or social groups belonging to the *entrepreneurial* prototype perceive risk taking as an opportunity to succeed in a competitive market and to pursue their personal goals. This group contrasts most with organizations or groups belonging to the *egalitarian* prototype which emphasizes cooperation and equality rather than competition and freedom. When facing risks they tend to focus on long term effects of human activities and equity concerns. The third prototype, i.e. the *bureaucrats*, relies on rules and procedures to cope with uncertainty. As long as risks are managed by a capable institution and coping strategies have been provided for all eventualities, there is no need to worry about risks. The fourth type, the group of *atomized or stratified individuals*, are often confused about risk issues, and are likely to take risks for themselves, but oppose any risk that they feel is imposed on them. Finally, the last group is the group of *autonomous individuals* in the center of the group-grid coordinates. They can be described as self-centered hermits or “born” communicators between the other groups.

The cultural theory of risk has its shortcomings and its merits. The reduction of cultural clusters to basically three important prototypes (entrepreneurial, egalitarian, and bureaucratic) may be a valid and intuitively plausible hypothesis in analyzing risk responses, but it should be treated as a hypothesis rather than the exclusive explanation. The emphasis on values and world views rather than interests and utilities (which in themselves are reflections of one world view) is a major accomplishment of this theory.

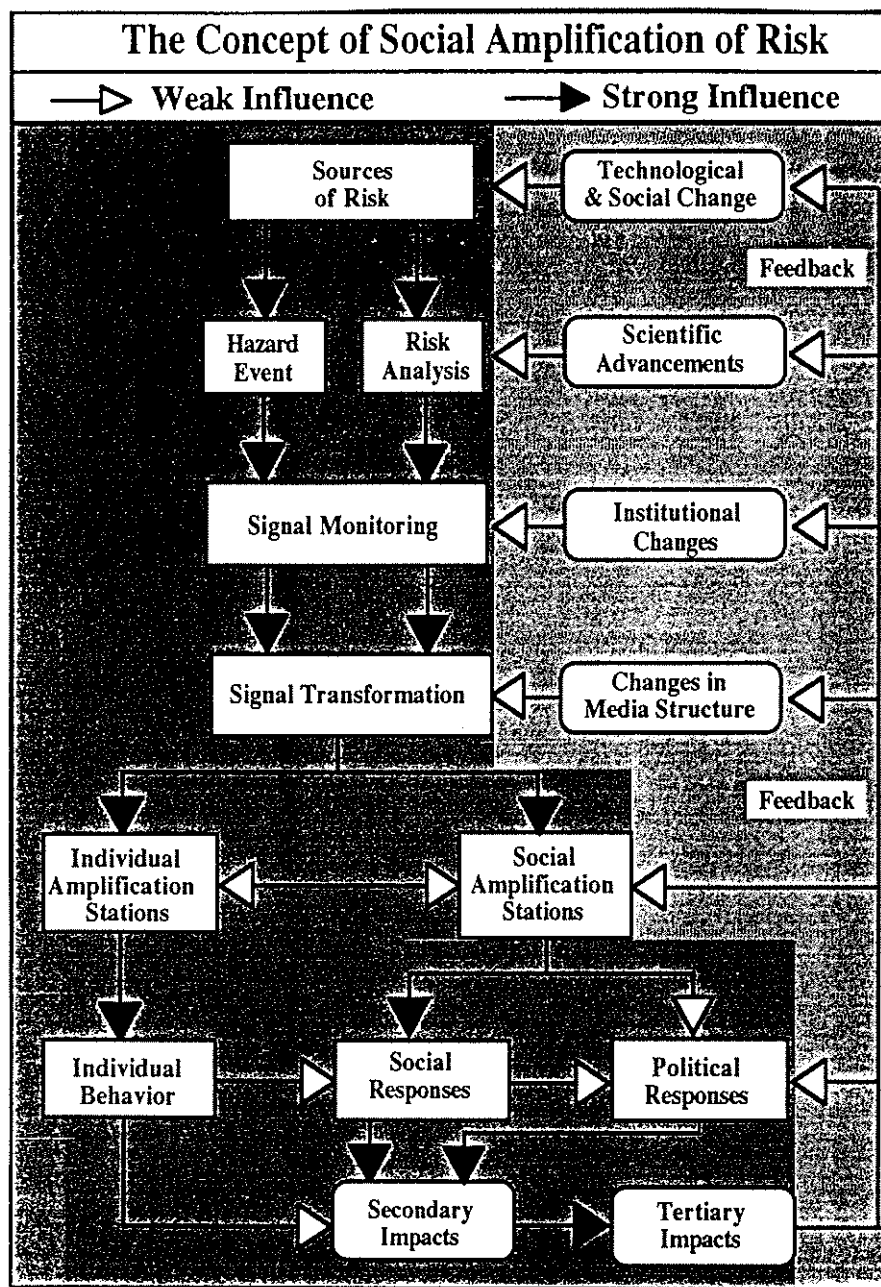
### 3.4 *Social Amplification of Risk*

In 1988, Kasperson and colleagues proposed a novel approach to study the social response to risk. The concept of *social amplification of risk* is based on the thesis that events pertaining to hazards interact with psychological, social, institutional, and cultural processes in ways that can heighten or attenuate individual and social perceptions of risk and shape risk behavior. Behavioral patterns, in turn, generate secondary social or economic consequences that extend far beyond direct harms to human health or the environment, including significant indirect impacts such as liability, insurance costs, loss of confidence in institutions, or alienation from community affairs (Kasperson *et al.* 1988).

Such secondary effects often trigger demands for additional institutional responses and protective actions, or, conversely (in the case of risk attenuation), place impediments in the path of needed protective actions. In accordance with the metaphor of amplification in the processing of electronic signals, amplification includes both intensifying and attenuating signals about risk. Thus, alleged overreactions of target audiences receive the same attention as alleged "down-playing".

Fig. 2. illustrates the process of amplification. The amplification process starts with either a physical event (such as an accident) or the recognition of an adverse effect (such as the discovery of the ozone hole). In both cases, individuals or groups will select specific characteristics of these events or aspects of the studies and interpret them according to their perceptions and mental schemes. These interpretations are formed into a message and communicated to other individuals and groups (Renn 1991). Individuals or groups collect and respond to information about risks and act in our terminology as "amplification stations" through behavioral responses or communication. Amplification stations can be individuals, groups, or institutions. Amplification differs among individuals in their roles as private citizens and in their roles as employees or members of social groups and public institutions.

The behavioral and communicative responses are likely to evoke secondary effects that extend beyond the people directly affected by the original hazard event. Secondary impacts are, in turn, perceived by social groups and individuals so that another stage of amplification may occur to produce third-order impacts. The impacts may spread or "ripple" to other parties, distant locations, or other risk arenas. Each order of impact will not only disseminate social and political impacts but may also trigger (in risk amplification) or hinder (in risk attenuation) positive changes for risk reduction.



**Figure 2:** *The basic concept of the social amplification of risk:* Signals from physical events are transformed in communication signals and forwarded to social and individual amplification stations. These stations process the information and respond to the messages by sending out new information or acting upon the content of the received message. Individuals may form or change their opinions on an issue or demand changes with regard to risk management practices. Social and political stations may use the incoming information to promote structural innovations that would help to cope with the respective hazard. All these social actions trigger secondary and tertiary impacts: Economic losses, institutional changes, social mobilization, and other consequences may occur.

Drawing upon this concept of social amplification of risk, Renn *et al.* (1992) investigated the functional relationships among five sets of variables that enter into the amplification process. The first class of variables included the physical consequences of 128 hazardous events (events that exposed humans or the environment to physical harm); the second class referred to the amount of press coverage about these 128 events; the third class entailed the individual layperson perceptions with respect to these events; the fourth class described the public responses (individual behavioral intentions and group mobilization potential) to these hazards; and the fifth class contained the socioeconomic and political impacts of these events as measured by documents and a Group Delphi with experts. The study investigated the structure of causal relationships among these variables classes.

The most interesting result of this study is the weak link between casualties and most of the other variables. The best physical risk predictor is exposure rather than any other indicator of harm. Exposure contributes to dread and is also highly correlated with media coverage. Its direct influence on intended action is small indicating that exposure operates through risk perception variables to influence personal actions. The link between exposure and societal impacts is not significant, however, despite the initially high correlation between the two variables (Renn *et al.* 1992). Exposure appears to shape societal experiences with risk through the media and through perceptions and intended individual actions. The data reflects the major assumption of the social amplification model. i.e., that physical events are observed and interpreted by groups and individuals, amplified through individual and social processors, and then expressed in terms of societal consequences.

The social amplification framework provides an integrative concept. The distinction between individual and social amplification stations corresponds with the two traditions in risk perception: the individual processing of information and the social responses to risk based on experience of (dis)trust, social values, and cultural affiliations. It provides a more holistic picture of the risk perception process and takes into account psychological, sociological, and cultural aspects.

#### **4. Lessons for Risk Management**

Should public perception of risk be the guiding principle for risk management? Should we allocate the budget for risk reduction according to the priorities recommended by the risk assessments of technical experts or by the risk perceptions of the lay public? Who should have the right to set standards that determine the division line between tolerable and intolerable risk levels?

All these questions have preoccupied the risk community for many years. A consensus among risk researchers or risk managers has not emerged since today. A major problem for both sides has been the philosophical question of whether technical risk estimates represent "objective" probabilities of harm or reflect only conventions of an elite group that may claim no more degree of validity or universalism than competing estimates of stakeholder groups or the lay public (Bradbury 1989: 383). This debate would have remained a mere side note in the history of risk management if the results of technical risk calculations and the intuitive risk perceptions of most people had been almost identical or similar. Our brief review demonstrated, however, that the perceived seriousness of risks has been almost diametrically opposed to the calculated risk numbers of the risk professionals (Covello 1983; Renn 1990). Juxtaposing professional estimates of risks and public perceptions of risk has been a popular activity among risk researchers ever since the EPA published its "Unfinished Business" report (US-Environmental Protection Agency 1987; 1990). Most of these studies confirm a clear discrepancy between the priority list of experts and the mean values of concerns among the general public. Obviously some risks are socially amplified in the public perception process, while others are attenuated (Kasperson *et al.* 1988; Renn *et al.* 1992). Risk management agencies are well advised to base their decisions on both, expert assessments and public concerns. How should risk managers integrate both inputs? What is the role of technical risk assessments and what is the role of public perception?

Technical analysis provides society with a narrow definition of undesirable effects and confines possibilities to numerical probabilities based on relative frequencies. The price society pays for this methodological rigor is the simplicity of an abstraction that we make from the culture and context of risk-taking behavior (Shrader-Frechette 1984: 281ff). The social experience of risk includes the perception of actual damage, but it is more focused on the evaluation of the risk context and the associations between the risk and social or cultural artifacts. Integrating all social science perspectives will be difficult as the assumptions behind each of the concept are incongruent with each other. Given the difficulties to capture and interpret public perceptions, would it be better to ignore them and rely only on technical expertise?

This would indeed be appropriate if society were only concerned about risk minimization. If all society would care about is to reduce the amount of physical harm done to its members, technical analyses and some form of economic balancing would suffice for effective risk management. Included could be the perspective of organizational sociology to make sure that technical safety measures are paralleled by institutional control and monitoring (Freudenburg 1989). The social sciences would only be needed to sell the risk management packages to the "mis-informed" public via risk communication.

However, society is not only concerned about risk minimization (Douglas and Wildavsky 1982; Thompson *et al.* 1990). People are willing to suffer harm if they feel it is justified or if it serves other goals. At the same time, they may reject even the slightest chance of being hurt if they feel the risk is imposed on them or violates their other attitudes and values (MacLean 1986; Linnerooth-Bayer and Fitzgerald 1996). Context matters. So does procedure of decision making independent of outcome. "Real" consequences are always mediated through social interpretation and linked with group values and interests. Responsive risk management needs to take these aspects into account. The social science perspectives on risk can help to enrich risk management.



How can this potential for enrichment be put into practice? Risk perception studies can help to create a more comprehensive set of decision options and to provide additional knowledge and normative criteria to evaluate them (Renn 1992a). Similar to the other perspectives, social science research can contribute valuable information for understanding risk responses and for designing risk policies, in particular:

- identify and explain public concerns associated with the risk source;
- explain the context of risk-taking situations;
- identify cultural meanings and associations linked with special risk arenas;
- help to articulate objectives of risk policies in addition to risk minimization, such as enhancing fairness and institutional trust and reducing inequities and vulnerability;
- design procedures or policies to incorporate these cultural values into the decision making process;
- design programs for participation and joint decision making;
- design programs for evaluating risk management performance and organizational structures for identifying, monitoring, and controlling risks.

Social science studies cannot resolve, however, conflicts about the acceptability of risk. They may assist risk managers to find the right procedures for conflict resolution or risk management planning. Such discursive procedures can be organized in form of advisory committees, citizen panels, formal hearings, and others. The necessity to base risk decisions on plural value discourse was highlighted in a recent report by the U.S. National Academy of Sciences (Stern and Fineberg 1996). The report emphasized an *analytic-deliberative process*, in which technical expertise and public value input should be integrated. Democratic values can provide the means by which to construct this dialogue and the social science perspectives can help to make these forms of dialogue work, i.e. to make sure that each group can bring their own interest and values to the process and yet reach a common understanding of the problem and the potential solutions (Fiorino 1989). Participation is not only a normative goal of democracy, it is also a requirement for rational decision making in situations in which risks need to be evaluated.

Risk management implies value judgments on three levels. The first set of value judgments refer to the list of criteria on which acceptability or tolerability should be judged, the second set of value judgments determine the trade-offs between criteria, and the third set of values should assist in finding resilient strategies for coping with remaining uncertainties. Using informed consent on all three value inputs does not place any doubt on the validity and necessity of applying the best of technical expertise for defining and calculating the performance of each option on each criterion. Public input is a crucial contribution for determining the objectives of risk policies and for weighing the various criteria that ought to be applied when evaluating different options.

## **5. Conclusions**

This list of individual and social factors that shape risk perception demonstrates that the intuitive understanding of risk is a multidimensional concept and cannot be reduced to the product of probabilities and consequences (Allen 1987). Although risk perceptions differ considerably among social and cultural groups, the multi-dimensionality of risk and the integration of beliefs related to risk, the cause of risk, and its circumstances into a consistent belief system appear to be common characteristics of public risk perception in almost all countries in which such studies have been performed. Furthermore, the experience of risk is

not limited to the threat of facing harm in the future. It includes subjective predictions of possible outcomes, the social and cultural context in which the risk is experienced, the mental images the risk situation evokes, the perception of the players who are involved in the risk situation and the judgments about fairness and equity related to the distribution of potential hazardous events. In this sense, risk is a social construct rather than a physical entity. However, this construct is not without physical foundation: people do face and fear disaster and natural degradation. Experiencing or expecting these physical changes are part of the social experience of risk, but it is not the only, in some instances not even the most important part of the social perception of risk.

Risk perception studies have revealed a multitude of elements that shape the individual and social experience of risk. First, individual and social risk experience appears to be stronger related to exposure than to actual casualties on which most risk assessment are based. Second, individual perception is highly influenced by qualitative risk characteristics and semantic images. Rather than evaluating risk with a single yardstick, most people use different mental tools when estimating and evaluating risk sources or activities. These tools are internalized through cultural and social learning. Third, the processing of risk by the media, social groups, institutions, and individuals shapes the societal experience with risk and plays a crucial role in determining the overall intensity and scope of societal impacts. The degree of perceived physical harm is hence only indirectly related to the individual and social response to risk; these responses are the result of intermediary processes such as amplification in the media, outrage by disadvantaged groups, and political maneuvering. In addition, the psychological mechanisms of information processing are influenced and affected by social value commitments; social, political, and cultural associations; trust in institutions as well as the struggle for social resources (Renn 1992b; Renn *et al.* 1992).

There is also no impartial referee available to judge the appropriateness of risk perceptions. Science may help to determine the magnitude of the risk but this information alone is not sufficient to make decisions about the acceptability of risks. The only viable resolution of these conflicts in democratic societies is by initiating a fair discourse among the major parties involved in the decision making process or affected by the decision outcomes (Habermas 1971; Renn *et al.* 1993). Risk communication and conflict resolution is therefore a crucial element of any risk management strategy. The goal of risk communication and conflict resolution should not be to persuade people to accept whatever the communicator thinks is best for them. The ideal communication program envisions a receiver who processes all the available information to form a well-balanced judgment in accordance with the factual evidence, the arguments of all sides, and his/her own interests and preferences. The ultimate goal of risk communication is to reconcile expertise, interests, and public preferences.

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**Communicating with Stakeholders and the Public at Large  
the Brent Spar Case**

**Margrethe M. Skov  
A/S Dansk Shell**



# What can be said can be said clearly

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## Abstract:

The biggest problem in nuclear technology is not the technology itself but the public acceptance. Most people cannot evaluate the radiation risk themselves, so they must trust experts. But, laymen do not understand the jargon of radiation experts, whose language is full of technical terms, unfamiliar units, abbreviations and acronyms etc.

After 26 years of experience in radiation protection and public information the author believes that the public has no possibility to leap over the broad communication gap. So, it is the experts who must do it. The author has discussed his communication strategy in his book *The Core and the Apple Peel* and implemented it in his book *Radiation in Everyday Language*. He stresses that *nothing has been taught if nothing has been learned* and claims that *everything that can be said can be said clearly*.

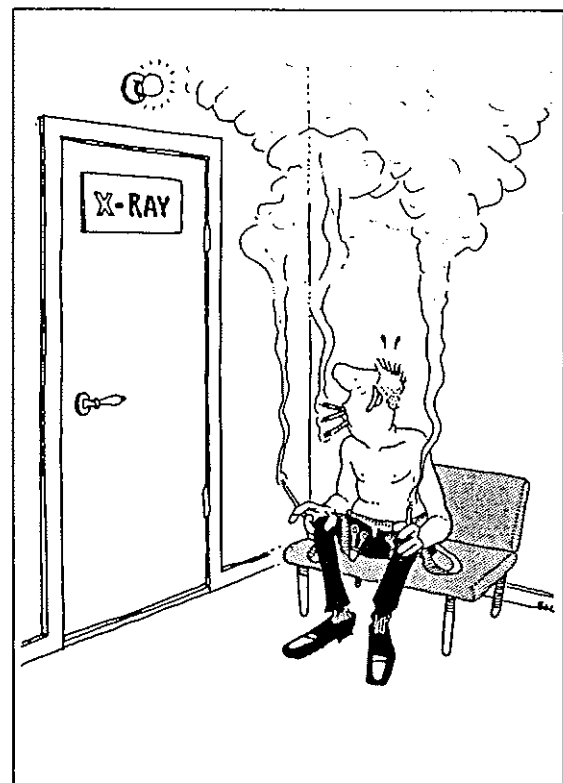
“Radiation is something that all people are afraid of, except for the experts.”

We have heard this aphorism before, and we know it is true. Actually this is how it must be.

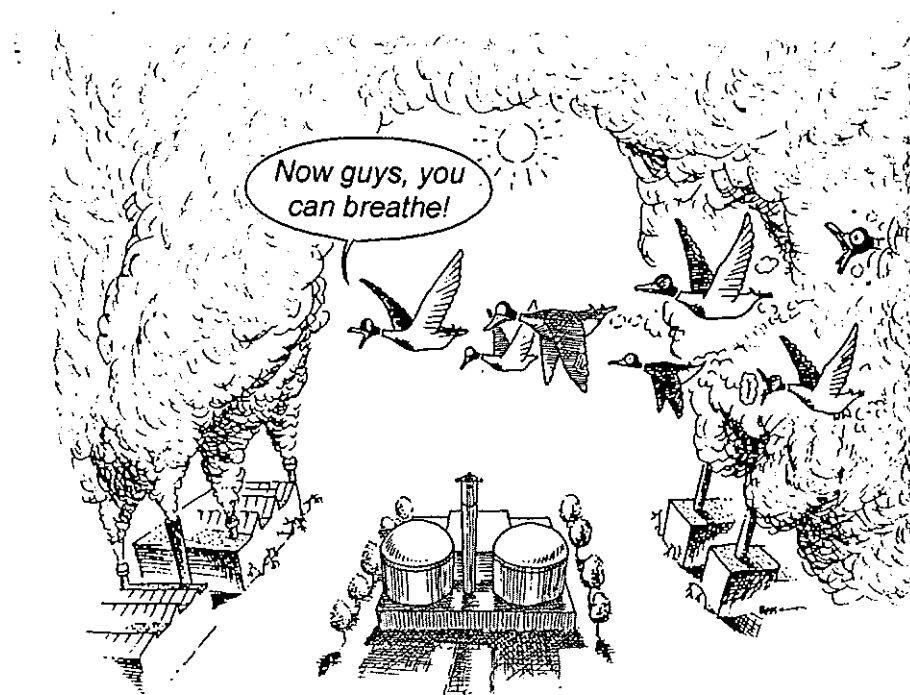
All people know that radiation can be harmful and dangerous to their health. They also know that radiation cannot be detected by human senses. In a radiation hazard or fall-out situation “ordinary people” cannot know if they are at risk or not. Laypeople cannot quantify the radiation risk.

As people know that radiation *can* be harmful, but they cannot know *when* it is, they must suppose radiation is *always* dangerous – so they are afraid of it.

Experts, on the other hand, can evaluate the risk in each situation. They know what is safe and what is not. They know which doses are acceptable and which should be avoided. Experts also trust other experts and they do understand the information they get from their colleges. So, instead of being afraid experts can act and react rationally in each situation. Experts feel safe.



But, most people are “ordinary” people and ordinary people are afraid of radiation. This is the biggest problem in the area of nuclear technology. The technology is not a problem itself. The problem is the public acceptance. All over the world we find millions and millions of people who, in spite of its pollution, prefer fossil power instead of nuclear power, who want to stop all safe handling of radwaste, who resist irradiation of food etc. And all this is only because they are afraid of radiation!



Isn't this frustrating! Experts know there is no reason being afraid of radiation. Experts know the risks and how to control them. But, our fellow-people, living in the same world - maybe living next door to us, are afraid.

I see the reason this way: Laypersons know, that radiation can be dangerous, even deadly, but they lack the possibility of making their own evaluations. So, they must trust the experts - but, they *do not understand* what experts say. Ordinary people a) don't have any chance of making their own observations or evaluations of radiation, and b) they do not understand what the experts say.

We have a very broad communication gap here, a gap that we, the experts, should do every effort to leap over.

## AN EXAMPLE OF MISUNDERSTANDING

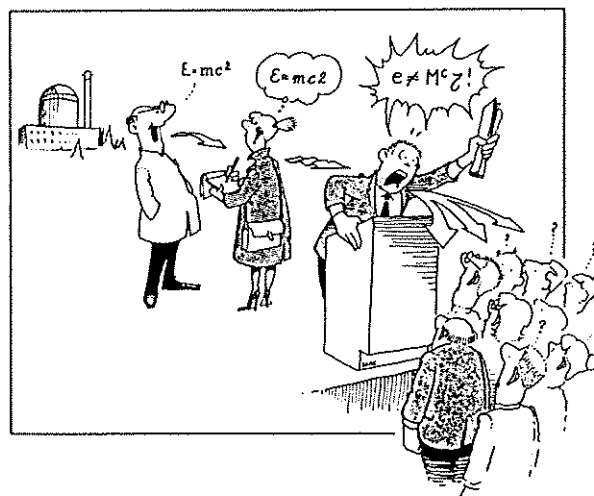
In one of my books on this subject I have illustrated the problem using a hypothetical example that could as well be authentic:

A local newspaper reporter interviews the manager of a nuclear power plant about the plant's radioactive releases into the sea. The reporter asks these things because people living nearby are worried. In fact, the plant has always performed very well, and no uncontrolled releases have ever occurred, so the expert feels very comfortable when giving his honest, pertinent and detailed answer:

Now, listen to what he says:

- “Annually about 50 000 cubic meters of purified, but still contaminated water is released from our nuclear power plant into the recipient. The concentration of radioactive corrosion products is, however, only a few hundred kilobecquerels per cubic meter. And, the concentration of fission products is two decades lower. Due to this the annual release of gamma and beta emitters is below 10 Giga-becquerels, that is 10 thousand million becquerels. The releases of alpha emitting transuranics are insignificant.”

The expert thought he had given a good answer. And his expert colleges would applaud him. We understood every word. But, the poor reporter didn't, because the expert used expressions, units and professional terms that the layman never learned in school nor later. Those are the words underlined. The expert could as well have given his answer in Greek or Swahili.



- *When talking to laymen, we must avoid words that were not familiar to us when we finished our school.*
- *We should not tell everything we know, just say what we mean.*

In this case the only notes that the reporter got down in his note book were the names of some substances that sounded dangerous, and some huge figures. Back in the news house, when reading his notes, his conclusion was that the people living nearby really had reason to worry. This was reflected in the headline on the front page of the local newspaper next day:

## **Large amounts of dangerous substances released into the sea**

If the expert had been able to put himself in the shoes of the reporter he could have said in everyday language:

- “When it comes to radioactivity, the water released into sea from our plant clearly meets the *purity standard for drinking water*, so there is really no reason for anyone to worry.”

From the facts given in the first answer we can easily verify that also this latter statement is true. Both answers are true. But, only the latter can be understood by a layperson.

Explaining to laypersons is very different from writing a doctor's thesis. When writing a thesis you must demonstrate how clever and educated you are. But, the public or the journalist already knows you are – that's why they ask you – so, now the *only important thing is that your message is understood*.

- *There has been no communication if the message did not get through.*
- *Nothing has been taught, if nothing has been learned.*

## **MEDIA RELATIONS**

When we talk to a member of the public we talk to *one* person. When talking to a journalist we talk to many thousand people. That's why our relations to media people are extremely important.

We should try to understand how media people work, and to serve them on their terms. A media representative cannot wait until tomorrow to get an answer. He must get it when he calls you. He must write his article now. What is big news in tomorrows paper is nothing one day later.

When a journalist turns to our opponents, our critics, he gets an answer immediately. It may not be the correct answer, but something anyhow. It's too bad if he doesn't have our view at the same time.

We can learn from how critics use media. They are eager to call, to write and to drop into a newsroom, to meet media people and to discuss with them. The opponents are ready to "wait in the rain" longer than experts. Their language is vivid and colourful and easy to understand for the media. As experts, again, we are often very careful about what we say. This is wise, of course, but it is definitely not what the media like or need. They may think that we turn and bend our words to hide something.

## **USE EVERYDAY LANGUAGE**

Experts in many areas make the life difficult for their fellow-people by using their own professional jargon. This is true for computer experts and lawyers, as well as for nuclear and radiation experts. Our jargon is full of expressions and units that are so usual and familiar to us, that we never even reflect about the fact, that laymen have no chance to get an idea of what we are talking about. Here are four categories:

- Acronyms & abbreviations:
  - LWR, ICRP, BEIR, N-16, ASME, TLD, NRC, ICRU, STUK, PWR, SSI . . .
- Unfamiliar units:
  - Mtoe, R, ppm, keW, gray, GWh, TBq, Bq/cm<sup>2</sup>, MWd/tU . . .
- Orders of magnitude, exponents:
  - Giga, Tera, nano, pico, 10<sup>-12</sup>, one per billion . . .
- Technical terms:
  - dose rate, reactivity, surface contamination, neutron flux density, burn-out, stretch-out, fall-out . . .

Just take such a usual word and magnitude as the *micro*. We talk about microSieverts and microGrays as easily as we talk about centimeters and kilograms. But the man in the street doesn't know the meaning of micro.

People have micro-chips in their calculator and Micro Soft in their micro computer. They have micro ovens in their kitchens to heat food with micro waves. But, most of our fellow people do not know that micro means  $10^{-6}$ . Please, reflect! Most of our fellow people do not know that micro means one millionth!

## OTHER EXAMPLES

Think about the following example:

Suppose we analyze for plutonium in the Atlantic Ocean and find 10 microBq of Pu-239 per liter sea water. To us this fact means, that there is only 1 decay of a Pu-239 nucleus in a liter of water every 28 hour. – And that is the same as “nothing”. But if we say to a layperson that there is 10 microBq plutonium per liter sea water close to a nuclear facility, he will think:

- *Oh, plutonium in the sea. That's terrible! That plant should be closed!*

The layperson understands “plutonium” but not “micro becquerel”. In this light, what do you think ordinary people know about nano and pico?

Again, going in the other direction, people understand kilo. They know what a kilogram weights and they know what a kilowatt-hour of electric energy costs. But, they have no idea of the Mega. People know that information technology is “mega-business” nowadays, but they do not know what a Megawatt or a Megabecquerel is. - And, do never mention anything about Giga or Tera to a layman!

After the Chernobyl accident the main newspaper of Finland, Helsingin Sanomat, showed a fall-out map of Finland with doserates in different regions filled in. Helsingin Sanomat is the newspaper, from which papers abroad pick out their Finland news.

Now, it happened so, that the dose rate figures reported in the newspaper were correct but - the unit was not. The dose rate unit should have been microRoentgen per hour. But, the editor did not see any difference between micro and milli, so the unit was said to be milliRoentgen per hour.

When these values - 1000 times too high - were published in USA, the most catastrophic consequence, from many Finns' point of view was, that the American swing quartet Manhattan Transfer, that was going to perform at the Pori Jazz Festival two months later, cancelled their participation . . .

## INSTEAD OF UNITS

In many cases we can do well by using the *per cent* instead of units. Everyone is familiar with the per cent. Everyone knows there is 11 per cent of alcohol in a good wine and 4 per cent in normal beer. We know how many per cent of our income we need to pay as taxes.

Now, if we find *12 Bq of cobalt-60 per kilogram in a fish sample*, and tell this to a journalist, the headlines in tomorrow's newspaper will be awful to read. Instead, we may say that “the activity of cobalt found in a fish sample was only *5 per cent of the natural potassium activity* in the fish”. That makes no headlines. The journalist understood the message: There was no danger. There was nothing to report about.



As often as possible, make *comparisons* to familiar things or normal situations. For instance, if a person is afraid of getting cancer because he lives close to a nuclear facility, he will not be helped a bit if you explain the cancer risk in the usual way:

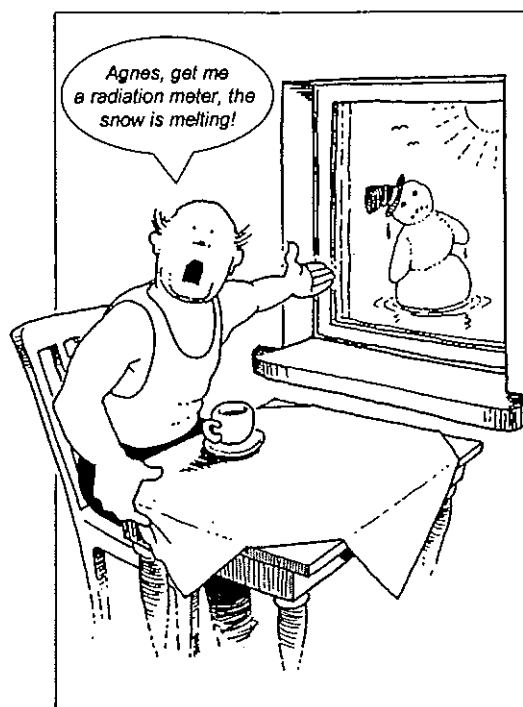
- "According to ICRP 30 the mortality risk of radiation induced cancer is only  $5 \times 10^{-2}$  after an exposure corresponding to a dose equivalent of one Sievert, while the maximum permissible radiation dose, caused by radioactive releases from the plant, to an individual of the critical group is 0,1 mSv per year."

Why not simply say:

- "The radioactive releases from that plant have less effect on radiation in the environment than have the changes in weather and seasons."

Nobody thinks he may get cancer because the snow melts or it starts raining...

But, it is true that these events cause higher increase in the outdoor radiation level than does the releases from most nuclear facilities. The dose rate of the background radiation may well increase 20 - 30 % when the snow melts.



## BE A GOOD LISTENER

We have got two ears and one mouth. We should use these organs in that proportion. To be a good communicator *one needs to be a good listener*. Never give your answer before you really understand what the person actually wants and needs to know.

- A difficult long question, that the person makes big effort to formulate, may need only a brief and short answer.
- A question that seems very simple – even naive – may be very serious to the person and based on a lot of thinking and wondering.

We must distinguish between questions of major importance, and those of exact scientific relevance. Public asks questions of the first category, and they should be answered in a simple way, leaving out details, exceptions and special cases. These only "contaminate" the message. I claim that *everything that can be said can be said clearly*. Everything that can be said can – *and must* – be said clearly.

A few years ago I wrote a book with the title *Radiation in Everyday Language* (see [www.kolumbus.fi/wahls](http://www.kolumbus.fi/wahls)). I thought this was what the public needed. By now I know it, because this book has been translated into eight languages and printed in USA and Europe as well as in Far East. Two new translations are in progress. I tell this only to

demonstrate what a huge need there is in the world for radiation information spelled out in a way that ordinary people can understand.

## BE PREPARED TO COMPARE!

When making comparisons to familiar concepts there is no need to compare normal radiation exposure to really dangerous activities - like smoking cigarettes or driving a car!

We are surrounded, all the time, by useful everyday radiation examples like

- the natural **activity** in our body, in a ton of soil, or in a cubic meter of sea water,
- the natural **radiation** in houses, on a hill, in a plane,
- the radiation **dose** one gets from an x-ray examination, from radon in homes, from a trans-Atlantic flight or when visiting a friend living in a high background area,

It may be a good idea to learn a few examples on beforehand. In my very newest book, *The Core and the Apple Peel* (see [www.kolumbus.fi/wahls](http://www.kolumbus.fi/wahls)), I have given some examples of how we could explain the risks, that the releases from a nuclear power plant causes in its environment, in a way that laymen can understand:

- **Releases of radioactive particles**

A nuclear power plant usually releases no more radioactive particles than a coal fired plant does. Most of the radioactive particles released from nuclear power plants are short-lived, but the radioactive particles from coal fired plants will remain radioactive for millions and billions of years.

- **Release of radioactive iodine**

The total amount of radioactive iodine released from a well performing nuclear power plant during one year, is usually smaller than the amount of radioactive iodine given to a patient for treatment of the thyroid.

- **Release of radioactive noble gases**

The natural concentration of radioactive noble gas (Radon) outdoors is about 10 becquerels per cubic meter of air. The normal releases from a nuclear power plant cause about 0,001 becquerel per cubic meter at the fence of the site, and much less further away.

- **Environmental dose from NPPs**

The extra annual radiation dose that people living close to a nuclear power plant receive corresponds to the dose one gets from natural background radiation in a few of hours. While visiting our friends in their home, we may receive an excess dose that is higher than what a nuclear power plant causes its neighbors during one year.

- **Plutonium**

The accusation that one microgram of plutonium kills a man doesn't mean what it sounds like. During the atmospheric nuclear weapon tests in the sixties five tons of plutonium were spread around the world. 5 tons is 5000 billion micrograms - that is one thousand times more micrograms than there are people in the world. Much of this fell to the ground, but people were not harmed by this.

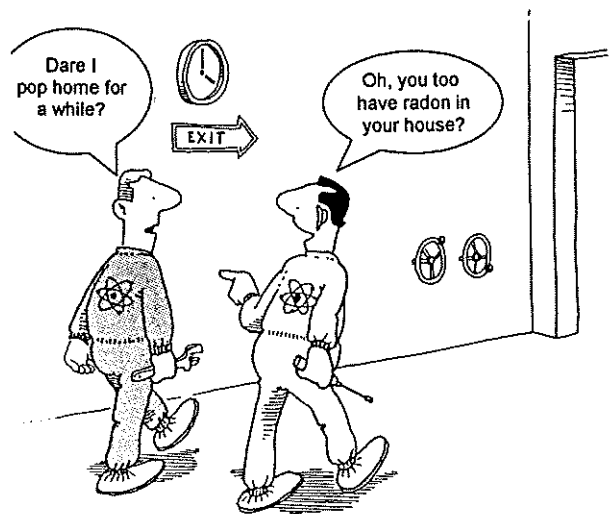
- **Natural versus artificial radiation**

It has been claimed that radiation from artificial, manmade, sources is more detrimental than natural radiation. This simply isn't true. Health effects from low rate radiation may occur only if a DNA molecule in a cell is damaged in a certain way. Of course, the damaged molecule cannot know the name or the origin of the substance that emitted the radiation quanta that hit.

I will not make the list of examples longer than so. These were only meant to give some idea of what kind of explanations ordinary people will understand. Many times it is enough for us, that our listener got the feeling that he understood. If our listener thinks: "*Oh yeah, now I see. I could understand that*", then it means that we did a good job.

If we discuss dose limits, occupational doses and **acute health effects** in terms of *mSv:s* and *Sv:s*, our listener will turn around and walk away. But he will stay and listen and feel he understands, if we compare the dose that gives you a 50-50 percent chance to survive or to die, with the Eiffel Tower. In this scale the annual dose limit to workers corresponds to the *height of a man* and the dose limits to public to the *thickness of a brick*.

We may also tell that the **average annual occupational dose** for exposed personnel in nuclear industry, medicine and science corresponds to what aircraft crews receive annually from natural cosmic radiation. This again, is much less than many people receive in their homes from natural radon gas.



Some people believe that **irradiated items** or food turn radioactive. Again, comparison to familiar situations is a useful tool. Everybody knows that an x-rayed patient will not turn radioactive. This is a convincing example!

There is one tool more that we may like to use. This tool is not too easy to handle, but if you succeed, the success is granted. This tool is the humor!

It is not forbidden to use humor to help your message get through. In *The Core and the Apple Peel* I have used cartoons as illustrations. Many magazines have reprinted them and some reviewers have said, that the cartoons make their point very clear. The illustrations in this paper are some samples from the book.

## **SOME CONCLUSIONS**

My conclusion, after 26 years of experience of public information is, that we need to step down from our ivory-towers, down to the grass-root level where the ordinary people stay, where our public lives, where massmedia operates. We need to "take off our white doctors' coats and show that we have blue jeans or colored skirts there under." Except for our special knowledge we, too, are "very ordinary" people.

- We must use two languages: One when we discuss with colleges, another when we communicate with the public. *If the public does not understand us, it is us to be blamed, not them.*
- We must concentrate on the message and how to get it though, how to get people understand what we really mean. *The important thing is not how something is expressed, but how it's comprehended.*
- We must avoid too much of data, figures, details and units when talking. *We have to sacrifice a little of the scientific exactness for the sake of clarity.*
- We must be emphatic! To be good communicators we need to be good listeners. We cannot give the right answer in the right way *until we fully understand what our listener actually wants or needs to know.*
- We must realize the importance of good media relations. We should try to *understand how media people work and to assist them on their terms.*

And finally: As we need to make some simplifications and generalizations we must always do this in an honest way. We may not simplify to change the truth or to avoid telling about problems, but only to make the idea clear.

When doing so our public will thank us, and – so I hope – our scientific colleges will forgive us.



## **SESSION II**

### **INFORMATION, EDUCATION, AND REGULATORY ASPECTS**

**Chairman: Sigurdur Emil Pálsson**



## Lunds universitets strålskyddsinformation på intranet

Christer Samuelsson  
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Universitetssjukhuset  
SE 221 85 Lund  
Sverige

### Abstract

Med ambitionen att göra lagstiftning, lokala regler och allmän information rörande arbete med joniserande strålning lätt och omedelbart tillgänglig för all personal vid universitetet, har relevant material överförs till länkad elektronisk form. Inslaget av lokal information, med bl.a. förteckningar över innehav av strålkällor hos enskilda institutioner, har gjort att vi för närvarande valt att begränsa tillgängligheten till datorer inom universitetes och landstingets domäner.

### Bakgrund

Lågdosexponering av personal för joniserande strålning är vanlig inom universitetsforskning och medför skapar i allt väsentligt ett informationsproblem, eftersom rena skyddsinsatser minskar i proportion. Den personal som arbetar med starkare strålkällor är oftast välinformerad om strålrisker, men behöver naturligtvis också tillgång till regler, strålningsdata etc. För att tillfredsställa detta informationsbehov har Lunds universitet fört över lokala strålskyddsregler, allmän information rörande arbete med joniserande strålning till länkad elektronisk form.

The screenshot shows a web browser window titled "sswebinfo - Microsoft Internet Explorer". The address bar shows "http://www.medfak.lu.se/stralskydd/FrameSet1.html". The page content is divided into two main sections:

**Strålsäkerhet vid LU**

**INNEHÅLL**

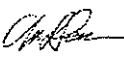
- [Introduktion](#)
- [Prefektansvar](#)
- [Avfall](#)
- [Facktermer](#)
- [Gränsvarde](#)
- [Personkontroll](#)
- [Organisation](#)
- [Röntgen](#)
- [Strålningsdata](#)
- [Strålningsrisker](#)
- [Tillstånd- och arbetsregler](#)
- [Tillståndsförteckning](#)
- [Transporter](#)
- [Länkar, kontakt, namn](#)

**Introduktion**

I större doser kan joniserande strålning bevisligen orsaka bl.a. cancer. För att skydda individ och miljö är därför all användning av joniserande strålning reglerad i lag. Arbete med joniserande strålning i forskning och undervisning kräver enligt strålskyddslagen särskilt tillstånd. Innehav av tillstånd för verksamhet med strålning är i sig ingen garanti för att strålsäkerheten är tillfredställande. Det krävs dessutom att tillämpliga arbetsregler och rutiner efterlevs, vilket naturligtvis underlättas om regelverket är snabbt och lätt åtkomligt. I detta syfte har jag i länkad form sammanställt strålningsdata, förordningar, tillståndsregler mm som gäller för arbete med strålning vid Lunds universitet.

Föreliggande information kommer att kontinuerligt uppdateras och en utvidgning till området fält och icke-joniserande strålning ingår i planerna. Jag hoppas också så småningom, till frömmen för alla gästforskare och -studenter, att kunna få tid och resurser för framställning av en engelskspråkig upplaga av informationen.

Med vänliga hälsningar



Christer Samuelsson, strålskyddsfysiker

Figure 1. Innehållsförteckning och introduktionssida



## Informationen

Informationen är huvudsakligen skriven i enkelt html-format och avsaknaden av avancerad grafik, gör att man även med enklare webbläsare/datorer kan snabbt bläddra bland sidorna. Om man använder en webbläsare med ramar, återfinns innehållsförteckningen i vänsterramen och externa länkar i överramen (Figur 1.) För närvarande består informationen av strax över hundra små länkade html-filer på totalt mindre än 600 kB. Hela materialet ryms på en diskett och kan utan olägenhet läggas in på små handdatorer för tillgänglighet i fält. Länksamlingen är upplagd på en server som administreras av medicinska fakulteten vid Lunds universitet och är tillgänglig för all personal som loggar in från landstinget, tekniska högskolan eller universitetet. Sajtens inslag av strålningsdata (se figur 2) är sådant att den också kan fungera som uppslagsverk för experten som konsulteras per telefon, någon sökfunktion ingår dock ej. Data som överförs till html-format täcker exempelvis attenueringskoefficienter, ALI-värden, toxicitetsklass, aktivitetesgränser för transportkollin och datablad över de vanligaste radionukliderna. Databladet har tagits fram av Bo-Anders Jönsson vid Radiofysik i Lund och täcker de radionuklider som är vanligt förekommande inom sjukvård och forskning (se Figur 3). På regelsidan återfinns givetvis de tillståndsanvisningar som gäller lokalt för universitetet,

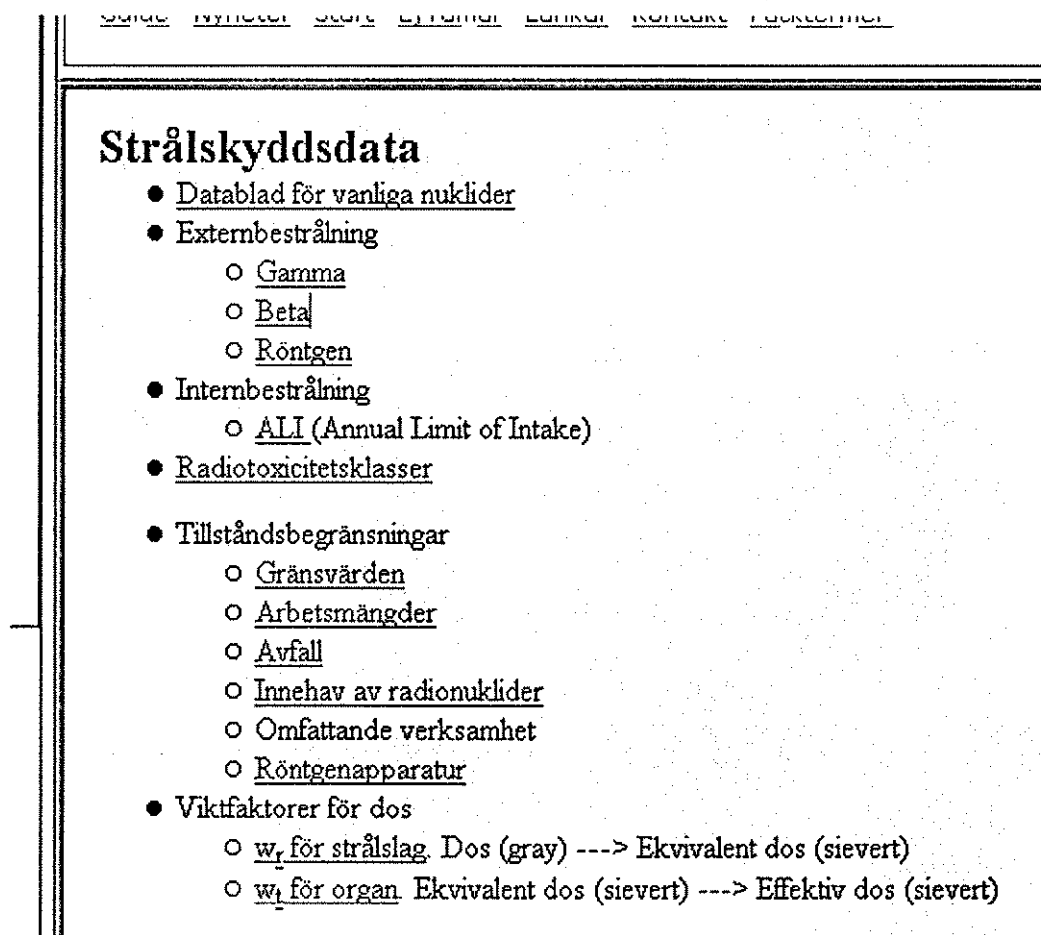


Figure 2. Innehållssidan för länk Strålskyddsdata.

men också generella anvisningar t.ex. för transporter och avfall.

För att göra informationen begriplig också för icke-specialisten, finns en utförlig lista över facktermer med tillhörande definitioner. Upplivningsdelen av webbplatsen diskuterar också grundläggande strålskyddspolicy och strålrisker. En kommenterad länklista till webplatser som innehåller strålskyddsinformation finns också med.

## Iodine-125      Jod-125      $^{125}\text{I}$

Z = 53    N = 72

Isotoper 55 < N < 88

Atomvikt = 126,90447

Skalelektroner = 2,8,18,18,7

©1992-1998. Bo-Anders Jönsson, Radiofysik, Lunds universitet.

Sönderfall:	Elektroninfångning till $^{125}\text{Te}$
Strålning:	Röntgen- och gammastrålning. Konversions- och Augerelektroner
Energi:	28 keV x-ray (140%), 35 keV $\gamma$ (7%)
Dosrat från 1 MBq punktkälla:	370 $\mu\text{Sv h}^{-1}$ på 1 cm avstånd
	0,037 $\mu\text{Sv h}^{-1}$ på 1 m avstånd
Fysikalisk halveringstid:	59,4 dagar
Biologisk halveringstid	80 dagar (sköldkörteln), 8 timmar (övriga organ)
Kritiskt organ:	Sköldkörteln (normalt upptag c:a 30-35%)
ATT	1 MBq

**Figure 3.** Del av databladet för  $^{125}\text{I}$ .

### Planer

Förnyelse och uppdatering av materialet är ständigt pågående, dock i oregelbunden takt pga av för små resurser. Hittills har inga särskilda resurser tilldelats informationsprojektet, utan allt arbete har skett inom ramen för den strålskyddsfysikertjänst som finns vid Lunds universitet. Framtida utvidgning till icke-joniserande strålning och en upplaga på engelska är påtänkt, men för närvarande utan tidplan. Inslaget av lokal information, med bl.a. förteckningar över innehav av strålkällor hos enskilda institutioner, har gjort att tillgängligheten till webbplatsen begränsats till datorer inom universitetes och landstingets domäner. Om önskemål uppkommer, kommer en uppdelning av websajtens informationen i en offentlig och en intern del att göras och då med den offentliga delen åtkomlig från internet.



## THE NORWEGIAN INFORMATION CAMPAIGN ON RADON

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### Abstract

The responsibility for providing an overview of *«all factors in the environment which are or may be having a direct or indirect influence on the health - - »* rests with the municipal health authorities. In order to enable the municipal staff throughout Norway to accomplish local radon surveys, an information campaign on radon, including printed information material and training courses, was carried out in 1998-99, primarily directed towards municipal civil servants.

The two-day training courses comprised of lectures and a compendium covering basic knowledge on ionizing radiation, sources of radon, measurement techniques, health risk, prophylactic and remedial measures, design and accomplishment of survey projects, and information strategy.

The printed information material includes booklets providing general information on radon (health risks, measurements, and mitigation), methods for measuring radon in indoor air and construction sites, action levels, and design of municipal radon surveys.

Two posters have been issued, one mainly intended for public offices and waiting rooms to motivate the public for radon measurements, the other one intended for municipal personnel and governmental offices, the latter also issued as a collection of fact sheets intended for schools etc.

The booklets are displayed on the Internet ([www.nrpa.no](http://www.nrpa.no)). The site also contains links to further information on mitigation techniques and economic support to remedial measures.

### The purpose

Norway, in similarity to Sweden and Finland, is among the countries in the world having the highest radon concentrations in indoor air. This situation is due to the composition and structure of bedrock and sedimentary deposits as well as traditional building methods and relatively high temperature indoors due to cold climate. The population average radon dose is about 2 mSv. Indoor radon thus contributes approximately half of the collective population dose from ionising radiation. By comparing results from municipal surveys conducted during the period 1992-98 with the nation-wide survey from 1987-89, it has been estimated that about 10% of the housing stock now exhibit season-adjusted average radon concentrations above the recommended action level of 200 Bq/m<sup>3</sup>. There are, however, great geographical variations connected to the problem, even at a very local level. In some areas, more than half of the residences lie above the action level, while in others, very few show elevated radon concentrations. It is not, therefore, possible to predict from a general and limited nation-wide survey where the radon-prone houses are most likely to be found.

In order to reduce radon exposure in the most rational way, it is important to identify the residences in which the radon concentration exceeds the recommended action level, and to contribute to the accomplishment of remedial actions in the local arena. The intentions formulated by superior authorities thus states that: *«Radon exposure in Norway should be surveyed in co-operation with the municipalities. The objective is that the major part of*

*residential buildings and water sources exceeding the action level should be identified in the course of 2001, so that remedial actions may be effectuated.*». Moreover, it is important to enable local authorities to supervise that builders comply with the building regulations demanding preventive actions against radon in the indoor environment.

According to Norwegian law, the municipal health authorities, being under the supervision of the Directorate of Health, and hence under the political superintendency of the Ministry of Social Affairs and Health, are responsible for providing an overview of « - *the standard of health in the municipality, and all factors in the environment which are or may be having a direct or indirect influence on the health, including biological, chemical, physical and social environmental factors.* » This rather comprehensive list of factors obviously includes radon. The municipal authorities are also responsible for public information on the same topics. This duty implies that all of the municipalities, as soon as possible, should carry out local surveys of the radon problem through measurements in a representative sample of the housing stock. The objective mentioned above will not become a reality without substantial effort within the individual municipality itself.

Therefore, on behalf of the Ministry of Social Affairs and Health, the Norwegian Radiation Protection Authority (NRPA) in 1998-99 launched an extensive information campaign on radon. The campaign was primarily directed towards municipal personnel who, according to their official duties, already are, or, in the near future, might be involved in official advisory services or executive procedures concerning protection against radon.

The most important parts of the campaign have been the accomplishment of training courses for municipal officials and development and issuing of printed information material such as posters and booklets.

### **Training courses**

Invitations to attend courses on radon management was emitted to both the Health Department and the Technical Department in all the 435 Norwegian municipalities both in 1998 and in 1999. As a consequence of the responses to these invitations two training courses were run for a total of 105 participants representing 77 different municipal and multi-municipal bodies. The purpose was to give a general overview over the radon problem, and to enable the participants to start up radon surveys in their home municipalities as well as providing general advice and guidance to how the radon problem might be managed by the individual house owner.

The courses were run during two days, comprising:

- basic knowledge on radioactivity and ionizing radiation,
- sources of radon in Norwegian dwellings,
- radon measurement techniques and elements of uncertainty,
- radon as a health risk, mode of operation, including the basis for risk estimates
- prophylactic and remedial measures,
- design and accomplishment of municipal indoor radon survey projects
- information and counselling strategy.

Teaching was performed in the form of lectures given by personnel from the NRPA and the Norwegian Building Research Institute. The lectures were supported by a comprehensive

compendium covering the same subjects in about 90 pages. Teamwork in smaller groups was also included. Participation was confirmed by a diploma. Similar courses have also been run for several years (1994-1997), and to this date 177 Norwegian municipalities and multi-municipal bodies have sent representatives to these courses, more than 300 participants in total, predominantly from the southern parts of Norway.

### **Printed information material.**

To assist the local officials in their work as far as public information on radon concerns, the NRPA has also issued different kinds of printed information material, comprising booklets (Strålevernhefte) mainly for the civil servants themselves, and posters directed towards the lay public.

The booklets cover:

- general information on radon (health risk, measurement, and mitigation, Strålevernhefte no. 9, from 1996),
- procedures for measuring radon in indoor air and building sites (Strålevernhefte no. 3, updated 1998),
- recommended action levels for radon in residential and working environment (Strålevernhefte no. 5, updated 1998),
- and design and accomplishment of municipal indoor radon survey projects (Strålevernhefte no. 17, new 1998).

All the booklets (nos. 3, 5, 9 and 17) are displayed on the Internet, to be accessed via the NRPA homepage ([www.nrpa.no](http://www.nrpa.no)), also containing links to further information on mitigation techniques and economic support to remedial measures. Another three booklets covering radon in household water, radon in kindergartens and schools, and radon in underground work-places are in preparation, due to be published in the course of 1999, summarising research reports on the same topics. These booklets will also be displayed on Internet.

Two posters have been produced and issued. One mini-poster was mainly designed for public offices and waiting-rooms, aimed at motivating the public to contact the municipal authorities or one of the dozen commercial radon measuring companies available, in order to have their homes measured. The second, more comprehensive, poster was primarily designed for the public servants themselves, but is also issued as a collection of fact sheets intended for schools etc., serving to some extent as an updated version of the already somewhat obsolete Strålevernhefte no. 9.

The four existing booklets and the mini-poster were distributed to both the technical staff and the health service offices of all Norwegian municipalities, as well as to relevant offices at the county level, about 1000 dispatches all together.

### **Information implementation - municipal radon surveys**

By August 1999, about 10 % of Norwegian municipalities have carried out radon surveys in accordance with NRPA's recommendations. One of the intentions with the information campaign was to persuade the municipalities to survey their domestic radon situation. Depending on the size of the municipality and the population density, it is generally recommended that between 2 and 10 % of the housing stock should be measured. In accordance with «Strålevernhefte no. 3» the measurements should be performed by etched

track detectors, only, and the integration time in the measurements should be at least two months, preferably in the winter time.

The aim of a general survey is the identification of particular radon prone areas, defined as areas where more than 10% of the housings show annual average (season-adjusted) radon concentrations above the action level of 200 Bq/m<sup>3</sup>. The extent of follow-up measurements, any problem-directed surveys, and public counselling will depend on the results from this first survey.

In this context, it is an important impetus that a governmental settlement for economic support to remedial measures in homes has been established, and will be put into force in the course of 1999. The support will cover 50% of the expenses up to a maximal contribution of 15 000 NOK. Support may be granted on the basis of estimated expenses to the owners of homes where the average season-adjusted radon concentration exceeds 400 Bq/m<sup>3</sup>. Measurement documentation, work specifications, and a binding tender should follow the application. The support will be paid after the remedial actions have been carried out, unconditionally whether the actions are successful or not. A research project, evaluating the efficiency of the remedial actions has been planned, however, which will be based on voluntary cost free post-hoc measurements and the data given with the applications. Recent calculations indicate that between 50 000 and 100 000 Norwegian homes should satisfy the criteria for such economic support.

In the course of the campaign, the general volume of radon measurements has increased. So far, however, no extensive municipal radon survey seem to have been launched as direct consequence of the campaign. However, although not part of the information campaign, but running in parallel to it, a county-wide survey has been conducted after a common initiative taken by the county medical officer and the county governor in Aust-Agder, starting up with a training course like the one described above. During 1998, and so far in 1999, 1909 homes have been measured, amounting to about 5% of the housing stock. All the 15 municipalities of the county have participated, about half through a survey of random samples, the others (but mostly to a lesser degree) through advertisement and voluntary enrolment. In the course of the progress of the project, the degree of participation also showed a clear domino effect as increasingly more municipalities joined.

### **Future challenges**

In order to fulfil the intentions formulated by superior authorities pertaining radon registration and mitigation, it is important to spread information to the public, thereby boosting the interest for radon measurements, and to stimulate to the performance of long lasting remedial actions. In addition to the governmental economic support to remedial measures, a strong incentive will be if the public administration of a region, for example a municipality or a whole county, undertakes the initiative to carry out a local survey, not only in known radon-prone regions. The progress of the Aust-Agder project has shown that co-operation between official bodies of different areas of responsibility may be productive, also indicating that continuous stimulation of inter-municipal or even inter-regional co-operation may be a useful way of achieving the results anticipated in the mentioned objective. In this context a thoroughly prepared information strategy towards officials as well as towards the public is important. It is a great advantage, therefore, if such projects have the opportunity to exploit an efficient training programme and information material like those produced as a part of the described information campaign.

# UNIVERSITY COURSES ON RADIATION PROTECTION IN ESTONIA

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**Abstract--**A brief overview is given on the educational courses containing topics in the field of radiation protection in the Tallinn Technical University and in the University of Tartu, Estonia. It follows from the analysis that at present there is no complete system for education or training of experts in the field. At the same time a significant deficit in specialists and experts is one of the major barriers in the development of an efficient radiation protection infrastructure in this country. A comparison of the course topics to the syllabus for the training of qualified experts recommended by EC and by IAEA demonstrates the items, which need a further development and an expanded coverage in the existing or in future courses. These items involve, e.g., operational radiation protection and its organisation, waste management, transport, quality assurance, etc. Upgrading of courses for the missing theoretical items is not difficult, but it is not sufficient. The largest void lies in the weakness / absence of an adequate basis for practical work or exercises for students. The examples of co-operation and help provided for the existing courses, especially by the Nordic countries, are encouraging.

After regaining independence Estonia has started a rather rapid establishment of her radiation protection infrastructure. As a legacy of the half-century foreign rule, almost everything in the field has to be started from the grass-root-level with much enthusiasm and limited funding, but without adequately trained local experts or relevant education/research facilities, etc. No Estonian terminology in radiation protection even existed. In the last few years certain positive changes in the situation have taken place. The basic legal document, Radiation Act, has been accepted by the Parliament (Riigikogu) in 1997. Together with about ten lower level regulations issued so far, this Act forms a legal basis for implementation of the concepts, principles and requirements found in the IAEA and EU basic safety standards. The Act also authorises a regulatory authority, the Estonian Radiation Protection Centre (formed in 1996), to enforce its provisions and requirements in the country. The basic Estonian terminology in radiation protection has been elaborated (Realo and Viik 1997).

In the establishment process a valuable help has been and is continuously given to Estonia by many international organisations (IAEA, OECD/NEA, etc.) and EC, as well as by authorities and institutions of the Nordic countries. The last statement is equally valid relative to education and training of local officials and specialists in the field of radiation protection. At the same time an existing significant deficit in local specialists/experts and a non-existing national program for their education and training may turn into a major barrier in the implementation and in the development of an efficient radiation protection system in the country. Hopefully the program for development of radiation protection (including also guidelines for education and training) will be elaborated and processed in the near future, in accordance with the corresponding provision of the Radiation Act. At present no university in Estonia prepares specialists in radiation protection. In an attempt to alleviate somehow the deficit, since 1992 a few courses containing topics in the field of radiation protection have been initiated in the curricula of a few existing specialities. The main purpose of these initiatives is to widen the scope of activities, i.e. to include radiation protection, where after passing specific training courses the graduates of these specialities may specialise.

Below a short overview is given on the courses presented for students at the Tallinn Technical University (TTU) and at the University of Tartu (UT). The Internet pages of both universities have been used as the main source of information presented in Table 1 (TTU 1999; UT 1999a, 1999b). Other Estonian universities have no such course in their curricula.



Table 1. Topics of the courses presented for students at the Tallinn Technical University (TTU 1999) and at the University of Tartu (UT 1999a, 1999b)

Title (lecturer), points/ frequency	Topics relevant to radiation protection (unofficial translation by the author)
<b><u>Tallinn Technical University</u></b>	
<i>Courses for students specialising in biomedical technology</i>	
<b>Special course on medical physics</b> ( <i>H.Hinrikus and K.Meigas</i> ) 5 p./ 2+2 h wk <sup>-1</sup>	Problems of radiology and medical physics; equipment, methods and quality improvement; radiation safety of patients and personnel; techniques of dose reduction
<b>Quality in radiology</b> ( <i>no data</i> ) 5 p./ 5 h wk <sup>-1</sup>	Ionising radiation and effects in living matter; radiation safety; equipment and methods in radiology and imaging; radiation dose and units, radiation risk; quality of x ray films; measuring equipment and exposure criteria; tests and phantoms; measurement standards; dose reduction methods, national and EU dose limits.
<b>Medical imaging systems</b> ( <i>no data</i> ) 3 p./ 2+2 h wk <sup>-1</sup>	x rays: scattering, absorption, generation, detection; filters and screens; nuclear radiation and detection; x ray and radionuclide diagnostics, imaging methods and image parameters; biological effects of ionising radiation
<b>Physical basis of imaging diagnostics</b> ( <i>no data</i> ) 5 p./ 5 h wk <sup>-1</sup>	x rays and their physiological effects; radiation safety of patients and personnel; dose and measurement; basics of x ray diagnostics, angiography and mammography; nuclear medicine, applications in imaging diagnostics
<b><u>University of Tartu</u></b>	
<b>(a) Courses for students of applied physics specialising in biomedical technology</b>	
<b>Radiations in medicine</b> ( <i>K.Kepler</i> ) 2 p./ 32 h	History; physics of ionising radiation: x rays, generation, beams; sources of radiation; interaction with matter; radioactivity; basics of dosimetry and radiation protection: quantities and units; dosimeters; external exposure protection, shielding; safety in medical applications; principles and legislation (IAEA, EU standards); introduction to radiobiology and radiation therapy: biological effects of radiation; radiation risks; dose in therapy
<b>Introduction to medical imaging</b> ( <i>S.Aid</i> ) 2 p./ 24+8 h	Medical imaging: x ray diagnostics and equipment; radiography and imaging, quality assurance, optimisation; fluoroscopy; mammography, angiography; dosimetry and safety; basics of CAT and image processing; radionuclide diagnostics; practical exercises
<b>Biomedical signals and methods</b> ( <i>J.Vedru and P. Alapuu</i> ) 4 p./ 6 h of total 64 h	Radio- and x ray therapy: safety requirements and techniques; effects of ionising radiation, tissue factors, NSD concept; radiation therapy planning, irradiation equipment and methods
<b>(b) Courses for students specialising in environmental physics</b>	
<b>Dosimetry in the environment and radiation protection I</b> ( <i>E.Realo</i> ) 3 p./ 36+12 h	Ionising radiation; interaction with matter; terms and units; natural exposure; antropogenic radionuclides; radiation sources; detectors; radon and measurement; methods of sampling and analysis; transport of radionuclides in the atmosphere; dose formation; biological effects; physical methods of radiation protection; fundamentals of radiation protection: principles, EU and IAEA BSS, limits; radioactivity in Estonia; practice in $\gamma$ spectroscopy
<b>Dosimetry in the environment and radiation protection II</b> ( <i>E.Realo</i> ) 4 p./ 48+16 h	Primary photon field and dose formation; compartmental modelling in radioecology; internal radiation protection: dosimetry of $\beta$ and $\gamma$ radiation, dose coefficients; protection optimisation; in-situ and laboratory $\gamma$ spectrometry; statistics and uncertainties in analysis; physics of radon in the environment and dose; neutrons and dose; basics of nuclear fission and nuclear safety; BSS and national legislation; practical quantitative $\gamma$ spectroscopy
<b>Radioecology and radiation protection</b> ( <i>M.Lust and E.Realo</i> ) 2 p./32 h	Natural and antropogenic sources of radiation; biological effects of radiation; basic of radiation protection; external and internal exposure; nuclear energy and nuclear safety; radioactive contamination; migration of radionuclides in the environment

As many curricula are under modernisation/upgrading, the author can not guarantee that all relevant courses are fully and comprehensively considered. It should be noted additionally that the course descriptions of UT in comparison to those of TTU are much more detailed and informative. The latter may cause a respective under- or overestimation of the topics.

TTU educates engineers specialising in biomedical technology. Their curriculum involves four courses, which contain several topics of radiation protection and safety. The Centre of Biomedical Technology of TTU is responsible for the courses. In the courses, a dominating consideration is given to physical, technical and measurement aspects of medical imaging methods and equipment, including those using ionising radiation. In addition, basics of radiation effects in biological matter, radiation safety of personnel and patients, dose limiting methods, etc., are involved. Although it is difficult to estimate the presented/ planned scope of this knowledge, all the courses together should give an educational basis suitable for further training of technological staff or technical supervisors for radiation protection.

In UT topics of radiation protection are included in the curricula of two specialities. The Department of Experimental Physics and Technology has a programme for education of applied physicists specialising in biomedical technology, while the Department of Environmental Physics educates environmental physicists. Both departments give relatively good education in basic physics, including basic atomic physics and nuclear physics, necessary for assimilating the courses in radiation protection.

At present three courses involving topics of radiation protection are available for the BSc physics students specialising in biomedical technology. One of them, "Radiation in medicine" by K. Kepler, might be considered as a basic introductory course to radiation protection, which covers broad scope in the field. The other two involve radiation safety topics important for specific technical and biomedical applications and equipment. One of the prepared courses plans also a considerable practice of students basing on the clinical facilities of UT.

In the Department of Environmental Physics, the first part of a course "Dosimetry in the environment and radiation protection" given by E. Realo presents the basics of radioecology and radiation protection and it belongs to the curriculum of the BSc students. The second part considers to a greater depth a few selected topics of both fields and it is given mainly to the MSc students. Both courses include some laboratory practice in gamma spectrometry using the equipment of the Institute of Physics. In addition, a short introductory course "Radioecology and radiation protection" is given in two versions: (a) as a basic curriculum course for the students of the Türi College, UT (M. Lust) and (b) for all students of UT interested in the subject (E. Realo).

A brief analysis of the presented courses was made to compare qualitatively their scope of coverage to the syllabus recommended by EC for the qualified experts in radiation protection (EC 1998) and to the one presented by IAEA (IAEA 1988). It should be stressed that the analysis is rather formal and superficial, because the author has no intention to compare the depth of presented knowledge, distribution features of lecture hours between various topics, etc., of the courses. The purpose is to find some general trends, missing and insufficiently covered items and the relation of theoretical vs practical education/training.

The following conclusions are drawn. Both scope and volume of the existing courses are insufficient for a full education of experts in radiation protection. At the present stage these courses may form only a background education for the further special training.

As a general approach, UT presents mostly theory-oriented courses with rather limited experiments and exercises, while the courses in TTU are planned to cover almost equally both theory and practice. At the same time UT offers a broader educational background that permits the student/graduate to grasp better the subjects needed for further training of a qualified expert or a health physicist. It is supported by the fact that a few UT physics graduates have successfully terminated their MSc studies in radiation protection in London

and Surrey (NB! with the help of SSI, Sweden!). This statement does not exclude an urgent need for the inclusion of standard laboratory and field experiments to the curriculum, i.e. for development of the necessary facilities and equipment. The lack of standard practical exercises seems to be the most serious void in the presented education and the most difficult one to overcome due to limited resources. In addition, practically no course covers such items, as operational radiation protection, organisation of radiation protection, specific safety features in industry, teaching and research, decommissioning, waste management, transport and quality assurance. Many of these items are important for Estonia at present and even more in the future. It would be advisable to incorporate the basics of these items into the existing or in future courses.

The author has a pleasure to point at the vital role of the international radiation protection community, especially of the specialists and institutions of the Nordic countries. Their contribution has been essential both in the establishment of the courses and in the development of the research groups, which form the needed scientific background for the courses. E.g., it is difficult to overestimate the significant help presented to UT by Geislavarnir Ríkisins, Studsvik AB, SSI, Lund University, STUK, University of Helsinki and others. Hopefully the prolific co-operation in the development of education and training in the field of radiation protection will continue in the future.

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# The System of Radiological Protection revisited: Are dose limits for the population really necessary?

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**Abstract.** The distinction between practices and interventions in the System of Radiation Protection has created a lot of confusion in the population and amongst decision-makers, especially with regards to the concepts of dose limits and intervention levels. The experience gained after the Chernobyl accident indicated that many actions taken led to an unnecessarily large expenditure of national resources, and many instances occurred of contradictory national responses. A major reason was the mixture of dose limits for the population, which apply only to exposures from practices, and intervention levels, which apply only to protective measures in de-facto exposure situations.

The existing System of Radiation Protection is revisited and it is suggested that the System can be revised with no dose limits for the public without causing a lower degree of protection of the population. With the widespread use of source-related dose constraints and practical restrictions on the sources of public exposure from practices, generally applicable dose limits are rarely limiting in any practical situation, even if dose constraints might, at least in principle, fail to take adequate account of the exposures from other practices. Constraints can be expressed as operational protection quantities, e.g. nuclide-specific release rates, dose rate at the fence of a facility or nuclide-specific surface contamination density in the environment.

A revised System of Radiation Protection without public dose limits would not cause any reduced protection of the public compared to the existing System, and it has a potential for removing much of the confusion with regards to application of intervention/action levels. It would also have the potential for improving public perception of radiation protection and radiation risks as well as for saving vast resources in intervention situations for better application in general health care of the public.

## 1 Introduction

Although the ICRP policy for radiation protection has evolved over the years, its main objective has remained basically unchanged namely to provide an appropriate standard of protection for man without unduly limiting the beneficial practices giving to radiation exposure (ICRP 60). The ICRP has indicated that its basic System of Radiological Protection is intended to prevent the occurrence of deterministic effects, by keeping doses below the relevant thresholds, and to ensure that all reasonable steps are taken to reduce the induction of stochastic effects.

## 2 The System of Radiological Protection

The System of Radiological Protection makes a distinction between source-related protection - which is concerned with the exposures of individuals resulting from a single source - and individual-related protection - which is concerned with the exposure of a single individual from many sources as illustrated in Figure 1.

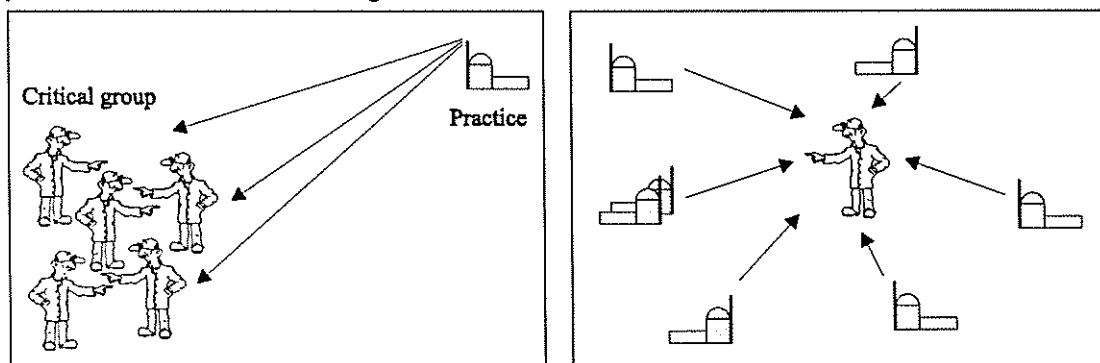


Figure 1. Illustration of the concepts of **source-related** protection of a representative member of a group of individuals exposed by a single practice (left picture) and **individual-related** protection of a single individual exposed to a number of practices (right picture).

Provided that the individual doses are well below the threshold for deterministic effects, the contribution to an individual dose from a single source has an effect that is independent of the doses from other sources. Each source, or group of sources, can then be treated on its own.

Source-related assessments make it possible to judge whether a practice or intervention is likely to bring benefits sufficient to outweigh any disadvantages and whether all reasonable steps have been taken to reduce the radiation exposures that a source will cause. They thus facilitate the justification of practices and interventions and the optimisation of protection at the source level. Source-related assessments take account of the magnitude (increase or decrease) of the doses attributable to the assessed source, and of the number of individuals exposed, but not of the influence on individuals of other exposure sources.

Practical application of the System of Protection, however, calls for a pragmatic combination of both source-related and individual-related assessments. To ensure compliance with individual annual dose limits, the System of Protection also requires individual-related assessments of the sum of annual doses from all relevant practices.

## 2.1 Practices and dose limitation

During its period of operation, a practice may add annual doses to the existing annual dose that people are incurring at the time of the introduction of the practice. The ICRP System of Protection for practices is concerned with the additional annual dose attributable to practice,  $+\Delta E$ , and not with the existing total annual dose - either that existing before the practice or that remaining after the practice as shown in Figure 2.

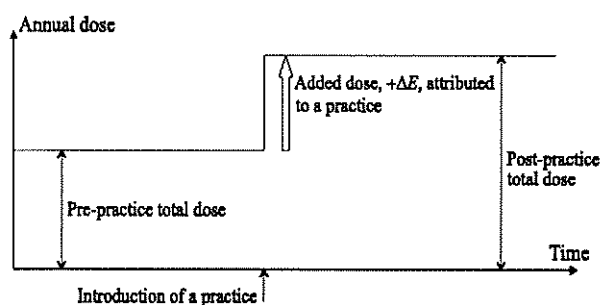


Figure 2. The introduction of a practice will **add** individual doses,  $+\Delta E$ , to the background dose (pre-practice total annual dose) existing before the practice is introduced. The System of Protection for practices is concerned **only** with limitation of the added annual doses,  $+\Delta E$ , attributed to practices and **not** with the pre- and post-practice total annual doses.

The individual dose to members of the critical group,  $+\Delta E$ , from a single practice should respect the dose constraint,  $f \cdot E_{\text{limit}}$  (see Figure 1, left-hand picture). Different national authorities have recommended values of  $f$  to be of the order of 0.1 - 0.3. In addition, the System of Protection requires that the sum of individual annual doses,  $+\Delta E_i$ , attributable to all relevant practices should not exceed individual annual dose limits:

<b>constraint</b>		<b>limit</b>	
$+\Delta E \leq f \cdot E_{\text{limit}}$	and	$\sum_{\text{all practices}} +\Delta E_i \leq E_{\text{limit}}$	

The existing total annual dose (either the pre-practice total annual dose or the post-practice total annual dose) is not subject to any dose restrictions other than the restrictions on those of its components which are attributable to the relevant contributing practices. In other words, the System of Protection for practices does not impose restrictions on the existing annual dose stripped of all the  $\Delta E$ s (see Figure 2).

## 2.2 Interventions and dose reduction

Interventions are intended to reduce existing doses by removing existing sources, modifying pathways or reducing the number of exposed individuals, thereby averting dose components of the existing individual doses. Once the intervention has been fully undertaken, the remaining individual doses are not subject to further consideration as illustrated in Figure 3.

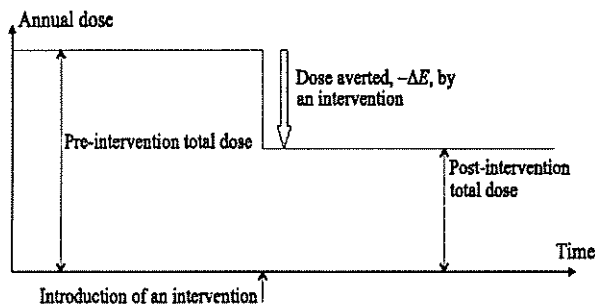


Figure 3. Undertaking an intervention will **re-**duce the existing individual doses at the time of intervention. The intervention is therefore expected to **avert** part of the pre-intervention existing individual doses. The averted dose is clearly **not** subject to annual dose limits or constraints. The residual post-intervention existing dose is also not subject to restrictions.

An intervention will usually be justified when the avertable dose,  $|\Delta E|$ , is greater than an optimised intervention level,  $IL$ :

$$|\Delta E| \geq IL$$

The use of dose limits as basis for intervention might involve measures that would be out of all proportion to the benefit obtained and would be in conflict with the principle of justification.

### 2.3 Distinguishing practices from interventions

There has been some uncertainty about the introduction and management of a practice in an area previously subject to intervention. It should be emphasised that individual doses from the residual exposure after an intervention has been fully withdrawn (or considered but not introduced) are not subject to restrictions and are out of the scope of the System of Protection. The post-intervention total dose is therefore the new baseline for considering any further human activities, including the introduction of a practice as illustrated in Figure 4.

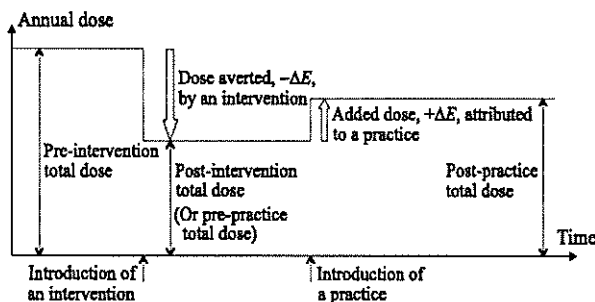


Figure 4. A practice is introduced after intervention were undertaken. The System of Protection restricts the **additional** annual doses attributable to the practice,  $+\Delta E$ , regardless of the post-intervention existing annual dose.

Usually there is no difficulty in distinguishing practices from interventions. However, there have been some misunderstandings over this distinction and some important issues should be emphasized:

- ▷ practices are adopted as a matter of a *planned choice* in order to gain some individual or societal benefit, in spite of the doses which a practice will add to existing annual doses.
- ▷ an intervention is intended to *reduce* existing individual doses caused by a de facto situation whose existence is not a matter of choice; in an intervention situation, the source (and/or the dose) *already exists* at the time when the situation is being considered.
- ▷ the clearest distinction between practices and interventions is the ability to choose a priori whether to accept beneficial sources and the consequent exposures; if a choice is still available, the exposure can usually be said to be due to a practice and subsequent steps to reduce doses are *improvements* in the practice and not an intervention; if there is no choice, because the sources already exist, any action taken to reduce exposures is an intervention.

When introducing the concepts of practice and intervention, the ICRP did not imply that any human activity that might cause increases in an individual's exposure is a practice, nor that any human activity that might reduce an individual's exposure is an intervention. For instance, normal modifications of living habits which may increase or reduce the individuals' background exposure (for example, a move to another part of the country or a change in the type of home) should not be treated either as a practice or as an intervention and should not be subject to the System of Protection.

### 3 Selection of a dose limit for public exposure

Because there is a risk of stochastic effects below the threshold for deterministic effects, the selection of a dose limit is only partially a scientific decision. It is mainly a value judgement being based not only on scientific information but also on knowledge of the level of risk that is usually considered unacceptable under normal conditions. According to the ICRP at least two approaches are possible in choosing a dose limit for public exposure (ICRP 60):

- ▷ judging at which level the radiation risk can reasonably be described as unacceptable
- ▷ judging the value of a dose limit by comparison with the variations in the existing level of annual dose from natural sources

The annual dose from natural sources may not be harmless, but it can hardly be called unacceptable as it makes only a small contribution to the health detriment which society experiences. The variation in the annual doses from natural sources around the world is of the order of 10 mSv/a, although much higher variations can be found.

The attributable lifetime probability of cancer death from an annual exposure of 10 mSv from birth over lifetime would be around 4%, applying the relative risk model and a dose and dose rate correction factor (DDREF) of 2. This probability corresponds to an average annual risk of about  $5 \cdot 10^{-4}$  with an age variation as shown in Figure 5.

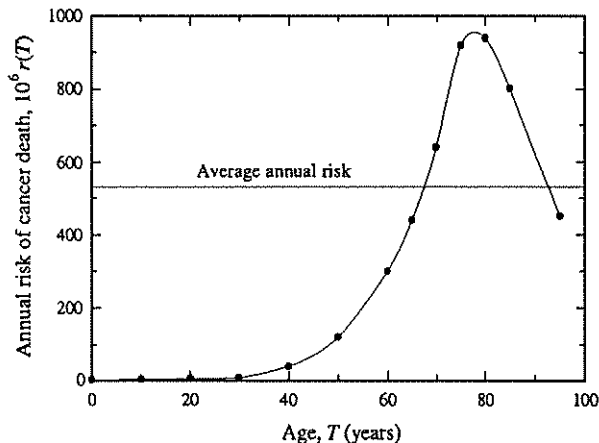


Figure 5. Variation of annual conditional cancer death probability (per million) with age for a standard population being exposed to an additional annual dose of 10 mSv per year from birth over lifetime, assuming a relative risk projection model and a DDREF of 2. The ICRP has stated that the consequences in terms of risk of continued exposure giving annual effective doses in the range from 1 mSv to 5 mSv do suggest a value of the annual dose limit for public exposure not much above 1 mSv.

ICRP has recommended an annual dose limit on effective dose of 1 mSv. In special circumstances, a higher value of effective dose can be allowed in a single year, provided that the average dose over 5 years does not exceed 1 mSv per year.

### 4 Discussion and conclusions

Public information on radiological protection has always been difficult due to the complex issues of radiation physics and radiation biology. Although the basic radiation protection principles and dose quantities have been developed further during the last two decades public information on radiation protection in normal and accidental situations has become even more difficult. At present, the system of radiation protection has reached a such high level of sophistication that even many radiation protection practitioners have difficulties in explaining all the details.

One of the basic concepts of radiation protection is dose limitation. However, public exposure can not be controlled by individual dose measurements, and the question is if dose limits for public exposure are needed at all. It is suggested - for at least the following reasons - that the question could be answered by a 'no':

- ▷ public exposure cannot be controlled by individual dose measurements, only by control of the individual sources contributing to the exposure of the public
- ▷ dose limits are supposed to ensure that the sum of doses to individuals from all practices should be kept below the limits, but no control scheme has yet been implemented to control this sum

- ▷ the potential for mixing public dose limits for practices and intervention levels for accidental and other de-facto exposure situations has created enormous problems, *e.g.* after the Chernobyl accident and in situations involving naturally occurring radiation exposure
- ▷ even if constraints do not preclude exposure from other practices, source-related constraints would ensure an adequate protection of individual members of the public, as long as the number of practices exposing the same critical group is limited

It has been suggested to the IAEA being the relevant body for the practical implementation of the radiation protection principles that IAEA should prepare a unified approach of radiation protection for practices and interventions in which dose limits for members of the public no longer exist but are replaced by constraints for source-related and individual-related protection<sup>1</sup>.

New ideas and guidelines are in fact being developed by the IAEA and ICRP on principles for protection of the public in extant chronic exposure situations. Both sets of guidelines include an individual-related reference level of 10 mSv in a year expressed as a **total** annual dose incurred by an individual. This reference level has been proposed to be used as a level of no concern, *i.e.* if the residual annual individual doses after the System of Protection has been applied are less than the reference level, further remedial measures are rarely required.

These ideas could be used also within the System of Protection for Practices. **Source-related** dose constraints could be derived from the **individual-related** reference level,  $E_{\text{ref}}$ , of 10 mSv/a, *e.g.* as a certain fraction of the reference level. In addition, if a very unlikely situation could arise where several practices might expose individual members of the same critical group, arrangements should be made so the total annual doses attributable to all practices and other sources - as far as possible - would be kept below the reference level,  $E_{\text{ref}}$ .

The ideas presented would constitute a major step forward towards a more comprehensible System of Protection that would not cause any reduced protection of the public compared to the existing System. Instead a revised System would probably remove a great deal of the misunderstanding which still exists today, especially on the principles for radiation protection against extant sources of both artificial and natural origin. A revised System would also remove the misuse of existing dose limits for the public in such situations.

When the ideas have been fully developed they would form a useful input to the decision-making process for introducing/terminating remedial measures in areas contaminated with long-lived radionuclides around the world, such as the Bikini Island and the territories in the former USSR contaminated by the Chernobyl accident. In addition, a revised System of Protection along the lines discussed here would have the advantage of being easier to explain to the public which probably would have a beneficial impact on public perception of radiation protection matters. Finally, a revised System would have a potential for saving vast resources in intervention situations for better application in general health care of the public.

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<sup>1</sup>Hedemann Jensen, P., *Radiological Protection Principles for Situations of Chronic Exposure to Residual Radioactive Materials*. International Conference on Topical Issues in Nuclear, Radiation and Radioactive Waste Safety, Vienna, 31 August - 4 September 1998.





## Effects of Low-Dose Ionising Radiation

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### ABSTRACT

Recent years there have been a number of discussions about the magnitude of the health risks resulting from the exposure to low levels of ionising radiation mainly within the American Health Physics Society, the French Academy of Sciences and the Canadian Radiation Protection Association. Essentially, the debate has centred on the question of linearity, i.e., whether to assume that the probability of radiation-induced cancer at low doses is proportional to the radiation dose received, without any threshold below which there is no risk, or to assume that a threshold does exist. In view of the current status of knowledge and of the established ethical precautionary principle, the use of the LNT assumption and the current "ICRP system of protection" is justified according to the author's view *for radiation protection purposes*. It has also a good acceptance among the health physicists all over the world. However, this approach to limiting the radiation risk should be used with great care. The collective dose should not be used to predict future detriment in the form of mortality numbers at very low doses say below a few millisieverts to large segments of populations.

Recently, there have been a number of discussions about the magnitude of the health risks resulting from the exposure to low levels of ionizing radiation mainly within the American Health Physics Society, the French Academy of Sciences and the Canadian Radiation Protection Association. Essentially, the debate has centered on the question of linearity, i.e. whether to assume that the probability of radiation-induced cancer at low doses is proportional to the radiation dose received, without any threshold below which there is no risk, or to assume that a threshold does exist. The International Atomic Energy Agency (IAEA) has in its Nuclear Safety Review 1996 provided a survey of present knowledge in the field (Ref.1).

Radiation protection standards assume that radiation doses over natural background doses cause additional health risks, notably an increase in the induction of cancers, that is proportional to the additional doses. The biological foundation for the standards includes the results of epidemiological investigations and fundamental studies on the cellular and molecular mechanisms involved in radiation damage and response. Furthermore, the results of studies with experimental animals provide further guidance.

Taking radiation such as X rays, gamma radiation and beta particles with a low energy transfer (LET radiation), epidemiological studies have, for some time, provided a substantial amount of direct, quantitative information on radiation risk. The main source of data is the Life Span Study of the survivors of the nuclear explosions at Hiroshima and Nagasaki in Japan 1945. These populations show a pattern of increasing mortality with an increasing dose for leukemia and most solid cancers, with a significant increase in the risk of cancer fatalities following acute doses in the range 200-500 mSv. With respect to cancer incidence, more recent data indicate a significantly increased risk at doses down to between 50 and 100 mSv (Ref. 2).

Information on cancer risks is also available from a number of epidemiological studies of patients irradiated for medical reasons. Many patients have received high doses to particular

organs. Results from pooling several studies have suggested a significant increase in the risk of thyroid cancer at doses down to 100 mSv, received in childhood.

A number of studies provide information on the risk of childhood cancer following exposure of the mother's abdomen during pregnancy. Detection of an elevated cancer risk after irradiation in utero is helped by the low background cancer rates that normally exist among children. These studies suggest that irradiation in utero increases the cancer risk. The Oxford Study of Childhood Cancer showed a 40 per cent increase in childhood cancer rate in children up to 15 years of age following in utero radiation doses within the range of about 10-20 mSv (Ref.3).

Direct information on the effects of low dose chronic irradiation is becoming available from studies of radiation workers. Some of these studies provide indications of excess cancer risks, notably for leukemia. Although the data are not strong enough to allow for quantitative risk estimates, the findings are consistent with ICRP risk estimates in Publication 60 and the assumption of a cancer risk even at low doses (Ref.4). However, below 10 mSv, it is not expected that epidemiological studies will alter the shape of the dose-effect curve for stochastic effects.

Experimental studies on animals cannot be used to obtain quantitative estimates of cancer risk for application to human populations because of the differences in sensitivity between species. They can, however, be used for examining the form of dose-response relationships and biological and physical factors that influence the radiation response. In a number of studies, the lowest acute dose to have a significant effect on the tumor yield falls within the range of about 100-200 mSv. This is similar to that found in studies on adult human populations. The lowest dose to result in a significant increase in risk following chronic irradiation is generally higher than that for acute exposure because of the reduced effectiveness of low dose rate radiation in inducing cancers. Animal studies, therefore, provide a broad support for the results of human epidemiological studies of the cancer risk of radiation at low to intermediate doses.

Studies at molecular, cellular, tissue and whole-animal level have made substantial contributions to our understanding of the radiation risk. In particular, they have demonstrated that the radiation damage increases with the dose and that, at least for low-LET radiation, it is often greater at high dose rates than at low dose rates, per unit of exposure. A Dose and Dose Rate Effectiveness Factor (DDREF) is commonly used to allow for the reduced effectiveness of radiation in inducing cancer in man at both low doses and low dose rates. However, only limited data are available on the effects of dose rate on the induction of tumors in human populations.

The available experimental animal data and limited human information have led ICRP to use a DDREF of 2 (Ref.4).

The developing understanding of the process by which damage to DNA may cause cancer has increasingly influenced the understanding of epidemiological and experimental studies. Neoplasia in tissues is now seen as a complex multistage process that may be subdivided into four phases: neoplastic initiation, promotion, conversion and progression. Although these are

simplifications of the overall process, they do provide a framework for interpreting the changes involved at the biochemical and cellular levels.

Neoplastic initiation encompasses the irreversible cellular damage, which provides the potential in cells for neoplastic development. There is good evidence that this initiation process results from damage to DNA leading to gene or chromosomal mutations in single cells in tissues. The critical event in relation to ionizing radiation is likely to be DNA double-strand breaks for which error-free repair is not likely at any dose.

Once the necessary gene mutation is present in a cell, further neoplastic development is believed to be highly dependent upon the cellular environment. Promotional events, influenced by growth factors in cells, dietary constituents, hormones, or other environment agents, may increase cell proliferation and may, in some instances, interfere with communication processes between cells that act to maintain cellular stability in tissues.

Conversion of these pre-neoplastic cells to a form in which they are committed to be malignant is believed to be driven by further gene mutations.

Progression of the disease, once the potential for a malignancy has been established, may depend upon further cellular changes that allow for the invasion of adjacent normal tissues, the circulation of neoplastic cells in the blood and lymphatic systems and the establishment of metastases at other sites in the body.

Radiation-induced mutations may influence all stages of the neoplastic process. Consequently, at the level of DNA damage, there is no basis for assuming that there is a dose threshold below which the risk of tumour induction is zero. For radiation protection purposes, it is appropriate to assume a progressive increase in risk with an increasing dose, with no threshold (the LNT hypothesis).

However, there is also some experimental evidence that low dose radiation may induce or activate cellular DNA repair functions, the so-called adaptive response. This effect is believed to be the result of the activation of signaling pathways. The majority of effects seen to date have essentially been short-term and act to modify the response to radiation rather than eliminate it (Ref. 5).

Although the ICRP does not employ the term precautionary principle, it does use the concept, at least implicitly (Ref. 6). In fact, the whole philosophy of protection against stochastic effects is not based on proven harm at low doses from radiation, because cancer and hereditary diseases from radiation have not yet been demonstrated conclusively, either in humans or animals at doses below 10 mSv. Experimental, ethical and practical reasons have led the ICRP and some other international agencies (FAO, ILO, OECD/NEA, PAHO, and WHO) to adopt the LNT hypothesis for low dose ionizing radiation.

The experimental molecular biology studies are broadly consistent with the thesis that, at low doses and low dose rates, the cancer risk of ionizing radiation increases as a simple function of dose and does not have a threshold-like component. Taken together with the epidemiological

information, there is no basis for arguing that low radiation doses below about 10 mSv would have no associated cancer risk at all.

In view of the current status of knowledge and of the established ethical precautionary principle, I believe that the use of the LNT assumption and the current "ICRP system of protection" is justified for radiation protection purposes. It is also largely accepted by health physicists over the world.

However, this approach to limiting the radiation risk should be used with great care. The collective dose should not be used to predict future detriment in the form of mortality numbers at very low doses, say below a few millisieverts to large segments of populations. In all cases, experts should use the best scientific information available concerning a given exposure situation. They may choose not to use the LNT assumption in their assessment. I refer, for example, to the Auger electron emitter isotopes bound to a chemical compound entering the DNA of cells and to radon in homes and workplaces.

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## **SESSION III**

- a) RADIOACTIVE WASTE**
- b) EMERGENCY PREPAREDNESS**

**Chairman: Karin Brodén**



# Omhändertagande av skrotade interndelar vid OKG

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## Bakgrund

1998 genomfördes på Oskarshamn 1, ett omfattande arbete med att byta ut interna delar, så som moderator-tank med lock samt ångseparator, i reaktor-tanken. Bytet var föranlett av att sprickor i svetsskarvar upptäckts på dessa. Enligt gällande koncept för mellanlagring av denna typ av avfall gäller att dessa ska sönderdelas i små segment, ca 400 x 400 mm, för att passa i skrotkassetter för att sedan transporteras till CLAB (Centralt Lager för Använt Bränsle) för mellanlagring i förvaringsbassänger. Till följd av den stora omfattningen av delar som skulle sönderdelas blev den planerade tiden för sönderdelning och transport så utdragen, ca 90 dygn, att alternativa lösningar började sökas. Resultatet blev att de interna delarna sönderdelades för att passa i avfallsbehållare tillverkade i stål med betongtanksformat, med måtten L=3,3, B=1,3 och H=2,3 m. Betongtankar är ett avfallsemballage som rutinemässigt används för omhändertagande av avfall i form av pulverformig jonbytar massa.

## Genomförande

De interna delarna sönderdelades till mindre delar under vatten med en kapmetod, som benämns Abrasive Water Jet (AWJ). Kapmetoden innebär att ett högt vattentryck (3800 bar) byggs upp med hjälp av högtryckspumpar.

Vattenstrålen passerar genom ett munstycke och fokuseras till en stråle på 0,07 – 1,2 mm. För att öka kaphastigheten tillsätts ett skärmaterial, abrasiv (stålflisor).

Delarna var kraftigt aktiverade och krävde omfattande skärmning vid hantering och transport. De placerades därför i speciellt framtagna stålbehållare med tre olika godstjocklekar 50, 100 och 150 mm beroende på de urkapade delarnas aktivitetsinnehåll. Stålbehållarna tillslöts med lock och transporterades ut från blocket till ett mellanlager på kraftverksområdet i väntan på slutdeponering i det planerade Slutlagret För Långlivat avfall (SFL). Eftersom interndelarna var kraftigt aktiverade laddades transportbehållarna med hjälp av en kassett som sänktes ner i bassängen och som när den lyftes upp strålskärmdes med hjälp av en transporthuv. Maxdosraten på moderator-tanken var 49 Gy/h.

Totalvikten för de skrotade komponenterna uppgick till 48 ton. Detta resulterade i totalt 26 avfallsbehållare med interna delar samt 23 kokiller med sekundäravfall i huvudsak abrasiv från kapningen. Eftersom avfallsbehållarna hade samma mått, som de avfallsbehållare som används i rutinemässig avfallshantering, kunde ordinarie transportsystem användas.

Behållarna lyftes ner från reaktorhallen med reaktorhallstraversen och därefter transporterades de med ett dynaliftfordon till BFA för nuklid-specifik aktivitetsmätning samt mellanlagring.



## Positiva erfarenheter

- Ingen spridning av radioaktivitet varken på ytor eller i luft under kapningsprocessen.
- Den framtagna lyft- och hanteringsutrustningen för laddning av behållare fungerade enligt framtaget koncept.
- Transportsystemet fungerade som avsett.
- Dosraterna på behållarna var låga och gav
  - mycket begränsad påverkan på omgivningen under transport till BFA
  - liten dos till transportpersonalen
  - inga problem vid uppställning i BFA.

## Negativa erfarenheter

- Reningssystemets kapacitet visade sig vara för dålig under kapning. Uppsamling av abrasiv och rening av bassängen fungerade inte enligt framtaget koncept. Detta ledde till att tidsplanen överskreds, 63 dygn istället för planerade 17 dygn.
- Återställning av arbetena efter kapning var inte planerad i detalj, varför detta tog längre tid.
- Service av kaputrustningen var mera omfattande än planerat. Åtgången av filterpatroner för rening av kapbassängen blev mer än dubbelt så stor som planerat. Detta ledde till en försening gentemot tidsplan med ca 10 dygn.
- Mängden förbrukat skärmaterial fördubblades p g a moderator-tanklockets konstruktion, vilket ledde till att kapsnitten blev mera omfattande än vad som planerats.
- Eftersom mängden skärmaterial och därmed antalet filterpatroner ökade blev avfallsmängden större än planerat.

## Slutsatser

Den använda metoden visade sig fungera på ett utmärkt sätt för att omhänderta skrotade interndelar, som klassas som SFL-avfall, till en avsevärt lägre kostnad än mellanlagring i CLAB. Hade interndelarna istället placerats i skrotkassetter för mellanlagring i CLAB skulle det krävs ca 45 kassetter. Detta innebär att OKG beräknas ha sparat ca 150 miljoner kronor jämfört med en alternativ förvaring i skrotkassetter i CLABs förvaringsbassänger.

Eftersom endast 10-20% av den ursprungliga radioaktiviteten, som har strål-skyddsmässig betydelse, återstår när de interna reaktordelarna ska deponeras i SFR är det inte nödvändigt att mellanlagra denna typ av avfall i CLAB. Metoden kan med fördel användas i fortsättningen för att omhänderta t ex hårdgaller, hårdstrilrör, provstavskanaler och moderatortanklock, som förvaras i interndelsbassängerna på reaktorblocken.

Erfarenheterna från omhändertagandet av interndelarna kan även användas i samband med framtida rivning av kärnkraftverk.

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# STATISTICAL DATA EVALUATION IN MOBILE GAMMA SPECTROMETRY

## *An optimisation of on-line search strategies in the scenario of lost point sources*

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### Abstract

There is a potential risk that hazardous radioactive sources could enter the environment, *e.g.* via satellite debris, smuggled radioactive goods or lost metal scrap. From a radiation protection point of view there is a need for rapid and reliable methods for locating and identifying sources. Car-borne and air-borne detector systems are suitable for the task.

The condition in this work is a situation where the missing radionuclide is known, which is not an unlikely scenario. The possibility that the source is located near a road can be high, and thus motivating a car-borne spectrometer system. The main object is to optimise on-line statistical methods in order to achieve a high probability for locating point sources, or hot spots, and still have reasonably few false alarms from variations in the natural background radiation. Data were obtained from a car-borne 3 litres NaI(Tl) detector and two point sources, located at various distances from the road. The nuclides used were  $^{137}\text{Cs}$  and  $^{131}\text{I}$ . Spectra were measured stationary on the road. From these measurements spectra we have reconstructed spectra applicable to different speed and sampling times; the time 3 seconds and 50 km/h are used in this work. The maximum distance a source can be located from the road and still be detected is estimated with four different statistical analysis methods. This distance is called the detection distance, *DD*. The method is applied on gross counts in the full energy peak window. For each method alarm thresholds has been calculated from background data obtained in Scania (Skåne), in the south of Sweden.

The results show a 30-50% difference in *DD*'s. With this semi-theoretical approach, the two sources could be detected from 250 m ( $^{137}\text{Cs}$ , 6GBq) and 200 m ( $^{131}\text{I}$ , 4GBq).

### Introduction

In the last decade the risk for radioactive sources adrift has increased. In Eastern Europe at least two severe accidents have occurred with stolen or dumped radioactive sources (Kivisääk 1995, Valentin 1998). Another potential risk for incidents with radioactive sources is the vast amount of transportations with radioactivity. From a radiation protection point of view there is an interest in improving the emergency preparedness against lost point sources.

Incidents have occurred where mobile gamma spectrometry has been used in the recovery of radioactive material and localisation of hot spots, *e.g.* the lost US Athena missile (Deal et al 1972), the Cosmos 954 incident (Bristow 1978) and the Goiânia accident (Moreira, 1991). An exercise, RESUMÉ 95, took place in Finland in 1995 where location of hidden sources was part of the agenda (NKS EKO-3, 1997). Apart from this exercise little has been done to systematically investigate search strategies and analysis methods for finding lost sources using mobile gamma spectrometry.

The scope in this work is narrowed down to a situation where the missing radionuclide is known. Furthermore it is not unlikely that the source is located near a road, which motivates the choice of using car-borne systems. A car also has other preferable features, compared to the frequently used air-borne techniques, *e.g.* lower costs, better spatial resolution and can easily be performed in the dark. NaI-detector based gamma spectrometry systems are commonly used when searching for point sources. The advantages, compared to semiconductor systems, are that NaI-systems are robust, have high sensitivity and are comparatively cheap; the main drawback is the inferior energy resolution. When searching for a known radionuclide the focus can be made on one gamma energy, or a few, and thus justifying the use of a low-resolution detector system.

The chosen approach to the problem is deciding if the content in a spectral window is caused by background radiation or not. In mobile gamma spectrometry the situation is often that there is rarely beforehand knowledge of the background in the area to be scanned. To avoid the need for a system calibrated for natural radionuclides the approach in this work is to evaluate the gross counts in the spectra. The only calibration needed to perform in advance is then an energy calibration. Another important feature when searching for point sources is that almost every spectrum collected contains pure natural background. The approach is then to make use of this continuing flow of consecutive spectra and apply on-line statistical analysis methods.

The main purpose of this paper is to evaluate and optimise different simple on-line statistical analysis methods for point source location, using a car-borne NaI-detector system.

### Theory

When processing data from NaI based measurements the most common technique is to evaluate the counts in a spectral window. All methods tested in this work are based on the gross counts in the  $^{137}\text{Cs}$  window (500-800 keV) and the  $^{131}\text{I}$  window (250-500 keV). Four different statistical methods are tested against the same data set to determine if a pre-set alarm threshold is exceeded. The data was obtained using two different sources; measurements were performed while they were placed at various distances from a road. In order to determine the alarm thresholds, car-borne measurements on natural background have been made. For each combination of method and source a detection distance,  $DD$ , is calculated. The maximum distance a source can be located from the road and still be detected with when driving by defines  $DD$ . For each method a test statistic,  $T$ , is calculated and compared to what would be expected in areas of pure background radiation.  $DD$ 's were calculated for several selections of input parameters. Only the best choices, resulting in the highest  $DD$ 's, are reported.

**Fix Background, (FB).** A very simple technique for analysing the data is chosen as the first method. It compares the gross counts,  $N$ , in a spectrometric window for every spectra against a pre-estimated fix background radiation level. The test statistic is calculated by

$$T_{FB} = \frac{N_X}{N_{X,fix}}$$

**Moving Background, (MB).** The average of  $N$  for the five latest spectra and the 20 spectra before is compared; the latter group is considering being natural background. The ratio is then calculated and compared to an in advanced estimated alarm threshold. The test statistic is

$$T_{MB} = \frac{\overline{N}_{X,1.5}}{\overline{N}_{X,6.25}}$$

**Mann-Whitney U test, (MWU).** MWU is a powerful nonparametric method. It tests if two groups of measurements are drawn from populations with equal distributions (Siegel 1956). Here the method tests the six latest collected spectra against the 18 spectra before. To apply the test, the 24 window counts are ranked. A test statistic,  $U$ , is then calculated by

$$U = \min [6*18+6*7/2-R_6, 6*18+18*19/2-R_{18}]$$

where  $R_6$  and  $R_{18}$  are the sum of the ranks for the group of six and 18 spectra. The MWU is a sensitive method. Applied on the data containing pure background,  $U$  reaches it's minimum ( $U=0$ ) too frequently. To avoid to many false alarms the test statistic,  $T_{MWU}$ , is set to

$$T_{MWU} = \sum_1^4 U_i$$

**Kolmogorov-Smirnov, (KS)** This is also an nonparametric method. If two groups of data have equal distributions the cumulative distributions are also equal, showing only random deviations. The KS test focuses on the "distance" between the cumulative distributions and determines the largest deviation at any point. The KS test is applied on the 20 (10+10) latest consecutive spectra. Cumulative step functions,  $S_i(N)$ , are determined for the two groups.

$$S_i(N) = K/i$$

where  $i$  is the number of samples and  $K$  is the number of scores  $\leq N$ . Let  $S_A(N)$  and  $S_B(N)$  be the step functions, where the latter one is the background group. The test statistic,  $T_{KS}$ , is then:

$$T_{KS} = \max [S_A(N) - S_B(N)]$$

### Material and equipment

The spectrometric system used was the GDM40RPS by GammaData. It is a portable PC-based system with GPS. It includes a special written software, NUGGET, for display and analysis (Mellander, 1998). In this work a three litres NaI(Tl) detector and 256-channel spectra were used. Two sources were used in the work:  $^{137}\text{Cs}$  (6 GBq) and  $^{131}\text{I}$  (4.5 GBq).

The detector was mounted in a GMC van. It was placed high in the car (ca. 2m) to minimise the influence of irregularities in the ground surface. The measurements were performed at a flat field near Revingehed, 15 km east of Lund, Sweden. The ordinary procedure to collect spectra in mobile gamma spectrometry is while driving the vehicle. At Revingehed a different approach was made. Stationary 60s measurements were performed 25m apart on the road. The gross window count rates were then interpolated to every five metres. These data were then used to estimate the window counts for arbitrary speed, sampling time and position of the car. The reason for this approach was to be free to choose the parameters afterwards.

### Results

The key to detect a source is to decide if an increase in the window counts is caused by natural variations or not. An alarm level for the different test statistics has to be determined. Ca. 5000 car-borne spectra (3s sampling time) have been collected in the south of Sweden in order to estimate alarm levels. The rate of occurring false alarms was estimated for different values of  $T$ . When calculating the detection distance the accepted rate were set to as low as possible, *i.e.* preferably no alarms at all on the background measurements. The minimum false alarm rate for the  $U$ -test is one per hour and for the KS method more then two per hour. One per hour might be acceptable but not two per hour, so the KS method was excluded.

**Table 1.** Alarm levels

<i>Method</i>	<i>Alarm level</i>	
	$^{137}\text{Cs}$	$^{131}\text{I}$
Fix Background	$T_{FB} \geq 1,92$	$T_{FB} \geq 2,00$
Moving Background	$T_{MB} \geq 1,43$	$T_{MB} \geq 1,59$
Mann-Whitney $U$ test	$T_{MWU} = 0$	$T_{MWU} \leq 4$

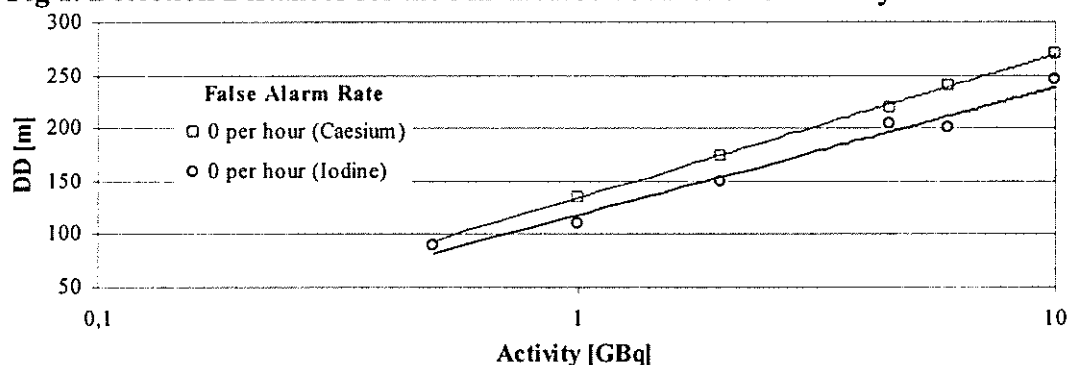
Now when the method strategies and their corresponding alarm levels are established the detection distances can be calculated. Two speeds, 9 and 15 m/s (*i.e.* 32 and 54 km/h) were used and a 3s sampling time. The three analysis methods were then applied on the reconstructed data series. The distance between the source and the point on the road where the car was furthest away from the source when the alarm level was exceeded determine the detection distance. Resulting  $DD$ 's are summarised in the table below.

**Table 2.** *DD's* for with an false alarm rate of 0 per hour, \*1 per hour

<i>Method</i>	<sup>137</sup> Cs (6 GBq)		<sup>131</sup> I (4,5 GBq)	
	32km/h	54 km/h	32 km/h	54 km/h
Fix Background	170 m	175 m	140 m	150 m
Moving Background	210 m	240 m	160 m	205 m
Mann-Whitney <i>U</i> test*	>200 m	<100 m	>150 m	>150 m

Data from the measurements with the two sources were used to estimate window counts for source activities between 0.5 and 10 GBq. The detection distances were then calculated in the same way as above. It was shown that the most effective method was *Moving Background* with 54 km/h. The resulting *DD's* for this method is plotted below against source activity.

**Fig 1.** Detection Distances for the MB method as function of activity for Cs and I.



### Conclusions

The choice of methodology when analysing spectra is important when searching for point sources. The method *Moving Background* has generally the highest detection distance and *Fix Background* the lowest. In mean  $DD_{MB}$  is about 30% higher than  $DD_{FB}$ . With the evaluated analysing methods, detector system and geometry in combination with a 3s sampling time the two sources could be detected from 250 m (<sup>137</sup>Cs, 6GBq) and 200 m (<sup>131</sup>I, 4GBq).

### Acknowledgement

This project has been supported by grants from the Swedish Radiation Protection Institute, SSI, (project number 710/2846/98).

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## WHO:s nya råd om jodprofylax vid kärnkatastrofer

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### Abstrakt

WHO har sammanställt nya råd om användning av stabilt jod som skydd mot utsläpp av radioaktivt jod vid kärnkatastrofer. Skriften har genomgått omfattande internationell evaluering i samråd med experter från IAEA. Den väntas bli utgiven detta år och kommer att läggas ut på Internet på WHO:s hemsidor ([www.who.ch](http://www.who.ch)).

Erfarenheterna från Tjernobyl visar att risken för sköldkörtelcancer efter exponering för radiojod i barndomen är betydande. En ökad incidens har iakttagits flera hundra km från katastrofområdet. Risken för allvarliga biverkningar av stabilt jod som engångsdos har konstaterats vara minimal. Stabilt jod är en säker, effektiv motåtgärd för att skydda sköldkörteln mot radioaktivt jod.

Det är motiverat att tillämpa olika kriterier för jodprofylax åt nyfödda, barn, unga vuxna och vuxna över 40 års ålder. För barn under 18 år borde jodprofylax övervägas redan vid 10 mGy sköldkörteldos. För unga vuxna rekommenderas en åtgärdsnivå på 100 mGy. För vuxna över 40 år är cancerrisken av radiojod mycket låg och jodprofylax är onödig, om den förväntade dosen inte överstiger 5 Gy, varvid exponeringen kan hota sköldkörtelns funktion.

Stabilt jod rekommenderas som engångsdos för att skydda mot radioaktivt jod i andningsluften. Som skydd mot radiojod i föda rekommenderas i första hand att intaget av kontaminerad föda (främst mjölk) begränsas. Om tillräcklig distribution av okontaminerad mjölk inte är möjlig, kan som alternativ åtgärd övervägas att ge dagliga doser stabilt jod till barn.

Den nya informationen om risk och nytta bör beaktas i planer för distribution och lagring av stabilt jod. Lagring är motiverad på mycket större avstånd från potentiella olycksställen än de som normalt ingår i zoner för beredskapsplanering. Lagringen bör vara så omfattande, att snabb distribution vid behov kan genomföras. WHO rekommenderar också att allmänheten ges möjlighet att fritt köpa jodtabletter på apotek.

### Bakgrund

Med anledning av olyckan i Tjernobyl utgav WHO:s Europa-avdelning år 1989 praktiska råd (*guidelines*) för jodprofylax vid kärnkatastrofer. Dessa råd baserade sig på ett WHO/CEC expertmöte i Bryssel föregående år. Mötesföredragen, vilka utgör den vetenskapliga basen för råden, gavs också ut i bokform (1).

Vid ett nytt expertmöte, sammankallat av WHO i Rom 1995, konstaterades att det föreligger behov av att revidera råden med anledning av ny information om den höga risken för sköldkörtelcancer hos små barn och den minimala risken för bieffekter av stabilt jod. Sköldkörteldoser till barn måste hållas så låga som det är rimligt att uppnå. Denna information bör beaktas, då råd ges angående lagring och distribution av stabilt jod. Emellertid förblir de tidigare råden giltiga i sina huvuddrag.



De nya råden har evaluerats av utomstående experter, som utsetts gemensamt av WHO och IAEA samt av tyreoida-läkarnas regionala förbund. Också två experter från IAEA har varit involverade. Det sista expertmötet år 1998 nådde samförstånd om innehållet. Dock har IAEA senare valt att avstå från gemensam publicering.

## Ändamål

Rådens ändamål är att sammanfatta dagens kunskap om nyttan och riskerna med stabilt jod för att blockera upptag av radiojod i sköldkörteln i fall av utsläpp vid en kärnkatastrof, ge information om rätt dosering och kontraindikationer till olika befolkningsgrupper, underlätta planeringen av jodprofylax vid en kärnkatastrof samt ge råd om praktiska aspekter för lagring och distribution av stabilt jod. Råden är avsedda bl.a. för beredskapsplanerare, befolkningsskyddspersonal, folkhälsomyndigheter och läkare.

## Innehåll

Dokumentet behandlar potentiell exponering för radiojod vid en kärnkatastrof, exponeringsvägar samt deterministiska och stokastiska effekter. Det noteras att deterministiska effekter förutsätter en sköldkörteldos på flera Gy, vilket kan förekomma genom inandning endast i närzonen. I denna zon är tidig evakuering eller skydd inomhus de primära skyddsåtgärderna, och jodtabletter kan vara ett viktigt komplement.

Risken för sköldkörtelcancer begränsar sig däremot inte till närzonen utan kan vara av betydelse t.o.m. hundratals kilometer från olyckan. Cancerincidensen hos barn runt Tjernobyl har visat sig vara förhöjd upp till 500 km från katastrofplatsen. Ingen zon för beredskapsplanering runt en reaktor sträcker sig så långt, och knappast något ställe i Europa ligger så långt från alla större reaktorer, att cancerrisken för barn saknar betydelse i händelse av en katastrof med stora jodutsläpp. Det går inte att fastställa ett avstånd, bortom vilket behovet av jodprofylax på förhand är uteslutet.

Dokumentet behandlar cancerrisken från radiojod i detalj. Risken för sköldkörtelcancer av yttre strålning är välkänd, och man vet att känsligheten är hög hos små barn, lägre hos äldre barn och låg hos vuxna. Den beräknade absoluta risken per år är mellan  $2.5$  och  $4.4 \times 10^{-4}$ /Gy vid exponering före 15 års ålder. Hos personer över 40 vid tiden för exponeringen är cancerrisken nära 0.

Klinisk användning av I-131 har inte visats orsaka cancer hos vuxna, men data för barn är mycket begränsade. Tjernobyl har visat att små barn är mycket känsliga. Cirka 1000 fall av sköldkörtelcancer har hittills konstaterats hos exponerade barn. Trots att tusentals barn fått mycket höga doser, har de flesta insjuknade fått mindre än 300 mGy sköldkörteldos. Detta beror på de stora arealer som kontaminerats och därmed på det stora antalet exponerade.

En analys av data från Vitryssland, Ukraina och Ryssland gav en riskfaktor på  $2,3 \times 10^{-4}$ /Gy per år vid exponering av barn under 15 år (1). En närmare analys av risken vid exponering av barn under 6 år i Vitryssland gav en riskfaktor på  $4,5 \times 10^{-4}$ /Gy per år (2). T.ex. hade bland 300 000 barn under 6 år med en medeldos på 60 mGy konstaterats 84 cancerfall på 7 år. Trots att dosberäkningarna i dessa analyser är osäkra och riskberäkningen baserar sig på medeldoser, är resultaten rimliga och stämmer anmärkningsvärt väl överens med riskkoefficienten för yttre strålning. Om den årliga risken förblir på samma nivå i 40-50 år, innebär det en livstidsrisk på cirka 1 %/Gy. Dödligheten i sköldkörtelcancer hos barn är

låg (hittills under 2 %) men sjukdomen är likväl allvarlig och innebär en stor börda både för de insjuknade och för hälsovården.

För beredskapsplanering är det skäl att använda samma riskkoefficient för I-131 som för yttre strålning.

Tjernobyli har också resulterat i en unik erfarenhet av att ge stabilt jod som engångsdos åt en stor befolkning. I Polen gavs jod åt 10 miljoner barn. Dessutom tog 7 miljoner vuxna in jod fastän det inte rekommenderats. Biverkningarna har utvärderats (3). Lindriga biverkningar, såsom utslag och illamående, var inte ovanliga. Däremot iaktogs inte en enda allvarlig reaktion hos barn, medan bland vuxna 2 fall av allvarlig allergi krävde sjukhusvård. Frekvensen av livshotande biverkningar från en engångsdos av stabilt jod kan nu fastställas till under  $10^{-7}$  hos barn och under  $10^{-6}$  hos vuxna.

För skydd mot inhalerat radiojod räcker sannolikt en engångsdos av stabilt jod. För att vara effektiv borde den intas innan det radioaktiva molnet anländer. I denna situation föreligger sannolikt inte tillförlitliga data om potentiella doser.

Senare är konsumtion av kontaminerad mjölk den viktigaste exponeringsvägen. Denna kan pågå i veckor och skulle förutsätta dagliga doser av stabilt jod. Frekvensen av biverkningar vid en sådan dosering är inte känd. Den är sannolikt låg hos barn men kan vara allvarlig hos vuxna. Som skydd mot radiojod i föda och mjölk kommer agrikulturella motåtgärder och kontroll av födan i första hand. Dock kan stabilt jod övervägas som alternativ för barn, om adekvat kontroll inte är möjlig eller skulle leda till otillräckligt näringsintag. Vuxna bör i en sådan situation hellre avstå från att dricka mjölk.

Den extremt olika utgångspunkten för olika åldersgrupper både när det gäller cancerrisk och risk för biverkningar gör det nödvändigt att behandla de olika grupperna skilt. Också doserna skiljer sig mellan olika åldrar i samma exponeringssituation. Små barn får de högsta doserna. I dokumentet behandlas gravida kvinnor, ammande mödrar, nyfödda, barn upp till 18 år, unga vuxna upp till 40 år och vuxna över 40 år. Jodprofylax är särskilt viktig för barn upp till 18 och för gravida och ammande. Dock bör upprepade doser av jod undvikas för gravida, ammande och nyfödda, på grund av risken för blockering av fostrets eller den nyföddas sköldkörtel.

### **Interventionsnivåer:**

#### **Balans mellan risk och nytta**

Basic Safety Standards for Protection against Ionizing Radiation (4) anger 100 mGy sköldkörteldos som generisk interventionsnivå för stabilt jod för alla befolkningsgrupper. Men i de allmänna anvisningarna för intervention konstateras, att nivåerna i praktiken kan vara högre eller i vissa fall lägre än de generiska värdena beroende bl.a. på närvaro av olika befolkningsgrupper. Det ger möjlighet att beakta den stora skillnaden i känslighet mellan olika åldersgrupper när det gäller exponering för radioaktivt jod, utan att BSS behöver ändras.

Om cancerrisken för barn är 1%/Gy och risken för allvarliga biverkningar av stabilt jod är  $10^{-7}$ , får vi en nettofördel redan genom att undvika doser på 0,01 mGy. I praktiken betyder det att risken för allvarliga biverkningar inte behöver beaktas. Lindriga biverkningar, som utslag eller illamående, har ingen större betydelse. Därför kan interventionsnivån fastställas enbart på basen av kostnader. Om jodtabletter har distribuerats på förhand, så att de är lätt-tillgängliga i händelse av ett nedfall, blir kostnaderna för den faktiska interventionen låga.

Tillämpning av den generiska interventionsnivån 100 mGy utan avseende på åldersgrupp innebär att vi accepterar en potentiell årlig incidens av sköldkörtelcancer hos barn på 20-50 per miljon inom några år, sedd mot bakgrunden av en spontan incidens på c:a 1 per miljon. Om vi sänker interventionsnivån för barn till 10 mGy, kan cancerincidensen hos de mest exponerade stiga till 2 – 5 per miljon per år, en ökning som eventuellt kan påvisas, om den exponerade populationen är mycket stor.

På grund av den relativt höga cancerrisken hos barn borde jodprofilax planeras redan för att undvika doser som överstiger 10 mGy. Även om de potentiella doserna i en reell situation skulle överskattas grovt, orsakas ingen betydande hälsorisk av att ge stabilt jod.

En motsvarande analys för vuxna kan inte göras, då den carcinogena effekten av I-131 inte har fastställts. På grund av den låga frekvensen av bieffekter av jod som engångsdos är det dock skäl att för säkerhets skull tillämpa den generiska interventionsnivån 100 mGy när det gäller unga vuxna.

För vuxna över 40 år risken för sköldkörtelcancer av I-131 sannolikt nära 0. Jodprofilax behövs bara för att förhindra deterministiska effekter. Detta garanteras av en interventionsnivå på 5 Gy.

För gravida och ammande kvinnor rekommenderas samma interventionsnivå som för barn.

### **Lagring av jodtabletter**

Jodtabletter måste vara tillgängliga för befolkningen med kort varsel, om profylaxen skall vara effektiv. Nära kärnreaktorer borde predistribution till hemmen övervägas. På större avstånd finns mera tid för distribution. Jodtabletter borde lagras på strategiska platser som skolor, daghem, sjukhus, apotek, brandstationer, polisstationer och befolkningsskydd, därifrån de vid behov snabbt kan delas ut. Tabletter borde också finnas fritt till salu på apotek.

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## EMERGENCY PREPAREDNESS IN FINLAND WITH SPECIAL EMPHASIS ON INTERNAL CONTAMINATION

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*Abstract* – Rapid development in the field of emergency preparedness has taken place during recent years. The very first measures in a possible emergency situation have been trained nationally and internationally. Less attention has been paid to measures in a later phase. To be able to react fast enough in an emergency situation it is essential to have well documented plans, written instructions and suitable measurement equipment ready for use. In the first phase of a nuclear accident iodine is of primary concern. The Finnish radiation protection authority –STUK has already some instruments calibrated for thyroidea measurements in field conditions outside the laboratory. In a Nordic project experts on internal contamination measurements will be trained to make rapid measurements with relatively simple instruments on large groups of people. The general public knows that it is possible to do direct measurements on people and will not accept prognoses based only on external radiation and foodstuff measurements. In the future it will be necessary to do also direct measurements on people for reassurance of the general public even if such measurements would not be necessary from a strict radiation protection point of view.

### INTRODUCTION

In the 1990's a rapid development in the field of emergency preparedness has taken place especially regarding transfer of information and 24 hour emergency service by radiation safety experts.

The very first measures in a possible emergency situation have been trained nationally and internationally. Less attention has been paid to measures in a somewhat later phase. One important experience after the Chernobyl accident was that the number of childhood thyroideacancer cases much exceeded the expected number. Unfortunately reliable direct measurements of I-131 in the thyroidea in Ukraina and Belarussia were done only to a limited number of children. Many uncertainties are involved in the data used for dose estimation.

To be able to react fast enough in an emergency situation well documented plans, written instructions and suitable measurement equipment must be available and ready for use. Equally important is that there is trained staff prepared to perform the measurements without delay. The Finnish Authority- STUK has already some instruments calibrated for thyroidea measurements outside the laboratory and will hopefully get more resources for this purpose in the near future. In a Nordic project internal contamination experts will be trained to make rapid measurements with relatively simple instruments on large groups of people. After the accidents at Chernobyl in Ukraina and Goiania in Brasil IAEA prepared a technical document (IAEA, 1994) with instructions for such measurements.

For groups of people also excreta measurements could be useful in a later phase of an emergency situation. Such indirect measurements are not dealt with in this presentation.

## EMERGENCY PREPAREDNESS AT STUK

The Finnish radiation and nuclear safety authority STUK is a regulatory body, a research centre and an expert centre with strong emphasis on preparedness for radiation accidents. After the Chernobyl accident a network of radiation safety experts was established to ensure fast response in emergency situations. After the reception of an alert the preparedness action should start immediately. This requires 24 hour expert services. Important tasks are maintenance of information concerning the accident situation and its consequences, safety assessment, recommendations and advice for protective measures and dissemination of information. The general principle for responsibilities in informing the public is that whoever directs activities is also responsible for informing the public. STUK always issues information with regards to radiation and its effects, and the other authorities on their own measures. STUK also gives recommendations to other authorities on local, county and national level. In Finland each authority decides upon measures concerning their own administration responsibilities and there is no centralised decision making.

The department for research and environmental surveillance at STUK is responsible for the preparedness for laboratory and field measurements in case of environmental contamination. This also includes an automatic radiation monitoring network for monitoring external radiation. This network gives the alert signal to the officer on duty.

The whole-body counting laboratory is prepared for measurements of internally, occupationally or via the environment, contaminated people.

### DIRECT MEASUREMENTS OF INTERNALLY CONTAMINATED PEOPLE

People become contaminated internally via ingestion, inhalation and to some extent via intact skin or wounds. Skin contamination is a disturbing factor when measuring people directly in an emergency situation with unavoidable environmental contamination.

For whole-body counting measurements in specialised laboratories there are only some details differing from routine operation. The surface contamination and the presence of the same radionuclides in laboratory air as in people have to be taken into consideration. This contamination level also varies with time. STUK has two systems for whole-body measurements one stationary and another mobile as shown in Figures 1 and 2. Special instructions have been prepared for emergency situations.

It is however not possible to do whole-body counting measurements in laboratory conditions on large groups of people rapidly. Rapid field measurements can be done of the whole body and especially of the thyroid with less sophisticated instruments. In the acute phase of a nuclear accident such measurements should be done without delay. Instruments and staff trained to use them should be available and plans for which groups of people to be measured prepared. Experiences after the Chernobyl accident showed lack of preparedness. Since then much work has been done. At STUK some monitoring instruments have been made available, calibrated and they are now ready for use. Figure 3 shows a schematic presentation of measurement arrangements.

The portable field instruments can be used for monitoring purposes to guide later actions such as repeated measurements with higher precision. The detection level for iodine-131 in the thyroid would be of the magnitude 10 kBq when using a GM-tube as detector, 1 kBq for a 2,5x2,5 cm NaI(TL) scintillation detector and 0,5 kBq for a HPGe semiconductor detector with a measurement time of 60 seconds. Places for arranging monitoring should be easily accessible such as hospitals, health care centres, schools etc. Preferably the place should be in an uncontaminated or only slightly contaminated area.

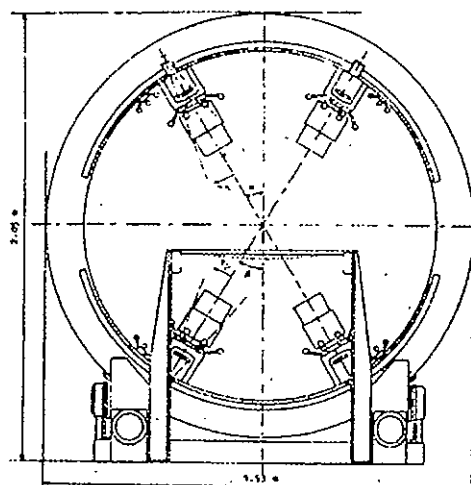


Fig. 1. Detector arrangement (head-end view) of the stationary whole-body counter IRMA 1.

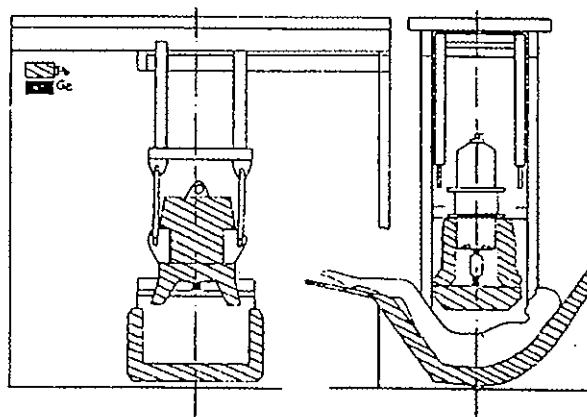


Fig. 2. Schematic drawing of the measuring geometry of the mobile whole-body counter IRMA 2.



Fig. 3. Schematic drawing of monitoring geometry.

## CONCLUSIONS

The results obtained with the rapid monitoring method described above can be used for different purposes such as:

- epidemiological studies based on results of measurements of large groups of people, when at least a representative part of the group was measured several times and the big number of results compensates for the statistical errors
- decision of need to administer stable iodine at places to which the radioactive cloud arrives later
- actions in a later phase especially the protection of special critical groups.

After the Chernobyl accident an active interest to improve whole-body counting techniques was seen. Intercomparison exercises were started by different organisations in many countries (. This is most important from a quality control point of view. The quality control aspects must be considered also when doing rapid measurements.

After the accident the general public learnt that it is possible to do direct measurements and will not accept prognoses based only on external radiation and foodstuff measurements. To avoid panic among the population measures that would not be absolutely necessary from a strict radiation protection point of view will have to be taken.

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## **SESSION IV**

### **CONTROLLABLE DOSE**

**Discussion of the Proposal by Roger Clarke, ICRP**

**Chairman: Per Hedemann Jensen**





## Panel and plenary discussion of *Controllable doses of low-level radiation - New ideas from ICRP*

The ICRP ideas on controllable doses were presented at a special session on 25 August 1999 at the ordinary meeting in the Nordic Society for Radiation Protection in Skagen, Denmark. After this presentation representatives from the Nordic radiation protection authorities (SSI-Sweden, STUK/Finland, Statens Strålevern/Norway, Geislavarnir ríkisins/Iceland and SIS/Denmark) each presented their views on the concept followed by a plenary discussion. The scientific secretary of ICRP participated in the panel and gave some additional information, also on the process that will lead to a revised System of Protection. He emphasized the extreme importance of a feedback to the ICRP. The majority of the Nordic authorities have not yet reached a final conclusion with regard to the ICRP proposal. Only at STUK the proposal has been widely discussed and also within the Governmental Committee for Radiation Safety in Finland. This paper gives a summary of the major comments given by the Nordic authorities as well as the reflections from the plenary.

### 1 The concept of collective dose

The major concern regards the proposal to abandon the concept of collective dose. The collective dose has always been regarded as a tool in the process of justification/optimisation. It is also a tool for the assessment of the general standard of radiation hygiene within the medical and the industrial use of radiation as well as within the nuclear field. There is no corresponding concept left if all the attention is shifted towards individual doses and the collective dose is abandoned.

The new principle in the proposal has been stated as: *If the risk to the health of the most exposed individual is trivial, then the total risk is trivial - irrespective of how many people are exposed.* The consequence of this principle would imply, for example, that if milk has been contaminated from radioactive fallout and would result in unacceptable individual doses, the principle of dilution would solve the individual risk problem without any reduction of the expected detriment being proportional to the collective dose. This would be in contradiction to the today's principles of public protection and a return to the "chimney policy" of the fifties, *i.e.* the higher the chimney, the lower the individual exposure.

As a consequence of the linear-non-threshold hypothesis the collective dose is a measure of the expected detriment, and comparison between different options of protection should still address detriment. It seems essential that the limitation of individual risk also in the future should be complemented by a source-related judgement of total harm, based upon the collective dose *and* the linear-non-threshold hypothesis of radiation risk, for which the owner of the source bears the full responsibility. To give up this principle would not be compatible with ethics prevailing today.

The collective dose concept should still be included in a revised System of Radiological Protection but it needs to be revisited and modified. The time-integration over very long time periods is meaningless and it seems prudent with a truncation of the time period to no more than a few hundred years.

### 2 The principle of justification

The present ICRP recommendations specifies that, when practices involving exposure or potential exposure to radiation are being considered, the radiation detriment should be explicitly included in the process of choice. The detriment being considered is not confined to that associated with the radiation. It includes other detriments and the costs of the practice. Often the radiation detriment will be a small part of the total. The justification of a practice like a nuclear power plant thus goes far beyond the scope of radiological protection and for such practices the principle of justification would play only a marginal role. However, in the medical field the justification principle would be essential. If a new practice is considered, *e.g.* screening for a given type of cancer, it is obvious that only if the expected number of cancers saved would exceed the expected number of cancers induced by the

screening itself plus other costs of the screening practice, the practice would be justified. Another example is the decision to introduce cleanup of contaminated land. Again, only if the net benefit of cleanup were positive, *i.e.* if the avertable doses and reduction in other negative attributes would exceed the costs, cleanup would be justified. Therefore, the principle of justification would still be needed, both for 'practice-' and 'intervention-like' situations although the principle may be redefined.

### **3 The principle of optimisation based on individual doses**

In the existing System of Protection optimisation is the process of deciding on the level of protection to obtain a *maximum net benefit*. In simple terms, the difference between the benefits and the disadvantages, expressed in the same terms, should be positive (justified) and should be *maximised* by setting the details of the protection measures. The process of optimisation includes the collective dose - both for practices and interventions. In the case of practices, the optimisation process will be subject to *constraints*. The ICRP introduced the use of dose constraints to provide a mean to deal with individual equity issues associated with the distribution of detriment from radiation exposure. Source-related dose constraints are applied in the process of optimisation to limit inequity. In the case of intervention the use of such constraints is not required. If the collective dose were abandoned in a revised System of Protection the process of optimisation needs to be revised as all attention is shifted towards individual doses. The principles of justification/optimisation always go together and these principles should be redefined to cope only with individual doses. The change of ALARA to ALARP and the consequences on the optimisation process should be elaborated. The role of source-related constraints should be revisited in the light of the proposed concept of investigation levels.

### **4 The distinction between practices and interventions**

Although the formulations of the optimisation principle differ for practices and interventions, the practical implementation of optimisation is essentially the same process, whether it is considered in the context of the continuing operation of a practice, as part of decommissioning of a practice, or for intervention. In all cases, it consists of looking at the different options available and how exposures might be reduced, and choosing the course of action, which results in the greatest net benefit, considering all of the relevant factors that influence costs and benefits. Some situations, *e.g.* clean-up situations, will clearly fall into one or other of the categories - practice or intervention - but for others it will not be so obvious. In other cases, although the distinction and choice is clear, it may not be acceptable to society to reach different conclusions for the level of protection that would depend on the origin of the source of exposure. The suggested removal of the concepts of practices and intervention would probably create more problems than it solves. It might be better to formulate a general framework (justification, optimisation, individual protection, limiting inequities) that should include the principles of practices and intervention, but place them in a wider context in which they continue to provide guidance for situations that fit well into one category or the other. For situations that do not fit well into either category, the framework should provide useful guidance that is independent of such a categorisation.

### **5 The Action Level/Investigation Level concept and dose limits for public exposure**

The proposed concept of controllable dose includes a maximum individual dose level, an *Action Level*, around some tens of a mSv in a year. If controllable doses are above this level, *action should be taken* to reduce the (individual) doses. The management of controllable doses below the Action Level would be by individual-related source-specific *Investigation Levels*. They would apply to different actions taken to reduce exposures at the source, in the environment or by moving people. They would cover, for example, occupational exposures, medical doses, doses from radon or from other elevated levels of natural radionuclides, and those after an accident. The action level/investigation levels would replace the existing dose limits for practices and intervention levels/action levels for intervention situations. The practical application of investigation levels with regard to dose addition in 'practice-like' situations and dose reduction in 'intervention-like' situations needs to be clarified. If investigation levels are to be applied both as 'dose limits' and 'intervention/action levels', the derivation of their numerical values needs further clarification. For instance, should they be based upon justification/optimisation (constrained/unconstrained) of protection of individuals or populations (in-

dividual- or source-related protection), levels of acceptable risk to individuals or just upon generic values of reference levels that have been inherited from the existing System of Protection?

## 6 General comments

The following general comments were made in the panel and plenary discussion:

- ❑ The existing System of Protection seems to be too complicated and difficult to explain even to professionals within radiation protection; there is a need for simplification and improvements of the System of Protection.
- ❑ The concept *controllable dose* may simplify the System of Radiological Protection provided that the concept can be properly defined; for instance, is it controllable at the source level or at the exposure level? The latter would mean a much wider application.
- ❑ The meaning of the word *controllable* is not unequivocal and may lead to misunderstandings because of translation problems in some languages.
- ❑ Including medical exposures into the general framework of controllable dose seems artificial; the clinical judgement of what is necessary and optimal for the patient does not need to be influenced by any 'action level' defined for other purposes; the distinction between medical, public and occupational exposure should be kept.
- ❑ Dose limits for public exposure might be unnecessary but excluding them from a revised System should be presented properly, emphasising that it does not mean any reduced protection of the public; it may have serious psychological and political consequences and the ICRP may lose credibility.
- ❑ The concept of action level/investigation levels directed towards the individual protection seems useless for the ongoing work in the former Soviet Union on remedial measures to reduce doses from the Chernobyl accident, e.g. decontamination of forestry, because of the lack of collective dose.
- ❑ The protection principles for individuals might be expressed as for other agents like chemicals.
- ❑ ICRP should be very careful with the introduction of a trivial risk (negligible dose), even if very small doses to many individuals might be of less interest. The question is if there should be room for a *precautionary principle* or *prudent avoidance*?

There were serious reservations with regard to the implementation of a completely new System of Protection within a time period of only a few years. The existing System was developed gradually over a time period of about 50 years. The road towards a new system seems rather long - a time perspective of 20 years was mentioned as realistic, also because the existing international directives and standards as well as national legislation just have adapted the ICRP 60 recommendations from 1990.

9 September 1999

Per Hedemann Jensen



# **SESSION V**

## **NATURAL RADIOACTIVITY**

**Chairman: Tua Rahola**



# **Radon-222 in soil, water and building materials: Presentation of laboratory measurement methods in use at Risø**

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## **Abstract**

Three methods for measurements of radon-222 in soil, water and building materials are described, and sample results are given. The methods are used both in connection with the health problem of radon-222 in houses and with radon-222 as a tracer of environmental transport.

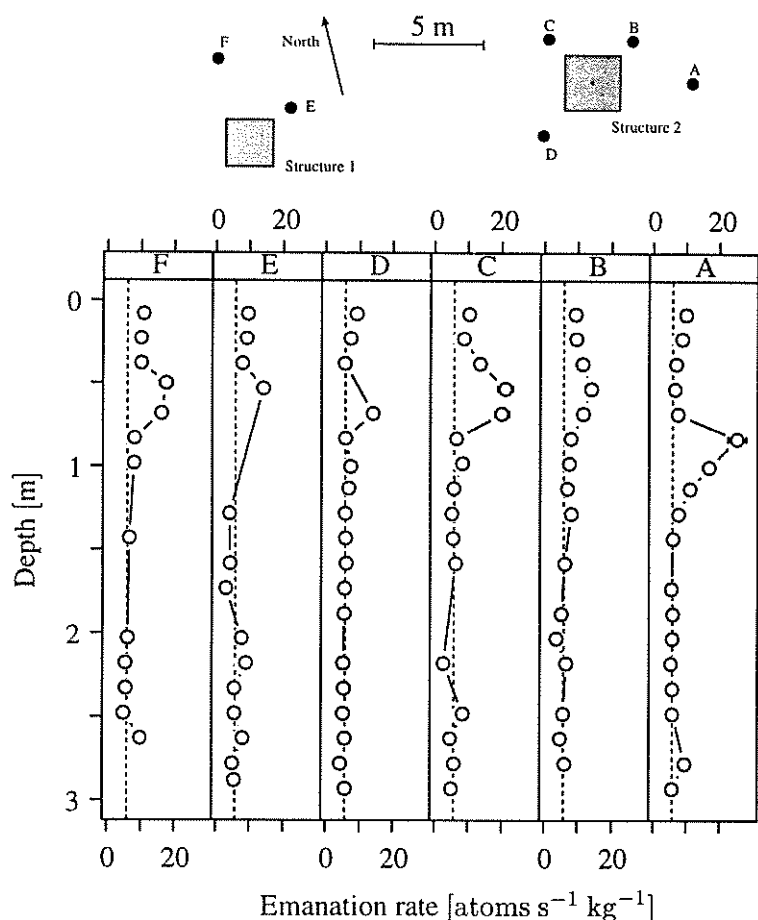
## **1 Introduction**

Radon-222 constitutes a dominant part of the life-time radiation dose for most persons in the Nordic countries. This is, however, not the only reason why radon-222 is an interesting object of research: radon-222 is also an excellent tracer of certain transport processes in parts of the environment. At Risø, we are engaged in studies of both of these aspects. To this end we have adapted laboratory methods relating to radon measurements in soil, water and building materials. The purpose of this contribution is to describe three such methods, and to outline some results.

## **2 Emanation from soil samples**

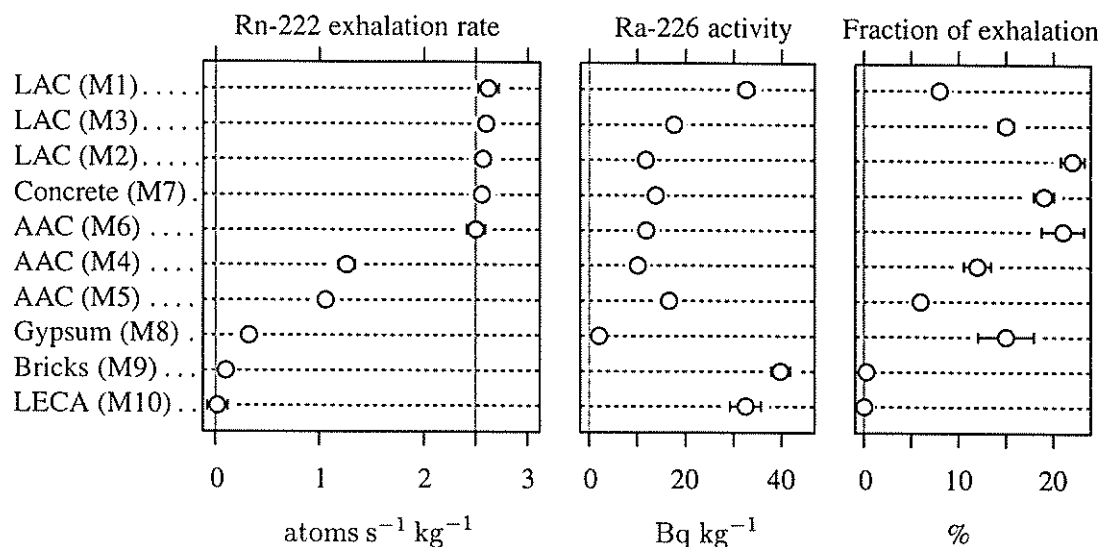
The emanation rate of soil is the number of radon atoms that (effectively) escape the soil grains into the pore system per kg dry mass per second. The emanation rate partly controls the soil-gas radon concentration. It is measured as follows: The sample is mildly disaggregated by forcing it through a brass plate with a 22 mm hole. The sample (typically about 300 g) is placed in a 6 L steel chamber. The chamber is flushed to near-zero radon concentration with about 30 L of aged nitrogen from a pressurized cylinder. Thereafter the chamber is closed, and radon starts to build up inside the chamber. Over the following days (or weeks) the chamber radon concentration is determined at selected times. Evacuated 200 mL scintillation cells are used for the purpose. An airbag with aged nitrogen is used to balance the pressure in the chamber after sampling. The scintillation cells are counted on a computer-controlled sample-changer with one photomultiplier tube. Samples are weighted before and after analysis. A moisture determination is also carried out. The analysis of the data includes corrections for dilution of the chamber because of sampling and pressure and temperature dependent transfer coefficients. Analysis of precision is carried out both for each chamber concentration determination (typically each cell is counted 3 to 4 times) and for the overall emanation-rate analysis (typically each analysis is based on 3 or more scintillation cell samples). The analysis-of-precision is a chi-squared test where the a priori uncertainty of the quantity of interest is compared with the experimental standard deviation. The final result of the analysis is reported on standardized computer-generated measurement sheets. Normally, batches of 12 samples are analyzed. In total, 480 analyses have been carried out. About 250 of these come from the Geological Survey of Denmark: To help understand differences in indoor radon potential for different geologies, samples from different





**Fig. 1.** Emanation rate results for six soil cores (named A to F) taken at the clayey till field site of Risø's radon test structures. The site is sketched above the graph. The dashed reference line at  $5.8 \text{ atoms s}^{-1} \text{ kg}^{-1}$  is the mean of all results below 1.4 m. Uncertainties (expressed as single standard deviations) are in most cases smaller than the plotted data points.

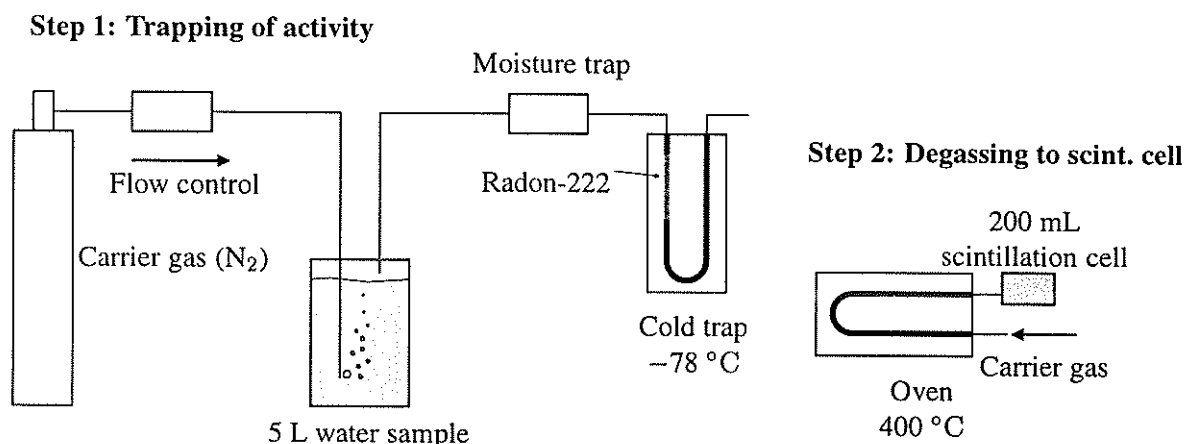
parts of Denmark have been analyzed. A small number of analyses have been done for engineering companies to help assess the risk for high indoor radon for larger construction works. About 200 emanation-rate determinations are from the site of Risø's radon test structures (the first structure, which is no longer in existence, is described in (Andersen, 1992)). Part of these analyses have been used to investigate the effect of disaggregation and moisture content (Andersen, 1998). Another purpose has been to provide soil parameters needed for the comparison of numerical models of radon transport in soil and entry into houses. Figure 1 shows 95 emanation rate results for six 3 m soil cores from the test structure site. Below 1.4 m depth, there is little variability with depth and from core to core. The pooled mean and standard deviation of these results amount to  $5.8$  and  $1.4 \text{ atoms s}^{-1} \text{ kg}^{-1}$ , respectively ( $N=48$ ). For the top (0–1.4 m) layer, the mean and standard deviation are  $10.1$  and  $4.4 \text{ atoms s}^{-1} \text{ kg}^{-1}$ , respectively ( $N=47$ ). It can be seen from the figure, that the emanation rate peaks in the layer between 0.5 to 0.85 m below the surface. The maximum value of  $25 \text{ atoms s}^{-1} \text{ kg}^{-1}$  occurs at 0.83 m depth for profile A. The high-emanation layer may result because the layer is more rich in radium than the other parts of the profile. Another possibility is that the fraction of emanation for the layer is high. The detailed mapping based on soil cores seems to allow for a much more clear understanding of the emanation-rate conditions of the site than previous measurements (cf. results given in (Andersen, 1992)).



**Fig. 2.** Exhalation rate results. The meaning of abbreviations is given in the text. M1 means material 1 etc. The fraction of exhalation is the exhalation rate divided by the radium-226 concentration. Uncertainties (expressed as single standard deviations) are in many cases smaller than the plotted data points.

### 3 Exhalation from building materials

The mass-specific exhalation rate of radon-222 from a building material sample (such as a brick or a concrete slab) is the amount of radon-222 that escapes the sample per kg per second. Studies carried out by Jonassen, Ulbak and coworkers in the 1970'ies and 1980'ies showed that the radon-222 exhalation rate of ordinary Danish building materials is low. Special building materials with large radon-222 exhalation rates do however exist (at least in other countries). For this reason, it is of interest for Danish producers of building materials to be able to quantify this aspect of their products. Risø has therefore set up a method for exhalation rate measurements. It is a closed-chamber method. The sample (typically 30 x 30 x 5 cm<sup>3</sup>) is placed in a 55 L stainless steel chamber together with a radon monitor that measures the radon-222 concentration every hour. Also temperature, humidity and pressure in the chamber is registered. The sample is conditioned for 24 h with a flow of aged nitrogen. The flow has a relative humidity of about 50 %. After conditioning, the chamber is closed and the radon concentration starts to build up. The measurement extends from 3 to 10 days. The sample is weighted before and after measurement. The method is documented in a report (Andersen, 1999) together with measurement results for 10 Danish building materials. All materials were found to have exhalation rates below 2.7 atoms s<sup>-1</sup> kg<sup>-1</sup>. The highest value were for ordinary concrete, lightweight aggregate concrete (LAC) and autoclaved aerated concrete (AAC). Bricks, gypsum and lightweight expanded clay aggregate (LECA) had values below about 0.3 atoms s<sup>-1</sup> kg<sup>-1</sup>. Under consideration of the application of the materials in a typical Danish single-family house, it was found that such materials cannot increase the indoor radon concentration by more than 10 Bq m<sup>-3</sup>. The Danish Institute for Radiation Hygiene (SIS) has measured the radium-226 concentration of the samples. The results are shown in Figure 2. The figure shows, for example, that some of the materials have a fraction of exhalation above 20 %. Other materials (such as bricks) have a very low fraction of exhalation. Although bricks have the largest radium concentration, very little radon exhale from the surface (the fraction of exhalation is less than 1 %).



**Fig. 3.** Radon-222 degassing of seawater samples. The water sample is stripped for radon by flushing it with 30 L of nitrogen. The activity is collected on activated charcoal cooled to dry-ice temperature. In step 2, the trap is heated to 400 °C, and the activity is transferred to an evacuated scintillation cell.

## 4 Radon in seawater

Groundwater tends to have a high radon-222 concentration compared with seawater. Radon-222 can therefore be used to trace the sub-marine supply of ground water to the sea. In 1998 Risø engaged in an EU-project called sub-Gate which deals with submarine ground water fluxes and transport processes from methane rich coastal sedimentary environments. In this project, an area of Eckernförde Bay in the western Baltic Sea is investigated. In the area many so-called *pockmarks* exist, and ground water (containing radon) is believed to seep into the sea. Because of the relative close distance from the study area to Risø, a method has been adopted where samples are transported back to the laboratory for radon analysis. The method is sketched in Figure 3. It has been modified from that described by Mathieu *et al.* (1988). Cruises have been made in December 1998 and in July 1999. Preliminary results indicate that excess radon-222 exist close to the sea floor. In December 1998, near sea-floor radon-222 concentrations as high as 30 mBq L<sup>-1</sup> were observed. In contrast, near sea-surface radon-222 concentration were only about 3 mBq L<sup>-1</sup>. The radium-226 concentration was found to be less variable. The average of 19 measurements was 3.3 mBq L<sup>-1</sup>.

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# COMPARISON OF INDOOR RADON CONCENTRATIONS MEASURED BY ACTIVE CHARCOAL-CANISTERS AND ALPHA TRACK DETECTORS.

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**Abstract:** Screening measurements of radon by using active charcoal-canister, was one of the method listed in Norway for several years, but is now removed from the official list. Miljølaboratoriet has more then ten years experience of using the method as a first step in radon investigations. We here present two different studies, where both short-term and long-term have been used in the same rooms and two studies where short-term measurements have been use to find areas with high radon risks. As these results shows good agrement, one can ask if it is reasonable not to have the active charcoal-canister listed as a screening method

## Introduction

The method of using ative charcoal to measure radon concentrations has some limitations. As the active charcoal allows continual adsorption and desorption of radon, the method does not give a truly integrated measurement over the exposed time. The adsorption is also affected by humidity. But if canisters are placed in conditions where there are small fluctuations in radon concentrations and we can correct for humidity, the charcoal-canister is a very useful screening-method; a big number of measurements within a defined area used to find smaller areas to be followed up with long-time measurements.

For years, the charcoal-canisters have been the most frequently used screening-method in the U.S., and joint analysis of long- and short-term monitoring data have been performed (Price and Nero 1996). Today, only methods that integrates over a period of more than two weeks are accepted as screening measurements in Norway. As Miljølaboratoriet has used charcoal-canisters for several screening studies during the last ten years, we want to present some data.

## Methods

The charcoal-canister used by Miljølaboratoriet is of the type Radios, developed with research funding support. Humidity corrections were done by weighing the charcoal-canisters before they were sent out and after returned. The radon concentrations could then be calculated with correction for watercontent according to A.C. George (1984). The gamma-detectors used were a 10" NaI-well-detecor and a 3" NaI-detector. The alpha track detectors used in most of the presented data are unfiltered detectors (LR-115), but the latest are filtered once (CR-39).

The first study was a screening short-term monitoring in 60 flats situated in three blocks built in terraces up a slope. The owners were asked to put one charcoal-canister in a livingroom with good contact to ground, and to have ventilation at a minimum during this period, in order to determine if there was risk of radon in the flat. Monitoring was conducted during three days in the end of January, with temperatures between 0 and 6°C, no wind. Long-term monitoring in two livingrooms over five weeks, was performed from the middle of March. Mean outdoor temperature was 6 - 7°C.

The second study was conducted in seven flats of a larger complex. Two short-term monitoring periods, the first in the beginning of January, temperature 3 to 5°C and windy, the last in the end of January, -4 to +3°C and nearly no wind. The charcoal-canisters were placed in a bedroom with reduced ventilation to find

possible radon risk. In the end of the year, the same bedroom and another livingroom had a long-term monitoring period of 6 weeks; middle temperature 5,5°C, 10 days with strong wind.

The third study was a short-term screening done by charcoal-canisters at the basement-level in 104 houses in March. These houses represented 10% of the houses in the municipality. Only a few family houses and occupational buildings in the different areas have been long-term measured till now.

The fourth study has only been started, the house owners are waiting for an organized long-term test. In an area of single family houses built up in a slope, half of the houses were tested for risk of radon by using charcoal-canisters at the basement level in rooms with good contact to ground, in end of March with outdoor temperature in middle -5°C.

## Result and discussion

In the first study, only 10 flats showed concentrations below 200 Bq/m<sup>3</sup> in the short-term monitoring. Therefore, we visited some of the flats, studied the ventilation system and looked for leakages. All flats were then followed up with long-term measurements, and the radon concentration values were compared. The results are shown in table 1.

Some of the flat-owners did not continue with the second step, and some changed the ventilation between the first and second measurement. Those who reported changes, are marked in the table. In the first step, the flat-owners were asked to keep ventilation at a minimum and choose a room with good contact to the ground, in the second they were asked to live as normal. The long-term monitoring was performed in two rooms, and one had to expect lower values. Two flats with low values during screening ended up with mid-level values, and two flats with mid-level values ended up at the highest level.

**Table 1.** Number of flats within the three radon-levels in short-term v.s. long-term measurements. Total 60 flats.

	<200 Bq/m <sup>3</sup>	200 - 400 Bq/m <sup>3</sup>	>400 Bq/m <sup>3</sup>
Screening	10		
Long-term <sup>a</sup>	5	2 (1 <sup>b</sup> )	
Room values	13	2	1
Screening		17	
Long-term <sup>a</sup>	13 (2 <sup>b</sup> )	2	2
Room values	22	8	3
Screening			33
Long-term <sup>a</sup>	11 (6 <sup>b</sup> )	11 (1 <sup>b</sup> )	10
Room values	20	26	18

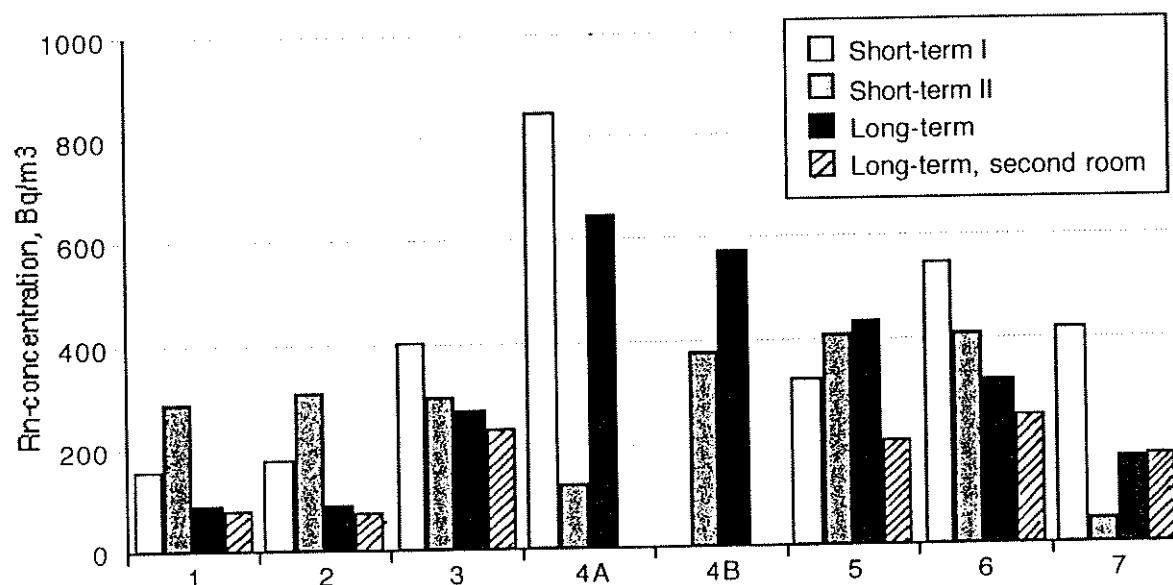
<sup>a</sup> Mean value of two rooms in the flat.

<sup>b</sup> Change in ventilation after screening.

In the second study, the flat owners had two screening periods in the same bedroom. Then, after ten months, this room and another livingroom had long-term monitoring. The results are shown in fig. 1.

After the first screening, the owner of flat 4 reported more ventilation during the second period, while in flat 7 they had kept the window open day and night. Flat 4 was also able to measure the second room both of the two latest periods. With the

measuring conditions taken into account, this study shows that the short-term monitorings provides useful information.



**Fig. 1.** Measured radon-concentration in seven flats. The two first values are short-term, third long-term in the same room as the two first, and the fourth is long-term measurement in another room of the flat.

The results of the third study are given in table 2. The authorities of the municipality wanted to use it as advice for new house-builders. The few long-term measurements that are performed, shows that this study also should have been used to advice the people who already live in the regions where radon-levels are high to perform long-term measurements.

**Table 2.** Mean radon-concentrations in different regions of a municipality, based on short-term measurements at basement level in 104 family houses.

Reg.	#	Min. Bq/m <sup>3</sup>	Max. Bq/m <sup>3</sup>	Mean value Bq/m <sup>3</sup>	Long-term measurements
1	3	20	85	50	26 <sup>b</sup> , 41 <sup>b</sup> 200 <sup>a</sup>
2	3	< 20	305	150	
3	13	< 20	195	80	
4	3	165	2130	955	
5	6	< 20	195	90	
6	7	< 20	370	140	<20 <sup>a</sup>
7	15	20	410	120	
8	6	< 20	105	60	
9	9	< 20	200	65	
10	10	20	150	70	
11	15	< 20	305	85	20 <sup>a</sup> 954 <sup>a</sup> , 708 <sup>a</sup> , 254 <sup>a</sup> , 92 <sup>c</sup> , 747 <sup>d</sup>
12	7	25	405	245	
13	7	65	4690	1300	

a Kindergardens

b School

c School, 2nd floor

d Livingroom, family house

The last study has only short-term monitoring results. They are shown in table 3. Almost 40% of the houses showed high radon concentrations at the basement level. Mapping the results gives a view of what areas should be followed with long-term measurements.

**Table 3.** Radon-risk short-term measurements in basements of 36 family houses built up a slope.

Rn-levels	<200 Bq/m <sup>3</sup>	200 - 400 Bq/m <sup>3</sup>	>400 Bq/m <sup>3</sup>
Number	<b>13</b>	<b>9</b>	<b>14</b>
Mean value	85 Bq/m <sup>3</sup>	285 Bq/m <sup>3</sup>	940 Bq/m <sup>3</sup>
Min. - max.	35 - 150 Bq/m <sup>3</sup>	200 - 355 Bq/m <sup>3</sup>	450 - 1785 Bq/m <sup>3</sup>

### Conclusions

The results given in this paper should show that the method of charcoal-canister can give useful information when used as a first step in locating areas of radon risk. One has to registrate temperature, wind, pressure and ventilation conditions, and to correct for humidity. Lots of time can be saved in the work finding areas with high risk for radon, and people can be informed if they live in a high-risk area if the method still can be used.

Acknowledgments: – The author wants to thank teachers and students of Hov school in Holtålen for organizing the practical part of the screening study.

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# REMOVAL OF NATURAL RADIONUCLIDES FROM DRINKING WATER FROM PRIVATE WELLS IN FINLAND

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## Abstract

Removal of natural radionuclides is often necessary in Finland when household water is taken from a drilled well. Removal of radionuclides by various methods from Finnish groundwaters were studied in a EU-research project, TENAWA. The results indicated that radon can be removed very efficiently (up to 99%) by applying aeration or granular activated carbon (GAC) filtration. Uranium and radium were also removed (over 94%) by using strong base anion (SBA) and strong acid cation (SAC) resins. The capability of reverse osmosis (RO) equipment to remove radionuclides was over 90% for uranium, radium and polonium. The water quality analyses indicated that water quality remained mostly good during the water treatment.

## Introduction

Finnish people are exposed to higher levels of natural radiation than people living in most other countries due to elevated levels of radon in indoor air and natural radionuclides in household water. According to the present estimates about 700,000 Finns regularly use water from their own wells and about 200,000 of them consume water from drilled wells. The mean annual effective dose for a Finn is 3.7 mSv from all sources of radiation. The dose from ingested water is estimated as 0.14 mSv and from indoor radon as 2 mSv. Only the water derived from drilled wells can elevate the dose considerably (the average dose by ingesting water is 1.6 mSv and the maximum dose 166 mSv).

Natural radionuclides occurring in groundwater mainly originate from the uranium series. The nuclides which contribute to the radiation dose are the isotopes of radon ( $^{222}\text{Rn}$ ), lead ( $^{210}\text{Pb}$ ), polonium ( $^{210}\text{Po}$ ), uranium ( $^{234,238}\text{U}$ ) and radium ( $^{226}\text{Ra}$ ). Rn concentrations in drilled wells can be high, average 540 Bq/L and max 77 500 Bq/L (Salonen 1994). Uranium quite often occurs simultaneously with Rn, while Ra, Po and Pb less frequently. In Finnish drilled wells uranium concentrations may be even a few mg/l, average 0.02 mg/L and max 12.4 mg/L (Salonen 1994) and the dose from uranium may thus be high. Fortunately, such high concentration occurs only rarely. The chemical toxicity of uranium is higher than its radiotoxicity and therefore the limit for uranium will be set on that basis. The concentrations of Rn and uranium in bedrock groundwater are clearly related to the rock type. The abundance of radionuclides of groundwater in granitic areas is higher than that in other types of bedrock (Lahermo et al, 1990).

Various radionuclide removal methods were studied in an EU- research project called TENAWA, "Treatment Techniques for Removing Natural Radionuclides from Drinking Water". The aim of the TENAWA project was to study the applicability of various treatment techniques for removing radionuclides from drinking water and it was carried out by partners from Finland, Sweden, Germany and Austria. The main result obtained in Finland are here presented briefly.

## Materials and methods

Removal of Rn by aeration was studied in seven private households in Finland. Four aerators from different companies were tested. Rn removal applying GAC filtration was studied in



twelve private households by using filters with bed sizes of 39 and 63 litres. Because Rn is released into indoor air during water usage, the aerators and the GAC filters were installed to treat all household water (point-of-entry). Ion exchange units for uranium and Ra removal were studied in six private homes to treat either all household water (filter sizes of 21 litres and 36 litres) or kitchen water (size of the filters 7 litres and 11 litres). The practicability of a tap filter was also studied in one leisure residence. The capability of RO equipment to remove radionuclides was studied in two private households. Two units of commercial RO equipment were installed for test purposes only and the treated water was not used as household or drinking water.

The test locations were selected so that water types most typical of Finnish bedrock were covered. The main water types were Fe and Mn bearing, humus-rich and slightly saline water and water low in mineral and organic matter content.

## Results

### Removal of radon

The two main methods of removing Rn from water are aeration and granular activated carbon (GAC) filtration. Aeration is based on the transfer of Rn from the aqueous phase to the gas phase. Aeration can be carried out either by dispersing water into air or by dispersing air into water. The mass transfer of Rn occurs over the water-air interface in the direction of decreased concentration. After aeration, the Rn-rich air is ventilated into the outdoor air.

In GAC filtration, Rn is retained on the carbon by physical adsorption and it decays in the filter. If the filter is used regularly, the decay rate of Rn equals the rate of adsorption in three weeks. After this, the activity of Rn accumulated in the filter remains near constant. Rn decays inside the filter and produces daughters ( $^{218}\text{Po}$ ,  $^{214}\text{Pb}$ ,  $^{214}\text{Bi}$ ,  $^{214}\text{Po}$ ,  $^{210}\text{Pb}$ ,  $^{210}\text{Bi}$  and  $^{210}\text{Po}$ ) that are also adsorbed on the carbon. Therefore, GAC filters emit gamma radiation and must be placed or shielded properly. Spent carbon from a GAC filter may also contain several hundred kilobecquerelles of  $^{210}\text{Pb}$  after a few years' service. Both the intensity of the gamma radiation and the amount of accumulated  $^{210}\text{Pb}$  depend on the daily water usage and the Rn concentration of the raw water. Therefore, GAC filtration is not recommended to be used when Rn concentration exceeds 5,000 Bq/L.

The average Rn removal efficiencies attained by the aerators varied between 61% – 99% depending on the type of the aerator and the aeration time applied (Turtiainen et al, 1999). The theoretical flow rates, by which the aerators attain a 99% removal, varied between 10-50 L/min. One of the aerators tested during the TENAWA project was poorly designed and constantly failed to reduce Rn levels below 100 Bq/L. Several technical malfunctions were documented for most aerators; e.g., breakdown of the water pump, clogging of the solenoid valves, insufficient water feed, and detached air tubes or O-ring seals. Iron and manganese precipitate partly during aeration and their precipitates may cause technical problems (e.g. clogging valves). Therefore high concentrations of Fe and Mn should be removed before the aeration unit. Aeration can only insignificantly decrease the concentrations of uranium, Ra, Pb and Po.

GAC filtration removed Rn efficiently (>99%) at most test locations (Turtiainen and Salonen, 1999). High concentrations of uranium, and possibly organic matter, were found to lower the adsorption rate of Rn. Therefore, an additional anion exchange unit should be placed prior to the GAC filter when water contains uranium. The concentration of iron (0.7 mg/L) and manganese (0.26 mg/L) had no effect on the adsorption of Rn. The external gamma dose rates

on the surface of the GAC filters could be very high, even over 100  $\mu\text{Sv/h}$ . With proper shielding and the placement of the unit elevated doses caused to the residents could be avoided. According to the results obtained during the TENAWA project, the GAC filtration is not a viable treatment alternative for removing uranium, Ra, Pb and Po from water. Their removal efficiencies varied greatly; 0%-100% for uranium, 0%-95% for Ra, 30%-100% for Pb, and 50%-100% for Po.

### **Removal of uranium, radium, lead and polonium**

The most commonly used method for uranium and Ra removal from drinking water is ion exchange. Removal of uranium and Ra by strong organic resins was studied with different types of ion exchange units and water (Perfler et al, 1999). The removal efficiency of uranium was over 95% at all test locations, independent of water quality, filter type and the treated bed volume. The strong base anion (SBA) resin had a very high capacity for uranium – no breakthrough occurred, even though more than 7,900 bed volumes of water had been treated. The removal efficiency for Ra applying strong cation exchange (SAC) resin was over 94%, which is sufficient to mitigate even the highest levels of Ra occurring in groundwaters. The removal efficiencies of Pb and Po was uneven due to the various speciations of these radionuclides occurring in natural waters.

Water quality was improved by the ion exchange treatment because the concentration of organic matter, Fe, Mn, turbidity,  $\text{PO}_4$ ,  $\text{SO}_4$  and  $\text{NO}_3$  was decreased. However, due to the decrease in alkalinity and hardness and the increase in chloride concentration the water becomes rather corrosive, especially when mixed-bed exchangers are applied. The increase in chloride was sometimes too great and levels higher than 100 mg/L were found in the treated water. Microbiological analyses did not indicate elevated heterotrophic plate counts. No coliform bacteria were found.

The dose equivalent rates on the surfaces of the unit varied between 0.13 and 11.3  $\mu\text{Sv/h}$ . The highest dose rates were measured on the mixed bed exchangers (Ra that is retained generates  $^{214}\text{Pb}$  and  $^{214}\text{Bi}$  which are gamma emitters) and at locations where high Rn levels occur in the water. At a distance of one meter the dose rate was close to the background level.

Membrane technologies can be utilised alone or in combination with other techniques. Of these reverse osmosis (RO) is one of the few methods which can be applied for the simultaneous removal of uranium, Ra, Pb, Po and water salinity. In Finland such waters may occur in coastal areas, where wells have been either drilled too deep or if their water level has decreased as a result of increased water consumption from the bedrock aquifer.

The RO equipment studied here were small point-of-use units, which can produce water for daily drinking and cooking purposes only. The ability of RO equipment to remove radionuclides was studied in two private households for a period of seven months (Huikuri et al, 1998, Jungclas et al, 1999). The reduction of uranium and Ra was above 99%. Rather low reduction for Rn (under 50%) was an expected result due to the high concentration of Rn and the fact that Rn passes through the membrane. The average reductions of Pb and Po were 92% and 93%, respectively. The RO units also removed most of the other water constituents. The reduction of Ca, Mg, Na, K, Fe, Cl,  $\text{SO}_4$ , TOC (total organic carbon) was above 94%. This is a clear disadvantage for the consumers because firstly, it is not advisable regularly to drink water that is de-mineralised and secondly, the water will become corrosive and may thus be harmful for the plumbing and also dissolve harmful metals.

**Table 1.** Radionuclides removal methods and their average reduction percentages.

Treatment method	Reduction (%)					Side effects
	<sup>222</sup> Rn	<sup>234,238</sup> U	<sup>226</sup> Ra	<sup>210</sup> Pb	<sup>210</sup> Po	
Aeration	61 – 99	-	-	-	-	Removes CO <sub>2</sub> , pH may increase, Fe and Mn precipitate, O <sub>2</sub> enrichment.
GAC-filtration	> 99	0-100 <sup>1</sup>	0-95 <sup>2</sup>	30-100 <sup>2</sup>	50-100 <sup>2</sup>	Reduced Fe, turbidity, organic matter. Deterioration of hygienic quality possible.
Ion exchange - anion resin	-	>95	35-60	20-70	50-70	Reduced NO <sub>3</sub> , SO <sub>4</sub> , organic matter, Fe, Mn, hardness, increases water salinity. Deterioration of hygienic quality possible.
-anion and cation resins	-	>95	95-98	20-90	50-70	Reduced Fe, Mn, hardness, increased water salinity, Deterioration of hygienic quality possible.
RO-filtration	-	>99	>95	>95	>95	Reduces total dissolved solids up to 98%, treated water corrosive.

<sup>1)</sup> high adsorption can be achieved only a few days after installation. A couple of months later adsorption is close to 0%.

<sup>2)</sup> depends on water quality

### Conclusion

Both aeration and GAC filtration can remove Rn from water very effectively (reduction up to 99%). For removal of uranium, Ra, Pb and Po, aeration and GAC filtration are not viable treatment alternatives. Water quality has an effect on removal efficiency and needs to be taken into account when Rn removal equipment is acquired. Ion exchange is a proper method for removal of uranium and Ra. Removal of uranium and Ra was over 94% independent of water quality. The removal efficiencies of Pb and Po vary greatly, since most of these nuclides are shown to be particle bound in natural waters. RO equipment removed most other natural radionuclides (>90%) from water, except Rn. Its main disadvantage is that water is almost completely de-mineralised. Post treatment would be preferable (re-hardening water).

The highest dose from radionuclides in drinking water is caused by Rn and the next highest doses by Pb and Po. Removal of all nuclides effectively by using one method is difficult, due to the various chemicals behaviours and their speciations in natural waters. Especially the removal methods of Pb and Po should be further studied in the future.

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## **DOSE FROM DRINKING WATER IN FINLAND**

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### **Abstract**

The dose from drinking water originates almost totally from naturally occurring radionuclides in the uranium-238 series, the most important nuclide being radon-222. Second comes lead-210, and third polonium-210. The mean age-group-weighted dose received by ingestion of drinking water is 0.14 mSv per year. More than half of the total cumulative dose of 750 manSv is received by the users of private wells, forming 13% of the population. The most exposed group comprises the users of wells drilled in bedrock, who receive 320 manSv while comprising only 4% of the population. The calculated number of annual cancer incidences due to drinking water is very sensitive to the dose-conversion factors of ingested radon used, as well as to the estimated lung cancer incidences caused by radon released from water into indoor air.

### **Introduction**

The concentration of natural radionuclides in Finnish ground water are high by world standards. The main reason is the bedrock of the Fennoscandian shield, which consists mainly of granitoids. Consequently, in Finland drinking water is the major path of radiation exposure through ingestion (Salonen 1994).

Based on the Radiation Act of 1992, STUK has issued safety guides to present the safety requirements for different practices. ST-Guide 12.3 of 1993 stated that annual doses from drinking water supplied by waterworks should be less than 0.5 mSv (STUK 1993). The recommendations of the guide have been implemented quite successfully. However, no official recommendations for private wells have been in force in Finland. STUK has proposed a reference concentration of 1000 Bq/l for private wells.

The Drinking Water Directive issued by the European Union in 1998 (Council Directive 1998) included a reference dose of 0.1 mSv per year. Radon and its decay products (including polonium-210 and lead-210) contribute most to the doses received, but they were excluded from the reference dose. In order to implement the directive, the Ministry of Social Affairs and Health in Finland intends to issue new national guidelines for radioactivity in drinking water. It will include both directives for waterworks and recommendations for small units (private wells). Consequently, it is a good time to reassess the occurrence, doses and risks from radionuclides in drinking water in Finland.

### **Materials and methods**

Measurements of radionuclides in drinking water have been carried out in STUK for 25 years at more than 1,000 waterworks, for about 6,000 wells drilled in bedrock and for 4,000 wells dug in soil. The measurements obtained at waterworks result from a comprehensive survey programme in the 1970s and later measurements at new waterworks. The measurements for private wells result from different survey programmes and measurements ordered by private well-owners (Asikainen 1992, Salonen 1994).

Practically all waterworks having more than 200 consumers are investigated. In this analysis, at least one radon measurement was available for 45% of all waterworks, covering 86% of consumers. The respective figures for other nuclides are: uranium-234 and uranium-238 5%

(21%), radium-226 41% (80%), polonium-210 and lead-210 14 % (41%), gross alpha count 45% (86%) and gross beta count 39% (82%). Most of the measurements of nuclides other than radon represent ground waters. In order to obtain representative estimates for water consumed, the proportions of surface and ground water at each individual waterworks were taken into account. The data base of the Finnish Environment Institute (SYKE) was utilised in these calculations. The missing values for radium, uranium, polonium and lead were obtained by using regression analysis with the values of gross alpha, gross beta and radon as arguments. To obtain values for waterworks without any measurements, means from the same type of waterworks were used.

As to private wells, the concentrations of radionuclides in well water were weighted using geographical frequency of wells in municipalities. The numbers of drilled wells were obtained from a representative indoor radon study (Arvela et al 1993), and numbers of both well types from the national rural well survey by SYKE (Korkka-Niemi et al.1993). To obtain values for missing radionuclides, the same kind of procedure was used as with the waterworks.

The age-dependent conversion factors presented by Kendall and his co-workers ( Kendall et al. 1988) were used to calculate the dose from radon, and those given by EU Basic Safety Standards (Council directive 1996) the dose from other radionuclides. Kendall's estimate for adults ( $10^{-8}$  Sv/Bq) is adopted by the publication UNSCEAR 1993 (UNSCEAR 1993). When estimating doses from radionuclides apart from radon, the amount of ingested water was assessed to be 2.2 litres per day for adults and children older than 12 years, and 0.8 litres per day for younger children. Because radon is easily released from water during heating, an effective amount of 0.5 litres of water per day was assessed in order to calculate radon doses for persons of all ages. The age distribution of the Finnish population in 1998 was utilised for weighting.

Radon is easily released from water into indoor air. The ratio of indoor radon concentration caused by radon released into the air and the radon concentration of water is called the transfer coefficient. In this study a value of  $10^{-4}$  was applied (NRC 1998).

## Results

The mean concentrations of radionuclides in different water sources are shown in Table 1.

**Table 1.** Average concentrations of naturally occurring radionuclides in drinking water supplied by waterworks and private wells in Finland.

Nuclide	Waterworks, Bq/l	Wells dug in soil, Bq/l	Wells drilled in bedrock, Bq/l	Weighted mean, Bq/l
Radon-222	27	45	540	46
Uranium-234	0.02	0.02	0.4	0.035
Uranium-238	0.015	0.02	0.25	0.025
Radium-226	0.003	0.01	0.07	0.006
Polonium-210	0.003	0.01	0.06	0.006
Lead-210	0.004	0.04	0.1	0.011

The collective, age-group weighted annual dose received by consumers of water from public waterworks is 350 manSv, to consumers of water from private wells dug in soil 80 manSv and to consumers of water from wells drilled in bedrock 320 manSv. Table 2 shows that the population weighted annual effective dose from drinking water in Finland is 0.14 mSv. Without using age-specific dose-conversion factors the dose would be 0.11 mSv. About 88% of the dose arises from radon, almost 10 % from polonium and lead, and the rest from uranium and radium.

**Table 2.** Age-group weighted annual doses from ingestion of natural radionuclides in drinking water in Finland. The amount of ingested water is assumed to be 2.2 l/day for persons older than 12 years and 0.8 l/day for others. In assessments of radon doses, the amount of 0.5 l/day direct from tap water is assumed for all.

Nuclide	Waterworks, mSv	Wells dug in soil, mSv	Wells drilled in bedrock, mSv	Weighted mean, mSv
Radon-222	0.07	0.12	1.4	0.13
Uranium-234	0.001	0.001	0.02	0.0014
Uranium-238	0.001	0.001	0.01	0.0009
Radium-226	0.001	0.003	0.02	0.002
Polonium-210	0.003	0.011	0.07	0.006
Lead-210	0.003	0.03	0.07	0.007
Total	0.08	0.16	1.6	0.14

The number of consumers exposed to different doses is shown in Table 3. It can be seen that most of the persons receiving more than 0.5 mSv per year use wells drilled in bedrock. The highest annual dose estimated from a single well was 166 mSv.

**Table 3.** Numbers of consumers exposed to different annual ingestion doses from naturally occurring radionuclides in drinking water in Finland, according to water source.

Water source	All consumers	Dose > 0.05 mSv	Dose > 0.1 mSv	Dose > 0.5 mSv	Dose > 1 mSv
Wells drilled in bedrock	200000	200000	160000	80000	40000
Wells dug in soil	500000	300000	150000	25000	4000
Waterworks	4500000	1600000	800000	5000	1000
Total	5200000	2000000	1000000	100000	50000

To assess the number of annual cancer deaths due to exposure to ingested radionuclides in water, the ICRP estimate of a 5% increment of fatal cancer incidences caused by 1 Sv was used. On the other hand, estimates of lung cancer risk caused by inhaled radon are available from direct epidemiological studies (Auvinen et al, 1996). The estimate used in STUK is 100-600 annual lung cancer cases from our mean indoor radon concentration of 120 Bqm<sup>-3</sup>.

The National Academy of Sciences (NAS) is preparing a new risk assessment for radon in drinking water, and their preliminary estimate (NRC 1998) differs from those put forward by Kendall and UNSCEAR 1993. Table 4 shows the discrepancies between cancer incidences in this study and those calculated according to the figures proposed by the NAS.

**Table 4.** Annual cancer cases due to naturally occurring radionuclides in Finland, estimated according to available cancer incidence estimates and dose conversion factors.

Reference	Ingested radon gas	Ingested, long-lived radionuclides	Inhaled water-born radon	Total inhaled radon
NAS (NRC 1998)	5		37	1000
This study	33	5	4-22	100-600

## Discussion and conclusions

The risk estimates of radionuclides in drinking water and indoor radon are important, because of the consequences to the policy and regulations adopted. In the U.S. NAS assessed the risk of ingested radionuclides in water to be negligible compared to the risk of radon released into indoor air. This led EPA to propose a so-called multimedia approach, which gives the water companies an opportunity to meet safety requirements in water by mitigating indoor radon concentrations. On the contrary, the risk estimates of this study, based on quite conservative conversion factors of ingested radon by Kendall and his co-workers, and on the rather modest indoor radon risk estimates used by STUK, result in higher risk from ingested radon than from inhaled, water-born radon. NAS decided not to give separate risk estimates for radon ingested

by children and other susceptible populations, whereas the risk estimates used in this study emphasise the risk to infants and children. A large source of uncertainty in all these figures arises, among other things, from the amounts of tap water consumed by persons of different ages.

According to this study, radon causes the highest doses in drinking water. However, long-lived radionuclides cause 10% of the dose ingested. If the dose conversion factors for radon proposed by NAS were to be adopted, the dose from long-lived daughters would equal that of radon. In any case, polonium-210 and lead-210 are the long-lived nuclides which we should recommend as subjects of further studies.

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# MEASUREMENTS OF AIRBORNE RADON-222, LEAD-210 AND BERYLLIUM-7 IN SOUTHERN FINLAND

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## Introduction

Beryllium-7 ( $t_{1/2} = 53.3$  days) is produced in the upper troposphere and lower stratosphere from atmospheric nitrogen and oxygen nuclei by nuclear interactions with cosmic radiation. The radioactive noble gas  $^{222}\text{Rn}$  ( $t_{1/2} = 3.825$  days) is a decay product of uranium-238.  $^{222}\text{Rn}$  enters the atmosphere by exhalation from the soil and decays to  $^{210}\text{Pb}$  ( $t_{1/2} = 22.3$  years) via several short-lived daughter nuclides. These three natural radionuclides ( $^7\text{Be}$ ,  $^{222}\text{Rn}$ ,  $^{210}\text{Pb}$ ) can be used as tracers in atmospheric processes.

## Materials and methods

The Finnish Meteorological Institute has been monitoring airborne natural radionuclides at its geophysical observatory at Nurmijärvi, southern Finland (60°30'N, 24°39'E,  $h = 105$  m a.s.l.). The activity concentration of  $^{222}\text{Rn}$  in surface air was monitored continuously by measuring its particle-bound short-lived progeny with a dual fixed-filter monitor (Paatero et al., 1994).  $^{222}\text{Rn}$  was assumed to be in equilibrium with its short-lived progeny. Daily aerosol samples (3500 m<sup>3</sup>/d) were collected onto glass-fibre filters. The filters were analysed for  $^7\text{Be}$  with  $\gamma$  spectrometry and for  $^{210}\text{Pb}$  with  $\alpha$  counting of the in-grown  $^{210}\text{Po}$  (Mattsson et al., 1996).

## Results, discussion and conclusions

The hourly  $^{222}\text{Rn}$  and the daily  $^{210}\text{Pb}$  and  $^7\text{Be}$  activity concentrations in the air were lognormally distributed. The lowest concentrations of  $^{222}\text{Rn}$  were observed in spring and early summer because of the efficient mixing of the atmospheric boundary layer with simultaneous reduced exhalation of radon due to the snow cover and frozen ground (Fig. 1). Maximum concentrations were recorded before sunrise in late summer due to the simultaneous strong radon exhalation and frequent nocturnal surface inversions. In winter the concentrations were relatively high with a small diurnal variation due to the low mixing height in the absence of solar radiation. For the same reason the activity concentrations of  $^{210}\text{Pb}$  in the air are high in winter and low in summer (Paatero et al., 1998). The maximum concentrations of  $^7\text{Be}$  are found in late spring and summer due to the tropopause folding events moving air from the stratosphere to the troposphere, and increased vertical mixing of the troposphere.

Air mass back trajectories calculated with the TRADOS model (Valkama et al., 1995) show that the highest  $^{222}\text{Rn}$  and  $^{210}\text{Pb}$  activity concentrations are found in continental air masses coming from southeast (Figures 2-4). The same air masses bring also high amounts of  $^7\text{Be}$  to the surface level air in southern Finland. High  $^{210}\text{Pb}$  and  $^7\text{Be}$  concentrations were also observed during air masses coming from northern Russia. These situations are often connected to high-pressure areas north or northwest of Finland and/or cyclones centered in northern Russia. The production of  $^7\text{Be}$  increases towards the magnetic poles of the Earth. This is not reflected to the source areas of  $^7\text{Be}$  in the surface air in southern Finland, because the concentration variations are due to synoptic-scale weather phenomena.



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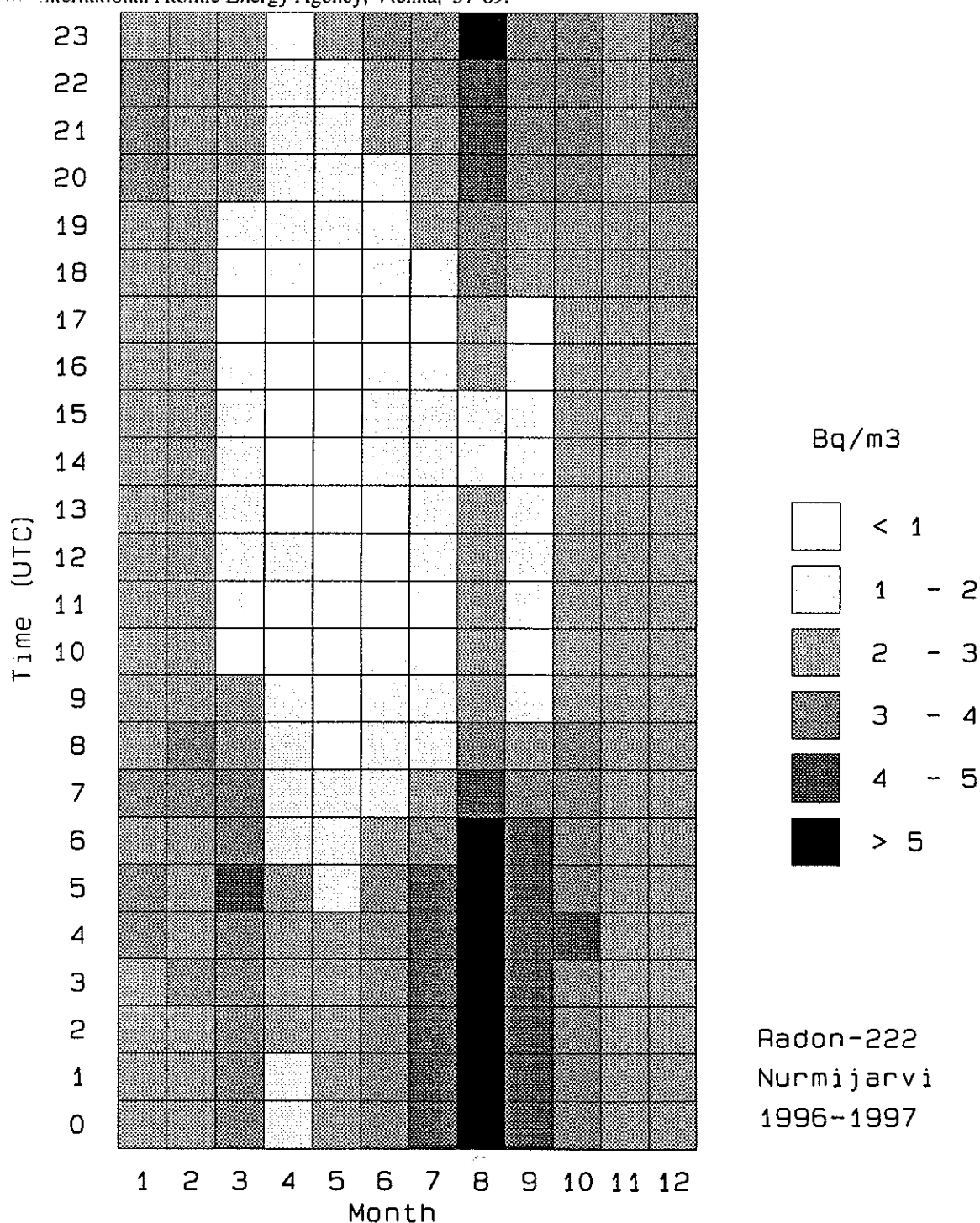


Fig. 1. Diurnal and seasonal variation of  $^{222}\text{Rn}$  in the surface air at Nurmijärvi, Finland, 1.1.1996-31.12.1997.

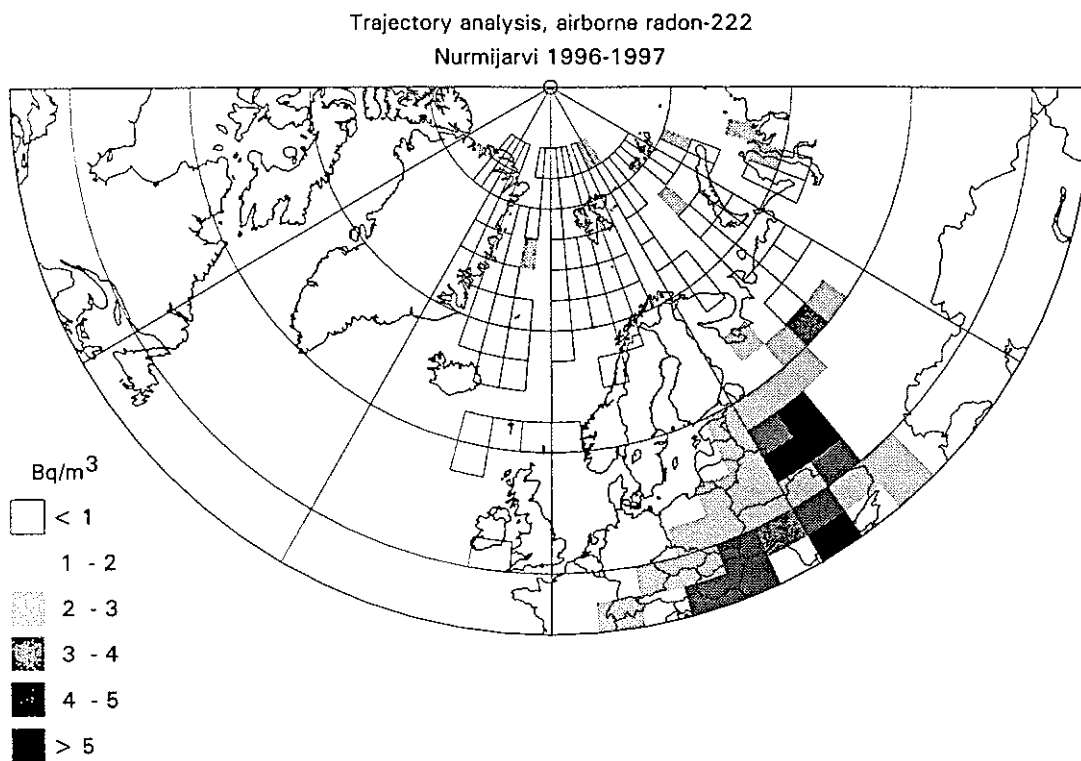


Fig. 2. Trajectory analysis of <sup>222</sup>Rn in the surface air at Nurmijärvi, Finland, X/1996-IX/1997.

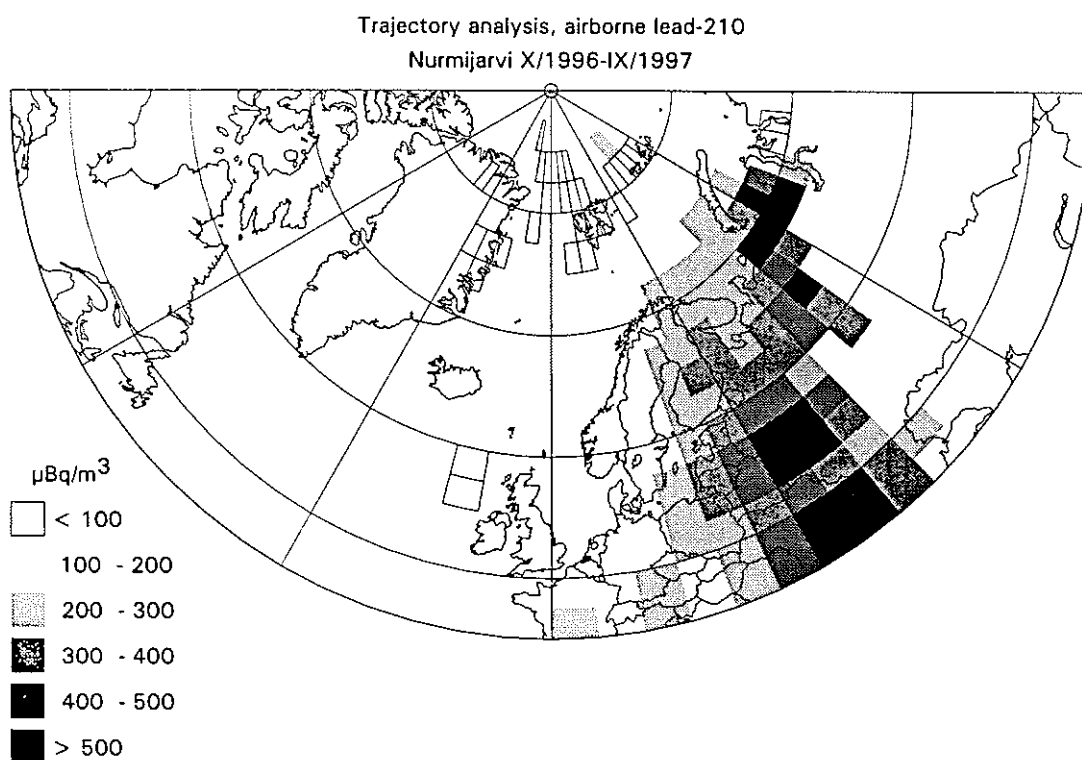
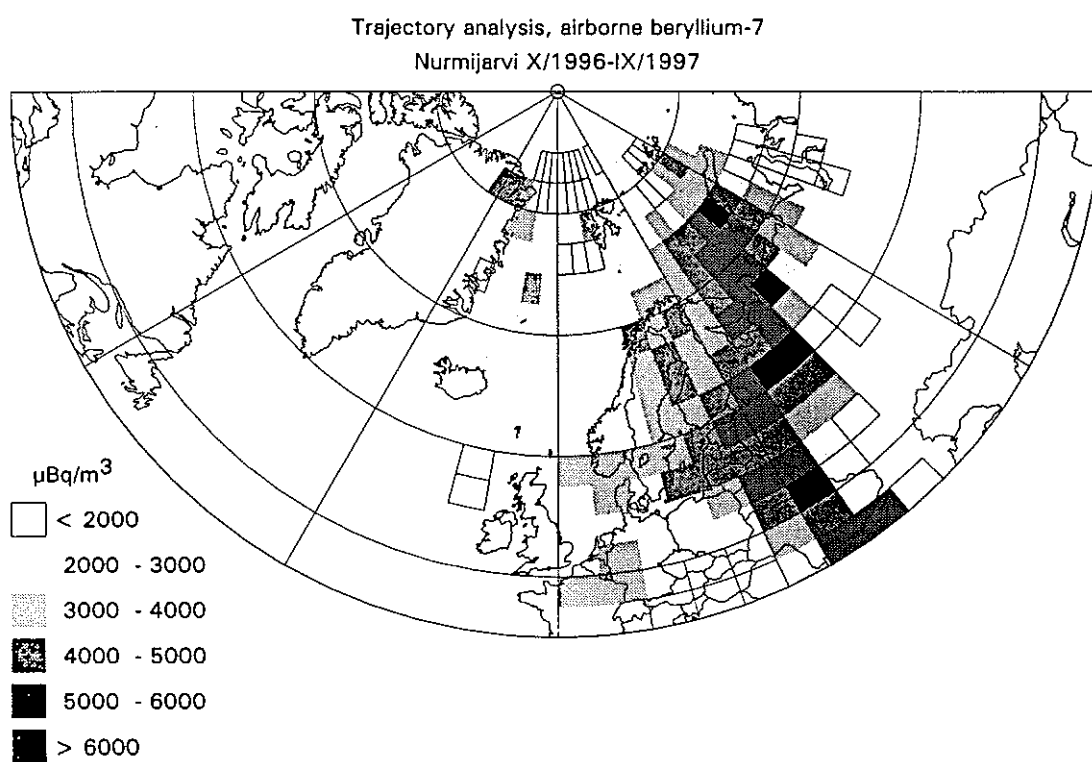


Fig. 3. Trajectory analysis of <sup>210</sup>Pb in the surface air at Nurmijärvi, Finland, X/1996-IX/1997.



**Fig. 4.** Trajectory analysis of <sup>7</sup>Be in the surface air at Nurmijärvi, Finland, X/1996-IX/1997.

# Forekomst av lavradioaktive blyavleiringer i petroleumsindustrien

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## SAMMENDRAG

Lavradioaktive avleiringer som inneholder  $^{210}\text{Pb}$  er blitt påvist i forbindelse med gassproduksjon på norsk sokkel. Blyavleiringer kan transporteres gjennom produksjonssystemet supportert av  $^{222}\text{Rn}$  eller usupportert. Radon supportert transport skjer hovedsakelig i gassfasen og kan forekomme over lange avstander. Usupportert bly transporteres mest sannsynlig i vannfasen og vil dermed opptre på utstyr som har vært i kontakt med produsert vann.

Stråledoser til personell offshore og onshore fra håndtering av komponenter belagt med blyavleiringer vil under normale forhold være lave. Det knytter seg imidlertid en viss risiko til inhalasjon av radioaktive partikler fra avleiringene.

## INNLEDNING

Forekomst av lavradioaktive avleiringer, LRA, har vist seg å representere et økende problem for petroleumsindustrien. Dette skyldes delvis at LRA akkumuleres etter som feltene har vært i drift over en lengre periode. Injeksjon av sjøvann i reservoaret for å utnytte olje- og gassforekomstene optimalt, øker også dannelsen av LRA.

Den vanligste typen LRA, sulfat avleiringer, inneholder radioisotopene  $^{226}\text{Ra}$  og  $^{228}\text{Ra}$ . Aktivitetsnivået i LRA kan variere fra 0 opp til 15 kBq/g. Norske myndigheter har satt friklassingsgrensen til 10 Bq/g for  $^{226}\text{Ra}$  eller  $^{228}\text{Ra}$  eller  $^{210}\text{Pb}$  (Statens strålevern 1997).

Undersøkelser har vist at det også kan forekomme en annen type lavradioaktiv avleiring som inneholder isotopen  $^{210}\text{Pb}$  og dens døtre  $^{210}\text{Bi}$  og  $^{210}\text{Po}$ . Betydningen av blyavleiringer på norsk sokkel kan være underestimert. I tidligere undersøkelser ble det bare påvist små mengder radioaktivt bly, nye studier har imidlertid avslørt verdier som ligger godt over friklassingsgrensen (Ramsøy et al. 1999).

## BLYAVLEIRINGER

Blyavleiringer kan være vanskelig å oppdage. Avleiringene er ofte svært tynne og skaper dermed ingen problem for produksjonen. I enkelte tilfeller opptre blyavleiringer i atomære lag og er dermed ikke synlige. Avleiringene lar seg heller ikke detektere på utsiden å produksjonsutstyret siden strålingen fra  $^{210}\text{Pb}$  eller dens døtre ikke er penetrerende. Heller ikke i kontakt med avleiringen vil vanlige doserateinstrument, som ofte benyttes offshore, kunne detektere strålingen.

Det antas at blyavleiringen kan oppstå på to måter:

1. Supportert av edelgassen  $^{222}\text{Rn}$  ( $T_{1/2}=3,83$  d)
2. Usupportert

Blyavleiringer supportert av  $^{222}\text{Rn}$  oppstår ved transport av radon gass fra nær-brønn sonen. Undersøkelser viser (Rood et al. 1998) at emanasjon av radon fra LRA normalt er lav. Imidlertid kan det antas at diffusjonsraten øker i nær-brønn området hvor det er høyt trykk og høy temperatur. Annen sammensetning av avleiringen kan også bidra til en høyere diffusjonsrate.

Transport av radon gass gjennom produksjonssystemet vil hovedsakelig foregå i gassfasen. Mindre mengder vil følge oljestrømmen, og vannfasen vil gi et ubetydelig bidrag (Bjørnstad et al. 1999).

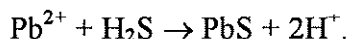
Selve blyavleiringen oppstår ved henfall av  $^{222}\text{Rn}$ , dette skjer mest sannsynlig i områder hvor gassen har lang oppholdstid.

Radon gassen kan transporteres over lange strekninger. Det er dermed en risiko for finne blyavleiringer på anlegg som ligger langt fra produksjonsstedet.

Selve mekanismen for dannelse og transport av  $^{222}\text{Rn}$  supporterte blyavleiringer er relativt godt forstått.

Når det gjelder dannelse og transport av usupporterte blyavleiringer er det imidlertid mangelfull kunnskap om mekanismene. Karakteristisk for denne typen blyavleiring er at  $^{210}\text{Pb}$  alltid finnes sammen med stabilt bly som fungerer som en bærer. I motsetning til Rn supporterte avleiringer skjer transporten hovedsakelig i vannfasen.

Usupporterte blyavleiringer er delvis påvist i form av blyforbindelser som PbS og delvis som rent metallisk bly. Dannelse av PbS (galena) kan forklares ved reaksjonen



Utfelling skjer ved fallende trykk og temperatur under transport opp fra brønnen.

Dannelse av metallisk bly kan imidlertid ikke forklares ved en overmetning  $\rightarrow$  utfellingsreaksjon. Det er foreslått (Hartog et al. 1998) at elementært bly dannes i en ko-reaksjon ved ordinær korrosjon:



## DETEKSJON AV $^{210}\text{Pb}$

Ved henfall av  $^{210}\text{Pb}$  sendes det bare ut svak  $\gamma$ -stråling (46,5 keV, 4,0%) som vanskelig lar seg påvise med vanlige håndinstrument. Datteren  $^{210}\text{Bi}$  sender imidlertid ut  $\beta$ -partikler med maksimal energi 1161 keV, denne strålingen kan lett påvises med en  $\beta$ -følsom monitorer.

Feltmåling av blyavleiringer kan dermed best utføres ved å måle på innsiden av komponenten med en  $\beta$ -probe. Formålet med denne type målinger er først og fremst å klassifisere komponenten til å være over eller under friklassingsgrensen. For Ra avleiringer er det utviklet en metode for å relatere telleraten til spesifikk aktivitet (Ramsøy et al. 1998; Ramsøy et al. 1999). Dette lar seg imidlertid vanskelig gjøre for blyavleiringer på grunn av beleggets varierende beskaffenhet. Det er derfor anbefalt å anse komponenten som over friklassingsgrensen hvis telleraten overstiger to ganger bakgrunnstelleraten.

Laboratoriemåling av avleiringer som inneholder  $^{210}\text{Pb}$  baseres på  $\gamma$ -spektroskopi. På grunn av den lave energien til  $\gamma$ -linjen bør det benyttes en Ge detektor med Be-vindu. Det er videre helt avgjørende å korrigere for selvabsorpsjon i prøven. Dette kan gjøres ved å beregne en absorpsjonskoeffisient ved å utføre målinger med en  $^{210}\text{Pb}$  kalibreringskilde på en standard og på selve prøven.

## FOREKOMST AV BLYAVLEIRINGER

Det finnes lite systematiske data om forekomst av blyavleiringer. I undersøkelser utført på nederlandsk sektor og i gassbehandlingsanlegg er det imidlertid påvist blyavleiringer (Hartog et al. 1998). På rør og plattformutstyr er det påvist usupporterte avleiringer med høy spesifikk aktivitet (opp til 1500 Bq/g) mens det i gassbehandlingsanlegg opptrer avleiringer supportert av radon gass. De sistnevnte har ofte relativt lav spesifikk aktivitet.

I et prosjekt utført på oppdrag fra Oljeindustriens landsforening (OLF) ble det påvist blyavleiringer på komponenter som stammet fra Ula og Gyda plattformene (Ramsøy et al. 1999). Plattformene ligger i Ekofiskområdet og produserer gass med en relativt høy andel produsert vann. Det var tidligere kjent at komponenter og rør var belagt med det tjærelignende stoffet Asphaltén som inneholdt ZnS.

Målinger avslørte blyavleiringer med spesifikk aktivitet fra 5-40 Bq/g av isotopen  $^{210}\text{Pb}$ . Det antas at avleiringen var dannet ved transport av usupportert PbS. På Fig. 1. er spesifikk aktivitet angitt ved forskjellige målepunkt. Blyavleiringene oppstår på utstyr som har vært i kontakt med produsert vann med tilnærmet samme aktivetskonsentrasjon fra brønn til utstyr på plattformdekk. Det ble også påvist radiumavleiringer (24 Bq/g) men bare i området nær brønnen. Dette bekrefter at blyavleiringer lettere transporteres med produksjonsstrømmen enn radiumavleiringer.

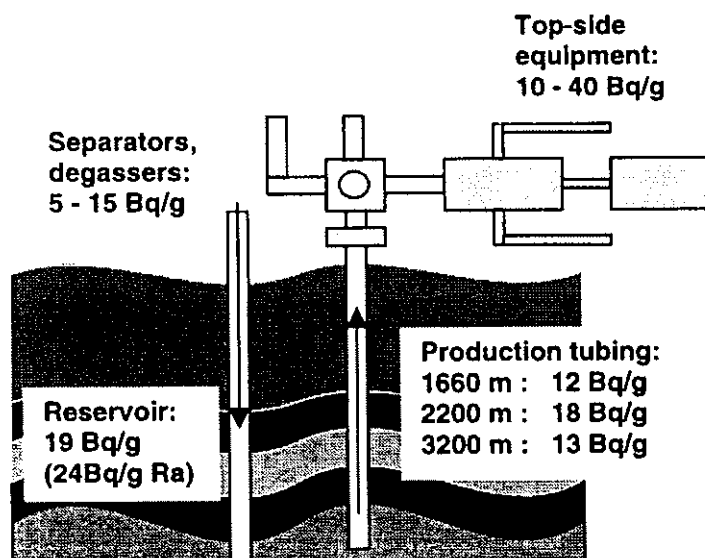


Fig.1 Spesifikk aktivitet i blyavleiringer fra Ula/Gyda plattformene

## STRALEDOSER FRA BLYAVLEIRINGER

Undersøkelser utført på norsk og britisk sektor har vist at årlig eksterndose til personell som arbeider med LRA typisk ligger i området 0,02 mSv til 0,3 mSv (Lysebo et al. 1998). Størst dose får personell ved renseanlegg i land. Det er også gjort beregninger av interndose fra inhalasjon av støvpartikler. De beregnede stråledoser ligger i samme området som eksterndosene.

Når det gjelder doser fra håndtering av blyavleiringer er det ikke utført tilsvarende undersøkelser. Avleiringer som inneholder  $^{210}\text{Pb}$  vil ikke kunne gi opphav til noen

eksterndose. Det er imidlertid en risiko for å bli utsatt for en signifikant dose ved inhalasjon eller oralt inntak av slike avleiringer. Inntak kan imidlertid lett unngås ved bruk av egnet verneutstyr og ved at arbeidsrutinene følges.

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## **FINNISH PRACTICE IN BUILDING RADON-SAFE HOUSES – A SURVEY OF MUNICIPAL AUTHORITIES**

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### **Abstract**

The study was carried out to chart radon prevention practices in municipalities with high indoor radon concentrations. The area under investigation consists of 95 municipalities in the most radon-prone area in Finland. About 75% of the municipalities investigated required or recommended reaction to radon at least in certain areas of the municipality, but only 30% of municipalities recommended such action in the whole area. The most important prevention measure against radon, radon-tight foundation construction, was recommended in only a few municipalities. Only about one fifth of the municipalities verified that the builders had succeeded in radon-safe construction by measuring indoor radon concentration in the new houses.

### **Introduction**

Indoor radon causes 100 - 600 cases of lung cancers each year in Finland, most probably 200. The action level of 200 Bq/m<sup>3</sup> for designing and building new houses has been in force for more than 10 years (Ministry of the Environment 1987). The action level for existing houses was lowered seven years ago and is now 400 Bq/m<sup>3</sup> (Ministry of Social Affairs and Health 1992).

Radon-bearing soil air is the main reason for elevated indoor radon concentrations in Finland. The pressure difference created by the indoor-outdoor temperature activates the soil air to flow into dwellings through the gaps and other leakage routes existing in the foundation structures. The main question in radon-safe building is how to prevent or reduce the flow of soil air through foundation constructions. The Ministry of the Environment published a guide concerning radon-safe building in slab-on-grade houses in 1994. The guide had been tested in the 50 most radon prone municipalities a few years earlier.

Radon-safe construction comprises two steps. First, airtightness should be achieved using bitumen felt and elastic sealants. The aim of sealing work is to achieve an airtight barrier preventing the leakage flow of soil air through the substructure. Second, preparatory perforated piping should be installed. Soil ventilation is a preparatory technique providing a method to prevent soil air leakage in case the sealing work does not bring the indoor radon concentration permanently below 200 Bq/m<sup>3</sup>. In STUK's previous study on radon-safe building it was noted that the failure of the sealing work is the main reason for high indoor radon concentrations (Ravea, Arvela 1997).

### **Materials and methods**

In 1998 STUK investigated (Voutilainen et al. 1998) the practices followed by municipal authorities in radon prevention. The aim was to find out how authorities advise builders and what kind of demands they place on them. The area investigated consists of 95 municipalities in the most radon prone area in Finland, which covers the Province of Southern Finland and

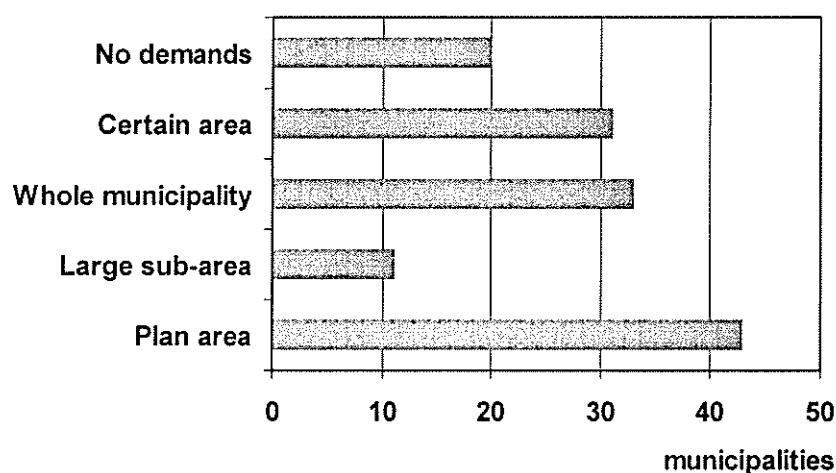


the Council of Tampere Region. The total number of the inhabitants in this area is approximately one million. About 50% of all indoor radon measurements in the study area exceeded 200 Bq/m<sup>3</sup>. The municipal building and health authorities were surveyed by interview. The questionnaire set consisted of seven questions for the building authorities and five for the health authorities. This article deals with certain issues raised by the report.

## Results of the survey

### The regional demands of radon-safe building

About 25% of the municipalities investigated did not recommend radon-safe building at all. Only 30% of municipal authorities recommended radon-safe building in the whole area of the municipality. Figure 1 shows the proportion of radon-safe building required by the municipalities. The answer covers 2 - 3 alternatives in many municipalities.



**Fig. 1.** The figure shows the answers of the health authorities to the question: In which areas is radon-safe building required or recommended in the municipality? The answer "Certain area" usually means an esker area or all eskers in the municipality and the answer "Large sub-area" usually means other soil types than clay or silt.

The main reason for recommending radon-safe building was if there had been indoor radon concentrations exceeding 200 Bq/m<sup>3</sup> measured in the area or in the neighbourhood. More than 10 municipalities applied the limit of 400 or 800 Bq/m<sup>3</sup>. 800 Bq/m<sup>3</sup> is the former limit for existing houses, in force from 1986 to 1992. In six municipalities the requirements were based on radon tests of the ground.

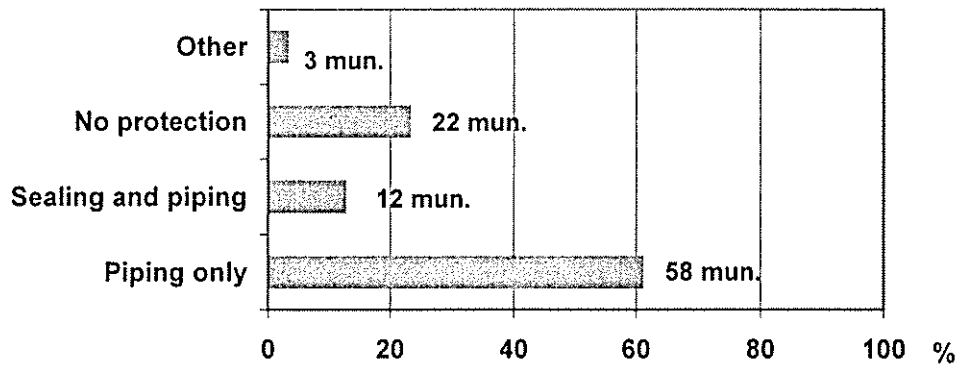
### Knowledge of radon in municipalities

Municipal authorities were quite well informed on the radon situation in their own region. Almost all municipalities used either STUK's indoor radon reports or reports they had made themselves. Only six municipalities lacked radon situation reports of any kind. Four municipalities did not know that STUK had a report especially produced for them.

### Different radon technical measures

Only a few municipalities recommended that builders make airtight construction, although this kind of construction is also a good protection against moisture. The most popular radon-safe

method used by builders was the installation of a piping system in the subsoil of the floor slab. 74% of the municipalities used the piping system. Figure 2 shows the answers of building authorities to the question of the different radon technical measures used.



**Fig. 2.** The figure shows answers of the building authorities to the question: What kind of radon technical measures do they advise the builders to use?

#### Radon technical plans

In 13 municipalities (14%) radon technical plans are required as drawings when applying for a building permit. In two municipalities they are required only for terraced houses. In the remaining 82 municipalities the prevailing practice is that no radon technical drawings are required when applying for a building permit. However, half of them demand that radon prevention measures should at least be mentioned.

The reason for this can be concluded from the open questions made during the telephone interviews. The building authorities felt that the law did not issue clear enough instructions. This is why they were quite cautious when advising builders and they tended to leave the responsibility to them. Furthermore, both the building authorities and the builders felt that the existing guide was too difficult to decipher.

#### Check-up measurements

Most of the municipalities did not monitor whether the builders succeeded in producing radon-safe construction. Only three municipalities (3%) verified the radon concentration in all new houses by a check-up measurement. The houses built in the most radon-prone areas were measured in 15 municipalities (16%). The remaining 77 municipalities did not suggest any radon measurements to builders. Despite this fact, 8 municipalities offered a free radon measurement if the builder was aware that this service was available.

#### Measurements funded by municipalities

In 1986 the municipalities started to make indoor radon measurements according to STUK's measurement plans. The purpose of these measurements was to find out which houses exceeded the action level. The municipalities funded most of the measurements. Over the past few years the number of measurements funded by municipalities has decreased.

In this study only about 38% of the municipalities offered a free radon measurement to private persons. A few municipalities offered a total of one hundred measurements per year. Normally the number of measurements per year was no more than a few dozen.

### **Conclusions and recommendations**

The study showed that the advice given by the municipal authorities to the builders is inadequate. The radon-safe building methods described in the guide published by the Ministry of the Environment should be recommended to all builders in the whole study area regardless of the soil type of the building site. More efficient methods might be needed in the most radon prone-areas.

Indoor radon concentrations are so high in Finland that the limit of 200 Bq/m<sup>3</sup> is most likely to be exceeded in almost the whole country when building new houses. This is why it is practical to build radon safe in all esker areas in the whole of Finland as well as at other building sites in most parts of Finland.

Builders should be given better instructions, both oral and written. The radon technical plans should be presented as drawings when applying for a building permit. Directions concerning the sealing should be written as simply as possible. The guide now in use provides constructors with a good basis for radon technical planning.

The success of radon-safe building should be verified by a check-up radon measurement. STUK recommends that building authorities should be obliged to remind builders to do so. The measurement should be made the winter after the sealing work on the foundation has been completed and the ventilation adjusted to the final conditions.

In many municipalities the health and building authorities co-operate in order to prevent any detriments of radon. However, in many other municipalities the practices of radon-safe building are still in their infancy. Co-operation between the building and health authorities should be enhanced and their obligations, especially in the case of the building authorities, should be clarified.

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## **SESSION VI**

### **ENVIRONMENTAL MONITORING AND RADIOECOLOGY**

**Chairman: Gordon Christensen**



## Plutonium in an Arctic marine environment 29 years after the Thule accident.

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### Abstract

*The nuclear weapons contaminated benthic marine environment in the 180-230 m deep Bylot Sound off Thule Airbase, NW Greenland, was revisited August 1997. Data on water and on brown algae indicates that plutonium from the contaminated sediments is not transported into the surface waters in significant quantities. Sediment core data only indicate minor translocation of plutonium from the accident to the area outside Bylot Sound. The present data support an earlier quantification of the sedimentation rate as 3-4 mm per year, i.e. 8-12 cm during the 29 years since the accident. Biological activity has mixed accident plutonium much deeper, down to 20-30 cm, and the 8-12 cm new sediment have been efficiently mixed into the contaminated layer. In addition to the classical bioturbation efficiently mixing the upper  $\approx 5$  cm, the plutonium data indicates the existence of a deeper bioturbation gradually decreasing with depth. Transfer of plutonium to benthic biota is low leading to lower concentrations in biota than in sediments.*

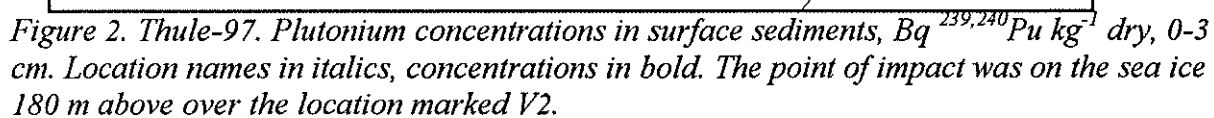
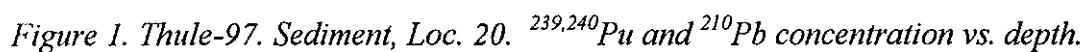
### Introduction

January 1968, a B52 plane carrying 4 nuclear weapons crashed on the sea ice in Bylot Sound off Thule Air Base, northwest Greenland. The bombs were destructed on impact. It has been estimated from earlier sample collections (1968, 1970, 1974, 1979, 1984, 1991) [1,2,3,4,5,6,7], that the pollution remaining in the seabed in Bylot Sound by 1968 amounted to approximately 1.4 TBq  $^{239,240}\text{Pu}$  ( $\sim 0.5$  kg), 0.025 TBq  $^{238}\text{Pu}$ , 4.6 TBq  $^{241}\text{Pu}$  and 0.07 TBq  $^{241}\text{Am}$ .

During August - September 1997 we took a new set of samples onboard the Greenland fisheries investigation vessel 'Adolf Jensen'. The task was devoted to water, sediments and biota. Two different sediment corers were used: most samples were taken with a Finnish 'Gemini' corer delivering 8-cm diameter cores normally divided in 1 cm slices on deck. At many locations, sediment coring was hampered by stones. In those cases the 'HAPS' corer, used during earlier sampling cruises, had a better success rate. HAPS cores without too large stones were divided in 3-cm slices onboard. Longline fishing -aiming at halibut and other large fish - with 4500 squid-baited hooks distributed at 3 locations overnight was unsuccessfull. Some shrimp and starfish were caught with baited crab-cages. Most benthic biota samples were taken by a Sigsbee trawl or a Van-Ween grab. As far as feasible, the caught biota was allowed to purge in clean surface water to reduce the importance of sediments in the guts.

### Plutonium in water

Except for a bottom water sample near the point of impact showing a total concentration of 30 mBq  $^{239,240}\text{Pu m}^{-3}$ , no clear effect of the accident was seen in water or seaweed samples. The general level inside as well as far away from Bylot sound was 5 - 10 mBq  $^{239,240}\text{Pu m}^{-3}$  unfiltered surface water. Plutonium speciation in water has been reported elsewhere [8].



## Sediments

Figure 1 shows an example of  $^{239,240}\text{Pu}$  and  $^{210}\text{Pb}$  concentrations versus depth for a contaminated sediment core, location 20. If the deeper part of the high-concentration layer is assumed to correspond with the accident in 1968, 29 years before the sampling, a sedimentation of 8-12 cm, i.e. 3-4 mm per year, has taken place since then. This corresponds well with  $^{210}\text{Pb}$  dating of earlier cores [9]. The penetration of plutonium to much deeper layers and the absence of very low concentrations in the top layers are both attributed to biological mixing processes performed by the rich benthic community.

Surface 0-3 cm sediment plutonium concentrations are shown in Figure 2. The accident site with the highest concentrations is situated at depth of 180 – 230 meters. It was not possible to catch sediments in areas shallower than 100 m due to rocky bottoms. The two “background sites” outside Bylot Sound, Ny-3 and 1412, are from 500 and 640 m. A surface 0-3 cm concentration of  $0.12 \text{ Bq } ^{239,240}\text{Pu kg}^{-1}$  dry was observed at Shades Øer 750 km further south. The surface concentrations outside Bylot sound is an order of magnitude higher (Fig. 2). Whether this is caused by accident plutonium or it is a natural perturbation caused by differences in sedimentation conditions is not clear. During the sampling expedition much effort was devoted to obtaining additional sediment samples from Upernavik, Melville Bay and the area between Carey Islands and the mainland. The success rate for this effort was very low due to stones in the sediments – even far from land and at large depths. The stones are probably dropped from the numerous icebergs in the area.

## Benthic biota

Where possible, plutonium concentrations in biota samples have been compared with concentrations in 0-3 cm surface sediments (Fig. 2) to give a “concentration Ratio”, CR,  $\text{Bq } ^{239,240}\text{Pu kg}^{-1}$  dry biota /  $\text{Bq kg}^{-1}$  dry sediment (Table 1). It is noted that although most of the biota is living buried in the sediments or on the sediment surface, the CR values indicates, that plutonium is not readily transferred to biota. One single bivalve sample showed a much higher level probably due to a hot particle. In Table 2, average concentrations

*Table 1. Thule-97. Plutonium Concentration Ratios, CR, between biota and upper 3 cm of sediments. Average values, dry weight basis.*

Class	Species	CR, $\text{Bq kg}^{-1}$ biota/ $\text{Bq kg}^{-1}$ sediment		
		Average	SD	n
Mollusc	Bivalves	0.025	0.024	13
Mollusc	Macoma calcaria	37*		1
Mollusc	Snails	0.0033	0.0018	9
Mollusc	Squid, Rossia sp.	0.00036		1
Echinoderm	Starfish	0.0094	0.0139	9
Echinoderm	Brittle stars	0.013	0.0159	4
Echinoderm	Feather stars	0.0070	0.0060	4
Echinoderm	Sea Urchins	0.12	0.16	4
Echinoderm	Sea cucumber	0.0080	0.0083	4
Crustacea	Shrimp	0.0048	0.0088	4
Crustacea	var.	0.038	0.039	4
Annelida	Pectinaria	0.068	0.05234	4
Annelida	var.	0.023	0.033	10
Annelida	Tube	0.28	0.29	6
Fish	Liparis sp.	0.00035		1

\*: Outlier, probably caused by hot particle



Table 2. Thule-97. Average concentrations of plutonium in biota samples and radiation dose\* expressed as the number of kg to be consumed to obtain 1 mSv from  $^{239,240}\text{Pu}$ .

	Bq kg <sup>-1</sup> fresh	SD%	n	kg/mSv	%dry
Bivalves	0.29	205	13	1.40E+04	18.2
Snails	0.13	97	9	3.17E+04	23.2
Starfish	0.94	254	9	4.24E+03	25.8
Brittle stars	0.29	88	4	1.38E+04	45.7
Feather stars	0.07	30	4	5.77E+04	35.0
Sea Urchins	1.57	114	4	2.54E+03	23.8
Sea cucumber	0.05	108	4	7.60E+04	14.9
Shrimp	0.06	141	4	6.90E+04	23.5
Crustaceans	0.18	47	4	2.17E+04	23.5
Pectinaria	4.92	149	4	8.13E+02	29.1
Worms	0.34	107	10	1.19E+04	26.4
Bivalve, outlier	83.14		1	4.81E+01	18.2
Squid, Rossia sp.	0.0047		1	8.49E+05	19.0
Fish, Liparis sp.	0.0363		1	1.10E+05	16.2

\*dosis factor: 2.5 E-07 Sv/Bq

of plutonium in different groups of biota samples are given on a fresh weight basis. Furthermore, radiation doses from a highly hypothetical consumption are given as the number of kg to be consumed to obtain 1 mSv from  $^{239,240}\text{Pu}$ .

### Acknowledgements

The present work was mainly financed by Dancea (Danish Co-operation for Environment in the Arctic). Please note that the content of this paper does not necessarily reflect the views of the Danish EPA. Part of the work, including participation from non-Danish laboratories, was financed under EC contract number FI4P-CT95-0035 (ARMARA).

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## Activity concentration measurements of $^{14}\text{C}$ in the surroundings of two Swedish nuclear power plants

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### Abstract

*In the nature  $^{14}\text{C}$  is continuously produced by cosmic ray bombardment of  $^{14}\text{N}$  in the strato- sphere.  $^{14}\text{C}$  is also produced in nuclear reactors and part of it is released to the atmosphere in the form of  $\text{CO}_2$ . Because of the biological importance of carbon and the long half-life of  $^{14}\text{C}$  it is of interest to measure the releases and to follow their incorporation in living material. Around Forsmark nuclear power plant we have collected sallow leaves and pine tree rings and found an excess of  $^{14}\text{C}$  up to  $25 \pm 5$  Bq/kg C. The  $^{14}\text{C}$  activity concentration, 0-4 Bq/kg C in annual tree rings around Barsebäck nuclear power plant as well as the back- ground  $^{14}\text{C}$  activity levels from two reference sites in southern Sweden during 1973-1997 have been measured. Accelerator mass spectrometry (AMS), as well as decay counting have been used in this investigation.*

### Introduction

$^{14}\text{C}$  is a beta-emitter with a maximum energy of 156 keV and with a relatively long half-life of 5730 years and it is produced in the nature by the cosmic ray bombardment of  $^{14}\text{N}$  in the stratosphere. The production rate is about 1 PBq per year. It is also produced in reactors in the nuclear industry and part of it is released to the atmosphere in the form of  $\text{CO}_2$ , CO or hydrocarbons. In the 1950's and the early 1960's a reserve of  $^{14}\text{C}$  was built up in the atmosphere from the nuclear bomb tests and has been incorporated in living material. This reserve is slowly decreasing while  $^{14}\text{C}$  is also mixed into the oceans. The total activity concentration is now 251 Bq/kg C (1997). The small excess activities emanating from Swedish nuclear power plants are difficult to measure. They can either be measured in the ventilation air at various distances from the reactor, or they can be measured in plants growing around the reactor. If the biological samples are perennial the activity concentration of  $^{14}\text{C}$  will reflect the conditions of the previous season. Several investigations have been made around boiling water (BWR) and pressurised water (PWR) reactors (Levin *et al.*, 1988; Loosli and Oeschger, 1989; Obelic *et al.*, 1986; Uchrin *et al.*, 1992; Uchrin *et al.*, 1998) and only rather small excess activities have been found. In the surroundings of heavy-water reactors and fuel reprocessing plants the excess activities are much higher (Milton *et al.*, 1995; Otlet *et al.*, 1990; McCartney *et al.*, 1986). The aim of this investigation is to find a release factor connecting the release rate of  $^{14}\text{C}$  (Bq/s) and the excess activity (Bq/kg C) in tree rings in trees at various distances from the reactor.

### Sampling sites

*Barsebäck* has two BWR:s, each 600 MW<sub>el</sub>. Tree ring samples of pine (*Pinus*) have been collected in a pine forest 1000 m and 3000 m from the nuclear power plant. The tree rings represent the growing seasons of 1981-97 and 1973-91 respectively.

*Forsmark* has three BWR:s, 970, 970 and 1155 MW<sub>el</sub> respectively Tree ring samples of pine (*Pinus*) have been collected in the forest surrounding the power station at various distances and directions in 1999. Sallow leaves have also been collected. *Måryd* is a "clean air" site about 25 km from Barsebäck where rush (*Juncus*) has been collected.

### Sample preparation and AMS measurements

Annual tree rings were cut from bore cores. This gives about 50-100 mg of wood per year. The samples were pre-treated with an acid-alkali-acid procedure followed by cellulose extraction (Olsson and Possnert, 1992). The AMS measurements were performed at the Lund AMS facility (Erlandsson *et al.*, 1992; Stenström *et al.*, 1995). Each sample consisting of a few mg of carbon is placed in the ion source of the accelerator and the  $^{14}\text{C}$  content is measured. The Måryd samples have been measured by proportional counters (Stenström *et al.*, 1998).

### Results

The transfer factor connecting the activity flow rate (Bq/s) of  $^{14}\text{C}$  in the stack air of the F1 reactor at Forsmark nuclear power station with the activity concentration in the air at the place where the tree rings were collected, was calculated using the continuous point source Gaussian plume model (Slade, 1968) adapted for measurements performed over a longer time period. The activity concentration was integrated over the different wind directions, which have prevailed during the measurements. A homogeneous distribution of wind directions within the individual wind sector was assumed

$$\chi = (2/\pi)^{1/2} (f n Q) (2\pi x \sigma_z u)^{-1} \exp -[h^2 (2 \sigma_z^2)^{-1}]$$

$\chi$	long-term average concentration (Bq/m <sup>3</sup> )	$n$	number of wind sectors
$f$	frequency of wind directions into the wind sector	$Q$	release rate (Bq/s)
$\sigma_z$	vertical standard deviation of the conc. distribution (m)	$x$	distance from source (m)
		$h$	release height (m)
		$u$	mean horizontal air velocity (m/s)

The wind direction data from the weather station Örskär (60°32' N; 18°23' E) 15 km NE of Forsmark were supplied by the Swedish Meteorological and Hydrological Institute. The frequency of the wind blowing towards a 60° sector centred on the tree during the growing season (April to September) was calculated. In 1992 we investigated the activity concentration of the stack air at Forsmark 1 (Stenström *et al.*, 1995). The results together with the stack air flow and the electricity production for the growing season of 1992 are given in Table 1. In column 2 is given the stack air flow activity concentration, in column 3 the release rate (Bq/s) for the time period, in column 4 the wind frequency in the sector (NW-NNW-N-NNE-NE), in column 5 the release rate into the sector (Bq/s), in column 6 the mean wind speed and in column 7 the calculated activity concentration (Bq/m<sup>3</sup>) in the air 2500 m from Forsmark, assuming category D weather (Slade, 1968). During the growing season the release rate into the sector varied from 0.066 to 8.3 10<sup>3</sup> (Bq/s) which gives an average concentration of 6.1 10<sup>-4</sup> (Bq/m<sup>3</sup>). Assuming the same output from all 3 reactors an activity concentration of 18.3 10<sup>-4</sup> (Bq/m<sup>3</sup>) is obtained.

The activity concentration of  $^{14}\text{C}$ , measured with the AMS technique, was 283±5 Bq/kg C in a tree ring from 1992 in a pine tree (*Pinus*) growing 2500 m from Forsmark power station. A background value (clean air site) of 258±2 Bq/kg C was taken from Måryd (about 10 km east of Lund). This gives an excess activity of 25±5 Bq/kg C. Although Måryd is about 600 km from Forsmark it can be used as background because of the small variations in this value over long distances (Stenström *et al.*, 1998).

The transfer factor,  $\tau_{\text{F,el}}$ , for  $^{14}\text{C}$ , from the activity concentration in the air, 18.1 10<sup>-4</sup> Bq/m<sup>3</sup>, at the site of the pine tree, to the tree ring, 25±5 Bq/kg C, is 1.38 10<sup>4</sup> m<sup>3</sup>/kg C. This factor can in, a more general way, be used to calculate the emission of  $^{14}\text{C}$  from the stacks of nuclear power plants if the energy production also is taken into account. During the growing season

of 1992 the energy production of F1 was 2.69 TWh<sub>el</sub>, of F2 2.58 TWh<sub>el</sub> and of F3 3.13 TWh<sub>el</sub>. The total produced energy was 8.30 TWh<sub>el</sub> and the transfer factor,  $\tau_F$ ,  $0.16 \cdot 10^4 \text{ m}^3/\text{kg C}$  per TWh<sub>el</sub>.

**Table 1.** Actual parameters for the reactor Forsmark F1 for 1992.

1992	Act. Conc. at outlet (Bq/m <sup>3</sup> )	Release rate (Bq/s) 10 <sup>3</sup>	Wind frequency	Rel. rate into sec. (Bq/s) 10 <sup>3</sup>	Wind sp. (m/s)	Act. Conc (Bq/m <sup>3</sup> ) 10 <sup>-4</sup>
6/4-21/4	123	15.7	0.53	8.3	7.9	12.5
21/4-4/5	87	11.1	0.29	3.2	4.2	9.1
4/5-18/5	63	8.0	0.25	2.0	6.3	3.8
18/5-1/6	85	10.8	0.21	2.3	6.3	4.4
1/6-15/6	72	9.2	0.29	2.7	6.3	5.1
15/6-29/6	123	15.7	0.35	5.5	6.3	10.1
29/6-13/7	85	18.8	0.34	6.4	7.5	11.8
13/7-27/7	4	0.5	0.13	0.07	7.5	0.1
27/7-10/8	54	6.9	0.19	1.3	4.0	3.9
10/8-24/8	58	7.4	0.23	1.7	7.8	2.6
24/8-7/9	58	7.4	0.28	2.1	7.8	3.2
Mean value				3.23 10 <sup>3</sup>	6.5	6.1 10 <sup>-4</sup>

As both Forsmark and Barsebäck are equipped with boiling water reactors the Forsmark transfer factor,  $\tau_F$  is assumed to be equal to that of Barsebäck  $\tau_F = \tau_B$ . During the growing season of 1997 (April to September) the reactor B1 produced 1.65 TWh and the reactor B2 1.78 TWh, together 3.43 TWh, which gives an transfer factor,  $\tau_{B,el}$  of  $0.56 \cdot 10^4 \text{ m}^3/\text{kg C}$ . The excess of <sup>14</sup>C is very small at Barsebäck power station (Table 2), but for the four years 1987-1990 the mean value is  $4 \pm 4 \text{ Bq/kg C}$ . Applying the transfer factor,  $\tau_{B,el}$ ,  $0.56 \cdot 10^4 \text{ m}^3/\text{kg.C}$  the activity concentration in the air around the Barsebäck pine trees is  $7.27 \cdot 10^{-4} \text{ Bq/m}^3$ , which is 1.7% of the normal <sup>14</sup>C concentration in air and in good agreement with 1.5% which is the excess activity in the pine trees. In order to calculate the release rate from Barsebäck wind data was taken from the weather station Oskarsgrundet NO (55°36' N; 12°51'E). The wind frequency into a 60° centered at the Barsebäck plant was during the growing season 1987 0.33, 1988 0.28, 1989 0.24 and 1990 0.28 with a mean value of 0.28. The mean wind speed was 6 m/s. The release rate for B1 is then 86 kBq/s for the distance 1000 m and 12 kBq/s for 3000 m, with a mean value of  $(49 \pm 30\%) \text{ kBq/s}$ .

According to the monthly report from the Barsebäck nuclear power station the release rate for December 1997 was 24 kBq/s for B1+B2. With regard to the energy production in the two reactors the release rate for B1 is 26 kBq/s and for B2 22 kBq/s. According to the calculations above based on activity concentration measurements in the tree rings the calculated release rate is a factor of 2 higher than the reported.

### Acknowledgements

This investigation was financially supported by the Swedish Radiation Protection Institute. Project SSIP 928.95.

**Table 2.** Comparison between the  $^{14}\text{C}$  content in tree rings of *Pinus* located 1 km and 3 km from the Barsebäck nuclear power plant and the “clean air” site at Måryd (*Juncus*). The excess activity at Barsebäck is the mean excess of the activity at Barsebäck compared to Måryd.

Year	Barsebäck 3 km <i>Pinus</i> (Bq/kg C)	Barsebäck 1 km <i>Pinus</i> (Bq/kg C)	Barsebäck Mean <i>Pinus</i> (Bq/kg C)	Måryd <i>Juncus</i> (Bq/kg C)	Barsebäck Excess act. (Bq/kg C)
1981	296±5	293±4	295±3	287±2	8±4
1982	280±4	282±9	281±5	283±2	-2±5
1983	279±4	271±5	275±3	279±2	-4±4
1984	277±4	276±5	277±3	276±2	1±4
1985	274±5	275±3	275±3	272±3	3±4
1986	277±4	269±5	273±3	269±2	4±4
1987	274±4	271±6	273±3	266±2	7±4
1988	262±5	270±5	266±4	262±2	4±4
1989	263±5	264±9	264±5	262±2	2±5
1990	265±4	265±6	265±4	261±2	4±4
1991	263±4	256±4	260±3	260±2	0±4
1992	263±5	262±3	262±3	258±2	4±4
1993	258±4	257±3	258±3	254±2	4±4
1994	252±4	260±4	256±3	254±2	2±4
1995	249±3	254±3	252±2	252±2	0±3
1996	252±4	251±4	252±2	251±2	1±4

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# Migration and chemical extractability of $^{137}\text{Cs}$ and $^{90}\text{Sr}$ in Swedish long-term experimental pastures

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## Abstract.

Vertical migration and chemical extractability were studied on two experimental pastures, a sandy soil and a clay soil, contaminated with  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  in 1961. Migration was studied by measuring the total nuclide content in soil samples to 55 cm depth, chemical extractability by sequential extraction. Both  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  were found at all depths in both soils, and  $^{90}\text{Sr}$  had moved deeper than  $^{137}\text{Cs}$ .  $^{137}\text{Cs}$  was mainly found in the upper 10 cm, and no difference was seen between clay and sandy soil, due to retention in the root mat of the sandy soil.  $^{90}\text{Sr}$  had moved deeper in sandy than in clay soil, the reason being the higher exchange capacity of the clay soil. Sequential extractions showed that  $^{90}\text{Sr}$  was much more extractable than  $^{137}\text{Cs}$ . 96-98% of  $^{137}\text{Cs}$  was found in the  $\text{HNO}_3$  and residual fractions.  $^{137}\text{Cs}$  was more extractable in the sandy soil than in clay, and in the sandy soil the extractability of  $^{137}\text{Cs}$  increased slightly with depth. Regarding  $^{90}\text{Sr}$ , 63-75% was found in the easily exchangeable fractions, the higher figure was in clay soil. A large proportion, 18-30%, was extracted in the reducible fraction, showing that availability of  $^{90}\text{Sr}$  might increase under reducing conditions.

## Introduction

Understanding the behaviour of radionuclides in different soil types is important for estimating the long-term consequences of a fallout. Two of the most important factors are the migration of radionuclides down through the soil profile and the physico-chemical form of the nuclides, as can be estimated through chemical extractions.

Migration of the radionuclides  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  in undisturbed soils have been studied on many occasions (Haak et al. 1973; Kagan & Kadatsky 1996; Rosén et al. 1999) and it has been shown that, generally,  $^{137}\text{Cs}$  migrates less than  $^{90}\text{Sr}$ , and that both nuclides migrate more in coarse soils than in fine-textured soils. Most earlier studies have been conducted less than ten years after contamination, showing that migration is most rapid during the first two or three years, equilibrium then being established (Kirchner & Baumgartner 1992).

Chemical extractability has been studied less, but results from the studies show that a few years after contamination  $^{137}\text{Cs}$  is strongly bound to the soil matrix, whereas  $^{90}\text{Sr}$  is much easier to extract (Riise et al., 1990; Vidal et al. 1993; Askbrant et al. 1996).

The aim of this study was to investigate the distribution and extractability of radionuclides in an old field experiment, to see if the standard conclusions are valid even after extended ageing of the nuclides.

## Materials and Methods

### Experimental sites

Two experimental pastures in the vicinity of Uppsala were used in the study, a neutral clay soil, and an acid sandy soil (Table 1). The sandy soil is a permanent pasture that has never been ploughed, the clay soil was established as a grazing ley in 1950. In 1961, the former fertilisation experiment was converted into a radioecological experiment and subplots at the two sites were contaminated with either  $^{137}\text{Cs}$  or  $^{90}\text{Sr}$  in water solution. The levels were 10  $\text{MBq/m}^2$  and 3.7  $\text{MBq/m}^2$ , respectively. The  $^{137}\text{Cs}$  plots on the sandy soil and half the plots on the clay soil were recontaminated with 12  $\text{MBq/m}^2$  in 1964, due to low activities in plant material. The clay soil was harvested annually until 1984, the sandy soil was still in experimental use, being both fertilised and harvested annually until 1997. The original experimental design is given by Haak et al. (1973).

Table 1. Soil data.

	Cs-plots No	Sr-plots No	Org mtr <sup>1</sup> (%)		pH H <sub>2</sub> O	CEC <sup>2</sup>	K <sup>2</sup>	Ca <sup>2</sup>
Sandy soil	4	4	11-21	7-8	5.0-6.1	14.3	0.4	12.1
Clay soil	4	2	14	9	6.2-6.3	26.0	0.6	21.5

<sup>1</sup> Loss on ignition<sup>2</sup> In 0.02M BaCl<sub>2</sub>, unit = m.e./100 g soil

### Soil sampling

Soil samples were taken with an auger to 55 cm depth and sliced into 5 cm sections. In each plot three samples were taken and the 5 cm sections from corresponding depths were pooled into a composite sample. For <sup>137</sup>Cs there were 4 replicates in each soil, for <sup>90</sup>Sr there were 4 replicates in the sandy soil and 2 in the clay soil. After drying, the soils were homogenised and sieved (2 mm). All samples, except two Cs-samples in clay soil, were taken in 1997, the remainder in 1998.

### Analyses

Chemical extractability was determined with a five-step sequential extraction method (Tessier et al. 1979; Riise et al. 1990). The extractants were: H<sub>2</sub>O (distilled), NH<sub>4</sub>Ac (1M), NH<sub>2</sub>OH·HCl (0.04M), H<sub>2</sub>O<sub>2</sub> (30%) and HNO<sub>3</sub> (9M). The residual fraction was determined by subtraction from the total content. The total content of <sup>90</sup>Sr was determined by extraction with 9M HNO<sub>3</sub> (hot, 20 h).

<sup>137</sup>Cs was determined using a high purity Ge-detector. Radiochemical separation of <sup>90</sup>Sr was carried out using a crown-ether liquid-liquid extraction method (Tormos et al. 1995). Counting was done on the daughter nuclide <sup>90</sup>Y after ingrowth and separation. The equipment was a low background GM detector, Risø GM-25-5.

## Results and Discussion

### Migration

<sup>137</sup>Cs was found at all investigated depths in both soil types. The activity was highest in the two upper layers and then decreased rapidly with depth (Table 2). Especially in the upper soil layers, the variability between replicates was very large. The difference in activity concentration between soils is due to the lesser contamination in two of the clay soil plots. All activities are normalised to 1996-06-30.

Table 2. Activity concentrations of <sup>137</sup>Cs and <sup>90</sup>Sr. kBq/kg ± SEM.

cm	<sup>137</sup> Cs		<sup>90</sup> Sr	
	Sandy soil	Clay soil	Sandy soil	Clay soil
0-5	84.9 ± 11.7	34.3 ± 6.3	2.9 ± 0.6	4.3 ± 0.2
5-10	59.7 ± 19.0	31.8 ± 6.4	3.1 ± 0.6	5.0 ± 0.2
10-15	20.6 ± 4.4	15.0 ± 5.1	2.9 ± 0.5	3.7 ± 0.4
15-20	8.7 ± 2.4	4.7 ± 2.2	1.8 ± 0.4	2.0 ± 0.3
20-25	3.8 ± 2.0	2.3 ± 0.8	-	-
25-30	1.5 ± 0.7	1.0 ± 0.4	-	-
30-35	0.9 ± 0.4	0.5 ± 0.3	0.9 ± 0.7	0.3 ± 0.0
35-40	0.6 ± 0.2	0.3 ± 0.1	-	-
40-45	0.6 ± 0.4	0.1 ± 0.0	0.8 ± 0.1	0.1 ± 0.0
45-50	0.2 ± 0.1	0.2 ± 0.1	-	-
50-55	0.1	0.1 ± 0.0	-	-

Approximately 70% of the <sup>137</sup>Cs was found within the upper 10 cm, and more than 10% was found below 15 cm (Figure 1). There was no difference in distribution patterns between the sandy and the clay soil, contrary to most previous reports. However, Haak et al. (1973) reported from a sampling of the same sites in 1967 that there was considerably less migration of <sup>137</sup>Cs in the sandy soil than in the clay soil. This might well be due to retention of nuclides

in the dense root mat in the old pasture on the sandy soil, compared with the more evenly distributed roots in the more recently established pasture on clay soil.

Also  $^{90}\text{Sr}$  was found at all investigated depths, but the activities were much lower, due to lower levels of contamination, and differences between soils were much larger (Table 2).

To achieve the distribution graph, we interpolated values at the lower levels using weighted means (Figure 2). In the sandy soil, only 27% of  $^{90}\text{Sr}$  is found in the upper 10 cm, in the clay soil 47% is found in the same layer. This difference is significant ( $p < 0.05$ ).

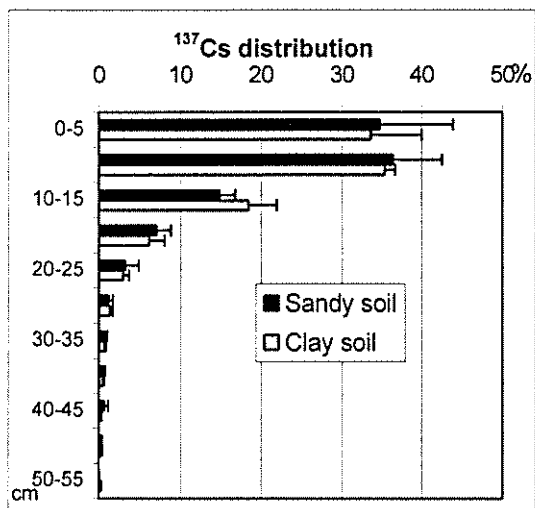


Fig. 1.  $^{137}\text{Cs}$  activity distribution in per cent of the total inventory. Error bars show SEM.

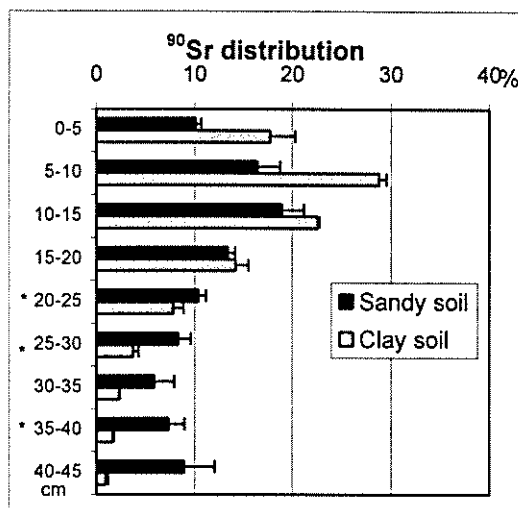


Fig. 2.  $^{90}\text{Sr}$  activity distribution in per cent of the total inventory. \* Depths with interpolated values. Error bars show SEM.

The recovery of the original deposition was 49-52% for  $^{90}\text{Sr}$  and 53-58% for  $^{137}\text{Cs}$ . About 15% of  $^{90}\text{Sr}$  and 5% of  $^{137}\text{Cs}$  may have been taken up and removed with plants since contamination. In sandy soil, some  $^{90}\text{Sr}$  might have migrated below 50 cm, but in clay soil this is improbable, and the same applies to  $^{137}\text{Cs}$  in both soils. We cannot explain the loss of 35-40% of the original deposition.

#### Chemical extractability

The chemical extractability of  $^{137}\text{Cs}$  was low in both soils. Only a few per cent of the total amount was found in the first four fractions ( $\text{H}_2\text{O}$ ,  $\text{NH}_4\text{Ac}$ ,  $\text{NH}_2\text{OH}\cdot\text{HCl}$ ,  $\text{H}_2\text{O}_2$ ), and the remaining part, 96-98%, was evenly split between  $\text{HNO}_3$  and the residual fraction (Figure 3a). Analysis of variance showed that there was significantly ( $p < 0.05$ ) more  $^{137}\text{Cs}$  in the  $\text{NH}_4\text{Ac}$  and  $\text{HNO}_3$  fractions in the sandy soil, and, correspondingly, more  $^{137}\text{Cs}$  in the residual fraction in the clay soil. The reason is that  $^{137}\text{Cs}$  is irreversibly fixed in clay particles, and then cannot be extracted even with strong acid. Also, in the sandy soil, the  $\text{HNO}_3$  fraction increases and the residual fraction decreases with depth. This might indicate that  $^{137}\text{Cs}$  is more mobile in sandy soil than in clay soil, and hence that migration could still be continuing in this soil.

The extractability of  $^{90}\text{Sr}$  was high in both soils, and there was no residual fraction in either of them (Figure 3b). The easily exchangeable fraction ( $\text{H}_2\text{O}$  and  $\text{NH}_4\text{Ac}$ ) was 63% to 75%, indicating the high availability of  $^{90}\text{Sr}$  for plant uptake. All fractions differ significantly between the soils. The  $\text{H}_2\text{O}$ ,  $\text{NH}_2\text{OH}\cdot\text{HCl}$  and  $\text{H}_2\text{O}_2$  fractions are larger in the sandy soil, whereas the  $\text{NH}_4\text{Ac}$  and  $\text{HNO}_3$  fractions are larger in the clay soil. The clay soil has a cation exchange capacity twice that of the sandy soil, which explains the considerably larger exchangeable fraction. The sandy soil has both a higher organic matter content and a higher content of iron oxides, which explains the larger oxidizable and reducible fractions, respectively. There were no differences in  $^{90}\text{Sr}$  extractability with depth.



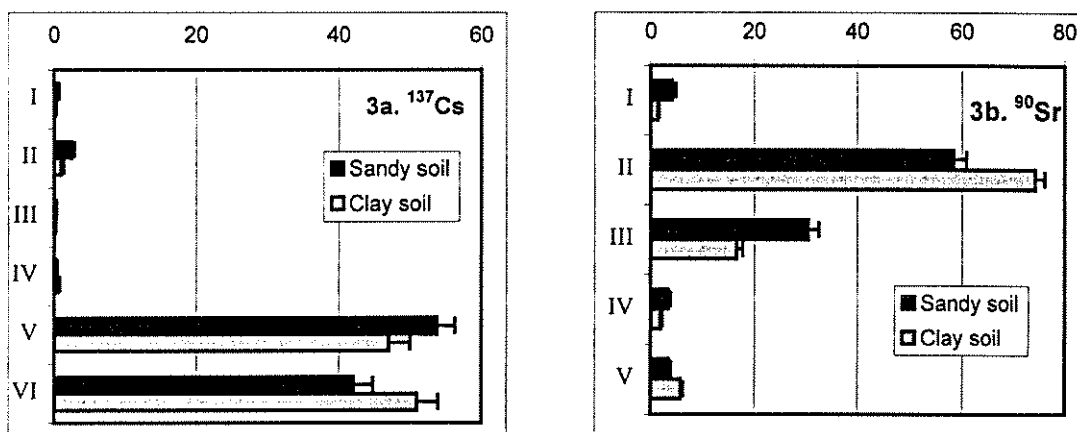


Fig. 3 a & b. Chemical extractability of a:  $^{137}\text{Cs}$  and b:  $^{90}\text{Sr}$ . Error bars show SEM. I= $\text{H}_2\text{O}$ , II= $\text{NH}_4\text{Ac}$ , III= $\text{NH}_2\text{OH}\cdot\text{HCl}$ , IV= $\text{H}_2\text{O}_2$ , V= $\text{HNO}_3$  and VI=residual.

In conclusion, in the 36 years since contamination,  $^{137}\text{Cs}$  has not migrated out of the root zone. Practically all of the nuclide content is contained in the least exchangeable or residual fractions. However, the residual fraction decreases with depth in sandy soil, indicating that migration may still occur.  $^{90}\text{Sr}$  has migrated considerably in both soils, but more in the sandy soil, as could be expected. Almost all  $^{90}\text{Sr}$  is found in the exchangeable fractions. There is also a large proportion in the reducible fraction, showing that  $^{90}\text{Sr}$  is readily bound to sesquioxides. This fraction may become available if reducing conditions occur, *e.g.* after prolonged wetting of the soil.

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# Bioaccumulation of Cs-137 and Co-57 by Marine Phytoplankton

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## ABSTRACT

Under controlled laboratory conditions we have examined the bioaccumulation of Cs-137 and Co-57 in three prymnesiophytes, the coccolithophorid *Emiliana huxleyi* and the non-calcareous species *Isochrysis galbana* and *Phaeocystis globosa*, and two diatoms *Skeletonema costatum* and *Thalassiosira pseudonana*. We measured uptake in growing and non-growing cells, and determined concentration factors on both volume and dry weight basis. For Co-57 uptake in non-growing cells, volume concentration factors (VCF) at equilibrium ranged from  $0.2 \cdot 10^3$  for *Emiliana huxleyi* to  $4 \cdot 10^3$  for the diatom *Thalassiosira pseudonana*. For Cs-137 uptake in non-growing cells the VCFs were close to zero. The results suggest that, in contrast to Co, the cycling and bioaccumulation in animals of Cs in marine systems is unlikely to be affected by primary producers.

## INTRODUCTION

The artificial radionuclides Cs-137 and Co-60 are important components of radioactive wastes discharged into the coastal waters of Northwest Europe. The European reprocessing plants at Sellafield and La Hague discharge radioactive wastes into the Irish Sea and the English Channel, respectively. Oceanic currents transport these wastes into the North Sea where they mix with waters from the Baltic Sea containing significant amounts of Cs-137 from Chernobyl fallout. The currents then pass along the Norwegian coast and enter the Barents Sea, where Norway has large investments in fisheries. Knowledge of bioaccumulation and various transport routes of radionuclides through marine food webs in these waters is therefore of special interest. Phytoplankton lie at the base of most marine food webs and could serve to introduce radionuclides into these food webs. This paper presents the results of laboratory experiments in which we studied the bioaccumulation of Cs-137 and Co-57 (as an analogue of Co-60) in five phytoplankton species, all of which can occur in the Barents Sea phytoplankton community.

## MATERIALS AND METHODS

We used axenic cultures of *Emiliana huxleyi* (Prymnesiophyceae), *Isochrysis galbana* (Prymnesiophyceae), *Phaeocystis globosa* (Prymnesiophyceae), *Skeletonema costatum* (Bacillariophyceae) and *Thalassiosira pseudonana* (Bacillariophyceae). All cultures were maintained aseptically in f/2 medium (Guillard and Ryther 1962) minus EDTA, Cu and Zn, prepared with sterile-filtered Southampton surface water (0.2  $\mu\text{m}$  pore size filter). To determine the bioaccumulation of the radioisotopes we basically followed experimental protocols described in Fisher et al. (1983). Briefly, we performed two experiments under

identical experimental conditions, one on non-growing cells and one on growing cells. Growing cells were exposed to cyclic light (L:D 14:10) provided by cool-white fluorescent lamps ( $170 \mu\text{Einst m}^{-2}\text{s}^{-1}$ ). Non-growing cells were kept in darkness to prevent growth. Both growing and non-growing cells were kept at  $12^\circ \pm 1^\circ\text{C}$ . In both experiments, growth was checked by counting cells microscopically with a hemacytometer. Radioisotope additions were in microliter quantities in 0.1 N HCl (Co-57) and 1.0 N HCl (Cs-137). The pH in the seawater solutions was unaffected by the additions. The activities of the isotopes added in both experiments were about 40 Bq/ml for Co-57 and about 300 Bq/ml for Cs-137. Over a six-day period, cells were periodically filtered onto Nuclepore polycarbonate membranes and the fraction of radioactivity associated with cells was determined for each isotope at each sample time (Fisher et al. 1983). Blanks that contained only media and isotopes were treated in the same manner and corrections were made for isotope sorption to the filters.

## RESULTS AND DISCUSSION

The results show that Co-57 is systematically taken up by all the species investigated, as illustrated in figs. 1-3. For growing cells (fig. 1 a), the uptake of Co-57 is highest in *E. huxleyi*, for which it reaches 1 mBq/cell. For non-growing cells, however, the uptake of Co-57 is highest in *T. Pseudonana* (fig. 1 b), whilst *E. huxleyi* in this experiment is taking up the lowest amounts of Co-57. Figure 1 b also show that there is a saturation in the uptake for non-growing cells after approx. 100 hours. The initial uptake in non-growing cells is slightly higher than the uptake in growing cells for some of the species. For Cs-137 however, the uptake is generally much lower in both growing and non-growing cells (figs. 1 c-d), and for

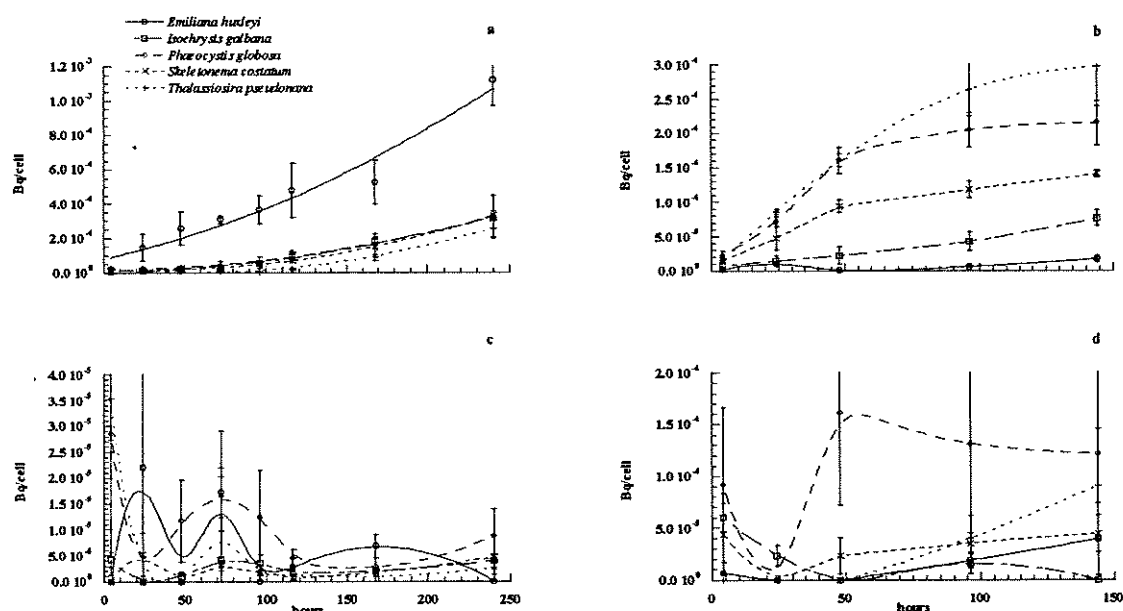


Fig. 1. a. Bq Co-57/cell, growing cells, b. Bq Co-57/cell, non-growing cells, c. Bq Cs-137/cell, growing cells, d. Bq Cs-137/cell, non-growing cells.

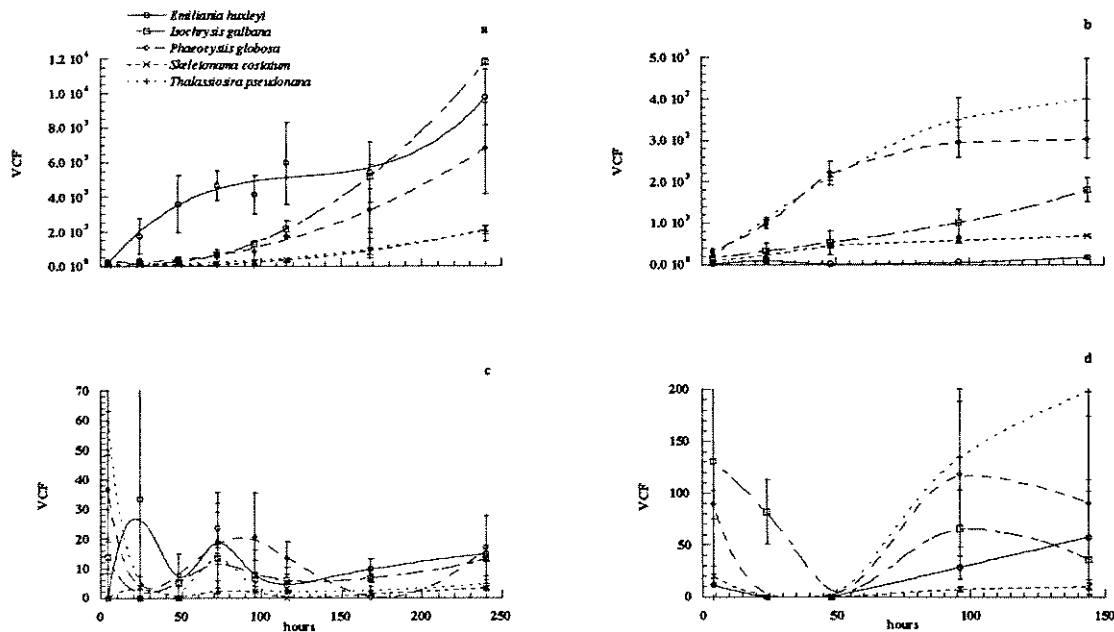


Fig. 2. a. VCF Co-57 (Volume Concentration Factors, radioactivity  $\mu\text{m}^{-3}$  cell divided by radioactivity  $\mu\text{m}^{-3}$  dissolved in water), growing cells, b. VCF Co-57 non-growing cells, c. VCF Cs-137, growing cells, d. VCF Cs-137, non-growing cells.

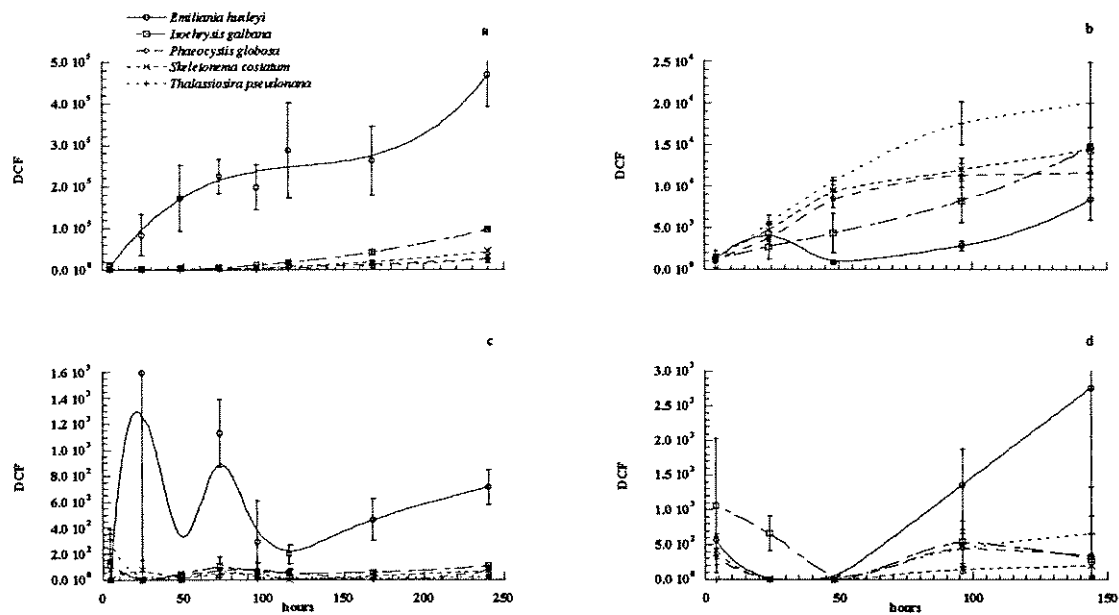


Fig. 3. a. DCF Co-57 (DCF = Dry weight Concentration Factors, radioactivity/g cell divided by radioactivity/g dissolved in water), growing cells, b. DCF Co-57 non-growing cells, c. DCF Cs-137, growing cells, d. DCF Cs-137, non-growing cells.

some of the species it is not significantly different from the blanks. It is difficult to see any significant differences in the uptake of Cs-137 between the species from these data. The volume-volume concentration factors for Co-57 (VCFs = radioactivity  $\mu\text{m}^{-3}$  cell divided by radioactivity  $\mu\text{m}^{-3}$  dissolved in water) vary from approximately  $0.2 \times 10^3$  for *E. huxleyi* to  $4 \times 10^3$  for *T. pseudonana* for non-growing cells (fig. 2 b). The VCFs for the growing cells

reached generally higher values, from  $2 \cdot 10^3$  for *S. Costatum* and *T. pseudonana* to  $12 \cdot 10^3$  for *I. galbana* (fig. 2 a). The VCF data also show saturation after approx. 100 hours for the non-growing cells, whereas the growing cells do not show any saturation since the uptake will increase while total cell volume of growing cells increases. The DCF data (fig. 3 a-d) give similar indications as the VCF data do, we therefore conclude that the volume to dry weight ratio is nearly constant for the phytoplankton species investigated. Our results are generally comparable to other results (Dahlgaard 1994; IAEA 1985).

Cs, an alkali metal, is a structural analogue of K, and could be thought to exchange with K and in this way enter the food chain. However, K is in excess in seawater (10 mM) and the Cs concentration ( $2 \cdot 10^{-6}$  mM), is too small to compete with K. Even though, the concentration of stable Cs in seawater is large enough to dilute the radioactive Cs, and this will also contribute to a decrease in the uptake of radioactive Cs. Thus our results that show low Cs concentrations in phytoplankton are therefore not surprising. Our results indicate that phytoplankton are unlikely to participate in the Cs build-up in marine food chains and Cs flux to deep waters. The bioconcentration of Co in phytoplankton is comparable to earlier findings; overall, Co shows moderate enrichment in phytoplankton relative to other metals (Fisher 1986).

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## **Radioecological Sensitivity: Danish Fallout Data Revisited**

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### **Abstract**

Danish fallout data covering four decades are interpreted in terms of radioecological sensitivity. The radioecological sensitivity is the time-integrated radionuclide concentration in an environmental sample from a unit ground deposition (e.g. Bq y kg<sup>-1</sup> per Bq m<sup>-2</sup>). The fallout data comprise observed levels of the radionuclides <sup>137</sup>Cs and <sup>90</sup>Sr in precipitation, grass, milk, beef and diet. The data are analysed with different types of radioecological models: traditional UNSCEAR models and more recent dynamic models. The traditional models provide empirical relationships between the annual fallout from precipitation and the annual average levels in grass, milk, beef and diet. The relationships may be derived from spreadsheet calculations. ECOSYS and FARMLAND represent more recent radioecological models, which are available as software for personal computers. These models are more mechanistic and require information on a range of topics, e.g. mode of deposition, nuclide dependent and nuclide independent parameters. The more recent models do not reproduce the fallout data better than the traditional models. But the general features of the more recent models make them suited for prediction of radiological consequences of routine and accidental releases in areas where limited radioecological data are available.

The work is part of the NKS/BOK-2.1 project on Important Nordic Food Chains aiming at characterising radioecological sensitivity and variability across the Nordic countries.

### **Introduction**

Data on environmental radioactivity in Denmark have been collected at Risø National Laboratory since the late 1950's covering abiotic samples, vegetation, foodstuffs and humans (Aarkrog et al., 1995, and previous reports). This collection has generated time series of data that illustrate the transfer of anthropogenic radioactivity through foodchains to the Danish population from a variety of sources including fallout from atmospheric nuclear weapons testing and from the Chernobyl accident, and discharges from European nuclear reprocessing facilities. The information obtained from the analysis of these data is useful in connection with predicting radiological consequences from potential nuclear accidents involving radioactive contamination of the Danish environment and for identifying effective countermeasures.

The radioecological sensitivity (Aarkrog, 1979) is a useful concept for characterising the transfer of radioactive contamination to different environmental compartments including man. The time-integrated dose rate to man is the relevant end point of a radiological assessment. Therefore the radioecological sensitivity is

defined as the time-integrated concentration in an environmental sample from a unit ground deposition (e.g.  $\text{Bq y kg}^{-1}$  per  $\text{Bq m}^{-2}$ ).

The data on environmental radioactivity are analysed with different radioecological models in order to obtain numerical values of radioecological sensitivities and to provide a comparison between models.

### Fallout Data and Models

The data used here comprise observed levels in Denmark of the radionuclides  $^{90}\text{Sr}$  and  $^{137}\text{Cs}$  in precipitation, grass, cow milk, beef and diet. The data are given for Jutland, the western part of the country, and for the Islands, the eastern part. Basic references for the data are the time series of annual fallout shown in Fig. 1 for  $^{90}\text{Sr}$  and  $^{137}\text{Cs}$  in Jutland. Figure 2 shows the time series for  $^{90}\text{Sr}$  and  $^{137}\text{Cs}$  in milk from Jutland.

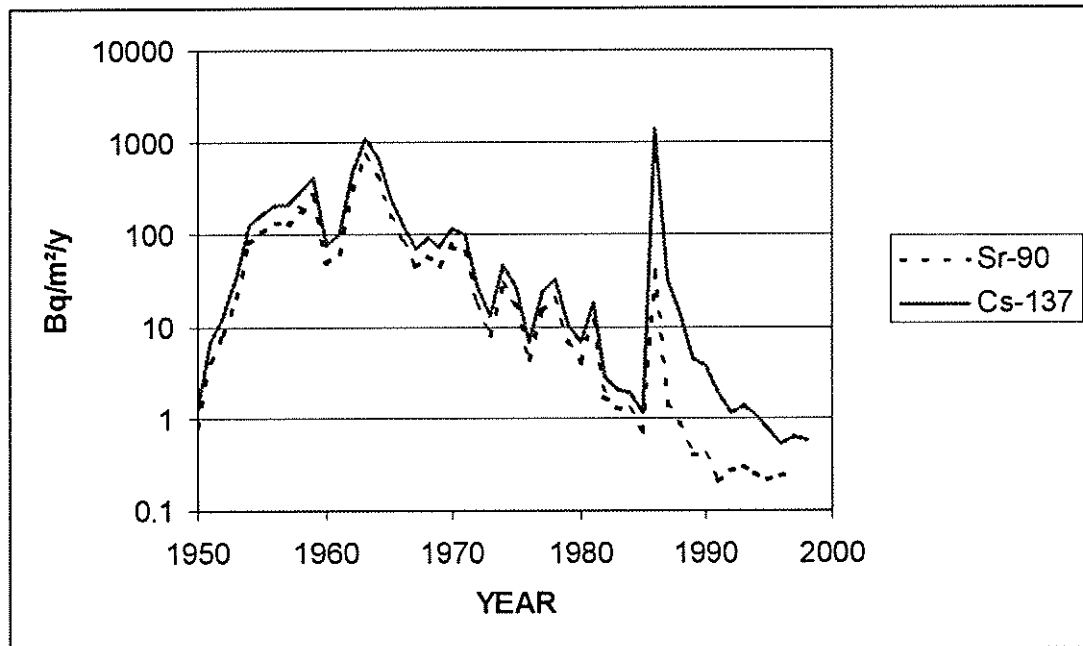


Fig. 1. Annual fallout ( $\text{Bq m}^{-2}$ ) of  $^{90}\text{Sr}$  and  $^{137}\text{Cs}$  in Jutland.

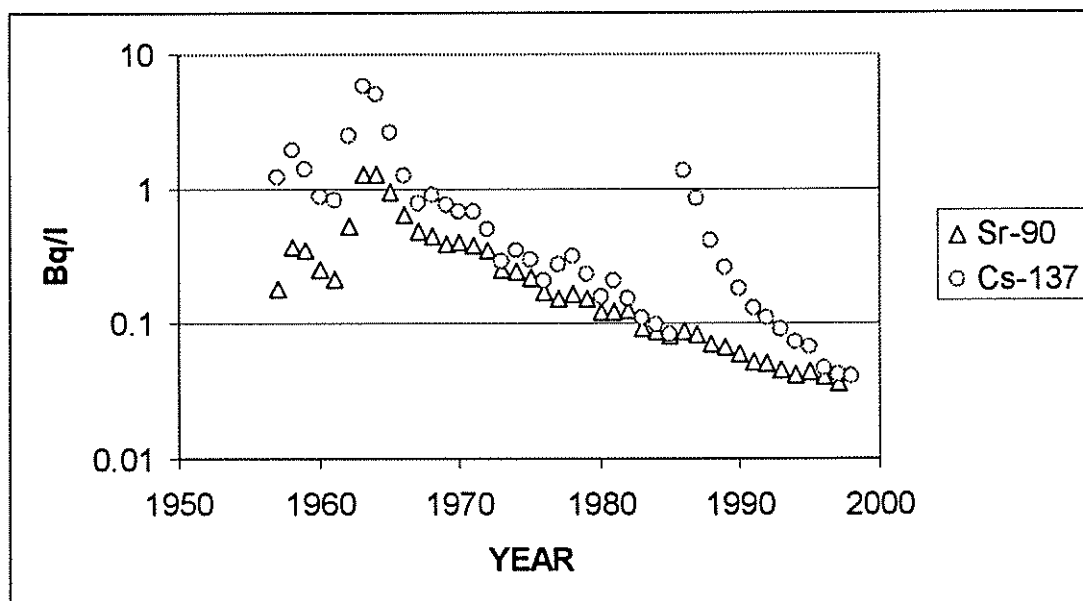


Fig. 2. Annual average concentrations ( $\text{Bq l}^{-1}$ ) of  $^{90}\text{Sr}$  and  $^{137}\text{Cs}$  in milk from Jutland.

Mathematical models accounting for the transfer of deposited radioactivity to the human diet and further to man were developed in the 1960's and adopted by UNSCEAR (e.g. UNSCEAR, 1972; Aarkrog, 1979). The derivation of these models involves the determination of parameters of transfer functions by least-squares fitting of diet and deposit data. This type of models may be applied for any environmental compartment for which data are available. Radioecological sensitivities are calculated from the parameters of the transfer functions. The derivation of these models may be carried out from linear regression analyses of the data by spreadsheet (e.g. Excel) calculations.

More recent radioecological models are dynamic and mechanistic in the sense that they incorporate explicit descriptions of individual transfer processes, e.g. deposition, resuspension, exchange between soluble and insoluble phases, root uptake, weathering, uptake and excretion by animals. These models are more complex and require more computing power. Two such models were used here: ECOSYS (Müller and Pröhl, 1993) and FARMLAND (Brown and Simmonds, 1995). ECOSYS was available in an Excel version and FARMLAND in a version imbedded in a software package (PC-CREAM, 1997). Neither of these versions were well suited for the purpose of making comparisons with the above mentioned data since the models were intended for other purposes. But it was possible from both model versions to extract the transfer of  $^{90}\text{Sr}$  and  $^{137}\text{Cs}$  to grass, milk and beef from a unit ground deposition and fold this transfer with the annual deposition data from Denmark (cf. Fig. 1).

### Radioecological Sensitivities

Transfer functions of the UNSCEAR model concept were fitted to the data, generally using all available data. In a few cases, however, data on  $^{137}\text{Cs}$  were omitted from the fitting for the years 1986 and 1987 due to the different environmental transfer found in these years from fresh Chernobyl fallout compared to fallout from nuclear weapons testing. The prediction models are shown in Tables 1 and 2.

Table 1. UNSCEAR prediction models for  $^{90}\text{Sr}$  in grass, milk, beef and diet.

	Jutland	Islands
Grass (mBq/kg fw)	$20 d_i + 13 d_{i-1} + 0.73 A_{28}$	$14 d_i + 28 d_{i-1} + 0.45 A_{28}$
Milk (mBq/l)	$1.2 d_i + 0.8 d_{i-1} + 0.21 A_5 + 0.013 A_{28}$	$0.9 d_i + 0.7 d_{i-1} + 0.14 A_5 + 0.015 A_{28}$
Beef (mBq/kg)	$0.45 d_i + 0.078 d_{i-2} + 0.027 A_{10}$	
Diet (mBq/d/cap)	$2.1 d_i + 1.8 d_{i-1} + 0.22 A_{10}$	$2.2 d_i + 1.8 d_{i-1} + 0.22 A_{10}$

Table 2. UNSCEAR prediction models for  $^{137}\text{Cs}$  in grass, milk, beef and diet.

	Jutland	Islands
Grass (mBq/kg fw)	$21 d_i + 0.9 d_{i-1} + 0.15 A_{30}$	$14 d_i + 1.1 d_{i-1} + 0.74 A_5$
Milk (mBq/l)	$5.3 d_i + 2.1 d_{i-1} + 0.12 A_2 + 0.032 A_{30}$	$2.6 d_i + 1.6 d_{i-1} + 0.017 A_{30}$
Beef (mBq/kg)	$32 d_i + 0.16 A_{30}$	
Diet (mBq/d/cap)	$5.4 d_i + 4.3 d_{i-1} + 2.4 d_{i-2} + 0.083 A_{10}$	$4.7 d_i + 4.3 d_{i-1} + 2.2 d_{i-2} + 0.11 A_{10}$

Table 3. Radioecological sensitivities from UNSCEAR prediction models.

Sensitivity	Sr-90		Cs-137	
	Jutland	Islands	Jutland	Islands
Grass (mBq y/kg fw)	$63 \pm 18$	$60 \pm 4$	$28 \pm 2$	$20 \pm 2$
Milk (mBq y/l)	$3.9 \pm 0.5$	$3.2 \pm 0.6$	$9.1 \pm 1.4$	$4.9 \pm 0.2$
Beef (mBq y/kg)	$0.9 \pm 0.1$		$38 \pm 3$	
Diet (Bq/cap)	$2.6 \pm 0.1$	$2.6 \pm 0.1$	$4.9 \pm 1.0$	$4.6 \pm 0.9$



The radioecological sensitivities calculated from these prediction models are shown in Table 3 giving the time-integrated transfer from a deposition of 1 Bq m<sup>-2</sup>. The sensitivities found from the ECOSYS and FARMLAND models are shown in Table 4.

Table 4. Radioecological sensitivities based on an application of the ECOSYS and FARMLAND models to Danish fallout data.

Sensitivity	Cs-137, Islands	
	ECOSYS	FARMLAND
Grass (mBq y/kg fw)	130	28
Milk (mBq y/l)	18	5.5
Beef (mBq y/kg)	47	28

### Conclusions

Traditional radioecological models of the UNSCEAR type have been applied to time series of Danish data on <sup>90</sup>Sr and <sup>137</sup>Cs in grass, milk, beef and diet covering four decades. The models fit the data well for <sup>90</sup>Sr for the entire period, but for <sup>137</sup>Cs the models generally perform somewhat poorer from 1986 and onwards. The reason for this is the different transfer through the environment particularly in 1986 and 1987 of Chernobyl radiocaesium compared to that from nuclear weapons testing. This difference is caused mainly by different seasonal modes of deposition. But overall, the models reproduce the data quite well. The radioecological sensitivity of the Danish diet for the transfer of <sup>90</sup>Sr and <sup>137</sup>Cs through Danish foodchains is found to represent an average individual intake of 3 Bq <sup>90</sup>Sr and 5 Bq <sup>137</sup>Cs for a ground deposition of 1 Bq m<sup>-2</sup> when no countermeasures are applied.

Dynamic radioecological models do not reproduce the fallout data better than the traditional models. But the general features of the dynamic models make them suited for prediction of radiological consequences of routine and accidental releases in areas where limited radioecological data are available.

### Acknowledgements

The work was supported by the Nordic Nuclear Safety Research programme (NKS/BOK-2 project) and by the Danish Emergency Management Agency.

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# **A compartment model for analysis of routine releases of radionuclides to a fjord**

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**Abstract** A compartment model has been used to calculate the radionuclide concentration in the local aquatic environment from routine releases of liquid radioactive effluents from the Halden Boiling Heavy Water Reactor and the subsequent radiation doses to critical groups from a number of exposure pathways have been estimated. The results will be used to define release limits for the reactor operation, implicating a conservative approach, and for each radionuclide the model input values have been varied to obtain the combination of values which gives the highest radiation dose.

## **Introduction**

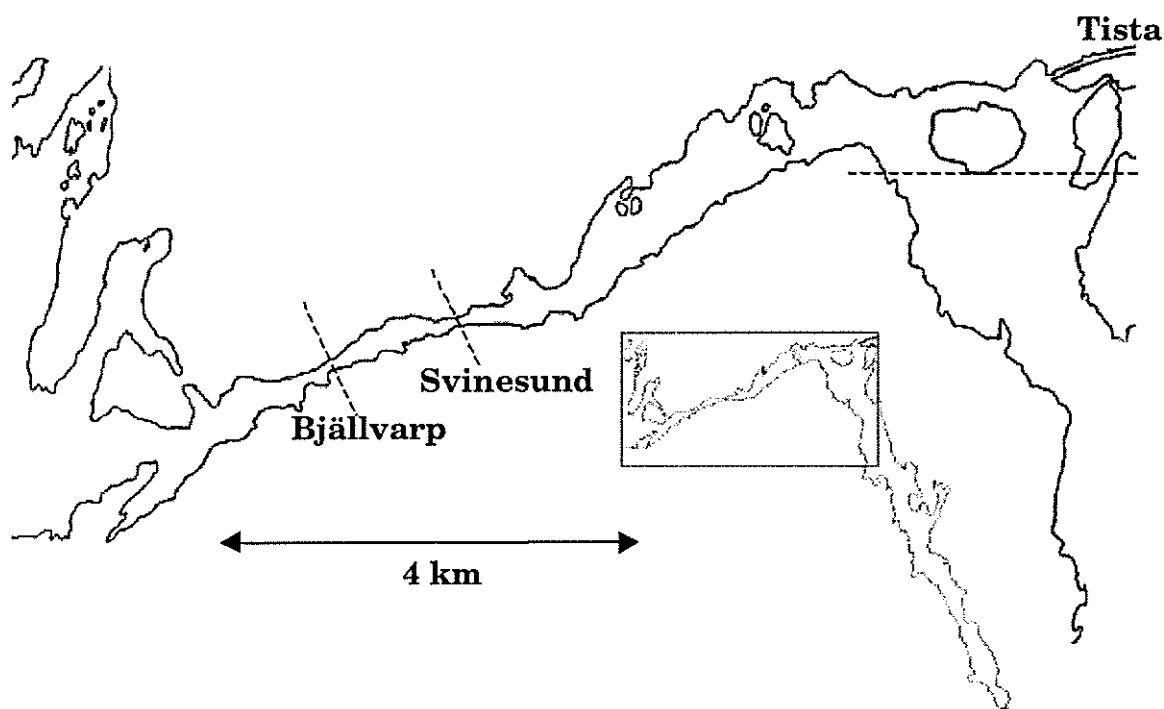
Liquid radioactive effluents from the operation of the Halden Boiling Heavy Water Reactor (HBWR) are routinely released to the river Tista, which discharges into the fjord Iddefjord. Current operational release limits are based on nearly twenty years old estimates of radiation doses to critical groups. At that time the Iddefjord was characterised as one of the most polluted fjords in Norway, mainly due to extensive discharges of pollutants from the local paper and pulp industry. The fjord exhibited a minimum of marine life and occupancy on the fjord shores was far from attractive. It was therefore difficult to find any reasonable radiation exposure pathway connected to the fjord itself. Instead the estimates of the radiation doses were based on the radionuclide concentration in the adjacent marine region. Since then the situation has drastically changed, with the closing down of the pulp mill and further restriction of the discharges from the industry, and today a great deal of recreational fishing, bathing and occupancy of the fjords shores takes place. This has necessitated a definition of a new critical group and a re-evaluation of the radiation doses and derived release limits.

In the following radiation doses following consumption of fish from the fjord, occupancy of beaches, boating and swimming are estimated. They are all based on the radionuclide concentration of the fjord water, which are calculated with a compartment model proposed by EU [1].

## **Methods**

The Iddefjord is 25 km long and rather shallow with a maximum depth of 40 meters, see figure 1. It is a typical sill fjord where the deeper layers are restricted from the adjacent ocean by two sills at a depth of 9 meters. The fjord is topographically divided in an inner and outer part by a sill at 20 meters depth at a distance approximately one third from the mouth of the fjord. The main fresh water input is from Tista, discharging into the outer part of the fjord, and a smaller river discharging into the inner part.

Vertically the fjord is divided into three layers. The lighter fresh water from the rivers is by winds and tides mixed with the underlying seawater and a brackish surface layer develops. Beneath the surface layer and above the sill depth, the intermediate layer, sea water is flowing inwards to an extent determined by the mixing of sea water into the surface layer, termed estuarine circulation. This layer also sustains an additional circulation system driven by pressure differences between the layer water and the coastal sea outside the fjord. The layer beneath the sill depth is termed the deep-water layer or basin layer. Since it is trapped by the



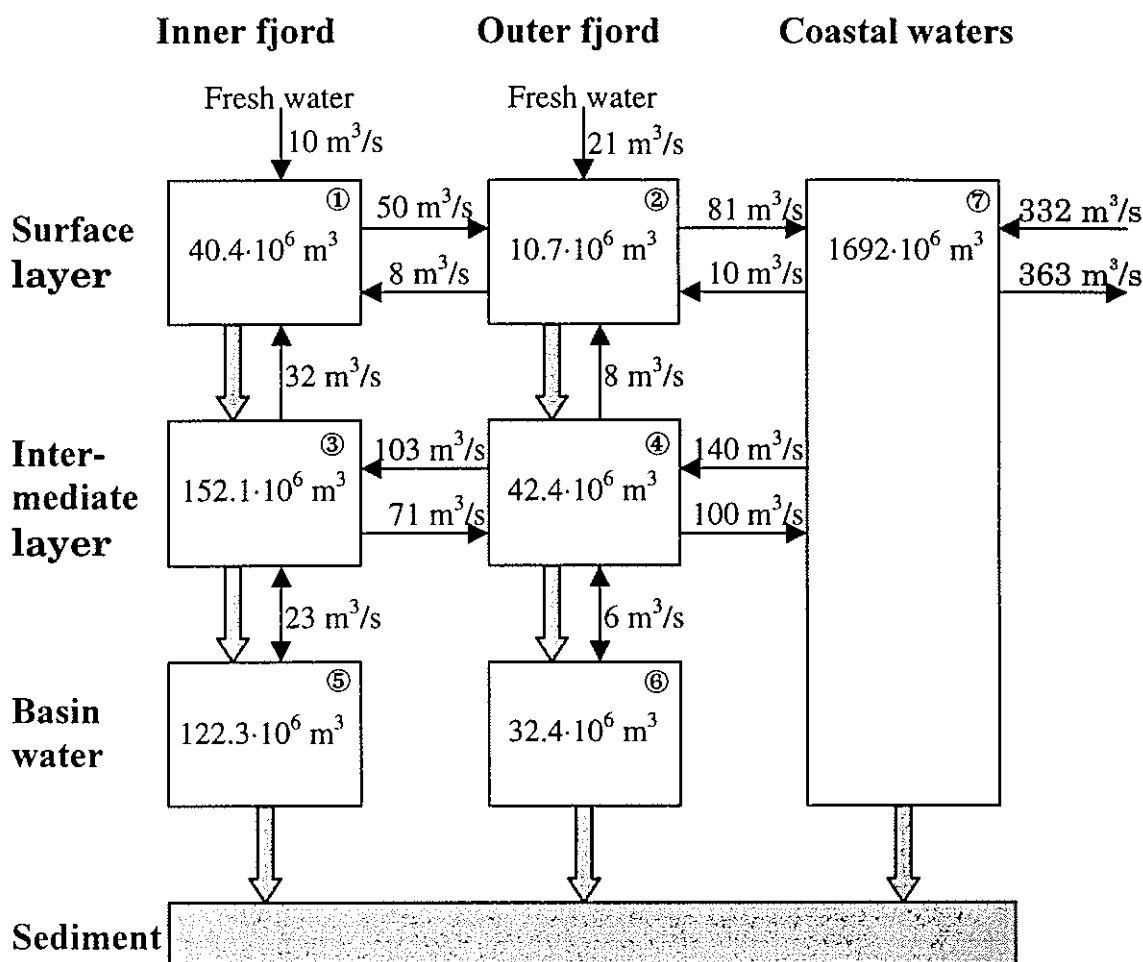
**Figure 1.** The Iddefjord. The localities of the sills are indicated with broken lines.

sills, renewal of the water only takes place at discrete times when the density of the coastal water above the sill exceeds the density of the basin water.

The discharge of radionuclides from HBWR goes to the surface layer of the outer fjord, where it is transported seawards. This layer has a high suspended sediment load and sedimentation rate, and radionuclides, which are sorbed to sediments, will partially be removed to the intermediate layer. Here the suspended sediment load are lower, but since the residence time of the water is longer, part of the radionuclides will enter the basin water with a subsequent settling at the bottom of the fjord.

The fjord topography and circulation system coincides in many ways with the generic compartment model analysis for a fjord proposed by EU [1]. This model assumes instantaneous uniform mixing within each compartment, with transfer between compartments being proportional to the inventory of material in the source compartment. The values for the transfer of the water between compartments as well as the volume of the different compartments are obtained from observations of the fjord by the Norwegian Institute for Water Research [2,3]. The most important data are the surface layer residence time of 11.8 days, an estuarine circulation of  $140 \text{ m}^3/\text{s}$ , and renewal of the basin water on an average of six times a year. The latter is approximated with a continuous interaction with the intermediate layer. The average freshwater input is  $21 \text{ m}^3/\text{s}$  from Tista and  $10 \text{ m}^3/\text{s}$  from rivers and creeks discharging to the inner fjord. The model is shown in figure 2.

The compartment model analysis uses a set of coupled first order differential equations to describe the transfer of materials between the compartments. The form of these equations, as well as methods for calculating the sedimentation of radionuclides, are summarised elsewhere [1,4]. The partitioning of radionuclides between the water phase and suspended sediment material is based on default values for the sediment distribution coefficient [1]. A suspended sediment load of  $2 \cdot 10^{-4} \text{ t} \cdot \text{m}^{-3}$  and a sedimentation rate of  $2.7 \cdot 10^{-10} \text{ t} \cdot \text{m}^{-2} \cdot \text{s}^{-1}$  have been used for the surface layer, while for the other compartments the values are  $5 \cdot 10^{-5} \text{ t} \cdot \text{m}^{-3}$  and  $1.6 \cdot 10^{-10} \text{ t} \cdot \text{m}^{-2} \cdot \text{s}^{-1}$  respectively.



**Figure 2.** Compartment model of the Iddefjord, showing the volume of the different compartments and the water transfer between compartments. Grey double arrows illustrate sedimentation. The radionuclide input into the fjord system is in the surface layer of outer fjord.

Radiation doses to critical groups can now be estimated from the radionuclide concentration of the different compartments. The total dose is calculated from a yearly consumption of 30 kg fish from the fjord, 200 h/y occupancy at the fjord shores, 50 h/y of bathing and 1000 h/y of boating. Since the results will define the release limits for the operation of HBWR, a conservative approach has been adopted, and the radionuclide concentration in fish is estimated from the highest concentration in the intermediate and basin layer, while the external doses are based on the highest concentration of the two surface layers. Default values were used for the fish concentration factor while the radionuclide concentration on beaches was obtained by the use of the default sediment distribution factors and applying a reduction factor of 10 to take account for larger sand grains and gravel on beaches opposed to the fine muddy particulates of bed sediment.

Several of the values of the parameters in the described compartment model are uncertain and are also expected to vary, and the influence of this variation on the calculated total radiation dose must be estimated. This was carried out by assigning a variability of 20 % to the water flow, and 50 % to the suspended sediment load and sedimentation rate. The variation of the water flow was performed in such a way that the frequency of water renewal of the different layers was held constant. The radiation doses were then calculated repeatedly (500 times) for each radionuclide. The standard deviation of the obtained values could then be used as an indication of the uncertainty of the results.

## Results

Table 1 shows the results of the dose calculation for some nuclides for a continuous release of 1 GBq/year.

**Table 1.** Water concentration and total radiation dose from a continuous release of 1 GBq/y, with the coefficient of variation of the values (standard deviation divided by the average value)

Nuclide	Concentration in water (Bq·m <sup>-3</sup> ) Coefficient of variance (%) in parentheses		Total radiation dose (nSv·y <sup>-1</sup> )	Coefficient of variance (%) of the dose
	Compartment 2	Compartment 6		
<sup>59</sup> Fe	0.37 (3)	0.04 (22)	25	6
<sup>60</sup> Co	0.39 (3)	0.21 (26)	158	7
<sup>131</sup> I	0.34 (0.5)	0.001 (29)	0.14	8
<sup>137</sup> Cs	0.43 (1)	0.12 (11)	5.6	11

It can be seen that the radiation doses are robust to variation in the fjord model data. One of the reasons is that the total dose consists both of doses connected to the surface layer concentration and to the concentration in the deeper layers. A decrease in the surface layer concentration, due for instance to a higher sedimentation rate, would lead to an increase in the concentration of the deeper layers, and conversely, leading to a total smaller variation of the radiation dose. For the purpose of establishing release limits for the reactor operation the maximum dose from the variation will be used.

The calculated radionuclide concentration of the sand of the fjord beaches matches well with measurements. In the period July 1998 to July 1999 the total release of <sup>60</sup>Co to water from HBWR was 0.3 GBq. The model predicts an activity concentration of 2.7 Bq/m<sup>3</sup> in the sand from that release, while the measured activity ranges from below detection limit of 0.3 Bq/m<sup>3</sup> up to 1.7 Bq/m<sup>3</sup> [5].

The results differ considerably from previous calculations, performed in 1982. The reason is not only the use of different marine models, but also the difference in the recommended parameters today, compared to the accepted values in 1982. Examples are the equilibrium sediment distribution factor for <sup>60</sup>Co, which is 20 times higher in current recommendations compared to previous, and the bio-accumulation factor for marine fish for the same nuclide, which has been increased by a factor 10.

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## **DIRECT MONITORING OF RADIOCAESIUM IN LIVE REINDEER AND REINDEER CARCASSES**

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The southern part of the Swedish reindeer herding area was severely affected by contamination from the Chernobyl fallout in 1986. Ground deposition levels of  $^{137}\text{Cs}$  up to  $100\,000\text{ Bq m}^{-2}$  were recorded in some areas. In the winter after the fallout, levels of  $^{137}\text{Cs}$  between  $10\,000$  and  $50\,000\text{ Bq kg}^{-1}$  were common in reindeer from southern parts of Västerbotten and northern parts of Jämtland county (Åhman and Åhman 1994). A large part of the reindeer meat produced in Sweden during the first years after the fallout was excluded from human consumption (destroyed or used as feed for animals). The contamination of reindeer and reindeer products naturally created great worry and anxiety among reindeer herders, as their possibility to produce reindeer meat as well as their own health seemed to be threatened.

Monitoring of reindeer and reindeer pasture started already during the summer 1986, and the control of reindeer meat has been very strict, not to allow reindeer meat with too high  $^{137}\text{Cs}$  levels to enter the market for human consumption. A strict control was approved by reindeer herders, since there was a fear for otherwise creating a negative attitude towards reindeer meat. A large amount of meat samples ( $62\,924$  samples during the first year and  $53\,544$  during the second) were taken from reindeer carcasses at slaughter (Fig. 1). These samples were sent to either of two laboratories (Malå, Västerbotten, and Uppsala). The cost for intense sampling was high (about  $100\text{ SEK}$  per slaughtered reindeer) and the need to retain reindeer carcasses at the slaughter house until the samples were analysed created practical problems.

Countermeasures, like altering the time of slaughter, feeding with uncontaminated feed and use of caesium binders, were gradually introduced during the first years after the fallout (Fig. 2). Good predictions regarding radiocaesium levels in reindeer had to be made for different areas and different periods of the year in order to choose the most relevant countermeasure.

During 1988, a handheld equipment consisting of a NaI detector connected to a robust field computer, GammaGeomacII (made by ABEM AB, Malå - now Malå GeoScience) was tested, with the aim to obtain an easy and quick way to monitor live reindeer before making decisions regarding slaughter and/or countermeasures. The detector measures all gamma radiation above  $0.05\text{ MeV}$  and the results were originally given as CPS (counts per second). The equipment and procedure was first tested during February–April 1988, by monitoring live reindeer before slaughter and then analysing muscle samples from the slaughtered reindeer at the laboratory.

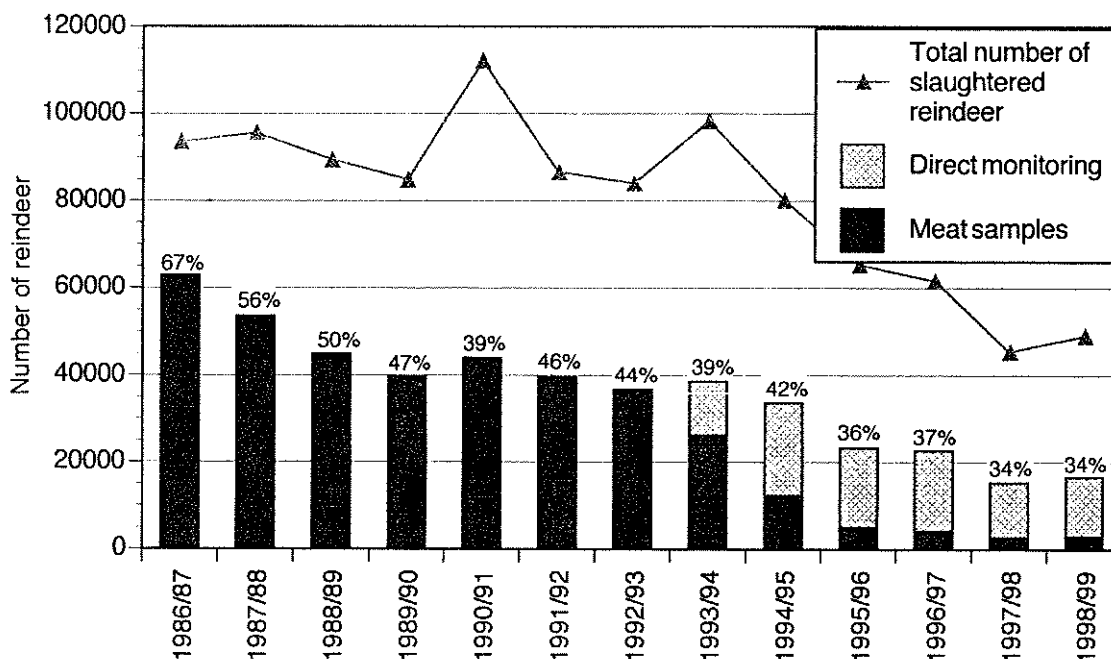


Fig. 1. Control of  $^{137}\text{Cs}$  in reindeer carcasses at slaughter. Figures at top of the bars show percent controlled animals of the total number of slaughtered reindeer.

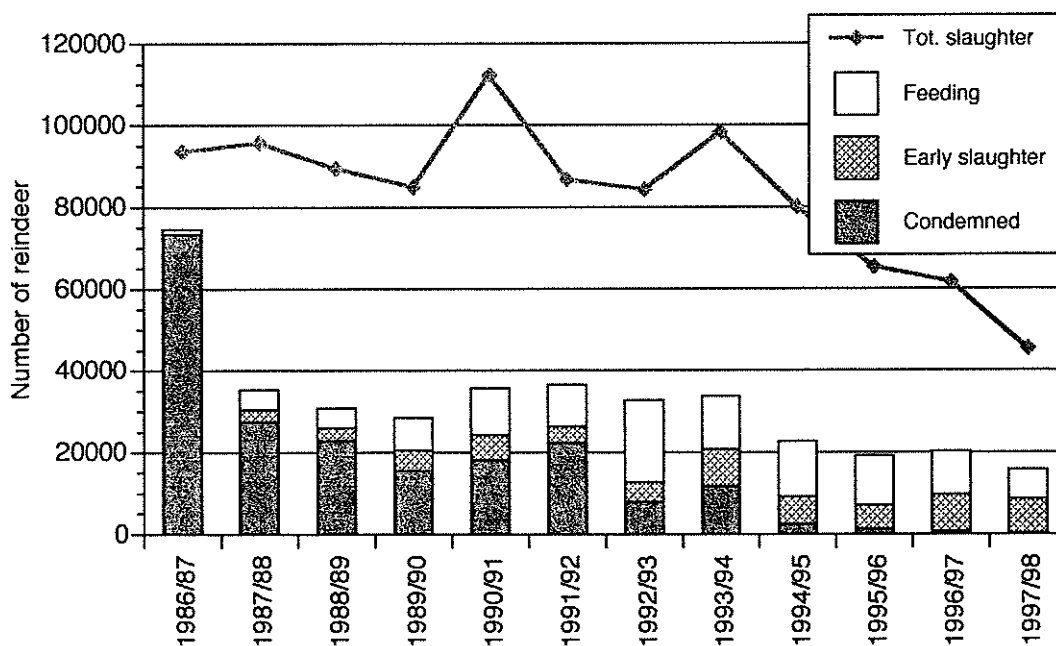


Fig. 2. Countermeasures applied in Sweden to reduce radiocaesium levels in reindeer meat.

The tests showed that the best results were obtained holding the detector against the most muscular part of the shoulder during 10 seconds. The method seemed to be reliable enough for the purpose down to  $^{137}\text{Cs}$  levels around  $300 \text{ Bq kg}^{-1}$  muscle. As expected, there was a significant effect of body size (weight) of the reindeer on the results, and the accuracy of the method was also affected by background radiation. Since the results of live reindeer monitoring seemed promising, the equipment was tested one year later for monitoring reindeer carcasses at slaughter.

From 1989 to 1993, standardised procedures were developed for monitoring both live reindeer and reindeer carcasses (Fig. 3). The equipments were individually calibrated and programmed to show the result as BqEqv (becquerel equivalents = equivalent to  $\text{Bq kg}^{-1}$ ) using different calibration factors for live reindeer and carcasses and for three categories of reindeer corresponding to different weight classes (calves, females + young males and adult males, respectively, Åhman and Åhman 1993).

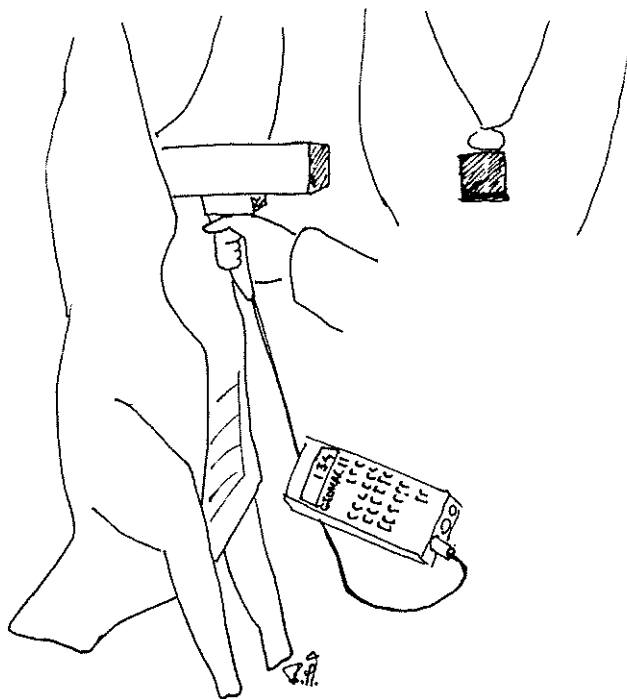


Fig. 3. Monitoring gamma radiation from a reindeer carcass

Currently (1999), 13 equipments are in use and they are handled by persons with special training. The accuracy of each equipment is regularly checked by comparing the direct monitoring with the analyses of muscle samples at the laboratory (Gammadata mätteknik, Uppsala). Since 1993, the method is approved by the Swedish Food Administration for control of radiocaesium in reindeer. However, carcasses that are close to the accepted limit for  $^{137}\text{Cs}$  in reindeer meat ( $1500 \text{ Bq kg}^{-1}$ ) must also be checked by analysing meat samples at the laboratory (SLV 1999). The use of direct monitoring has gradually reduced the need for laboratory analyses



of muscle samples (Fig. 1) and thus helped to reduce costs and facilitate the procedure at the slaughter house.

Live reindeer are monitored to see that all (or most) reindeer are below the accepted limit for sale before slaughter. If the levels of radiocaesium are too high, the reindeer could be taken to a corral and fed uncontaminated feed during a period of up to two month before they are slaughtered, while reindeer showing low levels of radiocaesium could be slaughtered immediately. During the last year (July 1998 to June 1999) almost 9 000 live reindeer were monitored before slaughter or feeding. The possibility to rapidly monitor a large number of reindeer has made it easier to choose the most appropriate countermeasures and to avoid slaughtering reindeer, if there is a risk that their levels of  $^{137}\text{Cs}$  are too high. The effect has been that gradually less reindeer are above the accepted level of  $^{137}\text{Cs}$  at the time of slaughter, and during the last year only 0.4% of the slaughtered reindeer had to be condemned because of too high radiocaesium levels (Fig. 2) and the costs associated with radiocaesium contamination of reindeer have been significantly reduced (Åhman 1999).

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## **SESSION VII**

### **DOSIMETRY, INSTRUMENTATION, AND QUALITY ASSURANCE**

- a) Diagnostic Radiology**
- b) Dosimetry**

**Chairman: Enn Realo**



# Generelle principper og problemer ved bestemmelse af den kollektive dosis til befolkningen fra røntgendiagnostik

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Når den kollektive dosis til befolkningen fra røntgendiagnostik skal bestemmes, er der flere aspekter der skal overvejes. Det skal bl.a. besluttet hvorledes oplysninger om frekvensen af undersøgelser fremskaffes. Skal der benyttes spørgeskemaer eller findes tallene allerede på sygehuse i en eller anden form? Hvordan udvælges de sygehuse der skal indgå i projektet og hvordan bestemmes landsresultatet ud fra denne stikprøve?

Næste skridt er at bestemme hvilken type målinger man ønsker at foretage. Mulighederne er mange, f.eks. måling af arealdosis, indgangsdoser eller beregning ud fra generator-indstillinger. Et andet spørgsmål er hvordan målingerne foretages i praksis? Skal det være op til det enkelte sygehus eller skal målingerne foretages af en medarbejder tilknyttet projektet?

Når alle data er indsamlet venter således den endelige behandling. Hvordan omsættes disse indsamlede målinger til effektiv dosis? Om man har mulighed for at gennemføre det ideelle projekt med hensyn til udvælgelse, målinger og databehandling, afhænger af mængden af de ressourcer der er til rådighed.

Statens Institut for Strålehygiejne er i øjeblikket i gang med at foretage en undersøgelse af befolkningsdosis fra røntgendiagnostik. Det er lykkedes projektgruppen at fremskaffe information fra sygehuse i Danmark om frekvensen af undersøgelser. Oplysningerne dækker ca. 95 % af den danske befolkning. Målingerne bliver udført på sygehuse, hos kiropraktorer og tandlæger. Dosismålingerne foretages dels med arealdosimeter, dels med TLD tabletter. Omsætningen fra arealdosis til effektiv dosis foretages ved hjælp af et til formålet udviklet program baseret på Monte Carlo-kode.

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## Indledning

I direktivet om beskyttelse af patienter i forbindelse med medicinsk bestråling (EU direktiv, 1997) står følgende:

*"Medlemsstaterne sikrer, at fordelingen af individuelle doser fra den i artikel 1, stk. 2, omhandlede medicinske bestråling vurderes for hele befolkningen og for relevante referencegrupper i befolkningen i henhold til, hvad medlemsstaten måtte finde nødvendigt."*

Denne artikel udtrykker at de enkelte medlemslande skal vurdere den kollektive dosis til befolkningen fra røntgendiagnostik. Inden man påbegynder et sådant projekt er der flere spørgsmål der skal besvares.

- Hvordan skal frekvensen af undersøgelser fremskaffes
- Hvordan findes de relevante doser
- Hvordan omsættes de målte værdier til effektiv dosis
- Hvordan skal målingerne indsamles

## **Bestemmelse af undersøgelsesfrekvens**

Frekvensen (eller antallet) af undersøgelser der foretages i det enkelte land skal kortlægges før man kan vurdere den effektive dosis hidhørende fra denne undersøgelse. En måde at gøre dette på er f.eks. at sende spørgeskemaer ud til de enkelte sygehuse og bede dem om at registrere hvor mange undersøgelser af en given type de udfører i en periode på f.eks. 14 dage. På baggrund af disse tal kan det samlede antal undersøgelser af en given type undersøgelse. Ulempen ved denne form for registrering er at sygehusene i mange tilfælde ikke har ressourcer til at deltage i et sådant projekt. En anden og bedre metode er at benytte de tal som sygehusene indberetter, til at vurdere undersøgelsesfrekvensen. Detaljeringsgraden af disse indberetninger varierer dog meget fra amt til amt.

## **Udvælgelse af sygehuse og gennemførelse af et pilotprojekt**

Når man har fastlagt frekvensen af undersøgelser på landsplan, er det nødvendigt at bestemme hvilke undersøgelser der skal måles på og hvor mange steder der skal måles for at opnå en god statistik. En mulighed er på dette tidspunkt i projektet at udfører et pilotprojekt for at bestemme omtrentlige doser fra forskellige undersøgelser. Det sygehus der udvælges til pilotprojektet må være af en vis størrelse, således at der udføres et bredt udsnit af undersøgelser.

Når et sådan pilotprojekt er gennemført, kan de fundne doser benyttes til at vurdere hvilke undersøgelser der bidrager med hvor mange procent til den kollektive dosis. Man kan herefter vedtage at man f.eks. måler på de undersøgelser der tilsammen bidrager med f.eks. 80 % af den samlede dosis. Den fejl man begår herved kan derefter bestemmes af statistisk vej. Alternativt kan der måles på så mange undersøgelser som muligt, indenfor en given periode.

Antallet af sygehuse der skal måles på er helt bestemt af de ressourcer der er til rådighed for projektet.

- ☐ Hvor lang tid ønsker man at måle
- ☐ Hvor længe skal der måles på hvert sygehus
- ☐ Hvor mange steder kan der måles samtidigt

Disse tre parametre bestemmer tilsammen antallet af sygehuse, der kan måles på indenfor en given periode. Udvælgelsen af de enkelte sygehuse bør foregå tilfældigt, dog således at alle typer sygehuse er repræsenteret dvs. små, mellem og universitetssygehuse. Udvælgelsen skal ske på en sådan måde at antallet af undersøgelser i de tre sygehus kategorier bestemmer hvilken vægt de skal have.

## **Målemetoder og databehandling**

Når man er kommet til det punkt i projektet hvor man skal til at måle skal man bestemme hvilken målemetode(r) man ønsker at benytte. Det kan være forskellige metoder til forskellige undersøgelser. F.eks. kan man på sygehuse med fordel

benytte arealdosimetre, idet disse ikke kræver de ressourcer, som f.eks. TLD tabletter. I andre tilfælde er TLD tabletter bedre, dette er tilfældet hvis målingerne skal foregå pr. post.

Hvordan skal målingerne så foretages ? Det enkelte sygehus kan foretage de nødvendige målinger, hvis de har kapacitet og ressourcer til dette. Hvis dette ikke er tilfældet, må en projektmedarbejder gøre dette i stedet. Set fra et ressourcemæssigt synspunkt er den første løsning at foretrække.

Efter endt dataopsamling skal data behandles. Dette kan ske ved brug af "Monte Carlo" metoder (MC) eller ved brug af faste omregningsfaktorer. Det er her værd at bemærke at udnyttelse af MC metoder er tidskrævende set fra et beregningsmæssigt synspunkt, idet beregning af den effektive dosis for en eksponering, nemt kan tage flere minutter.

### Befolkningsprojekt

I Danmark er Statens Institut for Strålehygiejne (SIS) ved at lægge sidste hånd på et projekt hvis formål er at kortlægge den gennemsnitlige dosis til befolkningen fra røntgendiagnostik. Projektet er planlagt til at strække sig over en 3 årig periode. Projektet er gennemført med støtte fra Apotekerfonden.

Projektet blev gennemført ud fra en modificeret protokol som NRPB i England benyttede i deres undersøgelse (P. C. Shrimpton *et al*, 1986). Det blev valgt at måle med en kombination af arealdosimetre og TLD tabletter. På sygehusene blev der benyttet arealdosimetre og hos kiropraktorer TLD tabletter.

Arealdosimetrene blev styret fra en PC, der således var i stand til at gemme resultatet af de undersøgelser der blev foretaget på det pågældende sygehus (fig. 1). Dette blev benyttet ved komplekse undersøgelser som f.eks. colon, intravenøs urografi etc. Ved simple undersøgelsestyper som f.eks. thorax, bækken, columna lumbalis etc. noterede personalet på de enkelte afdelinger de målte værdier ned på et dertil fremstillet skema.

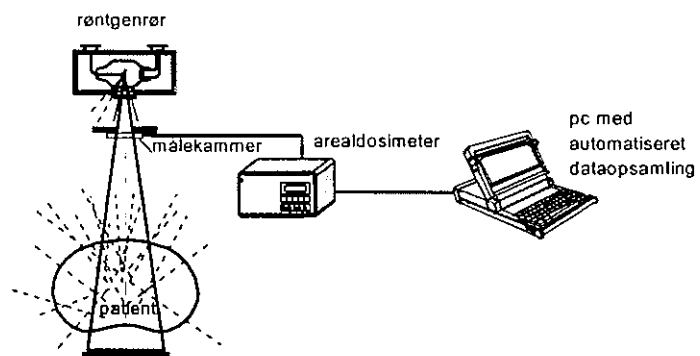


Fig. 1 Måleopstilling til bestemmelse af arealdosis

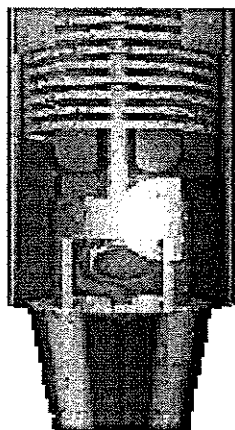
Når der skulle måles hos kiropraktorerne blev der valgt at gøre dette pr. post. Et antal udvalgte klinikker fik tilsendt TLD tabletter og fik besked på at måle på deres patienter. De udfyldte et skema hvorpå de benyttede tabletter blev fastgjort. Skemaerne blev derefter returneret til instituttet for videre behandling.

Et pilotprojekt blev gennemført på et større dansk provinssygehus. De indsamlede data blev brugt til at vurdere størrelsen på de doser der skulle bestemmes og til at fastlægge hvilke undersøgelser der skulle indgå i måleprotokollen.

**Tabel 1.** Eksempel på dosis og undersøgelsesfrekvens på landsplan

Undersøgelse	Kode	Antal	Effektiv dosis (mSv)	Total dosis (manSv)
Thorax	5010	590.518	0,10	63,8
Knæled patella	3074	100.865	$5,6 \cdot 10^{-3}$	0,57
Columna lumbo-sacralis	2012	100.389	0,86	86,3
Hånd / fingre	3054	82.772	$2,6 \cdot 10^{-4}$	0,02
Hofterled	3068	81.003	0,85	68,9

Det viste sig at undersøgelsesfrekvensen kunne kortlægges meget nøjagtigt, idet der kun var et enkelt amt, hvor det ikke var muligt at få oplyst antallet af undersøgelser ud fra den 4-cifrede EDB klassifikation (Sundhedsstyrelsen, 1989). Tabel 1 viser de fem mest almindelige undersøgelser i 1995 og de tilhørende doser fra pilotprojektet.



**Fig. 2.** MC fantom med billedfelt

De indsamlede data blev indtastet i en database og ved hjælp af et til formålet udviklet "Monte Carlo" program blev de effektive doser beregnet. For hver projektion blev der desuden genereret et billede af det matematiske fantom og feltprojektion (fig. 2). Der blev i alt foretaget mere end 23.000 beregninger og databasen rummer i dag oplysninger om doser til mere end 3.200 patienter. Desuden er der genereret næsten 46.000 fantom billeder som vist på fig. 2. Under lagringen af data er der taget hensyn til at disse skal være tilgængelige og kunne benyttes i andre sammenhænge.

## Konklusion

Erfaringer har vist at en omhyggelig planlægning er nødvendig. Det har især været vigtigt at få røntgenafdelingerne til at interessere sig for projektet. Hvis afdelingerne ikke havde været villige til at udføre de nødvendige målinger, ville det have været en besværlig og tidskrævende proces. Det viste sig desuden at netop det at have data samlet et enkelt sted, giver et overblik der ellers ikke havde været muligt.

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# A Practical Approach to Radiation Protection Information in Diagnostic Radiology

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## Abstract

In a benchmarking process, parameters related to patient doses and image quality were compared in x-ray examinations from 10 radiology departments in western Sweden. One main object of the project was to form a pedagogical process focussing on radiation protection and quality matters by engaging radiographers and radiologists from the departments in practical project work and optimisation discussions. Anatomical phantoms with simulated pathology were used for standardised entrance dose measurements and exposure of phantom images. Radiographer performance, such as centering and collimation, was evaluated by radiographers. Radiologists evaluated clinical images using revised CEC quality criteria.

The results of the measurements showed significant differences between the departments both for image quality, entrance dose and the performance of examinations. Explanations to these differences were in many cases found in the choice of equipment, working methods etc.

## Introduction

The legal system for radiation protection in diagnostic radiology prescribes quality assurance of equipment, working methods and competence of staff with the aim to accomplish good radiation protection conditions. This project was an attempt to contribute to all this parts by engagement of radiographers, radiologists and hospital physicists from 10 radiology departments in the newly formed region Västra Götaland in western Sweden, around Göteborg. The departments lie within a travel distance of 150 km and represent the whole scale from a big university hospital to a small private outpatient department. By performing a limited number of measurements in all departments and put them together for comparison the intention was to form a material for fruitful optimisation discussions within the region on joint project meetings and give new input to the local quality work by putting the measured local data from the own department in a regional perspective. The overall aim of the benchmarking approach is to use comparisons of measured quality parameters from the different departments together with explanations of the observed differences to facilitate the orientation towards methods giving good image quality at low patient doses.

## Materials and methods

The project work was managed from Sahlgrenska University Hospital by a project group (PG) consisting of a radiologist, a radiographer/instructor and two hospital physicists.

Every participating department assigned 1-2 radiologists and 1-2 radiographers to be locally responsible for the project. Thus a working group (WG) of about 25 persons was formed.

### Project plan

- An initial visit by PG to all departments for information on the project design to the locally responsible persons. Enquiry forms on equipment, exposure parameters and number of images in a standard chest and urography examination were also distributed.



- Collection of the x-ray images from 10 complete chest and urography examinations in each department.
- A visit to each department by radiographer and hospital physicist from PG for standardised dose measurements and exposure of images on anatomical phantoms and test plates.
- Two image viewing sessions of clinical and test images during two days for radiologists and radiographers in WG respectively.
- Collection of data from measurements and image viewing.
- Analysis of reasons for low image quality/high doses – suggestions for improvement.
- A joint meeting with the whole WG for presentation and discussion of results.
- Visits by PG to departments for presentation of results to the whole staff.

It was initially decided that each department should have access to all measured data from their own department and that data from other departments were presented anonymously, department A-K, for comparison.

#### Measurements and test images

The speed-class of the used film-screen systems was determined by measuring the dose to the cassette resulting in a film with net density 1.0, the so called absolute sensitivity. In digital systems no such figure can be determined.

To make standardised comparisons of patient doses and image quality two anatomical phantoms were used. The chest phantom was a humanoid phantom containing human skeleton and animal lungs. The trunk phantom used for urography was a plastic phantom with plastic skeleton but with no inner organs. Simulated pathological structures were attached to the surface of the phantoms. The used structures had been developed in previous studies (Månsson 1998, Bjurklint 1997). Images of the phantoms were taken by the local radiographers using normal settings for the corresponding clinical investigations in the department.

Images from all departments were viewed by all radiographers and radiologists in the WG in a joint viewing session. The phantom images were examined using visual grading analysis meaning that one of the images was picked as reference image. The viewers graded the visibility of the structures in the images as equal to the reference image (0) slightly better/worse (+/- 1) or considerably better/worse (+/- 2). The mean grade of all viewers became the grade of the department.

In connection with the exposures of phantom images the surface entrance dose to the phantom was measured with a solid state detector (Solidos, RTI Electronics, Sweden). To get an easily comparable dose measure of the total patient dose from a complete urography examination the dose concept "area dose" was used. The area dose is the product of the surface entrance dose to the phantom and the totally used film area for the complete standard urography for each department assuming that each film is completely exposed and the standard patient has the same thickness as the phantom.

#### Clinical images

From each department 10 clinical chest and urography examinations were chosen consecutively from the routine production. Two images from each examination were used in the viewing sessions.

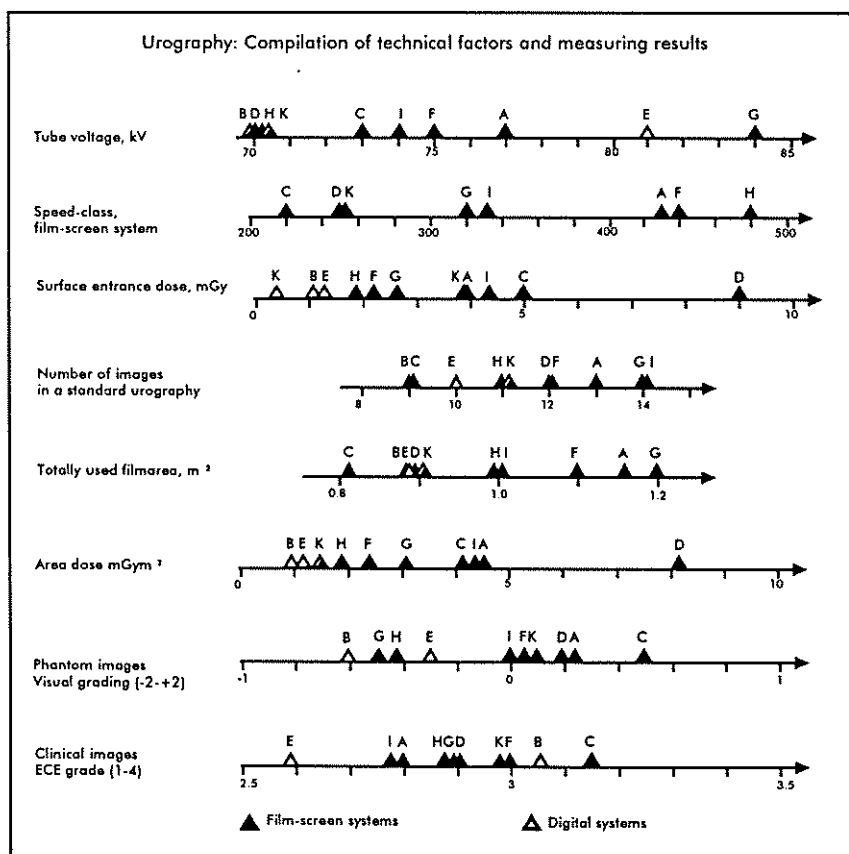
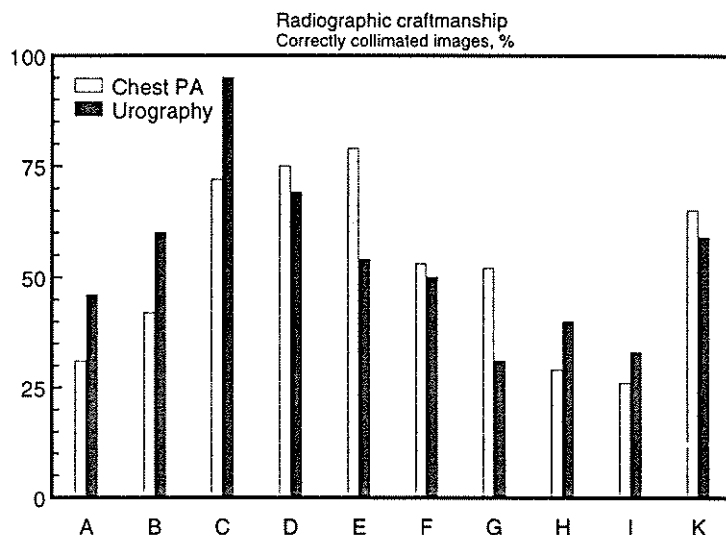
The images were viewed in two ways: radiographers looked at the quality of radiographic craftsmanship in the images and radiologists graded the image quality.

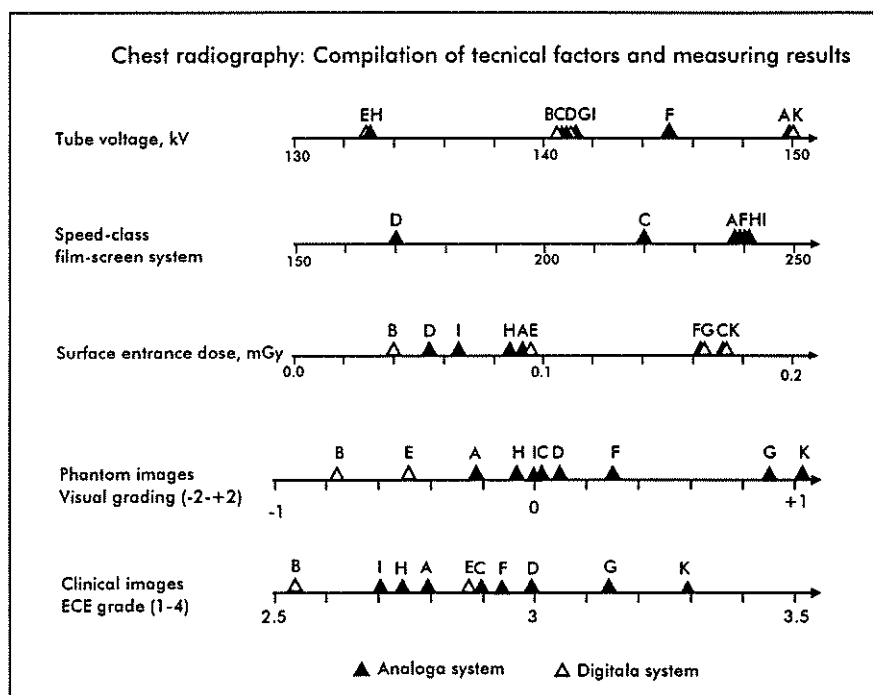
All radiographers in the WG graded images from all departments regarding centering, collimation and exposure on a three step scale: correct, slightly incorrect and definitively incorrect (should have been retaken).

All radiologists in the WG graded the visibility of a number of structures (grade 1-4 where 3 is "adequate") in the images from all departments using revised versions of "European guidelines on quality criteria for diagnostic radiographic images" (European Commission 1996 )

## Results

Examples of results from measurements and viewing session are shown below. The presentation of data is intentionally graphic and perspicuous, an important factor for the usefulness of the data in the discussions with radiologists and radiographers on the influence of different factors .





## Discussion

Measurement results showed, as in many other studies, a considerable variation between radiology departments regarding doses and image quality. The working group discussions clarified, using concrete examples from the "own and neighbouring hospitals", the consequences of different choices of equipment and methods. In some cases the proposals for improvement were obvious.

Studies of "radiographic craftsmanship" are rarely seen. The differences shown in our study could in the discussions often be related to equipment factors and internal department routines for quality assurance. The idea of a radiographers round focussing on methodological and technical aspects of radiographic imaging was proposed and discussed.

The project engaged many workers of different categories in practical project work and discussions on patient doses and image quality. We believe that the "practical approach" had a positive influence on the quality- and radiation protection work in the participating departments.

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European Commission. European Guidelines on Quality Criteria for Diagnostic Radiographic Images. EUR 16260 EN; CEC, Brussels; 1996.

# **EXPOSURE OF PATIENTS AND CREATION OF SYSTEM OF QUALITY ASSURANCE IN CONVENTIONAL X-RAY RADIOLOGY IN LITHUANIA**

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## **ABSTRACT**

One of the most important sources of exposure is medical application of ionizing radiation. X-ray examinations comprise a significant part of medical exposure. Doses received by patients and quality of diagnostic images are to be optimized. Measurements of these doses were started by the Radiation Protection Centre in 1997. These measurements are performed in randomly selected x-ray departments all around Lithuania during examinations of chest and lumbar spine. Dose and parameters related to exposure and patient are registered. Quality control measurements by PMX-III are being performed on each x-ray machine used for examination. The results show that in many cases the guidance levels are of entrance surface dose for standard patient determined by the Basic Radiation Protection Standard of Lithuania are exceeded. Quality control of x-ray machines performed in 1997-1999 shows that more than 30% of these machines did not comply with the requirements though in many cases shortcomings are minor and easily removed.

## **INTRODUCTION**

One of the most important sources of human exposure is medical application of ionizing radiation. X-ray examinations comprise a significant part of medical exposure. According to (IAEA, 1996) "registrants and licenses should ensure that guidance levels for medical exposure be determined ... and used as guidance" and "representative values for typical sized adult patients of entrance surface doses, dose-area products, dose rates and exposure times shall be determined and documented" in the case of radiological examinations. Determination of patients' doses helps not only to establish national reference levels but it is a powerful tool in creation of the system of quality assurance. It is possible to show variations of these doses and to prove efficiency and necessity of such system though patients' doses depend on quality of work of medical practitioners, techniques of examinations, performance of equipment.

The measurements of doses received by patients during x-ray examinations in Lithuania started in 1997. This report presents results of these measurements and implications of such survey.

## **METHODS**

Measurements are performed in randomly selected x-ray departments all around Lithuania during examinations of chest (PA and LAT) and lumbar spine (AP, LAT and LSJ). LiF pellets in black bags are taped on the body of the patient in the centre of x-ray field in the direction to the x-ray tube. The RADOS dosimetric equipment is used for registration of entrance surface doses. Since the system is calibrated in  $H_p(10)$  in the field of Cs-137 the coefficient of 0.68 has been used to calculate entrance surface doses (Ennow, 1999). Such parameters as sex of patient, his/her weight, height and thickness in the centre of x-ray field, exposure parameters such as kVp, mA or mAs, size of x-ray field, focus-film distance, total filtration are written down in the special protocols.

Quality control measurements (waveform, accuracy of kVp, reproducibility of kVp and dose, linearity of dose, accuracy of timer) are performed with the help of PMX III on each x-ray machine used for examination.

## RESULTS AND DISCUSSION

Guidance levels of doses should be established for standard sized patient (IAEA, 1996). However, measurements in distant hospitals have been performed for all the available patients in order to have more data. Weight distribution of patients presented in Fig.1 shows that in average patients have been of standard weight - the average weight was  $(71 \pm 2)$  kg (95% confidence interval).

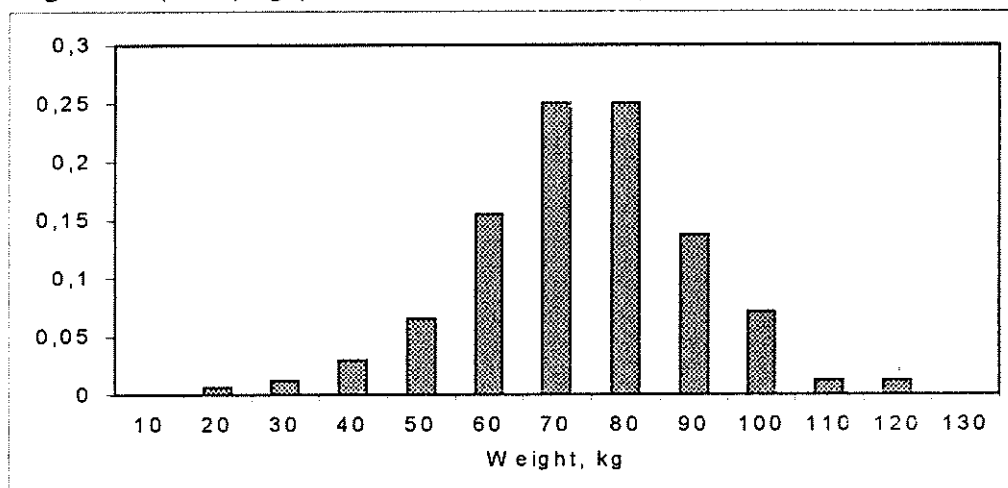


Fig.1. Weight distribution of patients

Results received in departments are presented in Fig.2. Departments are presented in numbers, type of x-ray equipment used for examinations indicated. Columns "All" represent the average entrance surface doses of all the measurements. Columns "Guidance" depict guidance levels recommended by the Lithuanian Basic Radiation Protection Standard.

It is evident that the average entrance surface doses show great variability from one x-ray department to another. It may be connected with different average weights of patients though differences of weight in different departments during different examinations were not essential (in many departments the average weight of patients was  $(66 \pm 12)$  to  $(77 \pm 18)$  kg). Another possible reason for these differences might be different techniques used in different departments and different speeds of screen-film combinations. The last parameter was rather difficult to record because in many cases staff of departments had not any information about it.

In some cases the average entrance surface doses were lower than guidance levels though the overall average with insignificant exception exceed recommended values. It indicates that it is possible to keep guidance levels. On the other hand, measurements of doses have been performed in departments which had turned to the Radiation Protection Centre for quality control measurements. Such departments may have higher level of quality assurance and it causes lower patients' doses.

It is important to emphasise that average entrance surface doses received by patients during photofluorographic examinations in two hospitals (x-ray machines 12F7C) were  $(3.2 \pm 0.7)$  mGy. These doses are well above the averages of chest PA examinations.

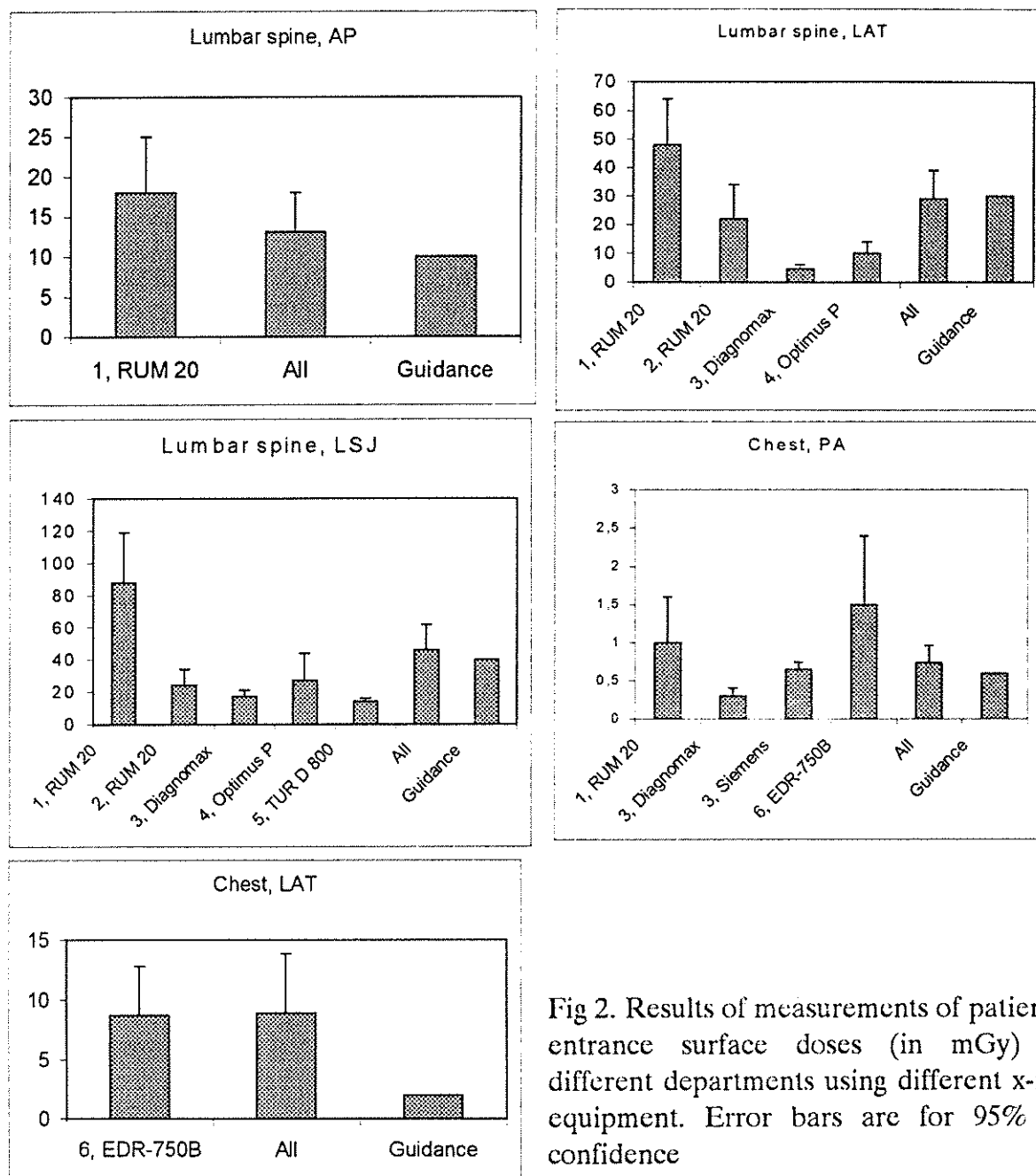


Fig 2. Results of measurements of patients' entrance surface doses (in mGy) in different departments using different x-ray equipment. Error bars are for 95% of confidence

On the basis of received results and (Hart et al, 1994) the effective doses due to chest (PA) examinations have been calculated (see Fig.3). This distribution is similar to the analogous distribution received in Sweden (Leitz, 1999). The average of effective dose received during chest PA examinations is  $(0.06 \pm 0.02)$  mSv. It is comparable with averages in Finland (national, 0.10 mSv), Greece (one hospital, 0.11 mSv), Ghana (12 hospitals, 0.10 mSv), New Zealand (national, 0.02 mSv), United Kingdom (national, 0.02 mSv), Brazil (3 hospitals, 0.021 mSv) (UNSCEAR, 1999). The average effective dose received during photofluoroscopy in two hospitals was  $(0.32 \pm 0.10)$  mSv per image. It should be pointed out that averages of effective doses in two hospitals are rather different -  $(0.46 \pm 0.14)$  and  $(0.14 \pm 0.03)$  mSv per image.

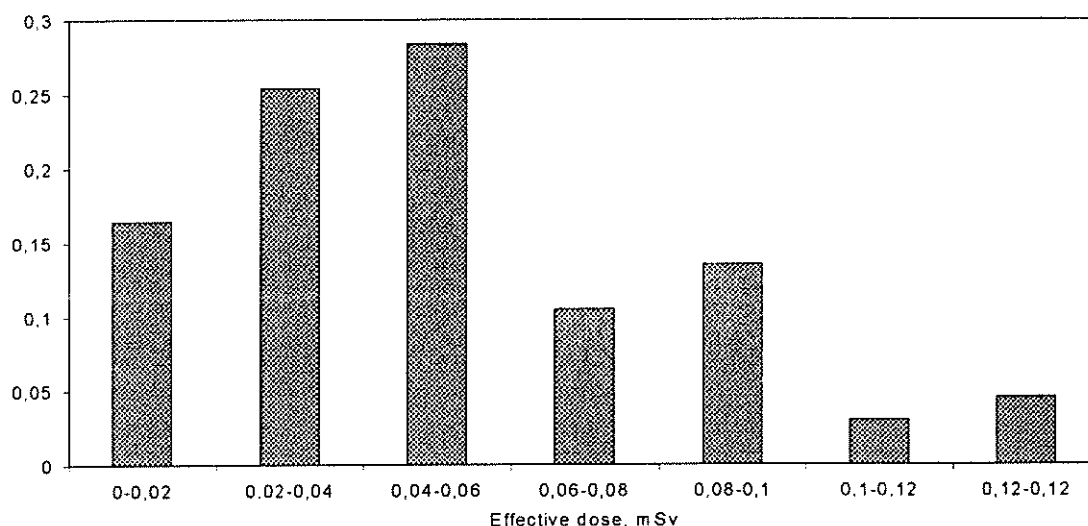


Fig.3. Distribution of effective doses received by patients during chest PA examinations in 10 hospitals

Quality control of 276 x-ray machines was performed in 1997-1999. More than 30% of these machines did not comply with the requirements though in 80% cases shortcomings were minor and easily removed.

The first steps of creation of system of quality assurance in x-ray radiology have been taken. The national standard on acceptability criteria for x-ray equipment has been adopted and guide on quality control by simple means prepared. However, human factor plays an important role, as well.

### CONCLUSIONS

The results show that there are many possibilities to decrease doses received during x-ray examinations. One of possibilities is to improve equipment performance. Quality control measurements may be powerful tool in it. Though equipment under usage is rather old doses received with it may be kept reasonably low in case of suitable technical maintenance of equipment. More close attention shall be paid to justification of photofluorographic procedures because effective doses during these procedures exceed effective doses caused by conventional chest radiography by almost 10 times. It is especially important when keeping in mind that healthy persons are usually undergoing photofluorographic examinations.

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# RADIATION DOSES FROM CONTAMINANT AEROSOL DEPOSITION TO THE HUMAN BODY

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*Abstract:* Nearly all assessments of radiation doses received following accidental airborne releases have focused on the contributions originating from the plume and from ground deposition. Very little thought has however been given to doses received from deposition directly onto humans. The results of recent experimental investigations of aerosol deposition to and clearance from human skin and clothing have been used to model the doses potentially received in an accident situation. It was found that both the skin dose from  $\beta$ -emitters and the whole body dose from  $\gamma$ -emitters may be significant compared with doses received through other pathways, such as external radiation from the environment.

## INTRODUCTION

Recently, an experimental study has been conducted to investigate the parameters and processes that govern the deposition and subsequent behaviour on human body surfaces of radioactive contaminants. The results of this work have been used to model the dose from dry deposition to skin and clothing after an accidental release of radionuclides. Both the dose to skin from  $\beta$ -emitters and the whole body dose from  $\gamma$ -emitters have been considered.

## METHODS

To facilitate comparison with doses received through other pathways, the doses received from direct deposition to humans were calculated relative to the near-surface air concentrations of the various contaminants measurable in Novozybkov, Russia over the first months after the Chernobyl accident, as reported by Kryshev (1996). This reference does not distinguish between the different chemical forms of iodine, but the EU COSYMA model suggests that 99% of the iodine is in the elemental form (with a dry deposition velocity of  $0.01 \text{ m s}^{-1}$ ), and the remaining 1% is in organic form (deposition velocity:  $5 \cdot 10^{-4} \text{ m s}^{-1}$ ). This partitioning has been assumed in the modelling.

With the exception of iodine, for which the chemistry is more complex, the different contaminant aerosol was divided into two groups, according to reported data after the Chernobyl accident. These are the volatile group ( $^{132}\text{Te}$ ,  $^{134}\text{Cs}$ ,  $^{137}\text{Cs}$ ,  $^{99}\text{Mo}$ ,  $^{103}\text{Ru}$  and  $^{106}\text{Ru}$ ) with lower Activity Median Aerodynamic Diameter (AMAD) values of the order of  $0.7 \mu\text{m}$  and the refractory group with higher AMAD values of the order of  $4 \mu\text{m}$ , to which  $^{140}\text{Ba}$ ,  $^{95}\text{Zr}$ ,  $^{141}\text{Ce}$ ,  $^{144}\text{Ce}$ ,  $^{89}\text{Sr}$  and  $^{90}\text{Sr}$  belong.



The estimates of dry deposition velocities to skin were based on the results reported by Roed et al. (1998).

It was assumed in the calculations that the people in the area stayed indoors during virtually the entire deposition phase. The relationship between indoor and outdoor air concentrations was based on typical measurement results in Western European houses of the main influencing factors: air exchange rate, indoor deposition rate and filtering factor, according to the expression of Roed (1990). Indoor surface deposition rates were mainly based on the work of Kocher (1980) and Roed (1990).

The total integrated contaminant skin depositions per unit of area were calculated by multiplication of the time-integrated air concentrations by the indoor/outdoor relationship and relevant skin deposition velocities.

The 'natural' decrease in contamination levels, by 'clearance' and radioactive decay, was taken into account in the modelling of the corresponding doses. This decrease is a critical parameter to the dose calculations. Recent findings (Fogh et al., 1999) suggest that the 'clearance' half-life is about 1.2 days for 2.5  $\mu\text{m}$  particles and only about 0.16 days for 4.5  $\mu\text{m}$  particles. Based on these experimental results and reported indoor resuspension rates for different particle sizes (Thatcher & Layton, 1995), it would be expected that the 'clearance' half-life of 1  $\mu\text{m}$  particles would be one or two orders of magnitude greater than that found for the 2.5  $\mu\text{m}$  particles, i.e. 10-100 days. The clearance is, however, here likely to be governed by the shedding of the stratum corneum over a few weeks, although especially small particles may migrate to deeper skin crevices and hair follicles. The COSYMA modellers estimate the 'clearance' half-life to generally be of the order of 30 days, and this value was applied in the calculations for the small (sub-micron) particles.

For the fraction of the body covered by clothing (here taken to be 80 %), the clearance half-life was based on the recent experimental work of the authors, showing that ordinary washing of cotton would lead to a reduction of the contamination level by a factor of at least ca. 1.4. The washing effect did not appear to be greatly influenced by the particle size in the relevant size range. If it is assumed that the clothes are washed at 2 days intervals, a washing reduction factor of 1.4 corresponds to a clearance half-life of 3.9 days, which was applied in the calculations. The applied deposition velocities to clothing are considered to be typical values, mainly derived from the results of Roed et al. (1998).

Gamma dose contributions were evaluated from Monte Carlo calculations using the MCNP code. A simplified model was applied, in which a tissue equivalent ICRP sphere at 1 m above ground was irradiated by a surface contamination of variable energy. It was found that the sensitivity of this model towards changes in sphere diameter was low.

For freely exposed skin, estimates of beta doses from the different isotopes to the basal layer of the epidermis were based on figures stated in ICRU report 56 (1997), which were found to be in fairly good agreement with the corresponding factors reported by Faw (1992).

Clothing greatly reduces beta skin doses. The Tables in Appendix A of ICRU report 56 show that due to the similar beta attenuation characteristics, it is reasonable to assume that equal mass-thicknesses of clothing and tissue are equivalent in dose considerations. From the dose rate factors given in ICRU 56 it followed that practically no beta doses would be received through 3 mm thick clothing. The beta doses through clothing, which are presented in this study, are for a clothing thickness of 0.4 mm. Again, the factors relating a contamination level with a dose rate were taken from ICRU report 56. These factors are in reasonable agreement with those reported by Taylor et al. (1997) for a 26 mg cm<sup>-2</sup> thick layer of cotton with a density of 0.7 g cm<sup>-3</sup>. Taylor et al. also demonstrated that an air gap of 1 cm between the skin and the clothing could halve the doses received from beta radiation with energies of about 0.4 MeV.

## RESULTS

The results of calculations of gamma doses to the human body and beta doses to the human epidermal skin layer from contamination on skin and clothing are shown in Table 1. These doses are related to measured integrated air concentrations in Novozybkov after the Chernobyl accident (Kryshev, 1996). Also shown in this table are some of the model input data. Complete input data lists are given in Fogh et al. (1999).

**Table 1.** Some parameters applied and results of modelling of radiation doses from airborne contamination of human body surfaces. For each isotope that was measured in near-ground level air in Novozybkov, the following factors are given: radioactive half-life, measured integrated contaminant air concentration at Novozybkov (from 28/04/86), indoor/outdoor air concentration relationships ( $C_i/C_o$ ) and typical deposition velocities ( $V_d$ ) to skin and to clothing. The last four columns show the calculated gamma and beta doses from contamination deposited to skin and clothing.

Isotope	$T_{1/2}$ (radio- active)	Integrated air con- tamination (Kryshev)	$C_i/C_o$	$V_d$ to skin	$V_d$ to clothing	Gamma dose from contamina- tion on skin	Gamma dose from contamina- tion on clo- thing	Skin beta dose from contami- nation on skin	Skin beta dose from contami- nation on clothing
	[ ]	[Bq s m <sup>-3</sup> ]	[ ]	[m s <sup>-1</sup> ]	[m s <sup>-1</sup> ]	[Sv]	[Sv]	[Sv]	[Sv]
Sr-89	50.5d	1.32E+05	0.12	1.20E-02	2.70E-03	0	0	1.80E-07	4.80E-07
Sr-90	28.8y	1.41E+04	0.12	1.20E-02	2.70E-03	0	0	1.67E-08	2.05E-08
Zr-95	64.0d	2.67E+06	0.12	1.20E-02	2.70E-03	4.95E-09	1.45E-07	2.43E-06	9.47E-07
Mo-99	65.9h	5.18E+06	0.50	1.00E-03	1.70E-03	1.03E-08	6.34E-08	3.35E-05	1.65E-05
Ru-103	39.3d	9.10E+06	0.27	1.00E-03	1.70E-03	2.16E-07	4.34E-07	8.95E-05	1.42E-06
Ru-106	374d	2.28E+06	0.27	1.00E-03	1.70E-03	3.24E-08	4.36E-08	1.08E-04	1.73E-05
I-131 (org.)	8.02d	1.79E+06	0.36	5.00E-03	1.00E-03	8.20E-08	3.86E-08	9.65E-05	1.76E-06
I-131 (elem)	8.02d	1.77E+08	0.07	1.00E-02	1.00E-02	3.15E-06	7.43E-06	3.71E-03	3.39E-04
Te-132	3.20d	1.81E+07	0.50	1.00E-03	1.70E-03	5.51E-08	3.21E-07	8.29E-05	6.23E-07
Cs-134	2.06y	8.93E+06	0.50	1.00E-03	1.70E-03	1.73E-06	2.25E-06	4.79E-04	2.65E-05
Cs-137	30.1y	1.78E+07	0.50	1.00E-03	1.70E-03	1.61E-06	2.02E-06	1.47E-03	7.86E-05
Ba-140	12.8d	1.08E+07	0.12	1.20E-02	2.70E-03	3.42E-09	8.26E-08	1.26E-05	1.77E-05
Ce-141	32.5d	3.17E+06	0.12	1.20E-02	2.70E-03	4.74E-10	1.33E-08	4.02E-06	2.09E-06
Ce-144	285d	2.14E+06	0.12	1.20E-02	2.70E-03	9.35E-11	2.87E-09	1.54E-06	2.64E-07
Np-239	2.36d	1.68E+07	0.12	1.20E-02	2.70E-03	2.58E-09	3.26E-08	2.15E-05	0

## CONCLUSIONS / DISCUSSION

From Table 1 it can be seen that in the example, iodine and caesium contribute most to both beta and gamma doses.

The total gamma dose contribution from body contamination amounts to some 20  $\mu\text{Sv}$ . This can be compared with for instance the external gamma dose contribution from contamination on surfaces in the environment (e.g., soil, roofs, walls of buildings and pavings). As it rained when the cloud from Chernobyl passed Novozybkov, the applied air concentrations would correspond to a total  $^{137}\text{Cs}$  dry deposition on a lawn of only 8  $\text{kBq/m}^2$ . The external gamma dose corresponding to this would be of the order of 10-50  $\mu\text{Sv}$  over the first year, depending on the degree of shielding provided by the dwellings and the fraction of time spent indoors (Andersson, 1996). Dry deposition after the Chernobyl accident actually led to contamination of living areas in Russia by levels which were several hundred times higher. Here, the skin contamination would give a gamma dose of several  $\text{mSv}$ .

Table 1 also shows that the skin beta doses amount to ca. 6  $\text{mSv}$  to directly exposed skin and 0.5  $\text{mSv}$  to skin protected by a 0.4 mm thick cotton layer. With a contamination level several hundred times greater, the skin dose would be several  $\text{Sv}$ . This dose may be considered in relation to the risk, according to ICRP, of skin cancer mortality and morbidity (respectively  $2 \cdot 10^{-4} \text{ Sv}^{-1}$  and  $9.8 \cdot 10^{-2} \text{ Sv}^{-1}$ ).

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# MIKROKJERNER OG APOPTOSE I HUMANE LYMFOCYTTER

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## SAMMENDRAG

Til oppbygging av biologisk dosimetri ved strålingsulykker eller strålingsuhell, har vi i Norge valgt å satse på mikrokjernemetoden brukt på humane lymfocytter fra perifert blod. Antall mikrokjerner influeres antakelig av hvilken tilbøyelighet et individs celler har til å gå i apoptose. For å få et mer fullstendig bilde av den skade som er oppstått etter bestråling, ville det derfor være interessant å registrere forekomsten av både mikrokjerner og apoptotiske celler.

Vi har bestrålt fullblod med røntgenstråling (0-2 Gy), og antall mikrokjerner i binukleære cytokineseblokkerte lymfocytter er telt. Frekvensen av mikrokjerner i lymfocytter fra fullblod og isolerte lymfocytter er sammenlignet. Isolerte lymfocytter er fiksert for apoptose 24 timer etter stimulering, og apoptose er detektert med in situ terminal-deoksynukleotidyl-transferase assay ved flowcytometri.

Foreløpige resultater viser at det ikke er vesentlig forskjell i antall mikrokjerner i lymfocytter fra fullblod og i isolerte lymfocytter. Videre har vi funnet forskjellig nivå av apoptose i ubestrålte isolerte lymfocytter, og at det er mer apoptose i bestrålte enn i ubestrålte isolerte lymfocytter. Det ser ikke ut til å være vesentlig mer apoptose etter 2 Gy enn etter 1 Gy, men det er foreløpig for tidlig å trekke noen konklusjoner om doseavhengigheten. Dette må studeres mer inngående.

Ved eventuelle strålingsulykker eller strålingsuhell, vil man i de aller fleste tilfeller ikke vite frekvensen av spontane mikrokjerner hos de bestrålte personer. Forekomsten av spontane mikrokjerner varierer mye fra person til person. Ved lave doser kan det derfor være vanskelig å anslå stråledosen kun ved hjelp av «tradisjonell» mikrokjernemetode. Ved også å vurdere andre tilgjengelige parametre, deriblant graden av apoptose, vil man kunne få bedre informasjon om hvor stor skade som er oppstått på grunn av bestrålingen.

## INNLEDNING

Et av problemene ved å bruke mikrokjernemetoden til biologisk dosimetri, er at man ikke vet frekvensen av spontane (dvs pre-eksisterende før bestråling) mikrokjerner hos den bestrålte personen. De siste års forskning har vist at spontane mikrokjerner langt oftere inneholder sentromerer enn det strålingsinduserte mikrokjerner gjør, og ved å finne metoder for å skille mellom de to typene mikrokjerner (sentromer-positive og -negative), har man klart å øke metodens sensitivitet noe. Det er imidlertid ikke sikkert om lymfocytter med pre-eksisterende mikrokjerner, som lar seg stimulere ex vivo, kan uttrykke sin opprinnelige mikrokjerne i resulterende binukleære celler eller om den dør av apoptose før kjernen deler seg (Fenech 1998). I de siste årene har det blitt allment akseptert at apoptose er den type celledød som forekommer etter biologisk relevante strålingsdoser for veldig mange celletyper, og da også for humane lymfocytter. Flere studier har indikert at strålingsindusert apoptose i humane lymfocytter kan være et potensielt biologisk dosimeter (Boreham et al. 1996, Fenech 1998).

Ved å studere forekomsten av både mikrokjerner og apoptose i lymfocytene, ville man kanskje få et bedre bilde av hvor stor strålingsskade den bestrålte personen har fått.

Hensikten med denne studien var å undersøke om det er noen sammenheng mellom frekvensen av mikrokjerner og apoptose i parallellprøver av humane lymfocytter fra den samme blodprøve etter røntgenbestråling.

## MATERIALER OG METODER

### Bestråling og cellekulturer

Prøver med heparinisert blod ble tatt ved venepunktering av to friske, ikke-røykende, kvinnelige donorer på 44 og 26 år (henholdsvis donor 1 og 2). Fullblod ble bestrålt i porsjoner à 2 ml med 0, 1.0 og 2.0 Gy røntgenstråling (250 kV, 12 mA, 3 mm Al-filter) med en doserate på 1 Gy/min i romtemperatur.

I forsøkene med isolerte lymfocytter, ble lymfocytene isolert umiddelbart etter bestråling med NycoPrep<sup>TM</sup> 1.077 (Nycomed Pharma) og tilsatt kulturmedium bestående av RPMI 1640 (BioWhittaker), 20% føytalt kalveserum, L-glutamin og antibiotika. For hvert dosenivå ble minst tre paralleller à 1 ml ( $4 \times 10^5$ - $1 \times 10^6$  lymfocytter pr. ml) satt opp for mikrokjernemetoden og én kultur med  $1$ - $2 \times 10^6$  lymfocytter i 5 ml kulturmedium for detektering av apoptose. For mikrokjernemetoden benyttet på fullblod, ble 0.1 ml fullblod tilsatt 1 ml kulturmedium rett etter bestråling. Phytohemagglutinin-P (PHA-P; Sigma) ble umiddelbart tilsatt alle kulturer til en sluttkonsentrasjon på 10 µg/ml.

### Mikrokjernemetoden

42 timer etter tilsats av PHA-P ble cytochalsin-B (Sigma) (sluttkonsentrasjon 6 µg/ml) tilsatt for å stoppe cytokinesen. Etter 70 timer ble cellene sentrifugert, inkubert i 0.125 M KCl i 6 min ved romtemperatur, deretter fiksert en gang i iskald fikseringsløsning I (metanol: NaCl: eddiksyre = 12:13:3) i 5 min og så i iskald fikseringsløsning II (metanol: eddiksyre = 4:1) i 5 min. Cellene ble deretter vasket flere ganger i fikseringsløsning 2 uten inkubering til supernatanten var klar, og applisert på kalde, rene objektglass. Preparatene ble farget med Giemsa (Sigma), og antall mikrokjerner i 500 binukleære celler pr. preparat ble telt i lysmikroskop.

### Terminal-deoksynukleotidyl-transferase (TdT) assay for detektering av apoptose

Prosedyren er hentet fra Gorczyca (Gorczyca et al. 1993) med visse modifikasjoner (Stokke et al. 1998). 24 timer etter stimulering ble lymfocytene fiksert i 1% paraformaldehyd i PBS ved 4°C etterfulgt av fiksering i 100% metanol, og lagret i -20°C. For å detektere apoptose ble de fikserte cellene preparert for to-farge fluorescensanalyse ved flowcytometri. Fargingsprosedyren inkluderte TdT, biotinyleret dUTP og streptavidin-FITC for å merke frie DNA-ender som er et resultat av apoptose. Cellene ble også farget for DNA-innhold med propidium iodid (PI). Rød fluorescens (PI) og grønn fluorescens (FITC) ble målt med et FACS Vantage flowcytometer, og tilhørende software ble benyttet til å estimere fraksjonen av apoptotiske celler.

## RESULTATER

### Mikrokjerner

Tabell 1 viser resultatene fra mikrokjernemetoden brukt på lymfocytter fra fullblod og isolerte lymfocytter fra donor 1. Forskjellen i frekvens av mikrokjerner i lymfocytter fra

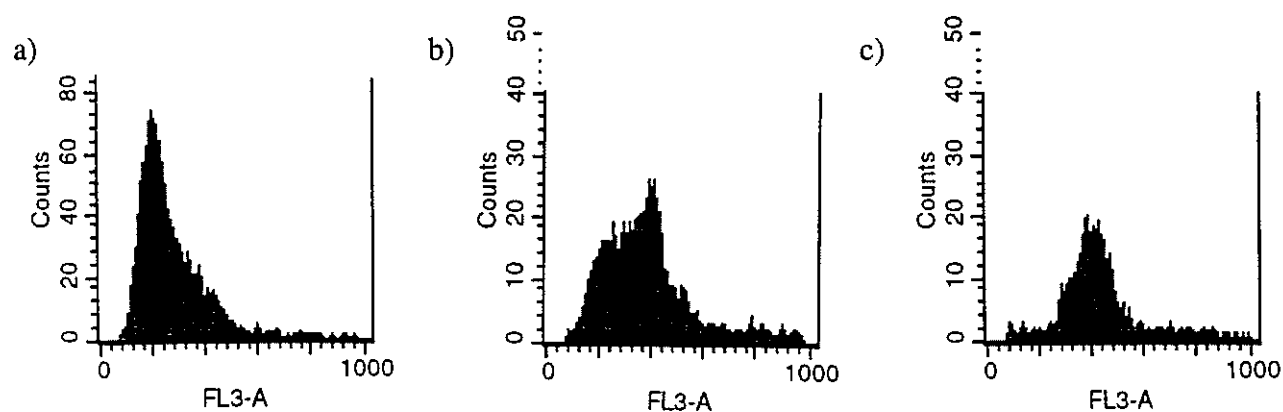
fullblod og i isolerte lymfocytter ikke er statistisk signifikant ( $p = 0.01$ ) etter bestråling med 0 og 1.0 Gy røntgenstråling. Forskjellen er statistisk signifikant ( $p = 0.05$ ) etter 2 Gy. For donor 2 har vi ikke tilstrekkelig med resultater til å kunne foreta denne sammenligningen.

**Tabell 1.** Frekvensen av mikrokjerner i binukleære (BN) lymfocytter fra donor 1.

Dose (Gy)	Antall mikrokjerner pr. 500 BN lymfocytter fra fullblod $\pm 1$ st.d.	Antall mikrokjerner pr. 500 BN isolerte lymfocytter $\pm 1$ st.d.	Statistisk signifikant
0.0	8.18 $\pm$ 2.23	7.38 $\pm$ 1.04	Nei ( $p = 0.05$ )
0.2	12.00 $\pm$ 4.58	-	-
0.5	14.60 $\pm$ 3.89	-	-
1.0	28.82 $\pm$ 10.83	23.00 $\pm$ 5.08	Nei ( $p = 0.05$ )
2.0	44.38 $\pm$ 7.63	65.60 $\pm$ 14.56	Ja ( $p = 0.01$ )

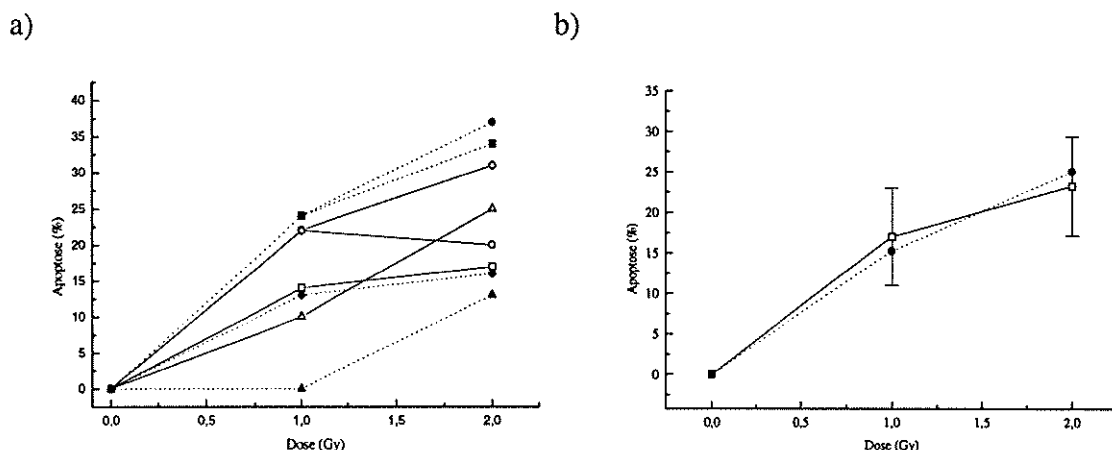
### Apoptose

Fig. 1 viser DNA-profil 24 timer etter stimulering for de isolerte lymfocytene fra ett av forsøkene med blod fra donor 1 målt med flowcytometer. De ubestrålte lymfocytene befinner seg hovedsakelig i G<sub>0</sub>/G<sub>1</sub>- og S-fase, mens man ser en tendens til forskyvning mot G<sub>2</sub>-fase etter bestråling. Dette tyder på at cellene arresteres i G<sub>2</sub>-fase for å reparere skader etter bestrålingen.



**Fig. 1.** DNA-profil for isolerte lymfocytter fra ett apoptoseforsøk med blod fra donor 1 etter bestråling med a) 0 Gy, b) 1.0 Gy og c) 2.0 Gy røntgenstråling. Abscissen viser DNA-innhold og ordinaten viser antall celler.

Fig. 2 viser resultatene fra de fire apoptoseforsøkene med hver av donor 1 og 2. Fra figuren kan det se ut som om prosent strålingsindusert apoptose for humane lymfocytter ikke øker proporsjonalt med stråledosen, men flater ut ved høyere doser (opp mot 2 Gy). Gjennomsnittet av fire forsøk med hver av de to donorene gir en nesten identisk dose-responskurve for de to donorene (fig. 2b).



**Fig. 2.** Apoptose (%) som funksjon av dose (Gy), normalisert til 0 for de ubestrålte prøvene, for isolerte lymfocytter fra donor 1 (—) og 2 (.....) detektert med *in situ* terminal-deoksynukleotidyl-transferase assay ved flowcytometri. a) Resultater fra hver av de fire forsøkene med hver donor og b) gjennomsnittsverdier for hver donor  $\pm$  1 standard avvik.

## KONKLUSJON

Ut fra våre foreløpige studier er det for tidlig å si noe om hvorvidt det er noen sammenheng mellom frekvens av mikrokjerner og apoptose i parallellprøver av isolerte lymfocytter fra den samme blodprøven. Kurveforløpet ser forskjellig ut for de to «endpoints». Prosent apoptose som funksjon av dose gir en kurve som ser ut til å flate ut for høyere doser, mens frekvensen av mikrokjerner fortsetter å stige i det doseområdet (0-2 Gy) vi har studert.

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# Uncertainties involved in the calculation of intakes and committed effective doses from whole-body measurements

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**Abstract.** Determination of committed effective doses from intakes of radionuclides can be done on the basis of measurements in a whole-body counter. Uncertainties in the determination of the content of radionuclides in the body, the uncertainties in determining the intake from the measurements and finally the uncertainties in the calculations of the committed effective dose are discussed. For radionuclides detected by whole-body counting at Risø National Laboratory it can be concluded, that the greatest contribution to the uncertainties in determining the committed effective dose is due to lack of knowledge of date of intake and route of intake.

## 1 Introduction

A whole-body measurement determines the activity of  $\gamma$ -emitting radionuclides in the body at the time of the measurement. Based on knowledge and or assumptions about the time of intake, the route of intake and the physical and chemical characteristics of the radioactive material an intake and a committed effective dose can be calculated. The assumptions and the use of biological models introduce uncertainties in addition to the uncertainty on the whole-body measurements themselves.

At Risø National Laboratory whole-body measurements have been made on members of the workforce. In the measurements small amounts of activity of the radionuclides  $^{51}\text{Cr}$ ,  $^{60}\text{Co}$ ,  $^{65}\text{Zn}$ ,  $^{124}\text{Sb}$ ,  $^{131}\text{I}$ ,  $^{137}\text{Cs}$ , and  $^{203}\text{Hg}$  have been detected. For these radionuclides the effect of different assumptions on the calculated intake and committed effective dose has been investigated.

## 2 Sources of uncertainty

The uncertainty in the determination of activity in a body using the whole-body counter arises from these contributions:

- 1) calibration of the whole-body counter
- 2) background correction
- 3) counting statistics
- 4) surface contamination on the person measured

1) includes the contribution from the uncertainty on the activity in the calibration source and on the source/detector geometry. The uncertainty is estimated to be 10-25%. The contribution from 2) varies with time and depends on  $\gamma$ -energy and on the concentration of radon and



daughter products in the counting room. The contribution from 3) depends on the amount of radioactivity in the body and on the counting time. When the detected activity contents are slightly above the level of detection 2) and 3) introduce an uncertainty of approximately 30-50%. 4) can introduce very large uncertainties and the contribution is difficult to quantify. At the Risø whole-body counter the persons to be measured are asked to wash and change to "clean clothes". The contribution from 1), 2) and 3) when measuring an activity content slightly above at the lower level of detection is therefore 30-55%.

When an intake and a committed effective dose are calculated from a whole-body activity content further sources of uncertainty have to be considered. To do the calculation intake parameters and biological retention model must be selected. Knowledge of the intake parameters and the proper biological model might be known to some extent but often they must be assumed. Finally the biological models are encumbered with sometimes large inherent uncertainties. These are seldom taken into account, as they are difficult to evaluate.

### 3 Calculations

The computer programme Ludep was used to calculate intakes and committed effective doses for the above mentioned seven radionuclides based on a body content of 1 Bq. Calculations were made for different values of the parameters: days since intake, route of intake, breathing mode, lung deposition, lung clearance, inhalation class and activity mean aerodynamic diameter. One set of parameters for each nuclide was considered to be the standard assumption relevant for Risø conditions. The standard assumptions are shown in table 1 together with the variations taken into account in the calculations.

*Table 1. Standard assumptions and variations in the parameters used in the calculation of intake and committed effective dose. (slow, medium, fast: lung clearance rates, D, W, Y: inhalation classes, amad: activity mean aerodynamic diameter).*

Radionuclide	Standard assumption common: inhalation, nose breathing, 45 days since intake, amad=5 µm	Variations common: ingestion, mouth breathing, 1, 7, 30, and 90 days since intake, amad=1 µm, 10 µm
<sup>51</sup> Cr	medium, W, bone volume seeker	slow, fast, D, Y
<sup>60</sup> Co	medium, W	slow, W, Y
<sup>65</sup> Zn	medium, W, bone volume seeker	slow, fast, Y
<sup>131</sup> I	fast, D	
<sup>137</sup> Cs	fast, D,	
<sup>124</sup> Sb	medium, W	fast, D
<sup>203</sup> Hg	medium, W, inorganic compound	fast, D, vapour

### 4 Results

Table 2 and table 3 gives the variations in the calculated intakes and the committed effective doses. The numbers in the tables are the ratios between the calculated values for the non-standard assumptions to the values for the standard assumption. The columns two to six in the tables gives the values when only one parameter (named in the column head) in the standard assumptions is varied. The last column in the tables gives the interval of variation when all parameters are allowed to vary at the same time.

**Table 2.** Variation in calculated intake when one of the parameters in the standard assumption is changed. Days since intake (dsi), route of intake (roi), clearance and inhalation class (msf/dwy), activity median aerodynamic diameter (amad) and mouth breathing (mouth) are considered. Also the maximum variation when all the parameters can be varied are shown. The numbers in the table are the intake values relative to the intake calculated when using the standard assumptions (np: variation not possible).

Radionuclide	Variation in calculated intake when a given parameter is change in the standard assumption					Possible maximum variation in calculated intake
	dsi	roi	msf/dwy	amad	mouth	
<sup>51</sup> Cr	0.04-3.79	25	0.84-1.09	0.59-1.88	0.43	0.02-100
<sup>60</sup> Co	0.04-2.25	5.4	0.88-0.92	0.58-1.88	0.42	0.04-9.0
<sup>65</sup> Zn	0.33-1.17	0.50	0.67-1.00	1.00-1.17	0.50	0.17-1.50
<sup>131</sup> I	0.02-62	0.47	n.p.	1.13-1.38	0.83	0.02-62
<sup>137</sup> Cs	0.55-1.32	0.48	n.p.	1.13-1.39	0.81	0.32-2.16
<sup>124</sup> Sb	0.04-2.51	115	3.8	0.53-2.17	0.40	0.03-300
<sup>203</sup> Hg	0.06-2.88	n.p.	0.41	0.62-1.74	0.44	0.03-5.29

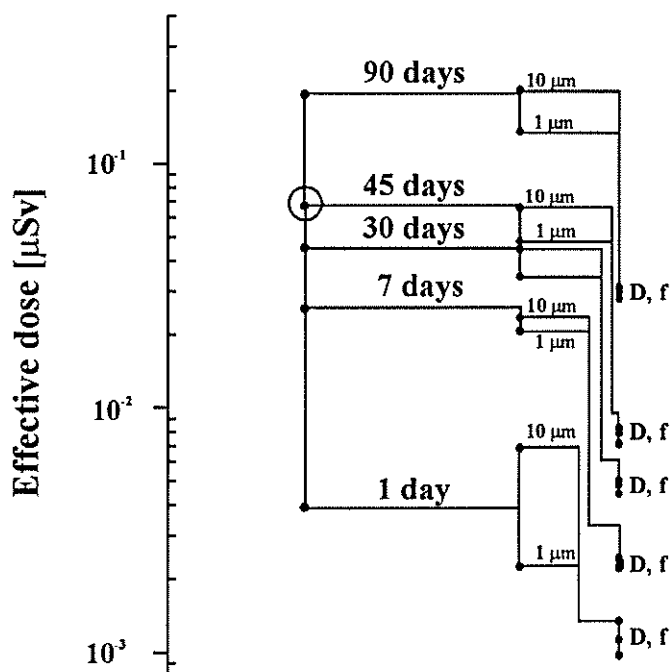
**Table 3.** Variation in committed effective dose when one of the parameters in the standard assumption is changed. Days since intake (dsi), route of intake (roi), clearance and inhalation class (msf/dwy), activity median aerodynamic diameter (amad) and mouth breathing (mouth) are considered. Also the maximum variation when all the parameters can be varied are shown. The numbers in the table are the committed effective dose values relative to the intake calculated when using the standard assumptions (np: variation not possible).

Radionuclide	Variation in committed effective dose when a given parameter is changed in the standard assumption					Possible maximum variation in calculated committed effective dose
	dsi	roi	msf/dwy	amad	mouth	
<sup>51</sup> Cr	0.04-3.79	28	0.64-1.07	0.53-1.54	0.67	0.02-100
<sup>60</sup> Co	0.04-2.25	5.4	1.00-2.43	0.93-1.00	0.42	0.02-3.46
<sup>65</sup> Zn	0.33-1.17	0.90	0.90-1.14	0.99-1.01	0.90	0.29-1.98
<sup>131</sup> I	0.02-62	0.96	n.p.	0.99-1.00	1.00	0.02-62
<sup>137</sup> Cs	0.55-1.32	0.98	n.p.	0.99-1.00	0.96	0.52-1.32
<sup>124</sup> Sb	0.04-2.51	59.6	1.12	0.69-1.50	1.06	0.01-100
<sup>203</sup> Hg	0.06-2.88	n.p.	0.12	0.72-0.99	0.44	0.02-4.16

From the tables it can be seen that a lack of knowledge on the parameters needed to calculate intake and committed effective dose introduce very large uncertainties in the calculations. The most important contributors to these uncertainties are lack of knowledge on the days since intake and the route of intake. Less important is lack of knowledge on lung deposition, lung clearance, breathing mode and activity mean aerodynamic diameter. However for the latter

three the uncertainties introduced are comparable or larger than the uncertainty on the activity measurement in the whole-body counter.

Figure 1 shows the committed effective dose for a body content of  $^{203}\text{Hg}$  when days since intake, activity mean diameter, and inhalation class and clearance are varied. It is clearly seen, that days since intake have the largest influence on the determination of the committed effective dose.



**Fig. 1.** Committed effective dose for a body content of 1 Bq  $^{203}\text{Hg}$ . Days since intake, activity mean diameter, and inhalation class and clearance are varied.

The seven radionuclides can be grouped into three according to the effect of the intake parameters on the calculated intake and committed effective dose. This is shown in table 4.

**Table 4.** The seven radionuclides grouped according to sensitivity of the calculated intake and committed effective dose on the intake parameters. The body contents of activity that would result in a committed effective dose of 1 mSv for the worst combination of intake parameters are also given.

Group I (most sensitive)		Group II		Group III (least sensitive)	
$^{131}\text{I}$	5 Bq	$^{60}\text{Co}$	1800 Bq	$^{137}\text{Cs}$	36950
$^{124}\text{Sb}$	50 Bq	$^{203}\text{Hg}$	3650 Bq	$^{65}\text{Zn}$	38450
$^{51}\text{Cr}$	5350 Bq				

The table also gives the body contents of the nuclides that in the “worst case” would result in a committed effective dose of 1 mSv. For body contents at this level or above detailed knowledge of the important intake parameters should be obtained.

## AlN CERAMICS AS A DETECTOR FOR UV EXPOSURE

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### Abstract

AlN-Y<sub>2</sub>O<sub>3</sub> ceramics is proposed for application in the field of UV detection and dosimetry. Both thermoluminescence (TL) and optically stimulated luminescence (OSL) signals from the material have been studied after exposure to UV light. AlN-Y<sub>2</sub>O<sub>3</sub> ceramics demonstrates very high sensitivity to UV light over a broad spectral region. The TL is characterized by a linear dose dependence over a large range. The fading rate of the UV-induced TL and OSL signals on storage at room temperature is lower than in the case of exposure to ionizing irradiation.

### Introduction

Recently it has been studied whether AlN can be used as TL and OSL detector of ionizing radiation (Trinkler *et al.*, 1998; Trinkler *et al.*, 1999). It was found that AlN ceramics has some advantageous features as a TL dosimeter such as very high sensitivity to ionizing radiation, large dynamic range of dose response, good re-usability, simple annealing requirements and small influence of heating rate on the TL signal. A serious disadvantage of AlN is a high fading rate of the TL signal after exposure to ionizing radiation occurring during storage at room temperature. The OSL signal was found to be much weaker and less stable than the TL signal.

AlN ceramics might also find practical application in the field of ultraviolet (UV) dosimetry. The present paper summarizes the results of a preliminary study of TL and OSL characteristics of AlN ceramics after UV irradiation in order to estimate its capability as a UV detector.

### Materials and equipment

AlN-Y<sub>2</sub>O<sub>3</sub> ceramics were prepared at the Institute of Inorganic Chemistry, Latvia (Palcevskis *et al.*, 1999) by sintering fine grain powders of AlN and Y<sub>2</sub>O<sub>3</sub>, the latter used as a sintering aid. Grown opaque ceramics cylinders were cut into 1mm thick tablets. Luminescence of the material (a strong broad luminescence band in the region of 400 nm dominating over weaker bands in the regions of 500-600 nm and 800-1000 nm) (Youngman *et al.*, 1990) is caused by the donor-acceptor pair recombination processes due to presence of oxygen-related defects in the AlN crystalline grains ( $E_g=6.2$  eV). As AlN ceramics is a light-sensitive material, special precautions have been undertaken to prevent samples from undesirable exposure to light.

For comparative measurements we used samples of Al<sub>2</sub>O<sub>3</sub>:C which is one of the most sensitive material introduced up to now for TL and OSL dosimetry of ionizing radiation and UV light (Akselrod *et al.*, 1990; Botter-Jensen *et al.*, 1997).

All TL and most of the OSL measurements were carried out using the Riso model TL/OSL-DA-12 reader with linear planchet heating, equipped with a built-in software controlled <sup>90</sup>Y/<sup>90</sup>Sr beta source and powerful blue LEDs (Botter-Jensen *et al.*, 1999). To separate luminescence from the stimulation light a U340 glass filter was used. For most of the measurements UV irradiation was fulfilled using a Sol 2 device (Dr. K. Honle GmbH), simulating a solar spectrum at the ground level, beginning from 300 nm. UV regions were selected with sets of glass filters. TL dependence on UV dose for the spectral region 300-400 nm was measured using a TL/05 lamp (Philips) and a UFS-1 glass filter, and estimating the intensity of the light by an optical power meter (model 840, Newport). OSL spectra were also measured in Latvia, using a deuterium lamp LDD-400 and a grating monochromator MDR-2.

The TL readings of AlN and Al<sub>2</sub>O<sub>3</sub>:C samples were performed by heating up to 600 °C in 90 s and 300 s, respectively. TL lightsum was completely bleached during the TL reading and no additional annealing was necessary. For all OSL measurements a pre-irradiation heating up to 600 °C in the reader was used.

### TL characteristics

The most attractive feature of AlN-Y<sub>2</sub>O<sub>3</sub> ceramics is a very high sensitivity to UV irradiation, demonstrated in Fig.1. The TL response of AlN to beta radiation is much higher, than that of Al<sub>2</sub>O<sub>3</sub>:C (1 and 3), but the effect of UV irradiation is even more striking, providing a sensitivity ratio of approximately 6000 (2 and 4). The TL glow curve of a UV-dosed AlN sample is a very broad, structureless band, shifted to significantly higher temperatures compared to that obtained for exposure to beta radiation. As a result a lower fading rate of the integrated TL signal was observed for AlN samples exposed to UV light (decrease to 60 % in 24 hours of storage) than for samples exposed to beta radiation (44 %).

The TL excitation spectrum is an important characteristic of a UV dosimeter, in that it determines the spectral region of its application. Solar emission is divided into UV-C (200-280 nm), UV-B (280-315 nm) and UV-A (315-400 nm) regions. Fig 2. (1) shows the TL excitation spectrum for AlN sample under solar emission at the ground level, simulated by a Sol 2 device. Each point corresponds to TL response after UV, normalized to integral transmission of the definite set of filters (TL response is not corrected to spectral distribution of the UV source), while the line segment indicates the transmission halfwidth of the set of filters used. The spectrum demonstrates that TL from AlN ceramics is sensitive in the UV-A region up to 360 nm, and especially in the UV-B region.

Different UV doses were obtained by varying irradiation time and distance between the sample and UV lamp and using sieve filters of different density. Fig.3 demonstrates a linear dependence of TL response versus UV dose over a wide range of at least 5 orders of magnitude. Deviation from the linear dependence observed at the highest doses could be explained by fading effects, actual for long irradiation time.

### OSL characteristics

Fig. 2. demonstrates the main spectral characteristics of OSL. OSL emission band (4) is centered at 390 nm, slightly shifted to lower energies compared to the photoluminescence spectrum (3), observed directly under UV irradiation. The stimulation spectrum covers a wide spectral range from 420 to 1500 nm, with a dominant band at 500 nm. The OSL excitation spectrum (2) was measured over a limited spectral range (200-300 nm). OSL can be stimulated over the whole region (UV-A) and has a well defined band at 245 nm.

A comparison of OSL decay curves obtained from AlN and Al<sub>2</sub>O<sub>3</sub>:C after both beta and UV light irradiation is shown in Fig. 4. In contrast to Al<sub>2</sub>O<sub>3</sub>:C, showing the same shape of decay curves for both types of irradiation, AlN appeared to have two different decay curves: 1) an intensive fast component that dominates in the beta-induced OSL curve and 2) a slow component that dominates in the UV-induced OSL curve. Although the initial OSL intensity of the AlN sample is significantly higher than that of Al<sub>2</sub>O<sub>3</sub>:C when exposed to beta radiation, this very fast AlN component yields less signal (approximately half of that of Al<sub>2</sub>O<sub>3</sub>:C). However, in the case of UV exposure, the total OSL sensitivity of AlN is about 60 times greater than that of Al<sub>2</sub>O<sub>3</sub>:C.

Fading of the OSL signal during storage of the dosed AlN samples originates from the initial part of the decay curve. That is why the fading effect is higher for beta-induced OSL which has a dominant fast component of the decay curve: only 10 % of the initial response is left after 24 hours of storage. The effect of fading is less pronounced for UV-induced OSL having a dominant slow component: 45 % of the initial response is left after 24 hours.

It should be mentioned that for both irradiation types, after the complete optical bleaching of the lightsum of AlN samples, a considerable TL signal is still observed.

The OSL response from UV exposure deviates from a linear response (Fig. 5.). This could be explained by the experimental procedure rather than the intrinsic property of the material: 1) the UV dose was varied by increasing the irradiation time, and the OSL signal was effected by the fading during irradiation, 2) all the OSL readings were performed during 300 s, which was not satisfactory for a complete read-out of the samples irradiated with large doses and therefore a part of the signal remained unregistered. Experiment should further be performed using varying UV doses obtained by changing the intensity of the incident light and monitoring OSL curves until complete bleaching.

### Discussion and conclusions

Application of AlN-Y<sub>2</sub>O<sub>3</sub> ceramics in the field of UV detection and dosimetry seems prospective, although the studies of the UV induced TL and OSL features of AlN ceramics are at the starting level and need further development.

The TL from UV irradiated AlN samples is characterized by an extremely high sensitivity (3 orders of magnitude higher compared to Al<sub>2</sub>O<sub>3</sub>:C), a broad excitation spectrum covering at least the UV-B and UV-A regions of solar emission, linear dose dependence in the range of 5 orders of magnitude. Although still considerable, the fading rate of UV-induced TL is lower, than that of beta-induced TL. These features suggest that AlN is a potential material for TL dosimetry of UV light.

The OSL response of AlN -Y<sub>2</sub>O<sub>3</sub> to UV exposure is lower than the TL response, but still essentially higher than that of Al<sub>2</sub>O<sub>3</sub>:C (approximately a factor of 60). It was found that the OSL excitation spectrum includes the 200-300 nm spectral region; however also the 300-400 nm region should be further studied. Using special precautions, a linear dose response can be obtained for OSL. Taking into account these characteristics, the material could be proposed for application in OSL dosimetry of solar emission using portable OSL equipment, which makes the read-out procedure feasible directly in the field.

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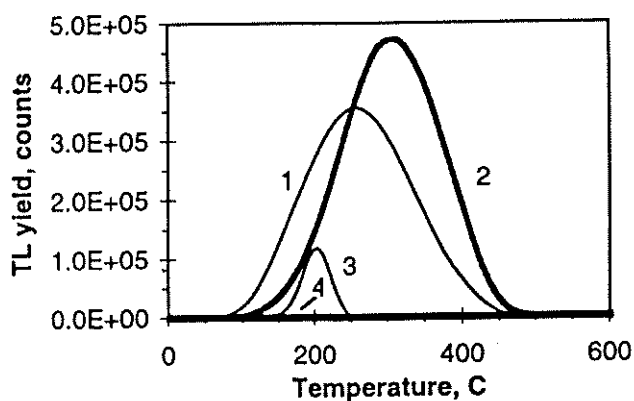


Fig.1. TL curves of  $\text{AlN-Y}_2\text{O}_3$  (mass 0.087g) and  $\text{Al}_2\text{O}_3\text{:C}$  (mass 0.078 g) after irradiation with beta 100 mGy (1 and 3) and UV light (1 min at 350 nm) (2 and 4).

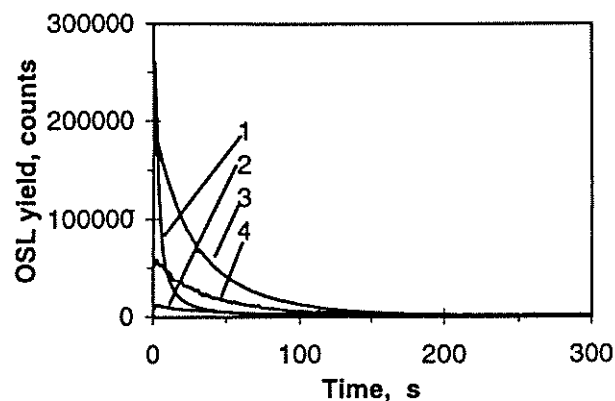


Fig.4. OSL decay curves of  $\text{AlN-Y}_2\text{O}_3$  (mass 0.087g) and  $\text{Al}_2\text{O}_3\text{:C}$  (mass 0.078 g) after irradiation with beta 100 mGy (1 and 3) and UV light (1 min at 350 nm) (2 and 4). Curve 4 is multiplied by a factor of 100.

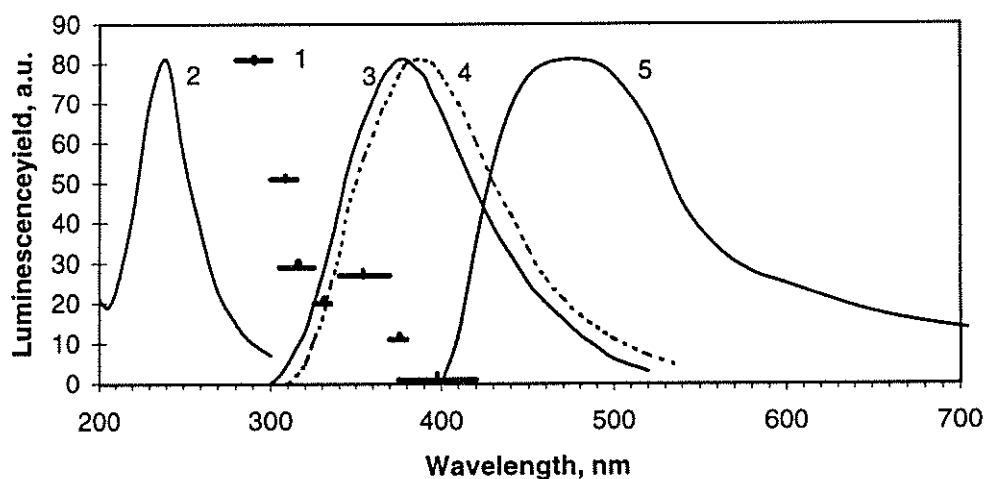


Fig.2. Spectral characteristics of  $\text{AlN-Y}_2\text{O}_3$ : TL excitation spectrum-1(details are given in the text); OSL excitation - 2; emission of photoluminescence-3 and OSL- 4; stimulation spectrum-5.

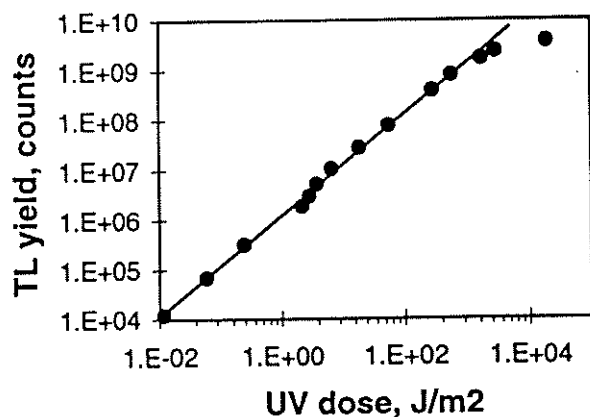


Fig.3. UV dose dependence of TL for  $\text{AlN-Y}_2\text{O}_3$ .

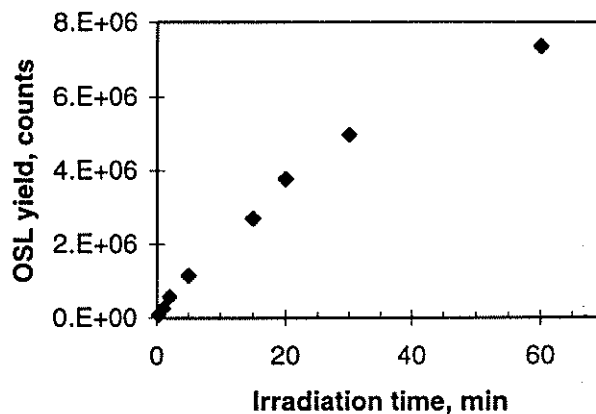


Fig.5. OSL dependence on UV irradiation time for  $\text{AlN-Y}_2\text{O}_3$ .

Nordic Society for Radiation Protection, 12<sup>th</sup> Annual Meeting  
23-27 August 1999, Skagen

## **New way of demonstrating the competence of a laboratory measuring radionuclides - The international draft standard ISO/IEC DIS 17025**

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### **Abstract**

In recent years there has been increased interest, and even need, amongst laboratories performing measurements of radionuclides to obtain accreditation. It has been discussed how this could be achieved with maximum flexibility for the laboratories and with minimum effort. The issuing of a new draft international standard, the ISO/IEC DIS 17025, created speculations whether it could offer a new and better way for laboratories to obtain accreditation. It was decided within the NKS/BOK-1.1 project to explore possible options for obtaining accreditation and what possibilities the new standard could offer. The benefits of computerised document control systems were also explored. The results were reported at the 12<sup>th</sup> Annual Meeting of the Nordic Society for Radiation Protection, 23-27 August 1999. Since then the final version of the standard has been published. The voting will continue until November 16<sup>th</sup> 1999 and is not clear at present whether the standard will be accepted or not. The original version of this paper was updated to reflect these recent developments.

### **Introduction**

Increased international exchange of data on environmental radioactivity has made it important for laboratories to be able to demonstrate the quality of their measurements. The formal way of demonstrating this is through obtaining accreditation for the type of measurements in question. The number of accredited laboratories is growing fast and this in turn increases the pressure on others. Authorities have also started to require laboratories that perform certain types of tests to be accredited.

A few radionuclide laboratories in the Nordic countries have already obtained accreditation for some (or all) of their measurements or are in the process of doing so. Many other laboratories are considering accreditation and how it is best to maximise the benefits of this time consuming and costly procedure. There has also been concern over to what degree accreditation limits the flexibility of laboratories to use best available methods for their work and to develop new methods.

The NKS (Nordic Nuclear Safety Research) has been a forum for discussing various aspects of quality assurance of radionuclide laboratory measurements in the Nordic countries. In the current project period this has been done within the **NKS/BOK-1.1 project "Laboratory measurements and quality assurance"**. This work was stimulated by statements that had been made by various people, indicating that a new international draft standard, the ISO (International Standard Organisation) DIS (Draft International Standard) 17025, would make it easier for laboratories to obtain accreditation, while allowing more flexibility than was previously possible. It was decided within the project to investigate this further, as well as what approach in general might be suitable for laboratories measuring radionuclides.



Representatives of laboratories and accreditation bodies were interviewed, and a professional consultant helped to introduce the various quality standards and their interpretation and use. The author must, however, be held responsible for the views put forward in this paper.

After the NSFS meeting in August 1999, the Final Draft version of the standard was published, ISO/IEC FDIS 17025. The standard was also published as a Final Draft European Standard, prEN ISO/IEC 17025. The voting on the draft standard began on September 16th 1999 and is to be terminated on November 16th 1999. At the time of writing it is not clear whether the standard will be accepted or not. If the standard will become accepted the abbreviation "FDIS" will be dropped from the international version of the standard "prEN" from the European version. The original version of this paper was updated to reflect the developments until October 1999.

### **General advice**

The general advice from those interviewed was to have the scope of the accreditation limited in the beginning, and then expand as needed. The laboratory is a good place to start in an institute. Many institutes and companies have made the mistakes of writing too elaborate quality manuals, which have then become far too cumbersome to use and maintain. Many of these quality manuals are now being trimmed down.

The high level of complexity of the work performed in many laboratories (even small ones) makes it nevertheless necessary to write detailed quality manuals. The comprehensive document control needed should not be underestimated. It is one of the biggest tasks in accreditation.

### **Computerised document systems**

Writing the quality manuals is maybe not the most difficult task. It takes a lot of resources, but while it is being done it is usually given top priority. Maintaining, updating and distributing quality documents can be a very demanding task, especially since this procedure has to be done in accordance with strict document control rules.

A computerised document system can make the tedious work of creating, revising, issuing and distributing quality documents much easier. A fully developed document control system should be used (not a "home made" application). Databases for quality documents are commercially available. First a skeleton of the quality manual can be created. Already existing documents can then be put into place and new documents written as needed. Modern computer technology means that a description can take various forms. It can include text, sound, photographs and even videos. Different types of views can then be tailored into the document database for each type of user.

But probably the most important advantage of using a computerised document control system is that many of the formal procedures for maintaining quality documents can be built into the system. The system can ensure that all the tedious formalities concerning the revision process are adhered to and it is easy to ensure that every user has access to the updated versions of the documents.

### **The draft international standard ISO/IEC FDIS 17025**

A new draft international standard has recently been issued, **ISO/IEC FDIS 17025 - General Requirement for the Competence of Testing and Calibration Laboratories**. This new standard is to replace the EN 45001:1989, which has been used as the basis for accreditation of laboratories in Europe in recent years. The new standard has various advantages over the previous one for laboratories in the process of seeking accreditation:

- The standard replaces EN 45001:1989 and ISO/IEC Guide 25:1990. Although this standard does not include all the requirements of ISO 9001 and ISO 9002, it includes all of those that are relevant to the scope of testing and calibration services.
- The new standard is far more detailed than the EN 45001:1989. There is much less need for other documents giving advice on the implementation of the standard.
- There are some differences in the new standard, compared with the previous one, which could be interpreted as making it easier for laboratories to adopt more flexible accredited procedures. Some aspects of the interpretation of the standard are however still to be agreed upon by the accreditation bodies.

One might ask, why use this new standard instead of the existing EN 45001:1989? The answer is simple, if and after it has been adopted there will be no choice, quality manuals will in time have to be revised to meet the requirements of this new standard. It makes good sense, however, to start using this standard right away, even though it cannot be used as the basis for accreditation for some time.

The accreditation of laboratories is currently based on the standard **EN 45001:1989 General criteria for the operation of testing laboratories**. The actual text of the standard is 9 pages. The brief text of the standard does not cover the actual requirements set by the accreditation bodies. The gap has to be bridged by:

- guidance documents
- services of consultants

The actual text of the new standard, **ISO/IEC DIS 17025 - General Requirement for the Competence of Testing and Calibration Laboratories**, is 22 pages. It is thus far more detailed than the old EN 45001:1989. There is much less need for guidelines and consultants and it is possible to structure a quality manual directly on the standard.

Another advantage is that the new standard replaces **EN 45001:1989** and **ISO/IEC Guide 25:1990** and includes all the requirements of **ISO 9001** and **ISO 9002** that are relevant to the scope of testing and calibration services

Some people have claimed that it may be easier for laboratories to adapt more flexible accredited tests. It was even said that a laboratory could now be able to get an accreditation for its work (due to its high level of competence), instead of just getting an accreditation for specified tests. In the new standard there is a section on taking laboratory-developed methods into use. This could be understood as if the laboratory could do so without prior approval by an accrediting body. This view has however been said to be contrary to the basic ideas of how quality is to be assured through

accreditation. It seems clear now that only individual tests will be accredited and not the operation of the laboratory as a whole.

If the standard will be accepted, the accreditation bodies have to form their interpretation of the standard. While flexibility can be desired, care must also be taken not to destroy the formal system that has lead to the recognition of competence of laboratories in various fields. Some time will therefore pass before the standard can be used as the basis for accreditation of laboratories.

### **Conclusion**

The ISO/IEC FDIS 17025 standard may be put into use later than originally expected and presently it cannot be used as the formal basis of accreditation.

The standard may still be the best guidance document generally available for laboratories on obtaining accreditation at present, compliance with it now will make the transition easier later.

Whatever standard is used, computerised document control systems can make the *preparation and maintenance* of quality documents much easier.

### **Acknowledgements**

The work was supported by the Nordic Nuclear Safety Research programme (the NKS/BOK-1.1 sub-project on "*Laboratory measurements and quality assurance*"). It is likely that this topic will be followed up to some degree within the project. For more information on this project please visit the web site:

<http://www.gr.is/bok-1.1/>

For information about the NKS (Nordic Nuclear Safety Research programme), please visit the NKS web site:

<http://www.nks.org/>

# POSTERS



# THE NATIONAL RADIATION PROTECTION INFRASTRUCTURE

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## Abstract

The state system of radiation protection is still being created after Lithuania regained its independancy and in connection with recommendations laid in the ICRP-60 publication and requirements of legislation of European Community. A new regulation institutions was established and a number of laws and regulations related to radiation protection was prepared. The Radiation Protection Centre of Ministry of Health is the regulatory authority responsible for radiation protection of public and of workers using sources of ionizing radiation in Lithuania. A new Radiation Protection Law, Nuclear Energy Law, Radioactive Waste Management Law and different regulations was approved. Preparation of legislation, creation of state system of radiation protection and its upgrading allow to presume that the necessary level of radiation protection is to be achieved.

## Introduction

Lithuania is one of the Baltic states. Our neighbours are Latvia, Belorussia, Poland and Russia. Lithuanian population is 3.7 million and square of 65 thousand square kilometres.

Together with old traditions of agriculture, production of textile, furniture Lithuania is famous as the most nuclear country in the world. Ignalina Nuclear Power Plant produces up to 85% of Lithuanian electricity. There are two RBMK (Chernobyl type) reactors of 1250 MW in the plant.

However, when considering problems of radiation protection and safety of sources it should be emphasised that we have more than 800 users of radioactive sources of higher or lower activity (Table 1).

## Discussion

A number of new institutions were established in independent Lithuania after March 11, 1990. The State Nuclear Power Safety Inspectorate is responsible for nuclear safety in Ignalina NPP and fields connected with its operation. The Ministry of Environment is responsible for environmental monitoring of radioactive contamination, environmental releases and discharges, state supervision of these processes, control of transport of radioactive sources. The Department of Civil

Defence at the Ministry of Defence is the co-ordinating institution in preparation of plan of actions and its implementation in case of accident in the Ignalina NPP (Fig.1).

Table 1. The Licensees Using Sources Of Ionizing Radiation (Sir) 1998

License	Number of licensees	Number of SIR under use		
		Total	Including	
			X-ray	sealed
Research institutions	42	519	94	380
Health care institutions	537	1437	1244	165
Education institutions	18	289	27	257
Food enterprises	5	29	-	29
Industrial enterprises	120	17544	164	17380
Other licensees	117	4978	48	4925
Total	839	24794	1577	23136

However the main responsibilities are laid on Radiation Protection Centre of Ministry of Health and its special structures of radiation protection in the regions (counties) (Fig.2). The total number of specialists in this service is 46, including 9 doctors-hygienists radiologists, 15 engineers-radiologists, 2 hygienic chemists with university degree, 13 technicians radiologists and technicians.

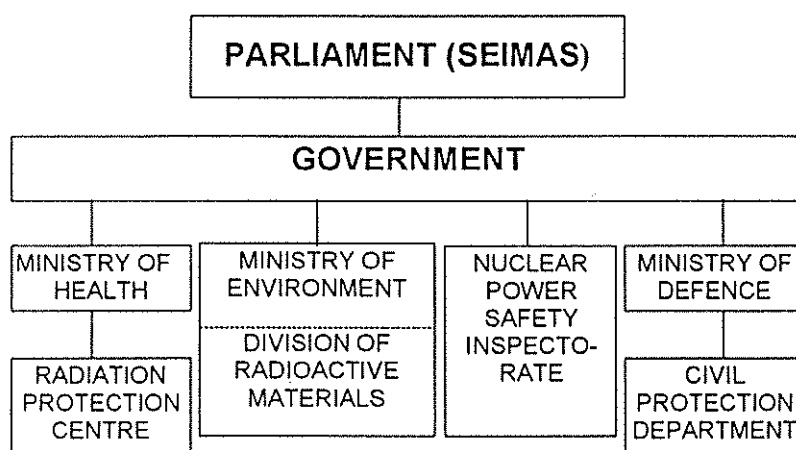


Fig. 1. Governmental institutions related to radiation protection

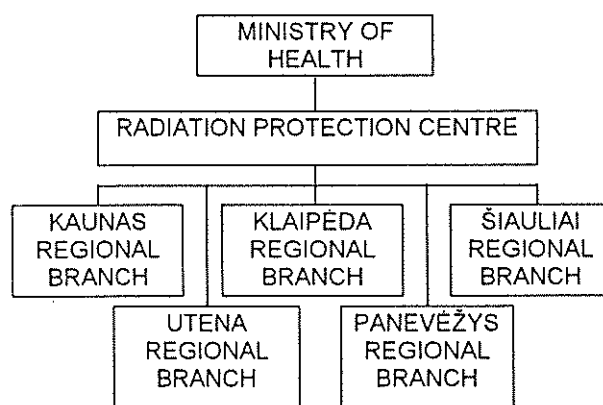


Fig. 2. General structure of Radiation Protection Centre

A number of laws (Table 2) and regulatory documents (hygiene norms and regulations) (List 1) are the legal basis for radiation protection, safety and registry of sources in Lithuania. The Basic Radiation Protection Standards have been prepared by the Radiation Protection Centre for implementation of the International Basic Safety Standards for Protection against Ionizing Radiation and for the Safety of Radiation Sources (BSS) and Council Directive 96/29/ Euratom. This standard was approved by the Ministry of Health.

Table 2. Laws related to radiation protection in Lithuania

Law	Date of approval
The Health Protection Law	July 19, 1994)
The Law of Health Institutions	July 6, 1996)
The Environmental Protection Law	(January 21, 1992)
The Law of Amendments and Changes to the Environmental Protection Law	(May 28, 1996)
The Nuclear Energy Law	(November 14, 1996)
The Radiation Protection Law	(January 12, 1999)
The Radioactive Waste Management Law	(May 20, 1999)

List 1. The Documents Prepared Or Being Under Preparation By The Radiation Protection Centre

Prepared and in force:

Hygiene standard HN 31:1998 "Radiation protection and safety in medicine X-rays diagnostic practice"

Hygiene standard HN 52:1995 "Industrial radiography"

Hygiene standard HN 54:1998 "Raw materials and foodstuffs. Maximum permitted levels of chemical contaminants and radionuclides"



Hygiene standard HN 72:1997 "Sampling methods of foodstuffs, feedingstuffs, soil and water for determination of specific and volumetric activity of radionuclides"  
Hygiene standard HN 73:1997 "Basic standards of radiation protection"  
Hygiene standard HN 77:1998 "Radiation protection and safety in nuclear medicine practice"  
Hygiene standard HN 78:1998 "Quality control in medical X-ray diagnostics. General requirements and evaluation criteria"  
Hygiene standard HN 83:1998 "Radiation protection and safety of outside workers"  
Hygiene standard HN 84:1998 "Maximum permitted levels of radioactive contamination of foodstuffs and feedingstuffs following a nuclear or radiological emergency"  
Hygiene standard HN 85:1998 "Natural exposure. Standards of radiation protection"  
Regulations of Licensing the practices with ionising radiation sources (Decision of Government-No.653, 25 05 1999)

Under preparation:

Hygiene standard on radiotherapy

According to the Radiation Protection Law the Radiation Protection Centre is responsible for preparation of laws and legal documents, preparation and presentation to the Government of principles of state strategy in radiation protection, registry and regulatory control of safety of sources of ionizing radiation, licensing of users and organisation of control of compliance with requirements, organisation of supervision and control of workplaces where sources of ionizing radiation are used. Besides, we are responsible for control of medical exposure, monitoring of radioactive contamination of foodstuff, its raw materials, drinking water and building materials, control of indoor radon concentrations, personal dosimetry control and emergency preparedness.

For implementation of the above mentioned means Lithuania received necessary help from the International Atomic Energy Agency, European Commission, other international organizations and countries in the framework of co-operation.

Much more plans are waiting for future. It is a need to continue preparing secondary regulations which should follow or harmonise with the directives and regulations of the European Commission. It is necessary to develop the system of licensing and inspections, use of modern information and communication technology. Our plans are to create the systems of education and training of professionals in radiation protection and internal dosimetry of workers. According to the Radiation protection Law the National Program of radiation protection will be prepared.

## Conclusion

Preparation of legislation, creation of state system of radiation protection and its upgrading, education and training of specialists, supply with necessary measuring and other equipment allow us to presume that the necessary level of radiation protection is to be achieved in Lithuania.

## Swedish Radiation Protection Institute: Information Activities

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### ABSTRACT

The purpose of SSI's Information and PR Service is to broaden public awareness of radiation and radiation risks as well as to fulfil other performance goals. SSI achieves this through its advisory, educational and informative activities. SSI publishes two external magazines, Strålskyddsnytt and SSI News. Strålskyddsnytt – which is available in Swedish only – has a circulation of 2,400 and is published four times a year. SSI News – which is in English – is published twice a year and has a circulation of about 1,500. Another important channel of communication is the web site ([www.ssi.se](http://www.ssi.se)). Taking advantage of PUSH technology, SSI also distributes, by e-mail, press releases and other important information on radiation to radiation protection professionals in Sweden. SSI continuously monitors news by subscribing to a press clipping service. SSI Training is a commercial unit within the Information and PR Service. A policy for mass media contacts exists as well as a policy for internal communication. SSI has a graphic profile. SSI has a specialised research library.

Information and PR Services is a separate department within SSI which is used as a strategic control instrument to attain the goals established by the Government.

SSI must become more visible in order to ensure that more of the general public knows about its existence and activities. Opinion polls show that SSI is one of the lesser-known government authorities. SSI must therefore become more offensive in its PR work. The mass media is an important information channel and we will work towards ensuring that journalists become more interested in SSI.

The purpose of SSI's Information and PR Service is to broaden public awareness of radiation and radiation risks as well as to fulfil other performance goals. SSI achieves this through its advisory, educational and informative activities. Through its communication work, SSI's Information and PR Service ensures that SSI can achieve its performance goals and that the public has the confidence that SSI's competence and integrity will ensure an adequate level of radiation protection.

SSI publishes two external magazines, Strålskyddsnytt and SSI News. Strålskyddsnytt – which is available in Swedish only – has a circulation of 2,400 and is published four times a year. This magazine primarily targets other authorities, radiation protection specialists in Sweden and the Nordic countries as well as journalists. SSI News – which is in English – is published twice a year and has a circulation of about 1,500. This magazine mainly targets radiation protection experts outside Sweden.

Another important channel of communication is our web site ([www.ssi.se](http://www.ssi.se)). SSI's intention is that the web site should be updated daily in order to display current information within SSI's area. The web site is, of course, an ongoing project. Taking advantage of PUSH technology,

SSI also distributes, by e-mail, press releases and other important information on radiation to radiation protection professionals in Sweden.

SSI continuously monitors news by subscribing to a press clipping service. News of interest is copied and distributed to the various SSI departments. Of the 2,400 clippings received during 1998, 35 per cent dealt with non-ionising radiation (cell phones and mobile masts, lasers, UV, electromagnetic fields) and 45 per cent dealt with nuclear power (the nuclear power debate, the phase-out of nuclear power, waste, final disposal, Chernobyl). The remaining 20 per cent concerned related topics such as radon, medical care and biofuels. SSI also provides training to improve the writing and communication skills of its employees and has issued an in-house style guide. Twenty issues of SSI's personnel magazine, *Curieren*, are published per year and the circulation is 200.

The Central Emergency Preparedness Organisation is placed in SSI. SSI and the Swedish Nuclear Power Inspectorate (SKI) participate jointly in the information unit in this organisation to inform the county administrations, the mass media and the public. The information unit is training a few times per year.

SSI and SKI collaborates on nuclear waste information. A joint program exists to inform the public in the districts where the SKB (the Swedish Nuclear Fuel and Waste Management Co) undertakes feasibility studies for siting a deep repository for spent fuel and also in other districts where nuclear installations exist and finally also inform the mass media on nuclear waste.

Together with the National Board of Health and Welfare, a manual has been published in Swedish and English on medical and information-related measures in connection with nuclear accidents.

SSI Training is a commercial unit within the Information and PR Service. To increase awareness of radiation and radiation protection, SSI Training consults with the various departments within SSI to provide courses and information for organisations, companies and the general public. Courses on radon and non-ionising radiation are primarily aimed at the municipal environmental protection offices. This year, for the first time, courses are being provided for people working within the civil aviation industry. SSI will also be offering training on health physics for X-ray personnel within the veterinary services as well as a new course on radiation protection and biofuels. All courses are advertised in the course catalogues that are published in spring and autumn.

SSI has a specialised research library, which receives state funding. The library's activities include acquisitions, lending services, database and literature searches as well as the monitoring of new publications. SSI's own publications are searchable online within the Swedish national LIBRIS catalogue.

### *SSI's Graphic Profile*

Our *profile* reflects who we are. One of the most important tools in establishing, reinforcing and maintaining SSI's profile as a professional and competent authority is a consistent graphic profile. The graphic profile establishes SSI's identity and presents an unambiguous picture of SSI to the outside world. Therefore, it is important for all of SSI's employees to consistently use the graphic profile, which has been developed.

Our *image* is the way in which we are perceived by the outside world. Examples of factors affecting our image include external information, mass media coverage, the quality of our services etc. Our image is reflected in all of the documentation produced by SSI. It is therefore important for letters, publications and information materials to consistently use our graphic profile since this projects a strong image to the outside world and reinforces our organisational culture

#### *Policy for Mass Media Contacts*

The mass media are senders as well as receivers of information. As opinion-makers, the media play a central role in society, and can thereby affect public perception of SSI. Consequently, it is important that information presented in the media should be correct in terms of facts as well as in terms of the position adopted by SSI on various issues.

The mass media are a valuable external information channel. By handling the mass media in a professional way, we can improve public awareness of radiation and radiation protection as well as reinforce SSI's profile as a regulatory and supervisory authority. Through our subscription to press clipping services, we can analyse the treatment of SSI and radiation protection issues in the media.

**Be Proactive:** We don't wait for the media to get in touch with us when something newsworthy happens in the radiation protection area. This applies to both positive and negative events. We must not damage our credibility by running the risk of the media believing that we are hiding information. We take the initiative in order to get publicity for positive news since the media show the greatest interest in radiation protection in times of crisis. We aim to maintain a proactive and long-term relationship with the mass media.

**Respond Quickly:** Information concerning different events should be provided as quickly as possible. If necessary, additional information can always be provided at a later stage. SSI must take, and preferably, keep the initiative.

**Be Factual:** All information provided must be factual, honest, true, unambiguous and must reflect the position adopted by SSI on the issue concerned.

Our right – as individuals - to provide information to the press is written into the constitution. Without renouncing this right, we must all carefully make a distinction between our own private opinions and opinions that we provide as representatives for SSI.

**Deal with Errors:** If serious errors are detected, we try to immediately take action to correct them. In the case of minor factual errors, it is often sufficient to phone the journalist concerned.

**Prepare an Answer:** When journalists call us up, we don't necessarily respond immediately to the question. In some cases, we ask whether we can get back to them with an answer in a little while. We then collect the necessary background information to be able to answer the question appropriately. If possible, we try to respond within half-an-hour.

**Co-ordinate Information:** SSI prefers to adopt a policy on how to respond to a particular issue. This minimises the risk of contradiction and ambiguity and enhances our credibility.

Brief the Information Staff: Contacts with the press are usually co-ordinated by the Information and PR Service. In the event of direct contacts between the press and other employees, our information and PR staff must be briefed so that they can follow up the interview and assess whether any special action is necessary.

SSI has issued a general information policy which covers both internal and external communications, contact with other agencies, information on research and information to the public

*The policy for internal communications*

A critical success factor for information work is that all employees should know and respect the goals and strategies established by the management for information work

Each employee is entitled to have access to the necessary information in order to be able to do their job and to get feedback on their performance. Each employee is entitled to know how his/her performance is being appraised and how it fits in to the "larger picture".

It is up to each line manager his/her employees so that they can do their jobs and can understand how they fit into the organisation as a whole. If information is not provided, it is the duty of the individual employee to try to obtain it. Each employee must know his/her responsibilities and limitations. On the basis of the right information, it is the duty of each employee to make decisions within his/her own area of responsibility

SSI aims to:

- Satisfy the needs of each employee for factual and open information concerning SSI's activities and the external environment
- Promote understanding across departmental boundaries
- Provide a better basis for decision-making
- Ensure that a consensus is reached concerning goals and information provided and thereby contribute to unified action to attain specified goals
- Reinforce shared values, motivation and job satisfaction within the organisation
- Ensure that a "united front" is presented to the outside world
- Facilitate communication between the management and employees as well as communication between employees
- Contribute to the personal development of SSI employees.

# Ändringar i reglerna för transport av radioaktiva ämnen

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## Sammanfattning

Föreskrifter för transport av radioaktiva ämnen ingår i regelverken för transport av farligt gods (ADR, RID, IMDG-koden och ICAO-TI) [1-4]. Dessa baseras samtliga på "IAEA Regulations for the Safe Transport of Radioactive Material" [5,6]. En ny upplaga av IAEAs rekommendationer publicerades 1996 och håller för närvarande på att implementeras. Ändringarna kommer att införas i regelverken för transport av farligt gods den 1 januari 2001. Ändringarna beror till stor del på anpassning till de grundläggande säkerhetsnormerna för strålskydd (BSS) [8].

## Inledning

Det internationella atomenergiorganet, IAEA, har sedan 1961 givit ut regler för transport av radioaktiva ämnen. Dessa bestämmelser revideras regelbundet och har sedan länge publicerats som "Regulations for the Safe Transport of Radioactive Material. IAEA Safety Series No.6" [5]. Nya upplagor har kommit ut med cirka 10 års mellanrum. IAEA-reglerna, som egentligen är rekommendationer, har sedan legat till grund för de delar av ADR [1], RID [2], IMDG-koden [3] och ICAO-TI [4] som gällt för transport av radioaktiva ämnen. Flertalet länder har även tillämpat dessa bestämmelser för inrikes transport. IAEA beslutade 1996 att ge ut nya rekommendationer. Dessa finns nu publicerade som "Regulations for the Safe Transport of Radioactive Material, 1996 Edition, IAEA Safety Standards Series No.ST-1" [6]. "IAEA Safety Series No.6" kommer således i framtiden att kallas "ST-1". De ändringar som kan antas innebära de största förändringarna kommenteras nedan.

## Ändringar beroende på Basic Safety Standards

### Allmänna bestämmelser

De nya transportreglerna baseras på de grundläggande säkerhetsnormerna, som de kommer till uttryck i "ICRP 60" [7] och "International Basic Safety Standards for Protection against Ionizing Radiation and for the Safety of Radiation Sources" (BSS) [8]. För att kunna behålla nuvarande gränsvärden för dosrat utanför kollin och fordon har det införts krav på att strålskyddsprogram skall finnas för verksamhet med transport av radioaktiva ämnen. Strålskyddsprogrammet, som skall anpassas till omfattningen av verksamheten, kan t ex innehålla uppgifter om ansvarsfördelning, dosuppskattning och kategoriindelning av personal, bestämning av ytkontamination, dosgränser, dosrestriktioner och optimering av strålskyddet, mätning och registrering av doser, segregationsavstånd, åtgärder vid missöde, utbildning samt skydd och säkerhet vid transport och hantering. Allt syftande till att undvika onödiga stråldoser till personal och allmänhet från verksamheten.

Strålskyddsprogrammet för verksamheten med transport skall finnas dokumenterat och kunna uppvisas för behöriga myndigheter om det efterfrågas.

Om dosen till transportpersonal med stor sannolikhet kommer att vara mellan 1 mSv/år och 6 mSv/år skall den effektiva dosen bestämmas genom mätning, antingen genom monitorering på arbetsplatsen eller genom användning av persondosmätare. I de fall den effektiva dosen till personalen kan komma att överstiga 6 mSv/år skall persondosmätning alltid användas. Uppmätta doser skall registreras och arkiveras enligt gällande regler i respektive land.

Studier av dos till personal sysselsatta med transport och hantering har visat på samband mellan erhållen dos och summan av transportindex (TI) för de transporterade kollina [9,10]. Dessa studier tyder på att det är osannolikt att någon ur transportpersonalen kommer att erhålla en dos överstigande 1 mSv/år om endast kollin för vilka summan av TI är mindre än 300 hanteras årligen. Om personen inte är sysselsatt med någon annan form av verksamhet med strålning, kan i de flesta fall detta samband användas för att avgöra vilken kategori personen skall tillhöra.

### **Undantagsgränser**

BSS [8] anger numera nuklidspecifika gränsvärden för såväl aktivitetskoncentration som total aktivitetsmängd. Dessa gränsvärden har beräknats med utgångspunkt från de grundläggande kriterierna. Undantagen verksamhet får t ex inte medföra en högre dos än 10 µSv/år till individer i kritisk grupp. Samma nuklidspecifika gränsvärden som anges i BSS kommer att tillämpas även för transport. De numeriska värdena för aktivitetskoncentration sträcker sig från 0,1 Bq/g för vissa alfastrålande ämnen till  $1 \times 10^6$  Bq/g för Tritium istället för nuvarande gränsvärde på 70 Bq/g oberoende av nuklid. Gränsvärdet för total aktivitetsmängd kommer att gälla för summan av samtliga kollin i en och samma sändning och ej för enstaka kolli. Mycket svaga strålkällor kommer genom detta undantag att vara tillåtna att transporteras utan tillämpning av alla detaljer i transportreglerna i framtiden.

### **Tillåtet innehåll**

$A_1/A_2$  värden har beräknats med hänsyn tagen till nya strålskyddsvärderingar, dosimetri och nukliddata. De numeriska värdena kan vara både lägre och högre än dagens  $A_1/A_2$  värden. Detta medför att tillåtet innehåll av vissa radionuklider i olika typer av kollin ändras. T ex kommer tillåtet innehåll i ett kolli med Mo-99/Tc-99m att minska, medan kollin med Am-241 strålkällor tillåts att innehålla mera.

## **Övriga ändringar**

### **Kolli av typ C**

En ny typ av kolli kommer att fordras för flygtransport av stora kvantiteter radioaktivt material. Utöver krav på typ B-provning skall ett sådant kolli även kunna klara fall med islagshastighet av 90 m/s mot stumt underlag, brand 800°C i 60 minuter samt nedsänkning i vatten på 200 meters djup. Transport av bl a Plutonium och stora Co-60 strålkällor kommer att omfattas av dessa regler.

Samtidigt införs en ny materialkategori kallad "Low dispersible material (LDM)" för vilka undantag från typ C-kraven får göras. Det skall då visas att materialet enbart, dvs utan emballage, ska ha en begränsad strålnivå (mindre än 10 mSv/h på 3 meters avstånd) och att

endast en begränsad mängd av ämnet kan frigöras i gas- eller partikulär form om materialet utsätts för fall- och brandproven motsvarande provningarna för kollin av typ C.

### **Kollin innehållande uranhexafluorid**

Speciella bestämmelser för transport av mer än 0,1 kg uranhexafluorid har införts. Detta har, på grund av ämnets mycket speciella fysikaliska och kemiska egenskaper, gjorts för att förhindra en riskabel tryckökning i händelse av brand. Kollin innehållande mer än 0,1 kg uranhexafluorid skall i framtiden godkännas av behörig myndighet även då det innehåller utarmat eller naturligt uran.

### **Kriticitetssäkerhet**

För undantaget klyvbart material införs en mängdbegränsning per sändning. T ex får kollin som ingår i samma sändning och innehåller vardera mindre än 15 g klyvbart material totalt inte innehålla mer än 400 g U-235 eller 250 g annat klyvbart material. Begränsningen kan i vissa speciella fall vara strängare. Bestämmelserna för hur kollin innehållande klyvbart material skall provas, utvärderas och märkas har också ändrats.

Vid flygtransport av klyvbara ämnen skall det visas att kollit förblir underkritiskt efter typ C provningar.

Ytterligare en ändring av reglerna är att om utbränningskreditering utnyttjas vid beräkning av mängden klyvbart material skall en mätning göras efter bestrålningen men före transport, för verifiering av beräkningarna.

### **Transportindex och kriticitetssäkerhetsindex**

Begreppet "Transportindex (TI)" har förenklats och kommer enbart att användas för strålskyddsändamål. Nya TI baseras på dosrat på 1 meter från kollits yta och tar ej hänsyn till kriticitetsegenskaper. Ett nytt index, kriticitetssäkerhetsindex (CSI) införs för klyvbart material och baseras på tillåtet antal kollin. En uppdelning på två index ger möjlighet att separera kollin av såväl strålskyddsskäl som på grund av kriticitetsrisk och antas i de flesta fall underlätta hanteringen. En ny etikett, där kriticitetssäkerhetsindex anges, ska fastställas utanpå kollin innehållande klyvbara ämnen.

### **UN-nummer**

Alla kollin, även s k undantagna kollin, skall i framtiden märkas med UN-nummer. Anledningen till kravet på denna märkning är att räddningstjänst skall få information oberoende av språk. För att kunna ge mer relevant information har listan över UN-nummer utvidgats och de nya UN-numren kommer att knytas till förteckningarna. Klyvbart och icke-klyvbart material, som transporteras enligt samma förteckning, får olika UN-nummer. För ämnet uranhexafluorid behålls de gamla UN-numren, oberoende av enligt vilken förteckning kollit transporteras, på grund av ämnets speciella kemiska egenskaper.

## **Införande och omstrukturering**

De internationella transportorganisationerna är nu i slutfasen med att överföra rekommendationerna i IAEA ST-1 till bindande föreskrifter ingående i ADR, RID, IMDG-koden och ICAO-TI [1-4]. Samtidigt kommer regelverken att få en ny utformning, som i stort överensstämmer med dagens ICAO-TI. Detta kommer att innebära att klass 7 (radioaktiva ämnen) inte längre har ett eget kapitel utan föreskrifterna kommer att återfinnas under rubriker



som "klassificering", "krav på kollin", "undantag och begränsade kvantiteter", "märkning och etikettering", "transportdokument" tillsammans med bestämmelser för övriga klasser av farligt gods. Omstruktureringen anses komma att underlätta för användarna att finna de regler som är tillämpliga i ett visst fall. Målsättningen är att de nya reglerna skall gälla från och med 1 januari 2001 och finnas publicerade i god tid före denna tidpunkt.

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# Kombinert lager og deponi for lav- og middels radioaktivt avfall i Norge (KLDRA-Himdalen)

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*Sammendrag* — Det nasjonale, kombinerte lager og deponi for lav- og middels radioaktivt avfall i Himdalen nær Kjeller ble etter ca 15 måneders byggetid innviet den 24. september 1998. Anlegget var resultat av en mangeårig, omfattende prosess som inkluderte utredninger om stedsvalg, miljøkonsekvenser og radiologisk sikkerhet, og der lokalbefolkning, aksjonsgrupper, fagfolk og politikere fikk uttale seg. Anlegget skal kunne ta imot i alt 10 000 tønner med lav- og middels radioaktivt avfall og skal etter planen være i drift fram til år 2030. Statsbygg har vært ansvarlig for byggingen, og Institutt for energiteknikk har fått ansvaret for driften av anlegget.

## HISTORIKK

Arbeidet med å etablere et nasjonalt deponi for lav- og middels radioaktivt avfall i Norge startet i 1988. Etter en omfattende prosess inkludert valg av til slutt tre mulige lokaliteter og en konsekvensutredning av disse, vedtok Stortinget i april 1994 at det skulle bygges et kombinert lager og deponi for lav- og middels radioaktivt avfall i Himdalen. Den kombinerte løsningen innebar at radioaktive stoffer med lang halveringstid (plutonium) skulle lagres i stedet for å deponeres. Samtidig ga Stortinget et pålegg om at det avfallet som i 1970 ble gravd ned på IFEs område på Kjeller, skulle graves opp og flyttes til det nye anlegget.

Stortinget ønsket også at det skulle gjøres en uavhengig vurdering av Himdalen-lokaliteten og den planlagte tekniske utformingen av anlegget. En internasjonal ekspertgruppe, under IAEAs "Waste Management Assessment and Technical Review Programme (WATRP)", gikk således i 1995 gjennom det norske forarbeidet og vurderte hele konseptet. Ekspertgruppen konkluderte med at Himdalen-lokaliteten og det tekniske konseptet kunne tilfredsstille kravene til et sikkert lager og deponi for de relativt små mengdene lav- og middels radioaktivt avfall skulle plasseres (IAEA 1995).

WATRP-rapporten dannet et viktig grunnlag for arbeidet med en sikkerhetsanalyse for anlegget, og for Statens stråleverns vurdering av konsesjonssøknader for bygging og drift. Den omfattende sikkerhetsanalysen bestod av beskrivelser av anlegget og dets funksjoner samt konsekvensene for omgivelsene ved normal drift og ved avvik fra den normale situasjonen. For de "mest sannsynlige" uhellsscenarier, som absolutt sett har lav sannsynlighet, har Statens strålevern satt en grenseverdi for effektiv dose til enkeltindivider i befolkningen på 1  $\mu\text{Sv}/\text{år}$ .

Etterat konsesjon for bygging var gitt, startet Statsbygg anleggsarbeidene i april 1997, og anlegget ble innviet i september 1998. Anlegget hadde da kostet 70 mill. NOK. Institutt for energiteknikk fikk konsesjon av Sosial- og helsedepartementet for drift av anlegget i april 1998. Den endelig tillatelse til driftsstart ble gitt av Statens strålevern i mars 1999, og den første transporten av avfall fra Kjeller til Himdalen fant sted 17. mars.

## ANLEGGSBESKRIVELSE OG DRIFTSFORHOLD

KLDRA-Himdalen er lokalisert i Himdalen i Aurskog-Høland kommune i et område med liten bebyggelse (Fig. 1). Avstanden fra Kjeller er ca. 25 km. Anlegget inkluderer 4 fjellhaller med 40 – 50 meter fjelloverdekning (Fig. 2). Det er bygget med et fall på 1: 50 fra hallene mot inngangsportalen. Dette gjør anlegget selvdrenerende med tanke på det vannet som ren-

ner inn fra fjellet rundt hallene. Alt vann som renner ut av anlegget kan kontrolleres. På denne måten unngås en ukontrollert lekkasje av radioaktive stoffer til grunnvannet i området. Slik anlegget er bygget, skal en slik lekkasje i teorien ikke skje.

KLDRA-Himdalen er dimensjonert til å kunne ta imot i alt 10 000 tønnekvivalenter med radioaktivt avfall. Disse fordeler seg med 7 500 på deponidelen (3 haller) og 2 500 i lagerhallen. Aktivitetsnivået av det avfallet som totalt skal deponeres og lagres i anlegget er beregnet til ca. 550 000 GBq, korrigert for desintegrasjon fram til år 2030.

Med unntak av periodene med innlasting og omstøping av avfallsbeholdere og rutinemessig inspeksjon, vil anlegget være ubemannet. Alle brann-, innbrudds- og luftaktivitetsalarmer og bilder fra TV-kameraer blir automatisk overført til IFE, Kjeller.

I de nærmeste 3-4 årene vil det avfallet som i dag er lagret over bakken og nedgravd på IFE bli transportert til Himdalen. Dette omfatter i dag ca. 3400 tønnekvivalenter hvorav ca. 2400 er lagret i lagerbygg og resten er nedgravd på IFEs område. Deponeringsprosessen og andre anleggsdetaljer er illustrert i Fig. 3.

Transportene fra Kjeller til KLDRA-Himdalen vil foregå med lastebil i henhold til gjeldende utgave av ADR-reglene for veitransport av farlig gods. Normalt vil transportene bli fulgt av følgebil med strålevernspersonell fra IFE. Følgebilen vil være fullt utstyrt for å kunne håndtere uhell under transport, og det er utarbeidet detaljerte retningslinjer for håndtering av slike uhell. Klargjøring av avfallsbeholdere og opplasting vil skje inne på IFEs område på Kjeller under overvåkning av strålevernspersonell (Raaum et al. 1999).

KLDRA-Himdalen skal etter planene være i drift fram til år 2030. Anlegget skal da stenges og være under institusjonell overvåkning i 300-500 år framover. Planer for lukking av anlegget skal utarbeides og godkjennes av myndighetene i god tid før år 2030. Hvordan overvåkingen av anlegget skal skje og hvem som skal utøve denne overvåkingen, vil bli avgjort i forbindelse med tillatelse til lukking av anlegget.

## **RADIOLOGISK KARTLEGGING**

I forbindelse med byggingen av KLDRA-Himdalen er det blitt gjennomført en kartlegging av radioaktiviteten i området der anlegget ligger. Undersøkelsen er gjort på oppdrag fra Statsbygg og hadde til hensikt å dokumentere bakgrunnsnivåene for radioaktivitet i området før drift av anlegget starter (Varskog og Bergan 1998; Strålberg et al. 1999).

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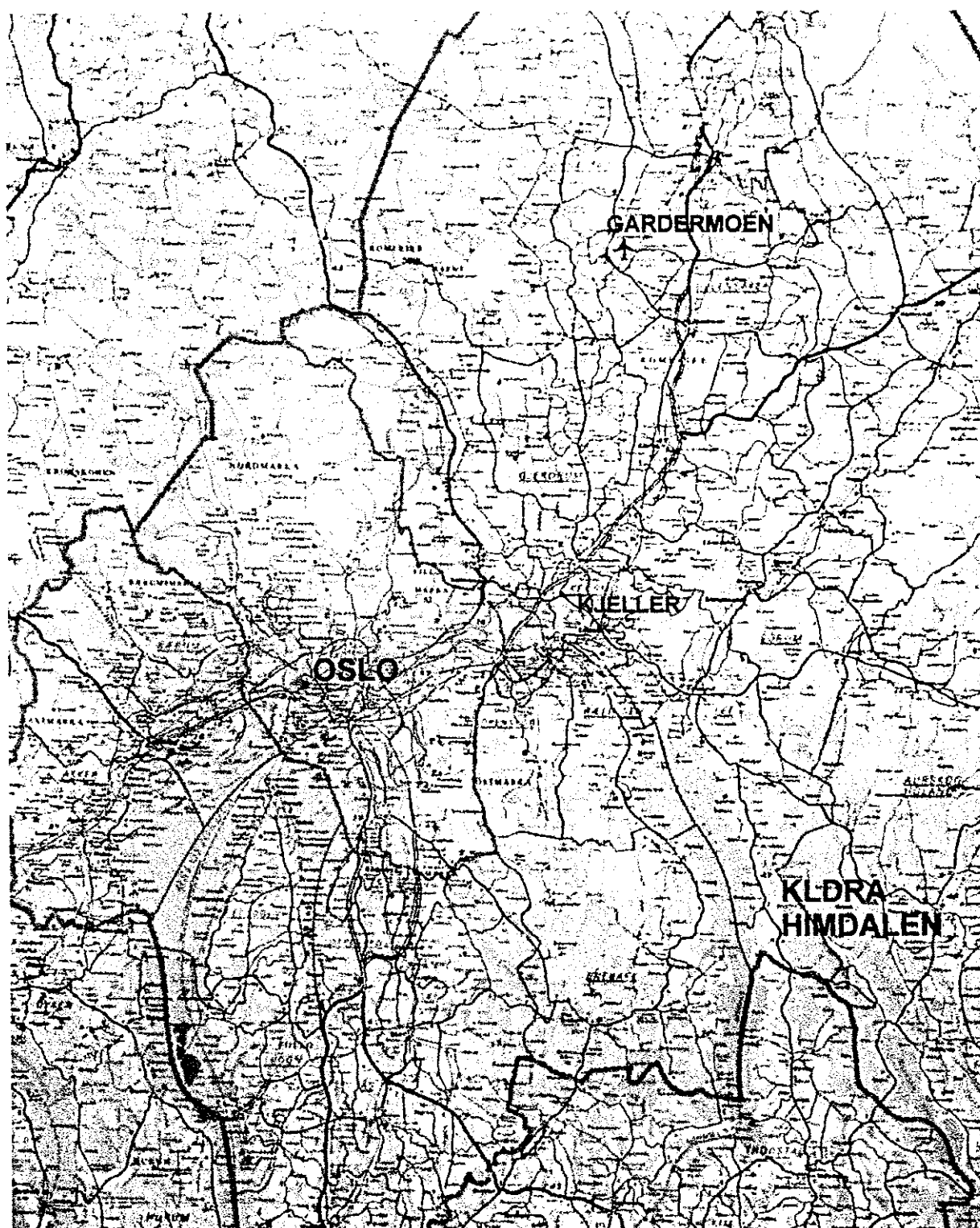


Fig. 1. Lokalisering av KLDRA Himdalen (omtrent ved bokstaven K i KLDRA).

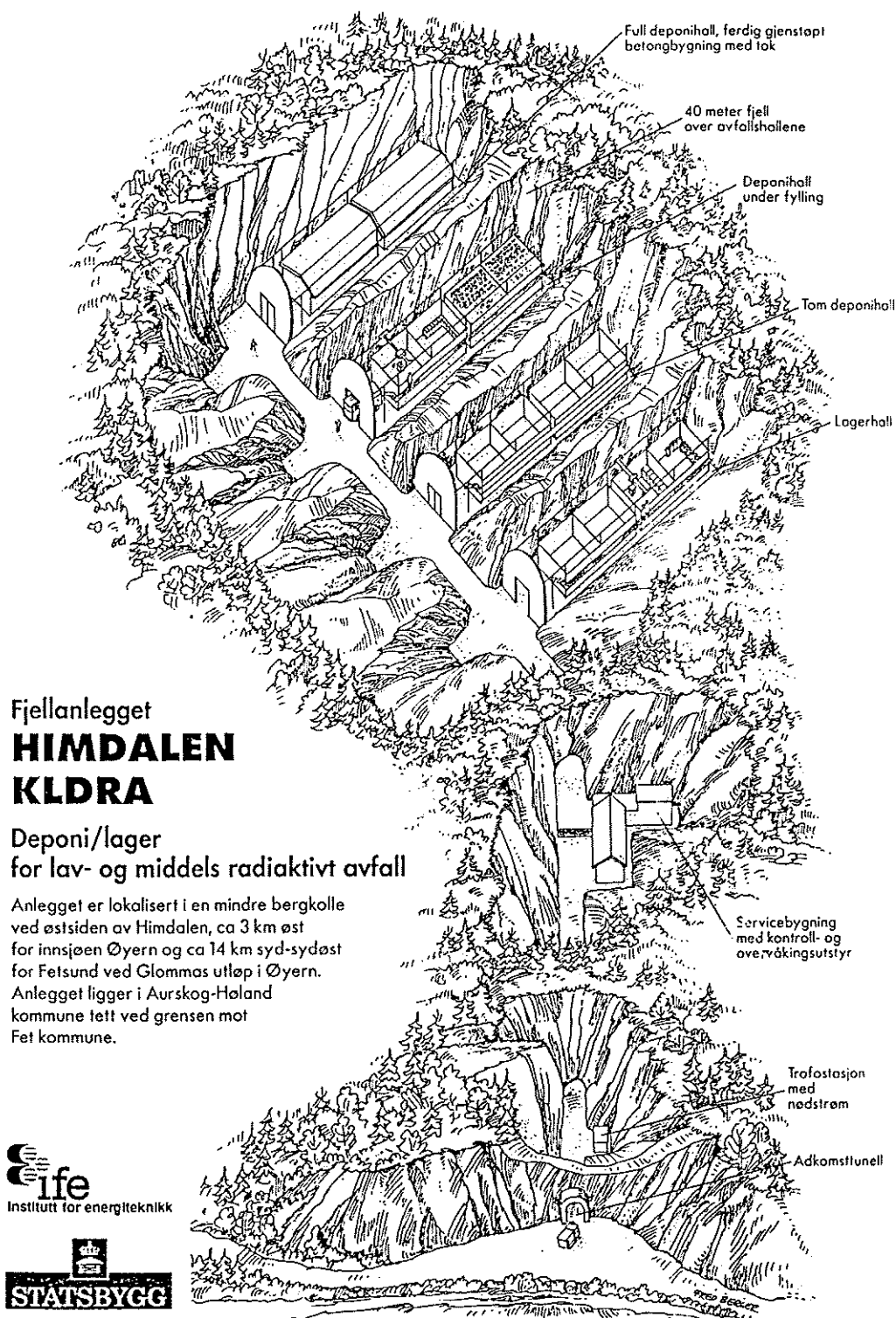


Fig. 2. Hovedinstallasjonene i KLDRA-Himdalen

## Deponering (innstøpning)

Innstøpningen gjøres i deponihallene.  
Avfallsbeholdere i lagerhallen støpes ikke inn.

### Sikkerhetsbarrierer for inneslutning av avfallet:

- ① Indre beholder
- ② Betong
- ③ Ytre beholder
- ④ Omstøpning
- ⑤ Betongrom
- ⑥ Fjell

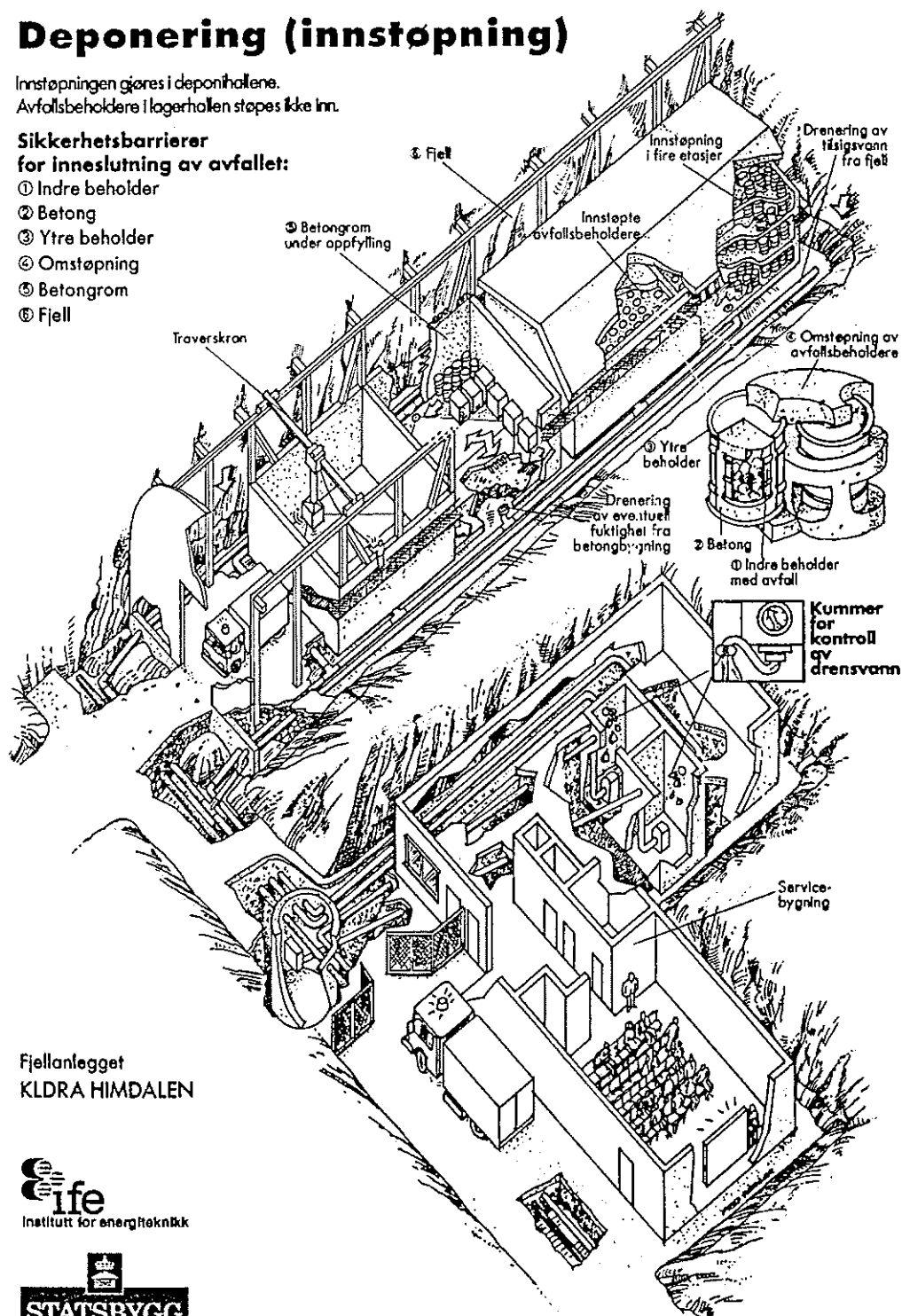


Fig. 3. Deponeringsprosessen og andre anleggsdetaljer i KLDRA-Himdaalen



# **Erfaringer fra transport av radioaktivt avfall fra IFE - Kjeller til KLDRA Himdalen**

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## **SAMMENDRAG**

Doseregistreringer ved hjelp av elektroniske dosimetre blir foretatt for å skaffe erfaringer med dosebelastning til personell under arbeidet med transport av lav- og middels radioaktivt avfall fra IFEs avfallslager på Kjeller til Kombinert Lager og Deponi for lav- og middels Radioaktivt Avfall (KLDRA) Himdalen. I løpet av en treårsperiode skal i overkant av 3400 avfallsbeholdere, hovedsakelig tønner, overføres fra Kjeller til KLDRA Himdalen. Dette utgjør ca. 170-190 transporter à 20 tønner. Transportprosessen kan deles inn i tre faser; opplasting som inkluderer uttak fra lageret på Kjeller, utklarering av avfallsbeholdere og plassering på lastepalet, transport på vei, og lossing, innbefattet plassering av avfallsbeholdere i riktig posisjon i Himdalen anlegget. Doseregistreringer fra 13 transporter à 20 tønner er analysert. Hver transport fører til en gjennomsnittlig kollektiv dose på ca.  $170 \cdot 10^{-6}$  manSv. De høyeste dosene fås i opplastningsfasen, og det antas at det er uttak av avfallsbeholdere fra lagerbygningen som bidrar mest til dosen. Selve transporten gir ingen dose utover bakgrunnen. Med utgangspunkt i 100 transporter i året blir den totale dosebelastning på ca.  $17 \cdot 10^{-3}$  manSv/år. Det tilsvarer en gjennomsnittlig årstdose på 4-5 mSv for hver av de 4 personene fra seksjon Radavfall som arbeider med transportene.

## **INNLEDNING**

Arbeidet med å transportere radioaktivt avfall fra IFEs avfallslager på Kjeller til Kombinert Lager og Deponi for lav- og middels Radioaktivt Avfall (KLDRA) Himdalen startet opp i mars 1999. I løpet av en treårsperiode skal i overkant av 3400 avfallsbeholdere, hovedsakelig tønner, overføres fra Kjeller til KLDRA Himdalen. Dette utgjør ca. 170-190 transporter à 20 tønner.

Arbeidet med håndtering av avfallsbeholdere i forbindelse med transporten antas å kunne medføre betydelig dosebelastning. Utstyr og arbeidsprosedyrer er derfor tilrettelagt med tanke på å redusere persondoser. For å kunne følge opp persondoser underveis, benytter alle som er involvert i arbeidet elektroniske dosimetre, og det registreres doser fra de ulike fasene i arbeidet. Doseregistreringer fra 13 transporter à 20 tønner er analysert og sammenfattet nedenfor.

## **TRANSPORTPROSESSEN**

Transportprosessen kan deles inn i følgende tre faser:

1. Opplasting inkludert uttak fra lagerbygg, utsjekking av avfallsbeholdere og plassering på lastepalet
2. Transport langs vei
3. Lossing og plassering av avfallsbeholdere i Himdalenanlegget



Avfallsbeholderne hentes ut fra lageret ved hjelp av truck. Før transporten blir hver enkelt avfallsbeholder kontrollert og merket i henhold til ADR-reglementet (Direktoratet for brann- og eksplosjonsvern, 1999). Dette arbeidet foregår i en spesialinnredet container. Ved hjelp av en dreieskive og fast montering av måleutstyr kan nødvendige målinger foregå med målepersonalet bak en betongskjerm og i ca. 1 m avstand fra tønner.

Ved plassering på lastepalet blir tønner med høy strålingsintensitet skjermet av tønner med lav strålingsintensitet. Tønnene med lavest strålingsintensitet plasseres nærmest sjåføren slik at strålingsbelastningen til sjåføren blir minst mulig.

Selve transporten foregår langs ca. 25 km vei fra IFE Kjeller til KLDRA Himdalen, og tar ca 45 minutter.

I KLDRA Himdalen blir avfallsbeholderne losset av og plassert i deponi-/lagerhall ved hjelp av kran.

Normalt har to til tre personer fra seksjon Radavfall og én person fra seksjon Strålevern, i tillegg til sjåføren, deltatt ved hver transport. Hovedoppgavene til personellet fra Radavfall er uttak fra lagerbygg, plassering på lastepalet og lossing og plassering av avfallsbeholdere i Himdalen. Seksjon Strålevern bistår med utsjekking og merking av avfallsbeholdere, og overvåker strålevern og arbeidsteknikker.

## MÅLING AV PERSONDOSER

I tillegg til TL- (termoluminescens) dosimetre som leses av hver måned, har elektroniske dosimetre (Stephen 6000 og Rados, Rad 50) blitt benyttet for å registrere doser fra de ulike fasene i arbeidet, henholdsvis opplasting, transport og lossing. De elektroniske dosimetrene skulle fortrinnsvis festes på brystet ved siden av TL-dosimeteret, men har tidvis blitt båret i lommer på grunn av dårlig informasjon.

Dosimetrene ble lest av før og etter hver fase, og resultatene ble registrert i eget skjema.

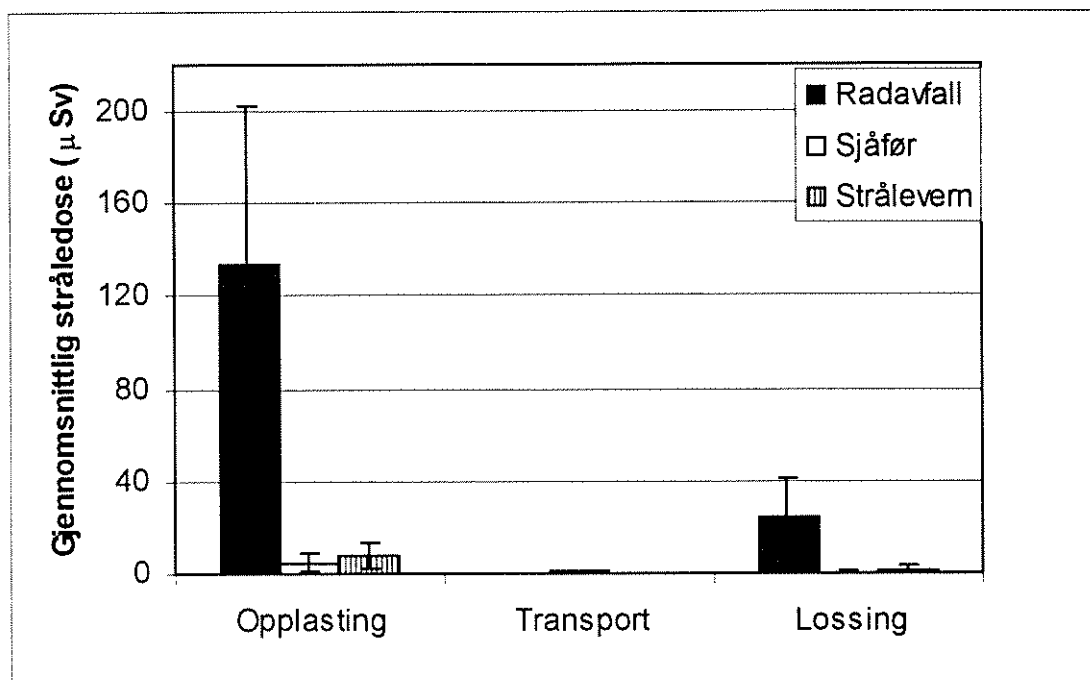
Ved beregning av persondoser er doser fra naturlig bakgrunnsstråling trukket fra. Doseraten fra bakgrunnsstrålingen ble målt ved hjelp av et Stephen 6000 dosimeter på to forskjellige steder innenfor instituttet, men i god avstand fra de nukleære anleggene. Målingene ble integrert over en time. På begge stedene var doseraten tett opp til 0,2  $\mu\text{Sv/h}$ .

## RESULTATER

Figur 1 viser gjennomsnittlige stråledoser fra 13 transporter à 20 tønner fordelt på de ulike fasene og personalkategoriene. Når det gjelder Radavfall, er dosen fordelt på to til tre personer, da det alltid er minst to som er til stede samtidig.

Resultatene viser følgende:

1. Hver transport fører til en gjennomsnittlig kollektivdose på ca.  $170 \cdot 10^{-6}$  manSv.
2. De høyeste dosene får personell fra Radavfall i opplastningsfasen. Det antas at det er uttak av avfallsbeholdere fra lagerbygningen som bidrar mest til dosen.
3. Selve transporten har ikke gitt noen dose utover bakgrunnen.



**Fig. 1.** Gjennomsnittlig dosebelastning per transport à 20 tønner

Med utgangspunkt i 100 transporter i året, som nok er i overkant av hva som er mulig, blir den totale dosebelastning på ca.  $17 \cdot 10^{-3}$  manSv/år. Det tilsvarer en gjennomsnittlig årtdose på 4-5 mSv for hver av 4 personer fra Radavfall som arbeider med transporten.

## REFERANSER

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# RADIOLOGISK KARTLEGGING I HIMDALEN

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## SAMMENDRAG

I forbindelse med byggingen av kombinert lager og deponi for lav- og middels radioaktivt avfall (KLDRA) i Himdalen, har Institutt for energiteknikk gjennomført en kartlegging av antropogen og naturlig radioaktivitet i området der anlegget ligger. Hensikten var å dokumentere bakgrunnsnivåer for en rekke radioaktive nuklider slik at disse skulle være kjent før driften av anlegget ble satt i gang. Bakgrunnsverdiene for de antropogene radionuklidene har videre blitt sammenliknet med aksjonsgrensene nevnt i IFEs sikkerhetsrapport for drift av anlegget.

Resultatene viser at bakgrunnsnivået for vannprøver ligger betydelig lavere enn aksjonsgrensene. Overvåking av eventuelle lekkasjer fra driften av anlegget vil konsentreres om prøvetaking og analyse av vann fra dreneringssystemet i deponiet og fra tilgrensende bekker og innsjøer. Det er rimelig å anta at en slik overvåking vil avdekke eventuelle lekkasjer fra anlegget lenge før aksjonsgrensene eventuelt blir overskredet.

## INTRODUKSJON

I konsekvensutredningen for KLDRA-Himdalen (1992) ble det anbefalt å ta prøver fra anleggets drensvann og nærliggende naturmiljø. Så lenge dreneringen i anlegget fungerer, skal ikke avfallet kunne være i kontakt med inntrengende vann. En eventuell lekkasje vil, om systemet skulle svikte, høyst sannsynlig først oppdages i drensvannet.

Himdalen har mottatt radioaktivt nedfall tidligere, både etter atmosfæriske prøvesprengninger av kjernevåpen på 1950- og 1960-tallet og etter Tsjernobyl-ulykken i 1986. Prøveprogrammet inkluderte derfor en grundig undersøkelse av hele området.

Undersøkelsen ble basert på måling av de vanligste antropogene radionuklidene. I tillegg ble bakgrunnsnivået for naturlig forekommende radionuklider etablert. Dette ble gjort fordi anlegget også vil ta hånd om en del lavradioaktivt avfall som inneholder uran.

## PRØVETAKING

Prøvetakingen ble i hovedsak utført vår og høst 1996. I tillegg ble det foretatt noen analyser av drikkevann fra brønner i området samt fra brønnen i deponiet våren 1998.

Følgende prøvetyper ble inkludert (lokalitetene 1-14 samt område A og B er vist i Fig. 1):

- bekkevann, bekkesedimenter (lokalitetene 1-6, begge sider av vannskillet). Bekkesedimentene ble delt i segmenter etter dybde med 5 cm intervaller.
- myrjord (lokalitetene 11-14, begge sider av vannskillet)
- drikkevann fra brønner i Himdalen (lokaliteter ikke vist på kart)
- vann fra brønn i deponiet
- overflatevann (lokalitetene 7-10)
- grunnvann og bergartsprøver fra borehull (ved deponiet)

- elg (kjøtt og knokler, område B)
- fisk (Dalatjern)
- kumelk (Bjørkefløtten gård, ikke vist på kart)
- vegetasjon (område A og B): torvmyrull (*Eriophorum vaginatum*), blåbær (*Vaccinium myrtillus*), tyttebær (*Vaccinium vitis idaea*), lav (hovedsakelig *Cladina stellaris*), furu (*Pinus sylvestris* L.), bjørk (*Betula pubescens*). I tillegg ble det samlet noe sopp under feltarbeidet høsten 1996

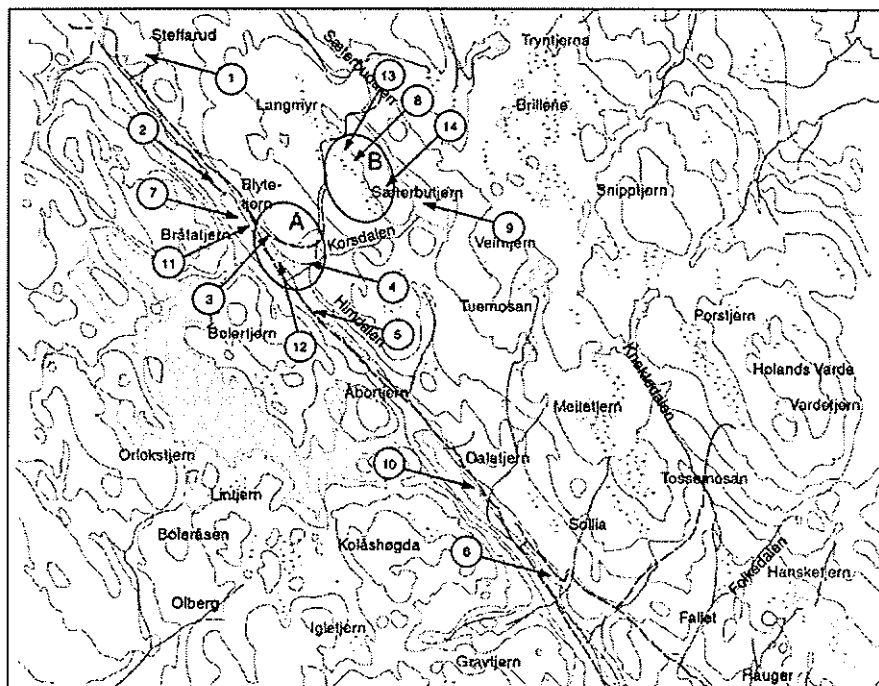


Fig. 1. Kart over Himdalen med angivelse av prøvesteder for overflatevann-, sediment-, myr- og vegetasjonsprøver.

## MÅLEMETODER

De vanligste bergartene i Norge inneholder mer  $^{238}\text{U}$  enn  $^{232}\text{Th}$ . Dette gjelder også bergartene i Himdalen (hovedsakelig amfibolittisk gneis (Statsbygg 1998)). Måling av naturlig radioaktivitet er derfor konsentrert om måling av nuklider i  $^{238}\text{U}$ -kjeden. Bestemmelsene er hovedsakelig basert på måling av  $^{214}\text{Pb}$ . Forutsetningen er da at radioaktiv likevekt er etablert i prøvematerialet. De forskjellige elementene i en radioaktiv serie vil imidlertid ha ulike geo-kjemiske egenskaper og vil være svært ulikt fordelt i naturlige systemer (berggrunn-vann-jord, økosystemer osv). I enkelte prøvetypene kan det derfor ikke antas likevekt gjennom hele kjeden. For alle vannprøver, samt prøver av vegetasjon, fisk, elg og melk, representerer aktiviteten av  $^{214}\text{Pb}$  alle nuklidene i  $^{238}\text{U}$ -kjeden fra  $^{226}\text{Ra}$  til  $^{210}\text{Pb}$ . For de andre prøvetypene representerer  $^{214}\text{Pb}$  aktiviteten av alle nuklidene i serien.

For best mulig å kunne avsløre eventuelle lekkasjer av uran eller uran-døtre fra anlegget, ble vannprøvene analysert med hensyn på både  $^{238}\text{U}$  og  $^{214}\text{Pb}$ .

Målingene av  $^{137}\text{Cs}$ ,  $^{60}\text{Co}$  og  $^{214}\text{Pb}$  ble utført ved hjelp av høyoppløselig gammaspektrometri.  $^{90}\text{Sr}$  ble målt i lavbakgrunns betakamre, mens  $^{239,240}\text{Pu}$  ble målt ved hjelp av alfaspektrometri. Både  $^{90}\text{Sr}$  og  $^{239,240}\text{Pu}$  ble før måling separert kjemisk fra prøvematriks.  $^{238}\text{U}$  ble målt ved hjelp av fluorimetri.

## RESULTATER

For antropogene radionuklider ble konsentrasjoner over deteksjonsgrensen målt for nuklidene  $^{137}\text{Cs}$ ,  $^{90}\text{Sr}$  og  $^{239,240}\text{Pu}$ . Resultatene for  $^{60}\text{Co}$  (deteksjonsgrenseverdier) er imidlertid også inkludert for vannprøvene. Resultatene er sammenfattet i Tabell 1 og 2.

Det er kjent at nivåene for antropogen radioaktivitet kan variere svært mye, selv over korte avstander (Gaare 1987a; Gaare 1987b, Varskog et al. 1994). Den observerte store variasjonen i nivåene for biologisk materiale, myrmateriale og bekkesedimenter gjør derfor slike prøver uegnet som referansemateriale.

Nivåene for  $^{137}\text{Cs}$  var som forventet etter kunnskap om nedfall etter Tsjernobyl-ulykken og atmosfæriske prøvesprengninger. Det er gjort få målinger av  $^{90}\text{Sr}$  og  $^{239,240}\text{Pu}$  i prøver av flora/fauna og jord-/ myrmateriale i Norge. Av den grunn er det ikke mulig å relatere resultatene med kunnskap om nivåene i Norge generelt. Nivåene er imidlertid sammenliknbare med nivåer som måles i området rundt Institutt for energiteknikk i forbindelse med overvåkingsprogrammet for instituttets nukleære virksomhet.

**Tabell 1.** Bakgrunnsverdier for antropogen radioaktivitet i ulike typer materiale fra Himdalen. Verdiene er gitt som Bq/kg tørrvekt for bekkesediment, myrmateriale, bergartsmateriale, vegetasjon, hel fisk og elg. For drikkevann, bekkevann, overflatevann, grunnvann og melk, er verdiene gitt som Bq/l. "–" angir ikke analysert.

Prøvetype	$^{137}\text{Cs}$	$^{90}\text{Sr}$	$^{239,240}\text{Pu}$	$^{60}\text{Co}$
Drikkevann	< 0,0028	< 0,002	< 0,00004	< 0,0024
Bekkevann	< 0,07	0,011 – 0,013	0,0002 – 0,0005	< 0,02
Overflatevann	0,009 – 0,02	0,004 – 0,011	0,00009 – 0,0002	< 0,01
Grunnvann	< 0,01	–	–	< 0,01
Bekkesediment	60 – 560	2 – 13	0,6 – 2,4	–
Myrmateriale (0 - 5 cm)	30 – 410	11 – 51	1,6 – 2,2	–
Bergartsmateriale	–	–	–	–
Vegetasjon (sopp unntatt)	2 – 330	2 – 50	0,02 – 0,04	–
Fisk	70 – 350	74 – 93	0,02 – 0,05	–
Elgkjøtt	$1,12 \pm 0,05$	$0,22 \pm 0,01$	$0,0029 \pm 0,0005$	–
Elgknokler	$1,00 \pm 0,05$	$78 \pm 3$	$0,033 \pm 0,005$	–
Melk	–	$0,049 \pm 0,001$	–	–

For vannprøvene er konsentrasjonene av de antropogene radionuklidene betydelig lavere enn de foreslåtte aksjonsgrensene (0,2 Bq/liter for  $^{137}\text{Cs}$  og 0,08 Bq/liter for  $^{90}\text{Sr}$ ) i IFEs sikkerhetsrapport for drift av anlegget. Men også for vannprøvene ble det dokumentert nivåforskjeller i innhold av radioaktivitet. Det er rimelig å anta at konsentrasjonene av radionuklider i vannsystemene i Himdalen reflekterer nivåene i hele nedslagsfeltet. Innholdet av radionuklider i overflatevann vil derfor variere med sesong og mikroklimatiske forhold.

I elgkjøtt ble det funnet små mengder av  $^{137}\text{Cs}$ ,  $^{90}\text{Sr}$  og  $^{239,240}\text{Pu}$ . Som forventet var konsentrasjonen av  $^{90}\text{Sr}$  og  $^{239,240}\text{Pu}$  større i knoklene. Mengden radioaktivitet i fisk og melk anses som normale.

Det ble ikke funnet detekterbare mengder antropogene radionuklider i bergartsprøvene. Nivåene for naturlig radioaktivitet ligger innenfor normalområdet (20 – 100 Bq/kg, Statens strålevern, 1994) for gneis i Norge.

**Tabell 2.** Bakgrunnsverdier for naturlig radioaktivitet i forskjellige typer materiale fra Himdalen. Prøvene er målt etter innstilling av radioaktiv likevekt med  $^{226}\text{Ra}$ . Verdiene er gitt som Bq/kg tørrvekt for bekkesediment, myrmateriale, bergartsmateriale, vegetasjon og hel fisk. For drikkevann, bekkevann, overflatevann og grunnvann er verdiene gitt som Bq/l. "–" angir ikke analysert.

Prøvetype	$^{214}\text{Pb}$	$^{238}\text{U}$
Drikkevann	< 0,006	0,02 – 0,40
Bekkevann	0,02 – 0,08	0,005 – 0,02
Overflatevann	0,02 – 0,04	< 0,005
Grunnvann	0,06 – 0,25	–
Bekkesediment	23 – 57	–
Myrmateriale (0 - 5 cm)	10 – 71	–
Bergartsmateriale	23 – 74	–
Vegetasjon (sopp unntatt)	6 – 26	–
Fisk	5 – 50	–

## KONKLUSJON

Overvåkingen av eventuelle lekkasjer fra driften av anlegget vil baseres på analyse av vann fra dreneringssystemet i deponiet og fra de tilgrensende bekker og innsjøer. En slik overvåking antas å kunne avdekke eventuelle lekkasjer fra anlegget lenge før aksjonsgrensene overskrides. Aksjonsgrensene er satt slik at dosen til enkeltindivider fra vannkonsum ikke skal overskride 1  $\mu\text{Sv}$  i året.

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## Nuclear threats in the vicinity of the Nordic countries

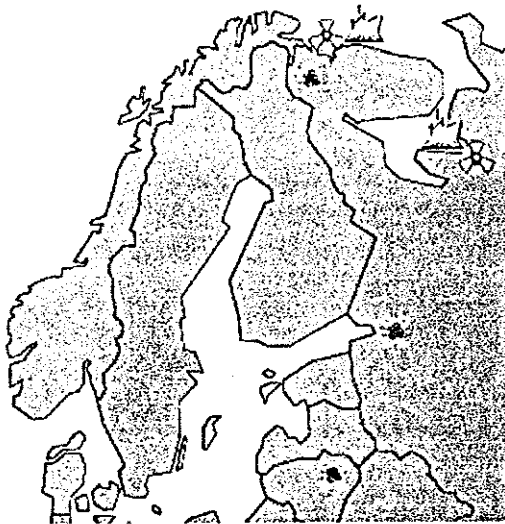
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### Introduction

This project is one of the new cross-disciplinary studies in the NKS research program 1998-2001. The main task for the project is to aggregate knowledge of nuclear threats in the vicinity of the Nordic countries, a "base of knowledge", and make this available for the Nordic authorities as a supplement for the national emergency preparedness work.

The project will focus on potential events in nuclear installations and the consequences for the Nordic countries especially on:

- vulnerable food chains
- doses to man
- environmental contamination
- the emergency preparedness system



*Fig 1: Potential threats in the vicinity of the Nordic countries*

The acute phase of an accident and the possibility of high exposure to the populations is always the most important factor in the emergency preparedness work. Radioactive contamination from an accident can however also cause long time effects for land use and enhanced doses to special population groups and economic problems for agriculture, reindeer industry, hunting, tourism and recreation. The project deals with nuclear installations in a geographical area including the north west of Russia and the Baltic states.



## Method

The project is organised with a project leader and a project group which will co-ordinate the different tasks and keep the whole project together. Working groups are established to focus on the different subprojects and perform different tasks such as organising workshops, making summary reports and identify gaps of knowledge.

The nuclear installations that will be investigated in the subprojects are:

- threats from nuclear power plants (Kola NPP, Ignalina NPP, Leningrad NPP)
- threats from ship reactors (icebreakers and submarines)
- threats from storage and handling of used fuel and radioactive waste

### Literature database

The purpose with this part of the project is to prepare a list of publications and projects that have been produced regarding possible nuclear threats in the vicinity of the Nordic countries. Publications with Nordic participants have the highest priority, but also publications from elsewhere which are relevant in this context will be included. The literature database will be part of the "base of knowledge".

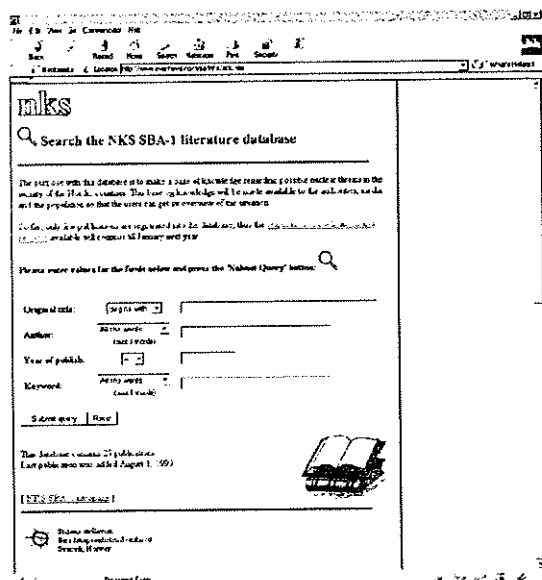


Fig. 2: Literature database

### Workshops

Based on the evaluation of the literature, work shops will be conducted to get specialists together to discuss the different findings and to aggregate new knowledge based on these investigations. If gaps of knowledge are identified, the project group will initiate further studies or identify the needs for the consideration of national organisations and authorities.

## Results

The scope of the project is to prepare a “base of knowledge” regarding possible nuclear threats in the vicinity of the Nordic countries. This base of knowledge will, by modern information technology as different web-sites, be made available to authorities, media and the population. The users of the web-sites can easily get information on different types of nuclear installations and threats.

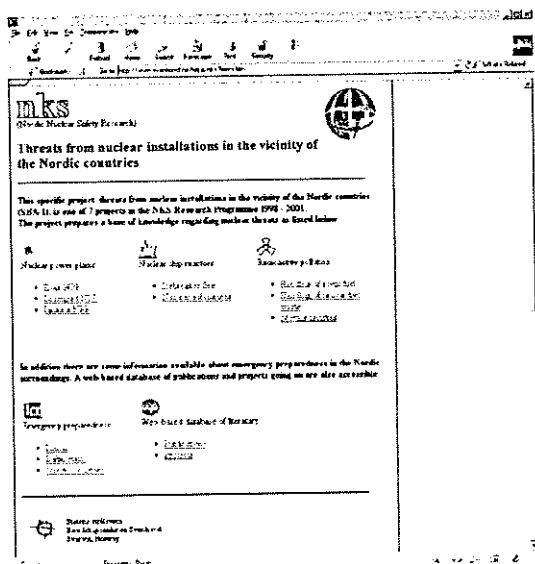


Fig. 3: Base of knowledge



# $^{210}\text{Pb}$ AND $^{238}\text{U}$ IN ESTONIAN FUEL PRODUCTS AND ASHES

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**Abstract**--Concentrations of  $^{210}\text{Pb}$  and  $^{238}\text{U}$  ( $^{234}\text{Th}$ ) in local fuels (oil shale, wood and wood-peat mixtures) and in their combustion products were determined using low-background low-energy HPGe gamma spectrometry. The samples were collected from a few operating power plants. In the spectrum analysis, self-attenuation corrections were modelled for each sample. The model took into account the measurement geometry and the determined attenuation coefficients of the sample. An analysis of the radionuclide balance in the combustion process of oil shale showed that approximately 30 % of  $^{210}\text{Pb}$  became volatile and its behaviour differed significantly from that of other natural radionuclides, e.g.  $^{238}\text{U}$ . While depleted in bottom ash in comparison to  $^{238}\text{U}$ ,  $^{210}\text{Pb}$  showed a considerable enrichment (up to 120 Bq kg<sup>-1</sup> or 7 times relative to fuel) in the electric filter ash fractions. Near the power plants enhanced  $^{210}\text{Pb}$  concentrations were observed in the surface soil layers, as a result of atmospheric dispersion and deposition of oil shale fly-ash. Even higher enrichment factors (up to 20) were found in ashes of local biofuels (wood, wood-peat mixtures), where the  $^{210}\text{Pb}$  concentrations reached values of 270 Bq kg<sup>-1</sup>. Annual population doses in the range of 20 .. 60 µSv a<sup>-1</sup> caused by technologically dispersed radionuclides were evaluated using a compartmental modelling method near the oil-shale-fired power plants.

## INTRODUCTION

Electricity production in Estonia is mainly based on the burning of oil shale in two large power plants located in the NE region. Local heat production all over the country uses imported natural gas or local biofuels (wood and peat or their mixtures). Both oil shale and biofuels contain trace quantities of natural radionuclides, the concentrations of which enhance considerably in their combustion products. These radionuclides released to the atmosphere via stacks with fly ash and flue gases or to the water pathways from the ash depositories, may cause radiological impact to the population (Realo *et al.* 1996). Insofar no similar studies are available for the Estonian biofuel-burning plants.  $^{210}\text{Pb}$  has relatively high values of dose conversion factors both for inhalation and ingestion (Keverling Buisman 1998) and it adds an important contribution to the technologically induced dose. In our preliminary study some specific features of radionuclide balance in the combustion processes of oil shale have been established (Realo *et al.* 1998). The present study has an objective to improve the data on the partitioning of  $^{210}\text{Pb}$  and  $^{238}\text{U}$  between various ash fractions and to assess their dose contributions.

## METHODS

Oil shale and ash samples were collected within 1.5-2 hours along the technological cycle of one of the furnaces of the Baltic Power Plant (Narva) in 1996 and in 1998. The following samples were collected: mill-pulverised oil shale, bottom ash, cyclone ash and electrostatic filter-precipitator ash from three (or four) successive stages, I, II, III (or IV). Wood, wood-peat mixtures and their ashes were collected from two thermal plants in Tartu in 1997-1998.

All samples were dried, homogenised, weighed and stored in sealed plastic or metal containers (at least two containers per each) for at least 3 weeks before measurement to allow for the in-growth of radon daughters. A low-background  $\gamma$  spectrometer with a planar 800 mm<sup>2</sup> HPGe detector was used for the analysis of  $^{210}\text{Pb}$  and  $^{234}\text{Th}$  ( $^{238}\text{U}$ ) nuclides with low  $\gamma$  energy. In addition, high-energy  $\gamma$  spectra for the other nuclides in the samples were also analysed by using a low-background  $\gamma$  spectrometer with a 42 % coaxial HPGe detector. The energy and

efficiency scales of the spectrometers were calibrated with the certified IAEA-RG-SET reference materials in identical containers. The IAEA GANAAS 3.1 software was used for  $\gamma$  spectrum analysis. The following method was applied for self-attenuation correction in low-energy  $\gamma$  spectrum analysis. Direct transmission measurements for each sample, reference material and empty container were made by using the sealed sources of  $^{210}\text{Pb}$  (or  $^{226}\text{Ra}$ ) and  $^{238}\text{UO}_2$  to determine the 46.5 keV and 63.3 keV peak attenuation coefficients  $\mu_I$ , respectively. The self-attenuation correction factors,  $F_{ref}(\mu_I)$  and  $F_{sample}(\mu_I)$  for reference and sample, respectively, were calculated using a numerical integration method for a cylindrical container and a disk detector coaxial combination (a four fold integral). The determined  $\mu_I$  values, the actual container and detector dimensions, incl. the distance between them, were used as input parameters for a MATHCAD routine to calculate the ratio,  $F_{ref}(\mu_I)/F_{sample}(\mu_I)$ . This correction factor is used to correct the self-attenuation in the measured sample activity. It should be stressed that a simple equation,  $F(\mu_I) = [1 - \exp(-\mu_I t)]/\mu_I t$ , where  $t$  is the sample thickness, offers practically coinciding correction factors for the used measurement geometry and samples. The maximum differences in the factors calculated by two methods are  $< \pm 5\%$  in the range of  $\mu_I$  from  $0.01\text{ cm}^{-1}$  to  $1\text{ cm}^{-1}$ .

## RESULTS AND DISCUSSION

### *Oil shale and ash products*

Mean concentrations of  $^{210}\text{Pb}$  and  $^{234}\text{Th}$  ( $^{238}\text{U}$ ) (in  $\text{Bq kg}^{-1}$ ) are presented in Table 1. For a comparison data for  $^{226}\text{Ra}$  are also included. The activity

Table 1. Mean activity concentrations ( $\text{Bq kg}^{-1}$  dry weight with  $\pm 1\sigma$ ) of  $^{210}\text{Pb}$ ,  $^{234}\text{Th}$  ( $^{238}\text{U}$ ) and  $^{226}\text{Ra}$  in oil-shale and oil-shale ashes, Baltic Power Plant, 1996 (Realo *et al.* 1996) and 1998.

	$^{210}\text{Pb}$			$^{226}\text{Ra}$			$^{238}\text{U}$		
	1996	1998	Average	1996	1998	Average	1996	1998	Average
oil-shale	$15 \pm 3$	$12 \pm 4$	13	$13.5 \pm 1.2$	$12.0 \pm 0.3$	12.8	$16 \pm 3$	$10 \pm 3$	13
bottom ash	$20 \pm 4$	$15 \pm 5$	18	$25.5 \pm 0.8$	$37.4 \pm 1.0$	31.4	$28 \pm 3$	$34 \pm 8$	31
cyclone ash	$29 \pm 4$	$40 \pm 5$	35	$48.2 \pm 0.4$	$46.1 \pm 0.9$	47.2	$48 \pm 4$	$43 \pm 6$	46
filter ash I	$46 \pm 3$	$64 \pm 5$	55	$50.9 \pm 0.5$	$49.8 \pm 1.2$	50.3	$57 \pm 4$	$51 \pm 9$	54
filter ash II	$80 \pm 5$	$95 \pm 8$	87	$57.1 \pm 0.5$	$55.6 \pm 1.4$	56.3	$64 \pm 4$	$53 \pm 9$	58
filter ash III	$102 \pm 4$	$110 \pm 8$	106	$61.1 \pm 0.4$	$56.8 \pm 1.5$	59.0	$64 \pm 3$	$60 \pm 10$	62
filter ash IV		$122 \pm 8$	122		$67.8 \pm 1.3$	67.8		$74 \pm 11$	74

concentrations of all radionuclides increase in ashes due to burnout of organic matter and as a result of various volatilisation-condensation processes. An increase in the activity concentrations from oil shale to filter ash and from filter ash I to filter ash III (IV) is common for all radionuclides. Nevertheless, for  $^{226}\text{Ra}$ ,  $^{234}\text{Th}$  ( $^{238}\text{U}$ ) the range and features of this increase are different in comparison to  $^{210}\text{Pb}$ . Approximately parallel behaviour with small systematic deviations (up to 9 %) from radioactive equilibrium is characteristic to the enrichment of  $^{238}\text{U}$  and  $^{226}\text{Ra}$ . A conclusion was drawn (Realo *et al.* 1996) about their predominantly non-volatile behaviour in the combustion processes.  $^{210}\text{Pb}$  and  $^{226}\text{Ra}$  are in secular equilibrium with  $^{234}\text{Th}$  ( $^{238}\text{U}$ ) in oil-shale only, while the ratio  $A(^{210}\text{Pb})/A(^{226}\text{Ra})$  decreases down to 0.5 in bottom and cyclone ashes and increases up to 1.8 in the finer fractions of filter ash. As the mean particle size decreases in the sequence: filter ash I > II > ...IV, it is reasonable to assume that at high temperatures certain volatilisation-condensation processes induce the observed particle-size-

dependent increase in the activity concentrations of radionuclides. Defining the enrichment factor of a radionuclide as a ratio of its activity concentrations in a given ash fraction to that of oil-shale, the changes in enrichment for different ash fractions are shown in Fig.1.

The enrichment behaviour of  $^{210}\text{Pb}$  in the oil-shale combustion is qualitatively similar to that observed for coal (e.g., Tadmor 1986) or peat (Hedvall and Erlandsson 1992). At the same time U is usually more volatile than Ra or Th in coal burning (e.g., Tadmor 1986). The observed similar enrichment behaviour of U and Ra (or Th) is a rather unique feature in oil-shale combustion. An estimation of the radionuclide balance has shown that the activities of  $^{226}\text{Ra}$ ,  $^{238}\text{U}$  and  $^{232}\text{Th}$  are preferably bound to solid ash particles. At the same time ~ 30 % of the total input activity of  $^{210}\text{Pb}$  is discharged in the volatilized form with flue gases from the stacks to the atmosphere. The calculated specific atmospheric discharges of activities via the power plant stacks amounted to 62 GBq  $\text{GW}_{\text{el}}^{-1}$  for  $^{210}\text{Pb}$  and 9 GBq  $\text{GW}_{\text{el}}^{-1}$  for  $^{238}\text{U}$ , respectively (Realo K and Realo E 1998). This radiologically important conclusion is also supported by our recent analysis of soil profiles collected in the vicinity of the power plants.

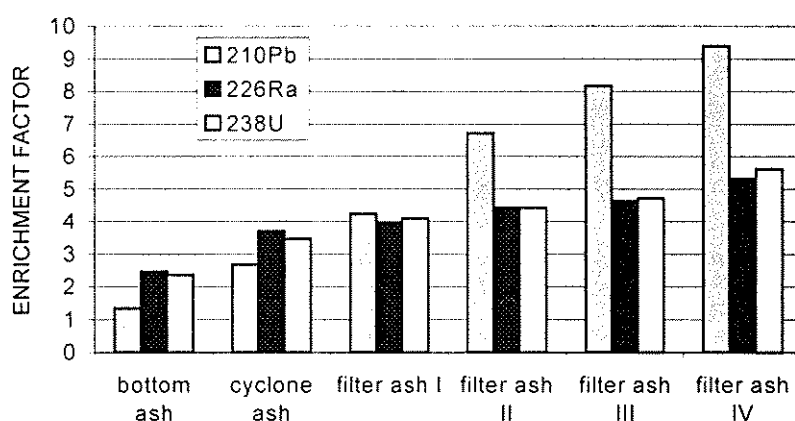


Fig. 1. Enrichment factors of  $^{210}\text{Pb}$ ,  $^{226}\text{Ra}$ , and  $^{238}\text{U}$  for different oil shale ash fractions.

Table 2. Average activity concentrations ( $\text{Bq kg}^{-1}$  dry weight with  $\pm 1\sigma$ ) of  $^{210}\text{Pb}$ ,  $^{226}\text{Ra}$  and  $^{234}\text{Th}$  in wood, wood-peat mixture and their ashes, Tartu, 1997 - 1998.

Product		$^{210}\text{Pb}$	$^{226}\text{Ra}$	$^{234}\text{Th}$
Aardla TP	wood (S Estonia)	< 12	< 2	< 12
	bottom ash	< 12	$52 \pm 1$	$32 \pm 6$
	cyclone ash	$242 \pm 13$	$38 \pm 1$	$26 \pm 8$
Luunja TP, 1997	wood (N Estonia)	$18 \pm 8$	$3 \pm 1$	< 12
	bottom ash	$24 \pm 4$	$18 \pm 1$	$14 \pm 7$
	filter ash	$268 \pm 15$	$36 \pm 1$	$49 \pm 12$
Luunja TP, 1998	wood-peat mixture	$24 \pm 7$	$9 \pm 1$	$22 \pm 9$
	bottom ash	$16 \pm 3$	$20 \pm 1$	$17 \pm 6$
	filter ash	$220 \pm 15$	$97 \pm 2$	$24 \pm 6$

### *Wood, wood-peat mixture and ash*

The preliminary mean activity concentrations are given in Table 2. Radionuclide content of biofuels, especially those from S Estonia, is rather low and often below the LLD of our method. The same conclusion applies also for bottom ash, which is actually partly recirculated sand contaminated with ash radionuclides. Activity concentrations of  $^{210}\text{Pb}$  in filter or cyclone ash are rather similar for wood and wood-peat mixtures. In general, radionuclide concentrations, incl. also  $^{137}\text{Cs}$ , in Estonian local biofuels and their ashes are rather low in comparison to those found by Hedvall and Erlandsson 1992 in Sweden or by Mustonen 1985 in Finland. In fly ash, the concentrations and enrichment factors of  $^{234}\text{Th}$  are lower than those of  $^{210}\text{Pb}$  and  $^{226}\text{Ra}$ . The enrichment factors for both two are rather similar: with values larger for wood (~12 .. 20) than for wood-peat mixture (~9 .. 10). The set of samples analysed so far does not exhibit a preferable volatilisation of  $^{210}\text{Pb}$  in the burning processes, which is characteristic to oil shale.

Our data can be applied to estimate the population doses caused by radionuclides released to the atmosphere as a result of oil-shale-based energy production. We have used the CAP-88PC 2.0 (with permission of US DOE) software to calculate the population doses and their geographical distribution. As follows from our estimation, the maximum annual individual effective doses amount to 20 .. 60  $\mu\text{Sv}$  near the power plants, while the contribution of  $^{210}\text{Pb}/^{210}\text{Po}$  (~80 %) dominates. This estimate includes no contribution from the water pathways. The latter may be important as most of the produced ash is deposited to ashfields. At present we can not yet estimate the specific discharges of radionuclides to the atmosphere via the stacks of biofuel-burning plants. But without any doubt the caused doses should be lower than those caused by burning of oil shale. The further studies are in progress.

*Acknowledgements* -- We are grateful to B.Erlandsson and R.Hellborg (Lund University, Sweden) for useful discussions and help in quality control in low-energy gamma spectrometric measurements. We gratefully acknowledge also a partial financial support from the ESF grant No 2770.

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# AFSKÆRMNING MED OLIVINSAND.

## Om opbygningen af Lavstrålingsrummet ved SIS

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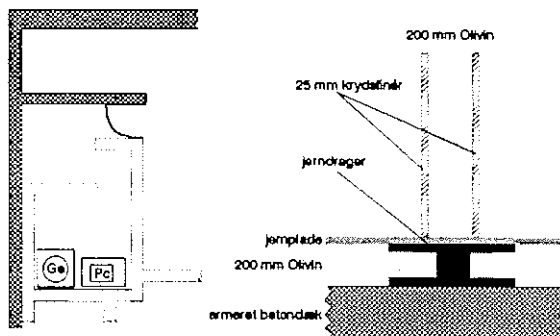
Da vi flyttede til vore nye lokaler (på Knapholm) i februar 1999 blev germaniumdetektoren anbragt i et nyindrettet og nykonstrueret "Lavstrålingsrum". Et af formålene med rummet var at reducere baggrundsstrålingen, specielt den del der stammer fra den naturlige radioaktivitet i de nærmeste omgivelser. Figurerne viser opbygningen og resultatet af målinger i rummet.

**Figur 1**

Lavstrålingsrummet.

Tegningen tv. viser hvordan lavstrålingsrummet er opbygget med vægge af et særligt afskærmningsmateriale, og har adgang gennem en "labyrint".

Som tegningen th. viser, består de skærmede vægge, og gulvet og loftet, af to plader mellem hvilke der er et 20 cm tykt lag af olivinsand.



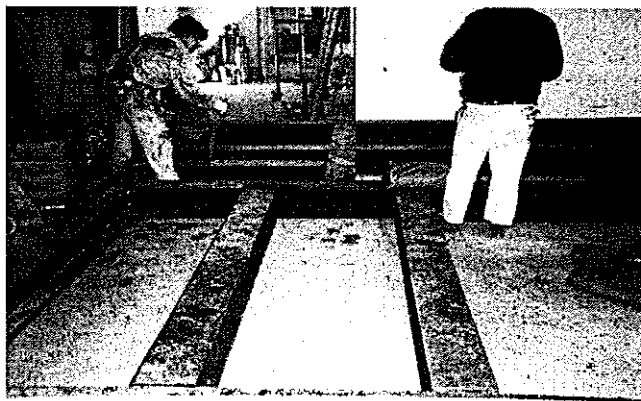
De bærende plader er af stål.

**Figur 2**

Lavstrålingsrummet bygges.

Olivinrummet ligger på 1. sal, og da det hele vejer 40 ton bæres det af solide jernbjælker der er fæstnet i søjlerne.

Olivinsandet fra Norge blev hældt ned mellem jernbjælkerne uden at blive blandet med cement.

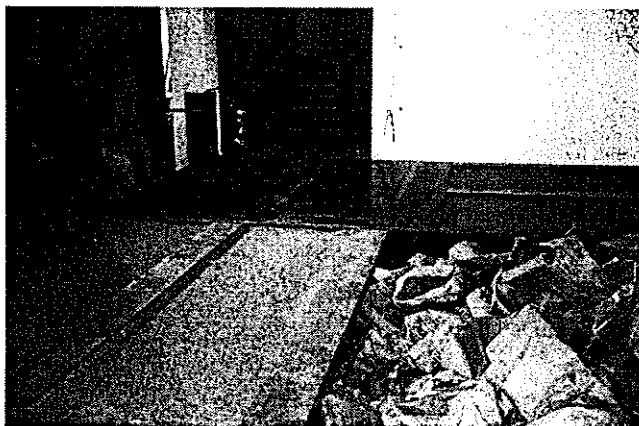


**Figur 3**

Gulvet bygges.

Her er lagt 2 af de jernplader der danner gulvet i de skærmede rum. Mellemrummet mellem jernbjælkerne er fyldt med olivinsand.

Man kan se hvordan det hele er spændt fast til en af søjlerne.





**Figur 4**

Væggene opbygges.

Vægpladerne omkring lavstrålingsrummet støttes af et metalskellet.

Man kan se hvor døråbningen ind til rummet skal være.



**Figur 5**

Noget om OLIVIN

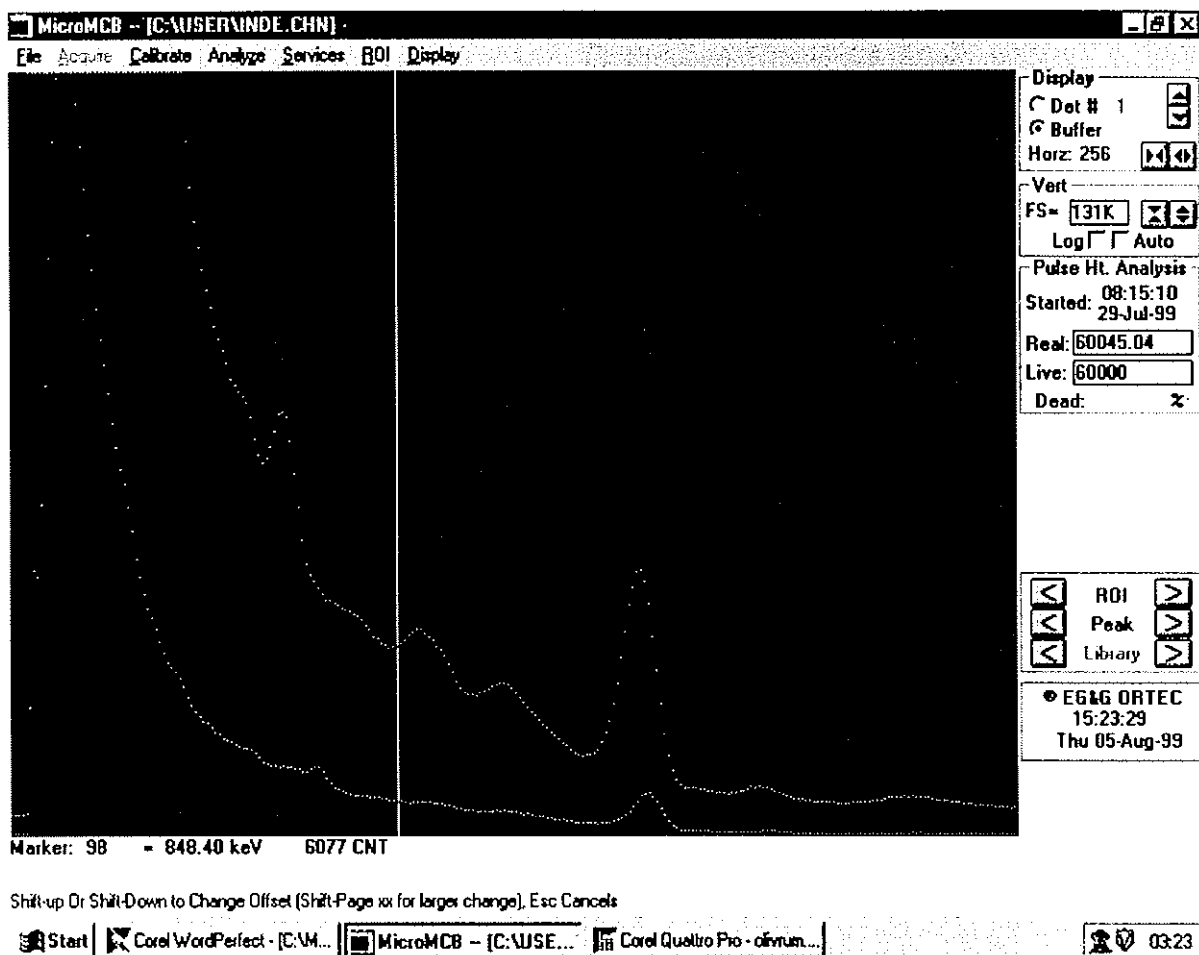
Olivin er et magnesium-, jernsilikat  $(\text{Mg,Fe})_2\text{SiO}_4$  med lavt indhold af kalium og tungmetaller som uran og thorium. Den krystallinske form kaldes "peridot", som har den flotte farve, der er vist på billedet.

I Norge findes olivin som kornede dunitt-masser der brydes til industribrug, og det er norsk olivinsand med kornstørrelse 2 - 5 mm vi har brugt til afskærmningen. Det egner det sig til vægge i rum som skal skærmes mod gammastrålingen fra de naturlige radioaktive stoffer, som K-40, Ra og Th, og da det uden cement pakkes til en massefylde omkring 2 giver det en god afskærmning af den samlede baggrundsstråling, sådan som målingerne gengivet nedenfor viser.

Det sand vi brugte havde mindre end 10 Bq K-40 pr. kg.



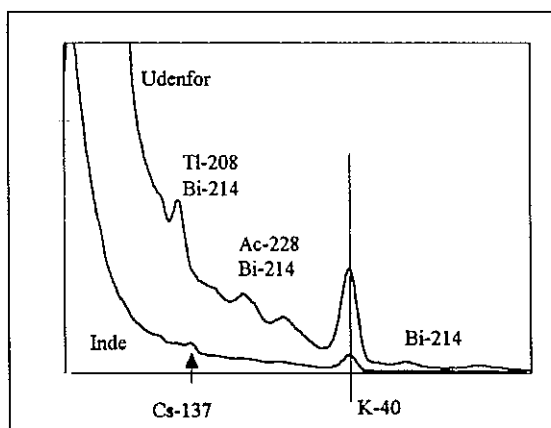
63. OLIVIN(peridot) AHE:PM 2 x 1,6 cm

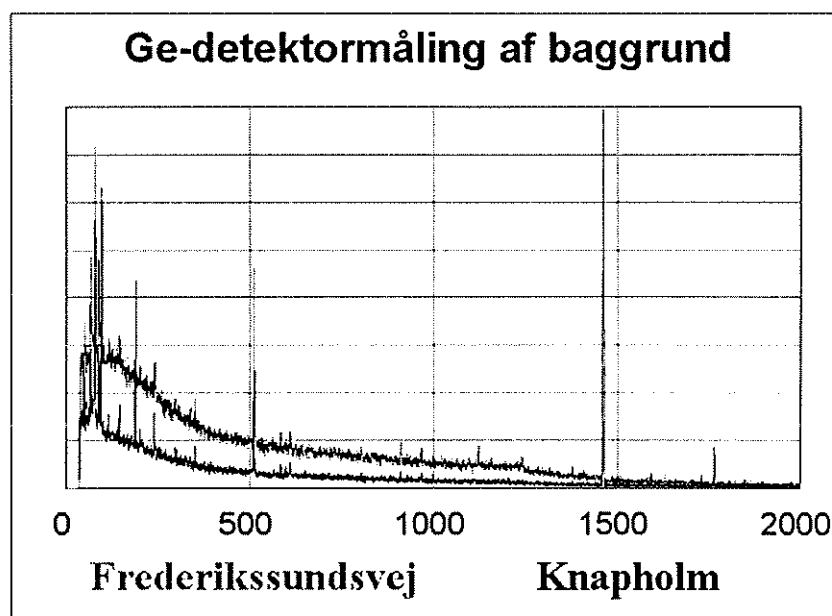


Figur 6

Spektre fra måling af  
baggrundsstrålingen i olivinrummet

Målingerne i rummet udenfor og inde  
i lavstrålingsrummet blev udført  
med en ukollimeret 3" NaI detektor,  
og illustrerer afskærmningseffekten  
af olivinvæggene.



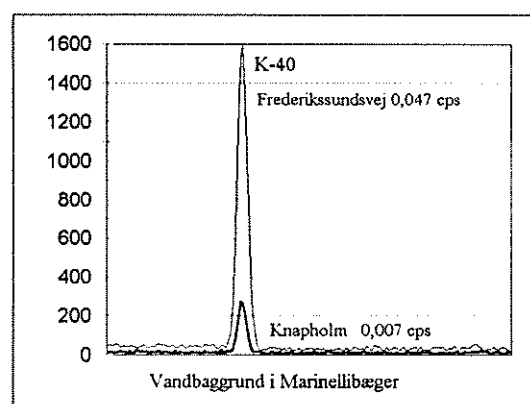


Figur 7

Ge-detektorens baggrund.

Her er en sammenligning af målingen af en vandbaggrund i det gamle laboratorium på Frederikssundsvej og i lavstrålingsrummet.

K-40 er fremhævet nedenfor.  
Tællertallet er reduceret ca. en faktor 7.



### Nogle måleresultater

Væggene med olivinsand og placeringen af lavstrålingsrummet på 1. sal har vist sig at give en lav baggrund, sådan som man kan se af de målinger udenfor og inde i rummet, der er vist på figur 6. Reduktionen er også vurderet ved måling af dosishastigheden. Med en Bicon monitor med vævsækvivalent plastikscintillator målte vi (inklusive den kosmiske stråling):

Dosishastighed uden for rummet	0,07 - 0,09 $\mu\text{Sv/h}$
Dosishastigheden inde i lavstrålingsrummet	0,01 - 0,02 $\mu\text{Sv/h}$

Den "baggrund" som skal fratrækkes ved måling af radium i prøver af vand og jord med Ge-detektoren i lavstrålingsrummet er reduceret så meget i forhold til bidraget på Frederikssundsvej, at det har indflydelse på de tælletal som spektrometerprogrammet Gammavision bestemmer ved rutinemålingerne. Til illustration vises her tallene for den Bi-214 gammalinie på 609 keV, der indgår i beregning af radiumindholdet.

	Baggrunden af brutto	1 std	MDA for jord
<b>På Frederikssundsvej</b>	ca 30 %	$\pm 2 \%$	1 Bq/kg
<b>I lavstrålingsrummet</b>	ca 20 %	$\pm 1 \%$	0,5 Bq/kg

### KONKLUSION

I de nye laboratorier ved Statens Institut for Strålehygiejne har vi indrettet et "Lavstrålingsrum". Det har vist sig at leve fint op til sit navn, og vil danne et godt udgangspunkt for en evt. installation af et "helkropstæller" til påvisning af interne forureninger.

## Mapping the Radiation Fields at a Research Reactor

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### Abstract

The DR 3 reactor at Risø National Laboratory is a multipurpose research reactor. It has the status of a Large European Beam Facility therefore its neutron scattering spectrometers are used by many visiting scientists. As a supplement to the routine health physics monitoring programmes a special survey has been made to get more detailed information of the radiation levels in the hall and of the most important sources of the radiation. The special survey consisted of three sorts of measurements: an extra set of thermoluminescence dosimeters, a set of continuous measurements of the dose rate at selected places and spot measurements with handheld instruments around the spectrometers. Some of the results from the survey are presented.

### 1. Introduction

The DR3 reactor at Risø is a 10 MW heavy water moderated and cooled multipurpose research reactor. The reactor operation cycle is 4 weeks of which 23.5 days is continuous operation and 4.5 days is shut down.

Three neutron scattering spectrometers are in operation inside the reactor containment. More spectrometers are placed in a building outside the containment. Neutrons for the latter spectrometers are transported through a neutron guide tube from a cold source in the reactor. Within recent years one of the spectrometers (TAS6) and the neutron guide has been completely renewed to enhance the neutron intensity in the connected spectrometers. This renewal has given an increase in the neutron intensity in the spectrometers but unfortunately also around the spectrometers and in the experimental halls.

The reactor is also used for silicon doping, isotope production and neutron activation analysis.

### 2. Routine monitoring

In the reactor hall 28 thermoluminescence dosimeters are placed to monitor the  $\gamma$ -radiation. They are renewed every 6-month and used for detecting long-term trends in the radiation level.

Daily around 100 measurements of the  $\gamma$ -dose-rate are made at fixed positions in the hall and are used for detecting short term changes in the radiation level.

### 3. Special survey

To get more detailed information on the dose rate levels and of the radiation sources a special survey has been conducted. It consisted of three sorts of measurements, which are described below.

**Dosimeters.** Some extra 50 thermoluminescence dosimeters were placed in the reactor hall. The dosimeters were placed at workplaces for the reactor staff and the researchers, at places where they would be a supplement to the routine dosimeters and where known sources were expected to give a significant contribution. The dosimeters measured the  $\gamma$ -dose and the dose from thermal neutrons accumulated over four three-month periods.

**Continuous dose rate measurements.** Continuous  $\gamma$ -dose rate measurements (5 minutes averages) were made at 10 places, to give information on the variations in the dose rate. This special attention was given to the workplaces for the scientists and the positions at the pump consoles for the silicon irradiation facilities. The minimum detectable dose rate was 1  $\mu\text{Sv/h}$ , thus values recorded as 1  $\mu\text{Sv/h}$  was between 0 and 1  $\mu\text{Sv/h}$ .

**Spot measurements.** Spot measurements around the neutron spectrometers were made. Changes in the  $\gamma$ -dose rate and the neutron dose rate were measured with handheld instrument to see the influence of different spectrometer settings.

## 4. Results

In the following the main results from the measurements at the experimental level in the reactor hall are reported. Figure 1 is a cross sectional drawing of the experimental level. The reactor block is in the middle. The triple axis spectrometers (TAS) are placed on platforms at face 1 and 3. The neutron guide tube starts from face 3. The PC cabins at the two corners houses the spectrometers controls. Two silicon doping facilities (Si) are placed at face 2 and 4. The ISB is storage for used fuel elements. The small triangles are building columns.

**Dosimeters.** The measured average dose rates are shown in figure 1. The numbers within the containment circle are from the special survey while the numbers outside the circle are average dose rate values obtained from the routine monitoring dosimeters placed on the containment wall at a height of approximately 3 m above the floor level. The numbers at the triangles are a representation of the dose rate along the walkway round the reactor block. The dose rates on the walkway are higher on the left-hand side of the figure, which is due to the presence of the neutron guide and the fuel storage. This difference is also reflected on the routine monitoring dosimeters. The neutron guide passes low over the walkway and the given dose rates are valid close to the outer surface of the guide.

**Continuous dose rate measurements.** Figure 2 shows the spectrum obtained in a position near the lift. It is seen that the dose rate varies in time and has significant peaks. Table 1 gives statistical information obtained from the spectrum in figure 2. The onset of the peaks could be correlated to pile in of crystals in the Si 1 silicon irradiation facility. When the peaks were removed from the spectrum, the average dose rate (during reactor operation) dropped 3 %. The spectra from the PC cabins were also peaked. Some of the peaks in one of the spectra could be correlated with the moving of crystals in a nearby irradiation facility whereas most of the other peaks must be ascribed to spectrometer operations. The mean values obtained for the PC cabins were in accordance with the means obtained by the dosimeters in the cabin.



**Table 1.** Statistics for the continuous dose rate spectrum shown in figure 2. Only data from the reactor running periods are shown.

<b>Minimum</b>	<b>1 <math>\mu\text{Sv/h}</math></b>	<b>Harmonic mean</b>	<b>2.9 <math>\mu\text{Sv/h}</math></b>
<b>Maximum</b>	<b>45 <math>\mu\text{Sv/h}</math></b>	<b>Kurtosis function value</b>	<b>44</b>
<b>Mean</b>	<b>4.9 <math>\mu\text{Sv/h}</math></b>	<b>Peak width 5-10 minutes</b>	
<b>Standard deviation</b>	<b>2.7 <math>\mu\text{Sv/h}</math></b>	<b>Mean without peaks</b>	<b>4.8 <math>\mu\text{Sv/h}</math></b>

**Spot measurements.** At the TAS6 spectrometer 5 different configurations were measured, while the TAS3 was closed. Table 2 shows the measured changes. The background levels refer to the levels measured when both TAS3 and TAS6 were closed.

**Table 2.** Background levels, absolute changes and relative changes in dose rate levels when using the TAS 6 spectrometer. Dose rates are  $\gamma$ -dose rates except when followed by: (n) when it is a neutron dose rate.

<b>Place</b>	<b>Background</b>	<b>Absolute increase</b>	<b>Relative increase</b>
<b>Platform</b>	<b>1-2 <math>\mu\text{Sv/h}</math></b>	<b>0-30 <math>\mu\text{Sv/h}</math> 0-5 <math>\mu\text{Sv/h}</math> (n)</b>	<b>0-1700% -</b>
<b>PC cabin 1/4</b>	<b>1-1.5 <math>\mu\text{Sv/h}</math></b>	<b>0-1.5 <math>\mu\text{Sv/h}</math></b>	<b>0-110%</b>
<b>Walk-way close to platform</b>	<b>0.5 <math>\mu\text{Sv/h}</math></b>	<b>1.5-4 <math>\mu\text{Sv/h}</math></b>	<b>300-800%</b>
<b>Walk-way further away from platform</b>	<b>1-3 <math>\mu\text{Sv/h}</math></b>	<b>0-0.5 <math>\mu\text{Sv/h}</math></b>	<b>0-40%</b>

## 5. Conclusion

A good consistency was found between the routine monitoring programme and the results from the special survey. All measured dose rates were in accordance with what is to be expected from the area classification. Although no major surprises were revealed the special survey has given a more detailed knowledge of the radiation fields in the reactor hall both regarding the spatial variation of the average dose rate and the time variation at specific places. This information will enable a more specific instruction of the visiting scientists and the reactor staff. The influence of the neutron guide and the neutron spectrometers has been quantified and will be helpful in planning and evaluating future modifications.

## Kartlegging av radioaktivt nedfall ved hjelp av ull fra sau.

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**Sammendrag:** – Det radioaktive nedfallet fra Tsjernobylulykken hadde virkning for utmarksbeitet i Røros kommune. Høsten 1986 tvilte saubøndene i Røros på at sau fra kommunen kunne leveres til slakting uten noen restriksjoner. Hele Røros kommune ble av landbruksmyndighetene erklært som "frisone", etter at kjøtt fra en sau og ett lam fra den sørlige delen av kommunen var blitt analysert med hensyn på total mengde radiocesium.

Saubøndene forsøkte å finne en enkel måte for å skaffe seg oversikt over hvordan situasjonen egentlig var. I 1986 og de to årene etter ble ullprøver fra samtlige beiteområder i kommunen analysert på total mengde radiocesium. Gjennom dette arbeidet kom en fram til en relativ kartlegging av nedfallsmønsteret i kommunen.

### Innledning

Røros kommune er ca. 2000 km<sup>2</sup> og ligger i sin helhet over 615 moh. Det høyestliggende saubruket ligger på 830 moh., og sauene beiter stort sett mellom 700 og 900 moh. på et areale som er ca. 1000 km<sup>2</sup>. Geologiske og klimatiske forhold varierer.

Prøver av fisk fra ulike deler av kommunen sommeren 1986, viste ved analyse av radiocesium, store variasjoner. Denne situasjonen skapte usikkerhet blant mange saueiere i distriktet. Analyse av et lam lengst nordvest i kommunen viste 630 Bq/kg allerede i juni, mens beslutningen fra landbruksmyndighetene om at Røros var "frisone", dvs. kunne levere slakt uten noen form for restriksjoner, var fattet ut fra analyse av radiocesium i en sau og ett lam i den sørvestlige delen av kommunen i august. Var hele kommunen likt rammet, og hvor lenge ville det eventuelt bli virkninger av nedfallet framover? Dette var spørsmål saueierne stilte seg og ansvarlige. Fantes det noen metode til enkelt å skaffe oversikt over forholdene? Det kom ikke på tale fra slakterienes side å analysere kjøtt fra hver enkelt av de ulike buskapene i kommunen.

Ideen om å benytte ull som testgrunnlag for innhold av radioaktive stoffer i sauenes beiteplanter, ble lansert. Det ble søkt, men ikke oppnådd forskningsmidler til en slik undersøkelse. Røros sau- og geitavlslag skaffet selv tilveie midler for å gjennomføre en undersøkelse i eget beitedistrikt.

### Metode

I Røros kommune var det 90 gårder som drev med sau i 1986. 30 av disse ble plukket ut til å være med i kartleggingsprosjektet. Med disse buskapene fikk en god dekning av de ulike beiteområdene. I utgangspunktet hadde en tenkt seg å bare velge sau av en rase, for å ha mest mulig lik ull, men det var umulig å få dekket alle områdene om en bare skulle bruke en sauerase. Derfor er både dalasau og spælsau med i undersøkelsen.

Etter høstklipping, der hver enkelt fell skilles ved pakking, samles all ull i Røros til ullstasjonen på Tynset. En person fra sau- og geitavlslaget er med på bilen ved henting av ull fra den enkelte gård. Nummererte etiketter ble festet til



inntakssedlene til de 30 utvalgte leverandørene. Ullklassifisørene ved Hed-Opp Tynset tok ut ca. 100 - 200 g ull av første klasse fra siden av fellen fra 3 tilfeldige feller fra hver av de 30 buskapene. Prøvene ble lagt i ferdigmerkede plastposer og sendt til analyse.

## Resultat og diskusjon

RadiOs miljølaboratorium as, Os i Østerdalen (nå Miljølaboratoriet) analyserte prøvene på sum cesium. Første året med 3" Nal-detektor, de to påfølgende årene med 10" Nal brønn-detektor.

Resultatene for første år er gitt i fig. 1. Konsentrasjonen varierte mellom 200 og 2000 Bq/kg, med middelerverdi ca. 600 (Aritmetisk middelerverdi 650 Bq/kg, medianverdi 550 Bq/kg). I de fleste buskapene er det godt samsvar mellom de tre prøvene. I minst tre tilfeller har en åpenbart tilfeldig feil ved at ullprøver fra væreren er kommet med. Dette gjelder område 2 der væreren kommer fra Rissa ("gråsons"), område 9 med vær fra Ålen ("kvitsone"), og område 23 der væreren har beitet på innmark og delvis stått inne i beitesesongen.

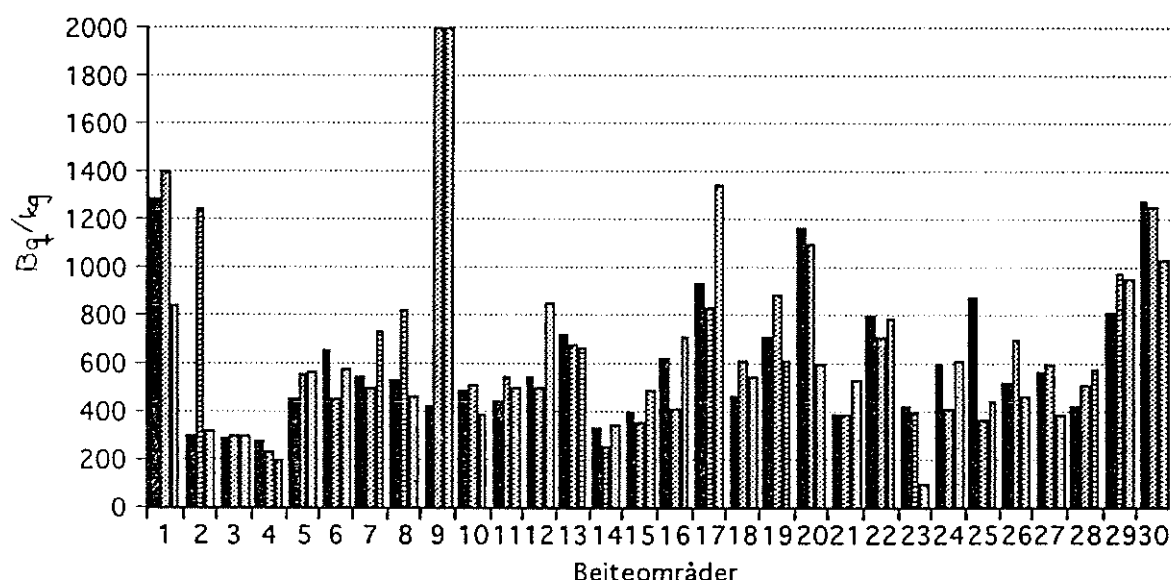


Fig.1. Radioesium i 3 ullprøver fra hvert av de 30 beiteområdene i Røros kommune, 1986.

Tsjernobylnedfallet kom mens det ennå var snø og sauene sto inne i Røros kommune. Sauene blir klippet omkring 1,5 måned før de ca. 1 juni slippes ut på beite. Sauen har svært lite ull når den slippes ut. Sauen går vanligvis 1 - 2 uker på innmark før de slippes i utmark. De sankes fra fjellbeite ca. 15. september, og beiter normalt på innmark til ca. 15. oktober. Sauen klippes ved innsett. Beite er gras og lyngmark, og ulla inneholder svært lite jordpartikler. På siden der ullprøvene ble tatt ut, renner regn lett av i overflata, og det er lite utvasking av ulla i løpet av sommeren.

Mulige feilkilder kan være forskjellig tidspunkt for klipping vår og høst i de ulike buskapene. Tidspunkt for slipp om våren varierer lite, mens sanketidspunkt kan variere noe mer. I forhold til hele beitesesongen vil dette likevel ikke utgjøre mye i veksten på ulla. Materialet inneholder to sauraser, dalasau og spælsau. Dette kan ha betydning innad i det enkelte beiteområdet. Dalasauen beiter ikke i så stor grad i flokk som spælsau.

Beitekartet i fig. 2 viser plassering av de ulike beiteområdene i Røros kommune. Resultatene fra fig. 1 er gruppert i 5 ulike klasser: under 400 Bq/kg,

400 - 600 Bq/kg, 600 - 800 Bq/kg, 800 - 1200 Bq/kg og mer enn 1200 Bq/kg og plottet inn i kartet. Områder som ligger inntil hverandre har tildels svært ulike ullverdier. Snøforhold, vindretning og nedbørsforhold da nedfallet kom, sammen med grunnforhold kan være med å forklare forskjellene mellom de ulike beiteområdene.

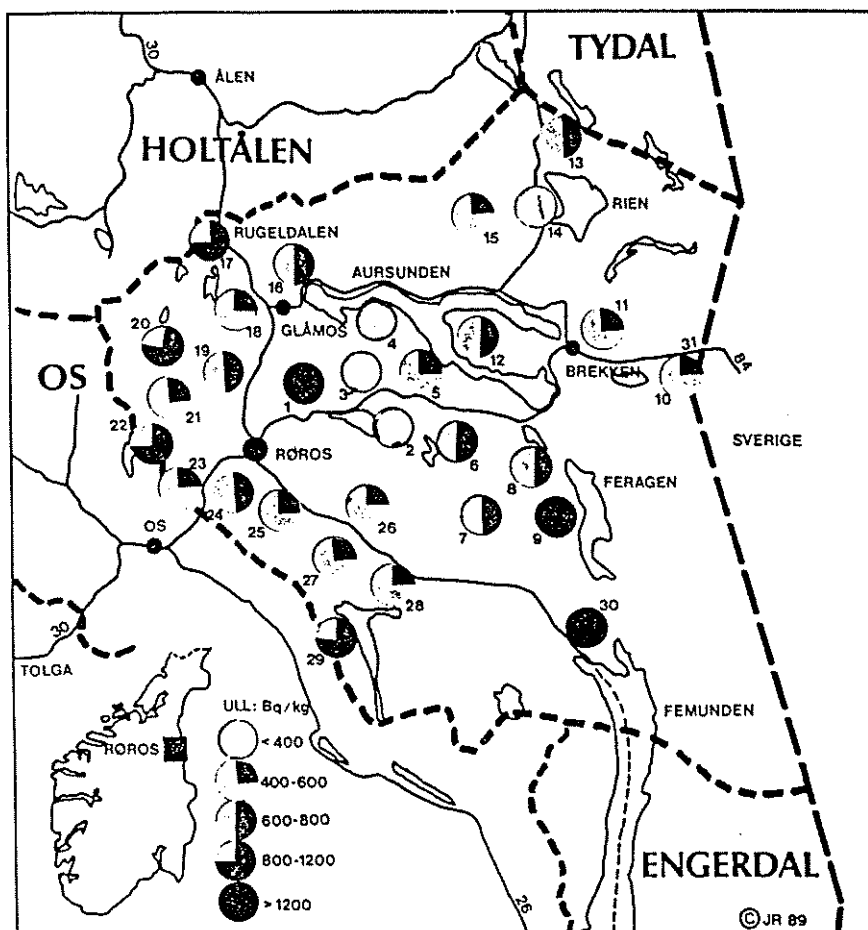


Fig. 2. Beitekart som viser fordeling av radiocesium i ull i de ulike beiteområdene i Røros kommune 1986.

Lammet som er nevnt innledningsvis med 630 Bq/kg kjøtt, kom fra område 17 som har en gjennomsnittsverdi på 1000 Bq/kg ull. Et lam fra den samme buskapen døde like etter sanking. Kjøtt viste 600 Bq/kg mens ull viste 1270 Bq/kg. Et hjemmeslaktet lam fra område 30 viste 280 Bq/kg kjøtt, men hadde da gått noen uker på hjemmebeite slik at en må regne med at radiocesium i kjøttet var vesentlig redusert. Ull fra denne buskapen hadde gjennomsnitt på 1200 Bq/kg. To andre lam som ble slaktet og analysert for sum cesium i kjøtt av næringsmiddeltilsynet viste henholdsvis 230 og 140 Bq/kg. Disse kom fra områder med ullverdi 500 og 400 Bq/kg. Den offisielle prøveslaktninga for Røros ble foretatt i et område som hadde 300 Bq/kg ull. Kjøttverdien var her 130 Bq/kg.

Ullprøver ble tatt ut på samme måte i 1987. Verdiene sank ca. 40 %. Dette var samme tendensen som ellers i Norge. Alle beiteområdene i Rørø ble erklært for friområder. 1988 var et godt soppår, og konsentrasjonene av radiocesium i kjøtt økte. Ullanalysene viste verdier litt i overkant av verdiene fra 1986. Resultatene fra de tre årene er vist i fig. 3.

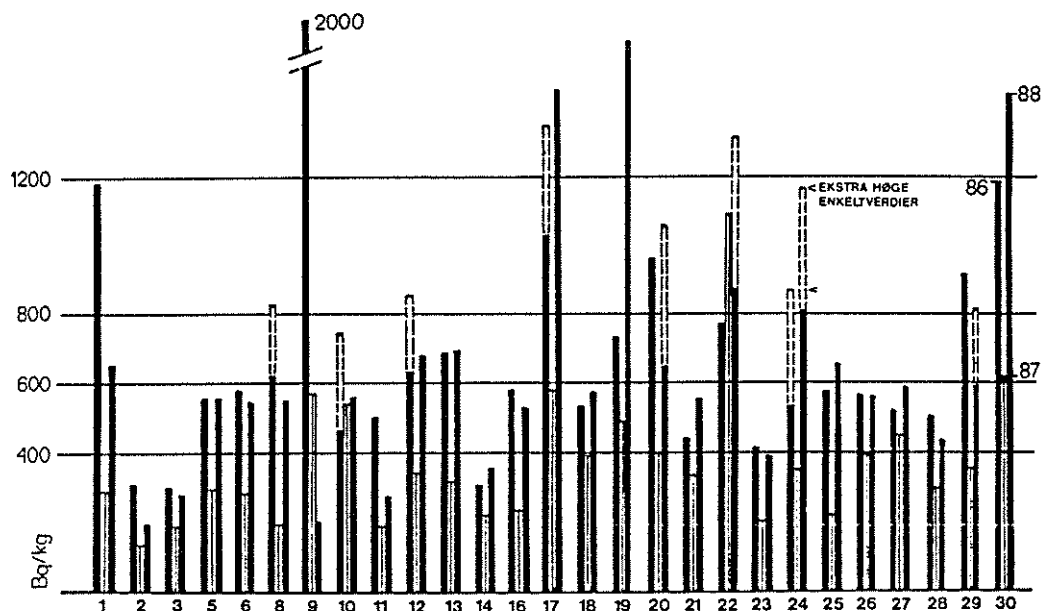


Fig. 3. Gjennomsnittsverdier av radiocesium i tre ullprøver fra hvert beiteområde for de tre årene 1986, 1987 og 1988.

Område 9 viser en dramatisk nedgang, men dette skyldes at saueholdet var under avvikling, buskapen gikk mye på innmark i 1987 og i 1988 ble sauen tatt tidlig hjem pga. ulvens herjinger i flokken.

Det ble fra myndighetenes side foretatt levendedyrsmålinger i september 1988 i noen buskaper også i Rørø kommune. Dette resulterte i 4 ukers nedfôring for område 17, resten av beiteområdene fikk 2 ukers nedfôring.

### Konklusjon

Ved hjelp av ull kan en foreta en relativ kartlegging av radioaktivt innhold i beiteplanter i et større beiteområde for sau. Det er en lite kostnadskrevende undersøkelse, og kan utføres enkelt når forholdene ligger slik tilrette som for ullhåndtering i Rørø kommune. Det er ingen enkel sammenheng mellom innhold av radiocesium i kjøtt og i ull. De analysene som er foretatt i dette materiale kan tyde på en ullverdi omkring dobbelt av kjøttverdi, men det vil kreve mer omfattende undersøkelser med nøyere registrering av bl.a. sanketidspunkt i forhold til slaktetidspunkt for å få bedre grunnlag til å si noe mer om dette.

Takk: – Forfatteren vil takke formann i Rørø sau- og geitavlslag Jon Ryen for utmerket samarbeid i planlegging og gjennomføring av undersøkelsen og Øystein Emanuelson for viktig arbeid i laboratoriet. Norsk sau- og geitavlslag, Landsrådet for organisert beitebruk, Hed-Opp, Rørø slakteri, Rørø sau- og geitavlslag og lokale sankelag har bidratt økonomisk til at undersøkelsen kunne gjennomføres.

# The inhabitants of Lund and Gävle and the legacy of Chernobyl a preliminary study

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## Abstract

*The body burden of  $^{137}\text{Cs}$  in the population of two Swedish cities, Lund and Gävle has been calculated from measurements of the activity concentration in the sewage sludge from sewage treatment plants serving these cities. Now 13 years after the Chernobyl accident the activity in the run-off water during the winter season, January to March, and during dry periods is very low. Already 1996, 10 years after the accident, the distribution of the mean value of the activity concentration of  $^{137}\text{Cs}$  has leveled out, and does not depend on the amount of precipitation. Therefore almost all  $^{137}\text{Cs}$  activity can be determined as coming from the inhabitants of the two cities. The number of inhabitants and the distribution of men, women and children are well known for both Lund and Gävle. It is therefore possible to calculate a mean value for the body-burden of  $^{137}\text{Cs}$  for Lund and Gävle. These values are in good agreement with values obtained from whole-body measurements.*

## Introduction

Most municipal sewage systems have the dual purpose of transporting waste water from households, industries and hospitals and of transporting run-off water after various forms of precipitation. It has been shown previously that sewage sludge is a very sensitive indicator for radioactive material released from hospitals (Erlandsson and Mattsson, 1978) and also for radionuclides spread via the atmosphere (Mattsson *et al.*, 1979). Radionuclides released into the atmosphere after nuclear weapons tests or from nuclear power stations both during normal operation and in connection with accidents are dry or wet deposited on the ground and buildings or washed out from the air by precipitation. The mean residence time of sludge in the plant depends on the type of plant, and varies normally from 2-3 days to 3-4 weeks. Much longer residence times are obtained when radionuclides are deposited on the ground and then washed off into the sewage system by precipitation.

After the Chernobyl accident Gävle, some 150 km north of Stockholm in Sweden was the most heavily inflicted town in Sweden and the deposition of  $^{137}\text{Cs}$  was 20-100 kBq/m<sup>2</sup>. At Lund in southern Sweden the deposition was only 2-3 kBq/m<sup>2</sup>. Now 13 years after the accident the activity in the run-off water during the winter season, January to March, and during dry periods is very low.

Already in 1996, 10 years after the Chernobyl accident the distribution of the mean value of the activity concentration of  $^{137}\text{Cs}$  for the winter months January – March has leveled out, and depends only to a smaller extent on the amount of precipitation. Therefore almost all activity can be considered as coming from the inhabitants. The aim of this investigation therefore has been to measure the activity that each day is leaving the inhabitants of Gävle and Lund and to calculate the body burden.

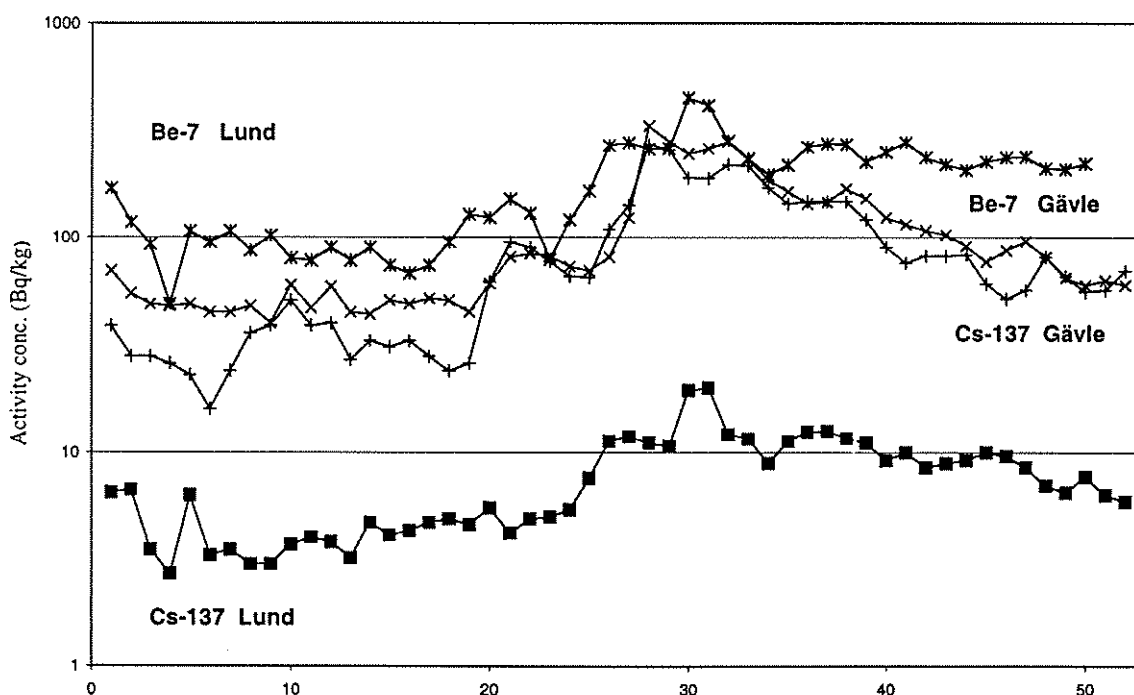
## Materials and methods

Samples of outgoing water and outgoing sludge were collected at the sewage treatment plants at Gävle and Lund. After filtering the water was passed through anion and cation exchangers,

and the resins were then dried and put into 60 or 180 ml tubs. The sewage sludge samples were dried at 105 °C. The dried samples were analysed with Ge(Li) and HpGe detectors and the activity concentration of  $^7\text{Be}$ ,  $^{137}\text{Cs}$ ,  $^{228}\text{Ac}$  and  $^{40}\text{K}$  were measured. Weekly samples of sewage sludge have been collected at Lund from 1983 and at Gävle from 1991. Simultaneous samples of sludge and outgoing water have been collected at both Gävle and Lund. The ratio between the activity in the sludge and outgoing water and sludge is 2.0 for Lund and 1.32 for Gävle (Erlandsson, *et al.*, 1989)

## Results

Just after the accident the activity concentration was about 200 kBq/kg dry weight (d.w.) in the sludge from Gävle and 1 kBq/kg d.w. in the sludge from Lund. Since the accident there has been a steady decrease, but with yearly variations at both Gävle and Lund. In Fig. 1 is shown the variations during 1997. After a period with small variations at the beginning of the year there is a peak at the end of the growth period at the end of the summer. This peak is getting smaller and smaller with each year. During the summer more of the  $^{137}\text{Cs}$  reservoir is put into circulation and can be washed off to the treatment plant. The variations of  $^7\text{Be}$  is also shown in Fig 1.

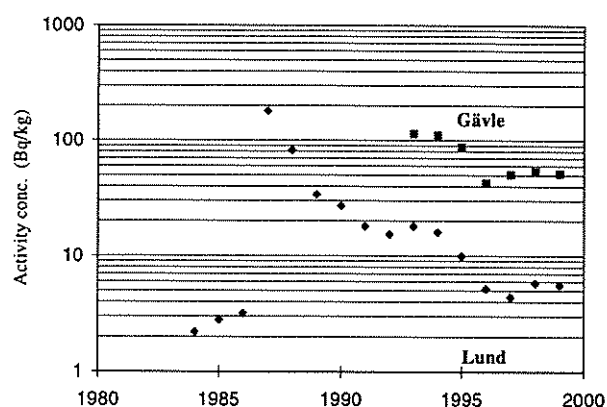


**Fig. 1.** Activity concentration of  $^7\text{Be}$  and  $^{137}\text{Cs}$  in sewage sludge from Lund and Gävle.

The two curves for Lund and Gävle are following each other and the difference is rather small. For  $^{137}\text{Cs}$  the difference is about a factor 20.

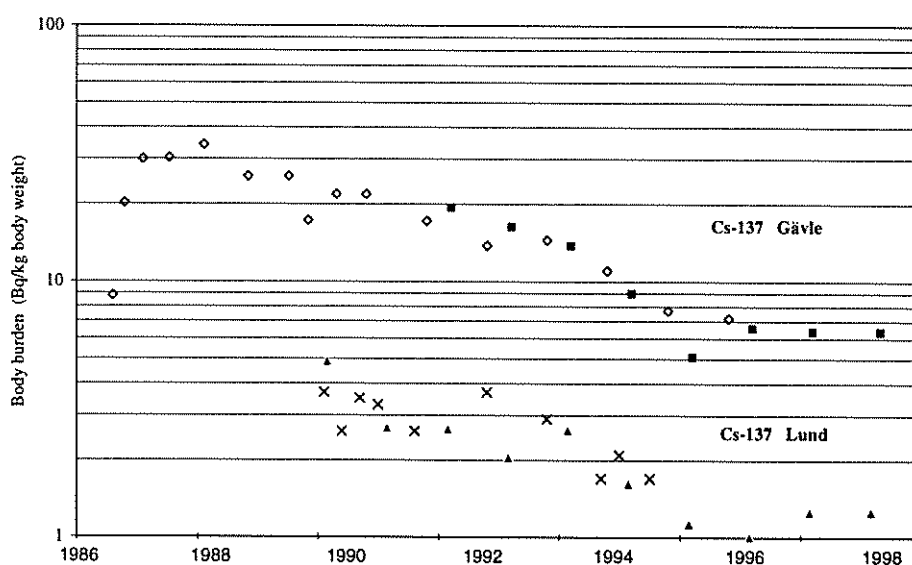
In Fig 2 the mean value of the activity concentration for  $^{137}\text{Cs}$  for the three winter months January, February and March is shown for both Lund and Gävle. For Lund there is a rapid decrease during the first three years after the accident with a half-time of 0.8 y and then a slower decrease with a half-time of 4 y. For both Lund and Gävle there is no decrease for the last 4 years from 1996 to 1999.

One can assume that most of the outgoing  $^{137}\text{C}$  activity during the winter months as far back as 1990 for Lund and 1992 for Gävle, is coming from the inhabitants.



**Fig. 2.** Mean value of the activity concentration for January, February and March for Lund and Gävle.

The total outgoing  $^{137}\text{Cs}$  in both water and sludge per day and person can be calculated and assuming a mean weight of 70 kg and a biological half-time of 110 days the body burden (Bq per kg body weight) can be calculated. The calculated body burden for the citizens of Lund and Gävle is shown in Fig 3.



**Fig. 3.** The Body burden in Bq per kg body weight. Black squares: calculated for Gävle, Black triangles: calculated for Lund, Open squares: whole-body concentration for "non farmers" Gävle, Crosses: whole-body concentration for SSI reference group.

The Lund citizens have been compared with the SSI reference group and the Gävle inhabitants with a reference group marked "non farmers" from Gävle (Nyblom *et al.*, 1998). Where there is an overlap the agreement is rather good for both Lund and Gävle. The biological half-time for the inhabitants of Lund is  $3.4 \pm 0.5$  y which is in good agreement with 4 y for the General Swedish population (Nyblom *et al.*, 1998). For Gävle the decrease is more complicated and

the half-time is  $3.8 \pm 0.5$  y for 1992 to 1994. Between 1996 and 1999 the body burden half-time is very long for both Lund and Gävle.

### Acknowledgements

The author is in great debt to the staff of the sewage treatment plants at Källby in Lund and especially to Ms Carin Eklund at Duvbacken in Gävle and also to Prof. S. Mattsson for many valuable discussions.

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# Cs-137 contamination of trees in the Briansk region

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## Sampling and Analysis

In 1997 a sampling programme was planned, aimed at obtaining a preliminary picture of the contamination situation in the forested areas around Novozybkov. Three areas were selected for sampling. One near each of the two settlements, Guta Muravinka and Novo Bobovich, where decontamination work was carried out, and a third area close to the village of Zaborie. This village received a very high level of contamination from the Chernobyl accident – about 4 MBq m<sup>-2</sup> on average. In Zaborie both trees planted before and after the Chernobyl accident were sampled. The ‘new’ trees were planted in an area that had been deep ploughed. This area was only about 600 metres from the ‘old’ forest where a pine was also sampled. The soil was in all sampling areas sandy with a thin organic horizon.

The result of soil sampling in the four locations is shown in Table 1. Four soil samples were taken at each location and analysed in the laboratory. In two locations only two soil profiles have been analysed so far. It can be seen that the variability of the results is very high in the area in Zaborie where there had been deep ploughed, whereas it is low in the three other locations where we have undisturbed soil profiles. The locations with undisturbed soil had very similar mean attenuation depths, about 1.7 g cm<sup>-2</sup>, with low variability. Overall, the surface contamination was three times higher in Zaborie than in the two other places and the attenuation depth was three times greater in the place in Zaborie that had been deep ploughed.

*Table 1 Review of analyses of soil samples taken at the wood sample sites.*

Location	Number of samples analysed	Mean surface contamination [MBq m <sup>-2</sup> ]	Mean depth [mm]	Mean attenuation [g cm <sup>-2</sup> ]
Zaborie Young	4	4.2 ± 2.2	48 ± 12	5.4 ± 1.1
Zaborie Old	2	2.95 ± 0.06	35 ± 3	1.8 ± 0.6
Novo Bobovich	2	1.041 ± 0.004	38 ± 2	1.7 ± 0.2
Guta Muravinka	4	1.2 ± 0.3	27 ± 8	1.6 ± 0.3

Pine, Birch and Oak trees were sampled. In total, seven trees were cut, as presented in Table 2. As pine is the most common tree in the Russian forests special focus was put on this species. A mature pine tree was sampled from all locations as well as a younger pine tree from one site. Birch is also very common and was sampled in two locations. Oak was only found in Guta Muravinka.

Sections of the trunks of the trees were sampled at a height of 1 metre above ground. For the pine tree additional pieces of the trunk were sampled at intervals corresponding to 5 years' growth in Zaborie and Novo Bobovich and with 2 year growth intervals in Guta Muravinka. Branches were sampled with leaves or needles. On location, the leaves and needles were separated from the twigs. For the pine trees the needles and branches were divided according to the year of growth, which can easily be identified on a pine tree. After returning to the laboratory the bio-material samples were dried to constant weight at 80 °C, except for the trunk sections, which were dried at



room temperature. The bio-material samples were homogenised and measured by gamma spectrometry in standard containers, and the results were expressed in relation to the dry mass.

The piece from 1 metre's height above ground and the 5 year old piece from the top of the tree were cut into discs and separated according to year rings.

A subset of pine samples from Zaborie and Guta Muravinka were selected for neutron activation analysis. Samples were sealed in plastic containers and irradiated for 2 hours at a flux of  $4 \times 10^{15} \text{ n m}^{-2} \text{ s}^{-1}$ . After a few weeks the Cs-134 signal was counted for 1 – 2 hours on a 7 % Ge(Li) detector achieving good counting statistics for all samples. The stable cesium content was determined from the Cs-134 count rate. Potassium content was also measured from the K-42 isotope. This gave a better determination than measuring K-40 that gave a relative weak peak.

Table 2. Trees sampled during the 1997 campaign

Location	Description	Species	Sample description
Zaborie	Young birch	<i>Betula pubescens</i>	10 year old and 3.5 meter high birch tree.
Zaborie	Young pine	<i>Pinus sylvestris</i>	10 year old
Zaborie	Old pine	<i>Pinus sylvestris</i>	~ 45 year old 15 meter high pine split in two at 5 metres' height
Novo Bobovich	Old pine	<i>Pinus sylvestris</i>	21 year old pine tree. 21 m high and 76 cm Ø at 1 m height
Guta Muravinka	Birch	<i>Betula pubescens</i>	Birch
Guta Muravinka	Oak	<i>Quercus robur</i>	13 year old Oak tree
Guta Muravinka	Pine	<i>Pinus sylvestris</i>	Pine

## Results and Discussion

The specific activities for eighth categories of samples are presented in Table 3. The corresponding transfer factors, TF, were calculated by dividing the specific activities by the contamination level in Table 2. The TFs are presented in 4. In the tree trunks, the specific activity was generally found to be highest in the cambium parts. For pine trees the specific activity was highest for fresh needles and twigs, followed by 2 and 3 year old needles and twigs, roots, bark and finally wood. In Zaborie the specific activity in the old pine was about 10 times higher than that in the young pine tree. This could be indicative of two phenomena: one is that the surface contamination on the tree is absorbed into the old pine tree, and the other is that uptake from a ploughed soil is considerably smaller than that from a virgin soil.

Table 3. Specific activities in different parts of the sampled species. The results are for mixed samples including bark for the wood and twig samples. The 'New needles' column refers to needles or leaves from the current year, whereas '3 year needles' refers to needles from 1995. Similarly the 1 and 3 year twig columns refer to twigs from the current year and from 1995.

Sample	Root	Wood 1 m.	Wood 5 y.	New needles	3 year needles	1 year twigs	3 year twigs	Bark 1 m.
	[Bq g <sup>-1</sup> ]	[Bq g <sup>-1</sup> ]	[Bq g <sup>-1</sup> ]	[Bq g <sup>-1</sup> ]	[Bq g <sup>-1</sup> ]	[Bq g <sup>-1</sup> ]	[Bq g <sup>-1</sup> ]	[Bq g <sup>-1</sup> ]
Pine Za-old	7.7	5.7	4.9	56.5	12.4	58.7	14.6	6.78
Pine Za- yng	1.76	0.31	0.41	4.9	-	4.64	-	1.46
Pine NB	0.40	0.78	0.23	1.17	0.25	1.17	0.38	-
Pine GM	-	0.81	0.82	9.1	2.01	5.69	1.48	-
Birch Za	-	0.044	0.052	0.25	-	0.176	-	0.089
Birch GM	-	1.11	0.83	2.6	-	1.13	0.67	-
Oak GM	-	1.83	-	12.9	-	3.13	2.71	9.39

Table 4  $^{137}\text{Cs}$  transfer factors from soil to wood matter. The specific activities shown in Table 7.3 were divided by the surface contamination obtained from analysis of soil samples. The TF's are expressed in units of  $10^3 \text{ m}^2 \text{ kg}^{-1}$ .

Sample	Root	Wood 1 m.	Wood 5 y.	New needles	3 year needles	1 year twigs	3 year twigs	Bark 1 m.
Pine Za-old	2.6	1.95	1.65	19.2	4.2	19.9	4.95	-
Pine Za-yo.	0.60	0.11	0.14	1.7	-	1.57	-	-
Pine NB	0.38	0.75	0.22	1.1	0.24	1.15	0.37	-
Pine GM	-	0.68	0.68	7.6	1.68	4.7	1.23	-
Birch Za	-	0.015	0.018	0.08	-	0.06	-	-
Birch GM	-	-	-	2.1	-	0.94	0.56	-
Oak GM	-	1.53	-	10.8	-	2.6	2.3	-

The birch trees had higher specific activities than had the pine trees, both in Zaborie and in Guta Muravinka. This is in good agreement with the results presented by Tikhomirov *et al.* (1993). However, the other deciduous tree (the oak in Guta Muravinka) had the highest specific activity in the sampled trees from this site.

When comparing the transfer factors for the pine trees from different sites it can be seen that the three older trees have similar TFs,  $0.7 - 2.0 \cdot 10^{-3} \text{ m}^2 \text{ kg}^{-1}$ , whereas the young pine from Zaborie has a considerable smaller TF,  $0.1 \cdot 10^{-3} \text{ m}^2 \text{ kg}^{-1}$ . The observed TFs for pine are in good agreement with the values reported by Belli *et al.* (1996).

The results of the analyses of the seven samples divided into year rings are presented in Figure 1. The results from analyses of sections of the old pine from 1997 back to 1958 show that the  $^{137}\text{Cs}$  is mobile in the tree trunk, as was found by other workers (e.g. Kohno *et al.* (1988) and Momoshima and Bondietti (1994)). No peak can be seen in the year ring corresponding to 1986, where the Chernobyl accident occurred. The activity is almost homogeneously distributed over the year rings, all the way back to the 1958 ring. The concentration seems to decline towards an equilibrium level in the centre of the tree.

The analysis of samples from different heights showed a different picture. Here a clear maximum in specific activity could be found in the samples corresponding to the growth in 1985 and 1986, that is the growing tree parts during the Chernobyl accident in 1986. This seems to indicate, that the vertical transport of caesium inside the stem is slower.

Based on the water content and potassium concentration it was possible to divide the samples analysed with neutron activation analysis into two distinct groups. The needles, cones, and twigs were termed the 'soft wood' samples and the branches and stem pieces were the 'hard wood' samples. The soft wood had dry masses around 39 – 52 % of fresh weight and potassium 3400 to 5400 PPM. The hard wood had dry masses of 88 – 93 % compared to fresh weight and potassium contents of 200 to 2000 PPM.

In Figure 2 the specific caesium activity has been plotted as a function of the stable caesium content. Two distinct lines can be identified and associated with respectively the soft wood and the hard wood samples. The soft wood has much higher activities with regard to the amount of stable caesium than the hard wood. Together with the knowledge from the year ring study presented in Figure 1, that the  $^{137}\text{Cs}$  migrates into the tree, this seems to imply that the Cs-137 activity in hard wood will continue to increase over the next years establishing an equilibrium between  $^{137}\text{Cs}$  and stable caesium.

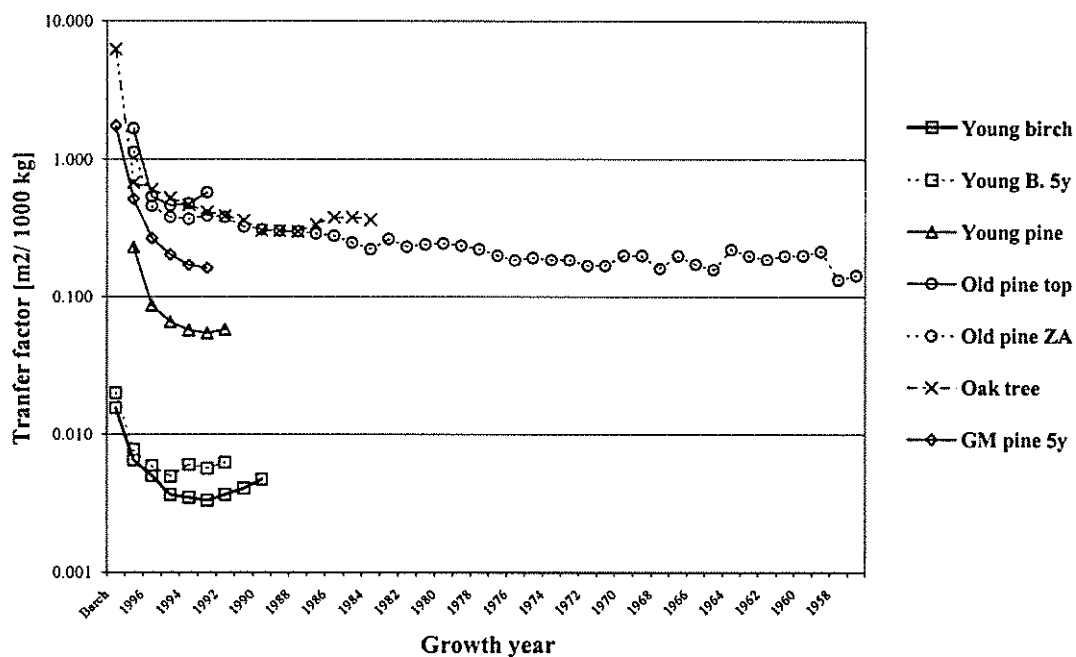


Figure1.  $^{137}\text{Cs}$  transfer factors for the analysed tree samples as a function the growth year.

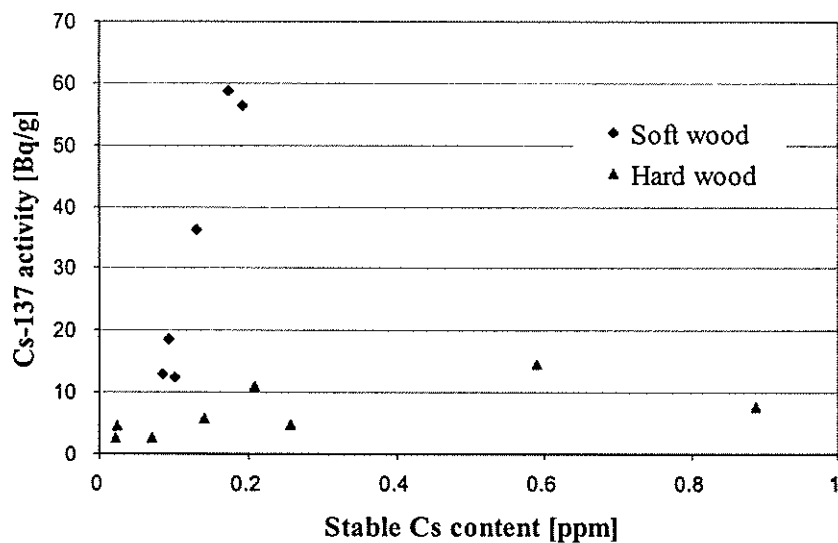


Figure 2 The specific activity plotted as a function of stable cesium content in the plant tissue.

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# LOW EMISSIONS OF RADIONUCLIDES FROM THE STUDSVIK SITE

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**Abstract** – Emission of radionuclides into air and water has been performed since 1959 at the Studsvik site, situated at the Baltic coast in the county of Södermanland, 80 km south of Stockholm. Several small nuclear reactors have been used at the site. Today only one 50 MW high-flux pool-type reactor (R2) and one 1 MW low-power reactor (R2-0) are used. In the R2 Materials Testing Reactor (MTR), fuel ramp testing, studies of high burnup fuel testing, radio-isotope production for medical and industrial purposes and neutron transmutation doping of silicon are performed. Treatment facilities for Boron Neutron Capture Therapy, BNCT, for medical treatments at the R2-0-reactor will be built during 1999. The University of Uppsala has a department of neutron research at the reactors. Other types of nuclear work done at Studsvik are testing and examination of fuel in the Hot Cell Laboratory, decontamination and waste management like incineration of waste from nuclear power plants and hospitals. Also melting and recycling of low-level metallic waste is performed.

## INTRODUCTION

The most dominant source of radionuclides found in the surroundings of Studsvik comes from natural radioactivity, nuclear atmospheric tests performed between 1945 and 1980 and after the Chernobyl accident in 1986. Yearly emissions are, and have always been, much lower than the limit, 100  $\mu\text{Sv}$ , set by the authority in 1981. The calculated effective dose to a hypothetical person in the critical group due to emissions was in 1998 only 0,4  $\mu\text{Sv}$ .

In the middle of the 1980's a new plant for treatment of liquid waste was built. The emission of radionuclides to the Baltic Sea then decreased by a factor of 10. In the beginning of the 1990 also emission to air was reduced, now by a factor of more than 10 when a method for testing filters with  $^{131}\text{I}$  was modified. This year a new two-step precipitation is made before the waste is emitted. The aim for year 2000 is an emission less than 1/1000 of the limit.

Regulatory limits:

1957-1964. Yearly emission limits: less than 7.4 GBq of gross alpha, 1.3 TBq of gross beta (and less than 0.55 TBq of Ce, Y and rare earth isotopes and less than 90 GBq of Sr)

1965-1980. Changed emission limits for beta: 222 TBq  $^3\text{H}$  and 1.2 TBq gross beta activity (instead of 1.3 TBq)

1981-. New emission limits: 0.1 mSv to individuals in the critical group

A critical group is defined as a hypothetical group of people receiving a higher dose than others from all relevant major exposure pathways and for the total nuclear site due to habits, age and place of living. The exposure to nuclides from external radiation, ingestion and inhalation is translated into a dose in mSv with the help of a site-specific model. In order to take into account the possibility of receiving doses from different sources, the Swedish authority has specified a dose constraint, which is one-tenth of the ICRP recommended limit of 1 mSv/year to members of the general public, i.e. 0.1 mSv/year from a nuclear site. The total radiation dose in mSv/year to the critical group is calculated from the measured actual emission of all relevant radionuclides to air and water by using a transport dispersion model and ICRP dose conversion factors between radioactivity (in Bq) and the effective dose (in Sv) to the critical group.

## METHODS

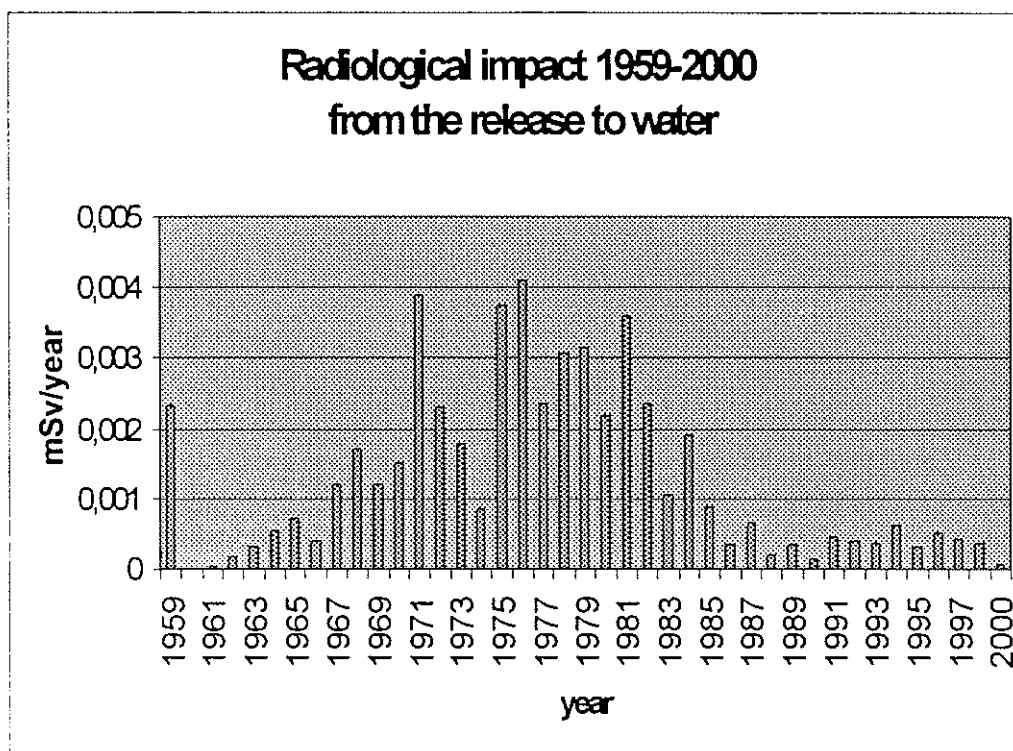
Of the total amount of activity that is supplied to the Baltic Sea from Studsvik and translated into a dose to a critical group,  $^{60}\text{Co}$  comprises a few per cent of the dose,  $^{137}\text{Cs}$  less than approximately 30 per cent and  $^{90}\text{Sr}$  less than approximately 40 per cent. This distribution will now be altered.

A new two-step precipitation method introduced in 1999 will decrease the emission of all nuclides in general. The precipitation is mixed with concrete to make the waste solid instead of liquid. The waste is then placed in the repository for radioactive operational waste at Forsmark (SFR). High level waste is stored at Studsvik and will later be placed in a future repository in Sweden (SFL).

The emission of  $^{90}\text{Sr}$  will also decrease by changing the handling method of the waste. Instead of chemical separation methods with the liquid waste the slurry is now mixed with concrete and stored for the coming SFL repository. The treated waste is a by-product from burnup investigations of nuclear fuel materials.

## EMISSION RESULTS

Figure 1 shows the radiological impact since 1959. Efforts were made in the 1970's to minimise emissions. In 1985 the treatment facility was modernised and in 1999 the precipitation is a two-step method. The aim for year 2000 is an emission less than 1/1000 of the limit.



**Fig. 1.** The dose in mSv to the critical group calculated from water emissions in Bq to the Baltic Sea between 1959 and 1998. The value for 2000 is an estimate.



## A long-term study of $^{137}\text{Cs}$ distribution in soil

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### Abstract

*During a period of ten years, soil samples have been collected at the investigated site and analysed for  $^{137}\text{Cs}$  from the Chernobyl accident and from the nuclear weapons tests. The distance between the sampling plots at the different sampling occasions has been no more than three metres. The results show that the depth distribution of  $^{137}\text{Cs}$  is practically the same from year to year, indicating that the migration of caesium at this site is very small. The total activity measured in the soil samples have also been compared with the deposition calculated from the amount of precipitation at the site and the results of this comparison shows a good agreement with measured soil caesium. In this calculation is used the previously determined specific activity of  $^{137}\text{Cs}$  in the precipitation from nuclear weapons tests and from the Chernobyl accident, based on air activity concentration and the amount of precipitation. For the period 1962 to 1986 the calculated activity of  $^{137}\text{Cs}$  amounts to  $1.41 \text{ kBq/m}^2$  (the decay of all activities has been corrected to 1986) which is to be compared with the measured activity of nuclear weapons fallout in the soil samples,  $1.06 \text{ kBq/m}^2$  in 1988 and  $1.60 \text{ kBq/m}^2$  as a mean value of the samples of the first four years. The calculated activity of  $^{137}\text{Cs}$  of Chernobyl origin is  $0.787 \text{ kBq/m}^2$  and the measured activity in the soil samples is  $0.789 \text{ kBq/m}^2$ . For the total activity the mean value of all soil samples give the result  $2.28 \text{ kBq/m}^2$  which is in good agreement with the calculated  $2.20 \text{ kBq/m}^2$ .*

### Introduction

The dominant radionuclide in the upper parts of the soil in southern Sweden is, besides the naturally occurring  $^{40}\text{K}$ ,  $^{137}\text{Cs}$  which mainly has been deposited after the nuclear weapons tests, performed in the atmosphere mainly in 1956-58 and 1961-62, and after the Chernobyl accident in 1986. Minor depositions occurred also in the seventies after nuclear weapons tests in central Asia.

The deposited  $^{137}\text{Cs}$  has, since the time of deposition, migrated down into the ground to an extent which depends on soil parameters and vegetation, as well as, on the mode of deposition: the old (bomb) caesium was deposited during a long period of time with the main peak in 1963 (DeGeer et al, 1978) whilst the Chernobyl caesium was basically deposited at a single event.

The aim of this work is to study the depth-distribution and cumulative activity of  $^{137}\text{Cs}$  in the ground at a given site over a long period of time and to test the agreement between the measured activity in the soil and the calculated deposition.

### Sample collection and preparation

Soil samples were collected on six occasions (January 1988, June 1991, November 1992, October 1994, November 1997 and November 1998) next to a spruce forest at Blentarp ( $55.063^\circ \text{ N } 30.617^\circ \text{ E}$ ) in southern Sweden. The soil down to about 2 cm depth can be characterised as organic-rich silt and the soil from deeper layers as sandy silt. The sampling site has been undisturbed since long before 1986.

When the first sample was collected in 1988, successive layers of soil within an area of  $0.25 \text{ m}^2$  were scraped off down to 0.5 cm and then further down to 2.8 cm depth. The soil sampling area was then reduced to  $0.063 \text{ m}^2$  and the soil down to 5.0, 7.2, 10.0 and 12.7 cm



was collected. At the following five occasions the soil was collected as bore cores with an 8 cm diameter metal tube (Isaksson and Erlandsson, 1995), which was pushed into the ground to a depth of 12-18 cm. Three bore cores were spaced in a triangular pattern with 30 cm between the cores and each core was divided into slices of 2 cm or 3 cm thickness. Slices from corresponding layers in the three cores were then pooled before analysis. All samples at the six sampling occasions were taken from an area of less than 10 m<sup>2</sup>.

The soil samples were dried and analysed for <sup>134</sup>Cs and <sup>137</sup>Cs with either a Ge(Li) detector (efficiency 18% and resolution 1.9 keV (FWHM) at 1332 keV) or a HPGe detector (efficiency 24% and resolution 1.9 keV). The detectors were surrounded by 10 cm thick lead shields. The samples from 1988, 1991 and 1992, initially analysed with the Ge(Li) detector, were all reanalysed in 1994 with the HPGe detector and the results agreed well. All activity measurements were corrected to April 26 1986 (the day of the Chernobyl accident).

## Results and discussion

The results from the analysis of the soil samples, taken at the six occasions, are shown in Figures 1-3. For the separation of the total activity in new and old, a ratio of 0.53 (Isaksson and Erlandsson, 1998b) was used to calculate the activity concentration of new <sup>137</sup>Cs from the activity concentration of <sup>134</sup>Cs, which was measurable in the samples from the first four sampling occasions.

Figures 1b, 2b and 3b show the accumulated <sup>137</sup>Cs activity (Bq/m<sup>2</sup>) vs. mass depth (kg/m<sup>2</sup>), for total, new and old <sup>137</sup>Cs. From Figure 1b it is clearly seen that, with the exception of the years 1988 and 1997, the activity distribution of total <sup>137</sup>Cs is almost the same. The discrepancies for the years 1988 and 1997 could be due to the different sampling method in 1988 or to the fact that even small variations in where the holes have been dug within the sampling area can lead to big variations in the total activity. This is also seen in Figure 3b but not in Figure 2b. An explanation may be that the spot where the samples for 1988 and 1997 have been taken have been covered with some sort of obstacle during a period in the 1960:s.

In Figures 1c, 2c and 3c the relative accumulated activity is shown as a function of the mass depth. It is clear that all the distributions have the same shape. The shape is also almost the same for the old deposition (Figure 3c), but variations in the shape appear for the new deposition. It is also evident that the <sup>137</sup>Cs is very superficially located and seems to have been fixed in the organic-rich silt.

The advantage of the soil core method is reflected by the small deviation in mass-depth. The accumulated mass depth, down to 12 cm depth, varied between 145 and 172 kg/m<sup>2</sup> with a coefficient of variation of 6%. The small variation in mass-depth indicates that the soil can be collected without significant compression and supports the findings from other sites (Isaksson and Erlandsson, 1995).

Since the second quarter of 1962 the activity concentration in air and the deposition of <sup>137</sup>Cs and other radionuclides, originating from nuclear weapons tests in the atmosphere, have been measured at Ljungbyhed, about 60 km from the sampling site at Blentarp (FOA 41, DeGeer et al., 1978). The variations of the air activity with time is rather slow and a comparison with measurements at other places in Sweden has shown that the differences over such a short distance as 60 km ought to be small (Bernström, 1969). Therefore the deposition depends only on the amount of precipitation. The deposition and the precipitation have been measured quarterly at Ljungbyhed and the activity concentration of <sup>137</sup>Cs in the precipitation has been calculated (Isaksson et al., 1999). Based on this, the deposition at Blentarp has been calculated as the mean for two precipitation stations (Ågerup and Sövde), which are only about 5 km from Blentarp and have been in operation for the actual period (Elleson, 1993), and for the period from 1962 to the first quarter of 1986 this deposition is (1411 ± 70) Bq/m<sup>2</sup>, decay corrected to 1986. This agrees well with the amount of bomb <sup>137</sup>Cs found in the soil

measurements, which was  $(1604 \pm 380)$  Bq/m<sup>2</sup> as a mean value of the first four years of sampling.

The deposition and the precipitation after the Chernobyl accident have been measured at Lund for the period 8 May to the end of December 1986 and the activity concentration has been calculated. For this period the precipitation has been measured at the three stations Ågerup, Sövde and Karup, all within 5 km from Blentarp. The mean value of the calculated deposition for Blentarp, corrected for decay to 1986, is  $(787 \pm 116)$  Bq/m<sup>2</sup>, which is in good agreement with the  $(789 \pm 50)$  Bq/m<sup>2</sup> obtained from the first soil measurement in 1988.

The total calculated deposition of <sup>137</sup>Cs at Blentarp is then  $1411 + 787 = (2198 \pm 86)$  Bq/m<sup>2</sup>, which is in good agreement with the mean value for the six soil measurements,  $(2284 \pm 340)$  Bq/m<sup>2</sup>.

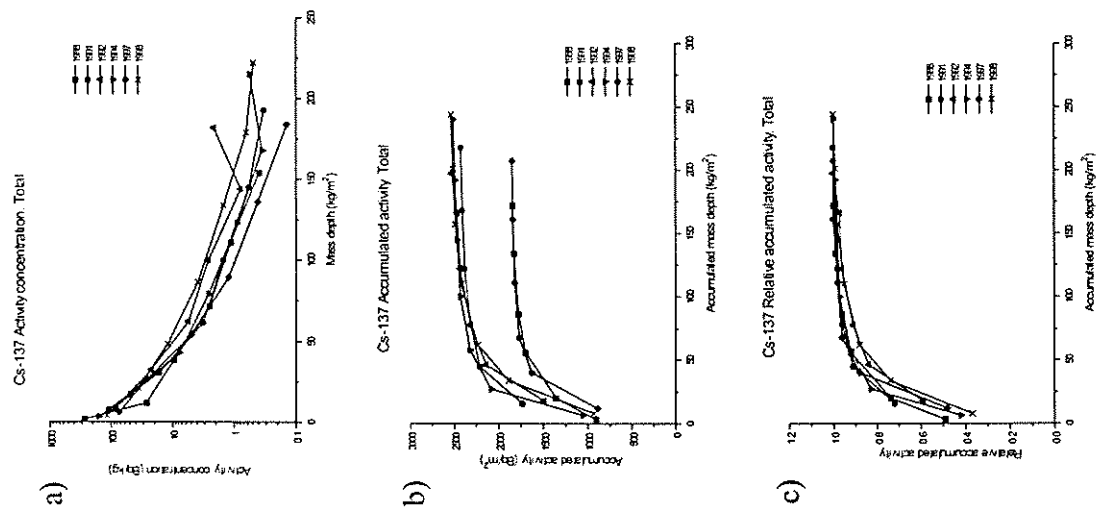
### Summary and conclusions

Soil samples have been taken, and analysed for <sup>137</sup>Cs, at the same site during a period of ten years. The depth-distribution shows great similarities between the years, implying that the caesium, once deposited, have migrated down to a certain depth and became fixed in the soil matrix. The analysis also showed that the method of soil-core sampling gives a good reproducibility concerning variations in mass depth. The variations in total activity is about 15%, which has also been found elsewhere (Isaksson and Erlandsson, 1995; Isaksson and Erlandsson, 1998a) on a variety of lateral scales.

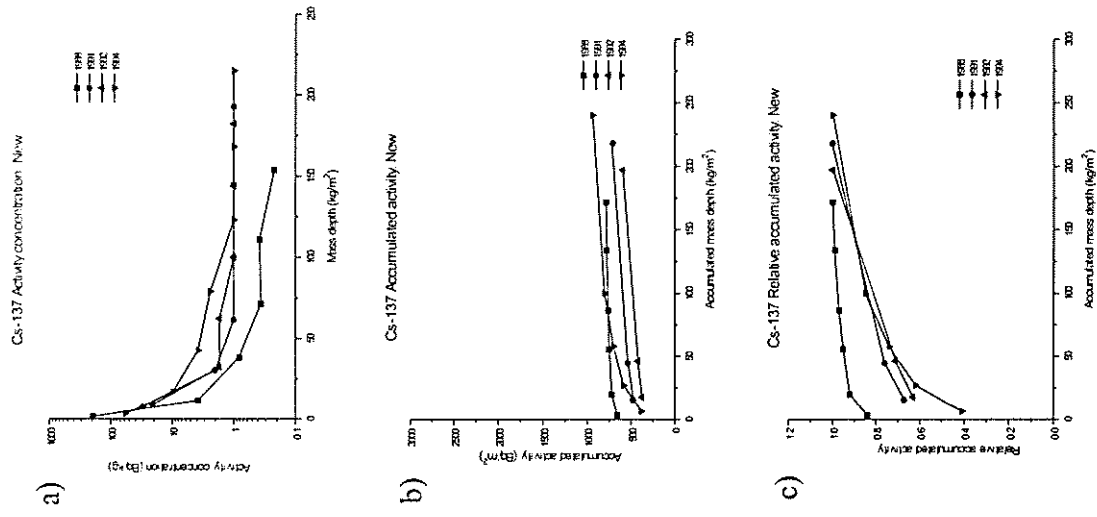
Knowledge of the amount of precipitation at various places and at various times together with a known activity concentration can also be used to calculate the deposition. Within the uncertainty limits, deposition of <sup>137</sup>Cs as calculated from the precipitation gives a good picture of the deposition determined by soil measurements with the three holes in a triangle method.

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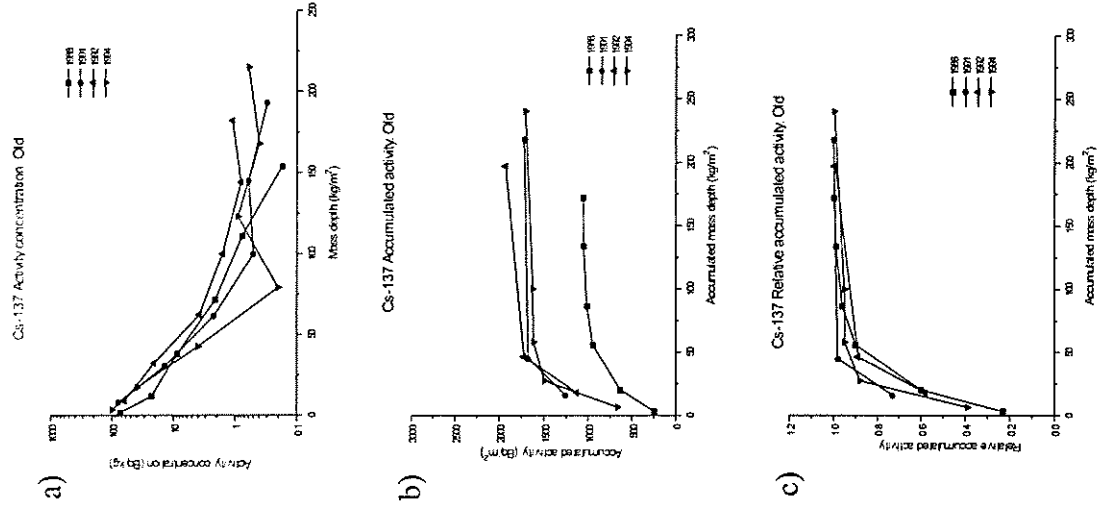
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**Fig. 1.** Activity concentration, accumulated and relative acc. activity for total  $^{137}\text{Cs}$  in soil.



**Fig. 2.** Activity concentration, accumulated and relative acc. activity for new  $^{137}\text{Cs}$  in soil.



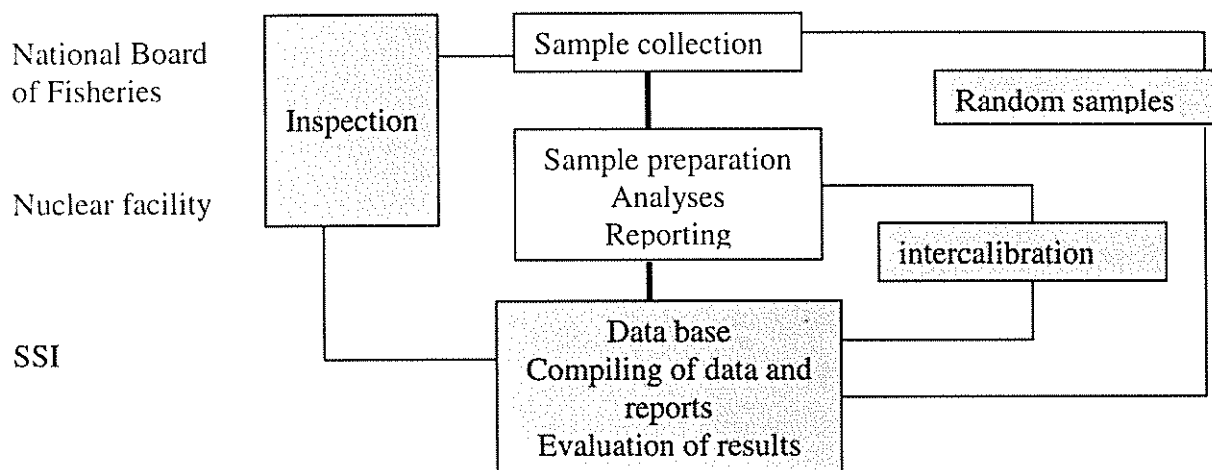
**Fig. 3.** Activity concentration, accumulated and relative acc. activity for old  $^{137}\text{Cs}$  in soil.

## Environmental Monitoring around the Swedish Nuclear Facilities

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### Introduction

The environmental monitoring programme for the Swedish Nuclear Facilities is issued by the Swedish Radiation Protection Institute (SSI) and managed by the facilities. To verify that the facilities comply with the programme, SSI makes annual inspections and also takes random double samples for measurements at the SSI laboratories. The programme is revised annually. A schematic picture of the different parts of the environmental programme for the Swedish nuclear facilities are shown in the figure. SSI's responsibilities are marked in grey.



### Environmental monitoring

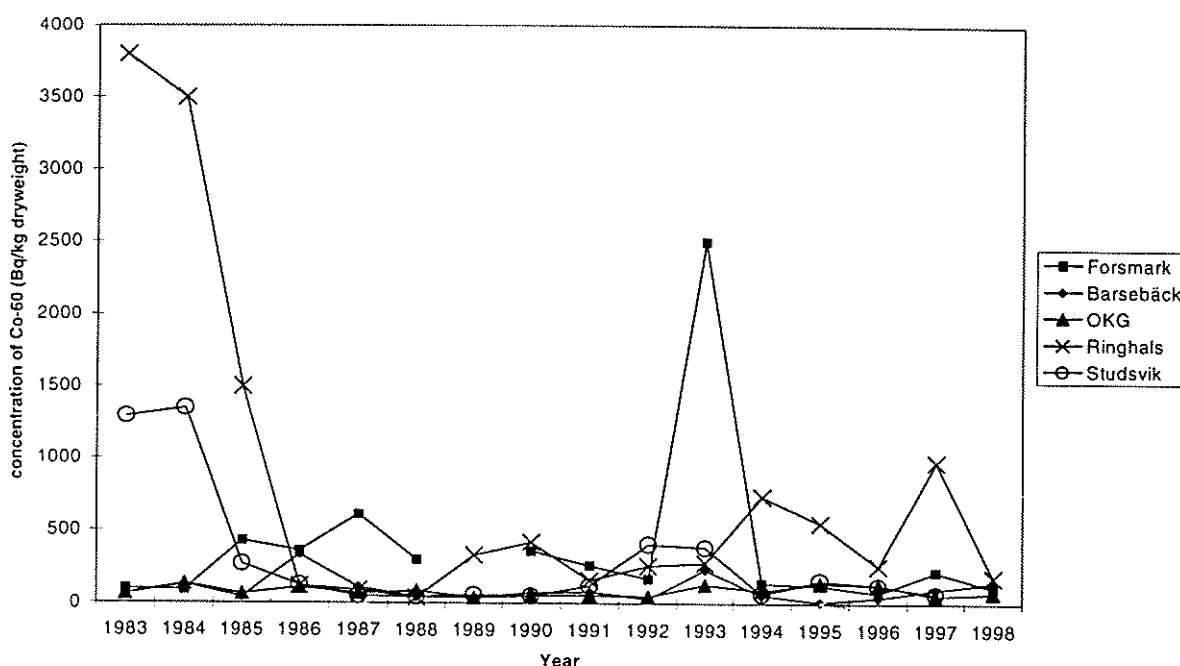
The aims of the programme are

- to detect large unmonitored or unreported discharges from the facilities
- to allow verifications of the model-based dose calculations
- to provide data for reporting to relevant international organisations.

Environmental samples are taken either by the operators or by contracted organisations, mainly the National Board of Fisheries. Radioactivity measurements are performed by the operators themselves or by contracted laboratories. The environmental samples consist of local fauna and flora, and sediments (Appendix 1). There are local variations between the sites as well as temporal changes, but the programme is taking into account both homogeneity and continuity between the sites. A regular programme is performed in essentially the same way every year. In addition to this, an intensive or extra programme is performed every fourth year at each site (Appendix 1). The regular programme includes about 380 samples per year in total for all nuclear facilities and the intensive or extra programme about 160 samples per four year period for all Swedish nuclear facilities. All data reported to SSI are stored in a database. From these data SSI compile and publish an annual report. In the report both environmental results and discharge data are presented and simple trend analyses are performed for a number of radionuclides.

## Some results

An increase of radioactive discharges is common during the annual outage of the nuclear facilities. Outages are usually carried out during the summer and the increase of radionuclide concentrations in biota can be seen in the autumn. Co-60 can be used as a marker for discharges from nuclear facilities. For the period 1983 to 1998 the concentration of Co-60 varied irregularly in environmental samples, but no long term trends can be discerned. The results for bladder wrack are shown in the figure.



For diatomic algae the radionuclide concentration in environmental samples usually corresponds to the activity discharged, but there may be substantial differences. One explanation to these differences is probably weather conditions. The growth rate of the algae is e.g. reduced in cold weather. Another explanation may be the location of the sample collector whether it is placed in strong currents or not, the distance from the outlet of the plant etc. The results of random samples measured by the facilities and SSI in general show good agreement. Most samples show very low activity levels and the Cs-137 activity is still due mainly to the accident in Chernobyl. Higher Cs-137 concentrations are particularly clear in samples from the Forsmark, Studsvik and Oskarshamn areas where significant levels of Cs-137 can be seen in sediments and diatomic algae.

## General conclusions

The environmental monitoring programme for the nuclear facilities has shown that the radioactive discharges increase the concentrations of some radionuclides in the local marine environment around the Swedish nuclear facilities. Samples from the terrestrial environment rarely show increased radionuclide concentrations. From a radiological point of view the most important nuclide in the environmental samples usually is Cs-137. However, the largest part of the present concentrations of Cs-137 in the Swedish environment originate from the Chernobyl accident. The concentrations of radionuclides that can be found in biota around the nuclear facilities are much lower than the concentration levels that are known to give acute damage to organisms. The total radiation doses for humans from the discharges of radionuclides are small.

## Appendix 1

Overview of the samples and number of stations at the Nuclear facilities.

Terrestrial programme	NUMBER OF SAMPLING STATIONS						PERIOD
	Forsmark	Oskarsh.	Barseb.	Ringhals	Studsvik	ABB atom	(S=Spring, A=Autumn)
Natural vegetation							
Hair moss, <i>Polytrichum commune</i> , Stair-step moss, <i>Pleurozium schreberi</i> alt. <i>Dicranum scoparium</i>	3	3	2	2	2		SA
Lichen, <i>Cladina sp.</i>	2	1		2	2		A
Shield fern, <i>Dryopteris filix-mas</i> alt. Bracken, <i>Pteridium aquilinum</i>	2	2	2	2			A
"Maritime grass", fam <i>Poaceae</i>	1	2	1	1		2	A
Cultivated vegetation							
Lettuce, fam <i>Brassicace</i>	1	1	1	1	1		July
Pasture grass, fam <i>Poaceae</i>	2	2	1	3	1		A
Cereals (grains of wheat/barley/rye)	2	1	2	1	2		A
Animal samples							
Sheep, muscle	1	1			1		A
Cattle, muscle	1	1	1	1	1		A
Elk/deer/roe deer, muscle	2	2	1	1	1		A
Pheasant, muscle			1				A
Milk	1	1	1	1	1*		Every fortnight during grazing *twice annually
Sludge	4	3	2	4	3	1	A
Precipitation						4	SA

Marine programme	NUMBER OF SAMPLING STATIONS						PERIOD
	Forsmark	Oskarsh.	Barseb.	Ringhals	Studsvik	ABB atom	(S= Spring, A= Autumn))
Seawater	1	1	1	1			Quarterly
Sediment	2 1 <sup>2)</sup>	1 1 <sup>2)</sup>	1	2 1 <sup>2)</sup>	5 <sup>1)</sup>		Quarterly <sup>1)</sup> twice annually <sup>2)</sup> A
Algae							
Green algae, <i>Cladophora sp.</i>	4	2	2	2	2		A
Bladder wrack, <i>Fucus vesiculosus</i>	2	7	5	7	4 <sup>3)</sup>		A, <sup>3)</sup> SA
<i>Fucus serratus</i>			2				A

Diatomic algae	3	1	1	2	2 <sup>4)</sup>		monthly <sup>4)</sup> monthly ice-free period
<b>Molluscs &amp; Arthropods</b>							
<i>Littorina</i>				2			A
<i>Radix/Theodoxus</i>	2	1					A
Sea mussel, <i>Mytilus edulis</i>	1	3	3	3	2		A
Baltic mussel, <i>Macoma baltica</i> ,	1	1			1		A
Lobster, <i>Homarus gammarus</i>				1			A
Crab, <i>Cancer Pagurus</i>				1			A
<b>Fish</b>							
Eel, <i>Anguilla anguilla</i>	2	3	2	3			SA
Cod, <i>Gadus morrhua</i>	1	1	1	1			A
Plaice, <i>Platichthys flesus</i>			1		1		SA
Corkwing, <i>Crenilabrus melops</i>				1			SA
Herring, <i>Clupea harengus</i>	1	1	1		1 <sup>5)</sup>		A, <sup>5)</sup> S
Pike, <i>Esox lucius</i>	1	1			1		SA
Perch, <i>Perca fluviatilis</i>	1	3			1		SA
Scorpion fish, <i>Myoxocephalus scorpius</i>			1	1			SA
Whitefish, <i>Coregonus sp.</i>	1						A

Overview of the samples and number of stations at the Nuclear facilities.

	NUMBER OF SAMPLING STATIONS				
<b>Intensive programme</b>	Forsmark	Oskarshamn	Barsebäck	Ringhals	Studsvik
Carried out every fourth year in the Spring. The latest and succeeding years for extra programme:	1995 1999 2003	1994 1998 2002	1997 2001 2005	1996 2000 2004	1994 1998 2002
<b>Algae</b>					
Green algae, <i>Cladophora sp.</i>	4	4	4	4	4
Bladder wrack, <i>Fucus vesiculosus</i>	4	6	9	9	6
<b>Molluscs</b>					
<i>Radix/Theodoxus</i>	3	3			2
<i>Littorina sp.</i>				3	
Sea mussel, <i>Mytilus edulis</i>	3	4	4	5	6
Baltic mussel, <i>Macoma baltica</i>	3	1			
Sediment	11	13	11	13	14

# Tsjernobylnedfallet i Midt-Norge: Undersøkelser i 1996

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## Sammendrag

Undersøkelser av radioaktivt cesium i reindriftsutøvere i Midt-Norge har blitt gjennomført siden 1987. I perioden 1987 til 1993 ble målingene gjennomført årlig, mens det deretter gikk tre år før neste måling ble gjennomført i 1996.

Våren 1996 deltok 110 personer i målingene. Gjennomsnittlig helkroppskonsentrasjon av  $^{137}\text{Cs}$  var  $88 \pm 7$  Bq/kg og  $164 \pm 11$  Bq/kg (gjennomsnitt  $\pm$  standard feil) for henholdsvis kvinner (44 personer) og menn (66 personer). Den midlere årlige stråledosen ble beregnet til 0.3 mSv. Konsentrasjonene av radioaktivt cesium var på sitt høyeste mellom 1988 og 1990 for denne befolkningsgruppen, og det har siden 1990 vært en jevn nedgang. Den effektive økologiske halveringstiden for  $^{137}\text{Cs}$  i personene i gruppen i perioden 1987-1996 var ca. 8 år.

Av de 110 deltagerne svarte 101 personer på en spørreundersøkelse om kosthold og bruk av mottiltak. Gjennomsnittlig konsum av reinkjøtt var 63 kg/år. En stor del av gruppen brukte også naturprodukter som vilt (60%), ferskvannsfisk (87%), sopp (36%) og bær (53-94 % for forskjellige bærarter).

De mest brukte og aksepterte mottiltakene var utvelgelse av matreiner etter konsentrasjon av radioaktivt cesium, etter beiteområde med lavere forurensningsnivå eller etter nedføring av matreiner. 64 % av husstandene i undersøkelsen mente at bruk av mottiltak ga dem muligheten til å påvirke eventuelle helseeffekter av Tsjernobylnedfallet for seg selv og familien.

## Introduksjon

Gjennom prosjektet «Overvåkning av stråledoser til utsatte grupper av befolkningen etter Tsjernobylnedfallet» har Statens strålevern siden 1987 gjennomført helkroppsmålinger av radiocesium og kostholdsundersøkelser blant befolkningsgrupper i Norge som ble spesielt rammet etter Tsjernobylnedfallet i april 1986. Områder som ble særlig utsatt for Tsjernobylnedfallet i Norge var Valdres og Gudbrandsdalen, samt Trøndelag og søndre deler av Nordland. Undersøkelsene i 1996 ble finansiert gjennom bidrag fra Statens næringsmiddelstilsyn og EU/NFR-prosjektet «SAVE» (kontrakt nr. F14P-CT95-0015).

Reindriftsutøvere i Midt-Norge (Trøndelagsfylkene, Hedmark og søndre del av Nordland) er blant dem som ble hardest rammet av Tsjernobylnedfallet, og helkroppsmålinger og kostholdsundersøkelser har blitt gjennomført årlig i Snåsa- og Røros-området i perioden 1987-93, deretter i 1996 og 1999.

## Material og metode

### Nuklider og doser

Tsjernobylnedfallet over Norge besto av en rekke ulike radioaktive stoffer. Av størst betydning er de langlevende radioaktive cesiumisotopene  $^{134}\text{Cs}$  og  $^{137}\text{Cs}$  med fysisk halveringstid på henholdsvis 2,1 år og 30,2 år. Den totale aktiviteten av radioaktivt cesium gikk ned forholdsvis raskt de første årene etter ulykken p.g.a den relativt korte fysiske halveringstiden til  $^{134}\text{Cs}$ , og etter 10 år etter ulykken var det svært lite  $^{134}\text{Cs}$  tilbake i miljøet. Kun målinger av  $^{137}\text{Cs}$  er presentert her.

Generelt gjelder det ingen absolutte grenseverdier i forbindelse med radioaktiv forurensning fra ulykker og andre hendelser som ikke er planlagte. I forbindelse med Tsjernobylnedfallet anbefalte myndighetene at stråledosen fra forurensningen ikke skulle overskride 5 mSv det første året etter ulykken og 1 mSv per år i de påfølgende år. For en «gjennomsnittsperson» (ICRP-23, 1975) vil en årsdose på 1 mSv tilsvare en konsentrasjon av  $^{137}\text{Cs}$  i kroppen på ca. 400 Bq/kg i middel gjennom hele året (dosekonversjonsfaktor 2,5  $\mu\text{Sv}/\text{år}$  per Bq/kg [UNSCEAR 1988]).



### *Helkroppsmålingene og spørreundersøkelsen*

Målingene i 1996 ble gjennomført på omtrent samme tid på året som målinger tidligere år (mars-april). I tillegg til å se på helkroppsinneholdet av  $^{137}\text{Cs}$  i gruppen ble det gjennomført en kostholdsundersøkelse. Et tillegg i forhold til kostholdsundersøkelsene tidligere år var spørsmål om bruk av andre naturprodukter enn reinkjøtt. Deltagerne ble også spurt om bruk av tiltak for å redusere inntak av radioaktivt cesium.

Invitasjon til å delta ble sendt ut til alle reindriftsenhetene i Midt-Norge. Representanter fra 63 hushold, eller 62 % av de tilskrevne, møtte opp ved undersøkelsene. Målingene ble utført på to steder (Snåsa og Røros) mens deltakerne i undersøkelsen kom fra en rekke omkringliggende kommuner i både Nordland, Nord-Trøndelag, Sør-Trøndelag og Hedmark. Telletiden under helkroppsmålingene var 10 minutter, mens kostholdsintervjuene tok noe lengre tid.

Totalt 110 personer ble målt i undersøkelsen i 1996. Av disse besvarte 61 menn og 40 kvinner fra til sammen 63 husstander spørreskjemaet om kosthold. Samtlige husstander som deltok i helkroppsmålingene ble altså representert også i kostholdsundersøkelsene. Personene i utvalget kom fra i alt 14 ulike reinbeitedistrikt i de fire fylkene.

Måleutstyret besto av en HPGe-detektor («high purity germanium») med 50% effektivitet, mangelkanalsanalysator kombinert med høgspenningsforsyning og forsterker (merke EG&G ORTEC NOMAD 92X), og mangelkanalsanalysatorprogrammet MAESTRO (også fra EG&G ORTEC). Beregningene skjedde i et PC-program med innlagte kalibreringsfaktorer, laget ved Forsvarets forskningsanstalt (FOA) i Sverige. Videre beskrivelse av utstyret er gitt av Ågren m. fl. (1996).

## **Resultat og diskusjon**

### *Helkroppsmålingene*

Det ble målt 66 menn og 44 kvinner. Det var stor spredning i alderen på deltakerne, fra 16 til 84 år. Kvinnene hadde lavere konsentrasjoner av  $^{137}\text{Cs}$  enn mennene, både gjennomsnittlig og i spredningen av verdiene. Resultatene er beskrevet nærmere i tabell 1. Den gjennomsnittlige konsentrasjonen av  $^{137}\text{Cs}$  for menn og kvinner var 126 Bq/kg, tilsvarende en midlere stråledose fra  $^{137}\text{Cs}$  på ca. 0,3 mSv. Resultatene er beskrevet videre i Mehli m.fl. (1998).

**Tabell 1** Beskrivelse av resultatene fra målingen av helkropsverdier av radioaktivt cesium blant reindriftsutøvere i Midt-Norge i 1996.

	Median	Gjennomsnitt ± st. feil	Spredning (min - max)
<b>Menn (n = 66)</b>			
$^{137}\text{Cs}$ , Bq/kg	153	$164 \pm 11$	24 - 472
Alder, år	43	44	17 - 84
Vekt, kg	75	75	50 - 102
Høyde*, cm		170	157 - 186
<b>Kvinner (n = 44)</b>			
$^{137}\text{Cs}$ , Bq/kg	76	$88 \pm 7$	15 - 238
Alder, år	35	41	16 - 80
Vekt, kg	60	63	46 - 104
Høyde*, m		158	145 - 172

\* Verdiene mangler for tre menn og fem kvinner.

### *Kostholdsundersøkelsen*

Kostholdsundersøkelsen ble gjennomført med en utvidet versjon av samme spørreskjema som er brukt ved tidligere undersøkelser av Statens strålevern (1989-1991) [Strand m.fl. 1989, Strand m.fl. 1992]. Undersøkelsen var konsentrert om forbruk av reinsdyrkjøtt, men det ble i tillegg spurt om bruk av ferskvannsfisk, bær og sopp.

Individuelle inntak av matvarer ble beregnet på grunnlag av den mengden mat husholdningen som helhet hadde skaffet seg. Mengden ble delt på antallet personer i husholdet og den antatte porsjonsfordeling mellom dem, og korrigert for spiselige mengder. Reinsdyrkjøtt var den eneste

matvaren fra skog og utmark som alle i utvalget brukte (tabell 2). Imidlertid brukte en høy andel av de spurte andre naturprodukter også, med blåbær (57 %) og sopp (38 %) som de minst vanlige.

**Tabell 2** Individuelle årlige inntak av matvarer fra skog og utmark beregnet på grunnlag av husholdningens totale anskaffelse (i kg).

	Prosent som bruker matvaren	Median	Gjennomsnitt	Spredning (min. - max.)
<b>Menn (n = 53)</b>				
Reinsdyrkjøtt, kg	100 %	53	63	10 - 149
Ferskvannsfisk, kg	87 %	3,2	4,2	0 - 17
Vilt, kg	60 %	1,6	4,6	0 - 29
Sopp, kg	36 %	0	0,70	0 - 12
Molter, kg	91 %	2,8	3,6	0 - 15
Tyttebær, kg	94 %	3,0	4,3	0 - 25
Blåbær, kg	53 %	0,30	1,4	0 - 20
<b>Kvinner (n = 37)</b>				
Reinsdyrkjøtt, kg	100 %	50	51	11 - 120
Ferskvannsfisk, kg	89 %	4,0	4,4	0 - 15
Vilt, kg	78 %	2,5	6,4	0 - 74
Sopp, kg	41 %	0	0,39	0 - 2,5
Molter, kg	95 %	3,0	4,6	0 - 29
Tyttebær, kg	91 %	3,0	4,1	0 - 15
Blåbær, kg	60 %	0,80	1,2	0 - 5,0

#### *Bruk og vurdering av mottiltak*

Det vanligst brukte mottiltaket var å velge ut matrein fra egne beitesteder med mindre forurensning og å velge ut dyr til eget forbruk etter å ha foretatt målinger av dyret. I tillegg oppga i overkant av 20% at de føret ned matrein til eget bruk før slakting. Under 8% av husholdningene tok hensyn til radioaktivitet i tilberedning av mat, og da kun en gang i blant. Det eneste tiltaket som ble nevnt i denne forbindelse var å bruke litt mindre reinsdyrkraft.

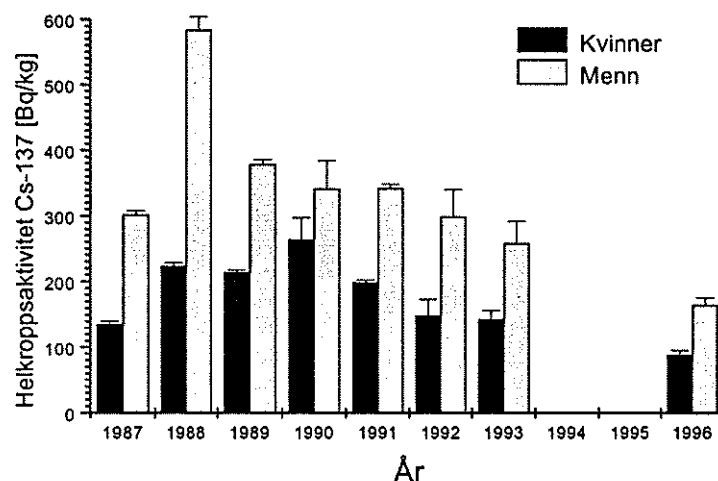
Majoriteten av husstandene i undersøkelsen (64 %) følte helt eller til en viss grad, at mottiltakene ga dem mulighet til å påvirke eventuelle helseeffekter av ulykken for seg selv og sin familie. Bare 3 %, to personer, trodde de hadde liten mulighet for å påvirke disse eventuelle effektene selv, mens 33 % av husstandene valgte å ikke svare på spørsmålet. Nedføring av matrein var det mottiltaket flest trodde var det beste, mens å velge slakt med lave verdier eller bytte reinsdyrkjøtt med annen mat ble foretrukket av en noe mindre andel av husstandene. Tidlig slakt av matrein ble også nevnt av en større andel. Resultatene er beskrevet videre i Mehli m.fl. (1998).

#### *Endringer i <sup>137</sup>Cs-konsentrasjoner over tid*

Utviklingen i konsentrasjon av <sup>137</sup>Cs i reindriftsutøvere i Midt-Norge fra 1987 til 1996 er vist i figur 1. Resultatene fra helkroppsmålingene i 1996 viser at konsentrasjonene ligger lavere enn ved siste måling i 1993. For begge kjønn var gjennomsnittsverdiene høyest mellom 1988 og 1990. Målingene har blitt gjennomført på omtrent samme tid av året (mars-april) ved alle målingene. For perioden 1988-1996 følger nedgangen i <sup>137</sup>Cs-konsentrasjonene en halveringstid på 7-9 år.

#### **Konklusjon**

Helkroppsmålingene i 1996 ga et gjennomsnitt på 126 Bq/kg. Resultatene viser en jevn nedgang i helkroppsinnhold av <sup>137</sup>Cs blant reindriftsutøvere i Midt- og Sør-Norge. Den effektive økologiske halveringstiden for <sup>137</sup>Cs i perioden 1987-1996 er ca. 8 år. Myndighetene anbefaler at stråledose fra radioaktiv forurensning etter Tsjernobyl-ulykken ikke overskrider 1 mSv per år. Midlere stråledose for denne befolkningsgruppen ble i 1996 estimert til 0.3 mSv, noe som er godt under denne verdien. Det var imidlertid enkeltpersoner som kan ha mottatt en årlig stråledose rundt 1 mSv.



**Figur 1** Gjennomsnittlig helkroppsverdier av  $^{137}\text{Cs}$  i reindriftsutøvere i Midt- og Sør-Norge i perioden 1987-1996. (Gjennomsnittsverdi  $\pm$  standard feil).

Kostholdsundersøkelsen som ble gjennomført i 1996 viste at samtlige av deltagerne konsumerte reinsdyrkjøtt. En stor del av gruppen brukte også andre naturprodukter som vilt (ca. 70 %), ferskvannsfisk (ca. 90 %), sopp (ca. 40 %) og bær (ca. 90 %). Dette viser at naturprodukter er en viktig del av kostholdet for denne gruppen.

De mest brukte mottiltakene var å velge ut matreir etter beiteområde med lavere forurensning eller etter målinger av dyr før slakt. Flere benyttet seg også av nedføring av matreir. Bare 8% brukte mottiltak i forbindelse med tilberedning av mat bevisst. Majoriteten av de spurte mente at de ved hjelp av mottiltak kunne påvirke eventuelle helseeffekter av ulykken for seg selv og sin familie.

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## THE ROLE OF BIOACCUMULATION IN MIGRATION OF TECHNOGENIC RADIONUCLIDES IN FRESHWATER ECOSYSTEM

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**Abstract** – The concentrations of  $^{137}\text{Cs}$ ,  $^{60}\text{Co}$  and  $^{54}\text{Mn}$  were estimated in bottom sediments and different species of macrophytes as well as fish of Lake Druksiai, the cooling basin for the Ignalina NPP. Besides, the concentration factors for radionuclides were calculated in sediments and macrophytes. The results suggested that macrophytes accumulated radionuclides from water more than from bottom sediments. The “trophic level effect” (biomagnification) was found for  $^{137}\text{Cs}$  accumulation in fish, while as for  $^{60}\text{Co}$  and  $^{54}\text{Mn}$  that effect did not occur.

**Key words:** radionuclides, concentration factors, bottom sediments, macrophytes, fish

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### INTRODUCTION

The distribution and transport of radionuclides in hydroecosystems are being affected by various abiotic and biotic environmental factors. The role of bioaccumulation, a biological factor of the radionuclide migration in freshwater ecosystem, depends on radionuclide redistribution between the main ecosystem's components: water, bottom sediments and hydrobionts. The origin, physical and chemical properties of radionuclides are one of the main factors causing this process. Living organisms change the radionuclide migration properties in environment by their inclusion into biological exchange mechanism (Polikarpov, 1964). Aquatic plants play an important role in this case by forming the greatest biomass, concentrating chemical elements and their radionuclides and serving as the first link in the aquatic food chain (Marciulioniene, 1994). Thus, much attention should be paid to the freshwater foodweb: *water– bottom sediments– aquatic plants– nonpredatory fish– predatory fish*.

The aim of this work was to establish the accumulation levels of  $^{137}\text{Cs}$ ,  $^{60}\text{Co}$  and  $^{54}\text{Mn}$  in bottom sediments, macrophytes of different ecological groups and fish of different trophic levels (rudd, roach and bream as nonpredators; perch and pike as predators) and to study the relationship of radionuclide accumulation between these components by evaluating the role of biological factors.

### MATERIALS AND METHODS

Samples of bottom sediments and macrophytes were collected in July 1993-1997, in Lake Druksiai. The bottom sediments were sampled in macrophyte habitats. Fish samples from the lake were collected in April, July and October 1994-1998. The samples were dried at 80°C and then concentrated by incineration at 400°C. Samples of standard geometry have been prepared for  $\gamma$ -spectrometric measurements carried out at the Institute of Physics. A low background high-resolution spectrometer with a Ge (Li) semiconductor detector was used for the purpose. The data about radionuclide concentrations in water of the lake were from pers.com. (Motiejunas; Mazeika). Radionuclide concentrations and concentration factors ( $\text{CF} = \text{sample/water}$ ) in samples were calculated for dry weight.

## RESULTS AND DISCUSSION

For the studies of radionuclide accumulation in the system *water – bottom sediments – plant*, the rhizomes of littoral halophytes (emergent macrophytes), *Typha latifolia* and *Phragmites australis*, were chosen for their being perennials and accumulating radionuclides permanently, like sediments. The results show (Table 1.) that the uptake of  $^{137}\text{Cs}$  by halophyte rhizomes is stronger ( $\text{CF}=5249\pm2009$ ) than that by sand ( $822\pm96$ ) and weaker than by clay fraction ( $14200\pm2890$ ). The uptake of  $^{60}\text{Co}$  and  $^{54}\text{Mn}$  by rhizomes is stronger ( $\text{CF}=40520\pm10449$  &  $62000\pm2065$ , respectively) than that by bottom sediments in both cases ( $12000\pm5324$  &  $653\pm112$ ;  $32000\pm8936$  &  $36000\pm12547$ , respectively).

Table 1. Concentration factors (mean $\pm$ SD) for radionuclides in macrophytes and bottom sediments of Lake Druksiai

Samples	$^{137}\text{Cs}$	$^{60}\text{Co}$	$^{54}\text{Mn}$
Cladophora spp.	7046 $\pm$ 1633	62412 $\pm$ 36351	98000 $\pm$ 2350
Nitellopsis obtusa	6111 $\pm$ 2811	9411 $\pm$ 1441	11176 $\pm$ 3150
Ceratophyllum demersum	8753 $\pm$ 2698	62000 $\pm$ 24005	82000 $\pm$ 3212
Myriophyllum spicatum	7743 $\pm$ 2882	18608 $\pm$ 6471	33315 $\pm$ 10973
Batrachium aquatile	4048 $\pm$ 2048	0	0
Potamogeton perfoliatus	2631 $\pm$ 189	12941 $\pm$ 2346	16000 $\pm$ 3142
Potamogeton lucens	2354 $\pm$ 234	0	0
Stratiotes aloides	2841 $\pm$ 340	0	0
Polygonum amphibium	2420 $\pm$ 410	0	0
Typha latifolia (rhizomes)	5249 $\pm$ 2009	40520 $\pm$ 10449	62000 $\pm$ 2065
Phragmites australis (rhizomes)	4378 $\pm$ 782	16667 $\pm$ 5735	33467 $\pm$ 1068
Sediments: clay fraction	14200 $\pm$ 2890	32000 $\pm$ 8936	36000 $\pm$ 12547
sand	822 $\pm$ 96	12000 $\pm$ 5324	653 $\pm$ 112

The results presented on Figure 1. suggest that  $^{137}\text{Cs}$  accumulation in *Typha latifolia* rhizomes does not depend on its concentration in bottom sediments ( $r=0.034$ ). Probably, the rhizomes accumulate  $^{137}\text{Cs}$  more from water, where it exist in exchangeable an ionic form, than from bottom sediments, where this radionuclide is strongly bounded and its bioavailability is lower. Besides, there is no correlation between  $^{60}\text{Co}$  concentrations in rhizomes and bottom sediments ( $r=0.260$ ). An absence of the above mentioned correlation and concentrations of  $^{60}\text{Co}$  and  $^{54}\text{Mn}$  that are higher in rhizomes than in sediments suggest that the uptake of these radionuclides by rhizomes is being supplied mainly from water due to the same reason as for  $^{137}\text{Cs}$ .

The results of the studies in the system *water– plant* show the highest CF for  $^{137}\text{Cs}$  to be characteristic of submerged *Ceratophyllum demersum*, *Nitellopsis obtusa* and *Cladophora spp.* as well as for subemergent *Myriophyllum spicatum*, which have a larger amount of potassium (Table 1). Direct correlation between  $^{137}\text{Cs}$  and its macroanalogue  $^{40}\text{K}$  concentrations in *Ceratophyllum demersum* (Fig.2.) and in other three species ( $r=0.641$ ;  $0.629$ ;  $0.413$ , respectively) was indicated. Although the same correlation was found for perennial *Typha latifolia* rhizomes ( $r=0.814$ ), in the latter the CF for radiocaesium was significantly lower than in the above mentioned annual macrophytes. Probably, the bottom sediments, especially those rich in organic, by accumulating intensely radionuclides act as a “shield” for radionuclide uptake by plant rhizomes and roots from water column.

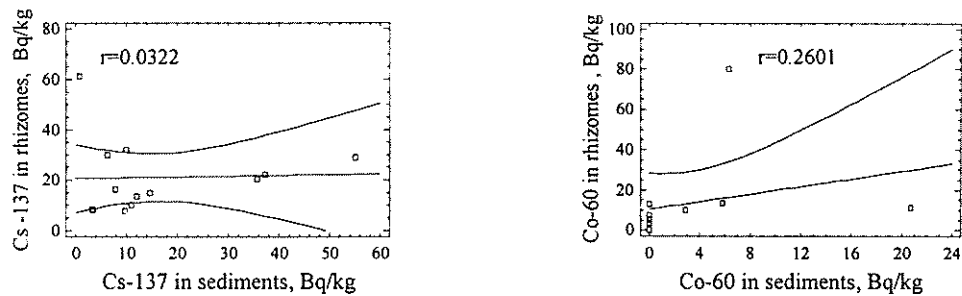


Fig. 1. Relationship between  $^{137}\text{Cs}$  and  $^{60}\text{Co}$  concentrations in *Typha latifolia* rhizomes and bottom sediments.

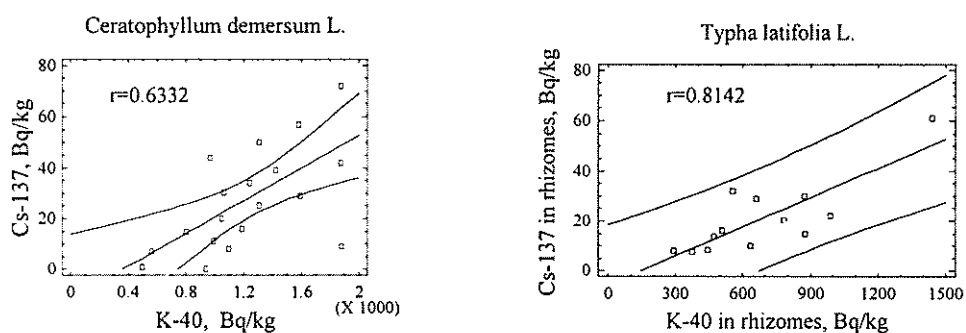


Fig. 2. Correlation between  $^{137}\text{Cs}$  and  $^{40}\text{K}$  concentrations in macrophytes.

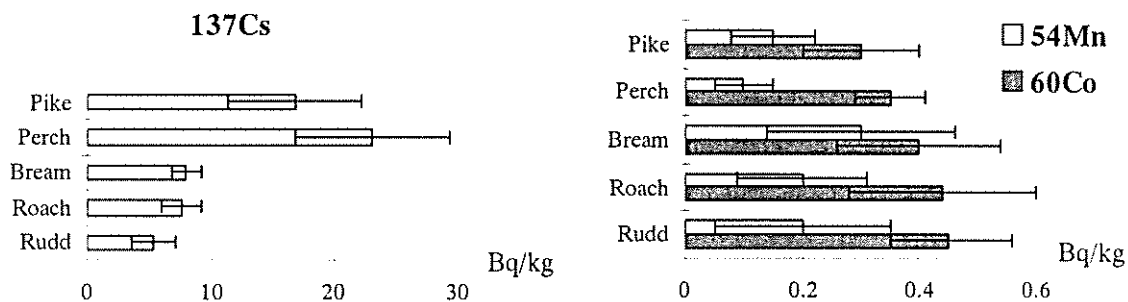


Fig. 3. Radionuclide concentrations (mean $\pm$ SD) in muscle tissue of fish from Lake Druksiai.

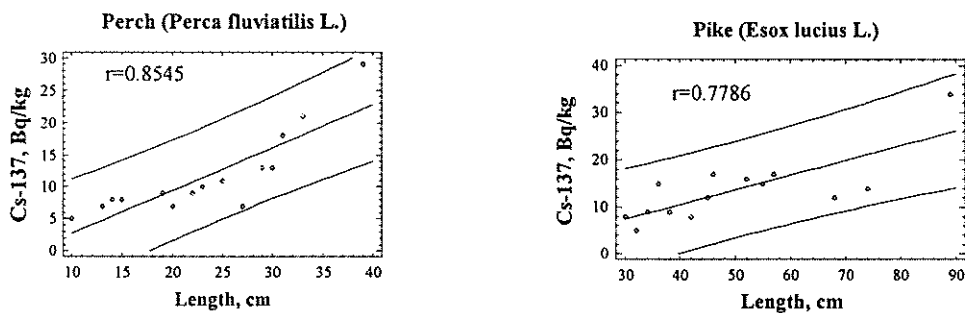


Fig. 4. Correlation between  $^{137}\text{Cs}$  concentration (in whole fish) and the body length of perch and pike.

The highest CF for  $^{60}\text{Co}$  and  $^{54}\text{Mn}$  were estimated in *Cladophora spp.* and *Ceratophyllum demersum*, which have a larger relative surface area per mass unit as the main route of  $^{60}\text{Co}$  and  $^{54}\text{Mn}$ , radionuclides prone to hydrolysis, uptake by plants is their surface adsorption.

The studies of radionuclide accumulation in fish confirmed the "trophic level effect" (biomagnification) to be characteristic of  $^{137}\text{Cs}$  accumulation. The highest concentrations of radiocaesium were found in predators, perch and pike (Fig.3). For  $^{137}\text{Cs}$  accumulation in different size classes of predators a positive "size effect" was clearly expressed (Fig.4). It might have been conditioned by changes in food composition resulted in by fish growth. For example, a perch moves from plankton (juveniles) and macroinvertebrates (smaller individuals) to fish (greater individuals), while nonpredatory cyprinids take more or less the same food at different life stages (Hadderingh et al., 1996). The higher concentrations of  $^{60}\text{Co}$  and  $^{54}\text{Mn}$  were found in nonpredators, rudd, roach and bream. Thus, for the mentioned radionuclides the "trophic level effect" does not occur. It is likely that the main uptake of radionuclides of corrosive origin by fish occurs from water as the annual investigations in Lake Druksiai have shown that  $^{60}\text{Co}$  and  $^{54}\text{Mn}$  levels in fish depend on the discharges of these radionuclides from the Ignalina NPP into the lake. The fact is confirmed by the accumulation in organs and tissues of fish as well (Petkeviciute & Marciulioniene, 1999).

## CONCLUSIONS

1. The main uptake of radionuclides by macrophytes in the system *water– bottom sediments – plant*, like in the system *water– plant*, eventuates from water. The "shield" of bottom sediments decreases the radionuclide uptake from water by plant rhizomes and roots. 2. The radionuclide accumulation in macrophytes depends on the physiological, morphological and ecological characteristics of the species. 3. Being the first barrier of radionuclides in the water basin, macrophytes reflect pollution with radionuclides, accumulating them during vegetation period, more efficiently than sediments. The long-term pollution of water basin with  $^{137}\text{Cs}$  is being better reflected by bottom sediments, while the long-term pollution with  $^{60}\text{Co}$  and  $^{54}\text{Mn}$  – by rhizomes of perennial halophytes *Typha latifolia*. 4. The "trophic level effect" is characteristic of  $^{137}\text{Cs}$  accumulation in fish. The "size effect" was found for predators. 5. Food has no considerable importance for the accumulation of  $^{60}\text{Co}$  and  $^{54}\text{Mn}$  in fish, and accumulation of these radionuclides is being caused mainly by their levels in water. 6. The results of the studies allow to assuming, that in the first stage of radionuclide dispersion and migration, when radionuclides enter the freshwater basin, intensive processes of radionuclide accumulation in bottom sediments and especially in aquatic plants take place, causing a great decrease in the radionuclide amount in water. In the second stage of radionuclide dispersion and migration, when radionuclides redistribute between components of freshwater ecosystem, a more important role is being played by the transfer of radionuclides from died plants to bottom sediments as well as through food chains to the hydrobionts of higher trophic levels.

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## LEVEL OF $^{90}\text{Sr}$ IN THE URINE OF A SMALL GROUP OF FINNISH PEOPLE

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**Abstract-** The aim of the study was to test the feasibility of the applied analysis method for  $^{90}\text{Sr}$  and if possible to estimate the current level of the  $^{90}\text{Sr}$  concentration in the urine. Urine samples were collected from seven Finnish volunteers in connection with studies of  $^{137}\text{Cs}$  body burdens. The activity measurements of urine samples were performed 14 - 18 days after chemical separation of  $^{90}\text{Sr}$  to allow ingrowing of  $^{90}\text{Y}$ . The  $^{90}\text{Sr}$  and  $^{90}\text{Y}$  activities were measured simultaneously using a Quantulus liquid scintillation spectrometer. The detection limit for  $^{90}\text{Sr}$  was 0.0033 Bq per sample, or 0.0007 – 0.0015 Bq l<sup>-1</sup>. The  $^{90}\text{Sr}$  activities in urine varied between 0.006 and 0.046 Bq l<sup>-1</sup>. The daily urinary excretion was found to be 0.007 – 0.018 Bq for the five volunteers that collected three-day urine samples. Assuming that the daily  $^{90}\text{Sr}$  intake was constant and that 18 % of the ingested activity was excreted in urine, the mean intake in the investigated group would vary between 0.039 and 0.1 Bq d<sup>-1</sup>. Based on these estimated intake values the respective annual effective internal doses from  $^{90}\text{Sr}$  and  $^{90}\text{Y}$  varied from 0.4 to 1 Sv during the sampling period.

### INTRODUCTION

The nuclear bomb tests performed mainly in the 50's and 60's caused a worldwide radioactive fallout from which radionuclides with long half-life are still found in the environment. In Finland the mean accumulated deposition of  $^{137}\text{Cs}$  was 1800 Bq m<sup>-2</sup> and that of  $^{90}\text{Sr}$  1100 Bq m<sup>-2</sup> at the end of 1985 (STUK-A54). In 1986 the fallout from the Chernobyl accident increased the mean  $^{137}\text{Cs}$  deposition by 10 700 Bq m<sup>-2</sup> but the  $^{90}\text{Sr}$  deposition was very low, only a few per cent of that of  $^{137}\text{Cs}$  (Arvela et al. 1990, Saxen et al. 1987). As a consequence only small amounts of these radionuclides can be found in foodstuffs and in man. The amounts of  $^{137}\text{Cs}$  in the body can be measured by whole-body counting but the amounts of  $^{90}\text{Sr}$  must be determined indirectly by estimating the intake via consumed foodstuffs or analysing daily excreted amounts.

Studies performed in 1964 showed that the mean  $^{90}\text{Sr}$  concentration of 18 females in southern Finland was 0,28 Bq l<sup>-1</sup> (Jaakkola et al. 1969). After the Chernobyl accident several studies on the whole-body contents of  $^{137}\text{Cs}$  of people have been made but the estimates of  $^{90}\text{Sr}$  have been based on intake evaluations.

It was decided to test the feasibility and sensitivity of the applied analysis method for  $^{90}\text{Sr}$  in urine and if possible to estimate the current level of the  $^{90}\text{Sr}$  concentration in the urine of the volunteers. In case of a possible internal  $^{90}\text{Sr}$  contamination it is important to be able to determine the  $^{90}\text{Sr}$  concentration in urine and thus the radiation dose. Urine samples of four volunteers were collected in a study of  $^{137}\text{Cs}$  biological half-life. It was decided to use these samples also for  $^{90}\text{Sr}$  determinations since the collection of excreta is a difficult task.

### MATERIALS AND METHOD

Urine samples measured for  $^{137}\text{Cs}$  concentration together with urine samples from three other persons were used for the  $^{90}\text{Sr}$  analyses. One of the persons was female with a typical Finnish diet and two of the male persons with a diet including much freshwater fish and other products from nature. All persons were adult and resided in Southern Finland. Three persons collected two three-day period samples each. The samples were taken between 1996 and 1999. From the persons with diets containing much products from nature only overnight collections of urine could be arranged.

The urine samples were first coprecipitated with calcium phosphate to concentrate the strontium and the precipitate then wet-ashed with nitric acid (Dietz et al. 1991). Radiochemical separation of  $^{90}\text{Sr}$  was carried through nitrate, chromate and carbonate precipitations (Bryant et al. 1959). The activity measurements were performed 14-18 days after chemical separation of  $^{90}\text{Sr}$  to allow ingrowing of  $^{90}\text{Y}$ . The  $^{90}\text{Sr}$  and  $^{90}\text{Y}$  activities were measured simultaneously using a 1220 Quantulus (Wallac Oy) liquid scintillation spectrometer. The measuring time was 300 min. The detection limit for  $^{90}\text{Sr}$  was 0.0033 Bq per sample, or 0.0007 – 0.0015 Bq l<sup>-1</sup>.



The results obtained with this method proved to be good in an intercomparison exercise arranged by Procorad (Association pour la promotion du controle de qualite des analyses de biologie medicale en radiotoxicologie) in which the  $^{90}\text{Sr}$  activity in urine samples was analysed.

## RESULTS AND DISCUSSION

The results of the  $^{90}\text{Sr}$  analyses of urine from the 7 volunteers are presented in the table below. The  $^{90}\text{Sr}$  activities in urine varied between 0.006 and 0.046 Bq l<sup>-1</sup>, the mean being 0.019 Bq l<sup>-1</sup> in 1996 – 1999. The comparison of this value with the earlier values from the year 1964 shows that the concentrations have decreased approximately by a factor of ten. The two men who had a diet including large quantities of freshwater fish had the highest activity concentration of  $^{90}\text{Sr}$  in the urine.

**Table.** Activity of  $^{90}\text{Sr}$  in urine mBq l<sup>-1</sup> and daily excreted  $^{90}\text{Sr}$

Person No. and Gender	Collecting time	Amount of Urine l	$^{90}\text{Sr}$ activity in urine mBq l <sup>-1</sup>	Excreted $^{90}\text{Sr}$ mBq d <sup>-1</sup>
1 M	14.2.1996 16:35 17.2.1996 10:00	5.00	6.4	11
2 M	19.2.1996 17:00 22.2.1996 12:00	4.93	6.0	10
2 M	27.2.1996 12:30 1.3.1996 0:00	2.18	7.9	7
3 M	21.2.1996 12:40 23.2.1996 8:20	2.75	12.6	17
4 M	10.2.1997 11:30 13.2.1997 13:00	5.30	10.1	18
4 M	13.2.1997 14:00 16.2.1997 12:00	5.20	6.6	11
5 F	19.3.1999 7:00 22.3.1999 7:00	3.42	9.9	11
5 F	28.6.1999 17:00 1.7.1999 17:00	4.45	10.0	15
6 M	1.7.1999	0.60 <sup>a</sup>	46.2	NA <sup>b</sup>
7 M	1.7.1999	0.57 <sup>a</sup>	25.3	NA <sup>b</sup>

<sup>a</sup> overnight collection only

<sup>b</sup> not available

The amount of  $^{90}\text{Sr}$  excreted per day was estimated on the basis of the collecting time, amounts of urine and  $^{90}\text{Sr}$  activity in urine. The daily urinary excretion was found to be 0.007 – 0.018 Bq for the persons 1-5. In the case of overnight samples no daily excretion value could be calculated. According to ICRP-23, the daily intake of stable strontium is 1.9 mg and the excretion via urine 0.34 mg d<sup>-1</sup> and via faeces 1.5 mg d<sup>-1</sup>. Thus in the equilibrium state, about 18 % of the stable strontium ingested daily is excreted via urine. Assuming that the daily  $^{90}\text{Sr}$  intake of the persons had been approximately constant for a long time and that 18 % of the ingested activity was excreted in urine, the mean daily intake in the investigated group would vary between 0.039 and 0.1 Bq. Based on the activity measurements of foodstuffs and mean consumption rates, the mean daily intake of  $^{90}\text{Sr}$  for adult Finns via foods was estimated to be about 0.3 Bq in 1988 (Rantavaara 1991). The results obtained in this study are in good agreement with that value when the small size of the group and the decrease of  $^{90}\text{Sr}$  concentrations in foodstuffs with time are taken into account. Thus the results obtained may be used as a current 'background' level for  $^{90}\text{Sr}$  in urine in Finnish persons.

The annual intake of  $^{90}\text{Sr}$ , based on the values estimated in this study, would vary between 14 and 37 Bq. Applying the dose intake factors given in The Basic Safety Standards (IAEA 1996) and

assuming that  $^{90}\text{Sr}$  and  $^{90}\text{Y}$  are in equilibrium in the body the annual effective internal doses from  $^{90}\text{Sr}$  and  $^{90}\text{Y}$  for the group studied varied would have been between 0.4 to 1 S v during the sampling period.

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# Uptake and activity concentration of $^{137}\text{Cs}$ and $^{90}\text{Sr}$ in *Salix viminalis*

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## Introduction

Due to overproduction and declining economy in agriculture, many farmers in Sweden are interested in changing production from food to energy on their farm land. Some of these land areas are contaminated with radionuclides and suitable crops are being looked for. The Energy Forestry concept is a potential candidate for these soils (Christersson et al. 1993, Sennerby-Forsse and Johansson 1989). Among the radionuclides, caesium,  $^{137}\text{Cs}$ , and strontium,  $^{90}\text{Sr}$ , are the most important for impact on human health risk. These nuclides have a high degree of mobility in soil- plant- systems. Our knowledge of the dynamics of radionuclides in salix-forest ecosystems is fragmented. In a previous investigation about uptake and distribution of radio caesium in *Salix* (Sennerby-Forsse et al. 1993) it was shown that the total plant uptake of  $^{134}\text{Cs} + ^{137}\text{Cs}$  was approximately 0.2% of the caesium present in the soil substrate. Almost 90% of the assimilated caesium were found in the roots. The total amount of caesium in the plants increased over time.

In order to gain a better understanding of the uptake, internal distribution of  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  and the effects of K-fertilisation on uptake of  $^{137}\text{Cs}$  in *Salix* plantations, a K-fertilisation experiment including  $^{137}\text{Cs}$ ,  $^{90}\text{Sr}$  and  $^{40}\text{K}$  was performed. Previous studies with K-fertilization in Sweden have mainly dealt with the flow of  $^{137}\text{Cs}$  between soil to plant in annual crops (Rosén 1991).

The aim of this study was to study the transfer of  $^{137}\text{Cs}$ ,  $^{90}\text{Sr}$  and  $^{40}\text{K}$  from soil to plants and to test the following hypothesis: i) that accumulation of  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  differs between plant organs, ii) that there is a seasonal variation in uptake and concentration of  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  and iii) that the availability of K is related to the uptake of  $^{137}\text{Cs}$ .

## Material and methods

The experiment was a micro plot design with the plot size 0.5\*0.5\*0.4. Three blocks, with 16 micro plots in each block were included in the experiment. The plough layer, 0-22 cm, of the soils consisted of loam or clay loam and the subsoil was a loamy sand. The treatments consisted of three different amounts of K with 4 replications. The topsoil layers were homogeneously contaminated with  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  in spring 1961. In 1994 the activity in the soils was 16.68 MBq/m<sup>2</sup> of  $^{137}\text{Cs}$  and 6.04 MBq/m<sup>2</sup> of  $^{90}\text{Sr}$ .

In the beginning of August 1993, two cuttings of *Salix viminalis*, clone 78183, were planted in each micro plot. During 1994 and 1995 all blocks were fertilised with nitrogen (N) (60 kg N/ha) and different amounts of K, 0 kg K/ha in block 1, 80 kg K/ha in block 2 and 240 kg K/ha in block 3. Before the onset and at termination of the experiment, soil samples (0-25 cm)

were taken in order to determine the chemical characteristics of the soil. Growth measurements of the plants were carried out regularly.

Sampling for radioactivity measurements in stems, leaves and roots were carried out during the different seasons in 1994, 1995 and 1996. Roots were separated into fine 1-2 mm, coarse 1-2 mm and coarse >2 mm roots. Activity concentrations of  $^{137}\text{Cs}$  and  $^{40}\text{K}$  in aliquots of biomass- and soil samples were determined by means of high-purity germanium detector systems housed in a low-background laboratory. The radio caesium measurement errors were in the range of 1-5 % for  $^{137}\text{Cs}$  and 1-15 % for  $^{40}\text{K}$ .  $^{90}\text{Sr}$  were analysed according to Suomela 1973. All activity concentrations in plant samples were measured on a dry weight (d.w.) basis and were corrected for decay back to the date of experiment start. measure

## Results

Chemical analysis of macro nutrients in stem samples before the fertilisation treatment started showed levels of about 9.2 mg N/g, 1.3 mg P/g and 5.5 mg K/g dry weight in all three treatments. During and after the first and second growing season, the concentration of  $^{137}\text{Cs}$  in stems and leaves was significantly affected by K supply, particularly in the treatments with 0 and 240 kg K/ha (Table 1). However, after the third growing season, the fertilisation effect had disappeared. In general, the  $^{137}\text{Cs}$  concentration was higher in the 0 kg K treatment than in the 80 kg or 240 kg treatment. No significant difference in plant growth between the two soil types was observed.

**Table 1.**  $^{137}\text{Cs}$  concentration (Bq/kg)  $\pm$ SD in *Salix* plants grown under different K fertilization. Means followed by the same letter within each column are not significantly different (LSD at 0.05 level)

Plant organ		Sampling date	0 kg/ha	K fertilisation		240 kg/ha	n
				80 kg/ha			
Stem		940819	800 $\pm$ 245 c	569 $\pm$ 276 b		368 $\pm$ 146 a	14
Stem		950830	1 306 $\pm$ 779 b	432 $\pm$ 239 a		497 $\pm$ 525 a	10
Leaves		940819	2 938 $\pm$ 1 313 b	2 272 $\pm$ 1 144 b		1 418 $\pm$ 69 a	14
Leaves		950830	3 522 $\pm$ 1 304 b	1 334 $\pm$ 531 a		1 279 $\pm$ 1 058 a	5
Roots		941117	2 954 $\pm$ 1 237 a	1 858 $\pm$ 1 040 a		3 162 $\pm$ 1 307 a	2
Fine	0-1 mm	960926	20 536 $\pm$ 4 878 a	20 483 $\pm$ 3 513 a		19 468 $\pm$ 4 208 a	8
Coarse	1-2 mm	960926	4 866 $\pm$ 1 280 a	4 005 $\pm$ 711 a		4 271 $\pm$ 640 a	8
"	>2 mm	960926	1 750 $\pm$ 284 a	1 622 $\pm$ 505 a		1 866 $\pm$ 559 a	8
Roots		Mean	9 051 $\pm$ 8 848 a	8 703 $\pm$ 8 796 a		8 535 $\pm$ 8 306 a	24

n = sampling number

The concentration of  $^{137}\text{Cs}$  was highest in roots and leaves. Among roots, fine roots had the highest  $^{137}\text{Cs}$  concentration. The concentration of  $^{137}\text{Cs}$  in stems and leaves decreased from summer to autumn. The  $^{137}\text{Cs}$  concentration in leaves and stems were higher during the second growing season than during the first growing season. This tendency was also showed in the roots.

There was no significant difference in  $^{90}\text{Sr}$  concentration between the K treatments. Leaves contained more  $^{90}\text{Sr}$  than the other organs. The  $^{90}\text{Sr}$  concentration in stems decreased from summer to autumn, while it increased in leaves. The  $^{90}\text{Sr}$  concentration in stems and leaves was higher after the first growing season than after the second. The concentration of  $^{40}\text{K}$  kept relatively stable throughout the whole experiment and in all plant organs. No significant difference in  $^{40}\text{K}$  concentration between treatments was observed. The  $^{40}\text{K}$  concentration in leaves and roots (0-2mm) were generally higher than in other plant organs. The  $^{40}\text{K}$  concentration of one-year-old stems was higher than in two-year-old stems.

## Discussion

The  $^{137}\text{Cs}$  concentration in all tissues was generally higher in the 0 K treatment. This indicates that uptake of  $^{137}\text{Cs}$  is negatively correlated with the K content in the soil solution. The  $^{137}\text{Cs}$  concentration was higher in roots than in the other plant organs. This observation agrees well with other studies of *Salix* (Sennerby-Forsse et al. 1993). Among roots, the  $^{137}\text{Cs}$  concentration in fine roots was about 2.5 time higher than that of the coarse roots, although the  $^{137}\text{Cs}$  content was almost the same. This may indicate that fine roots play an important role for  $^{137}\text{Cs}$  uptake while coarse roots are more important for  $^{137}\text{Cs}$  storage. The distribution of caesium within the roots indicates that  $^{137}\text{Cs}$  may have moved from fine roots to coarse roots before exudation and other losses occurred. The observation that  $^{137}\text{Cs}$  concentration in leaves decreased from summer to autumn supports the finding by Nylén and Ericsson (1996) on Scots pine and by Peter et al. (1969) on tulip poplar. This is probably due to retranslocation of  $^{137}\text{Cs}$  from foliage to woody tissue during mature and senescent stages of leaf growth. Also removal of  $^{137}\text{Cs}$  from foliage by rain and dew leaching prior to leaf fall may have an influence. Our observations indicate that the behaviour of  $^{137}\text{Cs}$  in physiological processes is similar or analogous to the behaviour of K in plants. Compared to  $^{137}\text{Cs}$ , the  $^{90}\text{Sr}$  concentration was generally higher in all plants. The reason for that could be that the transfer of radionuclides from the soil to the plants appears to decrease with time between soil contamination and harvest of plant, and this decrease is generally larger for caesium than for strontium. Previous studies have shown that after some years, the soil content of  $^{137}\text{Cs}$  gradually becomes less available for plant uptake than  $^{90}\text{Sr}$  (Krouglov, et al. 1997).

## Conclusions

Results from the present study showed that *Salix viminalis* plants accumulates radionuclides preferably in roots and leaves and less in the stems. In Energy Forestry plantations the stems are repeatedly harvested and used for energy production. We therefore suggest that production of biomass with fast-growing *Salix* clones should be considered as an alternative land use for contaminated agricultural soils.

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# DIFFERENCES IN THE ECOLOGICAL HALF-TIME OF RADIOCAESIUM FROM THE CHERNOBYL ACCIDENT AND FROM NUCLEAR WEAPONS FALLOUT AS MEASURED IN A SOUTH SWEDISH POPULATION

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**Abstract**—Between 1964 and 1994 the whole-body content of  $^{134}\text{Cs}$  and  $^{137}\text{Cs}$  was measured in a south-Swedish population living in the city of Lund, the so-called Lund reference group, in order to investigate the effective ecological half-time of  $^{137}\text{Cs}$  in that area and to assess the committed effective dose. The Lund area was subjected to a total deposition of about  $2 \text{ kBq/m}^2$  of  $^{137}\text{Cs}$  from the nuclear weapons fallout during the 1950s and 60s and an additional deposition of  $2 \text{ kBq/m}^2$  of  $^{137}\text{Cs}$  from Chernobyl in May 1986. The effective ecological half-time for Chernobyl  $^{137}\text{Cs}$  was found to be  $1.8 \pm 0.2$  years, which would correspond to an approximate average individual committed effective dose of  $0.033 \text{ mSv}$ . The time pattern of pre-Chernobyl  $^{137}\text{Cs}$  was however best described by a double exponential function, with a short term effective ecological half-time of 1.3 years between 1965 and 1970, and a corresponding long term component of about 10 to 20 years. The committed internal effective dose to an average individual from pre-Chernobyl  $^{137}\text{Cs}$  in the Lund reference group was calculated to be  $0.20 \text{ mSv}$ . The aggregate transfer factor,  $T_{\text{ag}}$ , was estimated to be  $2.5 \text{ Bq kg}^{-1} / \text{kBq m}^{-2}$ , compared to a value of  $10 \text{ Bq kg}^{-1} / \text{kBq m}^{-2}$  in 1965 at the time when peak activity concentration values were observed in the Lund reference group. An alternative way of describing the aggregate transfer of  $^{137}\text{Cs}$  to man, is by time integration of both the average whole-body burden and the deposition level during a certain time span. Integrating over a 50 y time span, it was found that pre-Chernobyl radiocaesium was transferred to man at least ten times more efficiently than Chernobyl caesium in the Lund region. Since the global fall-out in the sixties and seventies was more or less continuously deposited during the years, the uptake of the substance in agricultural crops during the growth season led to a more efficient transfer of  $^{137}\text{Cs}$  through foodstuff to man than the deposition from Chernobyl, which occurred just prior to the beginning of the growth season.

## INTRODUCTION

THE AIM of this study was to investigate the time variation of the whole-body burden levels of  $^{137}\text{Cs}$  in a south Swedish population after the Chernobyl fallout in 1986 (Rääf *et al.*, 1998; Rääf *et al.*, 1999) and to compare the results with data obtained from the same reference group during the 1960s and 70s (Lidén, 1961; Bengtsson, 1967; Lidén and Gustafsson, 1967), as well as with contemporary studies on other Swedish populations. Radiocaesium from the nuclear weapons fallout consisted almost solely of  $^{137}\text{Cs}$  (e.g. Cigna *et al.*, 1971) and it was estimated that the intake of dairy and beef products accounted for a large part of the  $^{137}\text{Cs}$  intake in the Lund reference group (Bengtsson, 1967), which is in accordance with international observations (Fredriksson *et al.*, 1966). Between 1960 and 1980 the cumulated deposition of fallout  $^{137}\text{Cs}$  from atmospheric bomb tests was about  $2 \text{ kBq m}^{-2}$  in the Lund region (Mattsson, 1975; Isaksson, 1997), which is in reasonable agreement with results from similar latitudes in the northern hemisphere (UNSCEAR, 1977), except for areas with very high precipitation (Ågren, 1998). The ground deposition levels from the Chernobyl fallout in the Lund region was typically  $2 \text{ kBq m}^{-2}$  (Isaksson, 1997), which is low in comparison with most other regions in Sweden (Ågren, 1998). Furthermore the aggregate transfer of caesium from soil deposition to man through the ingestion of contaminated foodstuffs was calculated



based on detailed deposition data in the region (Isaksson, 1997) in combination with the results of *in vivo* concentration in the control group.

## METHODS

The Lund reference group was formed in 1960 (Lidén, 1961; Bengtsson, 1966), and originally consisted of a small group of some 3 to 17 individuals who were subject to *in vivo* determinations of the body burden of  $^{137}\text{Cs}$  and  $^{40}\text{K}$  from 1960 to 1963. At the beginning of 1964, the group was enlarged to 34 urban inhabitants of Lund and its surroundings, of which a majority worked in one specific factory in Lund (Lidén and Gustafsson, 1966). Many of these 34 subjects remained in the group and were investigated at least once a year between 1964 and 1980 (except for 1969 and 1972) with a whole-body counter at the Department of Radiation Physics in Lund, and between 1987 and 1994 at the Department of Radiation Physics in Malmö. The duration of the period of investigation is thus longer than 30 years and consist of 34 different measuring occasions. The composition of the group by age and gender is given in Table 1. The experimental design of the study between 1987 and 1994 has been described in Rääf *et al.*, 1998. In this evaluation of data, the *effective ecological half-time*,  $T_{1/2,eff}$ , in the Lund reference group, was determined from the time dependence of the whole-body content values based on  $^{137}\text{Cs}$  determination in all the subjects in the control group.

**Table 1**

Composition of the Lund Reference Group by age [years] and gender at the beginning and end of the two study periods.

Period		No. of participants			Age (Mean $\pm$ 1 SD)		
		Females	Males	Both	Females	Males	Both
Pre-Chernobyl	Beginning:						
	1964-02	14	20	34	27.8 $\pm$ 12.0	41.2 $\pm$ 8.5	34.4 $\pm$ 12.4
	End:						
	1980-06	9	14	23	42.0 $\pm$ 13.7	60.2 $\pm$ 5.8	51.6 $\pm$ 13.9
Post-Chernobyl	Beginning:						
	1987-05	10	14	24	47.9 $\pm$ 14.1	63.4 $\pm$ 7.5	56.0 $\pm$ 13.5
	End:						
	1994-06	8	9	17	56.1 $\pm$ 13.6	67.3 $\pm$ 5.6	60.9 $\pm$ 12.2

The long-term transfer of caesium into man can be described by using the ratio between the time-integrated activity concentration, based on the mathematical fit to the measured *in vivo*  $^{137}\text{Cs}$  concentration in the control group,  $a_{pop}(t)$ , divided by the time-integrated number of disintegrations per  $\text{m}^2$  of  $^{137}\text{Cs}$  deposited on ground,  $A_{dep}(t)$ , from a specific fallout event. That is (Eq. 1):

$$\bar{T}_{ag} = \frac{\int_{t_0}^{t_1} \bar{a}_{pop,fit}(t) dt}{\int_{t_0}^{t_1} A_{dep}(t) \cdot e^{-\lambda_{phys} \cdot t} dt} \quad (1)$$

or (Eq. 2)

$$T_{ag} = \frac{\text{No. of disintegrations per kg body weight of a given radionuclide that will occur in individuals due to a single event or continuous deposition on a certain geographical area}}{\text{No. of disintegrations of deposited activity per m}^2 \text{ during a given time period from a single event or continuous deposition in that geographical area}} \quad (2)$$

where  $\lambda_{phys}$  is the physical disintegration constant ( $=\ln 2/T_{1/2,phys}$ ) for a given radionuclide and  $A_{dep}(t)$  [ $\text{Bq m}^{-2} \text{ y}^{-1}$ ] is the annual deposition. A similar procedure as Eq. 1 has previously been suggested by UNSCEAR, 1977. The principal difference between the UNSCEAR procedure and that used in this work, is that the annual deposition rate is here transformed into the cumulated activity on the ground by convoluting the annual deposition with the physical disintegration of the deposited activity,  $\lambda_{phys}$ , in order to obtain the cumulated number of disintegrations per  $\text{m}^2$  of soil during a given period of time. The fraction of the substance that is available to root uptake by crops and thus to the long-term transfer through the ecosystem into man, is partly related to the cumulated number of disintegrations in the upper layers of soil. The dimension of the time-aggregated transfer factor then becomes [ $(\text{Bq kg}^{-1})/(\text{kBq m}^{-2})$ ]. The advantage of this somewhat more cumbersome procedure is that the time pattern of the fallout rate is taken into account, which is useful when comparing two such different fallout patterns as the fallout from the nuclear weapons tests and Chernobyl.

## RESULTS

### Activity concentration of $^{137}\text{Cs}$ and $^{134}\text{Cs}$

Whole-body concentrations of pre-Chernobyl  $^{137}\text{Cs}$  in the Lund reference group reached their peak values of about  $12 \text{ Bq kg}^{-1}$  at the beginning of 1965, after which an exponential decline was observed with a half-time of  $1.3 \pm 0.2$  (1 SE) y. A slower decline was observed at the end of the 1960s and the 1970s (Table 2). Using annual deposition data from Ljungbyhed, 43 km north of the city of Lund (Isaksson, 1997), it was observed that the *in vivo* levels of  $^{137}\text{Cs}$  in the Lund reference group exhibited a one year delay in the time-pattern relative to that of the deposition in the Lund region.

The mean concentration of Chernobyl  $^{137}\text{Cs}$  *in vivo* in the Lund reference group reached peak values in spring 1987 of ( $\pm 1$  SE)  $3.7 \pm 0.7 \text{ Bq kg}^{-1}$  for  $^{137}\text{Cs}$  and  $1.5 \pm 0.3 \text{ Bq kg}^{-1}$  for  $^{134}\text{Cs}$  as average values for both sexes.

The corresponding ecological half-time for  $^{137}\text{Cs}$  during the pre- and post-Chernobyl study periods are given in Table 2.

**Table 2**

$T_{1/2,eff,conc}$  [years] for  $^{137}\text{Cs}$  in the LRG ( $\text{MV} \pm 1$  SE) for the two study periods.

Study period	Pre-Chernobyl		Post-Chernobyl
	1965 – 1980		1987– 1994
	Short-term component	Long-term component	
Females	$1.2 \pm 0.3$	$(13 \pm 20)$	$1.8 \pm 0.2$
Males	$1.3 \pm 0.3$	$(27 \pm 130)$	$1.8 \pm 0.1$
Both	$1.3 \pm 0.2$	$(18 \pm 50)$	$1.8 \pm 0.2$

### *Time-integrated aggregate transfer factor of $^{137}\text{Cs}$ in the Lund reference group*

The time-integrated aggregate transfer of  $^{137}\text{Cs}$  during the pre- and post-Chernobyl study periods are given in Table 3.

**Table 3**

Time-integrated aggregate transfer factor in [ $\text{Bq kg}^{-1} / \text{kBq m}^{-2}$ ] for  $^{137}\text{Cs}$  based on prognosis from fitted data of the mean *in vivo* concentration in the LRG and deposition data from Isaksson.

Period	Females	Males	Both
Pre-Chernobyl (1960-1981)	3.4	4.9	4.4
Post-Chernobyl (1986 – 2007)	0.39	0.36	0.38

## CONCLUSIONS

The large difference in the ecological behaviour of nuclear weapon's fallout and Chernobyl  $^{137}\text{Cs}$  found in this study is due to several reasons. The continuous nature of the pre-Chernobyl fallout resulted in significant contamination on growing crops, whereas the Chernobyl deposition took place just prior to the Swedish growing season. Other factors, such as the difference between the chemical properties of the pre-Chernobyl and Chernobyl  $^{137}\text{Cs}$ , could have importance in the fixation of the radionuclide in the soil and the root uptake rate by growing crops. Furthermore, human behaviour, in terms of countermeasures taken after the Chernobyl accident, might have led to greater consumer awareness of contaminated foodstuffs, which in turn could have reduced the total transfer rate of  $^{137}\text{Cs}$  from foodstuff to normal consumers.

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# Caesium-137 in Saamis and reindeer in Kautokeino, Norway, 1965-1999

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## INTRODUCTION

In the early 1960ies, during the period of the atmospheric nuclear weapons testing, it was realised that people consuming reindeer meat contained higher concentrations of radioactive caesium (i.e.,  $^{137}\text{Cs}$ ) than the rest of the population of Northern areas (Lidén, 1961). This is because reindeer graze significant amounts of lichen during winter (Gaare and Staaland, 1994), and lichen absorb and accumulate nutrients and contaminants directly from air and precipitation. In 1965, as a response to this knowledge, Norwegian authorities started a monitoring programme among reindeer herding Saamis in Kautokeino (69°N, 23°E), a 9687 km<sup>2</sup> municipality of Finnmark county where reindeer herding is an important way of living (Westerlund *et al.*, 1967). The aim of the programme was to assess internal doses to the group of the Norwegian population that was assumed to be subject to the highest exposures from  $^{137}\text{Cs}$  from the nuclear testing in general, and the testing at the Soviet test site at Novaya Zemlya in particular.

## MATERIALS AND METHODS

### Whole body monitoring of people

From the beginning in 1965 the monitoring of the reindeer herders and their families have involved a relatively stable groups of persons. Changes and additions of new subjects have been done to compensate for deaths and increasing age, as the group was supposed to be representative of all the reindeer herders and their families in Kautokeino. The number of investigated persons has most years been 40-60, with maximum participation of 92 persons in 1996 and minimum of 21 and 25 in 1965 and 1999, respectively. In 1996 invitations were distributed to recruit new person (Skuterud *et al.*, 1999). Every year the monitoring has taken place in end March or early April, a time of year which in 1970-71 was shown to give results approximately equal to the annual mean concentration (Berteig *et al.*, 1971).

Until 1993 the simplified whole body monitoring procedure by Palmer (1966) was applied, using NaI(Tl) detectors, as described by Westerlund *et al.* (1987). The total uncertainty in individual measurements using this geometry has been assessed to be about 25 %. In 1993 a 3"x3" NaI(Tl) connected to a multichannel analyser in a lead shielded chair geometry was used. The uncertainty of these measurements was estimated to 5-15 % (Andersson Sørli *et al.*, 1994). For the investigation in 1996 and 1999 the transportable whole body counter from the Swedish Defence Research Establishment (FOA) was used. This applies one HPGe detector in a modified chair geometry where the persons lie in a «cradle», as described by Ågren *et al.* (1996). No estimates of other uncertainty than counting errors is available for this setup. In 1999 a counting time of 900 s was chosen, giving an average error of 9 %.

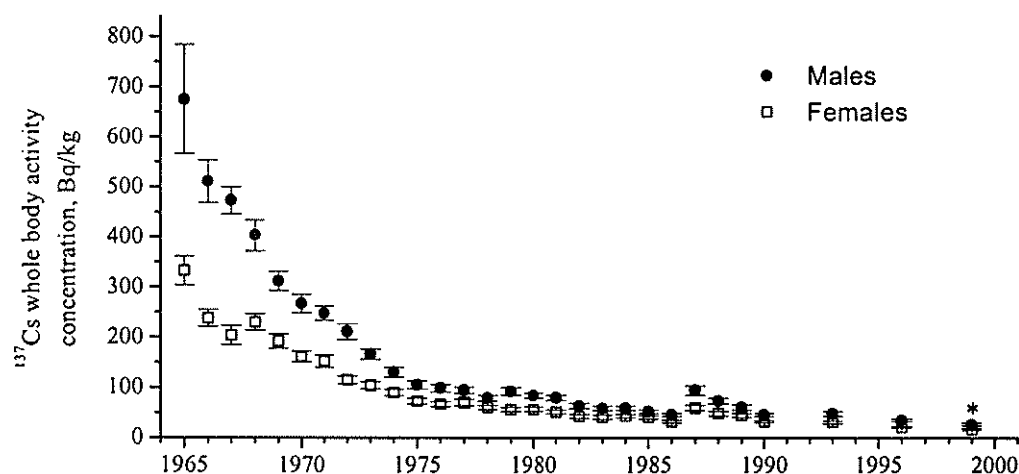
### Reindeer meat samples

Since 1967 the studied persons have been encouraged to bring a sample of reindeer meat, preferably from the winter grazing period. Most years 30-45 samples have been obtained. The maximum number is 48 (in 1981) and minimum 4 (1993). The radiocaesium activity in these samples have been analysed using 3"x3" NaI(Tl) detectors.

## RESULTS AND CONCLUSIONS

### Long term development in $^{137}\text{Cs}$ activity concentrations of humans and reindeer

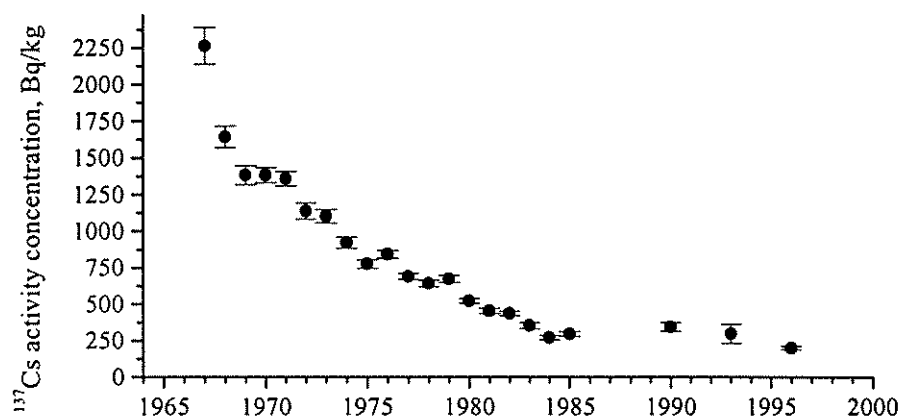
Many of the persons measured have been participating at several of the monitoring occasions, and one person has been monitored 27 times. A summary of the whole body monitoring results are presented in Fig.1. The average activity concentration in 1999 was  $16.9 \pm 1.9$  Bq/kg for females and  $26.7 \pm 2.9$  Bq/kg for males. Applying a dose conversion factor of  $2.5 \mu\text{Sv/y}$  per Bq/kg (UNSCEAR, 1988) this corresponds to an average effective dose of about  $0.055 \text{ mSv/y}$ . Comparing the results of 1999 with those in 1996 show that the male participants in 1999 in 1996 had an average whole body content significantly higher than the whole male group. Thus the results in 1999 are not directly comparable to those in 1996, and they are therefore not applied in further calculations at present.



**Figure 1.** Caesium-137 activity concentrations in reindeer herding Saamis from 1965 to 1999, mean  $\pm$  standard error. The asterisk for 1999 indicates that results for this year are not comparable to 1996, see text.

The  $^{137}\text{Cs}$  activity concentrations in reindeer meat are summarised in Fig.2. Samples from 1999 have not yet been analysed. The average activity concentration in 1996 was  $199 \pm 12$  Bq/kg. Comparing the time development in this figure with that in Fig.1 illustrates the obvious connection that exist between the concentrations in the people and one of their main dietary items.

From the knowledge about the fallout from the nuclear weapons testing, and estimates based on measurements and modelling, it was concluded that the activity concentration peaked in 1964-65 (Westerlund *et al.*, 1967; Westerlund *et al.*, 1987). This is in accordance with



**Figure 2.** Caesium-137 activity concentration in reindeer meat obtained from the people in Kautokeino, 1967-1996.

observations in Alaska and Canada (Hanson, 1982; Tracy *et al.*, 1997). Thereafter the concentration decreased until the mid 1970's, when the input from the 1973-74 Chinese testing caused a slower decrease and even increase in the activity concentrations. From 1979 onwards the concentrations decreased again, until the Chernobyl accident nearly doubled the concentrations in spring 1987 compared to one month before the accident in 1986.

Estimating effective ecological half-lives for  $^{137}\text{Cs}$  in people and reindeer in Kautokeino gives results as shown in Table 1. In these estimates the 1999 results are not included.

**Table 1.** Effective ecological half-lives ( $T_{\text{eff}}$ ) for  $^{137}\text{Cs}$  in people and reindeer, years. Estimates made on the basis of the average values each year.

Period	$T_{\text{eff}}$ people	Period	$T_{\text{eff}}$ reindeer
1979-86 <sup>1</sup>	7.9 (7.2-8.8) <sup>4</sup>	1979-85	4.8 (4.3-5.4)
1987-96 <sup>2</sup>	6.7 (5.7-8.2)		
1988-96 <sup>3</sup>	7.7 (6.3-9.8)	1990-96	7.7 (6.1-10.5)

<sup>1)</sup> The 1979-1986 period is the pre-Chernobyl period without significant direct concentration of vegetation and lichens

<sup>2)</sup> The 1987-1996 period is the whole period after the Chernobyl fallout

<sup>3)</sup> The 1988-1996 period is the time period after the Chernobyl fallout except the first year when direct contamination of plants grazed during the summer and autumn could have caused elevated concentration of  $^{137}\text{Cs}$ .

<sup>4)</sup> Range of minimum and maximum values in the standard error interval.

The estimates indicate that the  $^{137}\text{Cs}$  activity concentrations in people decrease with an effective ecological half-life of some less than 8 years when years with significant direct fallout are neglected. The reduction in the  $^{137}\text{Cs}$  concentrations in reindeer meat is approximately similar. This is in agreement with earlier estimates for Kautokeino reindeer of 7 years on the basis of the samples obtained from the early 1970's to 1983 (Westerlund *et al.*, 1987). No reason is known for the relatively rapid reduction in the concentration in reindeer meat during 1979-1985. A similar estimate (about 5 years) was also obtained for reindeer meat samples bought at the local slaughterhouse in the same period (Gjertsen, 1991).

Estimates on the basis of the monitoring of reindeer herders in Karesuando, Northern Sweden, indicated on effective ecological half-life of 5.9 years for the period 1991-1996 (Ågren *et al.*, 1996).

The data from Kautokeino and Karesuando indicates a slower decline in radiocaesium activity concentrations than found in more southerly reindeer grazing areas in Scandinavia. In Sweden values of 3-4 years have been estimated (Åhman and Åhman, 1994), and estimates in mid and southern Norway indicate 3-5 years (Amundsen, 1995; Hove *et al.*, 1999). The difference between these estimates and those for the more northerly areas might reflect differences in grazing areas, environments and climate. This shows that the application of genetic effective ecological half-lives is not straightforward even for relatively simple food chains, and that continued studies are needed to obtain knowledge of the long term trends relevant for radiological consequence analyses.

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# Fordeling og retensjon av radioaktivt Cs i naturlig jordsmonn i Norge

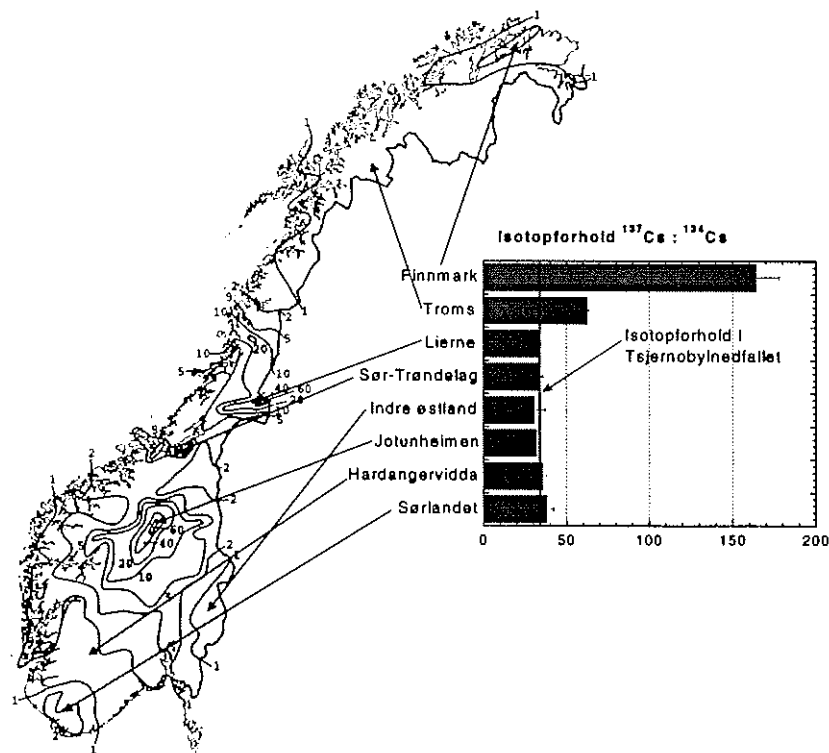
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## INNLEDNING

Stratosfæriske prøvespregninger av atomvåpen på 1950- og 1960-tallet og Tsjernobyl-ulykken (1986) har medført nedfall av menneskeskapte, radioaktive stoffer over hele Norge. Nedfallet har dels vært diffust fordelt over mange år og over hele den nordlige halvkule (bombe-nedfallet) og dels i form av ett enkelt regnskyll (Tsjernobyl-nedfallet). Avsatt på bakken er nedfallet utsatt for de klimatiske og andre påvirkninger som gjelder for stedet. Fordelingen av disse radioaktive stoffene vil derfor kunne variere i tid og rom avhengig av depositions-mønster, klima og geokjemiske egenskaper.



**Figur 1:** Fordeling av <sup>137</sup>Cs (kBq/m²) i Norge og isotopratioer for en del områder. Kartet er laget på basis av analyser av humusprøver tatt i 1995.



## METODER

De presenterte data er basert på analyser av 464 prøver av humus (0-3 cm dybde) tatt i områder med naturlig jordsmonn fordelt over hele Norge. Hver prøve er satt sammen av fire underprøver som alle ble tatt med bruk av prøvetakingssylinder ( $d = 10$  cm). Alle prøver ble tørket ved  $40\text{ }^{\circ}\text{C}$  og homogenisert etter fjerning av store planterøtter.

Aktiviteten av  $^{137}\text{Cs}$  og  $^{134}\text{Cs}$  i prøvene ble bestemt ved hjelp av høyoppløselig gamma-spektrometri. Normal telletid på våre Ge-detektorer (30 – 50 % effektivitet) var 1 til 8 timer. Cs-isotopforholdet ( $^{137}\text{Cs} : ^{134}\text{Cs}$ ) ble imidlertid bestemt for noen områder (fire prøver i hvert område, jfr. Fig. 1) ved bruk av tre-døgns-tellinger.

## RESULTATER OG DISKUSJON

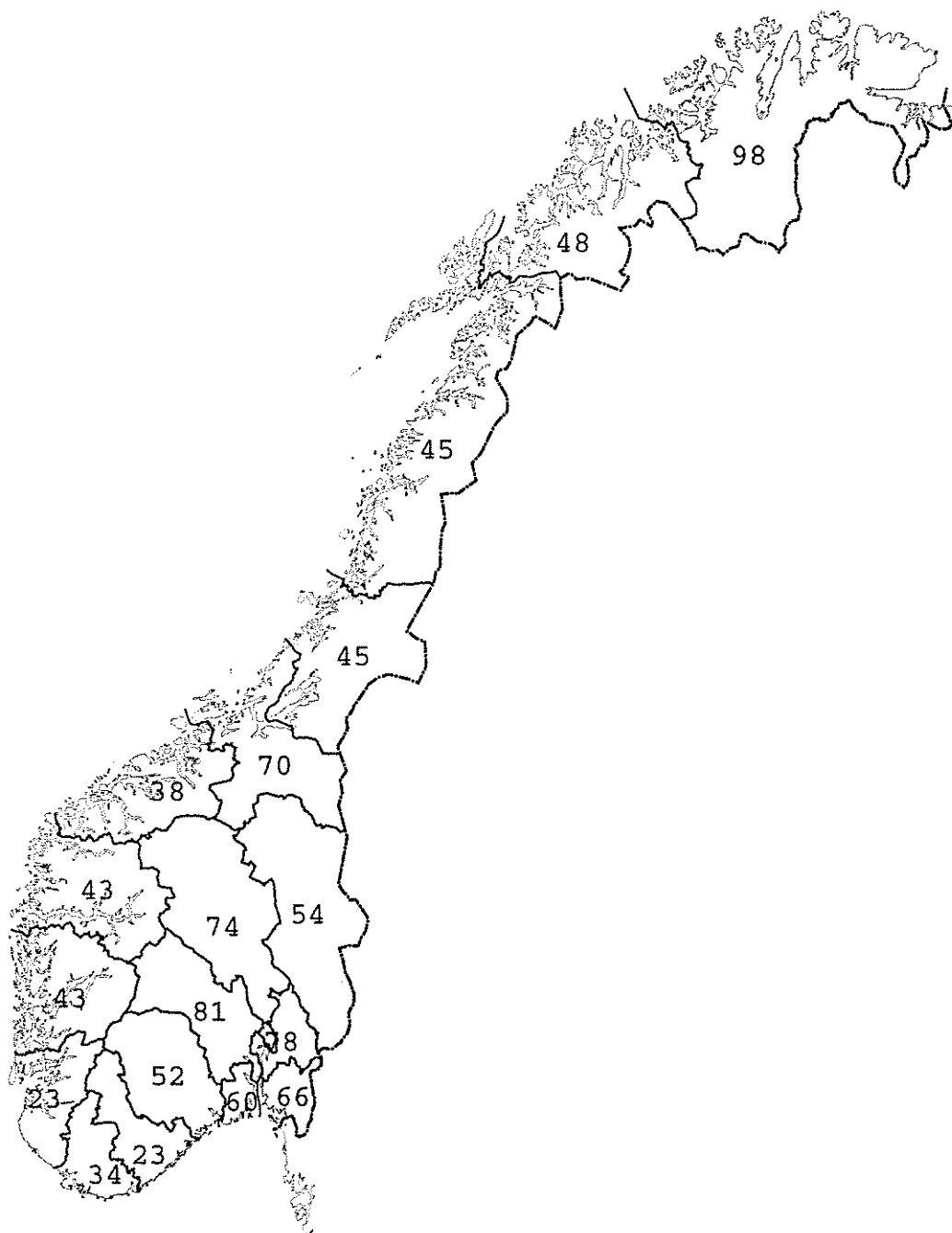
Fordelingen av  $^{137}\text{Cs}$  ( $\text{kBq/m}^2$ ) fremkommet i denne undersøkelsen er med få unntak i overensstemmelse med resultatene fra en undersøkelse gjort av Statens strålevern i 1986 bare to måneder etter Tsjernobylulykken (Backe et al. 1987). Det kan likevel poengteres at nedfallet fra Tsjernobylulykken ser ut til å være spredd over hele Norge (Fig. 1) med anomalier i Jotunheimen og det indre av Nord-Trøndelag ( $> 40\text{ kBq/m}^2$ ) i tillegg til noen områder med forhøyede verdier på Sørlandet, i Rogaland, i Sør-Trøndelag og Nordland. For alle disse ovennevnte områdene, i tillegg til "normalområdene" Troms, Indre østland og Hardangervidda, viser Cs-isotopforholdet (Fig. 1) at aktiviteten hovedsakelig skyldes nedfall fra Tsjernobylulykken.

Unntaket er et område med forhøyede verdier i Finnmark hvor isotopforholdet er så høyt at kontamineringen ikke kan skyldes Tsjernobylulykken alene. Det kan spekuleres i om dette kan skyldes direkte nedfall fra en prøvespregning på eller ved Novaja Zemlya.

Retensjon av  $^{137}\text{Cs}$  er beregnet fylkesvis basert på en sammenligning mellom våre data (fra 1995) og Statens stråleverns data fra 1986 (Backe et al. 1987). Prøvene i datasettet fra 1995 ble tilordnet fylker og deretter ble gjennomsnittsaktivitetet beregnet for hvert fylke etter tidsjustering av aktivitetene tilbake til 1. juli 1986. Dataene fra Statens strålevern var tatt kommunevis. Gjennomsnittverdier for hvert fylke ble derfor beregnet som gjennomsnitt av verdiene for hver kommune innen hvert fylke. Retensjonen ble så beregnet som kvotienten mellom aktivitetene i 1995 og 1986 uttrykt i %.

De fylkesvise data for retensjon av  $^{137}\text{Cs}$  (Figur 2) viser systematiske geografiske forskjeller med lavest retensjon i fylker som grenser til havet. Dette kan forklares med tilsvarende geografiske forskjeller i mengde og sammensetning av nedbøren i Norge (Berg og Steinnes

1997). Typisk for kystnære områder i Norge er store nedbørsmengder med høyt innhold av marint salt, som resulterer i stor gjennomstrømning i jordprofilen og i tillegg utvasking av kationer fra jordsmonnet (Låg 1968). Den spesielt lave retensjonen i de tre sydligste fylkene (Rogaland, Aust og Vest Agder) kan ha sammenheng med at det er disse fylkene som også er mest utsatt for langtransportert sur nedbør.



**Figur 1:** Retensjon av  $^{137}\text{Cs}$  (%) basert på gjennomsnittsverdier beregnet fylkesvis. Tallene angir andelen av  $^{137}\text{Cs}$  gjenfunnet i humussjiktet (0-3 cm) i forhold til det som ble deponert i 1986. Deposisjonsdata fra Backe et al., Rad. Prot. Dosimetry 18, 1987.

For innlandsområder, det er også de områdene som fikk mest nedfall fra Tsjernobyl, ser det ut til at mellom halvparten og tre fjerdedeler av  $^{137}\text{Cs}$ -deposisjonen fra Tsjernobyl fremdeles befinner seg i de øverste tre cm av jordsmonnet.

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# **Risk for Haematological Malignancies after Radiation Treatment of Painful Benign Conditions in the Skeleton**

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## Abstract

X-ray therapy of painful and inflammatory changes in joints and adjacent structures was very common in Sweden during the period from 1940 until the beginning of the 1960:s. These patients comprise a comprehensive material pertinent for epidemiological studies. The present study deals with a cohort of c. 27,400 patients from 3 hospitals in northern Sweden, who 1950 - 1964 received x-ray treatment for benign painful conditions. The distribution of the absorbed dose in the red marrow was estimated.

The average mean absorbed dose in red bone marrow was 0.4 Gy for the total cohort. The number of leukaemia cases observed in the cohort was obtained from the Swedish Cancer Register for the period 1958 - 1995. The study indicates a slightly increased leukaemia risk with borderline statistical significance in the highest dose group.

## Introduction

The present knowledge of leukemogenic and carcinogenic effects of ionising radiation in man derives from epidemiological studies. The most studied population is the Japanese A-bomb survivors, which have been followed-up for almost 50 years. However several studies on populations exposed to different types of ionising radiation for medical purposes have supplied supplementary knowledge. Concerning risk for radiation induced leukaemia in adults the so far most important of those is the British study of 14000 patients receiving X-ray treatment of ankylosing spondylitis. In this study a significant risk of leukaemia was shown (Court Brown and Abbatt 1955; Court Brown and Doll 1965).

In Sweden x-ray therapy of painful and inflammatory changes in joints and adjacent structures was very common during the period from 1940 until the beginning of the 1960:s. These patients comprise a comprehensive material pertinent for epidemiological studies. We have analysed a cohort of such patients from Skellefteå (n= 7419), Umeå (7529) and Gävle (12466). Results encompassing only patients from Umeå and Gävle were published in 1995 (Damber et al. 1995; Johansson et al. 1995).

### Material and Methods

The present study deals with patients who 1950 - 1964 received x-ray treatment for benign painful conditions in the locomotor system. The endpoints in the present study were haematological malignancies. The age distribution of the cohort is described in Figure 1, and the number of persons treated per year during is illustrated in Figure 2.

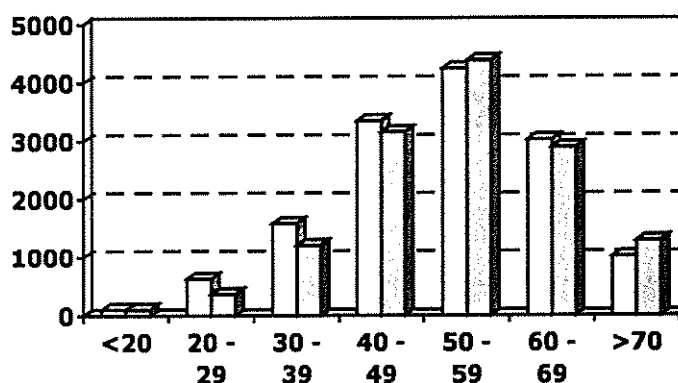


Figure 1. Age and sex-distribution of the cohort. Light grey: men, dark grey: women.

Data included on the treatment cards were entered in a computer database. These data was used to obtain individual estimations of the mean absorbed dose in the red marrow. A computer linkage was set up with the Swedish Cancer Register and the Swedish Cause of Death Register. For the statistical analysis the entire material was stratified with regard to mean absorbed red marrow dose into three classes; 0 - 0.2 Gy, 0.2 - 0.5 Gy, and > 0.5 Gy.

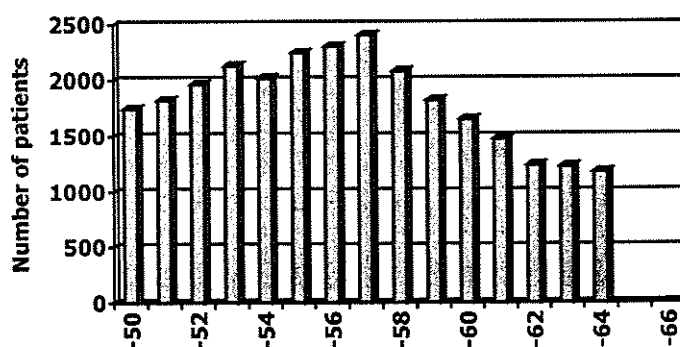


Figure 2. Distribution of year of first treatment in the cohort.

For the analysis of the distribution of the mean red bone marrow dose in the cohort, the treatments were grouped into ten categories according to the irradiation site. For six of those sites a fraction of the red bone marrow was located in the direct beam. For the remaining four sites, encompassing the distal parts of the extremities the red bone marrow dose was set to zero.

### *Dosimetry*

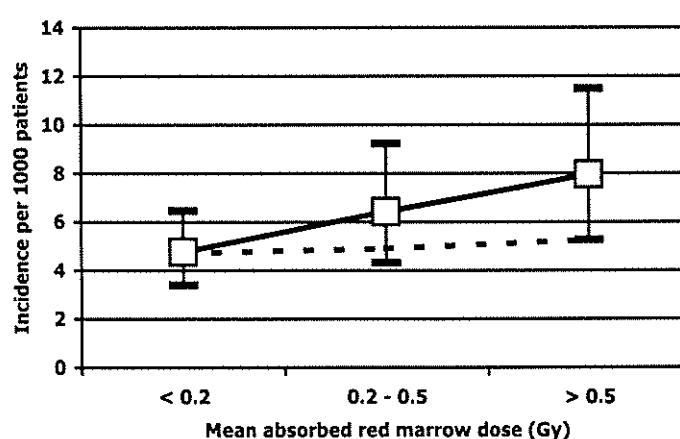
In order to estimate the mean absorbed dose in the red marrow an "average conversion factor," converting the surface dose, given in the treatment card, to mean red marrow dose, were assessed. Such a factor was derived for each of the six treatment sites with red marrow included in the direct beam. For this reason individual dose estimations were performed for 30 randomly chosen patients for each site.

The local absorbed dose to the red bone marrow was estimated using data from Ellis et al. (1976). Parameters such as HVL, FSD and field size were considered. The distribution of the red marrow within the body was taken from Cristy (1981), and these data were the basis for the estimation of the fractional red bone marrow included in the direct beam

The average mean absorbed dose in red bone marrow was 0.4 Gy for the total cohort. The number of leukaemia cases observed in the cohort was obtained from the Swedish Cancer Register for the period 1958 - 1995. The expected number was calculated using calendar and age-specific incidence rates for the relevant counties. A more comprehensive description of the dosimetric methods has been published elsewhere (Johansson et al. 1995)

### Results and Discussion

The standardised incidence ratio (SIR) for leukaemia was for the three dose levels: < 0.2 Gy 1.01 (95% CI 0.72-1.37); 0.2-0.5 Gy 1.31 (0.88-1.88); and > 0.5 Gy 1.51 (1.00-2.18). The study thus indicates a slightly increased leukaemia risk with borderline statistical significance in the highest dose group. The cohort is in a way similar to the British series of X-ray treated persons with ankylosing spondylitis but considerably lower doses were used and smaller red bone marrow volumes were exposed.



*Figure 3. Observed (solid line) and expected (dotted line) incidence of leukaemia per 1000 patients. Variations in expected rate is due to differences in age distribution.*

In ICRP 60 (ICRP 1991) a probability of lethal cancer is estimated to 0.40% Sv<sup>-1</sup> in a population of workers. Applying this risk figure on a population of 1000 individuals a mean absorbed red marrow dose of 0.1 Gy leads to 4 cases of fatal leukaemia, 0.3 Gy to 12 and 1.0 Gy to 40 cases. These figures may be compared to what can be derived from the present study (Figure 3). As can be seen the increase in number of observed cases with increasing red marrow dose is considerably lower than what can be expected from data in the ICRP 60. There may be several reasons for this difference, but an important one may be the age-distribution of the cohort, which significantly deviates from that of a normal working population. The older ages, for which the risk is smaller are over-represented in the cohort.

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# Ländryggsundersökningar - varför varierar stråldosen så mycket?

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## **Bakgrund**

Under det senaste årtiondet lades mycket arbete ner på att optimera röntgenundersökningar. Det ledde bl. a. till införandet av ett nytt koncept - diagnostiska referensnivåer - och återspeglas i all tydlighet i det nya direktivet om medicinska bestrålningar (MED97). Tidigare kartläggningar av stråldoser vid röntgenundersökningar har visat på stora skillnader för en och samma typ av undersökning, ett tecken på att optimeringstanken inte har slagit igenom. Orsaken till en stor spridning i stråldos kan ligga i nonchalans, okunskap eller kan kanske vara av mera inherent fundamental natur. Strategin om hur man kan förmå röntgenavdelningarna att optimera mera måste utgå ifrån orsakerna till dosspridningen. För att få underlag för detta samt för införandet av diagnostiska referensnivåer genomförde strålskyddsinstitutet en studie av ländryggsundersökningar, med detaljerade analyser som inkluderade både patient- och fantomexponeringar.

## **Material och metoder**

Grundidén i studien var att i samband med mätning av patientdoser registrera/mäta alla andra relevanta parametrar, och att komplettera dessa med fantommätningar. Fantomet - ett NRT standardtestfantom av PMMA, 20 cm tjockt och med en bildtestplatta innehållande diverse testobjekt - exponerades med samma driftbetingelser som för en AP ländryggsundersökning av en normalstor patient. Ingångs- och utträdesdosen till fantomet mättes liksom dos till bildmottagaren. Kerma-area-produkten (KAP-värdet) mättes med en KAP-mätare för ett strålfält av standardstorlek. Bilder togs av fantomet, och mätning av halvvärdesskikt samt kontroll av inställd rörspänning kompletterade fantommätningarna.

Vid varje röntgenstativ mättes patientdoser för ca 30 patienter som genomgick en komplett ländryggsundersökning. Patientens ålder, kön, längd och vikt registrerades, och för varje projektion rörspänning, rörladdning, KAP-värdet separat för genomlysning/exponering, användning av kompression och av exponeringsautomatik. Därutöver inventerades den utrustning som användes som t ex röntgengenerator, rastertyp, typ av bildregistrerande system.

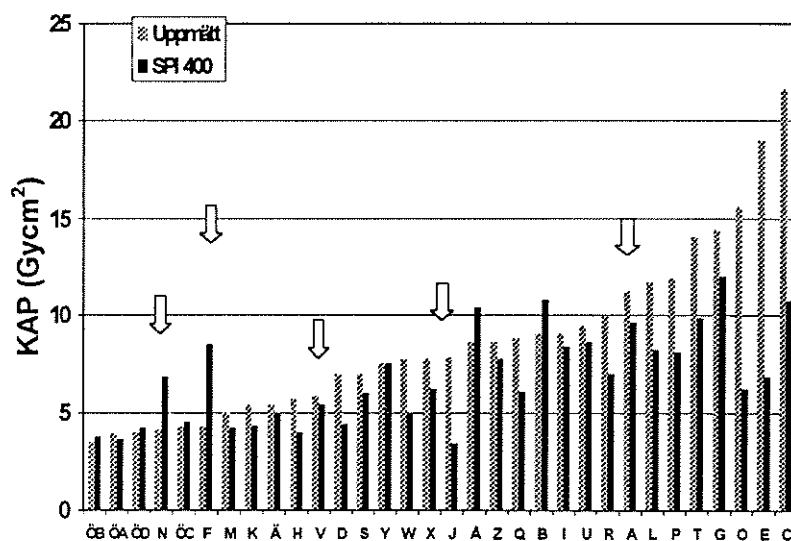
## **Resultat**

Från fantommätningarna beräknades ingångsdos och KAP-värdet för en normalexponering (dvs. motsvarande en AP/PA ländryggsundersökning för en normalpatient) och dito för en bild med nettosvärtning 1,0. Känsligheten för det bildregistrerande systemet beräknades, och attenueringen av strålningen mellan utgången av fantomet och bildmottagaren bestämdes.

Medelvärden för det totala KAP-värdet från de olika röntgenstativen återges i diagram 1. Värdena varierar inom en faktor 6. För att utröna orsaken till denna spridning har värdena normerats: till känslighetsindex (Speed Index) SPI = 400 (Diagram 1) och till både SPI = 400 och 4 projektioner (diagram 2).

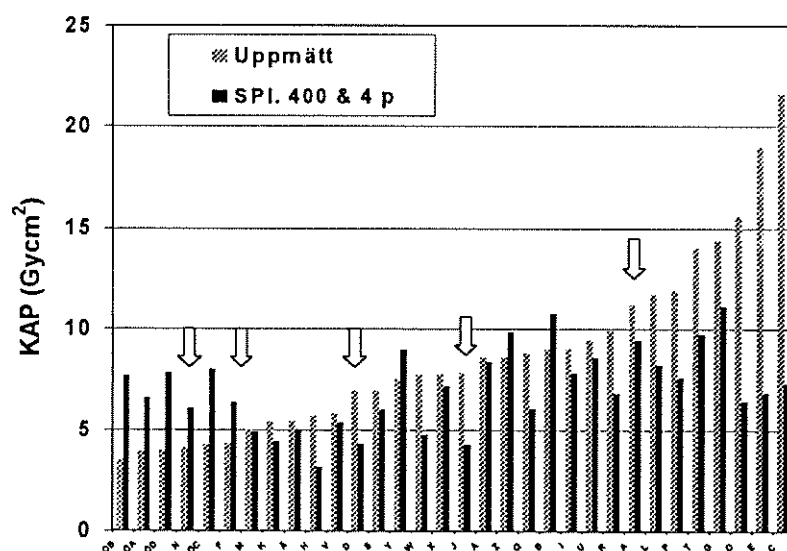


**Diagram 1:** Uppmätta KAP-värden och dito normerade till känslighetsklass (SPI) 400



↓ Indikerar utrustningar med bildplattor/bildförstärkare. Känsligheten för dessa system är ej definierad – för att kunna göra någon jämförelse postulerades en nominell känslighet som det reciproka värdet av dos till bildmottagaren vid normalexponering (kliniska exponeringsdata) med fantom.

**Diagram 2:** Uppmätta KAP-värden och dito normerade till SPI = 400 och 4 projektioner.



Normering till 4 projektioner skedde på följande sätt: Dosbidraget från omtag, är inte kompletteringar, genomlysning togs bort. Där en eller två projektioner inte genomfördes (sakrum AP/PA eller/och sakrum LAT) approximerades den som dosen för motsvarande ländryggsprojektion.

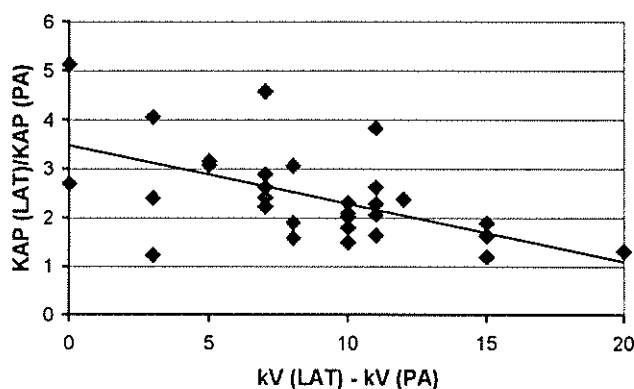
En rad andra parametrar som påverkar stråldosen till patienten har identifierats. Grupperade i kategorier och med angivelse av hur mycket stråldosen påverkas, uttryckt som kvot mellan högsta och lägsta stråldos när parametern i fråga varierar mellan sina extremvärden ges i tabell 1.

**Tabell 1:** Parameter som påverkar stråldosen - sammanfattning.

Variationen av parametrarnas värde beror på		
Utrustning	Val av inställning	Metod
Känslighet av bildmottagare*: 3 (8)	Strålkvalitet: 2	Antal bilder: 3
Attenuering i bord/raster: 2	Fältstorlek: 2	Kompression: 2
	Relativ strålkvalitet (mellan PA och LAT projektion): 2	Genomlysning: 2

\* siffran i parentes gäller när även bildplattor/bildförstärkare inkluderas, se anm. till diagram 1

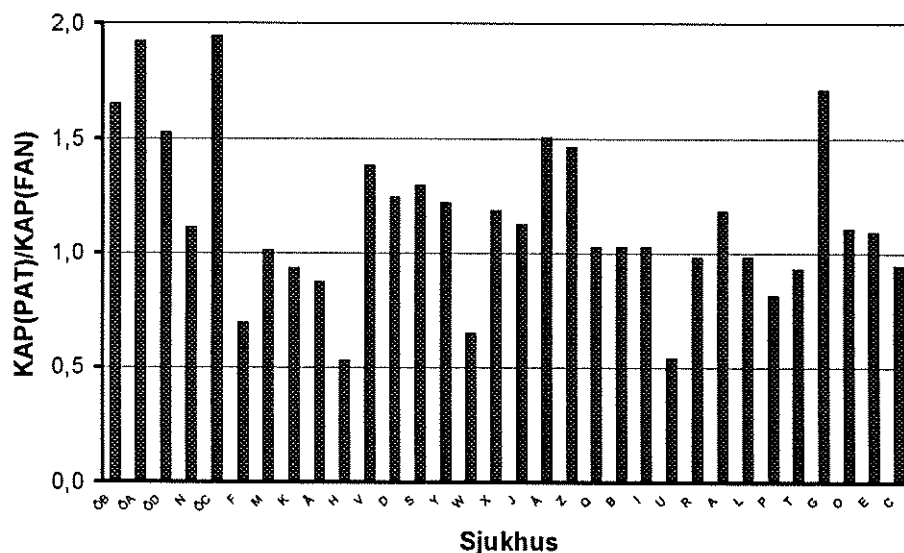
**Diagram 3:** Kvot mellan doser från LAT och PA projektionen som funktion av differensen av rörspänningen mellan dessa projektioner



Anpassning av strålkvaliteten till patienttjockleken är ett led i optimeringen. I diagram 3 illustreras hur detta påverkar stråldosen. En tydlig minskning av dosen för LAT projektionen relativt PA projektionen syns när differensen mellan rörspänningarna ökas, trots många fler faktors påverkan.

Fantommätningarna gjordes med samma inställning som för PA projektionen av en normalstor patient. Kvoten mellan rörladdningen från patientmätningarna (genomsnittet) och den från fantommätningen var 0,7 för anläggningarna som använder sig av exponeringsautomatik (AEC), ett tecken för att fantomet har större attenuering än normalpatienten. För att ta hänsyn till detta har i nedanstående diagram 4 KAP-värdena för fantommätningar multiplicerats med 0,7 för anläggningarna med AEC.

**Diagram 4:** Kvot mellan KAP-värden för patient och fantom (PA projektion)



## **Slutsatser**

Spridningen i patientdos är stor, vilket beror på en rad faktorer varav dock de flesta var för sig har en måttlig påverkan. De största enskilda bidragen är bildmottagarens känslighet och antalet projektioner för en komplett ländryggsundersökning. Om man som i diagram 2 normerar bort inflytandet av dessa två faktorer håller sig doserna inom en faktor 3, en relativt snäv fördelning med tanke på att minst 5 parametrar återstår som var och en påverkar dosen med en faktor 2. Åtgärder för att minska spridningen ytterligare måste inrikta sig på dessa återstående faktorer, vilket förutsätter kunskap om hur dessa påverkar stråldosen.

Den stora spridningen i patientdoserna visar tydligt att optimeringstanken inte har slagit genom än. Beror detta på slarv, nonchalans, okunskap eller ekonomiska skäl? Enligt författarnas uppfattning är det - naturligtvis - en blandning av alla. Dock verkar okunskap vara den dominerande faktorn - vid en del av anläggningarna med höga doser har på grundval av denna studiens resultat åtgärder vidtagits för att sänka dosen.

De Nordiska strålskyddsmyndigheterna har föreslagit 10 Gy cm<sup>2</sup> som den diagnostiska referensnivån för ländryggsundersökningar (NOR96). Detta verkar vara ett lämpligt värde för svenska förhållanden - 25 % av anläggningarna hade värden som överskred denna nivå, och det är i linje med hur referensnivåer bör fastställas. Normerat till känslighetsindex 400 och 4 projektioner hamnar endast 2 anläggningar knappt över 10 Gy cm<sup>2</sup>.

Den bristande korrelationen mellan dosen mätt med fantom och för AP ländryggsundersökning av patient - än sämre mellan fantom och den kompletta undersökningen - indikerar att fantommätning i sig har ett ringa värde som substitut för patientdosmätningar. Däremot är de värdefulla för att analysera de tekniska/fysikaliska faktorerna som påverkar stråldosen och på så sätt ge konkreta förslag om vilka förbättringar som bör göras och vilka effekter dessa kommer att ha. Med dylika studier kan man alltså både identifiera vilka anläggningar som ger höga stråldoser och vilka åtgärder som är lämpliga för att avhjälpa bristerna. Gott strålskydd innebär en systematisk genomgång av de många faktorer som påverkar bildtagningsprocessen med efterföljande optimering, och inte nödvändigtvis inköp av den modernaste utrustningen. Att engagera sig i optimering av både utrustning och arbetsmetoder är en av de viktigaste uppgifterna för myndigheten och initierade sjukhusfysiker.

## **Referenser**

MED97 Council directive 97/43/Euratom on health protection of individuals against the dangers of ionizing radiation in relation to medical exposure. Official Journal of the European Communities L 180, 22-27 (1997).

NOR96 Nordic Radiation Protection Authorities. Nordic Guidance Levels for Patient Doses in Diagnostic Radiology. Report on Nordic Radiation Protection Co-operation. No. 5 (Stockholm, SSI) (1996).

## **APOPTOSIS AND OTHER INDICATORS OF IONIZING RADIATION AT DOSES 0.1 – 2.0 GY IN FIVE CELL CULTURE SYSTEMS**

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### **ABSTRACT**

The most commonly used indicators of ionizing radiation exposure are cytogenetic measures and survival parameters. All these methods have their advantages, disadvantages and uncertainties, such that better biological estimators of the absorbed dose, especially in the low dose range, are being sought.

In this study we focus on apoptosis and several proteins involved in the regulation of apoptosis as possible indicators of irradiation after relatively small doses (0.1-2 Gy) of X rays. The studies were carried out in five lymphoid cell lines: two mouse lymphoma L5178Y, the human pre-B cell leukaemia Reh, and two human Epstein-Barr virus-transformed Ataxia-telangiectasia (AT) cell lines. We detected apoptosis with the in situ terminal deoxynucleotidyl transferase assay and flow cytometry, and measured the expression of several apoptotic-regulatory proteins (Bcl-2, Bax, Bclx) with Western blotting. The cytokinesis-block micronucleus assay and trypan blue test were also done for comparison.

Our results indicate that although there is an increase in radiation-induced apoptosis in all the cell lines examined by dose, there are marked differences in both the timing of apoptosis and the percentage of apoptotic cells, which do not always correlate with the above-mentioned parameters of radiation sensitivity. Moreover, when one compares different sets of experiments, the variation in the numbers of apoptotic cells in the controls is not very pronounced, but there is considerable variation for the same in the irradiated cells. These findings may be explained by the fact that the cell lines examined in this study were not synchronised, and that the number of cells prone to apoptosis after irradiation differed from experiment to experiment due to their cell cycle status. We did not find any association between the levels of expression of the Bcl-2 gene family and radiation sensitivity.

In conclusion, measurements of apoptosis and of apoptosis-related parameters can give supplementary information about radiation sensitivity, but it seems unlikely that they alone can be used as dose estimators.

### **INTRODUCTION**

In recent years it has been commonly accepted that apoptosis is the mode of death which is caused by biologically relevant doses of radiation in many, although not all cell types. In the cells in which apoptosis occurs both the extent and timing of apoptosis can differ dramatically. In many tumour cells apoptosis does not occur immediately following irradiation but after one or even multiple cell divisions (Muschel et al. 1998 and references herein). Although knowledge about the molecular mechanisms of apoptosis and interest in the possibilities of using apoptosis and proteins which are involved in this process as sensors or markers of radiation exposure has increased (Boreham et al. 1996, Olive et al. 1997), the overall complexity and heterogeneity of the apoptotic process in different cells may make it difficult to use it as a common irradiation sensor.

In this study we compare the occurrence of apoptosis and status of apoptotic-regulatory proteins (BCL-2, BAX, BCLx) with other radiosensitivity measures such as the micronucleus assay and trypan blue survival test for five different mammalian cell lines.

## METHODS

### Cell lines, culture conditions and irradiation

L5178Y-R and L5178Y-S, mouse lymphoma cell lines; Reh, a human pre-B leukaemia; and two human Ataxia telangiectasia(AT) Epstein-Barr virus transformed lymphoblasts; GM00736A, heterozygotic for the ATM gene, and GM00717C, homozygotic (Human Genetic Cell Repository, Camden, N.J.) were grown as suspension culture in RPMI 1640, with 10% FCS for L5178Y and Reh cells, and 15% for the AT cell lines at 37°C, 5% CO<sub>2</sub>. Exponentially growing cells were irradiated in 10 ml medium in T25 flasks with 0.1, 0.5, 1 and 2Gy of X-rays (250V, 12mA, 3mm Al, dose rate 1 Gy/min) at room temperature with the Philips MG 300 industrial x-ray unit.

### Micronucleus assay

Cytokinesis-block micronucleus assay was performed according to Fenech (Fenech 1990) with modifications for generation time, cytochalazyn B (cytB Sigma) concentrations, and mitotic delay. Briefly L5178Y-S cells were incubated with 1.5µg/ml cytB for 17-24h, L5178Y-R with 3µg/ml for 17-19h, Reh and AT with 3µg/ml cytB for 44h, and 48-50h for Reh and AT cells, respectively. For each cell line 2-3 replicate experiments were performed.

### Measurement of apoptosis with terminal deoxynucleotide transferase (TdT) assay.

The procedure was taken from Gorczyca et al. (1993) with slight modifications (Stokke et al.1998). To detect apoptosis the fixed cells were prepared for two colour fluorescence analysis using flow cytometry. The staining procedure included TdT, biotinylated dUTP and streptavidin - FITC to label free DNA ends resulting from apoptosis. The cells were also stained for DNA content with propidium iodide (PI). Red fluorescence (PI) and green fluorescence (FITC) were measured using a FACS Vantage flow cytometer, and instrument software was used to calculate the apoptotic fraction.

### Monitoring of cell survival

Viable and dead cell numbers were determined based on exclusion of trypan blue dye at intervals of 0, 24, 48h for all cell lines, and additionally 72h for AT. For assessment of antiproliferative activity, cell numbers were compared in control cells and irradiated cells after 48h for mice and 72h for human cell lines.

### Western blotting

For cellular extracts' preparation, cell cultures were incubated for 4 or 24 hours after irradiation and  $2-3 \times 10^7$  cells were used for one sample (method from Research Application, Santa Cruz Biotechnology, Inc.). Cell extracts were run on 12% SDS-PAGE using a Mini-PROTEAN II electrophoresis cell at 200V for 40 min. (Bio Rad). Blots of proteins on Immun-Blot PVDF membranes (Bio-Rad) were analysed for BCL2, BAX, BCLX<sub>S/L</sub> with rabbit polyclonal antibodies from Santa Cruz Biotechnology, Inc.: bcl-2(C21), bax (P-19) and bclx<sub>S/L</sub> (S-18) and with the rabbit polyclonal antibody for BCL-2 from Alexis (210-701-C100). Proteins were detected with the Bio-Rad Amplified Alkaline Phosphatase Immun-Blot Kit. Kaleidoscope prestained standards (Bio-Rad) were used as molecular weight standards.

## RESULTS

The cell counts derived from the Trypan blue test for these 5 cell lines illustrate the relatively high sensitivity to X-rays for LY-S and the AT cell lines, and less sensitivity for Reh and LY-R (Fig.1). As a measure of sensitivity to radiation, the Trypan blue viability test is useful.

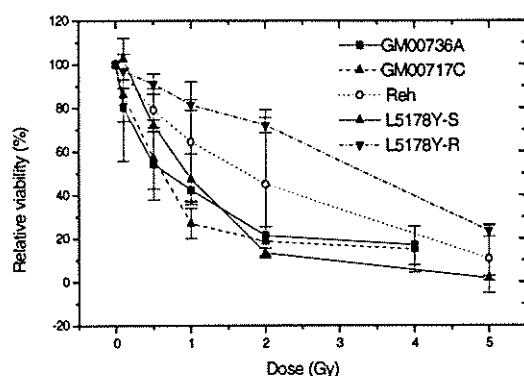


Fig.1. Trypan blue viability test for five cell lines irradiated with X-ray; each point represents data from 3 replicate experiments

Micronucleus studies (Table 1) in the small dose area do not show any significant differences between the lines for the doses  $\leq 1$  Gy, except that for the LY-S cell line it was not possible to perform the test for 1Gy due to high sensitivity to irradiation for this line (surviving fraction for 1 Gy is about 0.15). For 2 Gy the data may be underestimated for AT cell lines due to the very long G2 block in these lines for higher doses (in preparation).

Table 1. Frequency of micronuclei (MN) in five cell lines after X-ray irradiation with doses 0.1-2Gy; each value represents results from 2-3 replicate experiments. Numbers represent MN/1000 binucleate cells.

Dose (Gy)	L5178Y-S	L5178Y-R	Reh	G00736A	G00717C
0	11.5 $\pm$ 3.40	8.0 $\pm$ 4.6	11.8 $\pm$ 5.1	14.8 $\pm$ 3.6	13.6 $\pm$ 4.3
0.1	17.0 $\pm$ 6.80	24.5 $\pm$ 8.7	21.5 $\pm$ 6.8	30.9 $\pm$ 5.4	23.5 $\pm$ 2.8
0.2	23.3 $\pm$ 7.00				
0.3	38.7 $\pm$ 12.8				
0.4	41.3 $\pm$ 16.8				
0.5	42.7 $\pm$ 4,20	49.5 $\pm$ 7.9	49.0 $\pm$ 17.8	36.0 $\pm$ 10.8	56.3 $\pm$ 16.6
1.0		85.0 $\pm$ 6.2	77.5 $\pm$ 20.2	58,5 $\pm$ 17.1	68.0 $\pm$ 9.7
2.0		190.0 $\pm$ 0.3	116.7 $\pm$ 21.0	101.3 $\pm$ 23.5	124.8 $\pm$ 15.7

Apoptosis studies have shown that all five cell lines undergo delayed apoptosis, but the extent and timing of apoptosis differ considerably. (Table 2, Fig.2).

Table 2. Timing and extent of apoptosis as measured with the TdT assay in the panel of cell lines

Cell line	Time (hours)	Maximum of apoptosis observed at 2Gy (%) a
L5178Y-R	48	5,06
L5178Y-S	48	27,48
Reh	48	22,45
GM00717C	72	10,67
GM00736A	72	19,95

a – the highest value from 3 replicate experiments

Variation in the expression of apoptosis regulatory proteins BCL-2, BAX and BCLX<sub>L</sub> was detected in the controls; qualitative assessments of expression are presented in Table 3. We did not detect BCLX<sub>S</sub> in any of these lines. None of these proteins were induced 4h after irradiation in any of these lines, nor was there an increase of expression for any of these proteins relative to the controls.

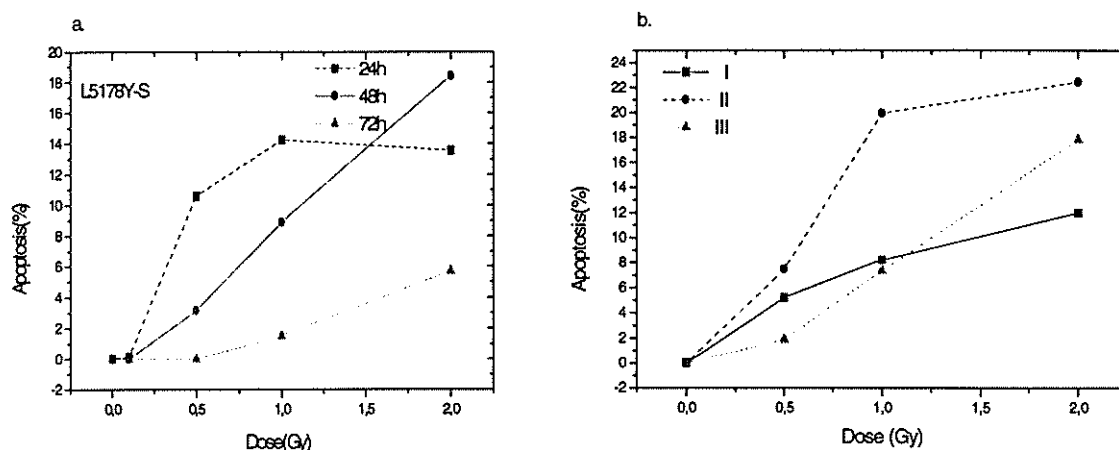


Fig 2. Radiation-induced apoptosis in L5178Y-S cells (most sensitive) at different times after irradiation (a), and variation in radiation-induced apoptosis at 48h after irradiation of Reh cells in three separate experiments (b).

Table 3. Relative amount of the apoptosis regulatory proteins in the panel of the cell lines

Cell line	Bcl-2	Bax	Bclx <sub>L</sub>
L5178Y-R	ND <sub>a</sub>	++++	++
L5178Y-S	+	++++	++
Reh	++++	+++	+
GM00717C	+	++	++
GM00736A	+	+	++

a - not detectable

## CONCLUSIONS

Although for the three examined parameters of radiation sensitivity-- micronucleus assay, trypan blue test and percentage of apoptosis-- there were observed clear dose-effect relationships for all cell lines examined, we did not find agreement between values for these measured parameters. There are marked differences in both timing of apoptosis and percentage of apoptotic cells. Variation in the apoptotic fraction in the controls for different sets of experiments is not very pronounced. There is however considerable variation for the same parameters in irradiated cells, possibly due to their cell cycle status during irradiation, as the cultures were not synchronised. Overall, neither the numbers of apoptotic cells nor the expression of apoptosis-related proteins can serve as dose estimators or sensors for these lines, but still these parameters can give valuable supplementary information about radiation sensitivity.

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# MOLECULAR DOSIMETRY BASED ON RADIATION INDUCED DEGRADATION OF POLYISOBUTYLENE

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## ABSTRACT

This project investigates the possibility of qualitative measurement of radiation doses through detection of changes in the average molecular weight in the polymer Polyisobutylene (PIB). Changes in molecular weight and molecular weight distribution is detected by Gel Permeation Chromatography (GPC). The aim of the project is to decide whether or not it is possible to determine a quality difference between  $\alpha$ -radiation ( $^{241}\text{Am}$ , 5,5 MeV) and  $\gamma$ -radiation ( $^{60}\text{Co}$ , 1,25 MeV) in the dose range 0,5 to 10 kGy by irradiation of PIB.

Irradiation with  $^{60}\text{Co}$  changes the average number molecular weight  $M_n$  by 12% per kGy and the average weight molecular weight  $M_w$  by 23% per kGy. The presence of antioxidant in the irradiated sample inhibits a change in average molecular weight by 5% and 16% for  $M_n$  and  $M_w$  respectively.

## INTRODUCTION

The biological damaging effects of ionizing radiation depends on the radiation's ability to induce molecular changes in the body's macromolecules - especially in DNA. Biological radiation damages arises as a consequence of direct action and indirect action. In the direct action the radiation interacts by excitation and ionization directly with the cell's macromolecules. In the indirect action the radiation interacts with the aqueous environment of the cell, whereby free radicals and poisonous chemical compounds are created, which in turn interact with the macromolecules of the cell. It is estimated that approximately 70% of the resulting damaged is due to the indirect action.

The purpose of this project is to find a method which in a direct manner is able to detect molecular change resulting from irradiation with ionizing radiation is sought after. Emphasis will be put on the investigation of the direct action.

In order to be able to simulate the damaging effects of ionizing radiation a good choice of material would be one where the molecules resemble the body's own macromolecules. Polymers possess this property. The chosen dosimeter material is the polymer Polyisobutylene (PIB) with a mean molecular weight of approximately  $10^6$ . Irradiation of PIB with  $\gamma$ -radiation predominantly leads to cleavage of the main chain and therefore a drop in the mean molecular weight. Besides the fact that the chosen PIB resembles the macromolecules of the body structurally, it has the advantage of having a high molecular weight. The idea is that in a given amount of a material with high molecular weight the number of molecules is relatively small. This means that few cleavages of the main chain leads to a relative large change of the mean molecular weight.

The quantity of ionizing radiation is not the sole determining factor for the ability to cause biological damage. The quality of radiation is also an important factor. The aim is to decide whether the method can be used to differentiate between  $\alpha$ - and  $\gamma$ -radiation or not.

## THEORY

Polymers are macromolecules consisting of a considerable number of simple repeating structural units (monomers). In a given polymer (except from certain polymers found in nature) the molecules will have a varying number of monomers. The varying length of the individual polymer chains is due to the element of randomness existing in the synthesis of the polymer. Therefore all synthetically polymers are said to be polydisperse.



In order to be able to characterize polymeric systems one must have knowledge of the average molecular weight as well as the molecular weight distribution (MWD). The parameters used in this project are the average number molecular weight  $M_n$  and the average weight molecular weight  $M_w$ .

$$M_n = \frac{\sum N_i \cdot M_i}{\sum N_i} \quad M_w = \frac{\sum N_i \cdot M_i^2}{\sum N_i \cdot M_i}$$

where  $N_i$  is the number of molecules with molecular weight  $M_i$ .

The polydispersity  $Pd$  is defined as the ratio between  $M_w$  and  $M_n$  and is used as a measurement of the width of the MWD.

Irradiation of polymers leads to an alteration in polymeric properties such as strength and flexibility. Such changes are related to a change in molecular size. During irradiation two effects both related to a change in molecular size take place: Cross-linking (gelling) and scission (degradation). In cross-linking individual molecules are linked together to form a three-dimensional network, while in scission the main chain is cleaved into smaller molecules. Both cross-linking and scission occur simultaneously during irradiation, but the relative magnitude of cross-linking to scission depends on polymer structure.

If scission is the main result of irradiation the following relationship between mean molecular weight  $M$  and absorbed dose  $D$  applies<sup>1</sup>:

$$\frac{1}{M} = \frac{1}{M_0} + k \cdot D$$

where  $M_0$  is the initial value of the mean molecular weight.

## MATERIALS AND METHODS

### Polyisobutylene

The tested PIB is from Vistanex and has an average weight molecular weight of approximately  $1,2 \cdot 10^6$  and a polydispersity of 3. The repeating structural unit of PIB (IUPAC name: 1,1-dimethylethylene) is shown on Figure 1. PIB is regarded as a non-gelling polymer, that is a polymer that primarily degrades upon irradiation (with  $^{60}\text{Co}$ ).

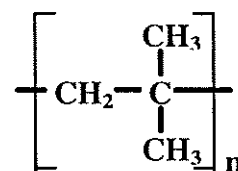


Figure 1. Chemical structure of PIB

### Gel Permeation Chromatography (GPC)

Molecular weight distributions are determined with a GPC system with a Differential Refractometer-Viscometer Model 200 from Viscotec. The two columns are Waters Styragel HT 6E (7,8\*300 mm). The solvent is stabilized Tetrahydrofuran (THF).

The method used to determine molecular weight and MWD is Gel Permeation Chromatography (GPC). GPC is an analytical chemical method, which separates molecules by size. The sample is dissolved in cyclopentane (1mg/ml) and is lead through a system of columns containing beads of a porous polymeric material. The pores of the beads vary in size. As the sample passes through the columns the molecules will by entropy controlled diffusion attempt to diffuse into the pores. Effectively the system has a variable length, the magnitude of which depends on the size of the molecules. The largest molecules will be eluted first and the

<sup>1</sup> "Irradiation Effects on Polymers", p.46, D.W.Clegg et. al, Elsevier Applied Science, 1991

smallest last. GPC retention is an equilibrium and entropy controlled size exclusion process. The separation itself is described by retention volume or retention time. GPC is not an absolute method for determining the molecular weight, hence a calibration of the system is required. The method is sensitive enough to detect differences in molecular weight within a few percent.

### <sup>60</sup>Co Irradiation

Before irradiation the sample is dissolved in Toluene. The antioxidant is removed from the solution by pouring it slowly into Methanol. PIB precipitates in Methanol and is easily extracted. Remaining solvent evaporates in vacuum at 40°C. The sample is irradiated in a <sup>60</sup>Co-chamber with a dose rate of 3 kGy/hr. Irradiations are carried out in the dose interval 0 to 100 kGy. After irradiation the sample is dissolved in Toluene and GPC analysis is conducted.

### <sup>241</sup>Am Irradiation

After the removal of the antioxidant and the remaining solvent, the sample is dissolved in cyclopentane. Films with a thickness of 25 µm are produced. The film is irradiated with a <sup>241</sup>Am source (diameter of 7 mm). Only a part of the irradiated sample is then dissolved in Toluene and GPC analysis is carried out.

## RESULTS

### <sup>60</sup>Co Irradiation

Irradiations in the dose interval 0 to 10 kGy gives an evident change in mean molecular weight. Displayed in Figure 2 are some of the achieved chromatograms. The higher the dose the lower the mean molecular weight. The change in  $M_n$  per kGy is calculated to be approximately 12% while the change in  $M_w$  is approximately 23%.

Commercial available polymers often contain antioxidant to inhibit autooxidation. The presence of antioxidant enhances the stability of the polymer by inhibiting degradation. In the initial experiments antioxidant was contained in the sample. The presence of the antioxidant revealed itself by plotting  $1/M$  against dose. In the dose interval 0 to 20 kGy the curve is evidently rising faster than in the dose interval 20 to 100 kGy. This indicates that the degradation is slower in the interval 0 to 20 kGy. In the following experiments the antioxidant was removed. Plotting  $1/M$  against dose for irradiated samples without antioxidant gives a straight line in the entire interval.

$$\frac{1}{M_n}(D) = 4,1 \cdot 10^{-7} \cdot D + 2,6 \cdot 10^{-6} \quad \frac{1}{M_w}(D) = 2,4 \cdot 10^{-7} \cdot D + 5,6 \cdot 10^{-7}$$

The uncertainty in determination of  $M_n$  and  $M_w$  is approximately 3%. The presence of antioxidant increases the mean molecular weight with 5% and 16% for  $M_n$  and  $M_w$  respectively.

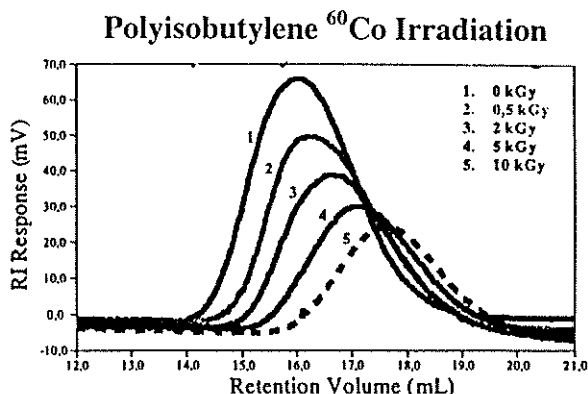


Figure 2. PIB irradiated with <sup>60</sup>Co. Larger molecular weight is displaced to the left.

## <sup>241</sup>Am Irradiation

In order to be able to compare the change in molecular weight resulting from  $\gamma$ -irradiation with  $\alpha$ -irradiation it is necessary to know the dose distribution through the sample for the  $\alpha$ -irradiation. Alpha particles have a relatively short range in matter and the rate of energy deposition vary greatly with the travelled distance. The range of <sup>241</sup>Am  $\alpha$ -particles in PIB is just 41  $\mu\text{m}$ . If the  $\alpha$ -source is assumed to be plane and infinite in size compared to the dimension of the analyzed material, then the dose distribution can be determined. The absorbed dose in depth  $x$ ,  $D_d(x)$  is hence given by:

$$D_d(x) = \frac{t}{\rho} \cdot \frac{d}{dx} \left( \frac{q}{2} \left( E_0 - \int_0^{\text{Arccos}(x/R_0)} E(R) \sin(\varphi) d\varphi \right) \right)$$

where  $t$  is the time of irradiation,  $\rho$  is the density of PIB (0,915 g/cm<sup>3</sup>),  $q$  is the activity of the source per unit area,  $E_0$  is the initial energy,  $R_0$  is the range and  $E(R)$  is the energy of the particle after having travelled the distance  $R = R_0 - x/\cos\varphi$ .

The  $\gamma$ -irradiation experiments have provided a relation between mean molecular weight ( $M$ ) and absorbed dose ( $D$ ). Assuming that  $\alpha$ - and  $\gamma$ -radiation degrades PIB in the same manner, the relationship between  $M$  and  $D$  can be applied to the  $\alpha$ -irradiation experiments to determine how the mean molecular weight will vary with the depth of the sample. For this distribution of mean molecular weights the average molecular weight is determined. This theoretically determined average molecular weight is then compared to the experimentally determined average value. If those values are significantly different from one another, is it possible to detect a quality difference between  $\alpha$ - and  $\gamma$ -radiation. At present time no  $\alpha$ -irradiation experiments have been carried out.

## DISCUSSION

The perspective of the project is to find a method using GPC to determine doses in the range from 0 to 100 Gy. The present results show that a dose of 100 Gy will cause a change in  $M_n$  of approximately 1,5% and a change in  $M_w$  of 4,2%. The relative uncertainty of the mean molecular weight ( $M$ ) is approximately 3% (for  $Pd \approx 3$ ). Hence detection of a dose of 100 Gy would be possible by measuring  $M_w$  but not  $M_n$ . However the narrower the MWD the smaller is the uncertainty on  $M$ . Hence using a PIB with a more narrow MWD it should be possible to determine lower doses.

During irradiation both cross-linking and scission occur simultaneously. Recent data<sup>3</sup> obtained for the polymer PMMA (Poly(methyl methacrylate)) show that the ratio between the amount of cross-linking and scission not only depends upon polymer structure but also upon the linear energy transfer (LET) of the radiation. PMMA is a polymer that degrades upon irradiation (as PIB does) at least when irradiated with low LET radiation (<sup>60</sup>Co). When PMMA is irradiated with high LET radiation (e.g. 2 MeV Ar<sup>+</sup>) excessive cross-linking takes place (the polymer becomes insoluble). A similar effect could exist for PIB.

<sup>2</sup> The dose contribution from the gamma lines in <sup>241</sup>Am is so small that it can be neglected.

<sup>3</sup> "LET effect on cross-linking and scission mechanisms of PMMA during irradiation", E.H.Lee et.al, Radiation Physics and Chemistry 55 (1999) 293-305

# Long-term decrease in the external radiation exposure, due to Chernobyl fallout, to inhabitants in the Bryansk area in Russia.

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## Abstract

*The Bryansk region in south-western Russia was one of the areas, which were highly contaminated with  $^{137}\text{Cs}$  and  $^{134}\text{Cs}$  after the Chernobyl accident in 1986. In 1990, a joint Nordic-Russian co-operation was initiated. Every year in September, we have visited this region and performed individual measurements of external absorbed doses. The participating persons were living in small villages with contamination levels between 0.9 and 2.7 MBq/m<sup>2</sup>. Some of the villages were decontaminated to various extents. Both school children and adults participated in the study. The absorbed dose from external irradiation was assessed with TL-dosemeters worn by the participants in a string around the neck for one month. In 1997, the external absorbed dose to the population in five different villages was in average around 50% of the 1990 and 1991 values. The decrease in external exposure may be described by an exponential function with a halftime of 3-6 years in the different villages.*

## Introduction

The Bryansk region in Russia was one of the most heavily affected area outside the 30 km zone after the Chernobyl reactor accident in 1986. After the first years, the radionuclide dominating the absorbed dose to the inhabitants in the area was  $^{137}\text{Cs}$ . Since 1990, we have made expeditions to the area in September every year to assess the body burden of  $^{137}\text{Cs}$  (Wallström et al. 1995, Zvonova et al. 1995) as well as the effective dose received by external radiation (Erkin et al. 1994, Wöhni 1995). The participating people were living in small villages with contamination levels of  $^{137}\text{Cs}$  ranging between 0.9 and 2.7 MBq/m<sup>2</sup>. Some of the villages were decontaminated to various extents.

In this report, results from measurements of the external dose will be given for the years 1990-1998 for people living in different villages.

## Material and methods

Thermoluminescence (TL) dosemeters based on 3x3x0.9 mm<sup>3</sup> LiF chips were used to assess the absorbed dose to individuals from external irradiation in their environment. The dosemeters were calibrated in Sweden and brought to Russia in a lead container with 1 cm thick walls. The final results are presented in terms of effective dose, according to the ICRP 60 definition, with the conversion factor of effective dose to absorbed dose to body surface ( $E/D_{\text{surf}}$ ) of 0.92 Sv/Gy (Wallström 1998).

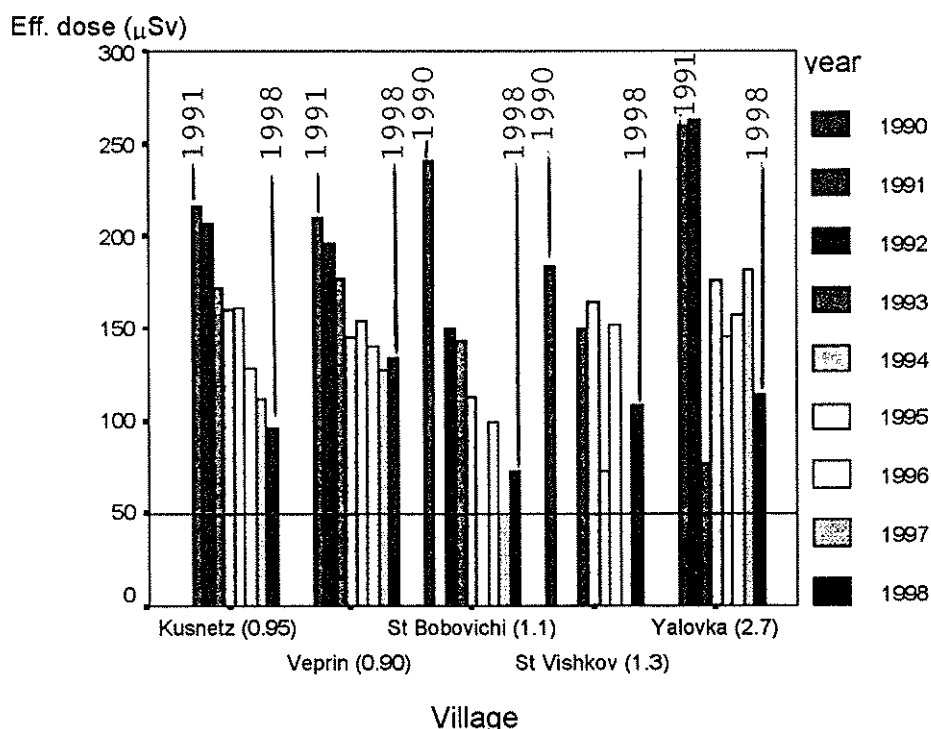
The LiF chips were mounted in a plastic dosimeter holder worn by the participants on a string around the neck during one month. The dosemeters were then put back in the lead container

and sent to Sweden for evaluation. During transport, the dosimeters were stored in the lead container together with "background" dosimeters used to estimate the signal accumulated during transport and storage.

The villages are situated in rural areas in the south western part of Russia, about 250 km north-east of the Chernobyl power plant. The participating people were in the age between 6 and 83 years, living in either brick or wooden houses, with agriculture being the principal occupation. A large percentage consisted of school children.

## Results, discussion

The mean values of the effective dose in September for people living in different villages are presented in Fig.1 for the nine year period 1990-1998. The reference line at 50  $\mu\text{Sv}$  represents an estimated level for the natural background radiation in the area. The numbers in parenthesis following the village name is the contamination level of  $^{137}\text{Cs}$  in  $\text{MBq/m}^2$  in 1986, before the decontamination efforts. Major decontamination efforts were carried out in Yalovka, while Kusnetz and Veprin were decontaminated to a very small extent. St. Bobovich and St. Vishkov could be considered partly decontaminated. The decrease of the exposure in the two villages without any major decontamination efforts show a more regular pattern than the rest of the villages. The spread in dose values for some villages (mainly St Vishkov and Yalovka) could be caused by some persons not wearing their dosimeters as instructed during one of the measurements, or being on leave from the area for some time during the measurement period.



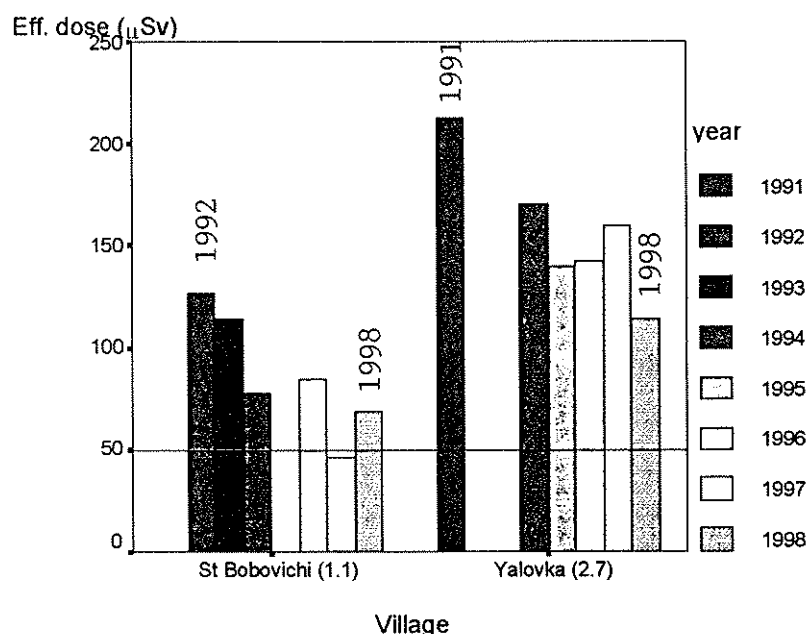
**Fig.1.** Mean effective dose from external irradiation during one month (September) in different villages for the period of 1990-1998. The reference line at 50  $\mu\text{Sv}$  represents the level for the estimated monthly natural background radiation in the area. The numbers in parenthesis refer to the contamination level of  $^{137}\text{Cs}$  in  $\text{MBq/m}^2$  in 1986.

The effective dose was reduced with an apparent half time of 2.8 - 6.0 years depending on village, see Table 1. The figures correspond to a yearly reduction of 10-20%. Kusnetz and Veprin had similar contamination levels, and were not decontaminated to any greater extent, still the doses seem to be reduced in different ways. The living conditions in the villages are not the same, Kusnetz being a village with many elderly inhabitants, who might have different outdoor habits than younger people. This could be one explanation together with the differences in the natural environment.

**Table 1.** The apparent half-time for the mean effective dose, received by external irradiation, in the studied villages during the years 1990-1998.

VILLAGE	HALF TIME (YEARS)
St Bobovich	2.8
Kusnetz	3.8
Veprin	5.8
St Vishkov	5.8
Yalovka	6.0

In Fig. 2, the effective dose to school children from external irradiation during 1991-1998 is presented for two villages. The difference in contamination level is clearly demonstrated, even though the reduction in effective dose is similar over time.



**Fig.2.** Mean effective dose from external irradiation during one month (September), for school children in two different villages, for the period of 1991-1998. The reference line at 50  $\mu\text{Sv}$  represents the estimated level for the monthly natural background radiation in the area. The numbers in parenthesis refer to the contamination level of  $^{137}\text{Cs}$  in  $\text{MBq/m}^2$  in 1986.

The distribution of the effective dose from external irradiation in all villages is presented in Fig 3 and 4, representing the years 1991 and 1998. There is a clear shift towards lower doses in 1998 compared to 1991, together with the fact that the 1998 values have a narrower distribution. There may be a tendency to a more lognormal distribution in 1998.

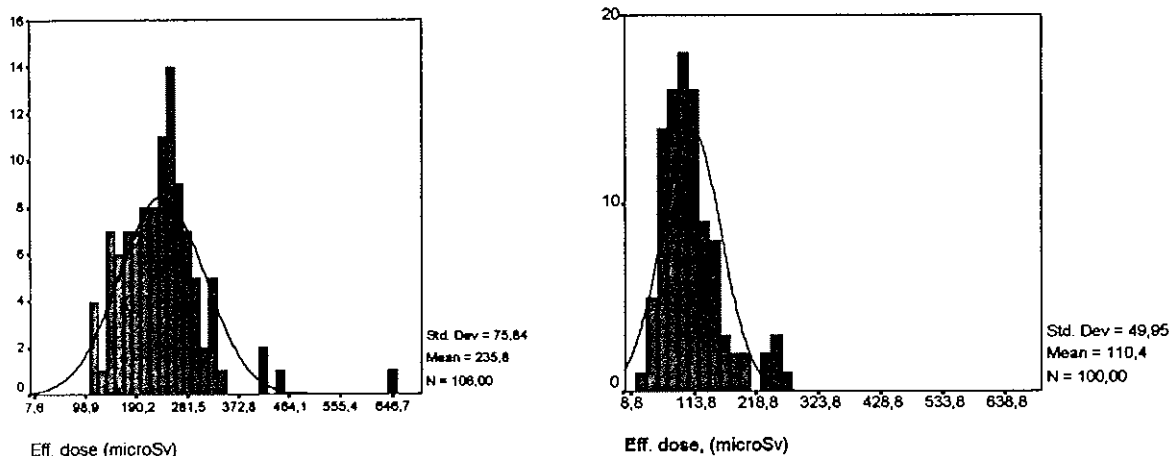


Fig. 3 and 4. Distribution of external doses in all villages in 1991 (left) and 1998 (right). The normal distribution curve is also displayed

### Acknowledgements

The project was financially supported by the Swedish Radiation Protection Institute.

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Holje, Gunilla	Lund University Hospital	Sweden
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Linde, Kenneth	Långnäs Eltekniska AB	Sweden
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Petersen, Søren Bøgvad	Dansteel Engineering A/S	Denmark
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Rahola, Tua	Radiation and Nuclear Safety Authority	Finland
Ramsøy, Tore	Institute for Energy Technology	Norway

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Realo, Küllike	University of Tartu	Estonia
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