PROCEEDINGS

Nordisk Selskap For Strålevern, det 11. ordinære møtet
Det 7. Nordiske Radioøkologi Seminar
26. - 29. august 1996, Reykjavík, Ísland

Nordic Radiation Protection Society
Nordisk Selskap For Strålevern

nrps

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Editors:
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Einar Páll Guðlaugsson

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Editorial

In June 1995 the Health Physics Society (HPS) released a special issue of their HPS Newsletter featuring several readers' viewpoints on the validity of the linear, no-threshold (LN-T) model for dose response at low levels and low-dose rate exposures to ionizing radiation. In the following months the topic was extensively discussed both in the HPS Newsletters and in other forums, such as the Radsafe computer bulletin board. The discussions revealed a variety of opinions. The LN-T model was challenged on scientific grounds, and the application of the model in radiation protection and risk analysis was questioned. On the other hand, several did also deliver convincing arguments for the established use of the model. In March 1996 an HPS position statement, "Radiation risk in perspective" [1], was issued, which recommended against quantitative estimation of health risk below an individual dose of 50 mSv in one year or a lifetime dose of 100 mSv in addition to background radiation. The position statement did not conclude the discussions, and at the 11th meeting of the Nordic Radiation Protection Society in Reykjavík in August 1996, several key persons involved in the previous months discussions were invited lecturers. Thus, at the first day of the conference, a thorough overview of the question was presented.

As in previous meetings of the Nordic Radiation Protection Society, the received papers spanned all aspects of radiation protection and radioecology. Of course, papers on radioecology are especially well represented, reflecting the fact that it was a joint venue with the 7th Nordic Radioecology Seminar. The papers appear in the proceeding as they were presented at the conference.

All, except invited papers, have been refereed. Approximately half were returned to the authors for improvement. This has without doubt heightened the quality of the proceedings. We would like to thank all the referees for their excellent work and commitment.

A copy of the proceedings has been sent to all that attended the conference. In addition several institutions involved in radiation protection or radioecology have received a copy. The proceedings are also available on the Internet (http://www.geirik.is/nsfs/). Detailed information on how to get printouts that exactly match the format of the proceedings can be found there.

Tord Walderhaug
Proceeding editor

[1] Mossmann K. L., Goldman M., Massé F., Mills W. A., Schiager K. J. and Vetter R. J., *Health Physics Society Position Statement, "Radiation Risk in Perspective"*, HPS Newsletter, Vol. XXIV, No 3, March 1996. (The text is also available on the Internet: http://www.inel.gov/resources/research/tschaech.html)

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Opening Address

Sigurður M. Magnússon

Dear friends and colleagues, ladies and gentlemen

It is a great pleasure for me as president of the Nordic Radiation Protection Society to welcome you to Iceland for the llth General Conference of the Society and the 7th Nordic Radioecology Seminar.

We are particularly happy that so many are attending our meetings this week. It was estimated at the initial planning that we could expect about 100-120 participants but it turned out that over 200 did register. I wonder why, hopefully it's the quality of the programme but the opportunity to visit a "different and exotic" country may also have something to do with it.

Questions have been raised why we have merged a meeting on radiation protection and a meeting on radioecology. Well, there is a straight forward explanation to that. The 7th Nordic Radioecology Seminar was to be held in Iceland in 1995 and the Society meeting this year. There is a considerable overlap both in topics and attendance between the two meetings. Hence it was decided, after extensive consultation, to merge the two meetings into one.

The Nordic Radiation Protection Society was founded in 1965. It is the regional IRPA society for the Nordic countries and has about 480 members. There are close links between our Society and IRPA. We are quite privileged to have here today 3 out of 12 members of IRPAs executive council, namely our own Jack Valentin of Sweden who was elected to the executive council at the Vienna meeting this spring and Bill Mills and Frank Masse of the Heath Physics Society.

There is a tradition for the general conferences and the presidency of the Society to rotate between the nordic countries. The last one was in Kristiansand in Norway in 1993 and the next will be in Denmark in 1999 if we continue with the tradition of a ordinary meeting every 3 years.
As I stand here today I can not help but think back to the year 1981 when the 6th general conference of the Society was held in Reykjavík. I am happy to see here today many of those that attend the meeting in 1981. The Nordic society is like a big family and our meeting express that being informal like a family reunion and stimulating both on a professional and personal level.

Nordic co-operation in radiation protection and radioecology has been extensive over the years and continues to be of high priority even more now that Denmark, Finland and Sweden are members of the European Community and Iceland and Norway not. However the co-operation is not static in nature but has changed over the years and adapted to changes that have taken place. In short it is an excellent example of a successful regional co-operation.

Turning my attention to scientific sessions of the meeting it is safe to say that we have stimulating week with many interesting presentations ahead of us.
First let me point out the highlight of the meeting which is " **the Bo Lindell lecture** ". Bo Lindells contribution to radiation protection over the years is momentus. He is a

former chairman and emeritus member of ICRP, former chairman of UNSCEAR, former Director General and emeritus worker of the Swedish Radiation Protection Institute to name a few of his activities. His profound influence on radiation protection and risk research in general includes a large number of high quality scientific publications on all aspects of protection of man against harmful effects of radiation. I should also like to add on a personal level, that Bo is a wonderful person of the highest integrity and that he has an excellent sense of humour.

At their 1995 meeting, the directors of the Nordic radiation protections institutes agreed to make a proposal to the board of the Nordic Radiation Protection Society that a lecture in honour of Bo Lindell be delivered at the general conferences of the Society, hence the name "the Bo Lindell Lecture" and that the institutes in turn would cover the travel costs of the lecturer chosen. It is the institute in the country that has the presidency of the Society and is hosting the meeting that covers the cost. The proposal was, of course, gratefully accepted by the board.

The Society is grateful to Bo Lindell for agreeing to a lecture bearing his name be delivered at the general conferences of the Society and to the directors of the nordic radiation protection institutes for their proposal and generous support. It is a particular pleasure that the first Bo Lindell lecture will be delivered this morning and who could be more appropriate as the first Bo Lindell lecturer than our good friend and colleague professor Roger Clarke, chairman of ICRP and director of NRPB. Later on this morning Bo will give a talk on "Elementary risk concepts".
We shall today hear of the activities of ICRP and NCRP but the main emphasis will be on issues related to the validity of the linear no threshold model for low doses. These issues are of fundamental importance for the future of radiation protection, especially with regard to the regulatory aspects and are being lively debated on both sides of the Atlantic. It is fortunate that we have been able to get so many of the leading experts in the world together. I hope that at the end of the day we shall have a better understanding of the fundamental issues involved both from a scientific and regulatory point of view.

For the rest of the week we have many presentations that reflect the diversity and dynamics of the Nordic research on the various aspects of radiation protection, emergency preparedness and radioecology. In this context the important role of the Nordic Nuclear Safety Research projects, with regard to research in the fields of nuclear safety, radioactive waste, radioecolgy and emergency preparedness must be emphasised. The 12 presentations on NKS supported projects as the many other presentations later on this week will clearly illustrate this importance.

In conclusion I want to take the opportunity to thank all my co-workers at Geislavarnir and all others that have been involved in planning of the meetings for their unselfish and excellent work. In particular I want to thank our scientific secretary Tord Walderhaug for his hard work and dedication.
With these words I welcome you to Iceland once again and declare the llth general conference of the Nordic Radiation Protection Society and the 7th nordic radioecology seminar to be open.

Thank you very much.

NORDIC RADIATION PROTECTION SOCIETY

GENERAL CONFERENCE

August 26, 1996, Reykjavik, Iceland

The Bo Lindell Lecture

Towards ICRP 2000: Issues that are of importance for radiological protection as the millennium approaches

Roger H Clarke
Chairman, ICRP

Abstract

The work of the Main Commission of ICRP is supported by its four Committees on: Biological Effects, Secondary Standards, Protection in Medicine, and Application of the Commission's Recommendations. This paper will discuss those aspects of the work of these Committees that are likely to be significant for the development of radiation protection policy and practice into the next century. It will particularly deal with the questions of risks at low doses of radiation and the "threshold controversy". The issue of genetic susceptibility to radiation induced cancer is another area in which ICRP is working and will be covered. There are also programmes of work concerning the development of internal dosimetry that will be described. Finally the programmes on radiological aspects of waste disposal and release of contaminated land are addressed since these are topics for which the idea of a threshold has recently been proposed. The paper will argue counter proposals.

INTRODUCTION

It is very appropriate that the Nordic Radiation Protection Societies have decided to honour Bo Lindell with the establishment of this lecture and it is a privilege to be the first to present it. Bo Lindell was a protégé of that founder member of the radiological protection profession, Rolf Sievert. Sievert was chairman of the Main Commission from 1956 to 1962, during which period Bo was the Scientific Secretary to the Commission. He was then a member of the Commission and Chairman of Committee 3 from 1965 to 1969, vice-Chairman of the Commission between 1969 and 1977, and its Chairman from 1977 through 1985.

The history of ICRP begins in 1928 and continues today, some 100 years after the discoveries of x-rays and radioactivity. In this paper some of the more important issues that are of importance in radiological protection, as we leave this century and enter the new millennium, are discussed. These centre around the knowledge of the risks associated with exposure to ionising radiation, the ability to estimate doses from exposures under different circumstances, and the control of those doses, which leads to the topic of risk management.

RISK COEFFICIENTS FOR PROTECTION

Committee 1 of the International Commission on Radiological Protection has the responsibility for maintaining the biological effects of ionising radiation under review and developing documents that relate such effects to the needs of radiological protection. During the last three years, new task groups have commenced work and working parties have been formed to review a broad range of biological, biophysical and epidemiological topics. The issues that may be important in the next few years are summarised here.

Genetic Susceptibility

The Task Group on Genetic Susceptibility to Cancer had been set up by the Commission in order to provide an overview of human genetic predisposition to cancer and to make interim judgements on the possible implications for the estimation of cancer risk after irradiation. The first draft of the main body of the review contains sections on: DNA damage and repair, mechanisms and genetics of solid tumours and lympho-haemopoietic neoplasia, tumorigenic radiosensitivity and heritable predisposition to cancer, together with computational modelling of genetically determined cancer risk after radiation.

There is particular emphasis on the potential importance of the repair of DNA double strand lesions for carcinogenic response and the relative genetic contributions to solid and lympho-haemopoietic neoplasia. Attention is also given to uncertainties on the apparently high genetic contributions to spontaneously arising breast and colon cancer and the limited extent of the database on which to estimate the degree to which cancer-prone human genetic disorders may contribute to excess cancer in irradiated human populations. These issues are critical to the computational modelling that has been proposed and will be the subject of further review and deliberation by the Task Group.

The tentative conclusions of the Task Group are that the incidence of genetic susceptibility to ionising radiation in a normal population is so low (<1%) that no special considerations need to be given for protection of workers or the public in setting normal standards of protection. However, in the case of therapy there may well be a need to consider individual sensitivity to

radiation, mainly because those who are genetically predisposed towards cancer will be those presenting for treatment and they may well be the ones with higher sensitivity to radiation.

Epidemiological data

By the year 2000 there will be available the mortality data from the Japanese A-bomb survivors to the end of 1995. What may these data indicate? The mortality data to the end of 1990 show an excess relative risk for solid tumours which is statistically significant down to 50 mGy, which compares with 200 mGy previously for data to the end of 1985. The results for solid tumours look remarkably linear against dose, but the risk for exposures at younger ages does appear to reduce at longer times. The next tranche of data may lead us to consider that for solid tumours, the DDREF may be less than the figure of 2 that was used in the 1990 recommendations, but that the extrapolation of constant relative risk over a lifetime may lead to an overestimate of risk.

Data will also become available from cohorts of chronically exposed nuclear workers, a greater proportion of whom are alive than the Japanese survivors. There is therefore less certainty, but it should be anticipated that the new analyses will support the A-bomb data and lead to the conclusion that the current figure of 5% per Sv is fairly robust.

Uncertainties in cancer risk estimates

The main uncertainties under review are those associated with the transfer of risk between different populations, calculation of lifetime cancer risks for tumour types and A-bomb dosimetry. The overall conclusion from the working party was that uncertainty on risk was not likely to vary from the current risk of around 5% per Sv for a whole population by more than a factor of around two in either direction.

Risk factors for individual organs

While the overall risk factor of 5% per Sv may be robust, it is to be expected that there will be data on risks for additional individual organs. Examples could be brain, kidney and some of the other organs that are currently in the Remainder category. This raises a possibility that new w_T values will be required. It becomes more difficult to fit additional organs into the present scheme, so that consideration may have to be given to each organ having its own value.

This raises another issue, since more specified organs means smaller w_T values for each and the possibility of high individual organ doses while the effective dose is low. Thus there is a question as to whether a limit on effective dose is a sufficient criterion, or whether an organ limit is required so as to prevent deterministic effects. This seems to be a question for Committee 1.

Molecular studies

Increasingly, interpretation of epidemiological and experimental studies at low doses is being influenced by accumulating information on the fundamental nature of the tumorigenic process. Neoplasia in tissues is now seen as a complex, multi-stage process that can be sub-divided into four phases: neoplastic initiation; promotion; conversion and progression. The sub-divisions are necessarily simplifications of the overall process which is, in any event, somewhat variable between different tumour types. However, they do provide a basis from which to interpret the cellular and molecular changes involved.

Neoplastic *initiation* encompasses the essentially irreversible cellular damage, which although not necessarily expressed immediately, provides the potential in cells for neoplastic development. There is good evidence that this initiation process results from damage to DNA leading to gene mutations in single target cells in tissues. It is argued that the critical damage is likely to be coincident damage to both DNA strands (DNA double strand breaks). Although a proportion of such double strand damage will be repaired, completely error free repair of such damage, even at low doses, is not expected.

There are 5,000-10,0000 DNA damage events per hour per cell due to thermodynamic instability and attack by chemical radicals via endogenous biochemical reactions. In a typical DNA repair interval of about 4 hours, some 30,000 "events" will occur. It has been suggested that, because of this high level of single strand DNA damage arising spontaneously in cells, the small increment of damage induced by low dose radiation will be insignificant for cancer risk.

This argument is not sustainable since it fails to recognise the very low abundance of spontaneous DNA double strand damage and the critical importance of these lesions and their misrepair for cellular radiobiological response.

Once the necessary gene mutation is present in a cell, further neoplastic development is believed to be highly dependent upon both the intra- and extra-cellular environment. As a consequence of *promotional* events, influenced by growth factors in cells, dietary constituents, hormones, or other environmental agents, there may be an increase in cell proliferation and in some instances interference with communication processes between cells that act to maintain cellular stability in tissues.

Once the potential for full malignancy has been established, the subsequent *progression* of the disease may depend upon further cellular changes that allow invasion of adjacent normal tissues, the circulation of neoplastic cells in the blood and lymphatic systems and the establishment of metastases (secondary tumour growths) at other sites in the body. It is this invasive process that provides principally for the fatal effects of most common human tumours.

Two classes of tumour-associated genes have been identified. *Proto-oncogenes* are subject to gain-of-function mutations while *Tumour suppressor genes* act as negative regulators of cellular processes that act to mediate cell division and development. It is loss, rather than gain, of function of these genes that contributes to the development of neoplasia. This loss of function may result from single DNA base changes, deletions of small regions of DNA or the loss of whole chromosome segments.

Although radiation-induced mutation may in principle influence all stages of the neoplastic process, it is argued that neoplastic initiation is the key stage that is primarily targeted by low doses of radiation. Thus, a single radiation track (the lowest dose and dose rate possible) traversing the nucleus of an appropriate target cell has a finite probability, albeit very low, of generating the specific damage to DNA that results in a tumour initiating mutation.

Dose and dose-rate, as well as radiation quality effects, are now seen to be determined by the failure of DNA to effect an error-free repair. Specific DNA deletions and/or rearrangements act as initiating events in radiation carcinogenesis. Following this, and again at a low probability, these initiated cells can develop by multistage processes into an overt malignancy. As was

mentioned in the Task Group on Genetic Susceptibility, inherited predisposition will influence radiation sensitivity.

There is some evidence that low dose radiation may induce or activate cellular DNA repair functions, the so-called adaptive response. The majority of effects seen to date have been essentially short-term, and the current consensus is that knowledge of their relevance to neoplastic processes is insufficiently developed and understood to influence current judgments on tumorigenic responses at low doses and low dose rates.

As a consequence, at the level of DNA damage, there is no basis for assuming that there is likely to be a dose threshold below which the risk of tumour induction would be zero. It is appropriate therefore to assume a progressive increase in risk with increasing dose .

The threshold controversy

The pursuit of a threshold is fraught with danger - it does not solve any problems; in fact it creates them! How would a protection system based on a threshold be implemented?

At what level of dose would it be set? And is it for a single acute dose or protracted exposure? Natural background radiation - excluding radon - gives everyone about 100 mGy over a lifetime, even twice that. Is the threshold to be set above this? No, surely lower, AND presumably we have to set a margin of safety - so where does it finish? Then exposures would not be additive and difficult to control - who released the photon that took the worker or child over the threshold.

Then there is the epidemiological evidence of risks at low doses in the range 10 - 100 mGy, thyroid and breast studies come to mind, as well as the *in-utero* series that indicated risks at doses as low as about 10 mGy. Now the latest Japanese A-bomb survivor mortality data to the end of 1990 show statistically significant excess risk at 50 mGy.

The pursuit of a threshold is seriously misguided. It is an attempt at a quick fix which does not address the real issues, return to which will be made later in this paper.

DOSE ESTIMATION

In the second of the three issues identified for the year 2000, the estimation of doses to exposed individuals, the topics that will be of interest centre around the acquisition of biokinetic data, the establishment of the Reference Person and the development of new phantoms.

Committee 2 has the responsibility for establishing secondary standards based on the Commission's recommended dose limits. The Committee and its task groups have been developing a series of documents related to both external radiation and internally incorporated radionuclides. While there is now a period of stability in the dose coefficients for use in protection, there remains a need to develop a full revision of Publication 30. With the move to physiologically-based biokinetic models, the aim is to develop models that are appropriate both for bioassay interpretation and for dosimetry calculation. Thus the full revision of Publication 30 will take into account the work done for the revision of Publication 54. As it will be necessary to review the biokinetic models for all the elements that have not been reviewed in Publications 56, 67, 69 and 72, it is unlikely that the work will be published before the year 2000.

It had been agreed at a previous meeting of Committee 2 that it would no longer be practicable to develop a full revision of Reference Man (Publication 23, 1975). Instead, a series of reports is in preparation covering the main information necessary for model development and for dose calculations. The first report, on the skeleton, was issued by the Commission as Publication 70 in 1995. The next report, on basic anatomical and physiological data, should be completed in 1996. A third report will cover the regions of the gastrointestinal tract, as well as the pancreas, gall bladder and liver.

In the development of ingestion dose coefficients for the public, the adult model for the gastrointestinal tract described in Publication 30 was used, as no age-dependent model is available. The Committee now intends to develop a new model for the gastrointestinal tract which will take into account the most recent information on anatomy and physiology as well as age-dependent characteristics. Additional information on the radiation sensitivity of different regions of the gastrointestinal tract and on the location of stem cells will also be considered. A working party has been set up, with membership from Committees 2 and 1, to develop the terms of reference for a task group. The aim is that the new model should be complete in time for the full revision of Publication 30.

The need to develop agreed phantoms for use in both internal and external dosimetry was considered by the Committee at some length. The MIRD phantom presently used for dose calculations was developed in the late 1960's and represents a mathematical triumph. It does however have severe limitations as it is based on fixed geometrical shapes. This makes it difficult to introduce additional organs and tissues when the need arises as a result of new information on the sensitivity of tissues to irradiation.

Comprehensive data are now becoming available from medical imaging and a number of laboratories are working towards developing realistic phantoms based on such data. Since absorbed doses have to be calculated by Monte-Carlo techniques, there is no reason not to move from mathematical descriptions of organs to real digitised data from MRI scans. Although there are difficulties in using such phantoms for dosimetry purposes, it was felt that this will be necessary before the revision of Publication 30. A working party is to consider how the Committee should proceed to develop new phantoms for the Commission. A 9 million voxel phantom called NORMAN, a digitised data set is being used at NRPB for non-ionising dose calculations. Each tiny cube is assigned one of 35 tissue compositions and the total phantom has been computer normalised to the height and weight of Reference man. Another advantage is that there should be no problem in introducing additional organs for which dose estimates are required. The digitisation of reference woman remains to be done.

ACCEPTABLE LEVELS OF RISK

The risk per unit dose has been established and the method of assessing doses from intakes of radionuclides as well as external exposures also is established. The next question is about risk, since in order to set dose limits or intervention levels, there is a need to decide upon the level of risk that is to be used.

Thus, the third issue identified for year 2000 is arrived at and it is probably the most crucial issue for radiological protection. It also happens to be the subject that Bo Lindell himself has concentrated upon in recent years through his involvement with the Swedish Risk Academy.

All human activities or lack of activities carry some risk. Some of the activities are accepted by most people even if the risks are rather high, eg. driving and traffic accidents. Other activities are not accepted because the risks are considered unjustifiably high in relation to the ensuing benefits even after reasonable attempts at risk reduction. Many attempts have been made to set an upper level of risk to an individual, i.e. a level of risk which would not be acceptable even if it could not be further reduced. There is a difference here between voluntary and imposed risks. For radiation protection purposes the relevant circumstances would be normal occupational or private life in what might be considered a safe society.

In fact there is no single figure for an 'acceptable' risk. The figure varies as between occupational and public exposures, between normal operations, potential situations, and intervention following an accident, between chronic and acute situations, and of course, acceptable levels of risk change from time to time to reflect societies' preferences.

In this framework, a dose limit is set at a level of risk selected at the boundary in the region between "tolerable" and "unacceptable" for the situation in which dose limits apply - ie the control of the total dose from all specified sources. The limit then protects the individual from all sources under control by ensuring that the total risk is not unacceptable. It follows that the optimisation process for any single new source should be "constrained" to restrict individual exposure so that the risk from that single source is in the "acceptable" range and the total risk does not become unacceptable.

For the year 2000, it may be interesting to reflect on the status of acceptable risks.

For workers it is possible to reflect the maximum level of fatal risk which is tolerated in industry and ICRP has done this to establish the dose limit for workers. An annual fatal risk of 1 in 1000 seems to be about the most that is normally tolerated in industry and this corresponds to the risk associated with the dose limit.

Industries that are regarded as very safe, such as shops and offices have annual fatality rates at or below 1 in 100,000. This means that for occupational exposure a trivial level of risk, or a level that is so low that no further optimisation is needed, would be where doses are below 1 mSv per year. Average annual risks in manufacturing industry are less than 1 in 10,000 which corresponds to a few mSv per year, and presumably reflects a degree of acceptability by workforces.

In the case of members of the public, it is much more difficult to decide what may be the level of unacceptability than it is for workers. A risk of 1 in 1,000,000 per year has commonly been regarded as trivial by many organisations and groups. A fairly widespread view seems to be that the maximum acceptable imposed annual fatal risk for a member of the public from a given source is around 1 in 100,000 and risks much beyond this probably verge on the unacceptable. ICRP has set the dose limit for the public at 1 mSv per year which represents a judgement on the borderline of the level of unacceptability. The associated average annual fatal risk is a few in 100,000 per year. This judgement is made bearing in mind the various attributes of the attributable risk from the radiation exposure, including lifetime risk, maximum annual risk, average annual risk and average loss of life expectancy. The Commission also took into account the existence of natural background radiation which gives an annual worldwide average effective dose of about 2.4 mSv. This natural background may not be harmless, but it makes only a small contribution

to the health detriment that society experiences. It may not be welcome, but the variations from place to place, excluding the large variations in the dose from radon in dwellings, can hardly be called unacceptable.

However, limits apply to the total exposure of individuals from all sources under control. The Commission has introduced the concept of a constraint which is a restriction of individual dose from a single source and is used in optimisation. A constraint is therefore seen as a prospective upper bound to optimisation that ensures the risk from that source is acceptable and the total risk does not approach the unacceptable.

It follows that there is no single value of a constraint but rather that there are practice-specific or even equipment-specific constraints. Values will be different as between, say, a nuclear power plant and a hospital diagnostic x-ray department.

In a number of countries a maximum value of a constraint has been set at 0.3 mSv per year corresponding to annual average fatality probability of 1 in 100,000. This figure would correspond to the tolerable level of risk for the public, while the trivial level of risk, 1 in 1,000,000 per year, would correspond to 0.03 mSv per year. This sets the acceptable level of dose somewhere in the range between 0.03 and 0.3 mSv per year.

So, how robust are the dose limits? It has been argued above that the risk coefficients are unlikely to change dramatically; thus, the major uncertainty is on the levels of risk that are likely to be tolerated. People are becoming more risk-conscious and reductions in acceptable levels might be anticipated.

MEDICAL ISSUES

One of the major issues regarding medical exposures is optimisation and the reduction of unnecessary exposures. Recently, reference levels have been introduced internationally as an indication of good practice in medical diagnosis and it will be interesting to see their effect. Within the UK, reference levels for common diagnostic procedures were specified some five years ago and a report published this year surveying 375 hospitals, showed a 30% decrease in average exposures in the last ten years. The overall exposure levels were in fact increased because of the increased use of CT and one of the greatest needs is for the development of reference levels for CT and an awareness of the need to ensure that it is only used when necessary.

Another medical issue is that of genetic susceptibility to which reference was made earlier. Here the problem seems more likely to arise in therapy than diagnosis, but the practical problems and ethical implications of screening will have to be addressed.

Interventional radiology also presents problems for protection as the millennium approaches. In this case there are deterministic effects on the patients from prolonged high exposures, often from cumulative, unrecorded treatments. In addition, the staff involved, while not approaching the limit on effective dose, are sometimes exceeding occupational deterministic limits for hands and lens of the eye. Optimisation here needs to be tackled with some urgency.

RISK MANAGEMENT

It is not simply in setting dose limits that consideration needs to be given to the question of acceptability of risk. In the discussion above on the threshold proposal in the dose-response relationship, it was indicated that effort should not be expended on fighting to prove that there are no effects at low doses since that is not the real issue. The real issue is risk management and this can be illustrated by examples of issues that are being addressed principally by Committee 4.

Potential Exposures

The question of the probabilities of accidents is covered by a topic that ICRP has called potential exposures. Restrictions on the annual individual risk from potential exposures should perhaps be of similar magnitudes to those for normal exposures. This leads to a risk criterion for potential exposure expressed as the annual probability of death attributable to a single source (installation) of 1 in 100,000. This would mean that the probability of receiving higher doses is accepted providing that the event has a correspondingly low probability. Thus, for example, a predicted effective dose of 100 mSv would have a fatal attributable probability of 3 in 1000 and the annual probability of receiving such a dose, without infringing the risk criterion, would be up to 3 in 1000.

What would be the acceptability of such actions. If the annual probability of an accident causing these consequences is 3 in 1000 ($3\ 10^{-3}$), as evaluated above, the probability that the accident will occur at some time in the lifetime of a single plant, say 40 years, is about 1 in 10 (0.1). For a programme of ten reactors, this implies that such an accident leading to a 100 mSv dose to a member of the public is almost certain to occur. However, in the range of doses from 10 - 100 mSv to a member of the public as the result of a nuclear accident, there will be additional considerations beyond those of individual risk. Regulatory authorities will have become involved and some form of off-site countermeasures (intervention) will be instituted. This frequency of intervention is almost certainly not tolerable and demonstrates that an individual risk criterion is not a sufficient requirement for managing those risks that can produce a significant impact on society from a large complex installation.

Individual risk will be a sufficient criterion for those sources that can affect only a small number of individuals, and for small accidents at complex plant such as reactors where off-site doses are less than about 10 mSv.

This leads to the question as to how to restrict the risk for larger accidents. There have been attempts over the years to develop `Societal' criteria to be added to the individual criterion. The difficulty in quantifying a single measure of societal risk is that there are so many different aspects to those consequences that arise from a major nuclear accident. There are late cancer deaths in the entire exposed population and most of these will be remote from the site; there may be early deaths close to the accident, if it is of sufficient magnitude; there will be disruptions from numbers of people evacuated and relocated; distribution of agricultural produce may be controlled; land may be interdicted; and there will be an enormous economic cost. How can all these different attributes be combined to set a target for design and control? Finally the aggregation of probability and consequence, which works for low consequence, high probability events, becomes increasingly less meaningful as probabilities decrease. For example, an accident with a probability rate of 1 in 1000 (10^{-3}) per year and an average consequence of 1000 deaths produces a mathematical expectation of one death per year. But the only available outcome is either zero

or 1000 deaths, with the most likely outcome being zero in any year. So the expectation value is of little use to us. Given these complexities, how can we manage risk for potential exposures and large accidents? A simpler, if cruder criterion of collective risk might be selected.

It is apparent that for some accidents the need for intervention becomes a more limiting feature of accident management than individual risk. Similarly for severe accidents, the societal risks directly related to the radiation doses becomes less important than other societal consequences. Thus there may be a limit to the probability of occurrence of major accidents because the consequences are sufficiently undesirable without quantifying those consequences. This approach requires the definition of serious accidents and the setting of risk criteria for the annual probabilities of such events.

Intervention

The intervention situation differs from that of the practice in that action is being taken to reduce the level of exposure as much as reasonable, whereas with the practice the objective is to restrict added exposures and the risk of being exposed. The two situations normally quoted for intervention are radon in buildings and countermeasures for the public after a nuclear accident. In both cases, protection measures taken are designed to lower the dose and hence dose limits do not apply. The question is by how much are the doses and therefore the risks to be reduced? Since there is some harm to the affected individuals from the disruption and risk of executing a countermeasure, our philosophy is to maximise the difference between the risk averted by the reduction of dose and the risk added by the countermeasure. In this scenario, there is no equivalent to the unacceptable level of risk although there must be a 'limit' to risk when the effects on the individual from the radiation are so severe, for example serious deterministic effects, that intervention is bound to be justified.

Simple countermeasures which involve little risk, such as sheltering, are recommended to avert doses of a few mSv, more disruptive countermeasures such as evacuation would not be contemplated unless doses of a few tens of mSv were averted, while international guidance on relocation sets a criterion of an averted dose over a lifetime of 1 Sv and the optimum figure for dose-rate at 10 mSv per month, say, 100 mSv per year. This figure seems reasonably robust considering that for radon in homes, ICRP has set an Action Level for simple remedial measures of, about, 1 mSv per month, or 10 mSv in a year.

The annual committed risk of attributable fatality associated with 10 mSv per year over a lifetime is about 3 in 10,000 ($3\ 10^{-4}$), and risk levels in excess of about one in 1000 (10^{-3}) per year are suggested as being acceptable before it is thought to be of overall benefit to relocate people from their homes. These figures may be compared with the risks thought unacceptable for practices.

Waste disposal

In the case of radioactive wastes the principles to be applied are those of optimisation and limitation. The principle of justification applied when the practice giving rise to the wastes was approved and would have involved considerations much wider than those of radiological protection. Governments justify the development of nuclear weapons, or nuclear submarines, or civil nuclear power, or the use of nuclear medicines to treat patients, on a broad basis of overall net benefit. The arisings of wastes are an inevitable consequence and their disposal does not need to be justified.

Committee 4 is considering a report on solid waste disposal. While it was felt that much of Publication 46 remained valid, it was recognised that there were questions over timescales and the use of collective doses. It was agreed that a short, concise, but comprehensible document on the application of Publication 60 recommendations as they apply to long-lived solid radioactive wastes should be produced and aimed at decision makers and their advisors involved in radioactive waste management. The Commission has now established a Task Group to undertake this work and a report is expected in 2 - 3 years.

The particular issues which the Task Group will need to address will be concerned with individual risk and collective dose. In the case of solid radioactive wastes disposed of deep underground, the question is how far into the future do you apply any individual risk criterion. Publication 46 gives no guidance and by implication the risk target applies for ever. In my opinion, given the increasing uncertainties on estimated risks as one goes farther into the future, one should put increasingly less weight on the associated individual risk.

The disposal of solid radioactive wastes does not provide a certainty of exposure, rather there is the potential for exposure at some point in the future. The individual is therefore best protected by a risk constraint, which can be set by analogy with the dose constraint. The maximum risk constraint from a single repository would then be set at no more than 1 in 100,000 per year. If the annual risk is predicted to be less than 1 in 1,000,000 per year, the risk could be regarded as trivial and there would be no requirement to demonstrate that protection could be improved. Between these two levels there would be the requirement to ensure exposures are as low as reasonably achievable.

The main input to optimisation of protection has generally been taken to mean the total (integrated) collective effective dose. However for solid waste disposal, the use of collective dose is far from ideal. This is because the collective dose is so dependent on detailed assumptions about the biosphere, human behaviour and population size that the results must be treated with extreme caution. A scenario can always be invented whereby a city of one million, or ten million people, or more, receive all of their drinking water from a river that flows directly through the repository. Publication 46 says collective doses should be used to choose between options, but if the absolute value of collective dose can be between zero and infinity, so can the differences. Collective dose seems to have no useful role in solid waste geological disposal. Perhaps the Task Group or members of the Main Commission will be convinced to the contrary.

Site-specific calculations relating to the biosphere and human behaviour cannot reliably be calculated for more than about 10,000 years and beyond this perhaps a reference hypothetical biosphere should be adopted and the risk constraint converted into a constraint on the radionuclide release rates from the geosphere. This would ensure the protection of future generations by assuming reference communities with habits broadly typical of those of subsistence communities in the present day.

The Commission meanwhile is reviewing the totality of its advice with regard to radioactive wastes, including effluent management. Once more the major issues are related to long timescales and collective doses. It may be that ICRP has aggregated too much information into a single quantity by integrating collective dose over all exposed individuals and over all time. If today's scarce resources are to be committed for future protection, it seems reasonable to wish to know how the doses are distributed in space and time. Let me be clear, I do not advocate a cut-

off in either individual or collective dose, but in order to make a decision I must know how many people get a given level of dose and when, so that decision makers may decide how much weight they wish to place on increasingly uncertain doses far in the future.

Restoration of contaminated sites

This is an area in which there is no international guidance and ICRP has established a Task Group to consider the issues. It seems that there can be at least three scenarios which may involve different approaches: the aftermath of an accidental release of radioactive material; the discovery of contamination at a site not known to have been contaminated, such as an old luminising plant; and lastly, the release of a decommissioned nuclear site.

Following the accidental release of radionuclides to the environment, intervention may be undertaken in the form of advice for the public to shelter, evacuate or in severe cases, to relocate. In this interventional situation dose limits will not apply and ICRP has recommended action levels, for example, the one for evacuation is set to avert a dose of more than about 50 mSv, while that to permanently relocate people is to avert doses above about 10 mSv per month. In this case decontamination of land may well be undertaken and the degree of decontamination may be decided by the principle of optimisation. The result of the optimisation may be to conclude that normal living should resume at a level of dose of a few mSv per year. This is above the dose limit for the public but it has to be remembered that those members of the public who were below the action level were not relocated and would have received nearly as high doses, and finally that the action level for radon levels in homes, where remedy involves minimal disruption, is about 1 mSv per month.

Collective dose in its normal sense is not applicable in an interventional situation although the number of people involved in a countermeasure does affect both its benefit and detriment and therefore is important in the optimisation. The criteria for intervention are therefore expressed in terms of individual dose averted.

In the case of old premises where radioactive contamination is found, the situation may be treated as an intervention by analogy with the ICRP advice on exposure to radon-222 since it will probably be the natural radionuclides which are involved and the premises are most likely to be industrial. An Action Level for the workplace can be set, below which the exposure is not treated as occupational. If the levels of dose are above the action level, the decontamination should be attempted to reduce the exposure below the action level. Only if the doses are above the action level should the workers be treated as occupationally exposed and dose limits applied. The action level at which intervention is certainly justified for Radon has been set by ICRP at 10 mSv per year and optimisation is unlikely to reduce this by more than a factor of three, so that the Action Level is expected to be set in the range between 3 and 10 mSv per year. The situation is less likely to arise for dwellings, but were it to do so, the same Radon philosophy might apply and an action level be set above which remedial measures should be undertaken, but the dose limit for the public would not apply.

When a licensed nuclear site is to be decommissioned and treated as unlicensed, the situation seems to be the same as a practice. The dose limits apply for the public and constraints can be set. Since during normal operation, a maximum constraint for environmental releases is not expected to exceed 0.3 mSv, it seems logical that the same figure, 0.3 mSv should apply as the maximum constraint for unrestricted release of the site. Similarly as the trivial figure is 0.03 mSv

and the exempt level is 0.01 mSv, these would appear logical lower bounds for constraints. There is no reasonable possibility of removing all traces of artificial radionuclides and it would be a mis-allocation of resources to try. Since the restoration is being dealt with as a practice, collective dose considerations will arise in the optimisation. In this case again, the collective dose may best be presented in ranges of individual dose and in discrete time frames so that in a multi-attribute decision process, less importance may be placed on very low doses or those received in the very far future.

CONCLUSION

In conclusion, there are several issues that are of importance in radiological protection that are primary policy issues for ICRP to deal with as the new millennium arrives. First and foremost, there is the increasing demand in some quarters to establish a threshold in the dose - response curve. However, at present it seems that DNA repair of double strand breaks is not error - free. It is fanciful to consider otherwise and anyway it is impractical to operate protection at low doses with a threshold.

The real issue to be addressed is what levels of risk are acceptable in different circumstances. There is a need to address normal and accident situations, workplaces, homes and medical situations.

The use of collective dose needs to be reconsidered and perhaps presented as a disaggregated quantity for decision making with regard to environmental releases and the treatment of risk far in the future needs to be clarified.

Whoever is the Chairman of the Commission in the new millennium, and whatever it's membership, I think that they will live in interesting times.

CURRENT ACTIVITIES OF THE NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS – CONTROVERSIES AND ALL

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Abstract

The National Council on Radiation Protection and Measurements (NCRP) and its predecessor organizations have been providing advice on radiation protection and measurement issues since 1929. The Council received a Congressional Charter in 1964 which predicates its structure and operation.

There are 75 Council members and more than 400 scientists working on about 40 Scientific Committees dealing with a wide variety of topics in the following program areas: Radiobiology and Epidemiology, Operational Radiation Safety, Metabolism and Dosimetry, Environmental Issues, Radioactive and Mixed Waste, Nonionizing Radiation, Radiation Protection in Medicine, Public Decision-Making and Radiation Measurement. As might be expected, several of the topics are the source of various levels of controversy. The health effects of ELF radiation and the adoption of the linear no-threshold model are among the most controversial at the present time. The approach to resolving these issues reflects the strengths in the NCRP Program.

The NCRP's relationship with the ICRP is strong, although the NCRP, while endorsing the basic quantities, units, and risk estimates, will occasionally make alternate recommendations as suggested by its Scientific Committees and approved by the Council members.

I am delighted to have the opportunity to tell you about the NCRP. I will begin with a little history, explain what and who we are today, explain our objectives, functions, and mode of operation, and finish with a discussion of a few of our recent reports, and a review of a couple of our current activities and controversies.

In 1928, the newly formed International Congress of Radiology's Advisory Committee on X-ray and Radiation Safety, the forerunner of the International Commission on Radiological Protection (ICRP), told the U.S. representative, Lauriston S. Taylor, to go back to the United States and form an organization which could bring a consensus opinion to the Advisory Committee. Dr. Taylor returned and formed the U.S. Advisory Committee on X-ray and Radium Safety in 1929 under the sponsorship of the National Bureau of Standards. This was largely made up of medical society representatives. After the war in 1946, it became the National Committee on Radiation Protection and Measurements, and in 1964 received a charter as the National Council on Radiation Protection and Measurements from the Congress. Dr. Taylor was Chairman/President from 1929 to 1977.

The four objectives given in the charter are the driving force for our activities. These objectives are:

1. Collect, analyze, develop, and disseminate in the public interest information and recommendations about (a) protection against radiation and (b) radiation measurements, quantities, and units, particularly those concerned with radiation protection;

2. Provide a means by which organizations concerned with the scientific and related aspects of radiation protection and of radiation quantities, units and measurements may cooperate for effective utilization of their combined resources, and to stimulate the work of such organizations;

3. Develop basic concepts about radiation quantities, units and measurements, about the application of these concepts, and about radiation protection;

4. Cooperate with the International Commission on Radiological Protection, the International Commission on Radiation Units and Measurements, and other national and international organizations, governmental and private, concerned with radiation quantities, units and measurements and with radiation protection.

Based on these objectives, we see that our function is first to provide reports, commentaries, and statements on topics of importance in radiation protection. With regard to our relationship with government, we address specific issues as requested by each of the federal agencies with responsibility in a given area. It should be noted here that the NCRP only provides recommendations. We are not a part of the regulatory process. The NCRP also prepares documents which relate primarily to the public. Things such as our public symposia and our newly instituted committee on public policy are two examples of this. Our traditional role is to provide guidance for the users. That is, guidance documents for physicians, medical physicists, health physicists, administrators, and the regulators. Our reputation and our value to the radiation protection community come from the scientific consensus obtained from the 75 Council members and from the work of the more than 400 scientists, physicians, and engineers working on our forty committees. The Council members are elected for six-year terms, and approximately six new Council members join us each year. We are organized based on a number of what we call scientific program area committees which are constituted to develop a program in the specific area and make recommendations on the work of the Council to the Board of Directors. These committees are chaired by a Scientific Vice President of the Council for the particular area over which he has responsibility. These committees are on:

Radiobiology, Epidemiology and Basic Criteria, chaired by S. James Adelstein
Environmental Issues, chaired by John E. Till
Radiation Measurement, chaired by Harold L. Beck

Metabolism and Dosimetry, chaired by Bruce B. Boecker
Nonionizing Radiation, chaired by Thomas S. Tenforde
Operational Radiation Safety, chaired by Kenneth R. Kase
Public Policy and Decision Making, chaired by Paul Slovic
Radiation Protection in Medicine, chaired by Fred A. Mettler
Radioactive and Mixed Waste, chaired by Donald G. Jacobs

The output of all of our effort, of course, is our publications. The most important of these are the NCRP reports, which, as we will see later, get a very rigorous review. For minor issues which need a very quick turn around, we publish NCRP Commentaries. These, however, do not get the review inherent in an NCRP Report. NCRP statements are a third category of publication which are very short but which have the same review characteristics as do the NCRP reports. In addition, we do publish our Annual Meeting Proceedings and Proceedings of special conferences. I think it is very important to understand the rigorous approach we take as to how reports are generated and as to how the reports are reviewed. For instance, a report only comes into being after it has first been proposed either by a program-area committee of the kind I just mentioned, or if it is a request from one of the funding agencies, or one of the Collaborating Organizations believes that it is an item which we should be reviewing. Of course, any individual may bring a proposal to us. All such proposals must then be approved by the Board of Directors. The Board consists of eleven Council members elected annually by the members. They will review the scope of the project normally prepared by one of the scientific program area committees and will appoint a chairman, normally suggested by the program area committee as well. Once the project has been approved, the secretariat of the NCRP goes about trying to obtain funding. The chairman then works with the program area committee to develop a list of candidates for committee membership which are presented to the Board of Directors which approves the members. Once the members are appointed and the committee is underway, they have full autonomy in their work. In fact, we believe that the autonomy of our scientific committees in their efforts to prepare a draft document is a keystone to the objectiveness of the scientific effort. Once the report has been completed by the committee, in draft form, it is sent out for a review by a selected group of critical reviewers. These are normally selected from the Council membership, although if there are not sufficient members of the Council who are well versed in the topic under consideration, we will go outside the Council for this review. Once the report has been revised based on the critical reviewers comments, it gets a final review by our 75 Council members. The 75 Council members will vote on the report and, in fact, no more than two people of the 75 can vote against the report if it is to move directly to publication. In addition to the 75 Council members, however, there are 39 Honorary members, 60 collaborating organizations, and 13 special liaison organizations who are given an opportunity to review these reports. All of their comments receive careful consideration in revising the report after review. This review process applies not only to reports but also to our statements.

As I mentioned earlier, the Commentary review process is less intensive. These documents are only reviewed by the Board of Directors, although at times they will also have critical reviewers who will be designated as selected experts.

I think it might be helpful if I review some of the existing reports and commentaries that may be of interest to the Nordic Society and then discuss reports in preparation, particularly those which address controversial issues. I will limit this particular presentation to the period of 1995 and beyond, although NCRP report 116 published in 1993 on *Limitation of Exposure to Ionizing Radiation* is of particular interest in that it reflects our approach to adopting ICRP recommendations. For example, we note specifically in that report that "the recommendations and concepts provided in ICRP Publication (ICRP, 1991a) have been carefully reviewed and in the interest of a uniform international approach to radiation protection have, in general, been incorporated in this Report. Deviation from their recommendations was deemed necessary in a few cases where greater flexibility could be obtained at similar or less risk (e.g., the occupational dose limits) or where increased protection was considered to be warranted (e.g., a monthly exposure limit for the embryo-fetus." It was noted that the publication also contains a table providing "a comparison of the radiation risk data, recommendations, and other factors used in NCRP Report No. 91 (NCRP, 1987) and ICRP Publication 60 (ICRP, 1991a) with those used in this Report." For example, the recommendation on dose limits contained in NCRP Report No. 116 provides more flexibility than that suggested by the ICRP in its Publication 60. ICRP had recommended a limit of 100 mSv in 5 years and no more than 50 mSv in any one year. The NCRP report recommended your age in tens of millisievert and 50 mSv per year which provides the flexibility for the further utilization of radiation and radioactive materials while still recognizing the need to keep lifetime exposures to individuals below 1 Sv as implied by ICRP recommendations.

NCRP Report No. 121 on "Principles and Applications of Collective Dose in Radiation Protection," was the first report issued in 1995. The report assesses the usefulness and limitation of collective dose in radiation protection. The report reviews the historical element of the term collective dose and then goes on to give the scientific basis for the use of collective dose which is summarized in the form of a major compilation of available human radiation epidemiological data. Limitations on the use of collective dose in radiation protection are then reviewed followed by a section on risk assessment and management together with conclusions and recommendations. They might be summarized as follows: The collective dose can be very useful in assessing the acceptability of the practice but not helpful as the basis for regulatory limitation. Collective dose is appropriately accurate when the demographic data is well known, whereas it is just as uncertain when the demographic data is uncertain or unstable. Also, the collective dose is limited to doses in the stochastic region where the assumption of linearity would pertain and should not be used in the areas where you would expect deterministic (nonstochastic) effects. The question of how the collective dose applies to exposures in the future is resolved when the discussion revolves around using collective dose when you have information about the demographics of the society or individuals that are summed in the collective dose estimate. It is therefore clear that as you go far into the future you won't have this information, and the use of collective dose for that purpose is probably inappropriate. Perhaps the most interesting recommendation is that when the collective dose is smaller than the reciprocal of the relevant risk coefficient, the risk assessment should note that the most likely number of excess cancer deaths is zero. For example, for the public, the fatal cancer risk is about 5×10^{-2} Sv; therefore, the collective dose is less than 20 Sv, the number of cancer deaths is less than one. ($20 \text{ Sv} \times 5 \times 10^{-2} \text{ Sv}^{-1} = 1$).

The second report issued in 1995 was NCRP Report 122 "Use of Personnel Monitors to Estimate Effective Dose Equivalent and Effective Dose to Workers for External Exposures to Low LET Radiation." The basic premise is that limitation based on effective dose means that the partial body exposure should reflect the risk based on the dose to the tissues exposed. Of particular interest is the recommendation on estimating the effective dose equivalent or the effective dose when a protective apron is worn during fluoroscopy. For a single dosimeter worn at the neck, one simply can divide the dosimeter estimate by 5.6 to obtain a conservatively high estimate H_E, or divide by 21 to obtain a conservatively high estimate of effective dose. If two personnel dosimeters are used in this situation, one under the apron and one above, the following relationships are recommended. $H_E = 1.5\ H_w + .04\ H_N$, where w is a subscript to denote the badge worn on the waist or chest and N to designate the badge worn on at the neck. The effective dose, E, is equal to $1.5\ H_w + 0.025\ H_N$. (It is my hope that this report will encourage other investigators to explore this issue for the spectrum of exposure situations.)

In this year the NCRP has released its report 123 on "Screening Models for Releases of Radionuclides to Atmosphere, Surface Water and Ground." This is a two-volume report which seeks to meet the need for simple authoritative screening techniques to address release of radioactive materials to the environment. The techniques can be employed to demonstrate compliance with environmental standards for release of radionuclides to the atmosphere, surface water, or ground. The report provides the derivation of screening models for the atmosphere and water, the derivation of source terms, the matter of usage factors which relate to the consumption of contaminated water and food by humans. The report provides worksheets which allow the user to easily carry out screening processes for a proposed release via a few multiplicative calculations using a minimum of site specific data and decisions.

NCRP's Scientific Committee 64 on Environmental Issues has a number of active working groups and a strong program plan for the future. The active committees are:

Scientific Committee 64-18	Ecological and Human Risks from Space Applications of Plutonium
Scientific Committee 64-19	Reconstruction of Historical Dose
Scientific Committee 64-20	Recommended Screening Limits for Contaminated Soil
Scientific Committee 64-21	Decontamination and Decommissioning of Facilities

The future plans of the committee are for work on:

- Design of an effective environmental monitoring program
- Contaminated ecosystems - remediation or management
- ^{137}Cs in relation to environmental remediation
- Case studies and lessons learned from remediation
- Radionuclides in the environment
- Radionuclides in environmental media

Of the work underway, perhaps the two closely related efforts in the area of our committee on Radiobiology, Epidemiology, Risk and Basic Radiation Protection Criteria should be of interest. The first is on the extrapolation of risks from non-human experimental systems to man, under the chairmanship of David Hoel. I have always been concerned that although we use animal, plant, and cell data to make our estimates of genetic risk and that associated with neutron radiation, there has always been reluctance to use anything but human data for our other risk estimates. In fact, we have no more data on the risk at very low doses, say below 5 mSv, then we have about neutrons or genetic effects. It is important, however, that the extrapolation be based on the best scientific approach, i.e., looking for comparable mechanisms influencing cancer induction rather than simply ratios of size or lifespan. This report will attempt to do exactly that. The second effort is on the examination of the scientific basis for the assumption of linearity at low doses which is under the chairmanship of Arthur Upton. This is an exciting project, and the committee hopes to get input from the entire spectrum of scientific information on this topic. A request for information has been published in scientific journals and, in addition, we hope to invite a wide spectrum of the research community to address the committee in person. This, of course, relates to the Health Physics Society's recent Position Statement. I have appended my edited version of that statement for discussion perhaps at this meeting. While on the subject of controversies, the NCRP's recent world wide press attention over our "recommendations on ELF" was perhaps one of our most difficult issues. In fact, there were no NCRP recommendations on ELF. This issue is contained in a press release which is attached to this report. I should only say that at this stage, the document in which the recommendations were contained, has now gone to some twenty-five critical reviewers. These reviews have been completed and the scientific committee responsible for that draft is working to resolve those comments. Once the comments have been resolved, it will go into the review process which I mentioned earlier in this paper. As I mentioned earlier, we have more than forty committees and I believe that virtually every one of them would be of interest to this audience. However, in terms of the time available, I think perhaps we have had enough.

ELEMENTARY RISK CONCEPTS

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Abstract The paper reviews elementary risk concepts and, in particular, the meaning of probability in risk assessments. The author makes a distinction between "macroscopic" and "microscopic" risks. With a *macroscopisk* risk he means any risk in a situation where a frequency generating mechanism can be identified or postulated. In contrast, with *microscopic* risks he means risks in situations where no frequency generating mechanism can be presumed. In the latter case, relevant for unique events relating to one particular individual or installation, the probability of a harmful event is entirely based on the subjective belief of whoever states a probability value, conditional to that person´s knowledge and experience. The evaluation of radiation risks is an interesting example.

What is "risk"?

"Risk" refers to something uncertain and threatening which might happen in the future. To the layman risk is no quantitative concept but merely a diffuse forebode of something unpleasant. To the professional "riskologist", risk is a compounded concept involving two types of components: *probabilities* and *consequences*.

Risk implies the possibility of any one of a set of unpleasant events E_i, each of which can be assigned a probability P_i and a consequence Q_i. Any attempt to treat a risk as a scalar quantity that can be described by one single number will therefore require subjective assumptions. In order to simplify the discussion, I shall here usually only deal with risks involving one particular type of event E with probability P and consequence Q.

Risk management involves steps for which there are established terms as follows:

> **risk assessment** involves *hazard identification* and *quantification* of quantifiable components (risk estimation).
> **risk evaluation** involves a *judgement* of the risk in connexion with decision making in the given situation.

It is important to realize that an assessed risk must be judged in its context. To ask whether a certain risk is acceptable is as nonsensical as to ask whether a stone is too heavy. The question cannot be answered unless the situation is described. What is in fact accepted in a given situation is not the *risk* (except indirectly) but the *situation* or *practice* that causes the risk. This means that the acceptance does not only depend on the degree of risk but also on a number of other factors, such as perceived benefits, costs, positive or negative attitude to the practice, etc.

In the following, I shall distinguish between two types of risk which I have given the homemade names "makroscopic" och "microscopisk".

Macroscopic risks

With a *macroscopisk* risk I mean any risk in a situation where a frequency generating mechanism can be identified or postulated. Frequency usually means number per time, but here it may also mean the number of specified events per trial. In throwing a homogeneous die, there is symmetry between the possible outcomes. The frequency of, for example, sixes can therefore be assumed to approach 1/6 of the total number of throws, and throwing the die defines the

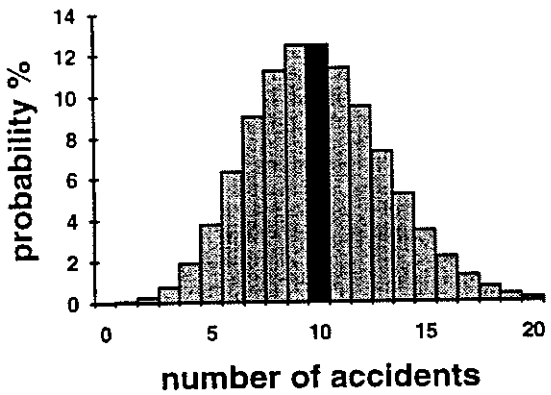
frequency generating mechanism. The registered number of impulses per time from beta particles hitting a measuring instrument from a radioactive source also implies a frequency generating mechanism, but here the frequency cannot be postulated but must be assessed by observations..

I both cases it is possible to assess a probability according to traditional definitions. In the case with symmetry between the possible outcomes, the probability, by definition, is the ratio between the number of special events and the total possible number of events. In the case of the beta particles, the *probability rate*, *i.e.* the probability per time, is the limit of the quotient of the number of registered pulses and the measurement time as the time tends to infinity (assuming, of course, that the decay of the radioactive source is negligible). In both cases, the mathematical expectation of the number of events (sixes, registered pulses) for a given number of throws or a given measurement period can be calculated by established methods. The actual number of events (n) may often be assumed to be Poisson distributed. If a denotes the mathematical expectation, each number n will then have the probability

$$P(n) = e^{-a} a^{n}/n!$$

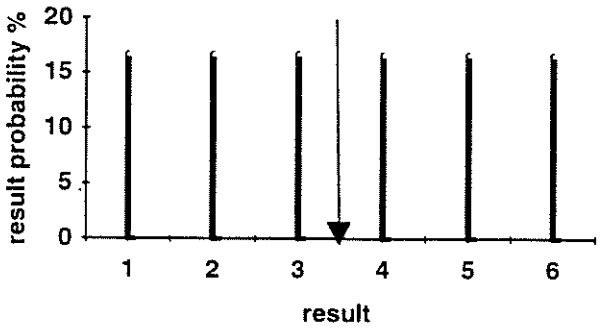
The value of a may be assessed by experiments and the assessment may be further improved by additional information. *In the case of macroscopic risks, a > 1.*

If the total number of events in a series of trials is such that it may also be seen as the consequence (number of accidents, injuries, deaths) we have a situation where n is the consequence of a situation that has the probability P(n). The mathematical expectation of the consequence is a. In the case of, for example, a = 10 (*e.g.* the expected number of accidents within a given area over a specified period of time), the probability density of the magnitude of the consequence will be:



Here, the mathematical expectation (a = 10, marked black) has a central position in the distribution and is a number which is at least as probable as any other number. It is also likely that the real outcome will not be very different from the expectation. The mathematical expectation can therefore be used for planning purposes and as the basis for decisions. The standard deviation (σ) from the mathematical expectation is less than the expectation as long as a > 1. The relative uncertainty σ/a is then less than 1.

The trivial fact that the mathematical expectation is not necessarily equal to the actual outcome, or even any of the possible outcomes, is of some importance for my presentation in the next section. For example, the mathematical expectation of the result of a throw of a die is 3.5 which is obiously not a possible result:



The mathematical expectation may be calculated as the product of the probability P and the number of "opportunities" N, individuals (humans or machine elements) or time elements, *i.e.* $a = P \times N$. Sometimes the actual outcome n may be seen directly as the consequence as in the previous example, but sometimes each one of the n events may be such (*e.g.* a major accident) which on the average causes a secondary consequence q (*e.g.* economic loss, or number of people killed per event). In that case, the outcome has the total consequence $Q = q \times n$ with the mathematical expectation $\overline{Q} = q \times a$ and the standard deviation $q \times \sigma$.

The interpretation of the probability concept causes some problems. Let us look at a situation where we want to assess the annual probability of death in a population of N individuals when we know the mathematical expectation, a, of the number of deaths. The consequence is *death* and the individual probability of death in the year in question is $P = a / N$. But for which individual is this valid?

Most populations are heterogeneous. Therefore, the probability of death in a given year in a human population must relate to the *average individual* in the population in order to make the relation $a = P \times N$ valid. In reality, few individuals, if any, are average individuals. For a real individual, identified in beforehand, it is unlikely that he or she is an average individual. If we nevertheless wish to use the probability $P = a / N$ in an individual case, we have to *postulate* that the individual runs the *average* risk.

A capable medical doctor who examines all the N individuals may perhaps identify a sub-population of N_1 individuals as carriers of the total risk. In this sub-population the death probability for the average individual will be $P_1 = a / N_1$, with $P_1 > P$. The more knowledgeable the doctor-examiner is, the better can he confine the risk-carrying part of the population and the greater becomes the probability P_1 for the average individual in the identified sub-population. If the examiner were omniscient, the risk-carrying sub-population would be identical to that comprised exclusively of those who will die. The probability is then replaced by certainty.

The concept of risk, however, presumes uncertainty. This means that any probability P_1 that we state as valid for a given individual is conditional to the *assumption* that the individual belongs to a sub-population for which that probability is valid. If our experience is limited to observations on the total population of N individuals and we have no further information about the designated individual, we can only assume that the probability $P = a / N$ applies. If we have some information about the individual in question, we may improve our probability assessment to P_1. However, any value we select for the probability in the individual case is conditional to our subjective view on the individual's risk. A doctor may say that the probability that a given

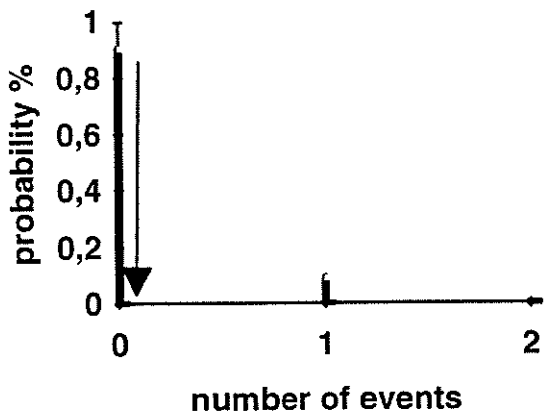
patient will die within a year is 100 per cent, because he assumes that the patient belongs to a sub-population for which experience has told him that this is the case. An outsider, who knows that doctors sometimes are wrong, may doubt the assessment of a 100 per cent probability and believe in a lower probability.

In the individual case, there is no "true" probability that a selected person will die within a year except zero and 100 per cent, corresponding to the only possible outcomes. Any other probability stated in beforehand will prove to be "wrong". However, it is possible to state a "true" probability for the *average* individual in a given population, because the stated probability will imply the generation of a frequency the existence of which can be verified by later observations. The in beforehand selected individual, however, is unique - he will either die or survive. It is not even possible to make a conceptual experiment with a population of exactly identical individuals. If the individuals and external conditions are *exact* the same, the fate of each individual will be exactly the same, they will all either die or survive. The exact identity will rule out the mechanism of chance that would have generated a frequency. But if the individuals are made different, to introduce an element of chance, this can be done in an infinite number of ways with an infinite number of different results. As soon as we permit chance, we have abandoned the unique individual.

The conclusion is that we rarely if ever deal with undisputable probabilities but rather with a probability stated by someone and depending on the reliability of the assumptions which are *always* behind probability statements. Often the uncertainty is not in the probability assessment but in the validity of the assumptions behind it. When experts are ridiculed because their stated probabilities seem to be in conflict with reality, it is not because the assessments were wrong (*e.g.* the very low probability that Titanic would sink) but because the assumptions were wrong (icebergs were not anticipated). Unfortunately, those who assess probabilites either take the assumptions for granted or are not even aware of their existence.

Microscopic risks

With *microscopic* risks I mean any risk in situations where no frequency-generating mechanism can be presumed. To the extent that it is nevertheless postulated, *e.g.* by assuming a Poisson distribution, the mathematical expectation of the number of events over available periods of time is less than one ($a < 1$). This is illustrated by the case that $a = 0.1$:



The fact that, in this case, the mathematical expectation is $a = 0.1$ event, however, is not too misleading, because no one is likely to believe that 0.1 event will happen (just as no one would expect a die to show 3.5). It is of course realized that the most likely outcome is 0 events (with a probability of about 90%) but that, with a probability of about 9%, there may be 1 event, and, with a very low probability, 2 or more events.

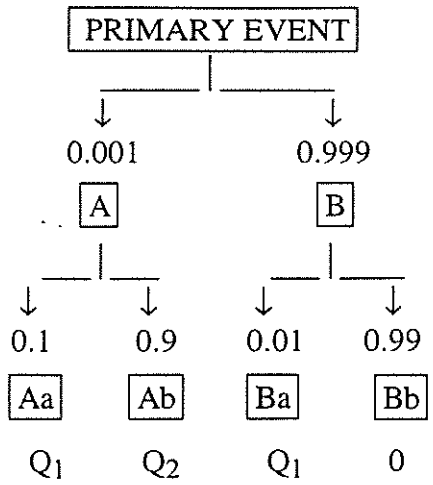
However, the perception will be different when we talk about *consequences*. Let us assume a situation where the probability P of an event is very small, say 0.00001 per year, but that 300000 persons would be harmed if the event occurs. The mathematical expectation of the annual number of *events* will be 0.00001 (which immediately tells us that the most likely outcome is zero events), but the mathematical expectation of the *number of persons* harmed willl be $0.00001 \times 300.000 = 3$.

While it is possible in the case of macroscopic risks to use the mathematical expectation as the basis for decisions, this is no longer possible at microscopic risks. It is meaningful to plan for handling 10 accidents per year as in my case with Poisson distributed events, because we know that there will be between 7 and 13 accidents most years. But to plan for 3 cases of injury per year is not meaningful: either there will be 300000, *i.e.* a big catastrophe, or no one harmed at all (with a 99,999 % probability). The usefulness of the mathematical expectation in the case of macroscopic risks has unfortunately invited the same routine for microscopic risks. Particularly in reactor safety assessments, the mathematical expectation of consequence has been used synonymous with "risk", but to talk about, for example, the mathematical expectation of a collective dose from a reactor accident with a very low probability is quite misleading.

The reason that I use the term "microskopisk" risk in this latter case is not that the risk is very small but that we are interested in a period of time (*e.g.* a year or the lifetime of an installation), which is small in relation to the inverted value of the probability rate. To speak about expected 10 accidents per year was meaningful, but to speak about 3 injured persons per year when this is derived from 0.00001 accidents per year is not meaningful The mathematical expectation of the number of injured persons becomes meaningful first if we are interested in this number over time intervals of more than 1/0.00001 year, *i.e.* up towards periods of a million years - and that is rarely the case.

It is obvious that the microscopic risk relates to unique situations because frequency experience is missing. That 4000 reactor years in the world has caused two serious accidents (TMI och Chernobyl), could be taken to imply an annual probability of the order of 1/2000 for the average reactor which, of course, does not exist. The Swedish reactor safety study of 1979 suggested an annual probability of about 1/10000 for a (less severe) accident in the average Swedish reactor, *i.e.* an annual accident probability of about 1/1000 for the total Swedish reactor program. For mathematical expectations of the corresponding consequences to be meaningful, they must be expressed over periods of several thousand years, while we are interested in what will happen on a "microscopic" scale, *e.g.* over the lifetime of the program. We are then back at the basic rule: risks comprise probabilities *and* consequences. Both must be assessed and presented separately to prevent the presentation from being misleading. Probabilities may be reduced by safety promoting measures (*e.g.* quality assurance), consequences may be reduced by mitigating systems (safety belts in cars, fire alarms in buildings, and vented containments in nuclear power stations).

When there is no prior experience of accident frequencies relevant to the situation which is at hand, the risk (probabilities and consequences) must be assessed by means of some form of analysis of conceivable sequences of events. Such analyses are common in air plane and reactor safety studies by means of "event trees" och "fault trees". In an event tree analysis the assessments starts from a primary event and continues by describing conceivable secondary events and the probabilities that can be assigned to each of the various branches of the event tree.



Given the primary event, the probability of consequence Q_1 (which may occur via either of two different branches) will be:

$$P(Q_1) = 0.001 \times 0.1 + 0.999 \times 0.01 \approx 0.0101$$

And the probability of Q_2:

$$P(Q_2) = 0.001 \times 0.9 = 0.0009$$

The mathematical expectation of consequence is:

$$\bar{Q} = 0.0101 \times Q_1 + 0.0009 \times Q_2$$

The event tree analysis is a very valuable method for identification of weak points in conceivable sequences of events and is therefore an important tool in safety management. However, the probabilities which may be calculated for various consequences (as well as the expectation of consequence) are uncertain since they are conditional to a number of assumptions. For some branches of the event tree there may be frequency experience of the reliablity of components such as valves, pumps, switches, etc., but it must still be postulated that this experience is relevant also for the future, which is by no means certain (there might be a new manufacturer, changes of material, inferior quality assurance, etc.). For other branches there may be no frequency experience and the probabilities will be subjective, based on knowledge and judgement (*e.g.*regarding the possibility of human errors). Each step in the event sequences is therefore subject to uncertainty, the biggest uncertainty being that some significant branch may have been neglected.

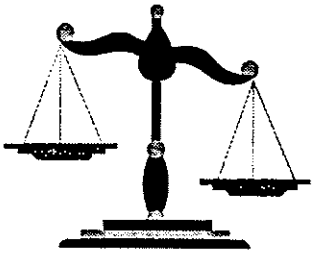
In a *fault tree analysis* the direction is the opposite. The analysis starts from a postulated accident and the task is then to review the faults that could have caused it. The fault tree analysis is also an inportant tool in safety wotk.

That the mathematical expectation ($\bar{Q}$) actually cannot be expected in cases of low probability is also demonstrated by the fact that the relative uncertainty ($\sigma/\bar{Q}$) is greater than 1, *i.e.* the standard deviation in the distribution of consequence magnitudes is larger than the expectation. However, there are also other factors which make the mathematical expectation of consequence unsuitable as a basis for decisions. The mathematical expectation is the sum of products of

probabilities and cosnequences. The requisite for this product to describe the "risk" is that it is considered immaterial whether a fractional change of this product is caused by a change of the probability or a change of the consequence. However, we usually attach more importance to the consequence, a fact which is demonstrated by our willingness to buy tickets to a lottery well knowing that the expectation of winnings is always below the ticket price (it would be simple to increase the expectation of winnings by promising all participants a prize equal to the ticket price, but who would then by a ticket?).

Risk evaluation

It was said in the beginning that it is not possible to determine whether a certain tisk (combination of probabilities and consequences) is acceptable without seeing the risk in it context. This means that it is not the risk in isolation but rather the practice and situation which is judged.



To judge whether a practice or a situation is acceptable involves balancing advantages and disadvantages. Both may be seen differently by different perrsosns. Risk is only one of the disadvantages, another one is the cost of the practice.

The perception of advantage and disadvantage differs between individuals. Their disposition, upbringing, education, training, social position, etc. will form their attitude, positive or negative, to a practice and will also influence their perception of risk and even their assessment of probabilities and their picture of the consequences.

There are many more factors which shape people's attitude to practices and situations which involve risk. Some of the most commonly mentioned factors are whether the risk is voluntary or imposed, whether the individual has a feeling of control, whether those who cause the risk or assess it are trusted, whether the nature of the risk is well known or difficult to comprehend, and whether the risk originates in human actions or is due to natural phenomena.

All this will influence people's risk perception, but should perhaps rather be seen as influencing their attitude to the risk-generating practice or situation. The willingness to accept the source of a risk seems to be rather little influenced by reassuring information about the magnitude of the risk. However, the willingness to accept a practice is influenced by attitude-shaping factors, *e.g.* by increasing people's influence and feeling of control or by demonstrating the reliability of those who cause or assess the risk.

A special question is whether there are practices which involve a risk with so large societal consequences, should an accident occur, that the practice should not be permitted even if the probability of the catastrophic consequence is extremely small. Behind this there may be a hidden distrust of assessments of very small probabilities (when somebody says that a probability is one in a million, it must be remembered that the person in question must be 99.9999 per cent confident in his assessment for the probability assessment to be useful). Prohibition of practices with too large potential societal consequences have been discussed in the Netherlands. The idea seems reasonable, but the practical application involves logical problems: Any human action is potentially capable of leading to an enormous catastrophe, it does not take much imagination to invent a scenario to that effect. If it is prescribed that no scenario may be omitted merely because of its absurdity (*i.e.* low probability), then nothing could be permitted.

A possibility of guidance may be found in fault tree analysis. If the assessment begins with a catastrophe which makes the city of Copenhagen uninhabitable for decades, a reactor accident at the Barsebäck nuclear power station is one of several possible causes, perhaps not more

unrealistic than other possible causes, *e.g.* a nuclear war. A limit of the magnitude of the possible consequence might therefore justify a prohibition of that power station. However, if someone claims that a child's play with a ball might result in a world war (the ball may roll out into the street and cause a traffic accident which kills a person headed for an important political appointment ... etc.), the fault tree analysis will show that other causes of a world war ar far more likely and that the child's play does not significantly alter the risk of war.

Evaluation of radiation risks

There are obvious examples of macroscopic and microscopic radiation risk. The risk in the case of a low-dose exposure of a large population may be a macroscopic risk (if the mathematical expectation of radiation harm is larger than the standard deviation). The risk of a major nuclear accident at a particular nuclear power station is a microscopic risk (the matematical expectation of the number of such events over a relevant period of time is less than one). In the latter case, there is no way of deriving an accident probability by means of freuqncy observations, any probability statement will be an expression of personal beliefs.

The first case is interesting because the mathematical expectation of events (*e.g.*deaths from radiation-induced cancer), even if it were larger than one, may not be detectable, for statistical reasons, against the random variation in the overall number of cancer deaths. Some people feel that what you cannot see should be of no concern. But they are then making the wrong comparisons.

There are three valid comparisons which may be made. A small increment to an individual's risk may be negligible to the individual because it is small in relation to the total, mostly unavoidable risk. For the same reason, it may be negligible to the society, because any society is the sum of its individuals. These two comparisons are quite valid and are often made and taken as an excuse for ignoring a small risk. But there is a third valid comparison. The small risk increment may be related to the *source* of the risk. That source may be a practice or an installation which brings little benefit. It might also be possible to improve the protection at the source. The valid comparison is then in relation to the source, to see whether the risk justifies the benefit or whether it is reasonably achievable to eliminate or reduce the risk. If some lives may be saved at a reasonable cost, why not save them even though many more people die for other reasons?

There is at present no way of proving whether the ICRP assumption of a non-threshold linear dose-response relation at low doses is valid, but there are a number of good reasons to believe that this is the case. Some people oppose to this assumption, often on the basis of "gut feelings" or in the belief that the postulation of a dose threshold would simplify radiation protection. However, it is very likely that, if a dose threshold could be demonstrated, linearity might still have to be postulated because the administration of radiation protection to observe a dose threshold would otherwise be extremely complicated (radiation doses from *all* sources, including natural and medical would have to be included under any dose limits)..

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COMMON SENSE, RISK ANALYSIS AND REGULATIONS

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Abstract - In times of limited resources, regulatory schemes are bound to come under close scrutiny. In the ideal world, scarce resources would be directed where they would do the most good. Recent tensions in the United States concerning budget and regulation have elicited much debate over a particular mode of regulatory decision-making called risk analysis, which is frequently espoused as the panacea to the often-voiced complaint that many regulations make no sense and frequently inhibit the goals they are designed to achieve. Risk analysis is an extremely valuable tool which, when used appropriately, can restore common sense to regulation.

Essentially, risk analysis can be divided into three distinct components: risk assessment, which attempts to measure and describe the risk or hazard associated with an activity or the use of a substance or technology; selection of an acceptable level of risk; and, risk management, the process and infrastructure used to keep the risk below the level determined to be acceptable. Each of these components involves an intermix of science and value judgements and cannot be left solely to experts but must in some way include politicians and the public if the approach is to succeed. Superimposed upon the approach of risk analysis are the needs to prioritize risks to assure that attention and resources are devoted to the regulation of risks that are truly significant and not to further reductions of minimal risks, and, to assure a uniform approach to risk-based decision-making among the various regulatory bodies particularly with respect to how common sense is inserted into the process.

The immediate challenge is to utilize the logic of risk-based decision-making to develop a practical overarching regulatory framework. Not a challenge for the meek, but one that must be met if both the economic and environmental well-being of our societies is to be assured in the future.

Introduction

In times of limited resources, regulatory schemes are bound to come under close scrutiny. In the ideal world, scarce resources would be directed where they would do the most good. Recent tensions in the United States concerning budget and regulation have elicited much debate over a particular mode of regulatory decision-making called risk analysis.

My talk is intended to respond to the complaint that too often regulations make no sense and actually inhibit the goals they are designed to achieve. Risk analysis provides an extremely valuable tool which, when used appropriately, can restore common sense to regulation and hopefully increase the confidence of the public in their government.

The Death of Common Sense

First, though, we need to understand the problem. It's clear that a broad segment of the American public believes that we are over-regulated, and that the way to cure this is through extensive de-regulation. It also seems clear that many believe that even where regulation may be needed, such as in the fields of health and safety and environmental regulation, our priorities are skewed. Resources are misdirected to minor problems in such a way that the costs of regulation don't justify the benefits to be derived.

Two authors have recently provided excellent summaries of the regulatory climate that has resulted in the present anti-regulatory mood. Philip Howard, in his 1994 book entitled The Death of Common Sense: How Law is Suffocating America[1], gives a variety of examples of how red tape at all levels of government puts a stranglehold on the actions of individuals and companies, including, for example, the nobel prize winner Mother Theresa. According to Mr. Howard, it seems that her order of sisters wanted to convert two old buildings in New York City into a homeless shelter. The sisters spent more than a year navigating the New York City bureaucracy and eventually obtained the necessary permissions. But their plans were upset by a requirement that a renovated four-story building have an elevator even though the sisters, for religious reasons, would never have used it and did not think it an appropriate expense. The bottom line - the homeless were deprived of a shelter altogether. Mr. Howard's general thesis is that regulatory requirements have become so detailed and inflexible that regulators cannot make use of common sense to devise regulatory solutions to real-life needs and problems.

The second author, Stephen Breyer, now Justice Breyer on the Supreme Court, focuses particularly on the problem of skewed priorities. His book Breaking the Vicious Circle: Toward Effective Risk Regulation[2] asks whether it makes sense to spend enormous sums of money to reduce environmental dangers that pose minimal risk. This problem arises when regulations that are originally adopted to reduce risks that most people would consider unacceptable, are ultimately made so unnecessarily stringent that large amounts of time and money must be expended to remedy the last little bit of environmental harm even though no significant safety benefit will result. He refers to this as the problem of "the last 10%" and gives, as an example, a lawsuit involving hypothetical dirt-eating children in New Hampshire. The case involved a private party trying to avoid the expenditure of $9.3 million dollars to clean up the last remnants of a toxic waste dump. The property had already been cleaned up to the point that the waste dump was clean enough for children playing on the site to eat small amounts of dirt daily for 70 days each year without significant harm. The additional cleanup that was being required would enable the children to get their daily dose of dirt for 245 days. Ironically, the site was actually a swamp, so there were no dirt-eating children around in any event. His example illustrates a common regulatory problem: at what point does it make sense to say that a given level of risk is acceptable because it doesn't make sense to expend further resources on the problem.

In sum, these two books bespeak a public plea to all regulators to return to common sense as a basis for regulation. While one can argue about what the anecdotal evidence employed by these authors proves, I think it is fair to say that much of the public shares their perception that greater common sense in government regulation is needed.

Risk Analysis as a Vehicle for the Return to Common Sense

So, let's return to the tool of risk analysis. As I've indicated, I view risk analysis, or risk-based decision-making, as a way to reincorporate common sense into the regulatory process. This mode of decision-making can be divided into at least three distinct components: risk assessment, selection of an acceptable level of risk, and risk management. Each of these steps involves an intermix of science and value judgments. It is in the value judgments part of the mix that common sense ought to be given considerable weight.

A. Risk Assessment

To illustrate my point, I'd like to look at a few of the places in the risk analysis process where value judgments must play a role. Lets start with the first component of risk analysis, risk assessment, which attempts to measure and describe the risk or hazard associated with the use of a substance or technology. It is basically a scientific endeavor which becomes more accurate as scientific knowledge grows, as methodologies are perfected and as measuring instruments become more refined. However, there are often considerable gaps and uncertainties in the scientific information that underlies most risk assessments. One must make certain assumptions to bridge these gaps and overcome these uncertainties which, in scientific terminology, are referred to as "default assumptions". A default assumption is essentially a policy decision which rests on value judgments. It is often the most conservative option, based on the principle that "it is better to be safe than sorry."

At first sight, that value judgment may seem to be just common sense. However, an assessment of a particular risk may need to incorporate many such assumptions, thereby building conservatism upon conservatism and producing an ultimate picture of the risk which may well be extremely disproportionate to its true nature.

To the extent that multiple conservative default assumptions render a given risk assessment incredible, surely it is reasonable to search for, and make use of more realistic alternatives. Doing so is simply common sense.

There are, I know, contrary views. Some contend that conservative default assumptions should be used wherever there are scientific uncertainties that must be overcome because this assures the greatest conceivable amount of public protection. In this view, the inherent biases of conservative assumptions are best dealt with by rendering them explicit; i.e., as long as the public is aware of where "guesstimates" have been made, they will know how to regard the risk estimate. However, I happen to think that this doesn't always work very well, especially when the risk is ultimately expressed in a number, because it is too easy to lose sight of the gaps in the evidence on which that number rests. Moreover, simply recognizing that there is a built-in bias doesn't reveal the extent of the bias and leaves both decision-makers and the public uncertain as to how the risk assessment should be used.

B. Selecting an Acceptable Level of Risk

This brings me to the second area in the risk analysis process, the selection of an acceptable level of risk, where value judgments are particularly important. Essentially, once the nature of the risk is characterized, an acceptable level of risk must be determined. This is, perhaps, the most

difficult value judgment to be made in the course of doing a risk analysis. It is difficult because the answer cannot be reached on the basis of scientific data alone but must take into account both expert and popular views on what type of risks are socially acceptable and the views of these two groups can be quite different. The matter becomes even more complex when the risk in question stems from exposure to a substance which is known to harm humans at high levels but where the risk at low levels is unknown, and possibly non-existent, but is assumed to exist down to zero.

An interesting example is the Environmental Protection Agency's effort to establish an air emission standard for the carcinogen, vinyl chloride, as required by the Clean Air Act. Congress directed that a standard be set at a level which will provide an ample margin of safety to protect the public health.

The problem came in determining what level of risk was acceptable, given the impossibility of establishing any definite level below which no adverse health effects would occur. EPA decided it had two alternatives. First, it could determine that the law meant that a complete prohibition was necessary because only a zero emission limitation would assure complete safety. Or, it could set the limitation at the lowest level achievable by use of the best available control technology if it found that the cost of achieving a zero emission standard would be grossly disproportionate to the benefits of removing the risk. EPA selected the latter course upon finding that the cost of a zero emission standard could be closure of an entire industry. EPA's choice was challenged in court[3]. The court was willing to give EPA broad discretion in determining the meaning of the term "ample margin of safety" but insisted that they decide what was "an acceptable level of risk" and not avoid the issue by simply basing the standard on whatever the best available technology could produce.

The court even suggested a two step method for accomplishing this task. The first step focuses solely on the risk to health. The agency would consider not only scientific information but also "what risks are acceptable in the world in which we live." And the court gave as examples of such socially acceptable risks daily activities like driving a car or breathing city air. Only after completion of the first step would the agency, in a second step, consider other factors like cost and technological feasibility to come to a conclusion as to what standard represented "an ample margin of safety".

Essentially the court was saying: Use your common sense! Being safe from possible harms does not mean living in suspended animation; it means accepting reasonable risks. As the court put it, "safety" does not mean "risk-free". When you must answer a difficult question like "what level of risk is acceptable" from the possible harms of a carcinogen, take a look at other risks people accept in daily life. This case deals with only one particular standard but it underscores a more general fact: "safety" is a social, as well as a scientific, construct and thus involves a value judgment which should incorporate common sense.

I am not saying that determining an acceptable level of risk is easy. In fact, a third area of the risk analysis process in which value judgments play an important role is in the comparison of risks. A number of studies have shown that many factors influence a person's willingness to accept a given risk, factors such as

- the certainty and severity of the risk,

- the reversibility of the health effect,
- the knowledge or familiarity of the risk,
- whether the risk is voluntarily accepted or involuntarily imposed,
- the benefit to be derived from accepting the risk, and
- who the responsible party may be.

There are also difficulties in comparing actuarial risks where available data give a very accurate picture of the risk, such as the risk of death in an auto accident, with risks that are substantially theoretical like the probability of getting cancer. Also, one of the most difficult problems in comparing risks is the problem of how to reconcile the views of the "experts" and the views of the lay public which are often quite different. Since we are dealing primarily with value judgments, the scientists cannot assume that science alone must dictate the solution.

Next, I want to mention the problem of prioritizing risks. This is a problem that transcends the issues involved in doing any particular risk analysis and that is the overarching problem of how society can assure that attention and resources are devoted to the regulation of risks that are truly significant and not have them diverted to further reductions of minimal risks. This is the problem that attracted the attention of Judge Breyer in his book Breaking the Vicious Circle. As he sees it, the results of America's investment in risk reduction have been very uneven but this.

> "...hodgepodge of results does not reflect a public that really wants dirty Boston harbors and superclean swamps; rather, such policy priorities more likely reflect the psychological and practical difficulties of making risk decisions one substance at a time."[4]

Breyer's solution is the creation of an elite corps of civil servants with interagency jurisdiction and authority whose mission would be to build an improved, coherent risk-regulating system with established priorities within and among programs.

But, there are other ideas out there. For example, the Harvard Group on Risk Management Reform, in its report called Reform of Risk Regulation: Achieving More Protection at Less Cost, makes a number of recommendations including this one:

> "Congress should authorize the President's science advisor to lead, integrate, and oversee the assessment and ranking of health, safety, and environmental risks in collaboration with federal agencies responsible for risk regulation."[5]

This effort would be administered by a Director of Risk Analysis whose duties would include preparation of risk assessment guidelines and creation of methods for citizen participation in risk-ranking exercises. Finally, I might mention the Committee for the National Institute for the Environment which advocates legislation to create a new Government agency whose mission would be to improve the scientific basis for environmental decision-making policy. It remains to be seen whether any of these ideas will be adopted.

C. Risk Management

Finally, the third component of risk analysis is risk management. This consists of devising a process designed to keep the risk below the level determined to be acceptable. This has traditionally been the primary task assigned to regulatory agencies by the President and the Congress. The political bodies have sometimes given agencies broad mandates which leave the agency with a good deal of discretion. In other cases they've given very detailed instructions as to how risk management is to be accomplished.

Thus a key problem involving value judgements in this third step is the lack of a uniform approach to risk-based decision-making among the various agencies. There needs to be agreement among the agencies with respect to how risk analysis is best done and particularly with respect to how common sense needs to be inserted into the process.

Currently, Federal agencies do risk-based decision making differently, thus it is no surprise that they wind up with different standards. However, this is exactly the situation that puzzles and frustrates the public.

Conclusion

The good news is that risk-based decision-making is now advocated, at least in broad concept, by both the Administration and the Congress. President Clinton issued an Executive Order in 1993 which requires the Federal agencies to incorporate many of the elements of risk analysis into the rule-making process. The House passed legislation on this topic, in March of last year and the Senate has considered its own version. However, the two bodies must agree on final legislation and then the President must approve before it can take effect.

So, what's the bottom line?

I have attempted to describe a particular method of regulatory decision-making, risk analysis, and to show you some of the points in the process where it may be possible to return to common sense in the course of making necessary value judgments. The issue of the near future will be to figure out how best to do this - both the environmental and the economic well-being of our society are in peril if we don't. Science, values and politics must find a way to come together.

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RADIATION RISKS AND LINEARITY: SOUND SCIENCE?

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Abstract-Significant questions have been raised about the appropriateness of the linear, no-threshold (LNT) theory as a predictive model to estimate health effects at low radiation doses and as a theoretical foundation for radiation protection standards and practices. The LNT model is based on three fundamental assumptions: (1) adverse health outcomes (e.g., cancer) might result from a single ionizing event in a critical cell, i.e., any radiation dose, no matter how small, is potentially harmful, (2) the probability of adverse health outcomes is linearly related to absorbed dose, and (3) radiation damage is not repairable. It is now well-known that the dose-response curves for some, but not all, health effects are non-linear in the low dose range, that radiation-induced damage (to DNA) can be repaired, and that cellular DNA is in a dynamic state in which damage, due to metabolic processes, is constantly occurring and being repaired. This paper explores some of the scientific issues at the heart of the LNT debate including limits of radioepidemiology studies, repair of radiation injury , and hormesis. It is concluded that the validity of the LNT model must be seriously questioned and that alternative approaches to regulatory decision making must be identified.

Introduction

Serious questions have been raised about the appropriateness of the linear, no-threshold (LNT) model to predict health effects (cancer) at low doses of radiation and as a theoretical foundation for radiation protection standards and practices. Concern has centered principally on two issues.

First, regulatory compliance costs too much. The U.S. federal government has promulgated regulations that compel private parties and state and local governments to allocate resources for the protection of the public health and safety. Such regulations depend upon the concept that reduction in dose leads to a concomitant reduction in health risk The LNT model is used widely to predict the reduction in risk for a given reduction in dose. However, there is little epidemiological evidence to show that reducing exposure to environmental carcinogens leads to a reduction in cancer risk. In a comprehensive analysis of the costs and benefits of federal social regulation, Hahn and Hird (1991) found that regulations aimed at reducing occupational safety and health risks have imposed costs of about $9 billion annually for negligible risk-reduction benefits (Table 1). Nuclear power regulations are also very costly. Although it has not been possible to estimate benefits of nuclear power regulations directly, such benefits are not likely to be much greater than those for occupational health and safety (Table 1). Environmental regulations (including air and water pollution control) have wide ranging costs and benefits; accordingly, it is difficult to draw conclusions about the cost effectiveness of these regulations.

Table 1. Costs and benefits of regulation

Regulations	Costs (1988 dollars)	Benefits (1988 dollars)
Environment	$55.4 - 77.6 billion	$16.5 - 135.8 billion
Occupational safety and health	$8.5 - 9.0 billion	negligible
Nuclear power	$5.3 - 7.6 billion	not available

Source: Hahn and Hird, 1991

Second, based on a growing body of scientific evidence, the LNT model appears to be an oversimplification of the dose-response relationship and overestimates health risks in the low dose range. The LNT model is based on three fundamental assumptions: (1) Adverse health outcomes (e.g., cancer) may result from a single ionizing event in a critical cell, i.e., any radiation dose, no matter how small, is potentially harmful. (2) The probability of adverse health outcomes is linearly related to absorbed dose. If the absorbed dose is doubled, the risk is doubled. (3) Radiation damage is not repairable. It is now well-known that the dose-response curves for certain radiogenic cancers are non-linear in the low dose range, that DNA damage can be repaired, and that cellular DNA is in a dynamic state in which damage is constantly occurring and being repaired.

This paper explores some of the scientific evidence bearing on the validity of the LNT model assumptions. The question of a threshold is not a central issue in the LNT debate The validity of the LNT model centers on the absence of nonlinear processes (e.g. repair) in the low dose region. In this context, a dose threshold may be viewed as the lower bound of nonlinearity. A brief historical overview of the LNT model in radiation protection, and a brief discussion of the current status of scientific knowledge pertinent to radiogenic health effects are provided in order to put the key issues in the LNT debate into perspective.

The assumptions underlying the LNT model can now be seriously questioned. There is sufficient scientific data to indicate that the LNT model is overly restrictive and that a less conservative approach to standard setting and radiation protection practice is appropriate. Adoption of a less conservative posture in radiation standard setting and radiation protection practice will have substantial economic benefits without compromising the public health.

Historical Perspective

The establishment of the LNT theory as the foundation for radiation protection standards and

practices began in the 1950s with the National Council on Radiation Protection and Measurements (NCRP) recommendation to replace "tolerance dose" with "maximum permissible dose," a new concept which established the idea of acceptable risk (NCRP, 1954).

Muller's experiments in 1927 were the first to demonstrate the mutagenic effects of x rays (Muller, 1927). The application of target theory as a biophysical explanation of x-ray mutagenesis was a key development in the evolution of the LNT concept. Timofeeff-Ressovsky first showed a linear relation between radiation dose and mutations and, based on mutagenesis experiments showing the lack of a dose-rate effect, thought that radiation damage was analogous to bombs hitting targets and, along with his co-workers, Karl Zimmer and Max Delbruck, set out the target or hit theory to describe radiation action (Timofeeff-Ressovsky et al., 1935). Target theory was based on the assumptions that a single ionizing event is sufficient to cause a "hit" (i.e., any dose can cause damage) and that once a target is "hit" by radiation it is permanently damaged (i.e., no repair occurs).

The belief that radiation induced mutations could occur at any dose and that damage was not repairable led Muller to continually warn the medical community and others who used radiation in the 1930s and 1940s about the potential health hazards and the need to limit the amount of genetic damage from radiation exposure (Carlson, 1981).

Support for the LNT concept was bolstered in the 1950s by studies of Japanese survivors of the atomic bombings which indicated that leukemia could be induced at low doses with no detectable threshold (Lewis, 1957). Further, radioactive fallout measurements, as a result of atmospheric nuclear weapons testing in the 1950s, indicated that large populations could be exposed to measurable radiation doses. In previous decades, the primary radiation protection concern was for individuals who might be exposed to relatively large doses of radiation exceeding established tolerance levels.

During the 1960s and 1970s the LNT model found broad applicability in radiological risk assessment and was also applied to chemical carcinogen risk assessment. Uses of the LNT model led to the widespread belief that extrapolated health effects were "real" rather than theoretical. Little attention was given to limitations and uncertainties in extrapolation or to the underlying assumptions of the LNT model. Predicted numbers of cancers at low dose were considered as valid as the directly observed numbers of health effects at high doses.

Serious concerns about the validity of the LNT model surfaced with the publication of the National Academy of Sciences' BEIR III Report (NAS 1980). The BEIR III Committee used a linear-quadratic model to provide a central estimate of health effects of radiation at low doses. The LNT model and quadratic models were used to establish upper and lower bounds, respectively, on risk. Furthermore, the BEIR III Committee recognized the difficulties in dose extrapolation and, accordingly, limited risk estimates for single exposures of 100 mSv or continuous exposures of 10 mSv/year. The BEIR III Report was marked by two minority opinions-one in favor of the LNT model, the other opposed to it.

The 1990 BEIR V Report (NAS, 1990), which updated the findings of the BEIR III Report, used

only the LNT model to estimate radiogenic risk for solid cancers in part to avoid the controversies that developed during the writing of the BEIR III Report. The BEIR V Committee also recognized the need to limit risk estimates to acute exposure of 100 mSv or chronic exposures of 1 or 10 mSv/year because of the substantial statistical uncertainties in risk at low dose. The BEIR V Committee also used dose-rate effectiveness factors (DREF) as an external correction to the LNT model to account for reduction in risk at low dose-rate due to repair of radiation damage.

The LNT debate reached the international arena in the 1990s. A number of national and international bodies have evaluated or are in the process of evaluating the appropriateness of the LNT model as a foundation for radiation protection. The International Commission on Radiological Protection (ICRP) continued to support the LNT model as a basis for its 1990 recommendations (ICRP 1990). The National Radiological Protection Board also reaffirmed the use of the LNT model in radiation protection in its recent review of radiogenic risks at low doses and low dose rates (NRPB 1995). Central to their position was the conclusion that, although repair of radiation injury has been well documented in laboratory investigations, there is little human epidemiological evidence to suggest that repair at low dose influences radiogenic risk. Canada's Advisory Committee on Radiological Protection (ACRP) recently endorsed the LNT hypothesis but recognized significant limitations. The ACRP recommended that below 200 mSv, health risks should be considered as hypothetical, and below 10 uSv/y individual health risks should be considered negligible (ACRP 1996). The Health Physics Society, in a recent position statement, has also questioned the validity of the LNT model at low doses and cautioned against the use of quantitative risk assessment at low doses because of the lack of health effects data below 100 mGy (Mossman et al., 1996). In a report on collective dose, the National Council on Radiation Protection and Measurements (NCRP) reaffirmed the LNT model as an underlying principle of radiation protection but noted that, at low levels, risk estimates are uncertain by a factor of two or more and a dose threshold cannot be excluded (NCRP, 1995). The NCRP has recently convened a committee to review specifically the LNT model in radiation protection.

What We Know and Don't Know about Radiogenic Health Effects

Any meaningful discussion of the appropriateness of the LNT model in radiation protection must be based on a common understanding of what is known and not known about radiogenic health effects. Based on a large body of laboratory animal and human radiation exposure studies, a number of generalizations can be made about cancer induction by ionizing radiation.

What Is Known:

1. Ionizing radiation is a relatively weak carcinogen. Relative risks (per Sv) for most solid tumors are less than or equal to 2 (UNSCEAR 1994). In comparison, the relative risk for cigarette - induced lung cancer exceeds 20.

2. Radiation induced cancers are non-specific and have a long latency. Because radiation-induced cancers are indistinguishable from naturally occurring cancers, it is impossible to show, in individual cases, that a particular cancer was radiation-induced. The latency period extends from a

few years to several decades depending upon the tumor.

3. Amongst the numerous tissues and organs known to be susceptible to radiation-induced carcinogenesis, there is a wide range of sensitivities. Female breast, thyroid, and bone marrow have high radiosensitivity. Cancers with low sensitivity include uterus and prostate.

4. Various host factors are known to contribute to radiogenic cancer risk. For instance, breast cancer and thyroid cancer have a significant dependency on gender and age at exposure. Age at the time of exposure is probably the most important determinant of radiogenic risk for cancer.

5. Radiogenic cancer risk is well-known at doses in excess of about 300 mGy. The BEIR V Committee has estimated the total lifetime cancer mortality risk to be 8% per Sv based on cancer mortality observed at 300 mGy and higher (NAS 1990).

What Is Not Known:

1. The mechanisms of radiation carcinogenesis are not fully understood. The origin of cancer is hypothesized to be the result of mutational events to critical genetic loci that lead to the development of cancer. The important and rapidly developing fields of molecular biology and molecular epidemiology are having a significant impact on our knowledge and understanding of the carcinogenesis process (Perera 1996). Recently, the gene that is mutated in the autosomal recessive disorder ataxia telangiectasia (AT) was identified (Savitsky et al., 1995). Individuals with AT have an acute sensitivity to ionizing radiation and a predisposition to certain malignancies including breast cancer. Further study of the AT gene may lead to a better understanding of human cancer and the role of ionizing radiation in carcinogenesis.

2. Although radiation damage to DNA is recognized as the key first step in carcinogenesis, ionizing radiation damage to DNA is non-specific and it is not known what critical lesions in DNA are responsible for gene mutations leading to cancer.

3. It is unclear why organs and tissues vary in radiosensitivity. Apparently, the sensitivity to radiation induction is unrelated to the spontaneous incidence for most cancers. For instance. thyroid cancer has a low spontaneous incidence but a high radiation sensitivity; prostate cancer has a very high spontaneous incidence and apparently cannot be induced by radiation.

4. Although repair of radiation damage (including adaptive responsiveness) has been well-characterized in cell culture systems, the influence of repair processes on human radiogenic risk at low dose and low dose rate is not fully understood. However, DNA repair is an important determinant of cancer risk since it is known that certain human diseases (e.g. ataxia telangiectasia) are characterized by defective DNA maintenance and repair capacity and high risk for cancer.

5. It is unknown whether ionizing radiation is a carcinogen below 100 mGy. The magnitude of radiogenic cancer risk below 100 mGy can only be inferred by extrapolating from epidemiological data obtained at high doses and dose rates. It is interesting to speculate that had none of the Japanese survivors received 100 mGy or more, there would have been little evidence of radiation-

induced cancer in the Japanese survivors. From 1950-1990, only 43 excess cancer deaths (out of a total of 5800 cancer deaths) have been found in a population of 69,000 exposed to doses below 100 mGy (Pierce et al., 1996).

6. Substantial uncertainty exits concerning the shape of the dose-response curve at low doses. Insufficient epidemiological data below 100 mGy exits to distinguish various biologically plausible dose-response models.

Limitations of Radioepidemiology

The non-specificity of radiogenic cancer, long latency, and large background rate of non-radiogenic cancer in the population necessitates the use of large epidemiological studies to quantify the carcinogenic effects of radiation in humans. At low doses of radiation (e.g. under 100 mGy) it is virtually impossible to detect a statistically significant increase in radiogenic cancer. In addition, competing risk factors, introduced because of the long latency period, may confound interpretation of radiogenic risk.

Radiation doses necessary to detect statistically significant increases in radiogenic cancer are much larger than doses typically encountered in environmental or occupational exposure settings (Table 2). The lowest dose consistently associated with radiogenic cancer is approximately 100 mGy for thyroid cancer in children based on a pooled analysis of seven studies (Ron et al., 1995). Doses in Table 2 do not represent thresholds but reflect the limits of detectability of radioepidemiological studies.

Table 2. Lowest Doses of Low Let Radiation Associated with Cancer

Health Effect	Population	Lowest Dose (mGy)	Reference
Total solid cancer	Japanese survivors of atomic bombings	200	Shimizu et al. 1990
Leukemia	Japanese survivors of atomic bombings	200	Shimizu et al. 1990
Thyroid cancer	Medically irradiated populations and Japanese survivors of atomic bombings	100	Ron et al.1995
Breast cancer	Medically irradiated populations and Japanese survivors of atomic bombings	200	Land et al. 1993

Proponents of the LNT model argue that there is epidemiological evidence of radiogenic cancer below 100 mGy. Several studies have suggested a correlation between *in utero* x-ray exposure and an increased risk of leukemia in childhood (see, for example, Stewart et al., 1958). Most reports seem consistent with a 40% increased risk of childhood leukemia associated with low dose intrauterine exposure to diagnostic radiation between 10-100 mGy just before birth. It remains to be established whether adult cancer is increased by fetal irradiation, although there has been recent evidence that this may be the case. In analyzing the childhood leukemia studies, it has been suggested that selection factors, related to the medical reasons why women received prenatal x-rays, were responsible for at least a portion of the increased leukemia risk. The absence of any childhood leukemia in atomic bomb survivors exposed *in utero* supports this selection theory. While it is now generally accepted that prenatal x-ray is associated with an increased risk of childhood cancer, the causal nature of this association and the level of risk remain uncertain (NAS, 1990).

A recent analysis of cancer mortality in atomic bomb survivors during the extended period 1950-1990 suggests that 50 mGy is the lowest dose at which there is a statistically significant excess risk (Pierce et al., 1996). However, this finding should be considered with caution since excess relative risk is strongly dependent upon gender and age at exposure. The lowest dose may be somewhat higher than 50 mGy depending on how age and gender parameter adjustments are made in the statistical analysis (Pierce et al., 1996). If dose response is expressed in terms of the ratio of observed to expected cancer deaths, the lowest dose associated with a statistically significant radiogenic risk is about 200 mGy, a finding consistent with the 1950-1985 cancer mortality analysis by Shimizu and colleagues (Shimizu et al., 1990).

Predicting radiogenic health effects at environmental and occupational exposure levels requires that directly observable dose response data be extrapolated 2-3 orders of magnitude (Table 3). This degree of dose extrapolation strains the credibility of risk assessment at low dose and is comparable to the dose extrapolations used to "demonstrate" the human cancer-causing effects of cyclamates, saccharin, and safrole based on laboratory animal data (Table 3). Accordingly, numbers of cancer deaths due to low levels of radiation exposure must be considered speculative; risk estimates at low dose have large uncertainties because they are theoretically derived. The possibility that there may be no health risks from radiation doses comparable to natural background radiation levels cannot be ruled out; at low doses and dose rates, the lower limit of the range of statistical uncertainty includes zero (NAS, 1990).

It is unlikely that future radioepidemiological studies will provide additional significant refinement of current radiogenic risk estimates. However, by focusing on early molecular events in carcinogenesis, molecular biology and molecular epidemiology studies may provide important new clues to the nature of the dose-response curves at low dose (Perera, 1996). Recently, radiation-specific mutations of the p53 and *ras* genes have been identified in radon-induced lung cancer (Vahakangas et al., 1992; Taylor et al., 1993). Although the significance of these findings remains questionable (Lo et al., 1995), confirmation of radiation-specific mutations will provide important molecular markers for heightened radiogenic cancer risk.

Table 3. Extrapolating Health Risks

Agent	Equivalent human experimental dose[a]	Typical human dose	Ratio[b]
Cyclamates[c]	~500 cans of soft drink daily	3 cans of soft drink daily	~150
Saccharin[c]	~1200 cans of soft drink daily	3 cans of soft drink daily	~400
Safrole[c]	~600 cans of root beer daily	<1 can of root beer daily	>600
Ionizing radiation	200-500 mSv	2-4 mSv/y	~100

[a] calculated human doses of carcinogen equivalent to actual doses used in animal experiments to demonstrate carcinogenic activity
[b] ratio of equivalent human experimental dose to typical human dose.
[c] modified from Kraybill, 1977

Repair of Radiation Injury

The LNT model does not account for repair of radiation injury. Accordingly, the model predicts that the frequency of radiogenic health effects is independent of dose-rate. For this reason an external correction factor, referred to as the dose-rate effectiveness factor (DREF), must be applied to LNT- derived risk estimates to correct for repair at low dose-rate. Risk estimates used in radiation protection are based on data from the Japanese survivors of the atomic bombings who were exposed to radiation delivered at high dose rate. The BEIR V Report provides a range of DREF values of 2 to 10 but offers little guidance on specific DREF values for use in radiation protection (NAS, 1990).

The LNT model also predicts that any dose, no matter how small, may produce health effects based on the hypothesis that a single ionizing event can result in DNA damage. However, DNA damage of importance in radiogenic diseases occurs at a high rate naturally even in the absence of radiation exposure (Table 4). Small doses of radiation (10-100 mGy) increase the spontaneous background rate by a small fraction of 1%, essentially contributing an insignificant increase to the DNA damage which occurs naturally. Accordingly, it is highly unlikely that small doses of radiation are of any practical public health importance.

Table 4. Intrinsic DNA Damage

Type of Event	Rate of Spontaneous DNA Events Per Day Per Cell	DNA Damage Per 10 mGy Per Cell[a]	% of Daily Spontaneous Rate
Single-strand breaks	121,000	10	0.008
Double-strand breaks	36,000	0.4	0.001
Base lesions	30,000	9.5	0.03
Total events	187,000	20.0	0.01

Source: modified from Billen, 1990
[a] low LET radiation

Various types of repair phenomena have been well characterized in mammalian cell systems. Subtransformational repair (Han et al., 1980) and adaptive responsiveness (Wiencke et al., 1986) are particularly relevant to radiation carcinogenesis. Both types of repair are measured by a decrease in radiation effects when specific radiation exposure conditions are used. In the case of subtransformational repair, the frequency of transformed cells is decreased when the dose rate is decreased. Although the exact nature of repair cannot be discerned from such experiments, it is theorized that the lower dose rate permits cells to repair DNA damage which would otherwise transform cells leading to cancer. In the case of adaptive responsiveness, cells exposed to low doses of ^{3}H are less sensitive to the mutagenic effects of a subsequent exposure to gamma rays. This acquired resistance is mediated by the repair enzyme poly-ADP-ribose polymerase since the effect may be prevented by treating cells with 3-aminobenzamide, an inhibitor of poly ADP-ribosylation (Wiencke et al., 1986).

Although these repair processes have been well documented in cell culture systems, their relevance to radiogenic cancer risk in humans remains unclear. Nonetheless, maintenance and repair of DNA are important in reducing cancer risk as demonstrated in several human diseases, including xeroderma pigmentosum, Bloom's syndrome, Fanconi's anemia, and ataxia telangiectasia. A common feature of these diseases is a heightened risk for cancer and an abnormally high rate of genetic damage in affected individuals. Ataxia telangiectasia (AT) is a particularly interesting disease because it is also characterized by increased radiosensitivity. The AT gene may be found in approximately 1% of the population but may be present in 9-18 % of persons with breast cancer in the U.S. (Swift el al., 1991). Although yet to be confirmed, this finding suggests that cancer is not homogeneously distributed within the population but appears to be clustered in genetically predisposed subgroups.

It has been suggested that low doses of ionizing radiation may be beneficial (hormesis). A large number of cell, animal and human studies suggest that DNA repair, enhancement of immune function, and other stimulatory effects may reduce the radiogenic and natural incidence of disease

(Luckey, 1991). Proponents of hormesis have argued that conventional risk estimates, based on the LNT theory, may be overstated because no allowance have been made for potential beneficial effects of radiation at low dose which may reduce risk.

Radiation hormesis remains controversial. Luckey cites over 1000 literature references documenting radiation hormesis in the context of growth and development, reproductive capacity, immunity, cancer, and lifespan (Luckey 1991). UNSCEAR (1994) acknowledged radiation-induced repair as an important cellular phenomenon, but found little evidence that cellular adaptive responsiveness and other forms of repair could convey beneficial effects to the organism that would outweigh potential harmful effects.

Demonstration of hormetic effects is a formidable task and hinges on the establishment of a causal relation between dose and beneficial effect. Evidence for causality should include demonstration of a strong statistical association, reproducibility of results, dose-response, and biological plausibility. Spontaneous incidence of disease can vary widely in human populations and studies involving an unusually high spontaneous incidence of disease may be misinterpreted as a hormetic effect. Hormetic effects at low dose may be difficult to demonstrate with statistical significance because of the large background incidence of disease (especially cancer). Accordingly, because of the paucity of statistically significant data in the low dose region, it may not be possible to distinguish an hormetic response from other biologically plausible dose-response curves.

Human diseases characterized by repair deficiencies and high rates of cancer clearly establish the importance of repair as a determinant of cancer risk. What is not known is the degree to which repair (as reflected by dose-response nonlinearity) reduces radiogenic cancer risk. Much work is yet to be done to fully understand repair of radiation damage and possible hormetic effects. However, enough is now known that serious consideration must be given to revision of the LNT model. Even if the LNT model is retained in some form, there is sufficient justification to review DREF values used to account for dose-rate effects.

Discussion

A substantial body of scientific evidence has now been accumulated indicating that the LNT model overestimates health risks at low doses and that less conservative nonlinear predictive models should be considered as the basis for standard setting and radiation protection practice. However, discarding the dominant ideology that any radiation dose may be harmful for a less conservative model is not an easy task. Linear models are simplistic and conservative and can be more readily justified to the public than the more complex nonlinear models.

Selecting a predictive model from among several biologically-plausible alternatives, for the purposes of setting regulatory policy, must be based on scientific, economic, social and political considerations. Non-scientific factors are important in decision-making because relevant scientific data are not sufficiently definitive to establish a clear model preference. For example, the BEIR III Report used three predictive models in the analysis of the Japanese atomic bomb survivor data base to estimate risk at low doses (NAS, 1980). Direct observations of radiogenic cancer deaths at low dose were insufficient to distinguish one model from another and, consequently, all

models could be satisfactorily fit to the data.

Regulatory decision-making must be based on the best science available. Establishing exposure limits based on technical capabilities of measuring and reducing doses down to specific levels is not sound scientific practice. In Executive Order 12866 of September 30, 1993, President Clinton identified 12 principles of regulation including the use of "the best reasonably obtainable scientific information concerning the need for, and consequences of ...regulation" (Clinton, 1993).

What is meant by the concept of "best reasonable obtainable scientific information?" Sound science, as a basis for regulatory decision-making, should be characterized as follows: (1) objective analysis of data, including appropriateness of experimental design and statistical tests, and careful attention to the uncertainties in data collection and interpretation, (2) appropriate consideration of underlying assumptions and limitations of theories and models used in the interpretation and analysis of data, (3) conclusions which are congruent with results and analyses, (4) publication in peer-reviewed technical journals or other suitable scientific publications, (5) repeatability of observations, and (6) reliance on multiple studies rather than individual studies. Credible scientific studies may lead to honest differences in data interpretation and support of competing theories. Risk estimates derived from various biologically-plausible theories may lead to significantly different predictions of health effects in the low dose region (NAS, 1980).

Risk assessment and establishing regulatory limits on exposure should be based on a balanced consideration of observable health effects data and theoretical predictions of health effects determined in the dose range of interest. Dose extrapolation of 2-3 orders of magnitude involve great uncertainty. In such cases, risk assessment is more of a guessing game than a science. When large extrapolations are involved, regulatory standards should be based on what is known rather than on theoretical estimates. The collective human epidemiological data indicate that below 100 mSv, health effects such as cancer are not observable in adult populations. The Health Physics Society's position to limit quantification of risk to doses above 100 mSv is based on this observable limit (Mossman et al., 1996). A recent U.S. federal court dismissed all 2,100 lawsuits against GPU Nuclear Corporation that claimed radiation injury from the 1979 TMI accident because of lack of evidence that anyone had received doses greater than 100 mSv. The court determined that there is consensus within the scientific community that "at doses below 10 rems, the causal link between radiation exposure and cancer induction is entirely speculative" (In Re TMI Litigation Consolidated Proceedings, 1996).

In the past, the LNT model provided a reasonable, conservative approach to risk assessment for the purposes of standard setting and radiation protection practices. The assumptions underlying the model were consistent with the scientific knowledge of the time. Substantial new information about human cancer, carcinogenesis, molecular biology of cancer, and human radiobiology at low doses now call into question the validity of the LNT model assumptions. Although much is yet to be learned about cancer at low doses of radiation, sufficient information is available now to apply less conservative predictive models with reasonable assurance that public health will not be compromised.

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RADIATION RISK IN PERSPECTIVE: THE HPS STATEMENT

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ABSTRACT. The Health Physics Society's Scientific and Public Issues Committee recently published its perspective on cancer risk from ionizing radiation at the low levels of exposure. [March '96 HPS Newsletter.] The purpose of this statement is to bring attention to levels of radiation dose below which estimates of risk cannot be stated with a level of confidence usually considered as "significant" in science. Estimates of adverse health effects, based on an overrsimplified, linear, no-threshold, dose-response model, are speculative and overestimate any cancer risks. The Committee recommends dose levels below which a range of health outcomes should be stated, including the possibility of zero health effects.

SCIENTIFIC AND PUBLIC ISSUES (SPI) COMMITTEE [HPS Rule:7.1.24]

Composition: Five members--President (Mills), President-Elect (Vetter), three most recent Past Presidents (Goldman, Chair; Mossman; Schaiger). [Past President Masse is a co-author]

Procedures: Release requires 4 affirmitive votes of members. Board of Directors receive information copy 48 hours prior to release. Communicated to members via HPS Newsletter.

Statement: Penultimate draft informally approved by Board with one dissent.

MAJOR POINTS

---Current radiation protection standards and practices are based on the premise that any dose, no matter how small can result in health detriment and the liklihood is directly proportional to dose received; i.e. the accepted dose-response relationship for estimating harm is the linear no-threshold (LNT) model.

---There is substantial scientific evidence that this mathematical model represents an oversimplification of the biological mechnisms involved and results in an overestimation of health risks in the low dose range.

---Radiogenic health effects (primarily excess cancers) are observed in human epidemiology studies only at doses in excess of 0.1 Sv (10 rem) delivered at high dose rates. Below this dose, estimation of adverse health effects is speculative.

CONCLUSIONS

---Limit quantitative risk assessment to situations where the doses above natural background are equal to or greater than 0.05 Sv in one year, or 0.1 Sv in a lifetime.

---Below these doses, expressions of risk should only be qualitative, emphasizing the inability to detect any increased health detriment and a range of possible estimates that includes no risk in the dose range of interest.

---"Collective dose" remains a useful index for quantifying dose in large populations and in comparing doses from other radiation sources. However, it should not be used to quantify health risks in that population when all individuals receive lifetime less than 0.1Sv.

COMMENTS

The SPI Committee believes the HPS statement best represents what is presently known in estimating human risks from exposure to low doses of ionizing radiation within a reasonable degree of acceptable scientific certainty. Also, that in publishing this statement we serve the radiation safety community, and the public at large, by stimulating and focusing open discussion on a topic fundamental to the policies for and practices of radiation safety.

Why did we take this somewhat bold and obviously controversial step?

While each member of the Committee might cite his own specific reasons for doing so, based on personal experiences and reasoning, we share a common one of recognizing a demanding and pervasive need for an open discussion on the issue of whether the current acceptance of the LNT model is based on "sound science" and provides a valid public perception of health risk when used in establishing radiation safety regulations.

My reasons are based on my own experiences in radiation protection during my years in the US Environmental Protection Agency (EPA) as contrasted to those during my earlier years in the US Public Health Service (PHS). In the former regulating for protection of the environment is of central importance; whereas, in the latter, "adequate"protection of public health is of central importance. As the Division Director (1970-1981) responsible for developing EPA's radiation standards and Presidential approved Federal guidiance, I found "sound science" and "common sense" to become less and less the applicable criteria within the Agency. It is in the context of these important criteria that I strongly endorse the HPS Statement for the following reasons:

(1) Prevailing public perception is that there is NO SAFE LEVEL of radiation exposure; a view encouraged perhaps by a misleading interpretation of the well-meaning guidance in radiation safety to AVOID UNNECESSARY EXPOSURES. The avoidance of unnecessary exposure is an action to be implemented at the operational level of control and should not be viewed in the context of there being "no safe level". Taken to the extreme, as is often done in the media, any exposure can be judged to be unsafe and therefore to be avoided. The regulatory requirement of ALARA practices below the established operational control limits enhances this preception, by undermining the limits as levels that are SAFE. [I'm reminded of the statement made by Walter Cronkite (one America's most respected newsmen) in reference to nuclear fuel reprocessing, implying that even a single atom of Pu in the lung would result in lung cancer **period**]

(2) BENEFITS of radiation sources [in generating electric power, exploring space, improving medical diagnosis and treatment, preserving food, making available thousands of industrial applications, and in ensuring national security and that of much of the free world] are not generally publicly stated in describing why radiation protection is necessary for society's health and welfare.[If there are no "benefits" there is no need for radiation sources to protect against!] In the US, health physicists have become overly sensitive to being viewed as "advocates" for these benefits or being accused of having some "conflict of interest." For this reason I personally prefer the term "radiation safety" rather than "radiation protection" in describing what I do as a health physicist. To the anti-nukes "protection" we appear as "for" radiation; whereas to the regulated we may appear as "against" radiation. "Radiation safety" implies caution in excersing "respect" but not "fear" of the source. Cooling towers and mushroom clouds have become symbols of unacceptable fear, rather than any recognition of the many societal benefits derived from splitting of the atom.

(3) Their is compelling scientific evidence at all levels of biology and from human epidemiology that supports a threshold non-linear (TN-L) dose-response model vis-a-vis the LN-T model. For example:

--At the molecular level, the number of spontaneous DNA damage events is of the order of 10E8/y per mammalian cell; whereas, for a dose of 0.01Gy only 20 additional events would occur, certainly in the range of being minimal , if not safe in the absolute. [Billen. *Rad Res 124*, 242, 1990.]

--At the cellular level there is evidence of radiation resistance to cell mortality in the dose range 0.1-0.8Gy; suggestive of adaptation. [Lambin. *Rad Res 138*, S40,1994.]

--At the human epidemiology level, there are two populations of special importance,

The US radium dial painters' data show no cancer below 10Gy (1341 subjects), all head carcinomas and bone cancer are in subjects (154) with doses in the range of 10.1-278Gy. [R.G. Thomas, 1994.] These findings, suggesting a threshold, are supported by beagle studies. [O. Raabe 1994.]

Estimates of cancer mortality in the A-bomb survivors *per Sv* are "statistically significant" at the 95% CL only for the total of solid tumors plus leukemias. Risk of leukemia is characterized as "strong evidence", major radiosensitve solid tumors as "significant association," and other solid tumors as "some indication," first time association," "strengthened finding," and many as "no significant effect." [Thompson et al., *Rad Res 137*, S17, 1994.] (I find these characterizations interesting of themselves.) Furthemore, the lowest dose for statistical significance in this population is reported to be 0.2-0.49Gy for all cancers except leukemia and lung; 0.5-0.99Gy for stomach, breast, and leukemia (kerma dose); and 2+Gy for colon. [Shimizu et al., Rad Res 121,120, 1990.]

It is noteworthy that cancer mortality radiation risk cofficients published in the NAS/NRC Committee's BEIR V report is per *0.1Sv* and that in the UNSCEAR 1994 report is *per 1Sv*. This suggests that application of these risk coefficients at doses below those cited by these distinguished scientific committees does not have a strong scientific base for estimating cancer risk and that perhaps only a dose should be estimated.

-My most compelling reason for supporting the HPS statement is that the interpretation of the linear, no-threshold dose response has resulted in overregulation of radiation in the U.S. Acceptance of this highly uncertain hypothesis as a matter of regulatory policy and the accompanying perception of "no safe level" is having serious consequences in the U.S. For example, disposal of much of the low level radioacitive waste is "storage on site," as is all high-level waste; costs of implementing the Clean Air Act standards for radioactivity are estimated to cost **$1-34 billion per year of life saved**; cleanup of radioactivity and other contaminants at DOE sites is budgeted at **$6 billion for fiscal year 1997** with no end-in-sight, and the "human radiation experiments" investigation is likely to expand in scope when compensation becomes available.

"Regulatory reform" is a phrase often heard in our Congress with the intent of usuing "common sense" and, where applicable, applying "sound science" in developing or more reasonable environmental and other health protection policies and regulations. Since many of these policies and regulations apply to radiation, **the HPS Statement is timely and the discussions that follow healthy to radiation safety and our profession.**

Nordic cooperation in nuclear safety

A historical overview

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1. Introduction

The Nordic cooperation in nuclear safety is a 40 year history of our five small countries, with a total of just over 23 million people, working together, sometimes amicably, sometimes less so. That they should work together is hardly surprising given their common historical and cultural background, and their similar organisation of democracy, the first example of which is found here in Iceland where the Alþingi met for the first time in the year 1000. Although the individual Scandinavian languages have moved away from the old Icelandic tongue, one extremely strong advantage for our cooperation is that with a little training most of us can communicate directly, each using our own languages without having to try to express nuances with idioms that are not our own.

Being a small group of closely related countries has not isolated us from what was happening in the rest of world, on the contrary, external factors have had a strong impact on the shape of this cooperation, and on its contents.

What may be surprising to the outside world is that these five countries often work as one single unit, which is demonstrated by this meeting of the NSFS where our countries appear as one single unit in the world assembly of radiation protection societies who together form the IRPA. Nordic work, and persons from our countries, have also had a significant impact on the development in the wider international development and on organizations shaping events - an impact much larger than their size would suggest.

It is this inter-relationship between the Nordic work, the external factors influencing it, and our input to international development which we shall be examining in this brief historical overview.

2. The beginning

So how did it all start over 40 years ago? Following the second world war, the prospects for collaboration did not seem quite so clear. There were large differences between the Nordic countries. Iceland had just taken the step to be independent from Denmark, which together with Norway had been held by Germany and soon after the war joined NATO. Finland depended heavily on the Soviet Union, while neutral Sweden was manoeuvring in between the blocks.

Atomic questions in that period were necessarily a mixture of defence and peaceful applications. Some of the first issues were taken up by foresighted personalities, who typically had played outstanding roles during wartime, some even in the resistance against the occupation.

These individuals saw both the promising peaceful uses of this new energy source, and its risks. Despite the fact that it was difficult to obtain not only relevant information but also the nuclear material which was necessary for experiments, e.g. uranium and heavy water, until after the Atoms for Peace programme was initiated in 1953, there was a quite early start in Sweden and Norway, already in the forties, with the start of reactor development and related research. There would seem to be a good basis for cooperation here.

Under the prevailing circumstances most of the early work was done nationally with only few Nordic contacts about such questions as heavy water, uranium, analysis, and calculations. The JEEP rector in Norway, the first one outside the big powers, was in operation as early as 1951 thanks to the foresighted Gunnar Randers. The next Nordic country to join the club was Sweden with its R1 reactor in 1954.

The second main outside incentive for cooperation was a consequence of the arms race "atoms for war": the fallout in the fifties over the Northern parts of Scandinavia where the Lapps lived and ate reindeer-meat. Concern for the consequences gave rise to consultations between Nordic authorities. Again it was influential personalities such as Rolf Sievert who actually started contacts with persons such as Jorma Miettinen in Finland and Thorleif Hvinden in Norway.

At the same time scientific and technical developments within the Nordic countries had an impact on the coming cooperation. On the theoretical side Niels Bohr and his institute in Copenhagen had gathered enthusiastic young physicists from all over the Nordic countries. On the practical side, from 1959 the Halden Project became a similar pole of attraction for young engineers from all the Nordic countries, who later returned to work on their national programmes.

When collaboration about peaceful nuclear questions became the subject of international consideration in the mid-fifties with plans for the IAEA, for ENEA (today's NEA), for Euratom, and for UNSCEAR, the Nordic countries became eager participants and players. In fact people from the Nordic countries had a strong influence on their coming into being. Here are some examples: The second director general of the IAEA was Sigvard Eklund from Sweden, and the second director of the ENEA was Einar Sæland from Norway. Sweden played an important role in giving birth to UNSCEAR.

3. Need for mutual support and joint action

We who live in the jet-age with its fax and e-mail, should remind ourselves of the practical difficulties with travelling and telephoning in the fifties. Leading persons - both on the administrative and on the technical side - felt isolated at home in their Nordic country when they had to face the multiple questions coming up in the atomic field. Personal contacts were essential, and these were being established in the Nordic situation.

This did not mean that all the Nordic countries would follow the same road. Each of the countries had its own objective in view of the coming nuclear age. In Denmark the goal was its multiple use for the benefit of society. Here in Iceland there seemed to be economic potential in the production of heavy water by using geothermal heat. In Finland a long course towards the

use of nuclear power started. Norway hoped for nuclear propulsion of its merchant fleet. And Sweden had ambitions to create its own nuclear industry.

It was the general political development that paved the way. With the founding of the joint parliamentary Nordic Council in 1952, a new promoter for Nordic ventures appeared. Nuclear energy was identified as one possible issue of common interest.

The Nordic Council, still in existence to-day, consists of parliamentarians and ministers. It takes initiatives and makes recommendations to the governments. It was this political body who in 1957 initiated the establishment of two committees, one to cover technical aspects, the so called *kontaktorgan*, the liaison committee for nuclear energy. The other dealt with nuclear physics research.

Two years later, in 1959, the Nordic Council recommended the government institutes responsible for radiation protection to work together. This led their directors to form their Nordic "chiefs" group which still exists to-day. It organises working groups with members of their institutes and consultations in matters of current interest.

The *kontaktorgan* was composed by top executives from ministries and other authorities. It provided a framework that permitted all types of questions to be taken up jointly. During the 33 years of its existence, the *kontaktorgan* laid the grand lines of the Nordic cooperation in its field, and also worked in some details, to discuss international ventures, and to further practical Nordic cooperation whenever warranted.

Its approach is characteristic of Nordic pragmatism. In some aspects it went quite far: thus its members arranged that the Nordic countries for many years shared one seat on the IAEA board.

4. External factors that influenced the Nordic cooperation

National ambitions evolved over time, and in some cases continued to differ. This resulted in positive and negative inputs to the various cooperation schemes which developed. Nordic cooperation has always been an "à la carte" process. Furthermore, Nordic cooperation was never seen as an end in itself, but as a means to strengthen the countries' positions internationally, to gain a stronger voice.

By the sixties, the economic growth was both raising demand for electricity and providing the funds for reactor development work at the nuclear research centres. Public opinion was in favour of the new energy source. This belief in nuclear power as a cheap and clean energy source led to cooperation in questions related to reactor design and performance. The research institutes worked together on heavy water reactors that could use natural uranium - thereby coming in a certain opposition to the utilities who were expecting the breakthrough of light water reactors using enriched uranium that had to be imported. At the end of the sixties, and after investigation by the *kontaktorgan*, the research institutes established a Nordic *coordination committee* to strengthen their positions and make use of their complementary knowledge. The *coordination committee* was active from 1968 until the time, in 1981, at which the research institutes had moved away from nuclear research and to-days NKS had taken over safety research.

The *coordination committee* organised several large joint projects at a total cost of the order of USD 1-2 million annually. Although much of the work headed towards reactor construction, it indirectly dealt with safety questions. At this time most questions of technical safety and waste

management belonged to the tasks of the research institutes and were taken up by the *coordination committee*. Collaboration in radiation protection was dealt with by the national authorities.

Concern about issues of safety and the capacity to evaluate the safety of nuclear reactors came up in connection with the planned visit of the US nuclear vessel N/S Savannah in 1964 to several harbours in the Nordic countries. This visibly put a strong burden on the authorities who were responsible for issuing the permits. The question of nuclear safety came up in the *kontaktorgan* in 1968. At its recommendation the competent authorities established the Nordic group on Reactor Safety, *NARS*. Its main purpose was to provide recommendations for the documentation to be used in license applications in the Nordic countries. Other fields of action were safety criteria and emergency provisions within nuclear sites.

In the seventies it was still believed that urban siting of nuclear power plants would be a realistic possibility. Experts from the other Nordic countries participated in the Swedish Urban siting study of a city-close location. The study was initiated when plans came up in 1968 for a combined heat-electricity nuclear power plant in central Stockholm. The outcome was that nuclear power reactors might well be located close to cities: the risk could be related to a loss-of-coolant accident, the risk of which was estimated to be of the order of 1-10 in a million, something which seemed acceptable. For the first time probabilistic methods were used.

This order of magnitude coincided with the finding of the NRC report WASH-1400 by Norman Rasmussen. A Nordic working group in 1975 undertook a systematic review of his report and confronted its results with the Swedish Urban study. Certain inconsistencies were actually found in Rasmussens study and notified to him.

However, in the seventies the opposition against nuclear power was already developing. An important person in this respect was the Swedish Nobel prize winner Hannes Alfvén who helped the anti-nuclear movement to gain political force. Serious protests arose in conjunction with Norwegian plans for a reactor site in the Oslo fjord in 1973.

In Denmark public debate increased strongly in the mid-seventies. Contacts had been established in 1968 between Swedish and Danish authorities concerning safety aspects of the planned Barsebäck plant, on the Swedish side, but close to Copenhagen. All relevant information was provided from the Swedish authorities and analysed in Denmark prior to the issuing of a construction permit, both in case of the first and the second unit.

Many calculations have been made over the years to estimate possible effects in Denmark following a hypothetical accident at Barsebäck. In 1978 Danish calculations were submitted to the Swedish Energy commission to show that calculations made by consultants for the Swedish authorities, predicting serious consequences, were unreasonable. Unrealistic assumptions must be made in order to arrive at the high number of casualties indicated in these reports.

Initially, one fifth of the output of the first Barsebäck reactor was contracted to a Danish utility. This was frequently noticed at a time of increasing Danish opposition against nuclear power. It was not very popular, during a moment of disturbance in the electricity supply in Copenhagen, to remark "it must be a failure at Barsebäck!"

One of the off-shoots of this public protest was an attempt to improve information from the research institutes. In 1973 the *coordination committee* established a contact group for environmental information. Its members were to exchange information related to risks and

environmental effects related to nuclear power. The group produced factual information, supposedly not biased in favour of nuclear.

The need for increased knowledge about accident sequences was strengthened by the concern in other countries. In the early seventies, voices of concern had come up, originating from the opposition movement in the USA, about the possible consequences of a loss-of-coolant accident following a pipe rupture in a reactor. A Nordic group in 1972 reviewed existing computer codes used in accident analysis. They recommended a cooperative effort "NORHAV" to produce a computational system for analysis of severe accidents. The NORHAV work was later offered as a Nordic in-kind payment when the USA team asked for competent input to the upcoming LOFT programme, the large Loss Of Fluid Test.

In the early seventies it became evident that computer codes simulating accident sequences became very complicated, and the need for experimental verification increased. The advanced Swedish heavy water Marviken reactor, once it had been decided that the original project would not materialise, offered a practically finished large power reactor where full-scale experiments could be arranged in its containment. The Marviken experiments got under way with a Nordic group to assist at their birth, and later they became international ventures with a strong Swedish leadership, for many years encompassing Nordic participation.

In the meantime, what in the public opinion had seen a unlimited source of power, was now also seen as a source of dangerous waste. It was the management of low- and medium level waste which was the immediate task, and here four of the Nordic countries had comparable situations. The advantages of working together on this issue were readily apparent. After the first of a long series of Nordic seminars on waste in 1974, a need for better regulations and methods for waste disposal appeared.

The *kontaktorgan* sat up a waste group to formulate proposals accordingly. Through its composition this group became an example of the constructive cooperation that could be achieved between research institutes and authorities. It recommended that a comprehensive Nordic waste programme be organised, where the practical work would be carried out by the *coordination committee.*

There have been many Nordic projects in the waste field, and some of the international waste programmes managed from the Swedish authorities (with names such as Hydrocoin, Intraval, Biomovs etc.) were helped in their start by including them in Nordic programmes.

Following the energy crisis in 1973 the concerns about safety were being balanced by the fear of energy shortage and dependence on vulnerable sources of energy supplied from abroad. The Nordic ministers for industry expressed their good-will for increased R&D on nuclear safety in order to open the way for replacing some of the oil-based generating capacity by new nuclear plants.

After long discussions in the now flourishing bureaucracy under the Nordic Council of ministers, this finally led to the start of an enlarged cooperation by the creation of the Nordic committee for nuclear safety research, *NKS*.

This was the start of a dynamic cooperation which to-day has become the most successful form of our Nordic schemes. *NKS* started its first programme in 1977 and has in the past 19 years carried out a number of four-year programmes, the emphasis of which have shifted over time.

The three first programmes were with Nordic financing from the Council of ministers, which implied that political goodwill was required at the annual approval of budgets. After the Brundtland report "Our common future" a fight between the energy ministers over the use of Nordic energy research funds started. This resulted in a withdrawal by Sweden from the *kontakt-organ* - and without the Swedes it is hardly possible to continue a Nordic cooperation in this field.

But at the same time *NKS* was transferred from the political level to the competent authorities, who together have entered a Consortium agreement. Since 1990 the programmes have been free from Nordic bureaucracy, which has been a great advantage. The Consortium group and a number of co-sponsors provide a basic finance of approximately USD 1 million annually, while the participating organisations make up to about an equal level.

In the eighties NKS' work has evidently been influenced by the accidents at TMI and Chernobyl. Earlier *NKS* programmes on control rooms were turned first towards human factors and then towards computerised support in the management of accidents. Reactor safety work, originally on quality assurance and core calculations, was extended to include probabilistic methods and accident sequences. Work on emergency, countermeasures, public information, and radioecology received a new impetus after Chernobyl. All of the five *NKS* programme have included waste management questions.

5. So what have been the results of this Nordic venture?

What has been achieved over so many years of sometimes cooperating and sometimes competing? Was it worth it, all these projects, meetings, conferences, journeys?

Looking back, a number of substantial results can be displayed. We shall first look at some of those which have made the Nordic label visible to the outside world and gained for them recognition and respect.

Emergency assistance
In 1959, when questions about reactor safety came up, a formal Nordic agreement on mutual emergency assistance in case of radiation accidents was drafted. According to this, the countries shall provide assistance - against reimbursement - to one of the contracting states in case of a nuclear accident. In order to enhance international cooperation, also the IAEA became part of the agreement, which was signed in 1963. This was a first-of-its-kind international convention in this field, to be followed by similar agreements much later, after the Chernobyl accident.

After the accident at Thule in Greenland in 1968, when a US bomber crashed and plutonium was spread over the ice, Iceland - who had no nuclear installations - saw the necessity of joining this arrangement. This was done in a typically Nordic fashion through telephone calls "among friends" and just confirmed by an exchange of letters with the Danish authorities.

NARS
The *NARS* recommendations on requirements for licence applications were issued in 1974. Their principles have been used as the individual authorities advanced with more detailed national regulations. All the Nordic safety authorities have taken the same approach in their relations to applicants and licensees: it is up to the applicant to demon-

strate the safety, not just by fulfilling certain requirements stipulated by the authorities. This approach has contributed to the good relations between authorities and utilities in the Nordic countries and the high safety level achieved.

Border agreement
During its work *NARS* found that there was a need to assure contact, in an early phase, with a neighbouring country's authorities if a nuclear plant were to be located close to its border. An exchange of information ought to be mandatory, and the neighbouring country should have the right to request consultations. The *kontaktorgan* took up this issue and an agreement was formalised in 1976 in the four countries and referred to as being a specific case in a more general Nordic environmental agreement.

Flag books
One outcome of the cooperation among radiation protection agencies is the series of Nordic "flag-books". They deal with international recommendations in the field of radiation protection, adapting them to the conditions prevailing in the Nordic countries. The first "flag-book" on radiation protection norms was issued in 1976. It was followed by others, dealing with more specific issues, in particular basic criteria for high level radioactive waste. As a result, national regulations, although still independent, are on the same basis.

"Nordic groups"
Several Nordic groups have made themselves known internationally over time. Three of them can be mentioned here: The NORHAV group, with specialist in accident sequence calculations; the Nordic transport group, at a certain time forwarding joint comments to proposed IAEA regulations; the "Nordic atomic libraries joint secretariat" who entered into international agreements on computerised documentation, including the safety and waste fields.

Other achievements are perhaps not known outside of the Nordic area but deserve to be mentioned. Perhaps the most important is that Nordic authorities have acquired a basis for uniform working patterns in fields such as quality assurance, environmental measurements, and emergency procedures. Here are some other achievements:

Contact forum
Following Chernobyl, a Nordic contact forum for information specialists has been established. The forum organises meetings with representatives from media to enhance their understanding of nuclear questions and risks. Its working group, with participants from the authorities, establishes Nordic contacts which now have a high priority in nuclear emergencies.

NKS reports
One main category of achievements is the material which has been made available - as a background material - for the regulatory authorities through different *NKS* projects. As an example, based on a post-Chernobyl project, there is now a system for transfer, among the Nordic networks of gamma measuring stations, of data about radioactivity in the atmosphere. There is also a catalogue with reactor data that helps analysing threats from nuclear installations in neighbouring countries.

Networks
Apart from such identifiable, substantial results as those mentioned, a major outcome of many years of cooperation is the contact net of authorities and experts which has proved its value many times, not least in the difficult days of the Chernobyl crisis.

International weight
The cooperation and the "aura" that surrounds it has helped the Nordic countries to make their voice heard internationally in questions about nuclear safety and radiation protection.

6. Some failures

Not everything can succeed in a cooperation scheme that is not based on integration but on case by case cooperation, which is typical for most activities in the Nordic setup.

Thus, in 1968 there was a major political attempt to establish what would have corresponded to a Nordic common market, the NORDEK plan. As part of this, attempts were made to create a Nordic reactor vendor and to combine the nuclear research institutes into one united nuclear development center. On the industrial side this was unsuccessful because the Swedish ASEA-Atom was already too far ahead of the other countries, and on the political side the plan collapsed. Instead, a Nordic Council of ministers was created.

Over the years, three separate attempts have been made to share the production of radioactive isotopes for medical use between the research reactors in the four countries, but each time it failed due to lack of sufficient commercial incentives.

When political viewpoints override technical or scientific initiatives, this may strike back on the success of cooperation. As the official attitudes to nuclear power changed in Denmark and Norway in the eighties, the countries' position in international discussions sometimes differed, for example in the London Dumping Convention.

We have also seen how authorities in the Nordic countries have taken different countermeasures in response to the Chernobyl contamination. With to-days cross-country TV and radio, this has given problems for the public opinion and harmed the confidence in the authorities.

Following Chernobyl, a "TRANSAM" project was proposed in order to verify models for long-range atmospheric dispersion. It was impossible to obtain financing, so it had to be dropped as a Nordic project. Later it was taken up by the IAEA in conjunction with other international organisations.

7. What did we learn from our specific Nordic way of working?

If we shall try to draw some conclusions from our cooperation over the last 40 years, some factors can be identified that have been helpful in obtaining our results.

Some of them are specific for the Nordic character of our cooperation. We benefit from easy mutual communication, most of us being able to use own languages. In our countries we have had - and have - personalities with strong pro-Nordic sentiments, and our cooperation got early political support, nationally and in the Nordic Council. We were helped by a generally accepted

desire for a certain independence from the great powers, and at the same time a desire to make our combined voices heard.

We have fairly unbureaucratic working habits which makes life for participants in our joint ventures easier than, e.g., in the EU. Also, we have learnt that organisational frames for the cooperation must be under constant review so that they can be adapted to changing circumstances.

A cooperation between independent institutes and authorities, such as we have them in our field, should be voluntary - not imposed. That implies that in each case there must be full national freedom to join a proposed venture or not. The matters dealt with should be in the pre-competitive stage. It is often easier to cooperate in producing surveys and drawing joint conclusions than to perform real research.

Dedicated persons, both in the participating countries and "neutral" ones are essential to look after upcoming issues and adapt them so that they will be suitable - or even attractive - for co-operation. Goodwill throughout the organisations can be encouraged if persons on several levels are engaged in some manner in the cooperation scheme. In spite of modern communication methods, the value of meeting activity should not be underestimated. Networks should be used in normal work so that they will function in emergency situations.

8. What can we contribute to the future?

Our Nordic cooperation is continuing and moving towards to-morrow's areas of concern. The recent events that will influence future Nordic programmes are, on one hand, our concern for safety in our eastern neighbouring countries and, on the other, the fact that all of our countries now join EU project activities and its nuclear safety work, although two of our countries are not members.

The sixth *NKS* programme, to start after 1997, will certainly be marked by this development. Can the Nordic countries improve their coordination of assistance to sources of concern in the east (e.g. Ignalina)? Can we help a coming EU framework programme with preprojects or test cases like our present project on an European intercalibration exercise that follows a successful Nordic experience? Can we make a contribution to a rapprochement between concepts used in nuclear safety and in radiation protection?

Our international contacts are essential for us. We are quite proud that our cooperation has international links, like the ones in the present conference. We feel that our regional scheme has much to give in the future, both for our own countries, and through the potential for impact it gives in a much larger framework.

ESTABLISHMENT AND MAINTENANCE OF PROFESSIONAL CREDENTIALS IN RADIATION PROTECTION

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ABSTRACT

Recognition of qualified experts in radiation protection is an issue IRPA has been concerned with from its inception. It has long been known that the recognition mechanism differs widely throughout the world community and IRPA Associated Societies have each dealt with the needs of their members in this regard in their own way. Some unification of the recognition of qualified radiation protection experts was first thought to be important with the organization of the Commission of the European Communities, anticipating the need for such expertise to be able to move freely within the EC states. A number of attempts have been made to determine the feasibility of such standardization by first intercomparing the existing systems internationally. Such intercomparisons have only verified the wide diversity of existing recognition systems, confirming the difficulty that would be associated with any standardization attempt. We are therefore shifting our focus to the issue of professional education and training as a means of gradual standardization in the profession.

INTRODUCTION

Establishment and maintenance of professional credentials by individual members of the IRPA associated societies has long been identified as an area in which IRPA might be of valuable assistance to it's membership. These issues have been identified repeatedly by the long range planning committee as important areas where IRPA can act as a useful resource. During the Associate Societies Forum at the Sydney congress in 1988, several societies expressed interest in developing criteria for certification and training which might be recognized internationally as well as nationally. In August of 1989 the Commission of the European Communities announced it's intention to conduct a survey on the radiation protection training and certification in it's member states. The purpose of that survey was to identify and review the training programs and methods of recognition utilized among members of the european community in order for the training organizers to assess and decide which of the programs might best fulfill the requirements of the newly organized community. The ultimate objective would be to ensure parity between the qualifications provided by training bodies in the various member states making it easier for workers to move around in the community. A similar survey had been carried out in 1972 among the six member states which then formed the European Atomic Energy Community. The 1989 survey was intended to include much of the information that had been gathered earlier, updating it and increasing the number of member states to the full european community.

Both surveys concentrated on training for specialists in radiation protection, those responsible for advising, supervising, carrying out or arranging for the radiation protection measures required in any activity utilizing ionizing radiation sources, and for disseminating to the workers involved the radiation protection knowledge required for their work. The surveys also covered training courses for doctors qualified to provide medical surveillance for exposed

workers. Neither survey included the radiation protection training inherent in activities utilizing radiation sources in limited fields such as nuclear medicine.

IRPA REVIEW OF THE CURRENT STATUS

In 1991, members of the IRPA executive council assigned to a task force to review the certification and training issue sent out questionnaires to all of the IRPA member societies. The purpose of the questionnaire was to determine the degree of formality used in the member societies for the recognition of a professional radiation protection specialist. Also included was a question on whether or not formal certification was required, who granted such certification, and the way in which professionals where trained or educated, whether or not a formal certification process was involved. The results of both the 1989 european community survey and the 1991 IRPA worldwide survey were similar in that they revealed the wide disparity of credentialing methods used throughout the radiation protection community. In view of the results of these surveys it did not seem appropriate for IRPA to try to promote an internationally recognized or standardized certification process. It appeared that it may be more useful for IRPA to seek a method whereby it could support the certification and professional credentialing efforts of its member societies by providing supplementary training and refresher courses through IRPA congresses held around the world.

During the 1992 World Congress in Montreal, the Executive Council again considered the certification and training issue and the decision was made to take another look and try to better focus on the best procedure that might be followed to provide the greatest benefit to the associate societies. A task group was formed to further the investigation and add to the earlier efforts. In early 1994 another brief questionnaire was circulated to all member societies and the results of that questionnaire reinforced the 1991 survey, updating earlier data and adding new input.

The results of the European Community study where formally published in a discussion document printed and published by the Health and Safety Executive in London in April 1992. That document provided detail on the common practices in the United Kingdom and the other European Community member states, providing separate information on the qualified expert in medical physics, which provides radiation protection for the patient, and the radiation protection professional, who provides protection to the worker, the public, and the environment. Since much of the data from that 1989 through 1992 study was also gathered in the 1991 IRPA survey we will not repeat the results of the CEC study here but provide reference to the original British document. Figure 1 lists in tabular form the information gathered in the 1991 and 1994 IRPA surveys. The listings appear to show more consistency than actually exists since closer scrutiny reveals that the term "certification" means many different things in different countries. As mentioned earlier the vast difference in formality, legal requirement, recognition, and training methods illustrate just how difficult a job attempting to unify professional recognition internationally could become. The 1994 survey indicated little change since the results gathered in 1991 and it became more apparent that assistance in refresher training and professional enrichment may be the most beneficial role that IRPA may participate in at this time.

PROFESSIONAL EDUCATION AND TRAINING METHODS

An example of the type of effort that may be undertaken is the professional training and assistance program provided by the Health Physics Society for the wide variety of health physics professionals in the United States. For more than two decades now the Health Physics Society has been involved in a growing program of training courses associated with its national meetings. These training programs began as a support mechanism for the American Board of Health Physics Certification program that was originated by and has been continually supported by the Health Physics Society. That independent certification process, which has now been in existence for 35 years, has grown to an active body of certified professionals now numbering over 1000 in active practice. While ABHP certification is often listed as a desired prerequisite or goal on professional placement listings, it is usually not required except for the most senior professional positions. However, ABHP certification is universally recognized and generally well rewarded in compensation scales. Recertification at 4-year intervals requires a significant assemblage of training and education credits during each recertification interval. The HPS refresher training programs were primarily organized to satisfy this need in addition to assisting members in initial preparation for the certification examination.

Such training courses now include the original continuing education classes of one hour duration typically held early mornings on the regular days of the HPS annual and midyear meetings, focusing on current topics, taught by recognized professionals in the field. A growing Professional Enrichment Program was begun about a decade ago, consisting of a variety of 2-hour sessions, many clustered on the day before the opening sessions for the annual and midyear meetings, during which several different training programs are offered in concurrent sessions to meeting attendees, with additional professional enrichment programs offered during the midday typically two at a time on the later days during the meeting schedule. These training programs have recently been supplemented by multiple concurrent all-day sessions organized by the American Academy of Health Physics, offered primarily to assist certified health physicists to maintain their certification requirements. Although it may seem that this rather ambitious venue of formal training programs providing individual opportunities that number into the thousands at a given meeting may appear as overkill, the Health Physics Society has yet to find the depth of the appetite of its members with respect to such educational programs. Most of these courses are filled to capacity well before meeting time and many applicants are disappointed and not able to attend the training session of their choice. For this reason, we believe the development of a comparable program for IRPA world and regional congresses would find substantial support and provide similar opportunities to a broader audience. Figure 2 shows a summary of the training and refresher opportunities available to HPS 1996 Annual meeting attendees.

THE INITIATION OF AN IRPA TRAINING EFFORT

IRPA has therefore begun to offer refresher training to determine how to best help its associated societies to maintain the professionalism of their members. Although we are beginning a program which might provide training resources to those attending regional and world congresses, we want to be sure it is the type of effort the associate societies would find useful and the individual members would find beneficial. Obviously such a program at the

international IRPA level would be restricted to scientific topics covering areas of the profession on a broad base, and could not dwell on the legal requirements or obligation in any given regulatory program. However there is much room for development of such a program and we feel that a reasonable effort could be productive. In general, we feel that continuing education and professional enrichment are a necessity to all members of our profession whether or not (1) certification or formal credentialing are required; (2) such certification or credentialing is granted by the government or by an independent body; and (3) such certification or credentialing requires continuing education for maintenance of that professional recognition.

Examples of how training courses or sessions might be conducted will include early arrival at world congresses or regional congresses for preliminary one or two days of refresher training sessions, such training interspersed with the technical program, early morning sessions during the meeting, one or two days of special training sessions at the end of the congresses, or some mixture of the above. Trainers could include recognized professionals in the country in which the congress is being held, recognized experts from throughout the world, or could take advantage of the presence of members of the IRPA executive council who would be attending the meetings for administrative reasons. Training programs of this sort would not be expected to be expensive. Typically charges are intended to offset the expenses of the speakers in the preparation of their program and the production of the necessary handout materials for the attendees plus expenses associated with the registration and conduct of the course. Once such a program is developed and the techniques refined, it is often possible to realize economies to the extent that a modest profit can be arranged to enhance the IRPA treasury while still maintaining very affordable training opportunities for individual members.

CONCLUSION

Hopefully, such a training and continuing education program will develop in such a way that existing certification programs requiring documented continuing education would recognize these sessions and give credit as appropriate to their certificate holders for participation or attendance. Such an international effort should in time yield some consistency among credentialing requirements, all of which may eventually help to consolidate the professional standards in the profession and begin to yield the consistency internationally people have been seeking. In addition, such an international training effort should be helpful in establishing more standardized treatment of current radiation protection issues.

Meanwhile, the current and recent efforts to document and catalog credentialing practices throughout the world will be followed by the establishment of a program to routinely gather such data on an appropriate updating frequency, standardizing the data gathering effort, and improving the classification system. The data bank created should be a useful reference resource to associated societies considering the establishment or improvement of a professional credentialing system. It could also be useful as a reference to neighboring countries considering a regional reciprocity scheme such as that considered by the European Communities. The reliability and consistency of the data should be gradually enhanced through the repetitious conduct of a gradually improving survey, the results of which should be routinely published and distributed to the IRPA membership.

REFERENCE

1. *Radiation Protection Recognition of the Qualified Expert*, Health and Safety Executive, UK, Crown, copyright 1992.

COUNTRY	CERTIFIED/LICENSED HEALTH PHYSICIST?	REQUIRED BY LAW?	CERTIFICATION/ LICENSING GRANTED BY	TRAINING METHODS FOR HPs
ARGENTINA				
AUSTRALIA	NO			ON JOB
AUSTRIA	YES	YES	GOVT	COURSE
BELGIUM	YES	YES	GOVT	COURSE
BRAZIL	YES	YES	GOVT/PROF	COURSE
CANADA	NO			COURSE/JOB
CHINA PR				
CYPRUS				
CZECHOSLOVAKIA	YES	YES	GOVT/PROF	COURSE
DENMARK	NO			COURSE
FRANCE	NO			COURSE
GERMANY	YES	YES	GOVT	COURSE
GREECE	YES(MEDICAL)	YES	GOVT	COURSE
HUNGARY	YES	YES	GOVT/UNI	COURSE
INDIA				
IRELAND	NO			COURSE/JOB
ISRAEL				
ITALY	YES	YES	GOVT	COURSE/JOB
JAPAN	YES	YES	GOVT	COURSE
KOREA	YES	YES	GOVT	COURSE
MEXICO	NO			COURSE/JOB
NETHERLANDS	YES	YES	GOVT	COURSE/JOB
NORDIC				
DENMARK	NO			COURSE/JOB
FINLAND	YES	YES	GOVT	COURSE
ICELAND	NO			ON JOB
NORWAY	NO			ON JOB
SWEDEN	NO			COURSE/JOB
PERU	NO			
PHILIPPINES				
POLAND				
PORTUGAL	NO			COURSE/JOB
SOUTH AFRICA	NO			ON JOB
SPAIN	YES	YES	GOVT	COURSE/EXAM
SWITZERLAND	YES	YES	GOVT	COURSE
UNITED KINGDOM	YES	NO	PROF	COURSE
USA	YES	NO	PROF	COURSE/EXAM

Figure 1

RESULTS OF IRPA QUESTIONNAIRES ON CERTIFICATION: 1991, 1994

Academy
- Saturday, 3 - 8 hr sessions

Professional Enrichment Program (PEP)
- Sunday, 30 - 2 hr sessions
- Monday-Thursday, 6 ea - 2 hr sessions
  - (3240 opportunities)

Continuing Education Lectures (CEL)
- Monday-Thursday, 2 ea - 1 hr sessions
  - (unlimited)

PEP Categories
- Communications/Training
- Computers in HP
- Radiological Evaluation
- Dosimetry
- Operational HP
- Industrial Hygiene/Non-Ionizing
- Environmental
- Medical
- Legal Aspects

Figure 2

1996 HPS ANNUAL MEETING TRAINING

NKS-programmet 1994 - 1997

Torkel Bennerstedt, exekutivsekreterare

Nordisk kärnsäkerhetsforskning

NKS - Nordisk kärnsäkerhetsforskning - är en frivillig nordisk samarbetsorganisation med rötter tiotals år tillbaka i tiden. NKS är ett serviceorgan för nordiska kärnsäkerhets-, strålskydds- och beredskapsmyndigheter, kärnkraftanknuten industri och andra berörda organisationer. Samarbetet gäller, som namnet antyder, forskning inom kärnsäkerhetsområdet. Definitionen på vad detta i praktiken innebär har varierat en del genom åren, men det grundläggande är reaktorsäkerhet, miljöeffekter och beredskap. För närvarande deltar cirka 250 nordiska experter i nio projekt. Genom den tydliga kopplingen mellan NKS å ena sidan och finansiärer och användare av resultaten å andra sidan förankras projektarbetet hos alla berörda och garanterar praktiskt användbara resultat. Genom ett försiktigt närmande till EU och länderna i östra Europa säkerställs att NKS även på sikt kommer att finnas kvar, till fromma för både Norden och övriga Europa.

Resultatspridning

Med relativt små ekonomiska medel utarbetas information, rekommendationer, databaser och praktiskt användbara manualer för myndigheter, kraftproducenter och andra organisationer som deltar i eller följer NKS' arbete. Utåtriktad information om NKS och resultaten av dess verksamhet sprids genom tekniska rapporter, vetenskapliga artiklar, broschyrer, lättlästa sammanfattningar, kurser, föredrag, seminarier, temamöten etc, både i och utanför Norden.

Därför samarbetar vi i Norden

Den som inte har förmånen att vara nordbo undrar ibland varför just Norden engagerar sig i gemensamma projekt av olika slag. Svaret står kanske att finna i det faktum att de fem nordiska länderna var sammanfogade i den så kallade Kalmarunionen i över 130 år (1389 - 1521). Därefter har de nordiska länderna ofta varit sammanslagna två och två i olika konstellationer under långa perioder. Nordbor förstår varandra lätt, både socialt och språkligt, i varje fall om det säregna men praktiska blandspråket "skandinaviska" bestående av danska, norska och svenska används. Tyvärr är finska, färöiska, grönländska, isländska och samiska inte användbara inom NKS. När så behövs tillgriper vi engelska.

Det historiska och kulturella arvet har gjort det självklart att diskutera med varandra, utbyta information och medverka i gemensamma projekt i de fall man ser att det leder till

praktiska resultat och samordningsvinster. Norden kan komma att bli en spjutspets, till exempel på kärnsäkerhetsområdet, inom EU och globalt. Det faktum att Danmark, Finland och Sverige är med i EU medan Island och Norge valt att stå utanför har inte minskat sammanhållningen, snarare tvärtom.

Den nukleära hotbilden

De senaste decennierna har den nordiska samverkan förstärkts av den gemensamma yttre hotbilden mot vårt territorium från nukleära installationer utanför Norden, inklusive mobila miljöhot som atomdrivna och/eller kärnvapenbestyckade ubåtar, fartyg, flygplan och satelliter. Inom Norden finns sex kärnkraftverk med totalt sexton reaktorer av kokar- och tryckartyp, ett antal forskningsreaktorer, en fabrik för tillverkning av kärnbränsle plus ett antal anläggningar för hantering och förvaring av radioaktivt avfall. Dessa nordiska installationer, tillsammans med transporter av radioaktivt material, utgör den inre hotbilden.

Ett utsläpp från en reaktolycka eller nedfall från en kärnvapensprängning respekterar inte nationsgränser. Nordiskt samarbete för miljöövervakning och utbyte av radiologiska mätdata har varit ett sätt att möta dessa hot och kunna sätta in motåtgärder snabbt och effektivt. Arbetet inom NKS stöder och underlättar sådan samverkan.

Nordisk säkerhetskultur

Till grund för NKS' verksamhet ligger en relativt enhetlig nordisk säkerhetskultur och gemensamma intressen. Syftet är att genom nordiska projekt öka kunskaperna och sprida information inom kärnsäkerhetsområdet, särskilt mot bakgrund av den yttre och inre hotbilden. Där så är möjligt och önskvärt ska NKS-projekten leda till ökad harmonisering av till exempel åtgärdsnivåer och skyddsåtgärder vid en kärnkraftolycka. Där man av geografiska, demografiska, sociala, ekonomiska, politiska eller andra skäl väljer nationella lösningar framför nordiska eller internationella, kan NKS hjälpa till att förklara och rättfärdiga skillnaderna.

Finansiärer

NKS förfogar över omkring 10 miljoner danska kronor per år i direkt finansiellt stöd. Detta kommer framför allt från de så kallade huvudfinansiärerna:

* Beredskabsstyrelsen i Danmark
* Handels- och industriministeriet i Finland
* Geislavarnir ríkisins (strålskyddsinstitutet) i Island
* Statens strålevern i Norge
* Statens kärnkraftinspektion och Statens strålskyddsinstitut i Sverige

Ytterligare finansiellt stöd erhålls från ett antal tilläggsfinansiärer:

*	I Finland:	Inrikesministeriet
		IVO (Imatran Voima Oy)
		TVO (Industrins Kraft AB)
*	I Norge:	Miljøverndepartementet
*	I Sverige:	Statens Räddningsverk
		Sydkraft AB
		Vattenfall AB
		Svensk Kärnbränslehantering AB (SKB)
		Kärnkraftsäkerhet och utbildning AB (KSU)

Till de likvida medlen från huvudfinansiärerna och tilläggsfinansiärerna kommer uppskattningsvis dubbelt så mycket (20 miljoner danska kronor) i icke fakturerade insatser från de medverkande organisationerna. Hit hör ett antal ministerier och departement, nationella kärnsäkerhets-, strålskydds- och beredskapsmyndigheter, kraftbolag och andra kärnkraftanknutna företag samt forskningsinstitutioner.

Projektkriterier

Vilka kriterier måste ett projekt uppfylla för att få ingå i NKS-programmet?

Ett självklart krav är att projektet är av gemensamt nordiskt intresse. Tre eller flera länder måste förklara sig villiga att delta i arbetet och/eller ha behov av resultaten. Ju fler länder som deltar, desto vidare blir perspektivet.

Vidare ska projekten ha ett tekniskt/vetenskapligt värde. Det rör sig minst lika ofta om tillämpad forskning som ren grundforskning. Systematisk sammanställning av redan existerande men geografiskt utspridd kunskap kan ge värdefull information till exempelvis beredskapsansvariga.

Varje projekt ska i största möjliga grad vara en sammanhållen enhet med en "röd tråd", även om det består av ett antal delprojekt där vissa är översiktliga och andra kan vara smalare, djupare.

Projekten ska vara kostnadseffektiva och utnyttja de samordningsfördelar som det nordiska samarbetet ger. Där så är möjligt och lämpligt samordnas projekten internationellt. Det gäller till exempel EU/Europeiska Kommissionen, OECD/NEA, IAEA. Även samarbete med Baltikum och andra öststater förekommer men måste då finansieras av någon annan part än NKS.

Pågående projekt

NKS' verksamhet delas in i fyraårsperioder. Just nu är vi inne i den femte fyraårsperioden, 1994 - 1997, med totalt nio projekt. Dessutom finns det en samordningsfunktion som bland

annat övervakar att projektplaner och budget hålls samt hjälper till med ekonomi och administration.

Reaktorsäkerhet

Ett projekt (RAK-1) behandlar strategi för reaktorsäkerhet: hur man undviker stora olyckor. Säkerhetsarbetet på kärnkraftverk och myndigheter kartläggs och värderas utgående från ett antal aktiviter och fallstudier, bland annat moderniseringen av Oskarshamn 1. Inledande händelser (med bestämning av frekvenser för stora rörbrott, LOCA), integrerad sekvensanalys (inklusive mänskligt felhandlande), underhållsstrategier, åldring samt ändringsarbeten vid kärnkraftverk studeras i ett antal delprojekt.

I ett parallellprojekt (RAK-2) undersöks möjligheterna att minimera konsekvenserna i händelse av en stor olycka. Fenomenologi vid svåra haverier är temat för ett delprojekt. När kan en överhettad härd fortfarande kylas med vatten? Vad kan bli följden av vattenkylningen? Härdsmältans utflöde från brottstället på reaktortanken simuleras i teoretiska och experimentella studier. Datoriserade system som stöd för beslut vid haverier (CAMS) ingår också i detta projekt, liksom en reviderad sammanställning av reaktorer i Nordens närområde.

Avfall

Flera nordiska länder måste snart fatta beslut om slutförvar av radioaktivt avfall. Ett projekt (AFA-1) ägnas därför åt slutförvaring av långlivat låg- och medelaktivt avfall. Karaktärisering av avfallet (radiologiskt och kemiskt) samt funktionsanalys och miljökonsekvensbeskrivning (MKB) för slutförvaret är de viktigaste sakområdena i projektet. Informationsspridning och kunskapsöverföring förväntas bli väsentliga resultat av arbetet. Eftersom högaktivt avfall främst är aktuellt för Finland och Sverige ingår det inte i detta projekt.

Radioekologi

I ett projekt om marin radioekologi (EKO-1) är huvudmålet att snabbare och säkrare än hittills kunna värdera effekterna av radioaktiva utsläpp i den marina miljön. Hänsyn ska då tas till både ekonomiska och hälsomässiga faktorer. Genom provtagning och analys tolkas sedimentprofiler och bestäms sedimenteringshastigheter. Växelverkan mellan sediment och havsvatten studeras. Genom modellarbete utvärderas de radiologiska konsekvenserna av dumpat kärnavfall i Karahavet. Kvalitetssäkring och utbyte av information är andra viktiga aspekter.

Provtagning, analys, modellarbete och kvalitetssäkring är väsentliga beståndsdelar även i projektet om långa ekologiska halveringstider i seminaturliga system (EKO-2). Överföringen av radioaktivt cesium och strontium i kedjorna mark - bete -

får (lammprojektet) och svamp - rådjur (skogsprojektet) studeras, liksom ekosystem i sötvattenmiljö (limniska projektet). Den limniska delen genomförs i samarbete med projektet om marin radioekologi för att jämföra resultat och se om några gemensamma slutsatser kan dras.

Beredskap

Ett projekt om beredskapsstrategi och -procedurer (EKO-3) inriktades under 1995 väsentligen på mobila mätningar, och i augusti 1995 genomfördes en stor internationell jämförande mätövning i Finland. Tillsammans med EU förbereds nu en liknande övning 1998. Kvalitetssäkring vid provtagning och analys, med ackreditering av laboratorier för livsmedelsmätningar som mål, är ett annat EU-inspirerat delprojekt. Operationella åtgärdsnivåer samt mätstrategi, beslutsunderlag och motåtgärder inom lantbruket är andra viktiga delprojekt.

I ett annat beredskapsprojekt (EKO-4) planeras och genomförs nordiska funktionsövningar. I samarbete med den internationella övningen ETEX-1 genomfördes nordiska spridningsberäkningar med olika meteorologiska modeller, och resultaten jämfördes. En nordisk konferens om harmonisering av åtgärdsnivåer har genomförts, likaså en övning avseende dosberäkning efter ett nedfall. En stor nordisk slutövning 1997 förbereds inom ramarna för den av OECD/NEA organiserade internationella övningen INEX-2. Ett förslag till nordiskt system för data- och informationsutbyte utarbetas.

I ett relativt nystartat beredskapsprojekt arbetar man med planläggning av tidig sanering efter ett radioaktivt nedfall (EKO-5). Man studerar effektiviteten av olika åtgärder samt utvecklar ett antal nedfallsscenarier med avseende på årstid, typ av bebyggelse (stad, förort, landsbygd), nuklider samt doser inomhus och utomhus. Saneringsstrategier utarbetas för olika nivåer av kontaminering. En kostnadsanalys genomförs och den avvärjda dosen uppskattas för de olika fallen. Slutmålet är att göra en checklista som lätt kan användas av planeringsansvariga och beslutsfattare.

Informationsfrågor

Det likaledes nystartade projektet om övergripande informationsfrågor (SAM-4) har flera syften. Dels ska man informera internt och externt om NKS' verksamhet, dels ska man utveckla nordisk riskkommunikation och NKS' informationsmål, dels ska man vid behov medverka med informationsinsatser i andra NKS-projekt (exempelvis den nordiska slutövningen nästa år) och utbilda projektdeltagarna i kommunikationsteknik. Ett nordiskt journalistseminarium med anledning av 10-årsdagen av Tjernobylolyckan har genomförts.

Projektstyrning

NKS' huvudfinansiärer utgör den så kallade konsortialgruppen, som beslutar om tilldelning av medel till NKS och fastställer budgeten för de pågående projekten.

NKS-arbetet leds av en styrelse med i genomsnitt tre representanter per land, däribland de fem konsortialparterna samt experter inom de fackområden som NKS-programmet omfattar. En del av styrelsens uppgifter delegeras till NKS' byrå, som består av ordföranden, sekreteraren och exekutivsekreteraren.

Varje huvudområde inom NKS-programmet (RAK, AFA, EKO) har en referensgrupp som följer projekten, diskuterar dem med projektledarna och rapporterar till styrelsen. Ordförandena för referensgruppen hämtas ur styrelsen.

Det tekniskt-vetenskapliga arbetet i de nio projekten organiseras, samordnas och följs upp i ett administrativt projekt, SAM. Till detta hör NKS-sekretariatet, som är placerat i Risø i Danmark. Det är SAM som ansvarar för att det publiceras hel- och halvårsrapporter för hela NKS-programmet och ges ut slutrapporter för de olika projekten. Andra viktiga uppgifter för SAM är att organisera styrelsemöten, referensgruppsmöten och seminarier, säkerställa finansieringen av NKS-programmet, hålla de deltagande organisationerna informerade om hur arbetet fortskrider samt i lämplig utsträckning initiera och stöda samverkan med EU och länder i östra Europa.

Klaus Sjöblom (LOVIISA NUCLEAR POWER PLANT)
Hannele Aaltonen &
Riitta Hänninen (FINNISH CENTRE FOR RADIATION AND NUCLEAR SAFETY)
Eldri Naadland (NORWEGIAN RADIATION PROTECTION AUTHORITY)
Åke Persson (SWEDISH INSTITUTE OF RADIATION PROTECTION)

NORDISKA SÄLLSKAPET FÖR STRÅLSKYDD, MÖTE I REYKJAVIK 26-30.8.1996

INTERNATIONAL EMERGENCY EXERCISE IN NORDIC COUNTRIES 17.4.1997

Abstract - On april 17th, 1997 there will be a large international emergency exercise which is also a part of Nordic co-operation. Nordisk kärnkraftsekärhet (NKS) has included this exercise as a part ("stor övelse") in EKO-4 project. The accident is postulated to happen at Loviisa NPP. The exercise will last one day with threat, release and immediate post-release phases. The radioactive release is going to be transported according to real weather conditions. Rescue service authorities in host, bordering and far-field activate their command posts. The main acitivities are: Real time information, decision making based on plant condition and public information exchange.

Participants and time schedule INternational EXercises of INEX 2

INEX 2 will be composed of serie of international exercises:

Accident host	NPP	Time
Switzerland	Leibstadt	November 7th, 1996
Finland	Loviisa	April 17th, 1997
Canada	Darlington	November 1997

At each nuclear power plant the utility emergency organization is activated and the national rescue service authorities will be alarmed. In addition to accident host country, bordering countries will participate simultaneously, activating their own emergency command posts to receive and transmit information on which to base their decisions as to protective countermeasures. More remote countries - called "far field countries" - will also participate

IVO GROUP

using their existing bilateral and multilateral communication agreements. These exercises will be sponsored by OECD/NEA. The IAEA, CEC and WMO will also participate.

Finnish rescue service authorities participation in INEX-2-FIN

The Finnish regulations require that off-site rescue service authorities have a large co-operation emergency exercise at least every third year for each nuclear power plant. The exercise on April 17th, 1997 is going to fullfil also this requirement and also for this reason the participation will be comprehensive:

- GOVERNMENT: responsible ministers & government office
- MINISTRIES of
  interior, forest & agriculture, trade & industry,
  social & health affairs, traffic, Foreign ministery
- NATIONAL BOARDS:
  Finnish institute for radiation protection and nuclear safety (STUK),
  Meteorological institute, Aviation board, broadcasting company,
  road department, Finnish news agency (STT)
- PROVINCEs (Uusimaa & Kymi)
- LOCAL RESCUE SERVICEs at Loviisa co-operation area:
  local alarm center, fire brigades, communal command posts, coast guard, police.

The planning is supervised by a direction committee that has representatives from ministery of interior, STUK, government office, Finnish rescue institute, IVO power company; also provincial & local authorities, that have rescue service responsibilities conserning the other Finnish NPP, contribute to exercise planning; this committee is chaired by senior inspector Tarmo Kopare from ministery of interior. More topical sections, like plant process and radiological scenario, are being planned in separate planning groups.

Nature of INEX 2 exercises

To be more realistic than INEX 1 exercises, INEX 2 exercises are based on a postulated accident at a **real nuclear power** plant with **real weather** conditions. The exercises will last one day with threat, release and immediate post-release phases. The exercise is going to be long enough so that release will affect a large area, at least to some extent. In key postitions the personnel should be rotated which urges a larger number of persons to prepare themselves for emergency preparedness activities.

The main acitivities of INEX 2 are:

- Decision making based on plant condition.
  - The decision makers have to make a compromise between early reactions based on less knowledge and late actions based on more complete knowledge. The potential release including its source term should be assessed; it might be a more important argument than the readings of radiation monitors on-site and near-site.

- Real time exchange of information
  - A severe radiation accident will have a world-wide interest far beyond the potential or postulated contamination area. The nature of the accident and it consequences should be assessed, informed and understood correctly in each command post.

- Public information
  - This may be the most visible part of emergency activities. The authorities and the power company have to demonstrate that they have made proper assessments and countermeasures.

The effect of real weather

Why real weather instead of postulated weather?
Traditionally emergency exercises have used postulated weather conditions because in this way

- Radiological monitoring data based on detailed calculation can be prepared carefully
- Radioactive releases and contamination can be directed to towns or across borders or to other "pedagocically interesting" locations

However, modern large area, two- or three dimensional computer models that calculate the transportation of radioactive releases make use of compehensive meteorological data that is transmitted automatically in file formats from a large number of weather stations. The computer models use both real time data and its history. It is impossible or at least inpracticable to generate enough postulated weather data or to use some old weather data. For this reason there is a general trend to use real weather in emergency exercises.

In case of Loviisa the potential exercise weather conditions can be classified to two categories:

1. Release towards inland: The population density near-site is relatively low and there are no state borders.

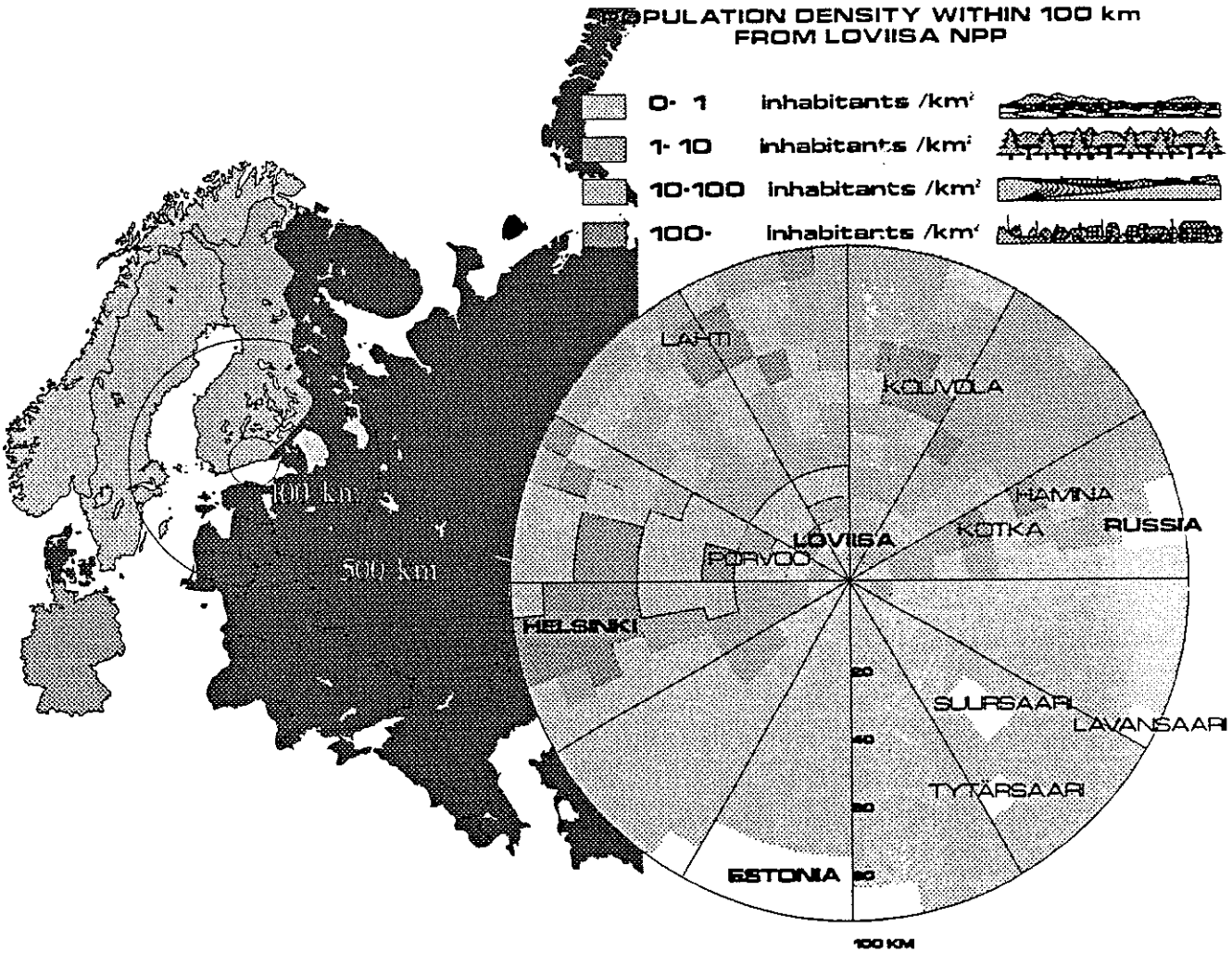
2. Release to the sea: Almost no population until Russian Islands (50 km) and Estonian coast (80 km).

* Extreme weather conditions (storms or very high/low temperatures or sea water levels) can be excluded; locally, mainly in bays, the ice cover may limit moving

The accident scenario is, of course, secret. However, one can speculate how far a release could travel during the exercise?

- Theoretically: 10 m/s * 14 h = 500 km e.g. to Stockholm
- Realistically: 5 m/s * 6 h = 100 km e.g. to Helsinki or Tallinna

These 100 km and 500 km circles have been indicated in the next figure.



The situation is very different from the INEX-2-CH exercise where the Swiss nuclear power plant is just at German border. It is probable that in INEX-2-FIN no neighbouring country will receive the postulated radioactive release during the exercise. For this reason the protective actions will be needed mainly in the national exercise and in the international part

the main activities will be, as stated earlier: decision making based on plant condition, real time exchange of information and informing public.

Nordic aspect

Nordic Nuclear Safety Research program 1994-1997 contains a large emergency exercise that is going to be combined with the INEX-2 objectives. This encourages all Nordic countries to participate actively in the exercise in April 1997.

Because of geography Scandinavian countries should concentrate on those activities that are not directly related to an immediate contamination situation. All five Nordic countries have representatives in the exercise planning committee and they will inform about the exercise in their home countries:

SVERIGE: Åke Persson & Åsa Wiklund / SSI
NORGE: Eldri Naadland & Finn Uggletvit / Strålevern
DENMARK: Vibeke Hein / Beredskapsstyrelsen
ICELAND: Elisabeth Ólavsdóttir / Geislavarnir Rikisins
FINLAND: Hannele Aaltonen, Riitta Hänninen /STUK

The information persons have planned their co-operation. Observers from Estonia, Latvia and Lithuania are going to be invited to assist them to organize emergency exercises in the home countries in the future.

Preparation for the exercise:

The exercise day will give valuable experience but the main benefit comes from the preparations. Each organization has good reasons to evalueate and modify their plans, train their personnel, assess the need of new equipment and check communication devices & contact numbers.

The exercise coordinators are not going to control each organizations activities in detail. Rather these should take initiatives of their own in order to have an effective exercise: What could our organization do in an exercise like this? How we simulate it? Could we take initiative to track problems and present them others?

The exercise organizers have a good opportunity to assess the exercise arrangement: how to reach a motivating and realistic exercise situation and how to develop the exercise planning in the future.

Nordisk riskkommunikation och utveckling av NKS' informationsmål

Informationschef Vibeke Hein
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Abstrakt

Atomenergiulykker respekterer ikke nationale grænser. Det er en af grundende til, at et nordisk samarbejde om befolkningsinformation er vigtigt. Sker der en ulykke, vil den få konsekvenser for alle/flere af de nordiske lande. Med de tætte grænser samt sproglige og kulturelle forbundethed vil oplysninger givet i et af de nordiske lande blive sammenlignet med oplysninger givet i nabolandet. De nordiske myndigheder opnår derfor en langt større troværdighed, hvis de kender til hinandens informationer, så eventuelle forskelle i foranstaltninger kan forklares. Information om risiko er svær. Det er den, fordi det ikke er entydigt, hvad der er farligt. Hvordan informerer man om et svært emne til en modtager, der ikke er interesseret - det er et af de spørgsmål, vi har stillet os. Det drejer sig udelukkende om forhåndsinformationen. I tilfælde af en ulykke er interessen pludselig **meget** stor, og informationsstrømmen synes ofte for ringe. Hvilke faktorer spiller ind, for at mennesker opfatter risiko på det rette niveau, bliver bange, hvis det er nødvendigt, eller undgår panik selv om situationen er ukendt - det er noget af det, vi i fællesskab forsøger at analysere. Hvad kan man egentlig opfatte, når man er i en presset situation - det ved man noget om også fra andre områder, fx fra hospitalsverdenen, en viden som vi samler ind for at blive bedre til at kunne tilrettelægge den information, der skal sendes ud, når noget sker. Ét vigtigt element er uden tvivl muligheden for at få svar på egne spørgsmål, at kommunikere og ikke bare modtage informationer.

Det arbejdes sideløbende med de generelle informationsspørgsmål med at forbedre kendskabet til NKS. Der oppstilles kommunikationsstrategi og grafisk profil, ligesom der afholdes kurser i formidling for projektledere, og nye kommunikationskanaler tages i brug.

Information exchange with internet technology, the NKS/EKO-4.2 project

Tord Walderhaug,
Geislavarnir ríkisins

Abstract - The EKO-4.2 project involves investigating approaches to setting up a new reliable system based on computer internet technology for exchange of information between the Nordic countries. Since a potential system will have to be implemented by the authorities in the respective countries, the first part of the project should survey their interest and emphasis, which was solved with meetings in the different countries. The conclusion from those meetings was an overall interest in establishing a system as described, and several requirements concerning security and reliability of the system were proposed. A practical solution of an information exchange system based on the World Wide Web, the File Transfer Protocol, and the Internet is outlined. The continuation of the project must be based on operable servers in the different countries, whose configuration and connection with the local network are the responsibility of the institution owning the server. The solution will depend on hardware and on the overall security policy of the institution involved. It is proposed that further work must concentrate on a general testing of a system, standardizing of server user interfaces and testing of encryption software for transfer of passwords.

Introduction

Information exchange between the Nordic authorities in case of nuclear accidents is today mainly based on the telefax. The obvious limitations of this technique, initiated the NKS/EKO-4.2 project, which should look into the possibility of using computer networking for the information exchange instead. The project was initially thought as a two-step process. First the authorities should express their expectation and requirements of such a system, and later on a system should be set up according to the specifications expressed by the authorities. The NKS should not invest in any equipment for the information exchange system, instead it should investigate the different approaches of setting up the system. By the fall of 1995 Geislavarnir ríkisins had set up an information server, which closely matched the requirement specified by the authorities. Further work, such as an overall testing of the system, transfer methods of passwords, encryption procedures and standardisation of user interfaces, must be based on operable servers, and is therefore delayed until the other countries have set up their servers.

Telefax vs. computer network for information exchange

The main advantage of the telefax, the current information exchange system between the Nordic authorities, is its reliability. It is based on the telephone network which has the very first priority of the telephone companies, and is in addition very easy to maintain. If a fax machine should e.g. break down, a new one can be plugged in without any complicated set up procedures. Its list of drawbacks, on the other hand, is extensive. The most important being the bad quality of the messages and the difficulty in processing the information further, all which is solved by the use of a computer network instead of the telefax. Data where a system based on computer networking is especially preferable, are graphical presentations and numerical data which can be processed directly. Examples are contamination maps, weather maps, and files containing nuclear inventories to be used directly in fallout/dose calculation software, but all types of computer files may be transferred, such as software, word processor documents and scanned pages from reports or text books.

The main drawback of an information system based on the Internet is the lack of reliability. One

solution to that is to set up a dedicated network, based on the telephone network, but this is very expensive. A better solution is to use the Internet, with direct dial-up connections as a reserve system. This reserve system may use the same server and software as the Internet connection, or may be a connection to a mirror server with remote access software. It must, however, be emphasized that though the communication line can be made highly reliable, the computer system itself may break down, and is today not as easily replaced as a telefax machine. It is therefore difficult to find a replacement for the telefax as an alarm system, and the telefax can also be considered the last resource if everything else breaks down.

Electronic information exchange can basically be set up in two ways, either as an electronic mailing list, or as a bulletin board system. The former method lacks an automatic acknowledgement procedure. This is not important for short messages, but is a problem in transfer of bigger files. On the other hand is the receiver notified of the existence of new information. The bulletin board system does not need any acknowledgement procedure, since it is the receiver who performs the actual information transfer. This system, however, has no automatic notification of new information available for transfer. It may therefore be of interest to have a combination of those two methods, where short electronic messages are sent with information about the content of the servers.

The emphases of the authorities

The following were emphasized by the authorities, when the plans for the system were presented:

- The security of the local network must not be compromised.
- Security of the data and the data server must be ensured
- Confidentiality of data must be ensured
- The system must allow access restriction with discretionary access control
- It must be highly available and reliable
- It must be easy to use

The security of a computer network engaged in communication with other networks, depends mainly on how the network is configured for the communication. Several options are available, from the data server being completely isolated from the local network, i.e. the data must be transferred to and from the data server by means of diskettes or some kind of dial up connection (a 100% secure solution), through different configurations with the local network involving specially configured gateway computers (firewalls) that check, route, and label all information passing through them, see figure 1, 2 and 3. In the first configuration the local network is actually not engaged in the communication at all, instead the data server operates as a network in its own right. The owner of the local network is solely responsible for the communication, and the configuration will thus depend on the institution's own network security policy.

The data server (the host computer) cannot be 100% protected. Firewalls can increase the security, but even then break-ins can be expected. They will however be rare (every few years), brief (the firewall system is quick to detect anomalies), and inexpensive (serious incidence involving system damage and data loss is very unlikely). When the data are transferred between networks, they will pass several intermediate networks, with the possibility of being tapped. The system must therefore be used with the risk in mind. A straightforward way of ensuring data confidentiality, e.g. with transfer of passwords, is encryption, and must be considered.

The authorities' requirements of access restriction with discretionary access control are easy to implement with the server software applied for the Internet. The newest software with graphical user interfaces is also easy to use, as shown in the next section.

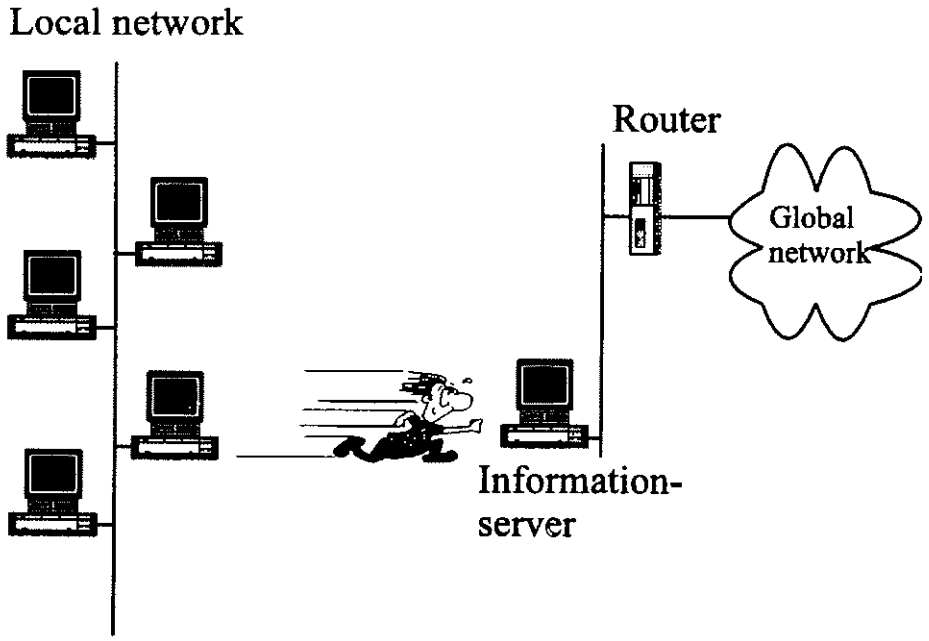

Figure 1. No connection between the local network and information server. Data is passed between the local network and server with diskettes.

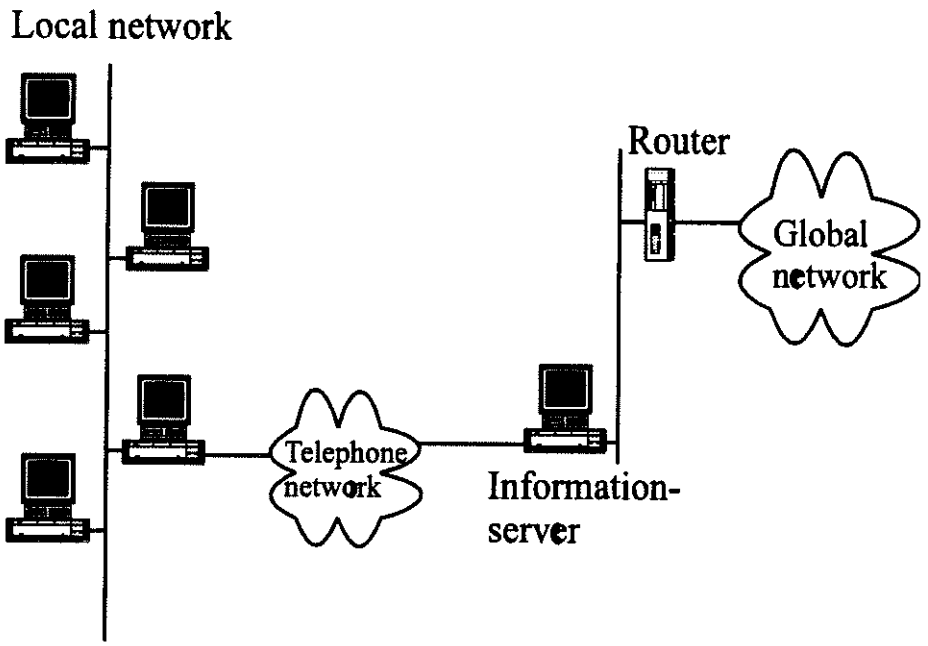

Figure 2. Direct dial-up connection between the local network and information server. By configuring the dial-up connection correctly, 100% security for the local network is available. The extra connection to the data server on the other hand, may decrease the security of that part of the system.

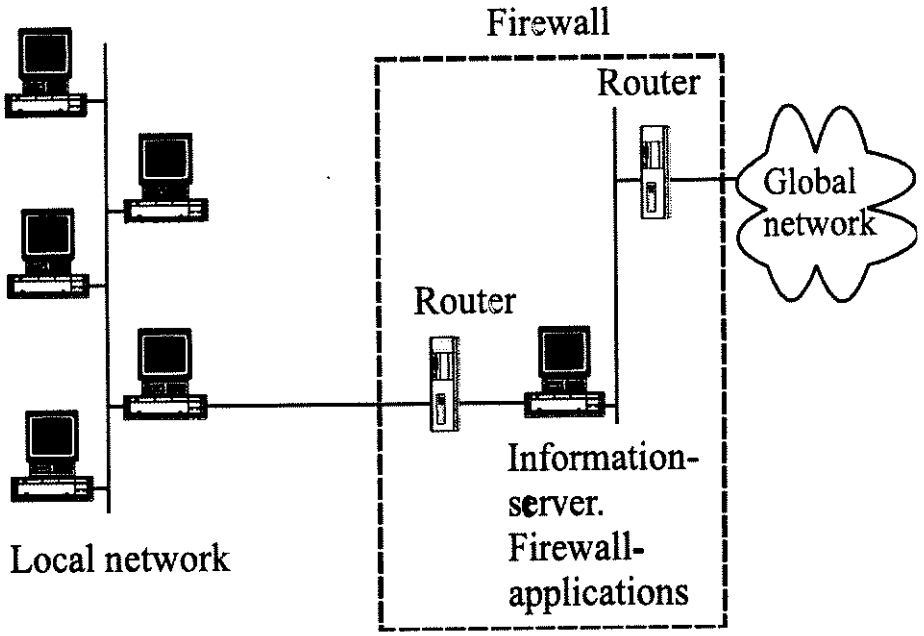

Figure 3. A configuration with a firewall. The local network is not 100% protected. However, it is very unlikely that break-ins will have serious consequences.

An example information server

An example data server which closely matches the requirements specified by the authorities, has been set up at Geislavarnir ríkisins. It is based on CERN httpd World Wide Web server software running on a Unix computer with a firewall connection to the local network. The transfer of data is accomplished with the File Transfer Protocol. The figures 4, 5 and 6 show how a session where a file is down loaded may appear, using Netscape Navigator browser software to attach the server. On figure 7 is shown how a new file is made available on the server.

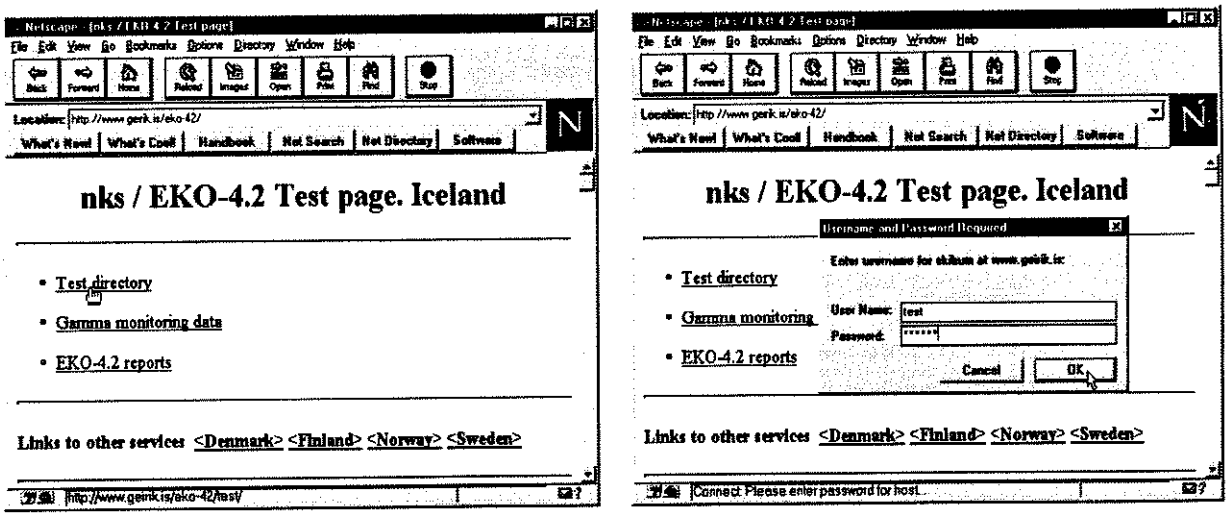

Figure 4. A user interface of an information server showing different directories containing data, as well as links to similar services in the other Nordic countries (left). The directories are password protected with different passwords for discretionary access control. When a directory is selected, a window pops up acquiring user identification (right).

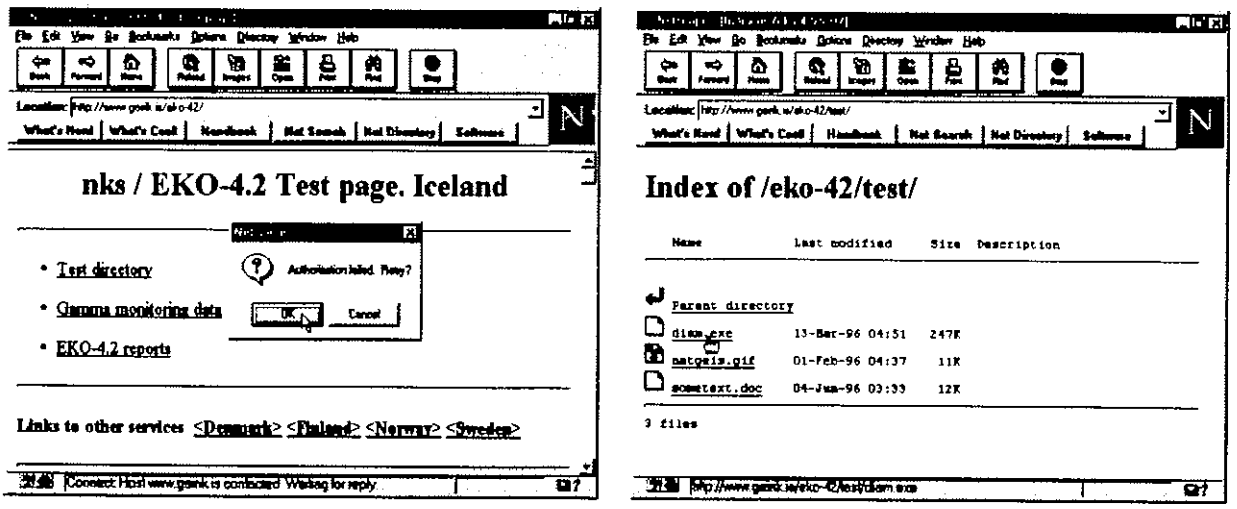

Figure 5. Incorrect user name or password is notified with an "Authorization failed" message (left), while a correct user identification gives access to the directory with the information (right). The different files can now be transferred to own system with a mouse click.

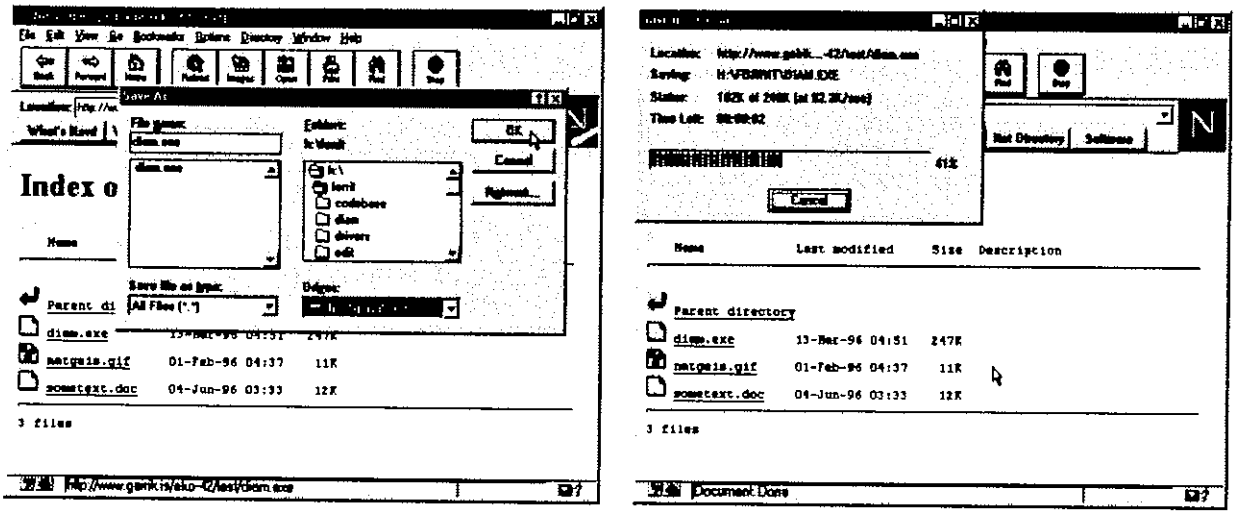

Figure 6. When a file is selected, the user is first asked where to save the file on own system (left), and the file is then transferred (right).

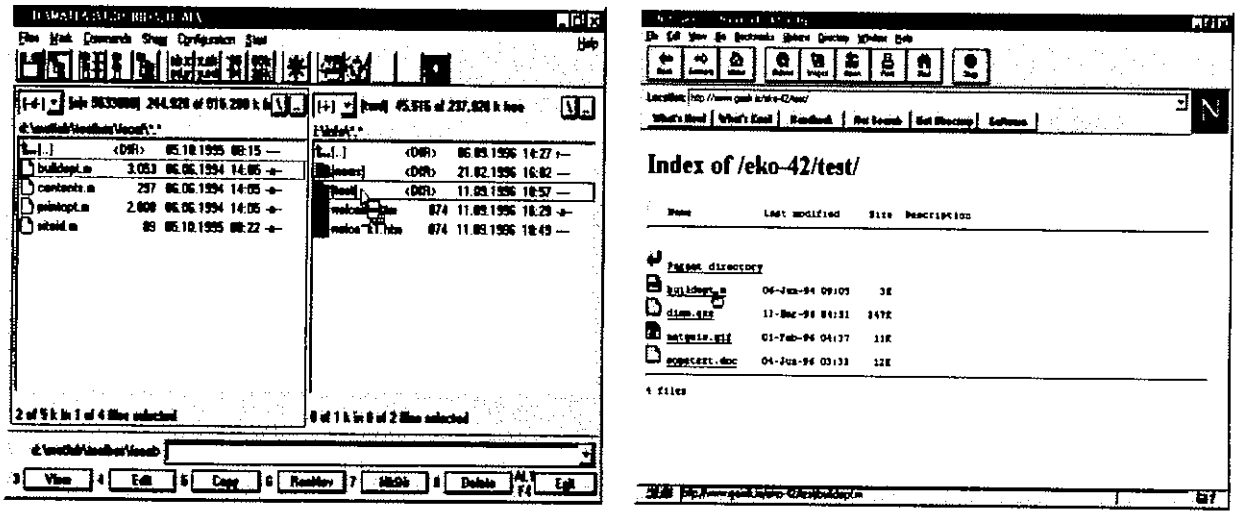

Figure 7. A new file is made available on the server. The server is set up as a drive on the computer network, and files can be copied across the firewall with a standard file manager (left). The directory listing is automatically updated and the remote user has now the possibility of approaching the new information (right).

Further work

It is important to note that the above example is far from the only way, and probably not the most convenient way of exchanging information. Different approaches are needed for different purposes. It may for instance be of interest to automatically transfer data of a certain kind, and then a solution with a direct use of the File Transfer Protocol must be applied instead of going through the World Wide Web. The current development in the field of telecommunication is very rapid, and better information exchange solutions will certainly be introduced in the future. An example is remote access to computer systems, which today is regarded as highly insecure, but which in the future may be regarded as an interesting alternative.

Further work must be based on operable servers in the different countries. It requires that every country both set up a server according to own criteria, and develop tools for approaching the information in the other countries. Proposed topics of interest to study further in addition to an overall testing of the system, are secure transfer of passwords, encryption, and a standardisation of user interfaces and directory tree for discretionary access control. All further work must be in close cooperation with the involved authorities.

Preplanning of early cleanup

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Abstract-A prestudy "Preplanning of early cleanup after fallout of radioactive material", made by Studsvik EcoSafe in Sweden, pointed out the need and request for preplanning of actions of cleanup. The prestudy proposed that a main study should be started to propose test methods, elaborate scenarios, describe relevant cleanup strategies in different environments and to prepare a checklist. Based on the prestudy and on discussions this project was started, with the goal as stated below. Because of the common interest between the Nordic countries NKS is the organisation responsible for the project.

The result of the project is intended to be a document pointing out what can be planned in advance regarding early cleanup actions after a nuclear plant accident. Early shall in this case mean the three first weeks after an accident. The project shall only deal with questions concerning external radiation.

The responsibility for these actions is differently organised in the Nordic countries and the document shall be usable to *persons in charge of planning* or *decision makers* on the appropriate level of organisation for each country. The document shall principally be aimed towards persons without professional competence in the field of radiology.

Results will be presented for a limited number of generalised environments and fallout situations
- urban/ suburban/ rural
- regional differences (in for example house types and constructing material)
- dry or wet deposition

For the time being work is going on in a project group with participants from Denmark, Finland, Norway and Sweden. There is yet no results to present.

Introduction

In Sweden the county boards (länsstyrelserna) are the authorities in charge for cleanup and other measures to be taken in the event of release and fallout of radioactive substances with origin in nuclear facilities. This is according to the Swedish law on rescue service. The same law requests the county boards to make up plans for cleanup actions.

In the other Nordic countries the responsibility for these actions lies on the national level, but there still is a person or a group of persons that have to or should do plans for early cleanup actions.

So whatever the organisation looks like, in our countries there are persons who would be helped to have guidelines and a checklist when they sit down to make plans for something that yet has not happened.

The prestudy "Preplanning of early cleanup after fallout of radioactive material", made by Studsvik EcoSafe, proposed that a main study should be started to propose test methods, elaborate scenarios, describe relevant cleanup strategies in different environments and to prepare a checklist. Based on the prestudy and on discussions this project was started. Because of the common interest between the Nordic countries, NKS is the organisation responsible for the project.

As the work is going on just now there are yet no results to present.

Purpose and limitations

There is a lot of knowledge and data dealing with different environments, various decontamination methods with their relative effectivness and a number of scenarios. Mostly this is in a form that makes it difficult to use as the needed guideline for preplanning of early cleanup actions. The task of the project is to work out these guidelines.

Which actions do we regard as relevant/necessary to be prepared for? What does these actions mean regarding the planning? Are there actions not justified to work out plans for? (There can be several reasons for this, for example that the action is relevant but need no planning or that the action not should be done).

The outcome of the project is intended to be a document that points out what can be planned in advance regarding early cleanup actions and containing guidelines and a checklist. The document shall have a form that makes it usable to planners and decisionmakers without professional competence in the radiological field.

The project will only consider doses to the public and working personnel due to external radiation and will not deal with actions aimed to reduce uptake and contamination in agricultural products and food. The result will be presented for a limited number of generalised environments.

Planning of the work and work so far

At a meeting in June the project group decided that the work should be concentrated on urban areas. Rural areas is a minor problem as long as you talk about preplanning because people can take actions by themselves after recommendations and have most of the equipment needed. Besides, it is a small part of the population that will be affected.

The group also decided not to use different contamination levels. The actions are neutral to levels, as long as you talk about relative effects of different cleanup actions, but it is necessary to consider the differences in efficiency for wet and dry deposition.

Five generalised environments for peoples living conditions will be used in the calculating of effects of various actions. They are: single family houses of wooden or brick/concrete construction, semidetached houses, large two-storey terrace houses and urban centre multi-storey houses.

'URGENT - a model for prediction of exposure from radiocaesium deposited in urban areas', developed by Risø, will be used for the calculations. For the environments mentioned above, and for the cleanup actions *triple digging; fire hosing of roofs, walls or roads; soil or snow removal; grass cutting; pruning of trees and bushes; ploughing; vacuum-sweeping of roads* the intention is to prepare a data sheet for each case showing the Expected effect (dose rate reduction, accumulated lifetime dose

reduction), Personnel requirements and costs, Equipment / other requirements and costs, Practicability, Waste-amounts and costs, Further remarks.

Together with views from some future users these data then will be edited to form a suitable final report. It is planned to be finished on March 31, 1997.

The project group

Thomas Ulvsand, FOA, Sweden	project leader
Kasper G Andersson, Risø, Denmark	
Jørgen Holst Hansen, Beredskabsstyrelsen, Denmark	
Kari Sinkko, STUK, Finland	
Svein Uhnger, Statens strålevern, Norway	
Jan Preuthun, Statens Jordbruksverk, Sweden	
Gun Svennerstedt, Statens Räddningsverk, Sweden	

Beskrivning av processen med ackreditering av laboratorier inom gammaspektrometri

Þorsteinn V. Jónsson (förk. ÞVJ), Geislavarnir ríkisins, Reykjavík, Island

1. Introduktion

1.1. Allmänt

Denna artikel beskriver ett delprojekt (betecknat med NKS/EKO-3.2.4) som bekostats av *Nordisk kärnsäkerhetsforskning*. I rapporten *Nordisk kernesikkerhedsforskning, Projektplaner for programmet 1994-1997, Rapport för 1994, NKS(94) 7 rev, 1994-10-20,* har projektet EKO-3 rubriken *Beredskabsstrategi og -procedure*. Ett av delprojekten har rubriken *"Kvalitetssikring ved måling, prøvetagning og analyse"*. I NKS rapport *NKS(95)1*, 1995-02-16, har delprojektet utdelats beteckningen NKS/EKO-3.2 och beskrivs bl a med följande ord: *"delprojektet kommer i första hand att koncentrera sig på gammaspektrometri vid laboratorieanalys och i fält."* Delprojektet, som är grunden till arbetet med denna artikel, blir senare ett delprojekt under EKO-3.2 och utdelas rubriken *"Kvalitetssäkring inom gammaspektrometri med ackreditering som mål"* och beteckningen NKS/EKO-3.2.4.

1.2. EU direktiv 93/99

EU direktiv 93/99, 29. oktober 1993 gäller för kontroll av livsmedel. I artikel 9 står (i dansk översättning):

"Medlemsstaterne sætter de nødvendige love og administrative bestemmelser i kraft for at efterkomme
– dette direktiv, bortset fra artikel 3, inden den 1. maj 1995
– artikel 3 inden den 1. november 1998"

I artikel 3 står bl a detta:
"1 Medlemsstaterne træffer alle foranstaltninger til at sikre, at laboratorier, der er omhandlet i artikel 7 i direktiv 89/397/EØF, overholder de generelle kriterier for prøvningslaboratoriers virksomhed, som fastlagt i europæisk standard EN 45 001 ..."

1.3. Målet med projektet

Ackreditering och kvalitetssäkring har stort intresse i beredskapssammanhang. Kvalitet av mätningar berör både laboratorier och mottagare av data. Den berör ej enbart de laboratorier som söker ackreditering. En ökad tyngd läggs på pålitlighet av snabbt utbyte av mätdata mellan land i fall av olyckshändelser. I sådana fall kommer tidsbrist att försvåra uppföljning och tester av mätdata. Det är dock nödvändigt att mottagaren kan lita på de data som han får i samband med olyckshändelser.

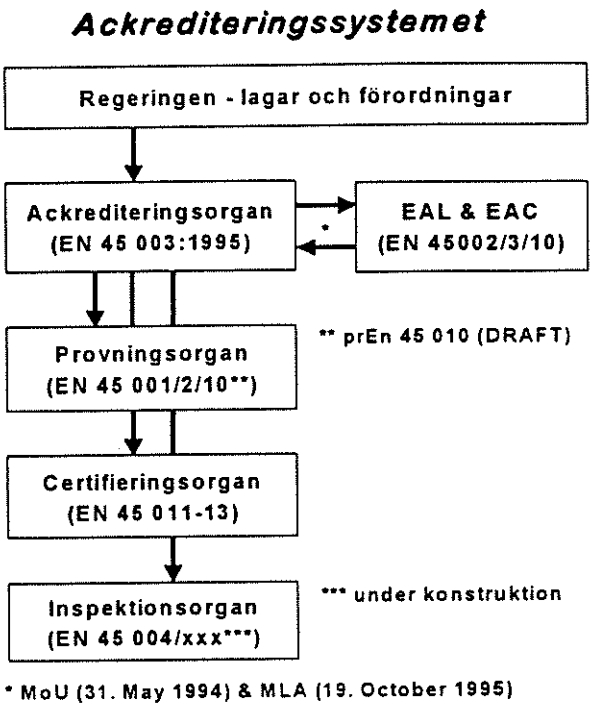
Gammaspektrometri kan ses som relativt enkel mätmetod jämfört med t ex mätning av alfa- eller betastrålande prover. Jämförelsetester avslöjar dock stora olikheter mellan laboratorier, även vid relativt enkla mätningar av f ex Cs-137. Det är därför ett viktigt mål inom beredskap och vid framtida utbyte av mätdata att laboratorier definierar och

om möjligt förbättrar den kvalitet som man vill erbjuda i beredskapssammanhang liksom vid "normal" drift.

2. Resultat

2.1. Allmänt

Resultat kan delas upp i fyra delar, (1) Information om ackrediteringssystemet i Europa, (2) information om ackreditering, (3) prototyper av kvalitetsmanual och teknisk manual och (4) rekommendationer till nordiska strålskyddsmyndigheter. Dessa delar presenteras här separat. Prototyperna av manualerna publiceras separat. De nordiska ackrediteringsorgan samarbetar till en viss mån men att få till stånd samarbete om ackreditering av laboratorier inom strålskydd verkade in vara realistiskt. Ackreditering inom ett land sköts av landets ackrediteringsorgan.



Figur 1 Ackrediteringssystemet

MoU : Memorandum of Understanding
MLA : Multilateral Agreements

Kostnader kan vara höga både när det gäller söknadsavgifter, bedömningsbesök och andra direkta kostnader. Arbetsinsatser av personalen kommer också att kosta tid och pengar, inte minst under fasen med förarbetet då en eller flera personer måste skriva manualerna och initiera nya kvalitetssystem. En person på laboratoriet, kvalitetschefen måste ägna omkring 25-50% av sin arbetstid åt kvalitetssystemet för att bibehålla ackrediteringen.

2.2. Ackrediteringssystemet i Europa

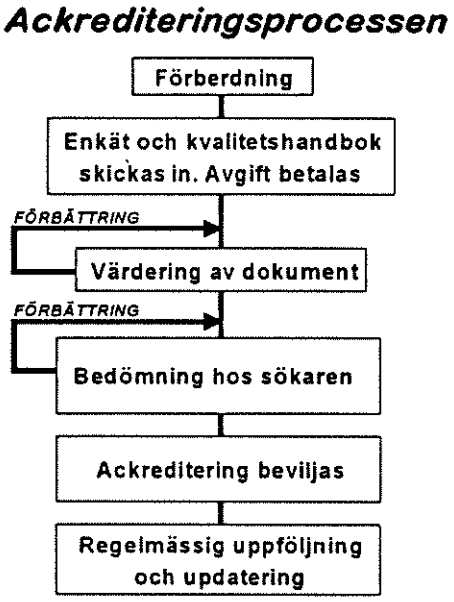
Informationen om hur ackrediteringssystemet i Europa är organiserat kom från ÞVJ's undersökning i juni 1996. I korthet kan man sammanfatta svaren således att ackrediteringsorganen tillsätts av regeringen i varje land och arbetar enligt lagar och förordningar. Samtliga ackrediteringsorgan arbetar under EN 45 000 standarderna och med hänsyn av ISO/IEC Guide 25. De Europeiska landen samarbetar inom området ackreditering inom Föreningen för ackreditering av laboratorier i Europa, EAL (*European cooperation for Accreditation of Laboratories*). Inom EAL har man dragit upp en multilateral överenskommelse (*MLA*). I introduktionen står bl a:

"*The cooperation of the Accreditation Bodies is based on a Memorandum of Understanding (MoU) dated of 31 May 1994. An on-going programme of cooperation has been set up which is aimed to maintain and develop Multilateral Agreements on the equivalence of operation of Calibration and Testing Laboratory Accreditation Schemes administered by EAL Members and on the acceptance of certificates and reports issued by laboratories accredited by them, based on mutual confidence.*"

Överenskommelsen trädde i kraft 30 maj 1995 och texten accepterades 19. oktober samma år. Alla Nordens land utom Island har undertecknat denna. För att tillåtas underteckna dessa avtal måste ackrediteringsorganen ha utvärderats under EN 45002/3/10 standarderna. Utvärderingen görs om vart fjärde år. En schematisk bild av ackrediteringssystemet i Europa visas i figur 1.

2.3. Ackreditering

Ackreditering är ett officiellt erkännande av att f ex ett laboratorium har nödvändig teknisk kompetens att utföra särskilda mätningar och möter kraven i europastandard EN 45 001. Erkännandet ges av ett ackrediteringsorgan som är ett organ som leder och administrerar ett system för ackreditering och ackrediterar laboratorier. En stor del av arbete innebär skrivning av en kvalitetsmanual och en teknisk manual. Ett kvalitetssystem måste införas och interna revisioner av verksamheten måste göras med jämna mellanrum. Ackrediteringsorgan genomgår ca en gång om året protokoll av interna revisioner. De kontrollerar även att registrering har gjorts enligt föreskrifter och att man åtgärdat avvikelser som uppkommit vid de interna revisionerna. Om ändringar ägt rum i laboratoriet bör ackrediteringsorganet informeras. Om mycket omfattande ändringar gjorts inom laboratoriet kan ackrediterings-organet kräva att en ny utvärdering utförs.



Figur 2 Ackrediteringsprocessen

Ackrediteringsprocessen medför detaljerad dokumentation av procedurer som skall genomgås av ackrediteringsorganet. Man kan grovt dela ackrediteringsprocessen i sex delprocesser (Se figur 2.):

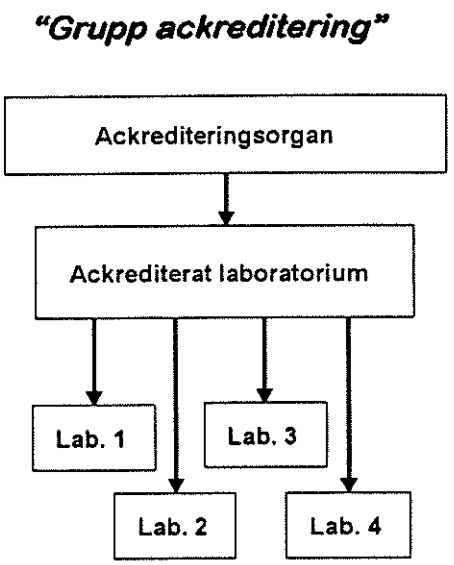
1. Förberedning
2. Inskick av söknadsenkät och kvalitetsmanual. (Avgift betalas)
3. Värdering av dokument hos ackrediteringsorganet
4. Bedömningsbesök hos sökaren
5. Ackreditering beviljas
6. Regelmässig uppföljning och uppdatering

Förberdningsarbetet kan vara arbetskrävande beroende på omfattningen av sökt ackreditering. Nödvändigt är att definiera omfattningen av ackrediteringen som söks, t ex provtagning, grovbearbetning av prover, slutbearbetning av prover och Gammaspektrometrisk analys av färdigbehandlat prov. Det är uppenbart att ackreditering av mätningar av livsmedel innebär i det minsta nr. 3 och 4.

En kvalitetschef bör utses vars uppgift är att ansvara för kvalitetssäkring. Han måste se till att personalen följer beskrivna arbetsrutiner, ta ansvar för interna revisioner av verksamheten och åtgärder för att korrigera ev motsägelser och fel som uppstår.

Kvalitetschefen skall ha direkt tillgång till högsta chefen för laboratoriet. En teknisk chef skall också utses vars ansvar är för den tekniska verksamheten inom laboratoriet.

Enligt europastandarden EN 45001 skall ett ackrediterat laboratorium ha ett fungerande kvalitetssystem och regler om sekretess, säkerhet och underleverantörer. Man uppmuntras delta i jämförelsetester och i samarbete med kunder (om de önskar efter det), ackrediteringsorgan och andra laboratorier om standardisering av mätprocedurer och ökning av kvalitet. Det är noggrant specificerat hur en rapport skal presenteras från ett ackrediterat laboratorium. Kalibrering av mätinstrument måste göras av ett ackrediterat laboratorium eller riksmätplats.



Figur 3 Grupp-ackreditering

På laboratorier använder man sig ofta av hemmagjorda datorprogram f ex vid analys av gammaspektra. Ackrediteringsorganet ser på datorprogram som en del av arbetsrutinerna som utförs. Datorprogram måste därför dokumenteras noga, dvs vilka ekvationer och algoritmer som används så att det är möjligt att verifiera att programmet ger de resultat som påstås av laboratoriet.

Det finns ett alternativ till full ackreditering för mindre laboratorier. De kan teckna kontrakt med ett annat laboratorium som skall gå genom full ackreditering. Det större laboratoriet, moderlaboratoriet, går igenom hela ackrediteringsprocessen och de övriga mindre laboratorierna, satellitlaboratorierna, gör sitt kvalitetsarbete i samarbete med moderlaboratoriet. När ackreditering beviljats får kontrakterade mätningar utförda av satellitlaboratorierna samma status som mätningar utförda av ett fullt ackrediterat laboratorium. Moderlaboratoriet däremot åtar sig ansvar för kvaliteten av satellitlaboratoriernas mätningar. Detta möjliga alternativ till full ackreditering kallas här "*Grupp-Ackreditering*" (Se figur 3).

2.4. Kvalitetsmanual och teknisk manual

Delprojektet resulterade bl a i prototyper av dessa manualer som publiceras separat.

Kvalitetsmanualen skal utförligt beskriva laboratoriets kvalitetssystem. Enligt EN 45001 måste en kvalitetsmanual i det minsta innehålla följande:

1. **Kvalitetspolicy**: Uttalande från laboratoriets högsta ledning om laboratoriets policy anseende kvaliteten i allt arbete som utförs inom ramen för ackreditering.
2. **Laboratoriets organisation**: Laboratoriets juridiska status och organisatoriska ställning i förhållande till resten av företaget samt laboratoriets interna organisation.
3. **Operativa och funktionella skyldigheter och uppgifter som rör kvalitetsfrågor**: Laboratoriet bör ha policy avseende personalrekrytering

och krav på personal. I detta sammanhang kan ingå krav om utbildning och introduktion av nyanställda samt vidareutbildning.

4. **Allmänna kvalitetssäkringsrutiner**: Utförligt dokumenterade rutiner.
5. **Rutiner för återinföring och korrigering i de fall man upptäcker motsägelser i resultat**: En person eller flera personer bör tillses för underhåll av interna revisioner och kvalitetssystemgenomgångar. Om dessa avslöjar motsägelser i resultat bör finnas dokumenterad beskrivning av åtgärder med syfte att korrigera motsägelserna.
6. **Hantering av klagomål**: Policy, ansvar, dokumentation och korrigerande åtgärder. I vissa fall policy angående ev skadestånd.

I tillämpliga fall måste även följande inkluderas i kvalitetsmanualen:

- Hänvisning till särskilda kvalitetssäkringsrutiner för varje provning
- Hänvisning till kvalifikationsprovning, användande av referensmaterial etc

Vid skrivning av kvalitetsmanual för ackreditering kan man dra nytta av ISO standard 10013 (*Guidelines for developing quality manuals*). Vid strukturering av kvalitetssystem kan ISO standard 9001 (*Quality systems – ...*) användas som referens. Det är annars fritt för laboratoriet att bestämma hur manualerna skall se ut såvida att de är konsekventa, klart skrivna och med utförliga innehållsregister så att de anställda kan utan besvär använda dem. Den tekniska manualen skall innehålla utförligt dokumenterade beskrivningar av arbetsrutiner, instrument, lokaler osv.

2.5. Diskussion

Det är möjligt öka kvaliteten utan ackreditering till en nivå jämförbar med ett ackrediterat laboratorium men det är omöjligt att jämföra kvalitet av två laboratorier utan en opartisk bedömning av en tredje part. För närvarande finns det inget bättre sätt att ge laboratorier, som mäter radioaktivitet (och biprodukter) i livsmedel, en kvalitetsstatus utan ackreditering. Det är dock klart att ackreditering kommer att vidmakthålla laboratoriets goda kvalitet och även öka den. Man bör ha i tanke att laboratorierna själva kommer att ha störst nytta av ackrediteringen och därför är det ej riktigt att betrakta den som ett onödig, byråkratiskt krav från EU som måste uppfyllas till varje pris. Att ackreditering skulle förhindra framtida utveckling inom laboratorier är fel. Det är inget i själva ackrediteringen som förhindrar att man ändrar arbetsrutiner och/eller mätprocedurer. Kravet från ackrediteringsorganet är att dessa ändringar är väl grundade och att man informeras innan de träder i kraft.

2.6. Rekommendationer till nordiska strålskyddsmyndigheter

Följande rekommendationer till nordiska strålskyddsmyndigheter är baserade på arbete med EKO-3.2.4 delprojektet:

På kort sikt (dvs inom 1½ år)

1. Söka ackreditering för rutinmässiga mätningar utförda av myndighetens laboratorier och de mätningar som kommer att göras i ev beredskapssituationer. Man bör sikta på ackreditering före 1. november 1998 enligt EU direktiv 93/99, artikel 9.
2. Teckna kontrakt med ackrediterade laboratorier eller laboratorier som söker ackreditering.

3. Uppmuntra redan kontrakterade laboratorier att söka ackreditering för den verksamhet som rör myndigheten.

På längre sikt (ca 1½-3 år)

1. Diskutera möjligheten att öka ackrediteringens omfattningen av myndighetens laboratorier så att denna även täcker mätningar inom forskning
2. Teckna endast kontrakt med ackrediterade laboratorier
3. Säg upp kontrakt med icke ackrediterade laboratorier

Det framtida målet för nordiska strålskyddsmyndigheter borde vara följande:
I fall av beredskapssituationer baserar myndigheter sina beslut och rekommendationer på bl a mätningar av radioaktivitet i livsmedel, gjorda av laboratorier som ackrediterats för sådana mätningar.

När man uppnått detta mål kan man konstatera följande:
I beredskapssituationer baserar de nordiska strålskyddsmyndigheterna sina beslut och rekommendationer på mätningar utförda i laboratorier ackrediterade för sådana mätningar. Detta garanterar en enhetlig kvalitetsnivå av de mätdata nordiska strålskyddsmyndigheter baserar sina beslut på.

3. Referenser

- EU-direktiv 89/397, 14. juni 1989
- EU direktiv 93/99, 29. oktober 1993
- EAL, *Memorandum of Understanding*, 31 May 1994
- EAL, *Multilateral Agreement*, text approved on 19 October 1995
- EN 45 001, *General criteria for the operation of testing laboratories*, September 1989
- EN 45 003, *Calibration and testing laboratory accreditation systems – General requirements for operation and recognition*, March 1995
- prEN 45 010, *General requirements for assessment and accreditation of certification/registration bodies*, DRAFT, January 1995
- ISO/IEC Guide 25, *General requirements for the copmpetence of calibration and testing laboratories*, Third edition, 1990
- ISO/IEC Guide 61, *General requirements for assessment and accreditation of certification/registration bodies*, First edition, 1996
- EN ISO 9001, *Quality systems – Model for quality assurance in design, development, production, installation and servicing*, July 1994
- ISO 10013, *Guidelines for developing quality manuals*, September 1995
- SWEDAC, DOC 92:5, datum 1992-08-21
- Norsk Akkreditering, NA Dok. 4: "*Vejledning ved søknad om akkreditering*"
- *Kvalitetshandbok för auktoriserade laboratorier*, Revision 3.0, Föreningen Auktoriserade laboratorier (FAL), Box 11 333, 111 81, Stockholm.

An Intercomparison Exercise on Radionuclides in Sediment Samples

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Abstract An intercomparison exercise on artificial and natural radionuclides in two sediment samples, one from the Baltic Sea and one from the Kattegat, has been carried out under the EKO-1 project of the Nordic Safety Research Programme (NKS) for the period 1996-97. The measurement techniques have included direct gamma-ray spectrometry with Ge and NaI detectors, and radiochemical procedures followed by beta counting and alpha spectrometry. The participants have comprised 21 laboratories. Results were submitted for Cs-137, Cs-134, Co-60, Sb-125, Pu-239,240, Pu-238, Am-241, Sr-90, Ra-226, Th-232 and K-40, Pb-210, Po-210 and U-235. The analytical performance of the participants was evaluated for those radionuclides where six or more data sets were received. Statistical tests were made to see if individual results agreed with overall average radionuclide concentrations in the two sediment materials within target standard deviations. The results of these tests show that for Cs-137, Cs-134, Ra-226, Th-232 and K-40 the analytical performance criteria were not met for 20-40% of the data sets. For plutonium isotopes the tests show that the performance criteria were not met for 13% of the data sets. Tests of overall analytical performance show that 61% of the data sets do not meet the combined performance criteria. This shows that there is room for considerable improvement of analytical quality for most of the laboratories that have participated in this intercomparison. The intercomparison exercise has furthermore demonstrated several elementary problems in analytical work such as interchange of samples, mistakes in calculations and correction of background levels.

INTRODUCTION

Under the EKO-1 Project on marine radioecology which is organised by the Nordic Nuclear Safety Research Programme (NKS) for the period 1994-97 an intercomparison exercise on radionuclides in sediments was initiated in 1995 by Risø National Laboratory. A total of 21 laboratories have participated in the present exercise: 14 from the Nordic countries, 6 from the Baltic republics, and one from Japan.

MATERIAL AND METHODS

Sediment samples

Two sediment samples were obtained: one from the Baltic Sea and one from the Kattegat. The Baltic Sea sediment sample was obtained from the German Federal Maritime and Hydrographic Agency (BSH) who collected the sample in the western Baltic Sea. The Kattegat sample was obtained from the Danish National Environmental Research Institute who collected the sample in the central Kattegat. Both samples were homogenised and tested for homogeneity by gamma-spectrometric measurements on 9 sub-samples. Homogeneity of the sediment samples was demonstrated for Cs-137 and natural radionuclides to within a few percent.

The sample material was distributed in June 1995 by sending 100 g Baltic Sea sediment and 200 g Kattegat sediment to the participating laboratories. Some laboratories shared the same samples. One participant received a Kattegat sediment sample only. During the intercomparison two of the participants submitted an extra set of results, so the number of results received per radionuclide varies from 1 to 23. For anonymity the data sets are identified by numbers.

Statistics

The data were evaluated by comparing the laboratory mean results of radionuclide concentrations in the sediment samples to the corresponding overall mean concentrations. The procedures were adopted from those used in a recent proficiency testing exercise on radionuclides in milk powder carried out by the Central Science Laboratory (CSL 1996). The overall mean concentrations were determined for those radionu-

clides where six or more sets of data were received and after exclusion of outliers determined on a subjective basis. No evaluation was carried out for radionuclides where less than six sets of data were received.

For each laboratory mean result a z-score was calculated according to

$$z = \frac{x - \bar{x}_{om}}{\sigma_t},$$

where x is the laboratory mean value, $\bar{x}_{om}$ the overall mean value and σ_t a target value for the standard deviation around the overall mean value for the analysis in question. The target values were chosen to be 5%, 10%, 15% or 20% depending on the radionuclide and represented uncertainties typical of those reported by the participants. The z-score may be used as a test-value in the normal distribution such that values between -2.0 and 2.0 would be acceptable at the 5% level.

Values of z-scores may be combined into a rescaled sum of z-scores, a RSZ-value which is calculated according to

$$RSZ = \frac{\sum z}{\sqrt{n}},$$

where n is the number of z-scores being combined. The RSZ-value may be used for a test in the normal distribution. The value is an indicator of analytical bias.

Values of z-scores may furthermore be combined into a sum of squares of z-scores, SSZ, which is calculated according to

$$SSZ = \sum_{i=1}^{n} z_i^{\,2}.$$

This value may be used for a test in the χ^2-distribution with n degrees of freedom. The SSZ-value is an indicator of analytical accuracy.

Two-sided tests have been made for the normal distribution and the χ^2-distribution. The following levels and symbols of statistical significance have been used: probably significant (P>95%, *), significant (P>99%, **) and highly significant (P>99.9%, ***).

The scores were combined across the two sediment samples for each set of results to allow an evaluation of the laboratory performances for each radionuclide to be made. Additionally, the scores were combined across all radionuclides to permit an evaluation of the overall analytical performance for each participant.

It is important to note the elements of subjectivity in the statistical tests. The overall mean values are in several cases determined after elimination of subjectively identified outliers, and the target values of the standard deviations (σ_t) around the overall mean values are chosen on a subjective basis.

RESULTS

Results were received on a wide range of radionuclides. The artificial radionuclides comprise Cs-137, Cs-134, Co-60, Sb-125, Pu-239,240, Pu-238, Am-241 and Sr-90. The natural radionuclides comprise Ra-226, Th-232, K-40, Pb-210, Po-210 and U-235. This paper presents results for some of the radionuclide and sediment samples where six or more data sets were received. The graphs show the sorted results and include an indication of the overall mean value and the 95% confidence interval for the data on which the overall mean is based. The results are shown with error bars that represent either standard errors of the mean in case of multiple determinations or reported uncertainties of single values. Furthermore, an evaluation of the laboratory scores is shown. The results are reported in detail in Nielsen (1996).

Cs-137

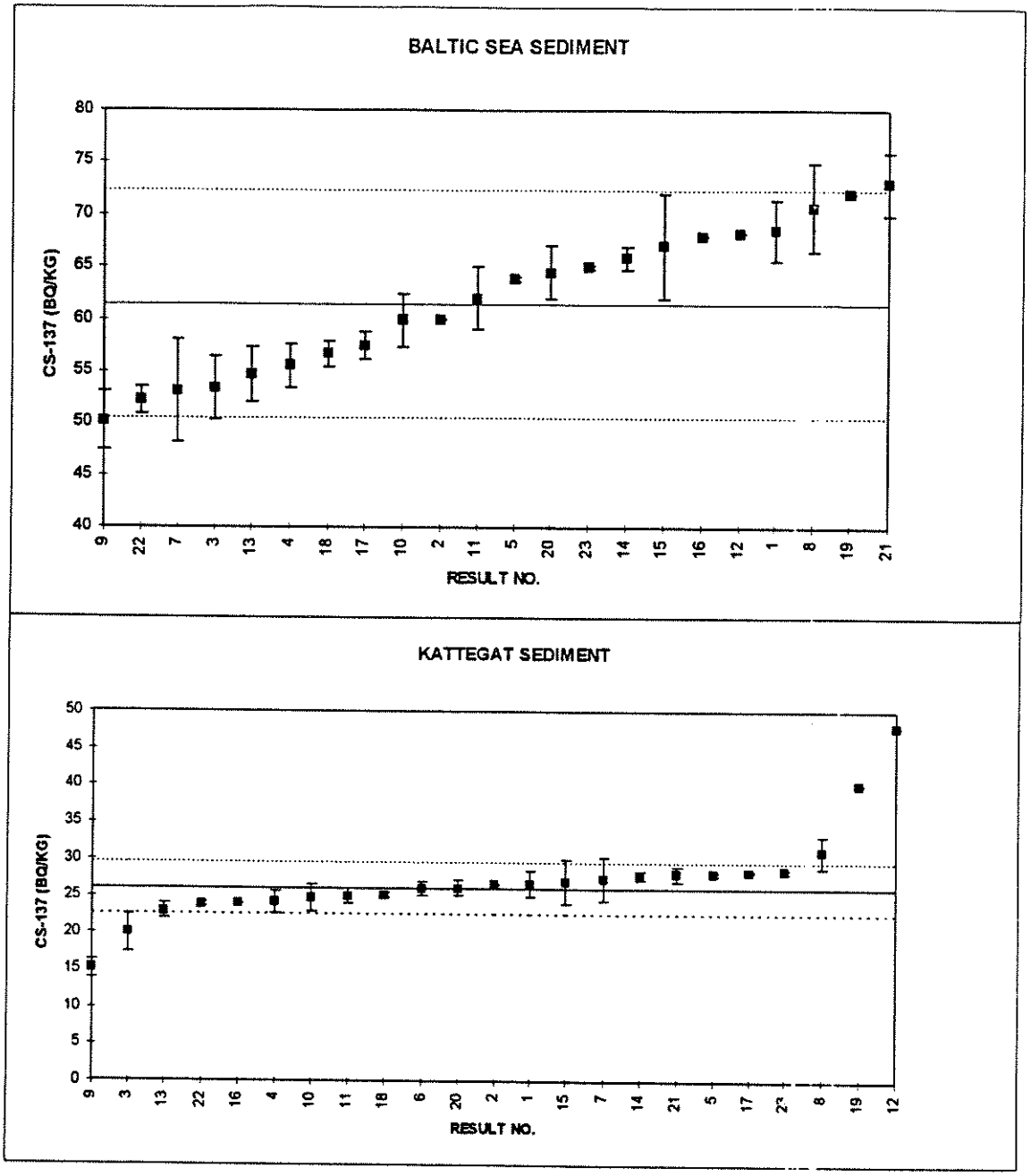


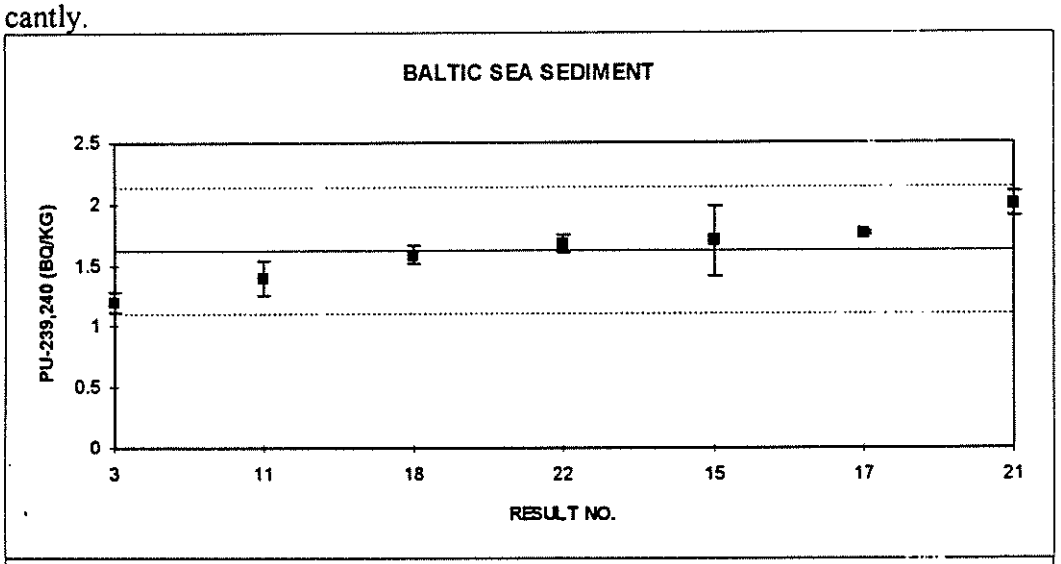
Table 1. Summary of combined z-scores for Cs-137 results showing rescaled sum of scores (RSZ), sum of squares of z-scores (SSZ) and the corresponding levels of statistical significance.

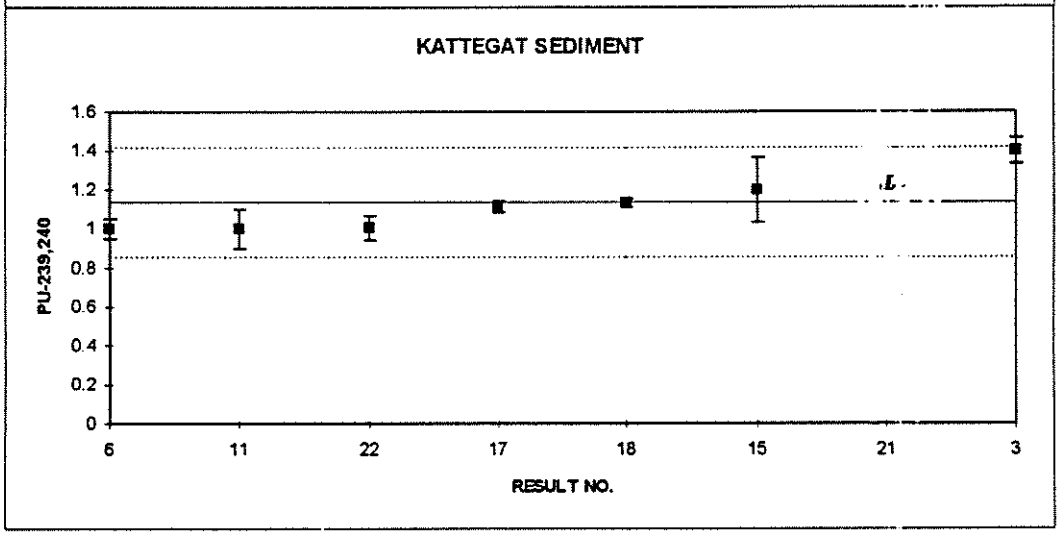
Res. No.	RSZ	Sign	SSZ	Sign	Res. No.	RSZ	Sign	SSZ	Sign
1	1.9	ns	5.6	ns	13	-3.3	**	10.7	**
2	-0.1	ns	0.3	ns	14	1.9	ns	3.8	ns
3	-5.1	***	28.7	***	15	1.8	ns	3.8	ns
4	-2.4	**	5.8	ns	16	0.4	ns	7.3	ns
5	1.6	ns	2.7	ns	17	0.3	ns	4.5	ns
6	-0.1	ns	0.0	ns	18	-1.7	ns	3.0	ns
7	-1.2	ns	8.2	*	19	10.0	***	125.1	***
8	4.8	***	23.2	***	20	0.7	ns	1.0	ns
9	-8.5	***	84.4	***	21	3.7	***	16.4	***
10	-1.1	ns	1.4	ns	22	-3.3	***	11.8	**
11	-0.5	ns	0.8	ns	23	2.1	*	4.4	ns
12	13.4	***	285.9	***					

The statistical tests in Table 1 show that 12 of the 23 data sets received on Cs-137 agree within 5% of the two overall mean values and that 9 of the data sets disagree significantly.

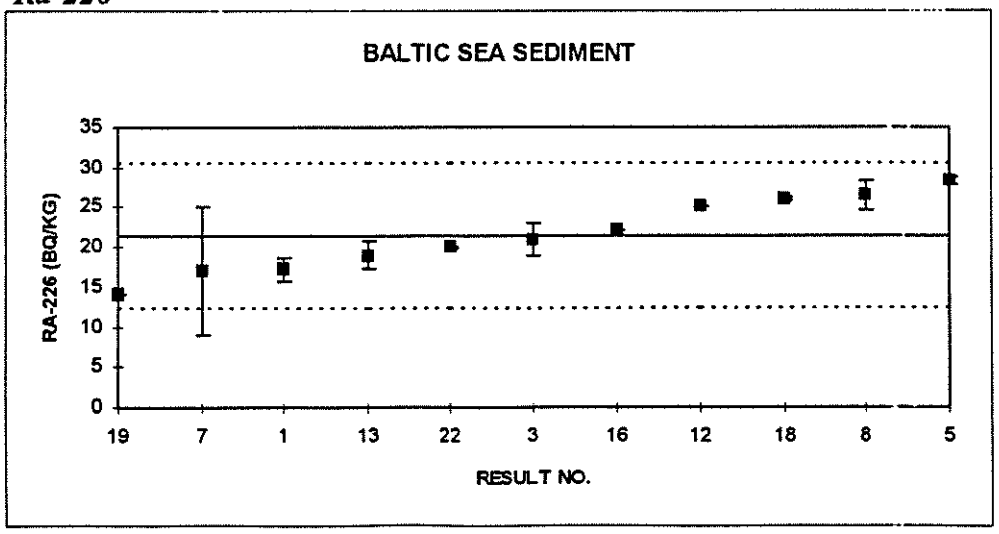
Pu-239,240

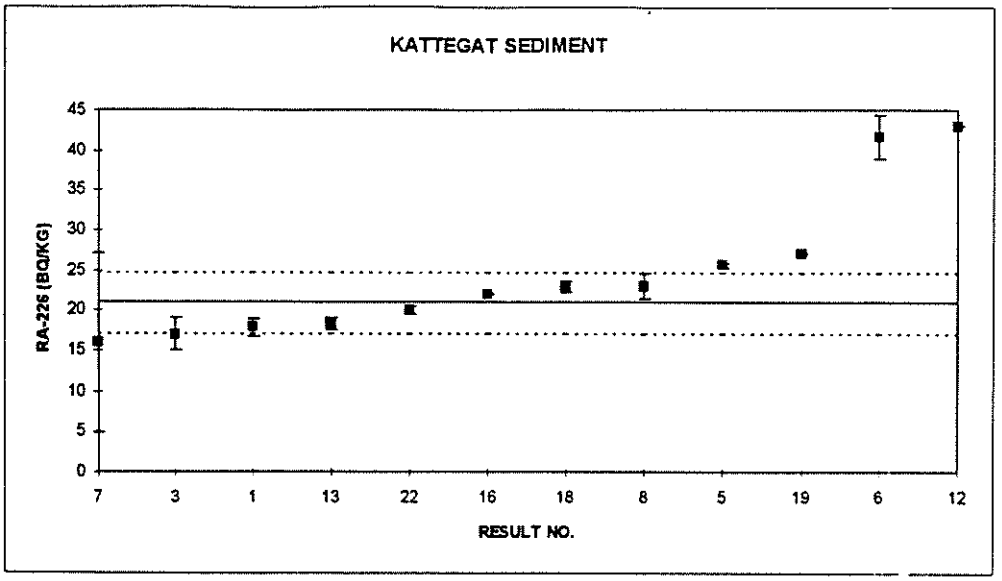
The statistical tests show that 5 of the 8 data sets received on Pu-239,240 agree within 10% of the two overall mean values and that one of the data sets disagrees significantly.





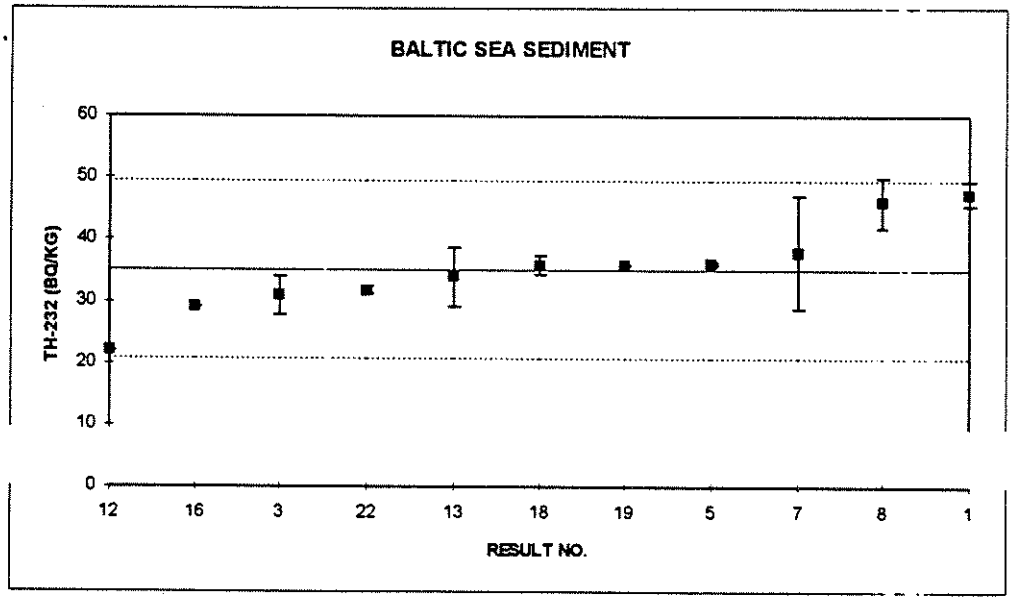
Ra-226

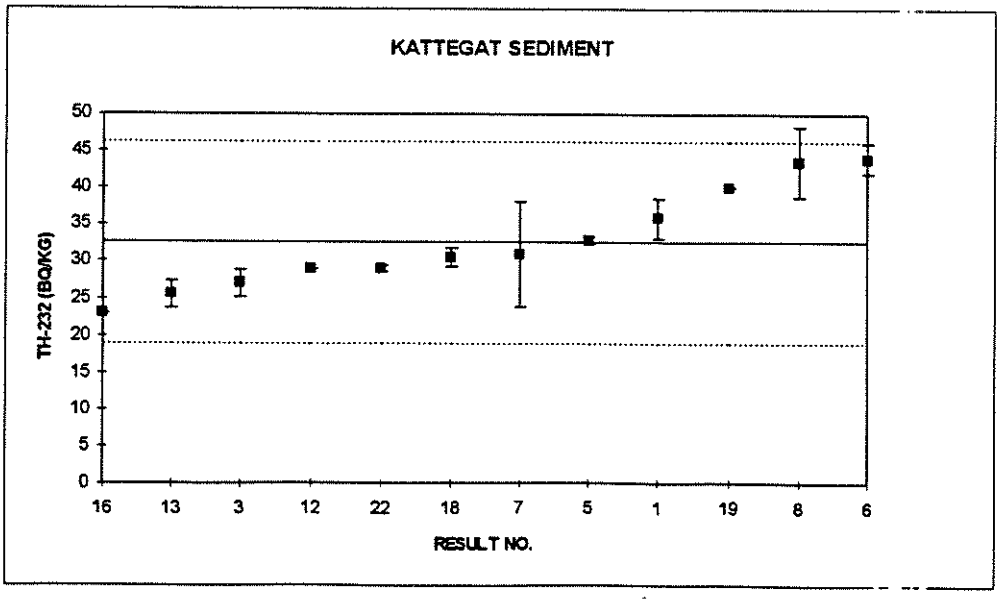




The statistical tests show that 3 of the 12 data sets received on Ra-226 agree within 10% of the two overall mean values and that 5 of the data sets disagree significantly.

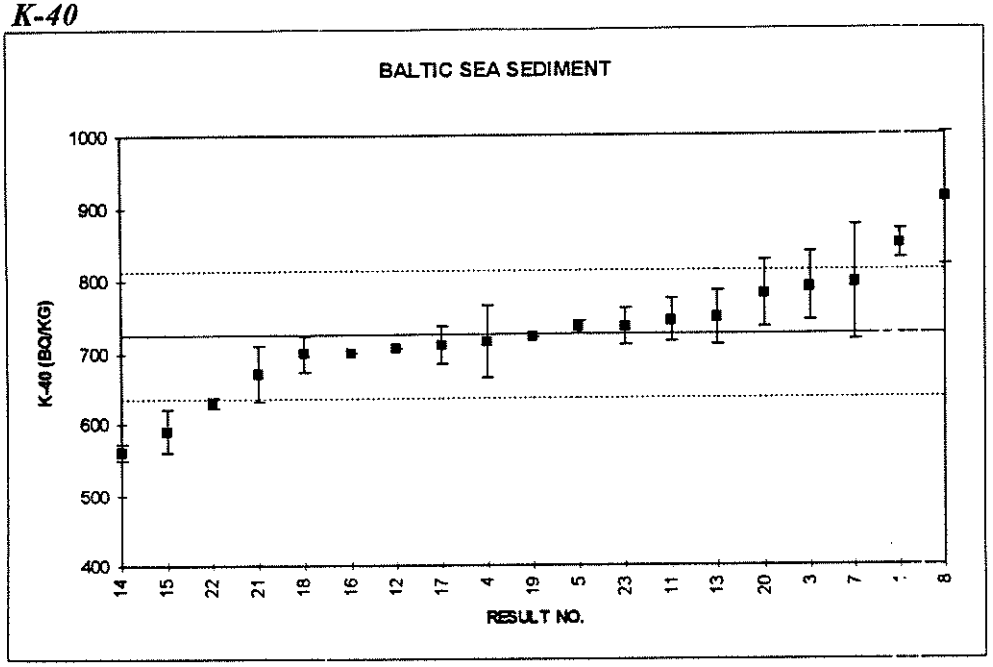
Th-232

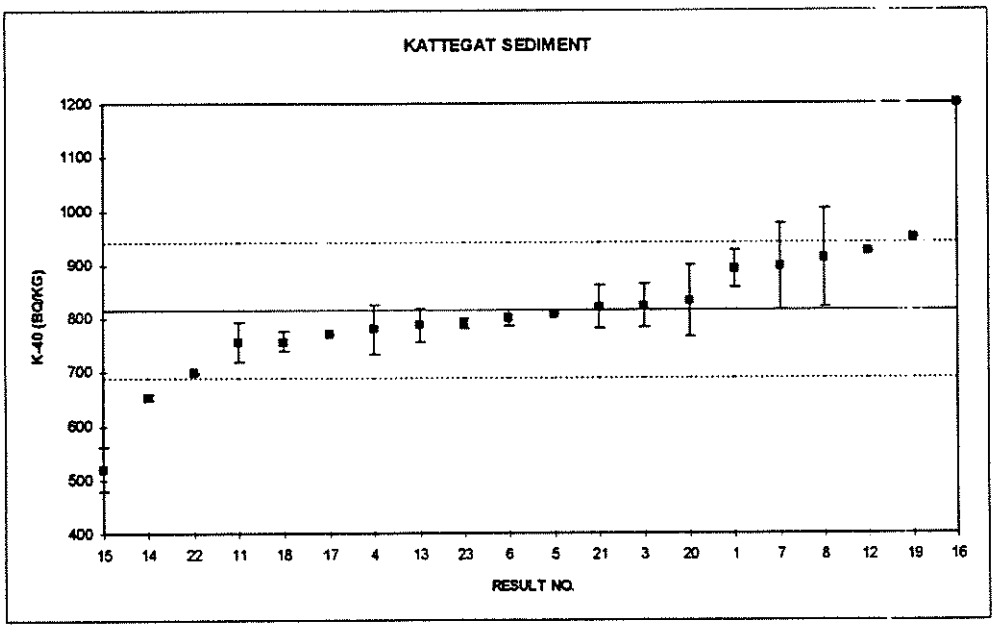




The statistical tests show that 6 of the 12 data sets received on Th-232 agree within 10% of the two overall mean values and that 5 of the data sets disagree significantly.

K-40





The statistical tests show that 16 of the 20 data sets received on K-40 agree within 10% of the two overall mean values and that 4 of the data sets disagree significantly.

Radionuclides combined

The overall analytical performance for Cs-137, Cs-134, Co-60, Pu-239,240, Pu-238, Ra-226, Th-232 and K-40 of each participant may be evaluated from the data in Table 2. The statistical tests show that 4 of the 23 data sets agree within target standard deviations of the overall mean values and that 14 of the data sets disagree significantly. It should be noted, however, that the number of results submitted by the participants vary considerably, from 2 to 14.

Table 2. Summary of combined z-scores for Cs-137, Cs-134, Co-60, Pu-239,240, Pu-238, Ra-226, Th-232 and K-40 showing rescaled sum of scores (RSZ), sum of squares of z-scores (SSZ) and the corresponding levels of statistical significance. The number of z-scores combined is given as n.

Res. No.	n	RSZ	Sign.	SSZ	Sign.	Res. No.	n	RSZ	Sign.	SSZ	Sign.
1	8	2.3	*	28.8	***	13	9	-2.4	*	25.1	**
2	2	-0.1	ns	0.3	ns	14	4	-0.8	ns	13.0	*
3	11	-3.2	**	50.2	***	15	8	25.8	***	3044.7	***
4	4	-2.0	*	6.0	ns	16	8	0.3	ns	41.8	***
5	9	2.9	**	18.5	ns	17	10	0.8	ns	8.8	ns
6	6	4.7	***	111.0	***	18	13	0.2	ns	16.0	ns
7	8	-1.4	ns	21.0	*	19	8	6.2	***	153.3	***
8	9	6.5	***	59.0	***	20	5	13.7	***	821.6	***
9	2	-8.5	***	84.4	***	21	7	3.2	**	23.9	**
10	2	-1.1	ns	1.4	ns	22	14	-3.2	**	26.1	*
11	8	-2.3	*	9.1	ns	23	6	-0.6	ns	15.0	*
12	8	9.7	***	416.0	***						

CONCLUSIONS

An intercomparison exercise has been carried out under the EKO-1 project of the Nordic Safety Research Programme (NKS) for the period 1996-97. The exercise has dealt with the analysis of artificial and natural radionuclides in two sediment samples, one from the Baltic Sea and one from the Kattegat. The measurement techniques have included direct gamma-ray spectrometry with Ge and NaI detectors, and radiochemical procedures followed by beta counting and alpha spectrometry. The participants have comprised 21 laboratories of which 14 are from the Nordic countries, 6 from the Baltic republics and one from Japan.

Results were submitted for a large number of radionuclides comprising Cs-137, Cs-134, Co-60, Sb-125, Pu-239,240, Pu-238, Am-241, Sr-90, Ra-226, Th-232 and K-40, Pb-210, Po-210 and U-235. The analytical performance of the participants has been evaluated for those radionuclides where six or more data sets were received. Statistical tests were made to see if individual results agreed with overall average radionuclide concentrations in the two sediment materials within target standard deviations. The results of these tests are summarised in Table 3 which shows the radionuclides, the selected target standard deviations, the number of data sets submitted and the number of data sets which disagree significantly with the overall mean values.

It is noteworthy that with the exception of Co-60 the performance criteria are not met in about 20-40% of the data sets for the relatively straight forward gamma-spectrometric analyses of Cs-137, Cs-134, Ra-226, Th-232 and K-40. For the more complicated radiochemical analyses of plutonium isotopes only 13% of the data sets do not meet the performance criteria. The test of the overall analytical performance shows that 61% of the data sets do not meet the combined performance criteria. This shows that there is room for considerable improvement of analytical quality for most of the laboratories that have participated in this intercomparison.

The intercomparison exercise has furthermore demonstrated several elementary problems in analytical work such as interchange of samples, mistakes in calculations and correction of background levels. High-quality analytical performance requires careful and dedicated staff using well-established laboratory procedures and frequent participation in international intercomparison exercises.

Table 3. Summary of analytical performance across all data sets.

Nuclide	Target standard deviation (%)	Number of data sets	Number of data sets that disagree significantly with performance criteria
Cs-137	5	23	9 (39%)
Cs-134	20	6	2 (33%)
Co-60	15	6	0 (0%)
Pu-239,240	10	8	1 (13%)
Pu-238	20	8	1 (13%)
Ra-226	10	12	5 (42%)
Th-232	10	12	5 (42%)
K-40	10	20	4 (20%)
Nuclides combined		23	14 (61%)

ACKNOWLEDGEMENTS

This work has been supported by the EKO-1 project on marine radioecology under the Nordic Nuclear Safety Research Programme (NKS) 1994-97. Thanks are due to Jürgen Herrmann from the German Federal Maritime and Hydrographic Agency for providing the Baltic Sea sediment material and to Gunni Ærtebjerg from the Danish National Environmental Research Institute for providing the Kattegat sediment material.

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Nielsen, S.P. (1996). *An Intercomparison Exercise on Radionuclides in Sediment Samples.* Risø-R-914(EN), Risø National Laboratory.

RADIOACTIVITY FROM THE NORTH POLE TO THE ANTARCTIC

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Abstract.The major scientific expeditions to the Antarctic and the Arctic, such as the Swedish Arctic Expedition (YMER-80), Swedish Antarctic Research Expedition (SWEDARP 88/89), Swedish Arctic Expedition (SWEDARCTIC-91 actually a Swedish-German-USA collaboration), the Swedish-Russian Tundra Expedition (TUNDRA-94) covered areas in the North and South Atlantic from 90° N to 72° S and in the Arctic Ocean from 15°W to 160°E.

During these expeditions, in particular, surface water was collected at sampling stations and during steaming from Sweden to the Antarctic and the Arctic and analysed for radiocaesium and plutonium. In combination with data published from other major expeditions such as for example the GEOSECS expedition 72/73 the distribution of radionuclides in time over 22 years and space can be assessed.

Data show that radiocaesium from nuclear detonation tests carried out in the late 1950's and the early 1960's shows a long residence time in surface waters in the North and South Atlantic. The half-life, corrected for physical decay is in the order of 100 years while plutonium has a half-life of 7-8 years.

Results from the central Arctic ocean and continental shelf along the European-Siberian tundra show that radiocaesium from European reprocessing plants is transported east wards along the coast to the Laptev Sea and then advected into the central Arctic ocean with the so called Trans Polar Drift. It has beeen shown from previous expeditions that the origin of plutonium in the Arctic Sea is mainly from nuclear tests, only 5-10% for $^{239+240}$Pu but 20-40% for ^{238}Pu and ^{241}Pu. The work also comprises transfer of radiocaesium in the Antarctic and Arctic food chains. Concentration factors are generally higher in the Antarctic than the Arctic.

INTRODUCTION

On a global bases nuclear test fallout constitutes still the most important source for contamination of the oceans with respect to radiocaesium and plutonium although from radiological point of view natural radionuclides such as ^{210}Po is more important. Locally other sources such as releases from nuclear reprocessing plants and the Chernobyl accident are more significant. Most of this activity is present in the oceans may be with the exemption for the Chernobyl accident.

Radionuclides are subject to a number of processes influencing their concentration in sea water including horizontal and vertical advection of large water masses, turbulent diffusion i sediments, reversible/irreversible scavenging by suspended particles etc. The knowledge of source terms and the physical decay of radionuclides provide an excellent tool for tracerstudies in time and space. Furthermore long-term studies of low activities in the oceans are necessary for predictions and dose assessment following eventual major accidents/releases. This paper describes the general latitudinal global of plutonium and radiocaesium in the South and North Atlantic from the North Pole to the Antarctic and the longitudinal distribution along the European -Siberian coast following releases from major sources

MATERIALS AND METHODS

Results are available from several important scientific expeditions such as YMER-80 to the Barents and Greenland seas, SWEDARP 1988/89 to the Antarctic Peninsula area, SWEDARCTIC 1991 to the North Pole, and the Swedish-Russian expedition 1994 along the coast of the European-Russian Tundra. The areas covered by these expeditions are shown in Fig. 1-4.

At the previous conference, 1991 at Norwich (1) results from the expeditions performed before 1991 were put in the perspective of the GEOSECS expedition 1972/73 to the North and South Atlantic and the Polish expedition 1977/78 to the Antarctic Peninsula area. The results from these expeditions showed an astonishing agreement of the latitudinal distribution of plutonium and ^{137}Cs after correction for physical decay.

During the expeditions surface water was generally obtained by direct continuos supply to a laboratory container. Water profiles were either collected by large volume bottles or in shallow waters, such as during the Tundra expedition, pumped from a hose submersed to the desired depth. Radiocaesium was precipitated on copper-ferrocyanide from 200 l water samples using ^{134}Cs as radiochemical yield determinant. The samples were analysed by HPGe gamma spectrometry for 2 days in 60 or 180 ml calibrated geometry. Actinides were preconcentrated on board from 200 l samples by precipitation with sodium hydroxide, at pH 9-10, after acidification, reduction with sodiumdisulfite and equilibration of the radiochemical yield determinant ^{242}Pu. For the assessment of the $^{134}Cs/^{137}Cs$ activity ratio large volumes, 1000 l, or more, were pumped through cotton wound cartridge filters impregnated with copperferrocyanide. Sediment samples were collected with a HAPS- or boxcorer during the Tundra expedition or regarding the central Arctic ocean collected by box corers or muliticorer devices and obtained from other research groups participating in expeditions not mentioned above. Plutonium was separated by radiochemical methods described previously (2) and analysed for 7-10 days by solid state alpha spectrometry.

RESULTS AND DISCUSSION

Radiocaesium.

Latitudinal distribution.

Caesium-137 is present today in various concentrations in surface sea water, from a few tenth of Bq m^{-3} such as in the Antarctic area to 100 Bq m^{-3} such as in the Baltic sea following the Chernobyl accident or even higher locally close to nuclear facilities. Levels of ^{134}Cs are in the order of 3% of those for ^{137}Cs if water masses are mainly contaminated from the Chernobyl accident (1986) and in the order of 0.3% if contaminated through the major period of releases (1974-1979) from European reprocessing facilities (mainly Sellafield).

Fig. 5 shows the latitudinal distribution of ^{137}Cs in surface water from the Antarctic Peninsula area to the North Pole. These data were obtained through the Swedish Expedition to the Antarctic 1988/89(SWEDARP 88/89) and the Swedish expedition to the Arctic and the North Pole 1991(SWEDARCTIC 1991). Activity concentrations are not decay corrected (about 5% in 2.5 years) since the two expeditions cover areas mainly contaminated from different sources during different periods.

Below 30 °S concentrations are low and in the order of 0.5 Bq m^{-3}. Significantly higher concentrations were found at 45 °N, which is difficult to explain but also ^{60}Co was detected in these samples. From 30°S to 25°N activity concentrations are significantly higher; around 2 Bq m^{-3}. With a lot of good will one can say that there is an increase in activity concentrations, to 2.5 Bq m^{-3}, between 30°S and 20°S which someone could interpret as a result of nuclear tests in the Pacific at these latitudes, but many other distribution processes might play a role. The effective ecological half life of radiocaesium in the North and South Atlantic was estimated to be as long as 100 years since the surface water is exchanged with water which is also contaminated.

North of 20°N activity concentrations are about 3 Bq m^{-3} if they are not influenced from European reprocessing facilities or the Chernobyl accident. Above 45 °N concentrations are very variable in the North and Norwegian Seas and activity concentrations vary with distance to sources (European reprocessing facilities) and distance from the coast. It has been shown that the activity is transported by the Gulf Stream and there is a very sharp gradient in activity concentrations of radiocaesium with distance from the Norwegian coast due to the coriolis force(3).

We have been collecting surface sea water samples during steaming along the Norwegian coast to and from the Arctic during the expeditions and data are shown in Fig. 6. In 1980 activity concentrations along the Norwegian coast reflect the releases from the Sellafield nuclear fuel reprocessing plant where the releases were the highest during 1974-1979 (annual discharges 3000-5000 TBq). The transit time from Sellafield to west Norway has been estimated to 3-4 years and the travelling time along the coast of Norway about 1 year(3,4). This resulted in activity concentrations of about 100 - 200 Bq m^{-3} or about twice as high as those in the Baltic Sea following the Chernobyl accident. The activity concentration decreased very little along the coast, less than a factor 2(5).

Further north the ^{137}Cs concentration C(x) varied with distance(x) from 70 °N for locations east of Svalbard according to the equation

C(x) = 141.exp(-000122 x)
(half distance 568 km)
and west of Svalbard according to.
C(x) = 148.exp(-0.00164 x)
(half distance 423 km)

This means that more radiocaesium went to the east than to the west of Svalbard.

As a result of the decrease in releases from the European reprocessing facilities activity concentrations along the Norwegian coast were only 10-30 Bq m^{-3} in 1991. The decrease would have been even more significant without the Chernobyl accident. There is an annual outflow of ^{137}Cs from the Baltic Sea to the North sea of about 50-60 TBq. The total nuclear test fallout of ^{137}Cs in the North Atlantic is estimated to 76 000 TBq and the releases from European reprocessing facilities (decay-corrected) 29 000 TBq(4). These sources are much smaller today. Fallout from nuclear test is almost negligible and releases from reprocessing plants have been reduced in such a way that outflow from the Baltic Sea today constitutes the most significant source for radiocaesium along the Norwegian coast and further into the Arctic.

It can be observed that activity concentrations of ^{137}Cs were higher in the central Arctic ocean than in the region between Norway and Svalbard. The highest concentrations were found around the North Pole, 14-19 Bq m^{-3}, between longitudes 50-180 °E and 0-15 °E but lower along the 0 meridian and 10-20 °W. The activity ratios $^{134}Cs/^{137}Cs$ (0.003-0-006) and the physical half-life of ^{134}Cs ($T_{1/2}$ =2.4 years) indicate that the origin of radiocaesium is mainly the Sellafield pulse 1974-1979 (ratio about 0.2). This means that it has taken about 14 years for radiocaesium from this source to reach this area.

In 1994 the ^{137}Cs activity concentrations along the Norwegian coast were quite similar to those in 1991. This means that present levels of radiocaesium are maintained by the outflow of Chernobyl related activity from the Baltic Sea.

Longitudinal distribution.

The distribution along the coast of the European Siberian Tundra is shown in Fig. 7. One could expect that run-off from the tundra is effective and the large rivers drain enormous areas which would result in a significant transfer of fallout deposited radionuclides to the Arctic ocean. West of 150 °E concentrations of radiocaesium are quite constant, 6-11 Bq m^{-3} while levels in the East Siberian Sea are in the order of 1-3 Bq m^{-3} where ^{134}Cs is absent which indicate that the only source is nuclear test fallout. The activity ratio $^{134}Cs/^{137}Cs$ west of 150 °E was about 0.015 indicating that about 30% of ^{137}Cs in these regions originated from the Chernobyl accident. A third source, tentatively European reprocessing facilities, is also present. The effects are even more pronounced when observing the bottom water, which is less influenced from the river water. The lower salinity coastal river water carries lower concentrations radiocaesium while the opposite is the case for Norwegian Coastal Current, Norwegian Sea and the Barents Sea (4) where coastal and Baltic water are

contaminated from the Chernobyl accident. Radionuclides are not accumulated at the continental shelf of the European- Siberian Tundra coast but washed off and advected together with the continental shelf water from the Laptev Sea along the Lomosov Ridge with the Transpolar Drift to the central Arctic towards Fram Strait.

Plutonium

Latitudinal distribution

Plutonium-239+240 is present in seawater from about less than 1 mBq m^{-3} to a few hundred mBq m^{-3}. A significant difference to radiocaesium is that caesium behaves rather conservatively in most waterbodies while plutonium is particle reactive and redox sensitive. This means that geochemical concentration processes of plutonium can take place. On the other hand plutonium in not concentrated in food chains like caesium.

Fig. 8. shows the distribution of plutonium in surface water from the North Pole to the Antarctic Peninsula Area obtained through the Swedish expedition to the Antarcic1988/89 and the Swedish Arctic Expedition 1991. As for radioceasium different symbols have been used for the two expeditions and decay correction would here in any case not be necessary due to the long physical half-life of the plutonium isotopes (5 670 years and 24 100 years respectively).

In the latitude band 0°S-60°S activity concentrations are about 1.5 mBq m^{-3} while concentrations suprisingly are higher, about 2 mBq m^{-3}, below the Antarctic convergence zone. There is no evidence of increased concentrations at latitudes where the pacific nuclear tests were performed but the relatively short residence time of plutonium in surface water would in any case have erased such signals. In the latitude band 5-25 °N we find concentrations about 3 mBq m^{-3} and at 25-50 °N around 8 mBq m^{-3}. Along the Norwegian coast from 60°N - 75°N activity concentrations of plutonium are decreasing while in the central Arctic ocean we find among the highest concentrations of $^{239+240}Pu$. This is rather unexpected since it has been shown that only a fraction (4%) of plutonium released from the Sellafield nuclear fuel reprocessing plant is present as soluble Pu and the remainder is deposited relatively close to the source (6). Results from the YMER-80 expedition showed that in 1980, 5-10% of $^{239+240}Pu$, 26-49% of ^{238}Pu and 19-45% of ^{241}Pu and in the Barents and Greenland Seas originated from European reprocessing facilities (7). Again in 1991 as for radiocaesium we find the highest concentrations around the North Pole between longitudes 0-15 °E, 9-16 mBq m^{-3}, while concentrations are lower between longitudes 50-80, 3-11 mBq m^{-3} where we actually found the highest ^{137}Cs concentrations. In Fig. 9 the distributions of plutonium in 1980 and 1991 along the Norwegian coast and in the Arctic Sea are compared. The mean activity concentrations decreased from 13 +/-2 mBq m^{-3} (n=48, 2 S.E.) to 8.4 +/-1.2 (n=38, 2 S.E.) which gives an effective apparent ecological half-life of 17 years or twice as high as for other parts of the North and South Atlantic.

There are several possible explanations for these results. The major pulse from the Sellafield reprocessing plants, 1974-1979, reached the central Arctic Ocean 10 years later and the fraction of the total plutonium from this source increased compared

to the figures above. On the other hand already in 1980 plutonium concentrations were much higher than expected above 75 °N and the isotopic composition indicated mainly that nuclear test fallout was the major source. A difference in source term behaviour might be possible, but the distribution of plutonium on suspended matter was not different for ^{238}Pu and ^{241}Pu compared to $^{239+240}Pu$. The activity ratio $^{238}Pu/^{239+240}Pu$ in releases from nuclear fuel reprocessing plants is in the order of 0.28 compared to 0.026 in fresh fallout. The corresponding figure for the activity ratio $^{241}Pu/^{239+240}Pu$ are 30 and 16 respectively. A more spectacular explanation is that during the former USSR tests at Novaya Zemlya they looked for that the wind was heading north. In this way activity contaminated Franz Josef land and its glaciers and constitute a continuos source to the Arctic Sea for plutonium and other radioisotopes. Another explanation is that plutonium behaves differently in the polar ice covered regions. The fraction of plutonium associated with suspended material was in 1980 in Barents and Greenland Seas 30+/- 8 % which is higher compared to other regions of the South and North Atlantic in 1989/90 except for around 30 °S and south of 60 °S in the Weddell Sea and the Antarctic Peninsula area. (Fig. 10). A higher fraction associated with suspended matter would actually normally decrease the residence time. These observations were done during the summer months and for the rest of the year the situation might be different. The composition of suspended material is certainly also different compared to other parts of the Atlantic. The more or less permanent ice cover, melting and freezing processes might also change the behaviour of plutonium and other radionuclides (oxidation state etc, prevention of vertical mixing etc.). Radiocaesium and radiostrontium would be expected to freeze out from the ice, but for particle reactive elements, in this particular environment, the behaviour is difficult to predict at this stage.

A closer look at the behaviour of plutonium compared to caesium in the South and North Atlantic is done by forming the Pu/Cs ratio (Fig. 11). Nuclear test fallout is the major source and the integrated ratio in 1989 is also indicated in the Figure. The figure shows the scavenging of plutonium relative to caesium which results in a shorter residence time for plutonium. The lowest Pu/Cs ratios are found around 30 °S indicate relatively higher scavenging of plutonium in this region, and the highest ratios are found in the Antarctic area indicating recirculation of organic material. The corresponding ratio for 1980 and 1991 in the Arctic Ocean were not plotted but were about $(0.5\text{-}1)\times10^{-3}$ and $(0.5\text{-}1.5)\times10^{-3}$ respectively which is the same as for the Antarctic area. This is similar to other areas of the North Atlantic where fallout is the major source of contamination but is much higher than expected since Sellafield is the major source and the ratio should decrease with distance. The Pu/Cs ratio would be in the order of 0.2×10^{-3} if plutonium had behaved as expected. This suggests a significant different behaviour of plutonium and possibly other actinides in the polar regions which should be investigated further.

CONCLUSIONS

The shorter effective ecological residence time of plutonium (8 years) compared to radiocaesium (100 years) can be demonstrated in the North and South Atlantic. The Pu/Cs ratios in surface waters are therefore well below integrated fallout ratios. The ratios are enhanced as are the fractions on suspended matter of plutonium

in the polar regions. The residence time of plutonium in the upper mixed layer is at least twice as high in the Arctic and possibly also in the Antarctic area as in other areas of the South and North Atlantic. This suggests a different behaviour of particle reactive and/or redox-sensitive elements in these region by influence of ice formation and/or the strong seasonal variation.

The releases of ^{137}Cs from the Sellafield reprocessing plant in 1974-1979 have now reached the central Arctic Ocean while Chernobyl related radiocaesium could in 1994 be found along the European- Siberian Tundra coast up to the Laptev Sea from where will be advected together with shelfwater into the central Arctic ocean with the Transpolar Drift. Plutonium has mainly nuclear test fallout as origin and is distributed differently compared to radiocaesium in the central Arctic Ocean. Outflow from the Baltic Sea into the North Sea is transported by the Norwegian current and constitutes today a major source for input of radiocaesium into the Arctic

ACKNOWLEDGEMENTS

These expeditions were made possible through the logistical assistance of the Swedish Polar Research Secretariat. Assistance from the organisers, supporters and leaders of the expeditions is thereby gratefully acknowledged as is the financial support from the Swedish Radiation Protection Institute, the Swedish Natural Research Council, the CEC Fission and Safety Programme, The Nordic Committee for Nuclear Safety Research (NKS), Crafoordska Stiftelsen, Magn. Bergvalls Stiftelse, Carl Jönssons Understödsfond.

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Figure captions.

Fig. 1. Cruising and sampling route for the expedition to the Barents and Greenland Seas, YMER-80, 1980.

Fig. 2. Areas covered by the Swedish Antarctic Research Expedition, SWEDARP, 1988/89.

Fig. 3. Cruising and sampling route for the Expedition to the North Pole, SWEDARCTIC-1991.

Fig. 4. Sampling stations along the coast of the European-Siberian Tundra during the Swedish-Russian Tundra Expedition, 1994.

Fig.5. Activity concentration of ^{137}Cs in surface water as a function of latitude from Sweden to the Antarctic Peninsula Area, 1988/89 (SWEDARP) and from Sweden to the North Pole, 1991, (SWEDARCTIC).

Fig. 6. Activity concentrations of ^{137}Cs in surface water along the Norwegian coast to the Arctic in 1980 (YMER-80), 1991 (SWEDARCTIC-91) and 1994 (TUNDRA-94).

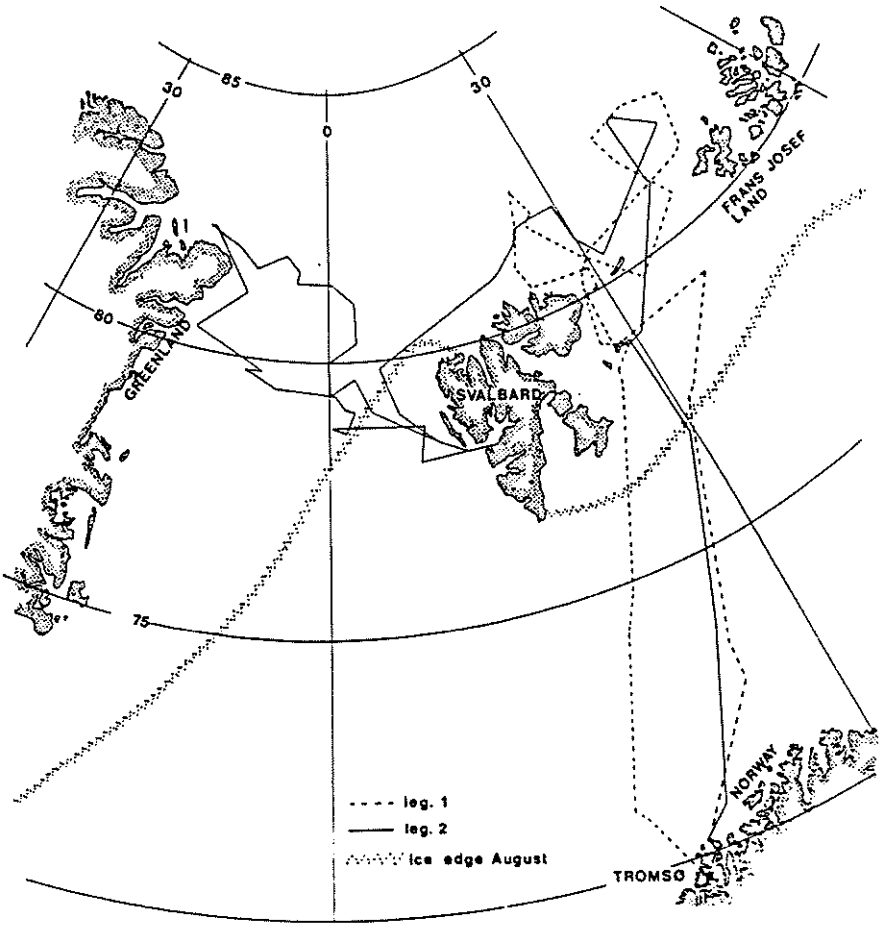
Fig. 7. Concentrations of ^{137}Cs in surface water in 1994 as a function of longitude along the continental shelf of the European-Siberian Tundra (TUNDRA-94).

Fig. 8. Activity concentration of plutonium in surface water from Sweden to the Antarctic Peninsula Area and from Sweden to the North Pole in 1988/89 (SWEDARP) and 1991 (SWEDARCTIC) respectively

Fig. 9. Activity concentration of $^{239+240}Pu$ along the Norwegian coast and in the Arctic in 1980 (YMER-80) and 1991 (SWEDARCTIC).

Fig. 10. Fraction of plutonium associated with suspended matter as a function of latitude from Sweden to the Antarctic Peninsula Area in 1991 (SWEDARP).

Fig. 11. The activity ratio $^{239+240}Pu/^{137}Cs$ as a function of latitude from Sweden to the Antarctic Peninsula Area in 1991 (SWEDARP).



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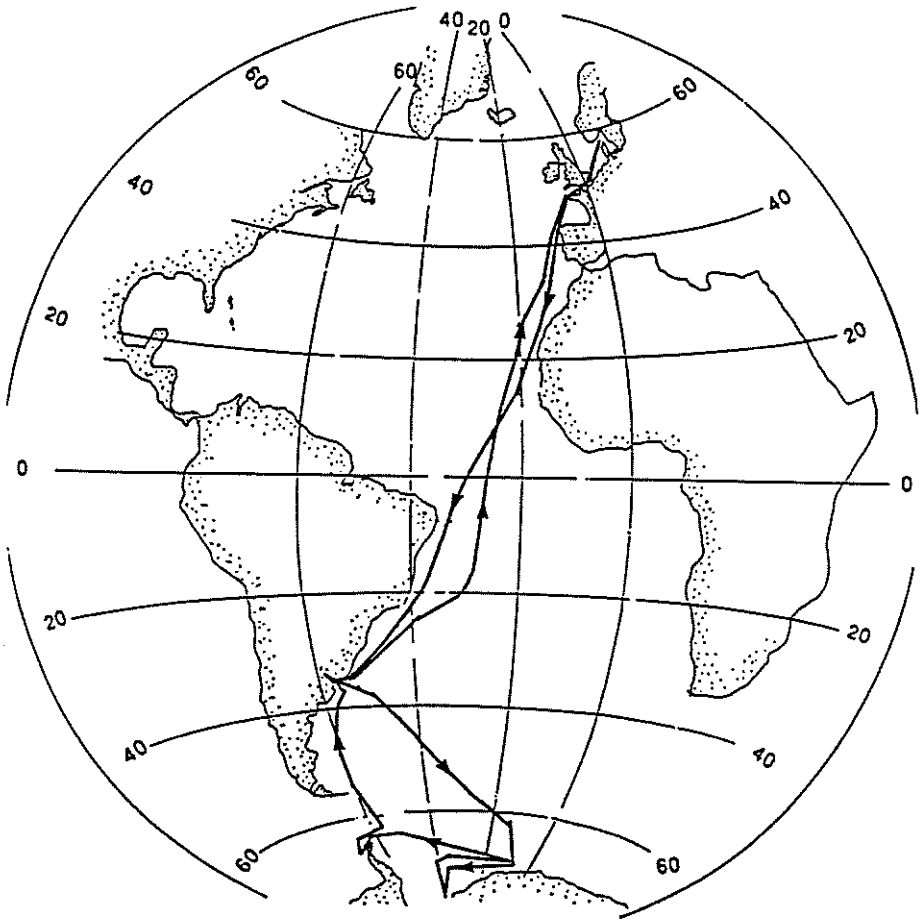

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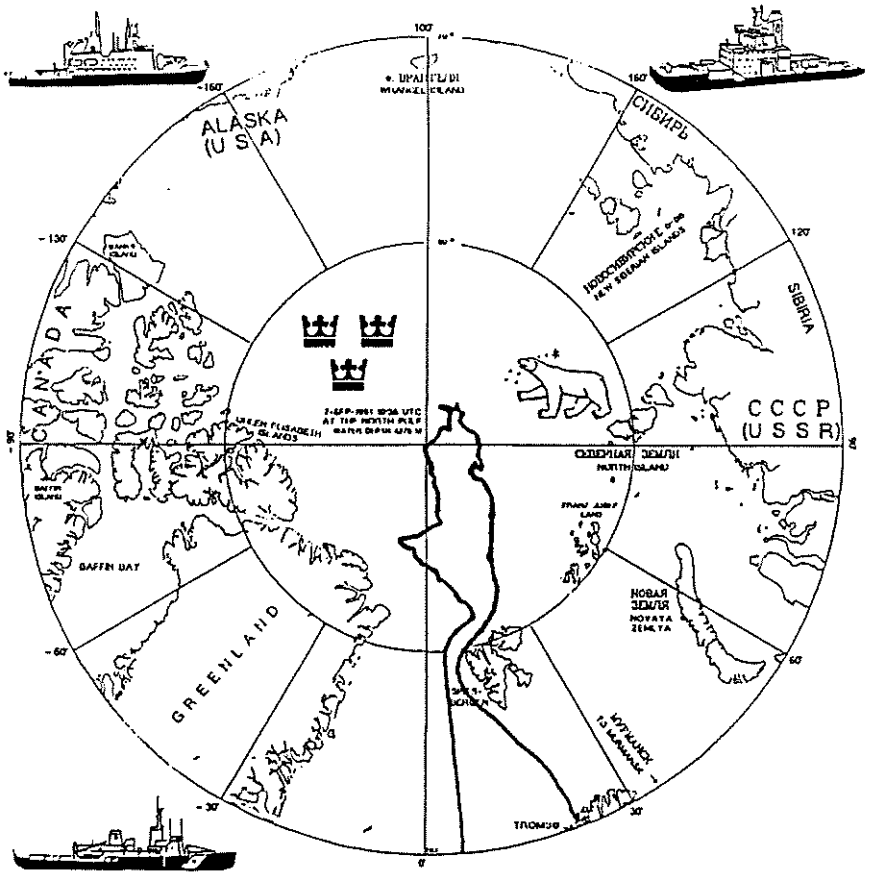

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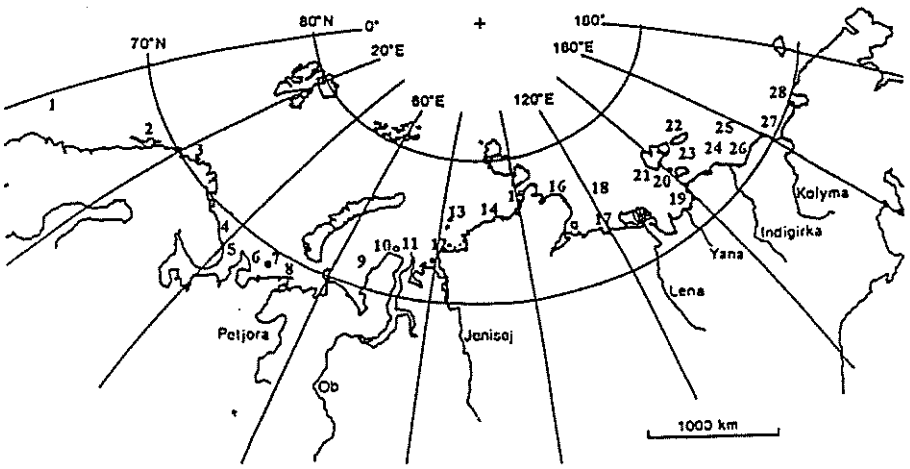


Fig. 4. Sampling stations along the coast of the European-Siberian Tundra during the Swedish-Russian Tundra Expedition, 1994.

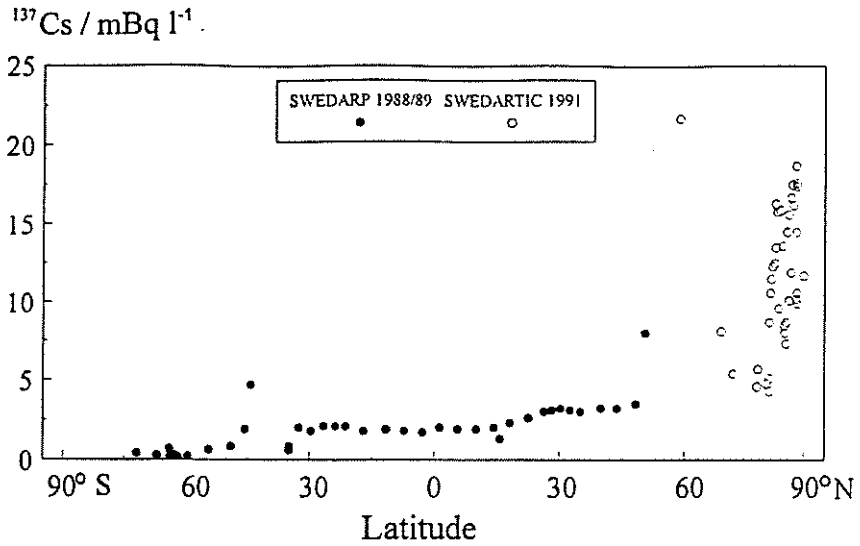
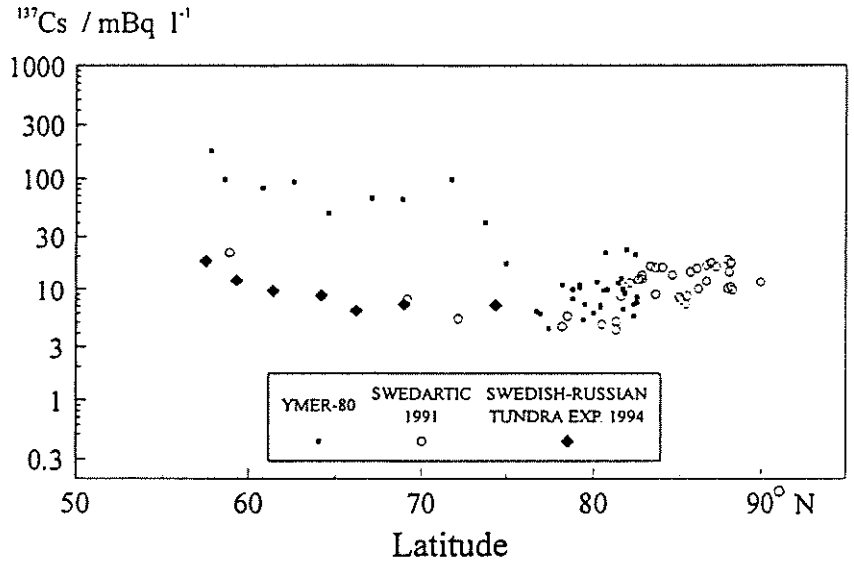


Fig.5. Activity concentration of ^{137}Cs in surface water as a function of latitude from Sweden to the Antarctic Peninsula Area, 1988/89 (SWEDARP) and from Sweden to the North Pole, 1991, (SWEDARCTIC).



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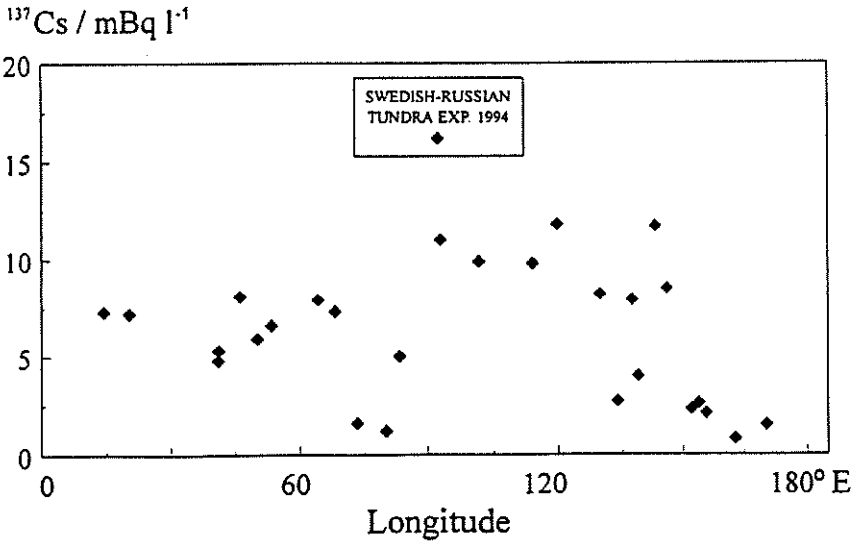

Fig. 7. Concentrations of ^{137}Cs in surface water in 1994 as a function of longitude along the continental shelf of the European-Siberian Tundra (TUNDRA-94).

Mobility of Cs and Sr in Soils: Implications for Transfer Factors and Ecological Half-lifes in Nordic Ecosystems

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The mobility and transfer of a radionuclide varies between sites and with time after deposition. Long-term changes in transfer factors in terrestrial ecosystems usually reflect changes in the speciation or distribution of radionuclides in soils. In soils, the mobility of a radionuclide is influenced by three main factors: 1) the initial physico-chemical form of the fallout; 2) transformation processes which change the original distribution of species; and 3) the radionuclide's subsequent reactivity with soil components. The reactivity of any element with soil components is a function of its reaction strength, rate and reversibility. Therefore, information on radionuclide speciation, the kinetics of transformation processes and soil-radionuclide interactions, together with the effect of environmental parameters is important for understanding variations between different sites.

This paper presents some studies on the mobility of radionuclides in environmentally (Weapons' fallout, Chernobyl) and experimentally (^{134}Cs and ^{85}Sr tracer) contaminated soils. Particular attention being paid to the influence of Cs and Sr mobility in soils on the transfer of radionuclides in Nordic soil-veg-lamb food chains. The main methods used include sequential extraction, dynamic tracer experiments and analysis of stable Cs and Sr distribution.

The results confirm the importance of the three factors in determining radionuclide mobility. Chernobyl contaminated soils are often associated with low mobility of ^{90}Sr due to the presence of fuel particles, and studies on Ukrainian soils indicate that transformation processes leading to particle weathering are dependent on particle composition well as environmental parameters (e.g. pH). Laboratory tracer studies on soils collected from various Nordic sites, show that ionic Cs and Sr are rapidly bound to soils: Sr remaining in exchangeable forms, but Cs becoming strongly fixed to soil components. Fixation of ^{137}Cs in soils seems to be the predominant process leading to reduction in soil-vegetation transfer factors. Soil chemistry and mineralogy clearly influences radionuclide mobility between the different sites. Analysis of stable Cs distribution shows that transfer factors at steady state vary between the different sites. Hence environmental differences and ^{137}Cs contact time rather than differences in fallout speciation probably account for the anomalies between Nordic sites. Finally the significance of soil ingestion on calculated veg-lamb transfer factors has been investigated using a recently developed, sensitive method using NAA analysis of Sc faeces.

The role of fungi in the transfer of ^{137}Cs from forest to man

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Abstract

The mushroom-picking habit was investigated in Avesta commune in central Sweden. In 1995, the three most common picked species were Cantarellus tubaeformis, 0.35 kg, C. cibarius, 0.17 kg, Boletus edulis, 0.04 kg and 0.19 kg of various species. By using transfer factors (TF_g) for these species the potential annual transfer to an average person in Avesta commune was calculated to be 263, 168, 56 and 152 Bq, respectively. In a normal mushroom year, the corresponding potential transfer is higher or about 1248 Bq. Additionally, quite a large part of the ^{137}Cs transferred by game animals to man is due to intake of mushrooms by roe deer and moose.

Introduction

In the forest ecosystems, fungi have a very important role both in the transfer of ^{137}Cs from soil to fruitbodies of fungi and to plants and further to game animals. A direct pathway to man is when people pick and eat mushrooms. The ^{137}Cs transfer this way can be quite high due to the high ^{137}Cs activity concentrations in many fungal species. Another pathway is by game animals since particularly roe deer eat quite large quantities of mushrooms, resulting in high levels of ^{137}Cs in game meat during certain periods. Furthermore, the mycelium of fungi may be the most important part in the soil responsible for the high uptake of ^{137}Cs, not only to fruitbodies of fungi but also to most of the plants in the forest ecosystem. An important role of the mycelium of mycorrhizal fungi in the soil is to take up mineral nutrients and deliver them to vascular plants in the formation of mycorrhizae.

Materials and Methods

Questionnaire

In 1995, a questionnaire was sent to 1000 families in Avesta commune in the central part of Sweden asking about their mushroom-picking habits. This questionnaire was answered by 52 % of the families. Based on the answers of the first questionary, a second was sent to those families who answered that they were picking mushrooms at least tvice a year. They were asked to answer in more detail how much of various species of mushrooms they collected in 1995 (a bad mushroom year) and to try to estimate how much they picked in a normal mushroom year.

Game animals

Results of the ^{137}Cs activity concentrations in moose were obtained from Harbo-Nora-Östervåla hunting organisation. Each year, between 190 and 275 muscle samples of harvested moose were collected. In 1992 and 1993, 199 respectively, 178 muscle samples of harvested roe deer were obtained from the local hunters in Harbo. For both moose and roe deer samples the ^{137}Cs activity concentrations were determined using high purity Ge detectors. Botanical analysis of moose (Johanson et al. 1994) and roe deer rumen content were performed in two other studies. Information on ^{137}Cs activity concentrations in various species of mushroom and vascular plants were also obtained from other studies.

Results and discussion

Of the families who answered the first questionnaire (52 %) a total of 59 % picked mushrooms more than twice per year and to them we sent a second questionnaire. The results of this questionnaire showed that the three most frequent species in the mushroom basket were Cantharellus tubaeformis, Cantharellus cibarius and Boletus edulis, totally 0.56 kg fw. Additionally, 0.19 kg of other species were picked in 1995.

Previously we had determined the TF_g for C. tubaeformis to be 0.94 m^2 kg^{-1} dw, 0.39 for C. cibarius and 0.28 for B. edulis. The TF_g for the other species was usually higher and we used 1.0. From these values and the mean ground deposition in Avesta commune - 10,000 Bq m^2 - the ^{137}Cs levels in the various mushroom species could be estimated. For C. tubaeformis it was 752 Bq kg^{-1} fw, 312 for C. cibarius, and 224 for B. edulis. For the other species, the mean level was 800 Bq kg^{-1}.

Table 1. The consumption (kg fw y^{-1} per person) and transfer factors (TF_g) of the three most important species of fungi and the calculated potential transfer of ^{137}Cs (Bq y^{-1})

Species	Consumption (kg fw)		TF_g	transfer to man Bq per capita	
	1995	normal year	m^2 kg^{-1}dw	1995	normal year
C. tubaeformis	0.35	0.84	0.94	263	632
C. cibarius	0.17	0.54	0.39	53	168
B. edulis	0.04	0.25	0.28	9	56
Other species	0.19	0.49	1.0	152	392

Combining these values with the consumption data the estimated potential transfer of ^{137}Cs to each person in 1995 was 263 Bq for C. tubaeformis, 53 Bq for C. cibarius, 9 Bq for B. edulis and 152 Bq for other species. Altogether, the estimated potential transfer to the average person was 477 Bq in 1995 and 1248 in a normal year, corresponding to 6.0 µSv in 1995 and 16 µSv a normal year. Assuming that Avesta commune is representative of all Sweden, the potential collective dose in Sweden was 48 manSv in 1995 and 128 in a normal year. The mean ground deposition of ^{137}Cs in Avesta commune is similar to that in all Sweden and the location being rather central in Sweden with a mixture of arable and forested land. In Avesta commune, there seem to be no countermeasures such as decreasing the picking of mushrooms compared with before the Chernobyl accident. However, the extrapolation should only be seen as a rough estimation of the potential collective dose. It is also important to realise that some ^{137}Cs activity will be lost in the preparation and cooking processes.

Roe deer

The ^{137}Cs levels in roe deer show a very marked seasonal variation with peak levels during the mushroom season in August to October (Karlén et al. 1991, Johanson 1994). During this period, often 20 to 30 % of the roe deer rumen content is found to be mushrooms so it is quite clear that roe deer consume large quantities of them.

In 1992 and 1993, the mean aggregated transfer factor, T_{ag}, for roe deer during August to October was 0.093 m^2 kg^{-1} and 0.027 during the rest of the year. Assuming that our research area is representative for Sweden with a mean ground deposition of 10,000 Bq m^2, the mean roe deer in Sweden has a ^{137}Cs level of 930 Bq per kg in August to October and during the other parts of the year 270. The annual harvest of roe deer in Sweden is about 300,000 animals and, assuming a mean meat weight of 10 kg per roe deer, the annual meat production is 3 million kg. About one-third of the harvested roe deer, corresponding to one million kg of meat, is harvested during August to October which will give a total potential transfer of 930 MBq. Of this 270 Mbq is due to intake of "normal" fodder and the rest, 660 MBq, due to ^{137}Cs coming from mushrooms. During the rest of the year, 2 million kg meat transfer potentially 540 MBq to man by roe deer. This will mean that the mushrooms were vector for 660 MBq corresponding to a potential collective dose of 8 manSv, and other vascular plants for 810 MBq corresponding to a potential collective dose of 10 manSv. The mean ^{137}Cs level in roe deer living in the Harbo area might be higher than the mean level in all Sweden since the major part of the roe deer population in Sweden was in the southern part of Sweden and probably on richer biotopes than found in Harbo.

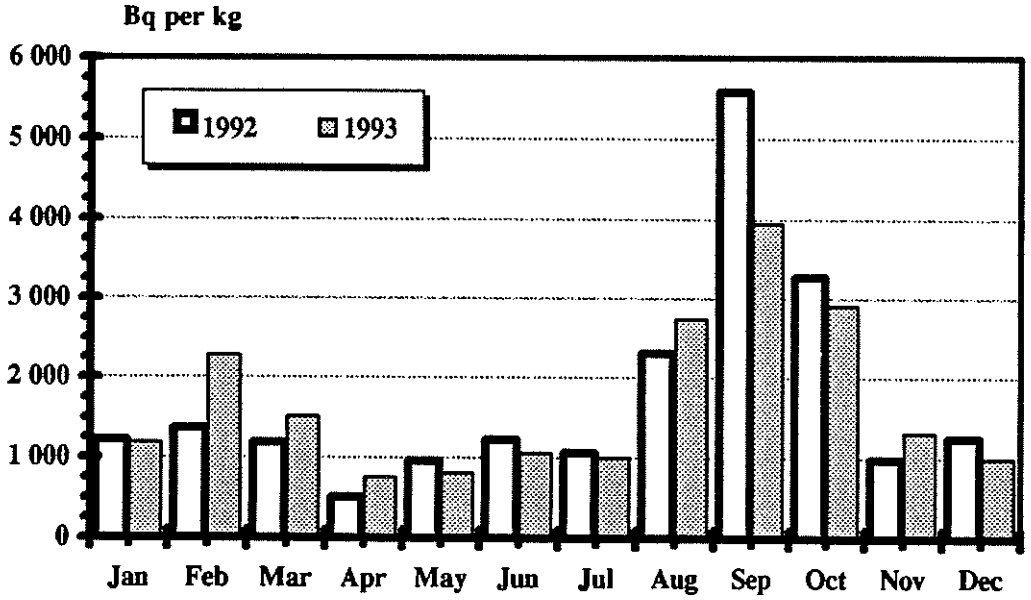

Figure 1. The seasonal variation of ^{137}Cs levels in roe deer.

Moose

The mean ^{137}Cs level in moose harvested in our research area during the period 1986 to 1993 was 750 Bq per kg and the ground deposition about 37,500 Bq m^{-2} given an aggregated transfer factor of 0.02 (Johanson & Bergström, 1994, Johanson, 1994). In 1994, there was a small decrease in the ^{137}Cs levels in moose and in 1995 a more pronounced decrease to about 450 Bq per kg or 40 % of the mean value of the period 1986 to 1993. In the moose rumen rather small amounts of mushrooms can be found (Johanson et al. 1994), probably due to the fact that most harvested moose had been ruminating the rumen content before they were shot. The decrease in 1995, however, strongly indicates that moose eat larger quantities of mushroom than found by botanical analysis of rumen content when they become available. In some years mushrooms appear

in August but more frequently in September. In 1995, we found less than 10 % of fruitbodies of fungi in our research area compared with 1994. The ^{137}Cs levels in moose during the hunting period starting in October are mainly dependent on the intake of ^{137}Cs during September.

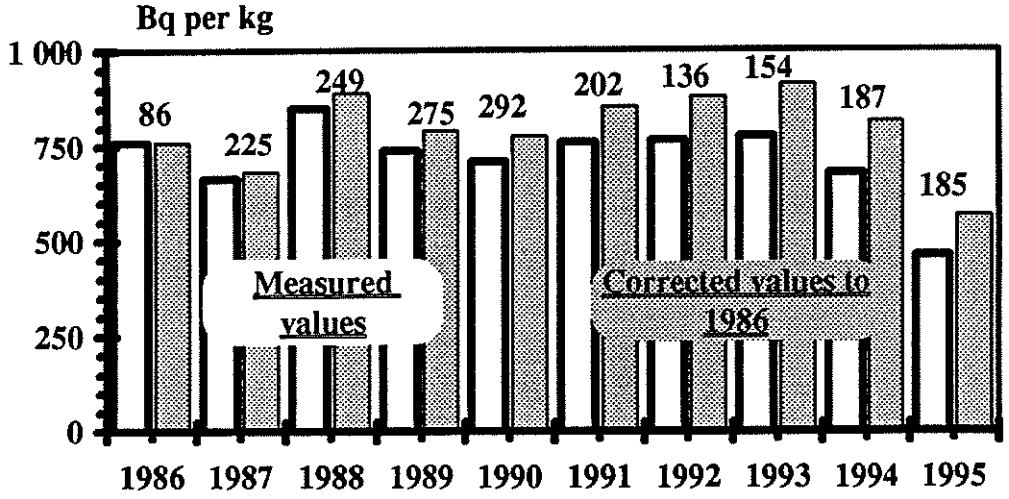

Figure 2. The mean ^{137}Cs levels in moose harvested by the Harbo-Nora-Östervåla hunting organisation during 1986 to 1995.

The annual harvest of moose in Sweden has been about 100,000 and the mean moose carcass weight is 135 kg or about 100 kg meat. The annual production of moose meat in Sweden is thus 10 million kg. The aggregated transfer factor for moose seems to be about 0.02, at least in central parts of Sweden where the moose-hunting period starts in October. The ^{137}Cs levels in a mean Swedish October moose should thus be about 200 Bq per kg and the total transfer of ^{137}Cs to man by moose should thus be 2 GBq. Of this roughly 40 % or 800 MBq is due to intake of mushrooms. This will correspond to 10 manSv due to mushrooms and 15 manSv due to vascular plants. Most moose are harvested in the northern part of Sweden and therefore our research area is probably rather good model for all Sweden.

The most important pathways seem to be the direct picking and eating of mushrooms although there is probably an upper limit, as given in this paper. Of the game animals, moose is of course the most important since the total amount of meat is much higher for moose than for roe deer. However, higher levels of ^{137}Cs are usually found in roe deer.

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Long-term behaviour of ^{137}Cs and ^{90}Sr in a large Finnish freshwater basin

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Abstract - Päijänne, a large lake in central Finland covers about 1100 km^2. The study area, Asikkalanselkä, is the southernmost, oligotrophic basin of the lake with an area of 52.8 km^2. The basin is important as a raw water source for the capital and environs and for freshwater fishing in Finland. The area received rather heavy deposition of ^{137}Cs from Chernobyl in 1986, about 64 000 Bq/m^2. Deposition of ^{90}Sr in the study area was about 420 Bq/m^2. After deposition there was a rapid decrease in ^{137}Cs concentrations in the water of the lake. The longer the time after deposition, the longer was the halftime of ^{137}Cs in water, being four-five years in 1996. A slight increase due to runoff from the catchment on the concentrations of ^{90}Sr in the near-surface water of the Asikkalanselkä basin was observed during the first years following deposition. In 1994, concentrations of ^{137}Cs were slightly higher in near-bottom than in surface water, implying resuspension from sediment. The relative distribution of ^{137}Cs between water, particles in water and sediment has changed gradually. Thus the fraction transferred to bottom sediment increased until 1988, but remained almost the same or decreased somewhat after that. For example, in 1994, 96% of ^{137}Cs was in sediment and only 4% in water and seston, whereas in August 1986, 55% of the ^{137}Cs was in water, about 30% in sediment and the rest in particles in water. ^{90}Sr showed less affinity for seston than ^{137}Cs. Gradual migration of the Chernobyl ^{137}Cs downwards in bottom sediment was found. ^{137}Cs in non-predatory fish was at its highest in the year of the accident, 1986, but in predatory fish two years later, when the average in pike was 4200 Bq/kg fresh weight. In fish as in water, the longer the time after deposition, the longer was the halflife of ^{137}Cs , being in 1996, ten years after deposition, typically 3-7 years depending on the species. In 1995 the values for pike varied from 300 to 600 Bq/kg. ^{90}Sr was low, about 0.3 Bq/kg in pike and 0.6 Bq/kg in perch. The results confirm the distinct differences present in the long-term behaviour of ^{90}Sr and ^{137}Cs in the aquatic ecosystem.

Introduction

Southernmost lake Päijänne is occupied by a large freshwater basin, Asikkalanselkä (Fig. 1). The basin is oligotrophic and its area is 52.8 km^2. It is the main source of household water for the Helsinki metropolitan area. Like the whole water system, the Asikkalanselkä basin is important for recreational and professional fishing, and so for decades has been a target of research in terms of general water quality issues and radioactivity. The considerable fallout of ^{134}Cs ($T_{1/2}$= 2 years) and ^{137}Cs ($T_{1/2}$ = 30 years) due to the Chernobyl accident in 1986, has allowed comprehensive studies to be conducted using radiocaesium as a tracer. The behaviour of gamma-emitting radionuclides and transuranic elements in water, particles in water and in sediment was studied during 1987-1991 by the Institute of Radiochemistry and Paavo Ristola Oy (1-8) and a sedimentation model was developed (9). The Finnish Centre for Radiation and Nuclear Safety (STUK) has analysed water and fish from the area since 1986 (10-15) and bottom sediment in 1988 and 1990 (16-17). A joint study between the laboratory of radiochemistry of Helsinki University and the foodchain laboratory of STUK was launched in 1991 on the long-term behaviour not only of radiocaesium but also of ^{90}Sr ($T_{1/2}$ = 28 years), which is known for having greater environmental mobility than radiocaesium. The study led to the gradual production of a versatile, site-specific database for environmental and radiological assessments that is also useful for radioecological modelling (18).

Sampling and analysis in Asikkalanselkä in 1991-1996

Temporal changes in the distribution of ^{137}Cs and ^{90}Sr between water, particles in water and sediment together with changes in the vertical distribution of ^{137}Cs in sediment and soil profiles were investigated. Water samples were taken from depths of 1, 10, 20, 30, 40 and 50 m in 1994. A sediment profile from sampling point A3 and soil samples from the catchment of the basin were taken in 1994 to supplement earlier samples. Several fish species were sampled and

analysed for ^{137}Cs and ^{90}Sr every year. Water samples from small rivers discharging to the lake and soil samples from the catchment of the rivers were taken in 1995 and 1996. As well as the collecting and analysis of new samples, many samples of different types from previous years, e.g. of lake water, seston, lichen and fishes, were analysed for ^{90}Sr. The analytical methods used are given in earlier reports.

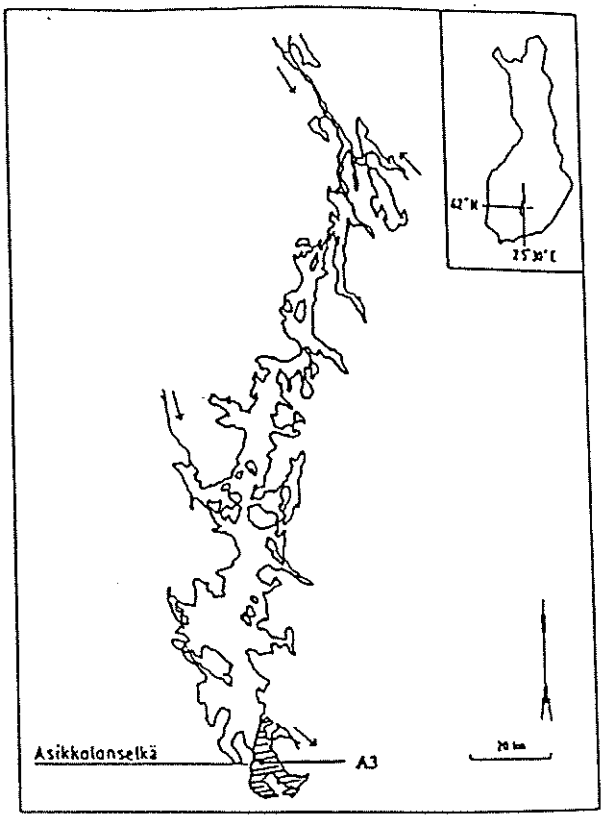

Fig. 1. Location of Lake Päijänne and the Asikkalanselkä basin in Finland. The area of the lake is 1100 km^2 and that of Asikkalanselkä 52.8 km^2.

Results for ^{90}Sr and ^{137}Cs

According to the areal distribution pattern of ^{137}Cs deposition, determined by Arvela et al (19), the Asikkalanselkä basin is part of the area most heavily contaminated by Chernobyl deposition in Finland. The average deposition of ^{137}Cs in the surrounding municipalities ranged from 60 900 Bq/m^2 to 66 700 Bq/m^2, determined by measurements of external gamma radiation. Depositions of ^{137}Cs and ^{90}Sr in the study area were checked by analysing lichen from the catchment area of the basin. The value for ^{137}Cs is consistent with the average value reported by Arvela et al (19) (Table I). Lichen samples of August 1986 describe the Chernobyl deposition of ^{90}Sr rather well, as the removal rate of ^{90}Sr from lichen is 4 - 5 years (20). The $^{137}Cs/^{90}Sr$ ratio in deposition in the study area in 1986 was about hundred times higher than that in the nuclear weapon test fallout (Table I). Since the end of April 1986, environmental ^{137}Cs in the catchment has originated mainly from Chernobyl fallout, whereas ^{90}Sr is dominated by nuclear weapons fallout (Table I).

Table I. Estimated ^{90}Sr and ^{137}Cs deposition (Bq/m^2) in the Asikkalanselkä catchment calculated for 1.10. 1986.

Origin	^{90}Sr	^{137}Cs	$^{137}Cs/^{90}Sr$
Nuclear weapons	1100	1790	1.4 ± 0.2
Chernobyl accident	425	64700 ± 3000	151

Water. ^{137}Cs in the water of Asikkalanselkä decreased quickly in 1986, when the halflife was about 50 days (9). The halflife has gradually increased, being 4-5 years in 1996. The main mechanism for ^{137}Cs removal was by sinking with the suspended matter of water to the bottom sediment. The increasing effect of runoff from the catchment is seen in concentrations of ^{90}Sr in the surface water of Asikkalanselkä in 1989 and 1990 (Fig. 2).

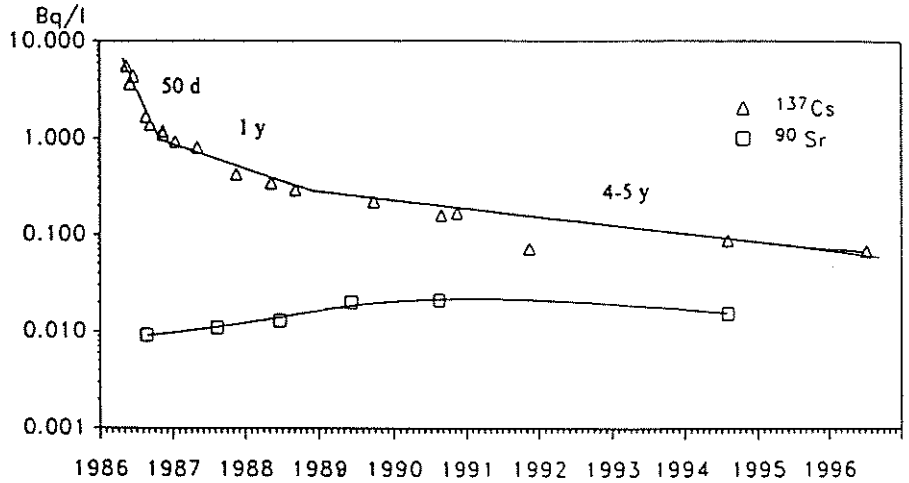

Fig. 2. ^{137}Cs and ^{90}Sr in water of Asikkalanselkä (Bq/l) in 1986 - 1996.

Concentrations of both ^{137}Cs and ^{90}Sr were lower in the near-surface water than at greater depths in August 1986. In 1994 ^{137}Cs was clearly higher in near-bottom water than at the surface, which means that ^{137}Cs detected in the near-bottom water has been resuspended from the sediment. Concentrations of ^{90}Sr in water were much lower than those of ^{137}Cs, and ^{90}Sr was rather evenly distributed throughout the water column.

Seston. Concentrations of ^{137}Cs in seston also decreased quickly after deposition (7). Although low, in 1994 ^{137}Cs concentrations both in water and in seston were higher near the bottom than near the surface. As to the relative distribution of ^{137}Cs between water and seston, ^{137}Cs is now almost totally in the water phase, with only a minor fraction being in particles. For example, in August 1994, more than eight years after the deposition, 3-8% of ^{137}Cs was in the particle fraction (> 0.45 mm) at different depths. A clearly smaller fraction of ^{90}Sr than of ^{137}Cs was incorporated in particles; only 0.03-0.7% at different depths in 1994 (Table II).

Table II. Distribution of ^{137}Cs and ^{90}Sr in water and seston at different depths at sampling station A3 in August 1994.

Depth m	^{137}Cs Water %	Seston %	^{90}Sr Water %	Seston %
1	96,8	3,2	99,97	0,03
20	94,1	5,9	99,5	0,5
50	91,9	8,1	99,3	0,7

Sediment. During six years ^{137}Cs has migrated downwards 2 - 3 centimetres in bottom sediment at a sampling point in Asikkalanselkä (Fig. 3).

With reference to the relative distribution of ^{137}Cs in water, seston and sediment, in August 1986, four months after deposition, 55% of ^{137}Cs was in water, 29% in sediment and 16% in seston (5). Thereafter, the fraction transferred to bottom sediment increased until 1988, but remained almost the same or decreased somewhat after that. In 1994, 96% of ^{137}Cs was in sediment and only a minor fraction was in water (3.7 %) and seston (0.3 %) (Fig. 4).

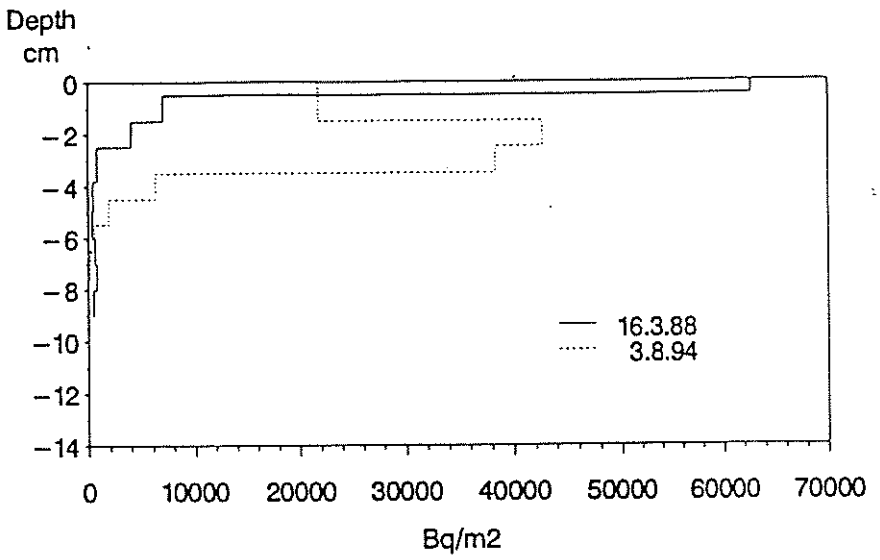

Fig. 3. Vertical distribution of ^{137}Cs in sediment profiles at Asikkalanselkä sampling point A3 in 1988 and 1994.

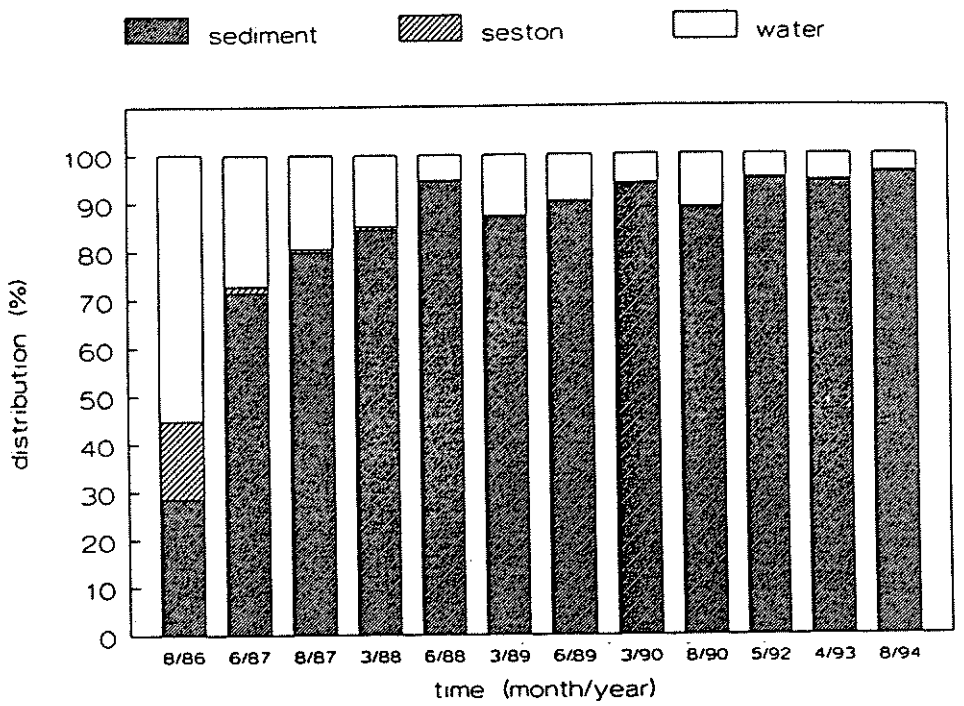

Fig. 4. Relative distribution of ^{137}Cs in the system water-seston-sediment at Asikkalanselkä sampling point A3 in 1986 - 1994.

Fish.. ^{137}Cs in fish in Asikkalanselkä was at its highest in 1986, 1987 or 1988, depending on the species (Fig.5). Concentrations were at a maximum in non-predatory fish as vendace, bream and roach in 1986, and then started to decrease. The maxima in pike and other predatory fish were not recorded until 1988, because these species obtain caesium along a longer pathway than the other species. Variation in the ^{137}Cs contents of a species was large, even during a short period, especially in other than nonpredatory fish, which showed the smallest variation.

^{90}Sr also varied greatly in fishes, from 0.1 to 10 Bq/kg fresh weight (Fig. 6). It was highest in perch and other small fish, partly at least owing to the higher proportion of bones in those samples. The fish were cleaned and gutted as normally in the kitchen. ^{90}Sr correlates with the bone contents of the samples, because strontium, like calcium, is mainly transferred to the bone tissue of fish.

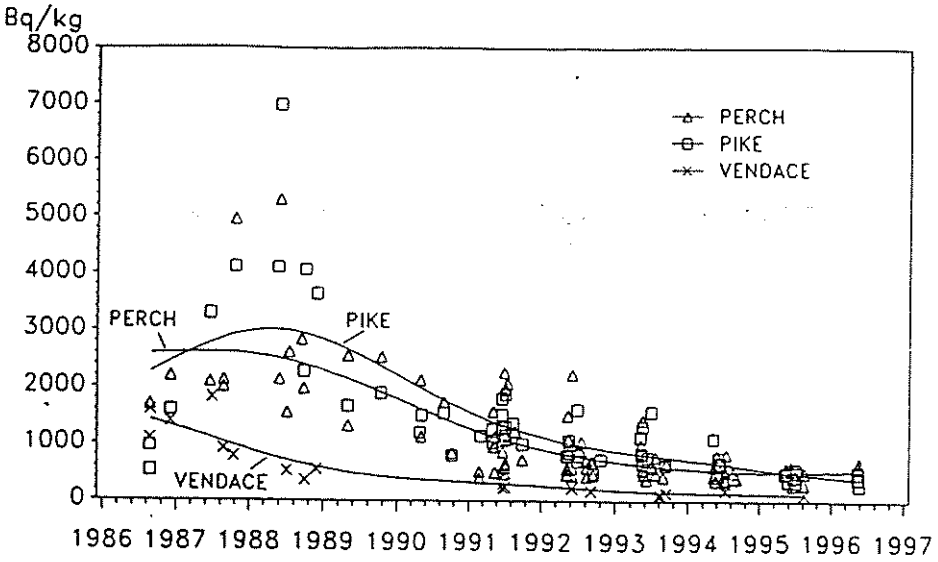

Fig. 5. ^{137}Cs (Bq/kg) in perch, pike and vendace, three species with different feeding habits, in 1986 - 1996.

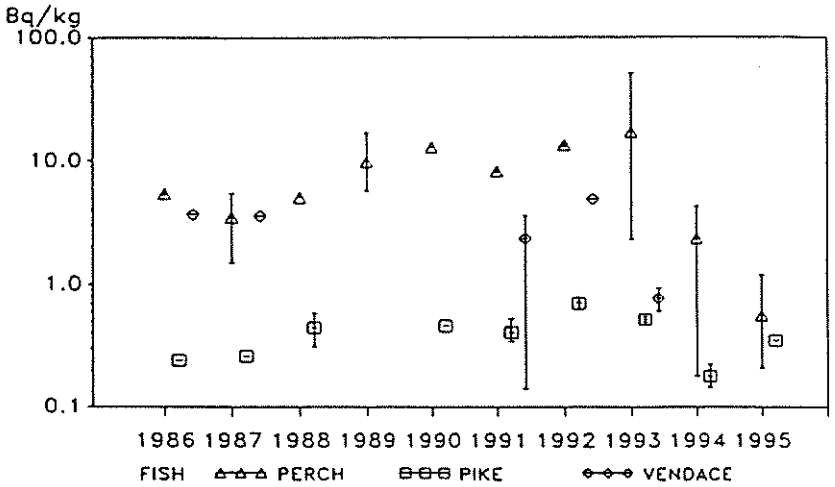

Fig. 6. Annual averages and ranges of ^{90}Sr (Bq/kg) in perch, pike and vendace in 1986 - 1995.

In the year of fresh fallout concentration ratios for ^{137}Cs from water to pike, perch and vendace varied from 380 to 710 and increased remarkably in the following years (Table III).

Table III. Concentration ratios for ^{137}Cs and ^{90}Sr (Bq/kg in fish/ Bq/kg in water) for pike, perch and vendace in Asikkalanselkä basin in 1986 - 1994 with corresponding pre-Chernobyl values, calculated from reference (21).

	1986	1987	1988-1994	1982
^{137}Cs				
Perch	710	3900	9400 ± 2400	4200
Pike	380	5200	11700 ± 3800	3700
Vendace	500	1600	2600 ± 1300	2600
^{90}Sr				
Perch	590	320	650 ± 200	60
Pike	30	20	30 ± 7	10
Vendace	440	330	240 ± 100	50

In 1988-1994 the figures for ^{137}Cs were 2-3 times higher than the pre-Chernobyl values. Values for ^{90}Sr were almost the same before and after the Chernobyl fallout for pike, but for perch and vendace post-Chernobyl values were one order of magnitude higher, possibly due to differences in cleaning and gutting of the samples.

Conclusion

The results confirm the fundamental differences in the long-term behaviour of ^{90}Sr and ^{137}Cs in natural aquatic systems. The relative fractions of ^{90}Sr released from the catchment area to the water basin are higher than those of ^{137}Cs. Cs-137 is more closely associated with sediment than is ^{90}Sr, which therefore favours the liquid phase. The relative uptake of ^{90}Sr and ^{137}Cs by fish and their contents in the edible fraction of fish differ essentially, reflecting the difference in the metabolism of alkaline and alkaline earth elements, Cs and Sr. The dynamics of radionuclides during the first weeks and months after deposition emphasize the importance of early countermeasures against ^{137}Cs in water in the event of an emergency. In extreme cases ^{90}Sr would cause long-term human exposure to internal radiation via fish and drinking water after heavy fallout.

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SHEEP GRAZING AND RADIOCAESIUM, SOME EXTERNAL FACTORS

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ABSTRACT

Iceland has been participating in NKS's lamb projects since 1990, when the RAD-3 project was started, and later in the EKO-2 project. As both projects concentrate on nuclide transfer from soil to grazing lambs, the Icelandic Radiation Protection Institute (IRPI) suggested collaboration with the Agricultural Research Institute's (AGRI) Animal Nutrition Division. In 1994 50% of all meat production was lamb meat, making it the main meat product in Icelandic Agriculture. The annual lamb meat sale in that year was 27.2 kg capita^{-1}, 46% of the total meat sale. It is thus clear that lamb meat consumption must be considered as one of the main pathways for radionuclides into the human foodchain.

One of the aims of the Icelandic study have been to collect data from as broad spectrum of land types as possible. The RAD-3 project was carried out at two different locations, in highland pasture, previously used for grazing research and at the experimental station Hestur on lowland mire. The EKO-2 project is being conducted at the experimental station Stóra Ármót on uncultivated dry lowland pasture. Thereby it is assumed that a wide range of the land types, used for sheep grazing, are covered.

AGRI has been responsible for providing animals and experimental sites, as well as monitoring of animals (live weight, live weight gain), estimation of standing herbage (biomass) and part of the sampling, as well as part of the sample preparation. Caesium analysis of all samples has been carried out at IRPI. Chemical analysis of vegetation are carried out at AGRI. Collected samples include soil, milk, vegetation and faeces.

Relatively high variation has been observed in caesium content, both in soil and vegetation, as well as between animals, due to heterogeneous distribution within the soil and vegetation. This emphasises the importance of connecting the data with the environmental factors on each site. Factors connected with the animals, i.e. grazing pattern, plant selection, intake and digestibility must also be considered. As the sampling for the RAD-3 project have been done on experimental farm where other experiments have been conducted at the same time, data on intake, digestibility and grazing behaviour are available. A broad information is also available on the genetic traits of the Hestur sheep flock, as the same herd has been bred there since 1951.

This study was a part of the NKS / RAD-3 project in the 1990-1993 and it now forms a part of the NKS / EKO-2 project.

Deterministiske skader og dosisreduktion i interventionel neuroradiologi i Danmark.

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RESUME

Deterministiske skader i røntgendiagnostikken har igennem mange år været få. Med interventionsradiologien, der medfører meget lange gennemlysningstider og adskillige optagelsesserier, foreligger der oplysninger om skader fra flere lande. Der redegøres for de relevante skader, deres dosistærskler og latenstider.

Beregning af patientdoser i interventionsradiologien kompliceres af, at de moderne apparater er mikroprocessorstyrede og dermed giver mulighed for brugeren til at vælge mellem adskillige programmer til styring af højspænding, rørstrøm, filtrering m.m. Der bringes et eksempel på et sådant reguleringssystem.

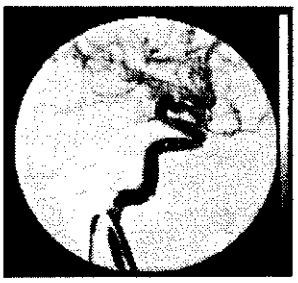
I Danmark har én røntgeninstallation givet anledning til to tilfælde af epilation (håraffald) i forbindelse med interventionel neuroradiologi. Der redegøres for de dosismålinger der er foretaget, de tiltag og anbefalinger der er gjort med henblik på at reducere doserne fremover og dermed forsøge at undgå de deterministiske skader.

INDLEDNING

Allerede kort tid efter at røntgenstrålingen var opdaget i 1895, blev man klar over, at den kunne forårsage hudskader. Specielt fik mange læger og forskere skader på deres hænder i de første år.
Et dansk udvalg, der forberedte den første strålebeskyttelseslovgivning, gør i en betænkning fra 1928 opmærksom på opståede røntgenbeskadigelser hos patienter, der kan være meget alvorlige, og som medfører erstatninger.
Forholdsvis enkle strålebeskyttelsesmæssige foranstaltninger, som filtrering af strålingen, stor fokus-hudafstand og indblænding af strålefeltet bevirker, at hudskaderne ikke opstår.
Men med indførelsen af den digitale billedbehandling accelererede interventionsradiologien med meget lange gennemlysningstider og optagelsesserier, og beretninger om deterministiske skader hos patienter som følge af de store doser dukker op. I Danmark har Statens Institut for Strålehygiejne kendskab til tre tilfælde af deterministiske skader i form af total epilation og desuden viden om flere tilfælde af delvis epilation, alle opstået i forbindelse med interventionel neuroradiologi.

Interventionel radiologi omfatter behandling af patienter gennem perkutan adgang uden åbning af kroppens kaviteter. Typisk vil det dreje sig om at lukke deformiteter i karsystemet eller skaffe gennemstrømning i forsnævrede kar. I interventionel neuroradiologi kan eksempelvis aneurismer i hjernens karsystem lukkes med såkaldte coils, der anbringes ved hjælp af et kateter under gennemlysning se figur 1 og 2. Proceduren omfatter også lange optagelsesserier.

Figur 1. Aneurisme fyldt med kontraststof.



Figur 2. Aneurisme lukket med coils og derfor ikke længere fyldt med kontrast.

Deterministiske stråleskader er karakteriserede ved, at skaden først opstår, når dosis overskrider en vis tærskel, samt at graden af skade stiger med stigende dosis. I figur 3 er angivet de skader som er aktuelle i interventionsradiologien samt tærskeldosis og latenstid. Tærskeldoserne afhænger af forskellige faktorer som fraktionering og hudtype hos den bestrålede person. Latenstiden vil være afhængig af dosis, således at en høj dosis giver en kort latenstid.

Skade	Tærskeldosis (Gy)	Latenstid
-Tidligt transitorisk erytem	2	timer
-Transitorisk epilation	3	3 uger
-Erytem	6	10 dage
-Permanent epilation	7	3 uger
-Tør dermatitis	10	4 uger
-Invasiv fibrose	10	--------
-Hudatrofi	11	>14 uger
-Telangiektasi	12	>52 uger
-Ekssudativ dermatitis	15	4 uger
-Sent optrædende erytem	15	6-10 uger
-Hudnekrose	18	>10 uger
-Sekundær ulceration	20	>6 uger

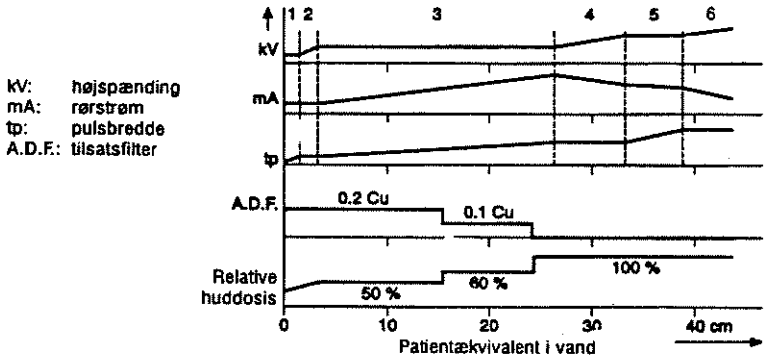
Figur 3. Deterministiske skader, tærskeldoser og latenstider ved interventionsradiologi.

Ved gennemlysning med traditionelt røntgenudstyr kan man normalt danne sig et skøn over størrelsesordenen af huddosis, idet de væsentligste parametre der varierer er højspændingen, rørstrømmen, gennemlysningstiden og fokus-hudafstanden. Noget tilsvarende gør sig gældende ved optagelser, enten dette er enkeltoptagelser eller serier.

Forholdene er væsentlig mere komplicerede ved moderne mikroprocessorstyrede røntgenapparater. Ud over at de ovennævnte parametre kan have en meget kompliceret sammenhæng styret af programmer, der kan vælges af brugeren, er det muligt at ændre

pulsbredde og filtrering. Visse apparater har mulighed for filtrering på over 0,4 mm kobber, hvilket naturligvis giver en huddosisreducerende effekt.

Et eksempel på et moderne reguleringssystem er vist på figur 4. Røntgenapparatet "måler" patientens transparens udtrykt i vandækvivalent, og parametrene indstilles på basis heraf. I område 1 forøges pulsbredden, mens rørstrøm og højspænding holdes på deres minimumsværdier. I område 2 stiger højspændingen, mens rørstrømmen stadig er på minimum og pulsbredden konstant. I område 3 holdes rørspændingen konstant, og pulsbredde samt rørstrøm stiger, ind til den maksimale rørbelastning nås. I område 4 stiger højspændingen, rørstrømmen falder og pulsbredden er konstant. I område 5 er højspændingen konstant, pulsbredden stiger, og rørstrømmen falder. Endelig stiger i område 6 højspændingen, mens rørstrømmen falder og pulsbredden holdes konstant. Endelig er det værd at bemærke, at der ved bestemte vandækvivalenter ændres på filtreringen.



Figur 4. Reguleringsprincip for røntgengeneratorer..

MATERIALE OG METODE

To af tilfældene med total epilation i forbindelse med neuroradiologisk intervention har været med det samme røntgenapparat. Der er foretaget en opmåling af ydelsen fra røntgenapparatet med henblik på at reducere doserne. Desuden er der udført en række patientdosismålinger for at se, hvor store de aktuelle huddoser er.

Det anvendte røntgengenerator er en Shimadzu HD 150-G 60, som ikke er mikroprocessorstyret. Røntgenrøret sidder under lejet på en Shimadzu MH-10 C bue. Dosisreguleringssystemet fungerede med fastholdt rørstrøm.

Til opmåling af apparatets ydelser blev anvendt et Capintec 172 dosimeter med et 30cc ionisationskammer og et Alderson hovedfantom. De målte ekspositioner blev omregnet til huddoser inklusiv tilbagespredning. Til måling af et antal patienters huddoser blev anvendt TLD i form af LiF, der blev tapet til patienternes hud, således at et dosimeter var i indgangsstrålefeltet ved både PA og LAT projektionen.

RESULTATER OG DISKUSSION

For at gøre det muligt for personalet, der udfører interventionerne, at foretage et skøn over patienternes huddoser, blev der foretaget opmålinger ved de mest almindelige serie- og gennemlysningsparametre, som det fremgår af figur 5 og 6. Desuden blev der opmålt ydelser med en totalfiltrering på 4 mm aluminium i stedet for de 3 mm aluminium, som målingerne i figur 5 og 6 var baseret på. Disse målinger viste en dosisreduktion på ca. 20%, og billedkvalitetsvurderinger viste efterfølgende, at billedkvaliteten stadig var acceptabel på trods af den hårdere strålekvalitet. Inden opmålingerne påbegyndtes, var der foretaget en reduktion af rørstrømmen på 20% med henblik på at reducere huddoserne. Som en yderligere dosisreducerende foranstaltning blev det anbefalet at øge fokus-hudafstanden, idet den anvendte apparatopstilling gav mulighed herfor.

Fokus-hudafstand (cm)	Serielængde (s)	Antal billeder	Højspænding (kV)	Rørstrøm (mA)	Huddosis (Gy)
56.5	10.5	40	74	80	0.20
56.5	7.6	29	74	80	0.15

Figur 5. Huddosis ved optagelsesserier

Fokus-hudafstand (cm)	Højspænding (kV)	Rørstrøm (mA)	Huddosishastighed (Gy/min)
56.5	88	0.9	0.03
56.5	73	2.0	0.05
56.5	69	3.0	0.05

Figur 6. Huddosishastighed ved gennemlysning

Efter indførelse af de dosisbesparende foranstaltninger blev der med TLD i indgangsstrålefeltet målt et antal huddoser på patienter. Ved de første 4 patienter blev der ikke opgjort serietider og gennemlysningstider separat for AP og LAT projektionerne. Huddoserne lå imidlertid i intervallet 0,027 til 1,1 Gy, hvilket er under tærskelværdien for deterministiske skader.

For de følgende 4 patienter er serielængde, gennemlysningstid og huddosis vist for både AP og LAT projektionen i figur 7. Det fremgår, at man for patient B's vedkommende er oppe på så høj en dosis, at der kan forventes en deterministisk skade. Dette er da også en interventionel arteriografi, men A og C er diagnostiske arteriografier. Selv efter indførelsen af de

dosisbesparende foranstaltninger, er der således stadig mulighed for at give patienterne høje doser.

Patient/projektion	Serielængde (s)	Gennemlysning (minut)	Huddosis (Gy)
A/PA	34.5	2.0	0.43
A/LAT	14.4	0.5	0.38
B/PA	167	27.2	3.1
B/LAT	37.3	3.0	1.2
C/PA	41.4	2.8	0.68
C/LAT	8.7	1.2	0.24

Figur 7: Huddosis målt på patienter med TLD.

Endelig er der i et røntgenrum på hospitalet med moderne udstyr, som normalt ikke anvendes til interventionel neuroradiologi, foretaget pilotmålinger, der indikerer, at patientdoserne kunne nedsættes ved interventionel neuroradiologi ved anskaffelse af nyt røntgenudstyr.

KONKLUSION

Ved interventionsradiologi, hvor der arbejdes med lange optagelsesserier og gennemlysningstider bør der anvendes mikroprocessorstyrede røntgengeneratorer med brugervalgte programmer, der giver mulighed for at optimere dosis og billedkvalitet. Herunder må filtreringen være en af de parametre, der kan optimeres på. Endelig må der udføres patientdosismålinger, der gør det muligt at vurdere huddoserne og dermed muligheden for deterministiske skader.

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Strålrisk vid mammografiscreening - försumbar eller ett stort problem?

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Sammanfattning - När helst mammografiscreeningens vara eller icke vara diskuteras kommer även strålrisken med in i bilden. Uppfattningar om dess betydelse har gått - och går - isär, från "helt utan betydelse" till "strålrisken överstiger nyttan". Orsaken till att dessa motstridiga slutsatser är olika antaganden om stråldos, strålrisk per dosenhet och nyttan med screening i form av minskad dödlighet i bröstcancer. I denna rapport diskuteras de senaste rönen om strålriskerna, bl a resultaten av den svenska fibroadenomatosstudien, och utvecklingen av stråldoserna under de senaste årtiondena. Beräkningarna visar, att, med realistiska antaganden avseende stråldos och risk, den förväntade nyttan med screeningen väl överstiger de skadliga effekterna, men att å andra sidan försumlighet av strålskyddstänkandet lätt kan leda till stråldoser som stjälper detta förhållandet.

Inledning

Främsta syftet med screening med mammografi för tidig upptäckt av bröstcancer är att minska dödligheten i sjukdomen och att drastiska terapeutiska ingrepp kan ersättas med mindre. Ett absolut krav för berättigandet av ett screeningprogram är att nyttan är större än de negativa effekterna. Bedömningen kan vara svår eftersom de flesta effekterna är svåra att kvantifiera och mäts dessutom ofta i olika enheter och är således svåra att jämföra.
Ett undantag är strålrisken. Den kan mätas i antal strålinducerade bröstcancerdödsfall eller förlorade levnadsår som kan ställas mot den förväntade nyttan av ett screeningprogram uttryckt i antalet förhindrade bröstcancerdödsfall eller vunna levnadsår. I denna artikel diskuteras just detta för olika screening scenarier med utgångspunkt från nya rön både vad gäller strålrisk, stråldos och effekten av screeningprogrammet i form av reduktion av bröstcancerdödlighet.

Bakgrund

Allmän screening.: Screening med mammografi för tidig upptäckt av bröstcancer har diskuterats - inte minst i Sverige - för mer än tjugo år sedan. Även om stråldoserna då var högre än de är idag var inte strålrisken främsta motargumentet utan bristen på beviset att screening för med sig medicinsk nytta. Den s k WE-studien visade med statistisk signifikans 1985 [1] att bröstcancerdödligheten kan minskas med mammografiscreening. Detta var startskottet för allmän screening med mammografi i Sverige, som i dag (augusti 1996) omfattar hela landet med undantag av Gotland.

Utveckling av stråldoser. På 60-talet användes direktfilm utan förstärkningsskärmar. Stråldosen per bild, uttryckt som genomsnittlig absorberad dos till bröstkörtelvävnad i ett normalt stort bröst, var då större än 20 mGy. På 70-talet kom de första film-skärm system, speciellt avsedda för mammografi, med en dos på 2 - 3 mGy. Tack vare vidareutvecklingen av dessa system med bl a speciellt anpassad framkallning kunde dosen pressa ner till ca 0,6 mGy i slutet av 80-talet. Då ökade å andra sidan användningen av raster, som tidigare endast användes för s k klinisk mammografi. Idag används raster nära nog till 100 procent vid screening med en dos per bild på ca 1,5 mGy.
I screeningsammanhang är förutom dos per bild också antalet bilder per undersökning och intervallet mellan två konsekutiva screeningomgångar av betydelse för dosen. I WE-studien användes en-bildsteknik, och det genomsnittliga intervallet mellan screeningomgångerna var 33 månader för kvinnor äldre än 50 år och 24 månader för de yngre. Socialstyrelsens allmänna råd

om mammografiscreening från 1986 [2] blev vägledande vid införandet av den allmänna screeningen 1986. Där rekommenderas en-bildsteknik för kvinnor över 54 år och två-bildsteknik vid initialscreening för de yngre kvinnorna. Bedömningen av bilderna avgör om de följande undersökningar ska göras med en- eller två-bildsteknik. Screeningintervallet rekommenderas till 18 månader för kvinnor under 55 år och till 24 månader för de äldre kvinnorna. Screening sker idag i regel enligt dessa rekommendationer.
Strålrisker. Kunskapen om risken för strålinducerad bröstcancer kommer främst från studier av de överlevande i Hiroshima och Nagasaki [3 4], från tuberkulospatienter som genomlystes med röntgen [5 6], och för patienter med benigna bröståkomma som strålbehandlades [7 8 9]. Speciellt i de tidigare publikationerna är resultaten för kvinnor som bestrålats i åldern > 40 inte statistiskt signifikant, och uppföljningstiderna är där relativ korta.

Nuläget och framtidstrender

Stråldos. Nya film-skärm system introduceras på marknaden. Preliminära utvärderingar [10] visar att den absorberade dosen till ett standardbröst kan sänkas till 0,8 mGy eller mindre med användning av raster under bibehållen bildkvalitet. De övriga trender som påverkar stråldosen går i motsatt riktning. Främst i Storbritannien och Nederländerna övergår man till allt högre filmsvärtning, från nettosvärtning mellan 1,0 och 1,4 till mellan 1,5 och 1,8, vilket motsvarar en dosökning mellan 50 och 100 procent. I ett snart publicerat dokument från EU om dosimetri för mammografi [11] "tillåts" värden upp till 4 mGy per bild vid nettosvärtning 1,8.
I diskussionen om och hur screening ska bedrivas för kvinnor i åldersgruppen 40 - 49 år framhålls att screningintervallet måste vara kortare än för äldre kvinnor för att uppnå önskad hälsoeffekt - ett år anses vara optimalt [12]. Och två-bildsteknik används mer och mer rutinmässigt i screeningsammanhång, speciellt för kvinnor under 50 år.

Strålrisk. Mattsson et al [9] citerar några av de mest relevanta studierna om strålrisk för kvinnor över 40 år vid bestrålningen och räknar om resultaten till samma enhet, excess bröstcancerrisk per 1 Gy. Dessa värden återges i tabell 1.

Tabell 1: Risk för strålinducerad bröstcancer från olika studier (ur ref [9])

Ålder vid exponering	Excess relativ risk per 1 Gy	referens
≥ 40 år	0,48	4
≥ 40 år	0,46	8
≥ 35 år	0,16	6
≥ 35 år	0,10	5

Mortalitetsreduktion. Ehuru den positiva effekten av hälsoundersökningar med mammografi med avseende på reduktion av bröstcancerdödligheten har kunnats verifieras för kvinnor i åldersgruppen 50 till 70 år med statistisk signifikans redan 1985 [1] finns fortfarande frågetecken för kvinnor under 50 år. En sammanslagning av samtliga randomiserade, populationsbaserade studier gav för 40 - 49 åringar en mortalitetsreduktion på 22 procent med ett konfidensintervall (95% nivå) 36 till 4 [13], dvs det är nätt och jämnt säkerställt att det överhuvudtaget finns en positiv effekt. Modellberäkningar, förutsatt ett screeningintervall på högst ett år, förutspår en mortalitetsreduktion på 36 procent [12].

Strålrisk kontra nyttan vid mammografiscreening.

Det finns inte något allmängiltigt svar på frågan "hur stort är förhållandet mellan nytta och risk vid mammografiscreening", även om man inskränker sig till att jämföra räddade liv genom tidig diagnos med antalet strålinducerade bröstcancerdödsfall. Det är många parametrar som ingår och de kan varierar mycket i olika screeningscenarier. Fem av dessa parameter listas i tabell 2, tillsammans med den spridning de kan ha under realistiska antaganden. Som framgår ur sista raden kan ratio benefit/risk skiljer sig med en faktor 450 mellan extremfallen.

Tabell 2: Variation av faktorer som påverkar benefit/risk ratio vid mammografiscreening

	min/max värde	Ratio
Dos per bild	0,8 ...4 mGy	5
Antalet bilder	1..2	2
Screeningintervall	3...1 år	3
Excess relativ risk / 1 Gy	0,1 ...0,48	5
Mortalitetsreduktion	30...10 %	3
Totala maximala relativa skillnaden		**450 !!**

För att belysa hur det kan slå i verkligheten har beräkningar genomförts för tre olika scenarier: ett "högrisk", ett "lågrisk" och ett "medelrisk", se tabell 3. Förutsättningar var att alla värden hamnar inom ett rimligt intervall, såsom beskrivet i föregående avsnitt. För "hög-" respektive "lågrisk" alternativet valdes de mest gynnsamma respektive ogynnsamma värden, för medelalternativet det som enligt författaren är realistiskt för screening i Sverige, förutsatt dock en hög ambitionsnivå, både vad gäller bildkvalitet som stråldos.

Tabell 3: Tre olika scenarier för mammografiscreening av 40 - 49 åriga kvinnor

	"högrisk"	"lågrisk"	"medelrisk"
Dos per bild	4 mGy	0,8 mGy	1 mGy
Antal bilder	2	1	1,3
Screeningintervall	1 år	3 år	1,5
Excess relat. risk/Gy	0,46	0,1	0,2
Mortalitetsreduktion	10 %	30 %	20 %

För de olika scenarier har antalet förhindrade respektive inducerade dödsfall samt antalet vunna respektive förlorade levnadsår beräknats. Beräkningarna gjordes av Anders Mattsson [14] som använde detaljerade data om åldersfördelningen av bröstcancerincidens och -mortalitet samt tidsberoendet däremellan för Stockholmsbefolkningen som underlag. Latenstiden för strålinducerad bröstcancer har antagits till att vara 5 - 10 år. Reduktionen i bröstcancer-dödlighet genom screening börjar för 40 - 49 åringar 7 år efter första screeningen och kvarstår livet ut. Tabell 4 visar resultaten. Jämfört med förhållandena för bröstcancerröd är benifit/risk

ratio för vunna/förlorade levnadsår cirka två gånger gynnsammare, beroende på att "livräddningen" inträffar långt tidigare än de strålinducerade cancerdödsfall.

Tabell 4: Benefit/risk förhållandet för mammografiscreening under 10 års tid av 100 000 40-åringar

	"högrisk"	"lågrisk"	"medelrisk"
A Räddade liv	60	190	125
B Förlorade liv	150	1,2	5,5
C Ratio A/B	0,4	160	22
D Vunna levnadsår	1500	4500	3000
E Förlorade levnadsår	2000	15	70
F Ratio D/E	0,75	300	40

Diskussion och slutsats

Ur tabell 3 framgår att det kan gå så illa att skadan uttryckt i antalet strålinducerade bröstcancerdödsfall eller förlorade levnadsår kan bli långt större än nyttan i motsvarande enheter. Om inte omöjligt så är dock detta scenario rätt så osannolikt att inträffa i verkligheten, åtminstone om det finns någon form av kvalitetstänkande bakom screeningprogrammet. Å andra sidan är också "glädjekalkylen" osannolik, så verkligheten ligger nog närmare den sista kolumnen.

Mer än tjugo gånger så många räddade liv än strålningsinducerade dödsfall, och mer än fyrtio gånger så många vunna levnadsår än förlorade borde kunna rättfärdiga ett screeningprogram, åtminstone om man endast tar hänsyn till denna aspekt. Marginalerna är dock inte stora. Med en fyra gånger högre stråldos - vilket lätt kan hända i ett program med t ex genomgående två bilder, screeningintervall på ett år och en dos per bild på 2 mGy - sjunker benefit/risk ratio till 5, och då är det inte självklart att ett sådant screeningprogram är etiskt försvarbart.

Sammanfattningsvis kan konstateras att strålrisken inte utgör något hinder för genomförandet av ett screeningprogram. Detta förutsätter dock att stråldosen inte väsentlig överstiger de värden som anges i "medelscenariet" - stråldosen får även i fortsättningen inte negligeras. Lika viktigt är det att uppnå och upprätthålla hög kvalitet i alla de andra aspekterna, som t ex diagnostisk säkerhet för att minimera antalet falsk positiva och falsk negativa diagnoser.

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Doses to the Staff in the Center for Positron Emission Tomography at the University Hospital in Copenhagen (RH).

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INTRODUCTION

This paper describes a survey of radiation and radiation levels in the center for Positron Emission Tomography (PET) scanning at the University Hospital in Copenhagen. The survey was conducted as a Master Thesis at the Department of Electrophysics (AEF) at the Technical University of Denmark (DTU), April 1995. The project was made in collaboration with the University Hospital in Copenhagen (RH) and the Danish Institute for Radiation Protection (SIS), with Uffe Korsbech (DTU) and Klaus Ennow (SIS) as supervisors.

In Denmark the radiation protection of workers is based on the recommendations of the International Commission on Radiological Protection (ICRP). According to the ICRP the use of X-ray apparatus and radiopharmaceuticals should follow the 3 general principles for the radiation protection of workers:

- the use is justified (the advantages must at leastcompensate the disadvantages)
- all doses should be kept as low as reasonably achievable with regards to economic and social conditions
- no workers should receive doses higher than the dose limits

The recommended dose limits for occupational exposure are: 20 mSv per year for the effective dose, 150 mSv to the lens of the eye, 500 mSv to the hands and feet and 500 mSv to the skin.

The investigation in the PET- center was initiated due to measurements of increased doses to the staff. In general, in Danish hospitals, film and pocket dosemeters are used for monitoring the doses to the employees. Thus, the staff at the PET center have worn dose measurement equipment to control their doses since the start in 1992.
A typical effective monthly dose to a laboratory technician during the start up of the center in 1993 (while making radioactive liquids for injection and while performing injection) was : 0.1-0.3 mSv, whereas the monthly dose to the skin was less than 0.1 mSv. By 1994 these doses had increased to 0.4 - 0.6 mSv and 0.3 - 0.8 mSv respectively. In addition stains were observed on the films, i.e. apparently the radioactivity had in some way settled on the films before decaying.

At the center the positron emitters are used as tracers. The elements N-13, O-15 and F-18 are attached to molecules that are part of the human metabolism. The PET-scanning is

based on the detection of the decay from these positron emitters. Certain conditions are of pratical importance for the radiation protection:

- the decay of the positon emitters gives complex radiation, fields i.e. β^+-radiation (positrons) as well as high energy γ -radiation (annihilation).

- the preparation procedure is complicated: After the elements have been produced in the cyclotron, several manual chemical procedures follow before the product is ready for injection.

- all of the positron emitters have relatively short half-lifes: 124 s - 2 h. Therefore high start activities are required, otherwise there will not be sufficient time for the scanning process.

MATERIAL AND METHODS

Different methods were used to determine the level of radiation doses to the employees: The Air Monitor, Smear Tests, The Pocket Dosemeter and Thermo-Luminescence Dosemeters.

The Air Monitor was used for monitoring the airborne radioactivity in the rooms of the PET-center during work with the pharmaceuticals marked with N-13, O-15 and F-18. By use of a pump, the air from the room was lead to a scintillation detector window. The detector signal was amplified and analysed by a counter. The count rate was converted into volume activity. To do this the system was calibrated first. The calibration was made by circulating a known concentration of radioactive CH_4 to the detector window. Doses to the sensitive part of the skin and effective doses were finally calculated from the measured volume activities using approximating formulae.

To determine the contamination of surfaces, Smear Tests were used. Four different ways of exposure from surface contamination were investigated; that is the external irradiation of skin from the surface contamination, inhalation of evaporated activity, external irradiation of skin from contamination transferred to the skin and finally from intake of surface contamination. A certain surface area was wiped off, and the activity was detected in a NaI-detector. The NaI detector was calibrated using a known concentration of F-18 activity.
The objective of the derived limits is to establish guidelines for which doses are acceptable. These guidelines ensure that the ICRP recommendations are not exceeded. The guidelines are calculated using knowledge of the Average Dose Rate Equivalent per Surface Contamination Unit, the Annual Limit on Intake and the Derived Air Concentration (the model is conservative i.e. it overestimates the dose).
A comparison between the measured surface contaminations, 0 - 4000 Bq/cm^2 , and the derived limits was then made.

The staff was furthermore asked to wear a pocket dosemeter (type 21/Stephen 6000, P.E.-medical Aps), which made it possible to follow the doses received in different handling

situations second by second. For example the preparation of O-15 (water) and the injection of F-18 (FDG) was monitored.

Finally TL materials was used for measurements of exposures of the staff when handling radiopharmaceuticals in different situations. First 7 measurements were performed by using the TL-meterial LiF-100 (the measured effective doses were corrected for the background contribution). Normally dosemeters of this kind are sufficient for strongly penetrating radiation. For weakly penetrating radiation, problems arise, because the detector can not differentiate between the dose absorbed in different layers of the detector. In this situation a detector made of a very thin tissue equivalent thermo luminescent (TL) material, covered with a thin tissue equivalent filter is a more appropriate solution.
To be able to differentiate between weakly and strongly penetrating radiation, a new dosemeter made from LiF:Mg,Cu,P and MgB_4O_7:Dy was developed. Two LiF:Mg,Cu,P chips were placed in a plastic pocket on each side of a 3mm perspex plate.
The diameter was 8 mm and it shielded the back chip against beta radiation. A MgB_4O_7:Dy pellet was placed in the pocket next to the LiF:Mg,Cu,P chips. The plastic material of the pocket was non transparent. The purpose of the pocket was to hold the tablets, to function as detector window and to shield against UV-light. UV-light would affect the read out of the tablet.
In the new dosemeter the upper LiF:Mg,Cu,P chip received both beta and gamma radiation while the lower LiF:Mg,Cu,P chip only received gamma radiation. The MgB_4O_7:Dy pellet received both beta and gamma radiation. This pellet was used as a control of the dose measured with the LiF:Mg,Cu,P chips.
A new serie of measurements was finally performed with the new dosemeter.

RESULTS
The effective doses due to airborne radioactivitity, measured by using the Air Monitor were found to be negligible. The highest measured doses to the sensitive skin was found in situations handling H_2O (O-15), 0 - 8 μSv, whereas the sensitive skin doses the staff received when handling N-13 or F-18 were all negligible. None of the dose levels were critical compared to the recommendations from the ICRP.
(Since the investigation at the center, the staff have started using C-11 (^{11}CO gas). Permanent use of this gas may increase the amount of airborne radioactivity in the center, so new monitoring with the Air Monitor could be necessary).

A comparison between the measured surface contamination and the dirived limits gives the following results:

- The doses from external irradiation of the skin are of the same order of magnitude as the calculated derived limit $DL_{surface\ to\ skin}$ = 147 Bq/cm^2 and $DL_{skin\ cont.\ to\ skin}$ = 340 Bq/cm^2 respectively.

- No important surface contamination regarding inhalation or intake was registered. The limits are here: $DL_{inhalation}$ = 140000 Bq/cm^2 and $DL_{intake,\ N\text{-}13}$ = 28000 Bq/cm^2 and $DL_{intake,\ F\text{-}18}$ = 80000 Bq/cm^2

- Typical contaminations are:

Transport containers: 40-200 Bq/cm^2
The Dose Calibrator: 500 Bq/cm^2
The handle of the LAF-closet: 4000 Bq/cm^2
Work tables: 0 - 300 Bq/cm^2

not contaminated is:
the handle of the HOT-cell-closet: 0 Bq/cm^2
and transportation wagons: 0 Bq/cm^2

The results obtained by measuring doses with the pocket dosemeter is that during the preparation of O-15 the highest dose comes from the trial run and the quality tests: ca. 1.5 μSv. Each production gives a dose of ca. 0.8 μSv. From the dose for the total preparation: 5.4 μSv, the average dose rate was calculated: 5.4μSv * (3600/4500)/h = 4.3 μSv /h. So one person can work 4651 ~ 4600 hours with the O-15 preparation each year without exceeding the limit.
While, as another example, during the injection of FDG the higest momentary dose comes during filling of the injector: 5.3 μSv, the injection itself gives 2.4 μSv. It is remarkable that staying near by the patient after injection gives as much as 9.7 μSv.
None of the situations measured with the pocket dosemeter indicated that workers would receive doses that, during a working year, would exceed the ICRP recommendations.

The effective doses to the staff in different handling situations, measured by use of the TL material LiF-100, are all in the range of 0-2 mSv.
In general the beta doses found by use of the New Dosemeter, i.e. by use of the TL-materials LiF:Mg,Cu,P and MgB_4O_7:Dy, are three times higher than the gamma doses for different handling situations, i.e. the need for a dosemeter measuring the beta dose was confirmed.

CONCLUSION

The conclusion concerning the radiation levels in the PET-center is that the protection arrangements at the center, for the present, limit the doses sufficiently compared to the ICRP recommendations. But there is a need for a better protection against the dose to the sensitive layer of the skin - especially to the hands. As cheap solutions, the staff could use thicker plastic gloves and introduce the use of tongs.
The staff could when possible use both hands (instead of only the right) while preparing or injecting the positron emitters. Finally the staff could when possibly try to spread the dose more evenly among several individuals.

A more expensive solution would be to make the preparation, as well as the injection, more automatic.

Abstract
This paper describes a survey of matters concerning radiation levels in the center for Positron Emission Tomography (PET) scanning at the University Hospital in Copenhagen. Since the launch of the PET center in 1992, by means of film and pocket dosemeters, an incresing level of radiation to the employees has been observed. Within the survey, the sources of this increasing radiation level are investigated. While making radiopharmaceuticals for injection, as part of the preparation for the scanning process, the employees inevitably receive radiation doses. Futhermore, while performing injections of radioactive preparations to the patients, the employees are exposed.

A survey of matters concerning air and surface contaminations in the center, has been conducted. For the purpose of monitoring the air contamination a dedicated Air Monitor was constructed. The surface contamination was measured using Smear Tests. Doses as well as doserates were registed by means of a Pocket Dosemeter and analyzed at times of production and injection of the radiopharmaceuticals. Doses to the fingers and hands of the personel, during handling of radiopharmaceuticals, were investigated using thermo-luminous materials. A finger dosemeter, that takes into account both the contributions from beta and gamma radiation, was developed.

The doses that were measured, were compared to the recommendations concerning protection against ionizing radiation by the "The Commission on Radiological Protection". On these grounds, it has been assessed, if there in certain working situation is a need for improved radiation protection. In this paper proposal have been made, where to act, and which measures can be taken in order to reduce the radiation doses, the personnel is exposed to.

Uppföljning av datortomografianvändningen i Sverige

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Sammanfattning - Statens strålskyddsinstitut samlade 1991 in uppgifter om användningen av datortomografi i svensk sjukvård. Dessa kompletterades sedan med dosmätningar och dosberäkningar. Resultatet från denna kartläggning redovisades bl.a. vid Nordiska Sällskapet för Strålskydds förra möte i Kristiansand i Norge 1993.

Slutsatserna från kartläggningen ledde till att Statens strålskyddsinstitutet bl.a. efterlyste en mer individuell anpassning av exponeringsparametrar till kroppsstorlek, undersökt område m.m.

Under våren 1996 har en uppföljning gjorts med en enkät om maskinpark, antalet undersökningar och undersökningsteknik. Svaren visar en fortsatt ökning av datortomografianvändningen både i antalet tomografer som nu är 112 st och antalet undersökningar som nära nog fördubblats sedan den förra kartläggningen. Vad gäller undersökningsteknik kan skönjas att införandet av anpassade undersökningstekniker kommit igång såsom sänkning av doserna vid undersökning av barn och i övrigt göra skillnad för skillnader i kroppsstorlek och anatomi.

Inledning

Datortomografin är den enskilda teknik inom röntgendiagnostiken som ger det ojämförligt största bidraget till kollektivdosen. Det är därför viktigt att följa utvecklingen inom detta område. SSI gjorde för 5 år sedan en kartläggning av datortomografiverksamheten i Sverige kallat *ACTRIS*[1]. Resultatet från denna kartläggning har presenterats vid Nordiska Sällskapet för Strålskydds möte i Kristiansand i Norge för 3 år sedan[2], vid en EU-workshop i Grado, Italien 1993, vid ett par tillfällen på Svenska läkarsällskapets årsstämma i Stockholm[3,4], vid ungerska strålskyddssällskapets 20:e workshop[5] samt i olika publikationer.

Denna kartläggning bestod sammanfattningsvis av följande. Under en "normalvecka" samlades data in för samtliga undersökningar från 89 av landets datortomografer vilket gav 3.881 protokoll. Dessa protokoll omfattade patientdata och exponeringsparametrar. Detta kompletterades sedan med dosmätningar i fantom så att patientdosen för varje enskild undersökning kunde beräknas enligt en egen enkel dosberäkningsmodell kallad *practical approach*[6].

Sammanställning av kartläggningen visade att det 1991 gjordes ca 200.000 undersökningar, vilket då motsvarade ca 4 % av det totala antalet från röntgendiagnostik. Kollektivdosen uppskattades till ca 1.000 manSv, vilket var ca 20 % av den totala från röntgendiagnostik. Sammanställningen visade att stora skillnader i doser fanns mellan olika användare och slutsatsen blev att exponeringsparametrarna ej var optimerade och individuellt anpassade till patientstorlek, anatomi och snittjocklekar.

Projektet innefattade även respons till användarna, där varje användare erhöll ett antal dosdiagram för några undersökningstyper med den egen placering relativt andra användare. En SSI-rapport[7] gavs också ut, där bristen på optimering presenterades men med rekommendationer om dosbesparingsåtgärder tillsammans med en lathund för att på ett enkelt sätt kunna göra egna dosuppskattningar.

Uppföljning

För att se om de efterlysningar om optimering givit några resultat och för att följa den fortsatta utvecklingen gjordes en uppföljning våren 1996. Med hjälp av en enkät till alla användare begärdes information in enligt följande:

- En inventering av apparatparken, dvs vilket nytillskott som kommit i form av utbytta apparater och nytillkomna användare samt vilka apparater som tagits ur drift.
- Antalet undersökningar för 1995 som utförts vid varje tomograf.
- Vilka undersökningsparametrar som används för några vanliga undersökningstyper samt hur dessa ändras med patientens storlek och anatomi.

Det gjordes dessutom kompletterande dosmätningar i fantom för de nya apparattyper som introducerats samt nya dosberäkningar för de angivna standardundersökningarna.

Svarsfrekvensen har tyvärr varit dålig och efter påstötningar har i dagsläget bara 2/3 av användarna svarat. Det kan därför bli vissa justeringar men förmodligen inte så stora då de som svarat representerar alla kategorier av sjukhus som universitetssjukhus, länssjukhus, länsdelssjukhus och privata sjukhus.

Följande slutsatser har kunnat göras:

- antalet datortomograferna ökat med ca 25 %.
- antalet undersökningar per apparat har ökat med ca 50 %.

Detta innebär att antalet undersökningar 1995 var ca 350.000

Patientdoserna

Äldre tomografer med färre detektorer har bytts ut och ersatts med nyare med fler detektorer vilket har medfört högre doser till patienterna. (p.g.a. brus-skäl krävs högre dos för tomografer med fler detektorer - vilket naturligtvis ger bättre upplösning).

Däremot har vi sett att många av de som använder samma apparat som vid förra kartläggningen tagit intryck av våra synpunkter och sänkt sina doser för samma typ av undersökning. Men även många av de nya användarna har också anammat anpassade exponeringsparametrar.

	Antal	Förändring i medeleff.dos	Intervall
Utbytta tomografer	33	+ 80 %	-40 / + 650 %
Samma tomograf	56	- 20 %	-60 / +70 %
Nya användare	23	-	
Summa	112	+ 20 %	

Tabell 1. Förändringar i medeleffektivdoser.

Om man antar att utvecklingen är den samma för de som ej svarat och att den relativa fördelningen av olika undersökningstyper är som den var 1991, ger detta att kollektivdosen för 1995 var ca 2.000 manSv.

Det bör dock påpekas att det finns vissa indikationer på att ökningen av antalet undersökningar inte skett proportionellt utan att bål-, thorax, och framför allt ryggundersökningar ökat relativt mer än skallundersökningarna. Detta kan förklaras av att CT-tekniken infördes som en skallundersökningsteknik och ersatte först de konventionella undersökningarna av skallen innan den infördes som helkroppsscanner.

Minskning av mAs	Andel '91	Andel '95
Vuxna / Barn	< 10 %	65 %
Thorax / Buk		40 %
Smala / Tjocka		85 %

Tabell 2. Vi ser här hur införandet av individuellt anpassade exponeringsparametrar har slagit igenom.

Om inga dosbesparingsåtgärder vidtagits, dvs de som sänkt sina doser skulle ha kört sina undersökningar som tidigare kan kollektivdosen uppskattas till över 2.200 manSv. Detta innebär att åtgärderna bidragit till en besparing på ca 200 manSv d.v.s. 10%. Det bör dock påpekas att i denna uppskattning inte räknat in besparingar beroende på patientstorlek vilket skulle ge en högre siffra på besparingarna.

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Metode til separation af gennemlysnings- og fotograferingsdosis ved arealdosis målinger på patienter.

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Resume

I takt med den stigende brug af arealdosimetre til kvalitetskontrol på hospitaler, er det ofte nødvendigt at kunne skelne mellem dosis fra gennemlysning og fotografering. Separationsprocesen er triviel når der kun benyttes almindelige filmkassetter. Ved brug af spotfilm og digitalteknik er problemstillingen en anden, idet arealdosis pr. tidsenhed fra henholdsvis fotografering og gennemlysning, er af omtrent samme størrelse.

Ved hjælp af en computer kan data fra arealdosimeteret opsamles hvert sekund, derved fås en kurve over arealdosis pr. tidsenhed (mGy x cm²/s). Disse data sorteres efter størrelse. Computeren analyserer nu de sorterede data, og finder herved det sted, hvor ændringen i arealdosis pr. tidsenhed er særlig stor. Dette punkt repræsenterer overgangen fra gennemlysning til fotografering. Niveauet for hvor stor ændringen skal være, bestemmes ud fra målinger på et antal patienter, hvor antallet af billeder er kendte.

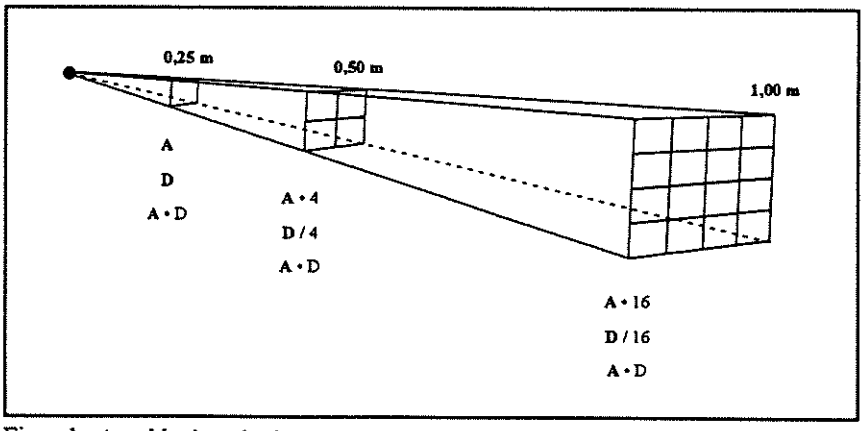
De data der er benyttet til at teste den nuværende version af metoden er indsamlet på fire danske hospitaler. Der er indsamlet data fra både colon og ventrikel undersøgelser. Ialt er der på nuværende tidspunkt analyseret data fra ca. 40 undersøgelser hvor der indgår fotografering og gennemlysning. Data er indhentet i forbindelse med et projekt hvis formål, er at kortlægge arealdoserne fra seks udvalgte undersøgelser.

Det har vist sig at metoden er istand til at angive antallet af billeder rimelig nøjagtigt. I de tilfælde, hvor det beregnede antal billeder er sammenlignet med det antal man visuelt kan bestemme ud fra hastighedskurven, er den maksimale afvigelse ± 1 billede.

Alt tyder på, at man langt nemmere kan bestemme antallet af billeder ved en kompleks undersøgelse hvis instrumentet kan samples hurtigere (op til 200 gange pr. sekund). Dette er i øjeblikket ved at blive undersøgt.

Indledning

Ved optimering af patientdoser ved røntgenundersøgelser på hospitaler, er det nødvendigt at kende bidraget til dosis fra både fotografering og gennemlysning. I stedet for TLD tabletter kan man benytte et arealdosismeter der er enkelt at anvende, og som tillader aflæsning af den målte størrelse umiddelbart efter undersøgelsens afslutning. Princippet ved måling af arealdosis er vist på figur 1. Arealdosis måles normalt i enheden (mGy·cm²).



Figur 1. *Arealdosis princippet.*

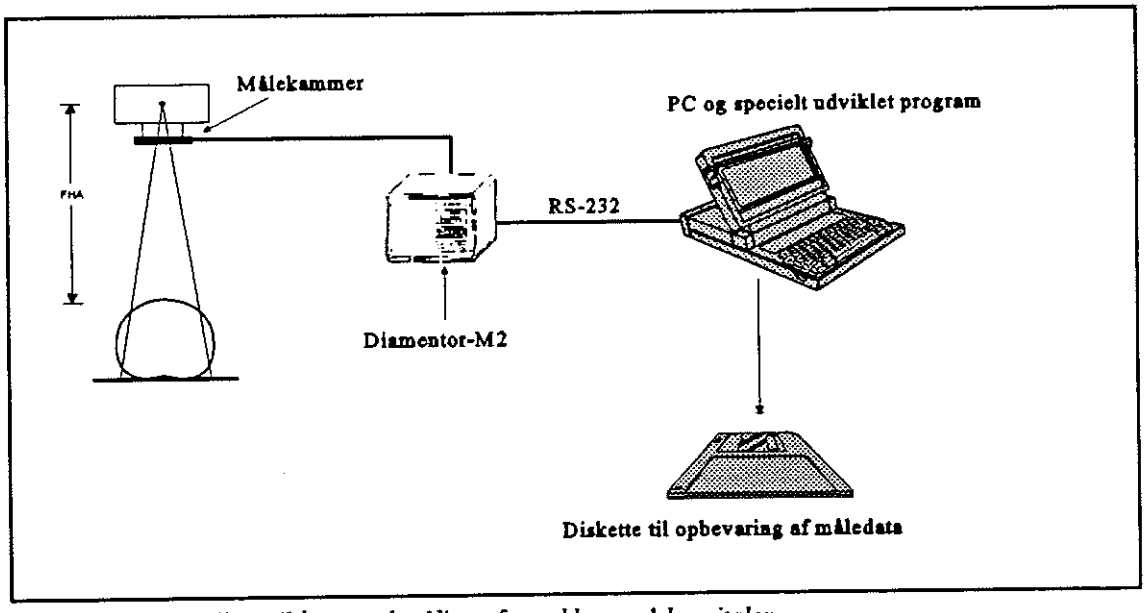
Som det ses er den målte arealdosis uafhængig af den benyttede FHA og den egner sig derfor godt til at sammenligne målinger på flere patienter. Når man har målt den gennemsnitlige arealdosis fra et antal patienter kan denne dosis sammenlignes med et sæt referencedoser der er vedtaget på nordisk plan. Hvis den gennemsnitlige dosis er væsentligt lavere eller højere end referencedosis for den pågældende undersøgelse må årsagen findes, og om nødvendigt må undersøgelsesteknik eller apparatur ændres. Af faktorer der har indflydelse på arealdosis kan nævnes:

- Film/folie systemets hastighed.
- Antal billeder.
- Gennemlysningstid.
- Benytte kVp.
- Indblænding.
- Personalets erfaring.

I det efterfølgende beskrives en metode til at bestemme den tid der gennemlyses og den hertil knyttede arealdosis.

Udstyr og målemetoder

Til måling af arealdoser benyttes et arealdosimeter fra PTW Freiburg af typen Diameter M2. Dette instrument kan styres fra en PC ved brug af en seriel kommunikations udgang (RS-232). Måleopstillingen er vist på figuren nedenfor (figur 2).
Til styring af måleinstrumentet og til opsamling af patient- og måledata, er der udviklet et



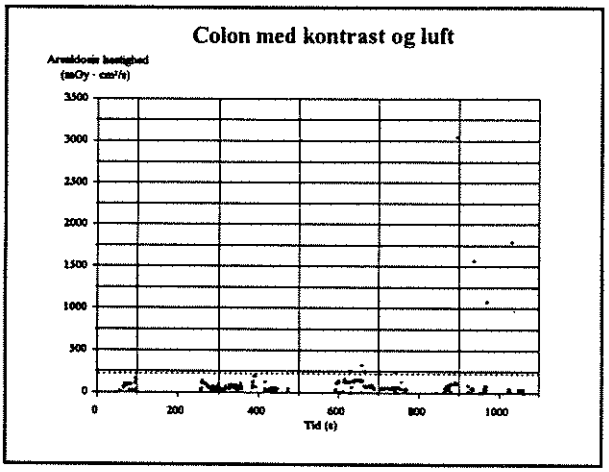
Figur 2. *Måleopstilling til brug ved måling af arealdoser på hospitaler.*

program. Programmet gør det muligt af indtaste patient data og starte en måling. Der er lagt stor vægt på at gøre programmet så brugervenligt som muligt, så det personale på røntgenafdelingen der udfører undersøgelsen selv kan betjene udstyret. Programmet er designet som et vindues system hvor brugeren indtaster data ved brug af mus og tastatur. Programmet aftaster arealdosimetret hver sekund. Ændringen i arealdosis samt det relative tidspunkt skrives til en datafil, der senere kan benyttes som input til beregningsprogrammet. Alle oplysninger gemmes i en database af typen Microsoft Access. Når apparaturet flyttes fra en afdeling til en anden sørger

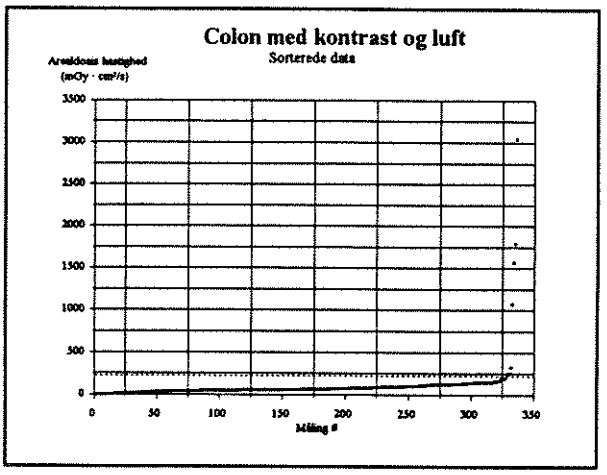
en automatiseret rutine for at gemme data på en diskette og derefter klargøre systemet til at måle på den næste afdeling. Data på disketten overføres på Statens Institut for Strålehygiejne til en PC hvor resten af databehandlingen finder sted.

Beregningsmodellen

Næste skridt er at bestemme bidraget fra henholdsvis fotografering og en evt. gennemlysning. Hvis der til undersøgelsen kun benyttes almindelige film er denne proces triviel, men hvis der benyttes digitalteknik eller spotfilm, er problemstillingen en anden p.g.a den lavere dosis disse typer kræver. På figur 3 er vist en undersøgelse hvori der indgår både gennemlysning og fotografering med normale film og med spotfilm. Eksemplet er en colon undersøgelse med kontrast og luft.



Figur 3. *Eksempel på en colon-undersøgelse med indhældning af kontrast og luft.*



Figur 4. *Data ordnet efter stigende arealdosis-hastighed. Mindste værdi først.*

Som det ses af figur 3, er der foretaget fire eksponeringer med normale film og fire eksponeringer med spotfilm. Disse eksponeringer er de målinger, der ligger over den stiplede linie. Men som det også fremgår af figuren er grænsen mellem gennemlysning og fotografering meget smal. Det er denne grænse det udviklede beregningsprogram finder.

Analyseprogrammet er opbygget af følgende elementer:

- Indlæsning af data fra disk.
- Sortering og differentiering af måledata.
- Bestemmelse af grænsen mellem gennemlysning og fotografering ud fra de behandlede data.
- Udregning af arealdoser fra gennemlysning og fotografering.
- Opdatering af database.

Sorteringen af data foregår v.h.a en implementeret QuickSort algoritme. QuickSort algotitmen er en optimeret sorteringsalgoritme. Resultat af denne sortering er en tabel, der indeholder data sorteret efter arealdosishastigheden. Mindste værdi først. De sorterede data fra figur 3 er vist grafisk på figur 4.

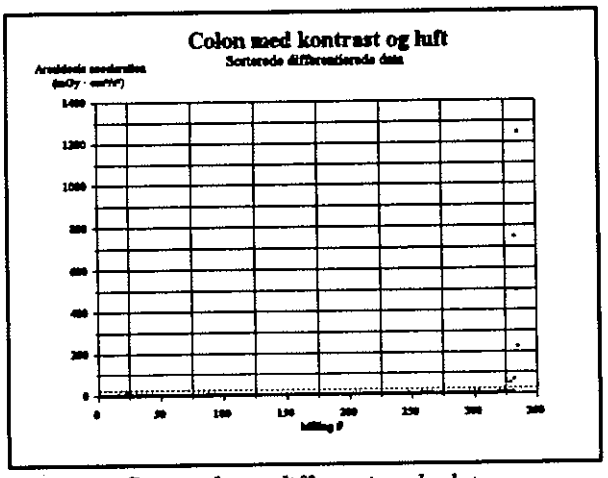
Efter sorteringen bliver data differentieret efter følgende formel:

$$Diff(i) = Sort(i) - Sort(i-1), \qquad \text{for } i = 2 \text{ til antal målinger}$$

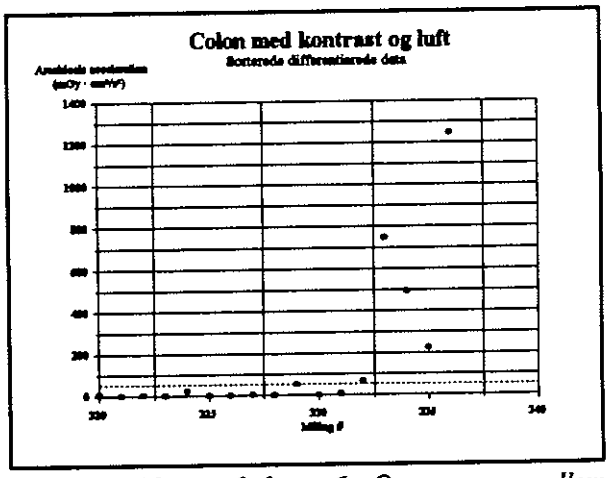
Resultatet af denne differentiering er vist på figur 5 og figur 6.

For at bestemme de målinger hvor der er foretaget en eksponering, udregnes variansen af et antal målinger. Antallet af målinger der beregnes ud fra, kaldes vinduet. Udregning af variansen for n målinger, sker ud fra følgende formel:

$$\sigma^2 = \frac{1}{n} \cdot \sum_{i=1}^{n} (x_i - \bar{x})^2$$

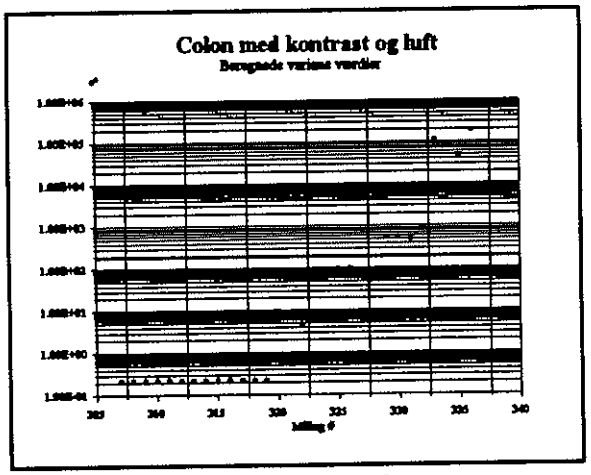


Figur 5. *Sorterede og differentierede data.*



Figur 6. *Udsnit af figur 6. Overgangen mellem gennemlysning og fotografering.*

Hvor x_i er et målepunkt indenfor det betragtede vindue og $\bar{x}$ er middelværdien af vinduets målepunkter. Vinduets bredde (n) er ved gentagne afprøvninger fundet til optimalt at være 3 målepunkter.



Figur 7. *Beregnede variansværdier.*

Vinduet flyttes fra de højeste værdier og nedad. Når den beregnede varians (vist på figur 7) bliver mindre end en fastsat grænse, som i vores tilfælde impirisk er bestemt til 200 (vist med en kraftig streng på figur 7), og når den har holdt sig under denne grænse i et antal flytninger af vinduet (f.eks 20), er grænsen mellem gennemlysning og fotografering fundet.

Når man nu kender en værdi for overgangen mellem gennemlysning og fotografering er det en triviel sag at bestemme de værdier der ligger over denne værdi.

De beregnede værdier opdateres i databasen og programmet er klar til at beregne endnu en undersøgelse.

Resultater og diskussion

Beregningsmodellen er indtil nu blevet afprøvet på ca. 40 colon og ventrikel undersøgelser. Antallet af billeder beregnet af programmet, er blevet sammenlignet med antallet billeder der rent visuelt kan bestemmes. Resultatet er at usikkerheden på denne sammenligning er på ± 1 billede. Denne nøjagtighed er tilstrækkelig til at foretage en optimering af patientdoserne på de danske hospitaler.

Statens Institut for Strålehygiejne, foretager i øjeblikket en estimering af den danske befolknings kollektive dosis, hidrørende fra røntgenundersøgelser. Til dette formål benyttes et arealdosimeter fra Gammex - RMI. Dette arealdosimeter er istand til at sample op til 200 gange per sekund. Alt tyder nemlig på, at samplingshastigheden er afgørende for, hvor simpel en genkendelsesalgoritme kan gøres.

Referencer

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Analyse av data fra arealdosemålinger i röntgendiagnostikk

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Sammenfatning - Metoder for analyse av data fra arealdosemålinger i röntgendiagnostikk beskrives. Analysen er for en stor del automatisert og består blant annet av en separasjon av fotografi fra gjennomlysning. Separasjonen baseres på et tidsbilde av röntgenundersökelsen (arealdose per tidsenhet), og fotografier skjelnes fra gjennomlysning ved en korrelasjon med en kassebölgefunksjon med samme bredde som den fotografiske topp. En benytter også fordelingen av arealdosene til et estimat av sannsynligheten for spesielt höye pasientdoser.

Innledning

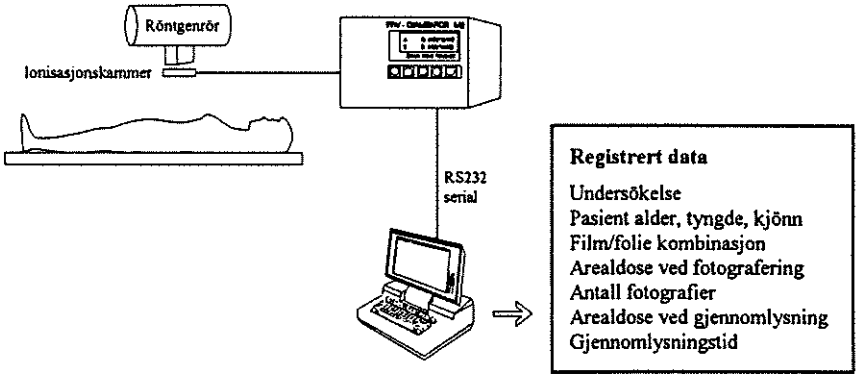
Et landsomfattende program med målinger av arealdose i röntgendiagnostikk har pågått i Island siden 1994. Data fra undersökelser av mere enn 2000 pasienter fra 10 forskjellige röntgenavdelinger fordelt på 40 undersökelser foreligger. Målingene skal benyttes som en del av et större program for å anslå bidraget til den kollektive strålingsdose fra medisink virksomhet, hvor den effektive dose beregnes fra målingene med omregningsfaktorer utviklet ved NRPB [1].

Målingene kan i tillegg benyttes til å vurdere de enkelte röntgenavdelinger i forhold til et landsgjennomsnitt [2], og et utvalg av undersökelsene kan også vurderes med hensyn til nordiske anbefalinger om stråledose i röntgendiagnostikk [3]. Slike sammenligninger bör imidlertid komplementeres med en analyse av faktorer av betydning for stråledosen, så som rörspenning, film-folie kombinasjon og pasient tyngde. En separasjon av arealdose fra gjennomlysning og fotografi, samt antall fotografier vil være spesielt nyttig.

For noen strålingstunge undersökelser, så som intervensjonell neuroradiologi, er det av interesse å vurdere virksomheten med hensyn til sannsynlighet for at deterministiske skader skal oppstå. Ved å forutsette at pasientdosene fordeles log-normalt, kan sannsynligheten for at en spesifikk stråledose overstiges, beregnes på grunnlag av et begrenset antall målinger.

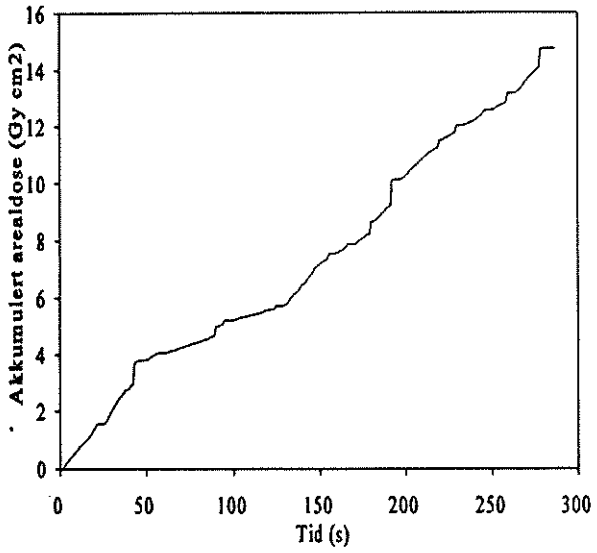
Metode

Målingene er utfört med PTW Diamentor M2 elektrometer med serielt grensesnitt til dator og med mulighet for tilkobling av to transmisjonskammere. Måleoppstilling er vist i figur 1.

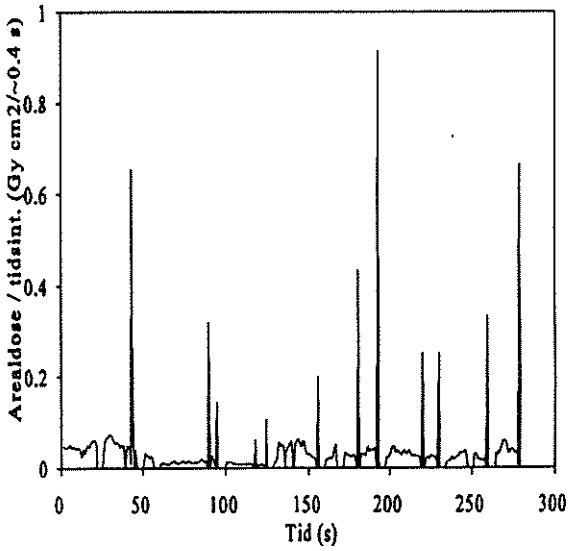


Figur 1. Måleoppstilling ved arealdosemålinger.

Datakommunikasjonen er basert på "handshaking", dvs. datoren sender en karakter som besvares av elektrometeret ved transmisjon av de akkumulerte arealdosene fra de to röntgenrörene. Elektrometeret er ikke nullstilt mellom transmisjonene slik at et plott av data som en funksjon av tiden fra undersökelsens start, vil få en form som vist i figur 2. En kan her se fotografier som områder i plottet med öket gradient, og identifikasjon av fotografi og gjennomlysning kan baseres på den antatte forskjell i eksposisjonsrate mellom de to teknikker. Det er imdlertid en begrensing av hvor ofte en kan få overfört data, og elektrometeret tåler ikke en frekvens over 2.5 transmisjoner i sekundet.



Figur 2. Akkumulert arealdose som funksjon av tiden fra undersökelsens begynnelse for en ventrikkel dobbeltkontrast undersökelse.



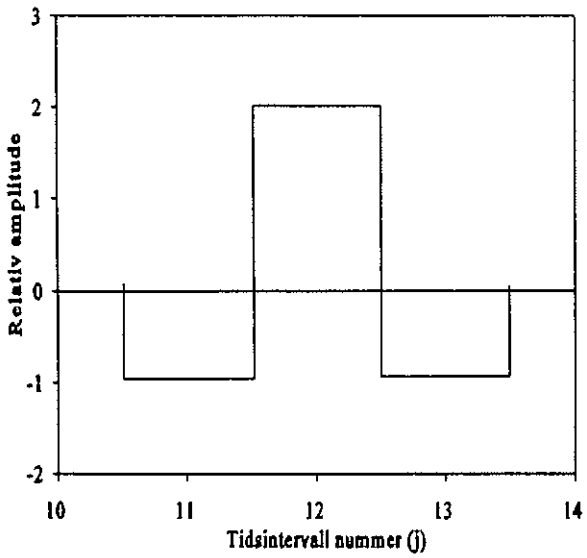
Figur 3. Arealdose for hvert tidsintervall for samme undersökelse som i figur 2.

For å lette den videre diskusjon overföres plottet i figur 2 til et tidsbilde, dvs. den akkumulerte arealdose mellom hver datatransmisjon plottes som en funksjon av tiden fra undersökelsens begynnelse, se figur 3. Fotografiene vil her komme fram som topper over gjennomlysningen. Imidlertid kan også sees at begrensingen i transmisjonsfrekvensen medförer at korte eksponeringer, mindre enn 0.1 sekunds varighet, og med liten eksposisjonsrate vil få en tilsvarende amplitude som i gjennomlysning. Den mest nærliggende metode for å skille mellom fotografi og gjennomlysning, å definere et nivå slik at amplituder overstigende nivået klassifiseres som fotografi, kan derfor ikke umiddelbart benyttes. Imidlertid kan forskjellen mellom fotografiske amplituder og gjennomlysning ökes med följende metode, hvor en benytter den forskjellige form på et fotografi og gjennomlysning. Tilsvarende metode har vært benyttet til lokalisering av topper i et gammaspektrum [4]. Mens gjennomlysning gjerne strekker seg over flere tidsintervall, begrenses et fotografi seg til et intervall. Ved å multiplisere verdiene i hvert intervall med faktoren

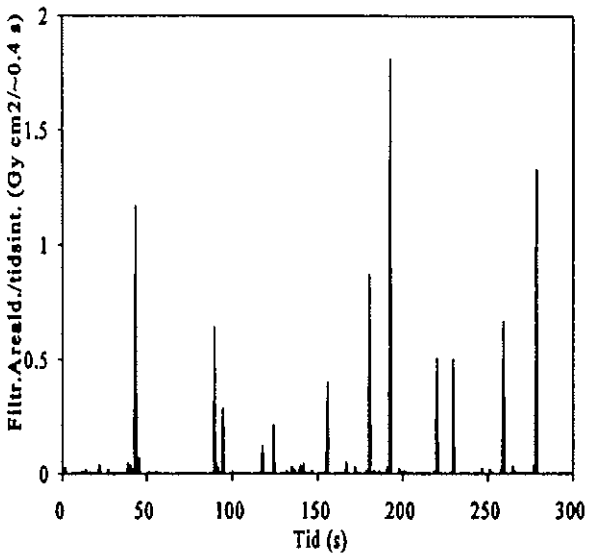
$$c_j = 2 \cdot y_j - y_{j-1} - y_{j+1}$$

hvor y_j er amplituden i intervall nummer j, se figur 4, transformeres plottet i figur 3 til figur 5. En automatisk prosedyre som legger inn et nivå som adskiller gjennomlysning og fotografi vil nå være mere robust dersom den baseres på verdiene i figur 5 enn i figur 3. Et nivå lik

gjennomsnittsverdien pluss et halvt standardavvik av de opprinnelige verdier, dvs. fra figur 3, har f.eks. vist seg å fungere tilfredsstillende for undersökelser inneholdende både gjennomlysning og fotografi.



Figur 4. Kassebölgefunksjon benyttet som filter til å öke fotografiske amplituder.



Figur 5. Plott av undersökelsen fra figur 3, hvor forskjell i amplitude fra gjennomlysning og fotografi er økt ved å benytte filteret fra figur 4.

Resultater og diskusjon

I figur 6 og 7 er vist eksempler på fordelingen av arealdoser fra to undersökelser, thorax og colon dobbeltkontrast. Fordelingene er basert på det totale antall målte undersökelser fra alle hospitaler. En kan spesielt legge merke til den betydelige del av arealdosen som fremkommer fra gjennomlysning, nesten 70%, for colon dobbeltkontrast, som viser at dosereduksjon ved en forbedret teknikk er mulig for denne undersökelse.

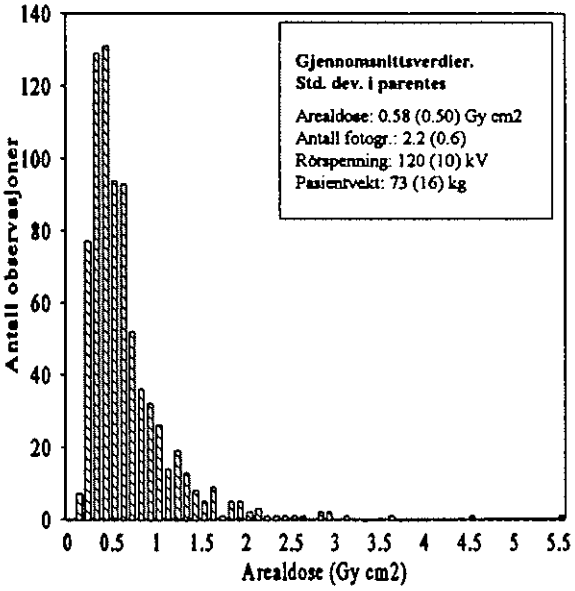


Figure 6. Fordeling av thoraxundersökelser, 774 observasjoner fra alle hospitaler.

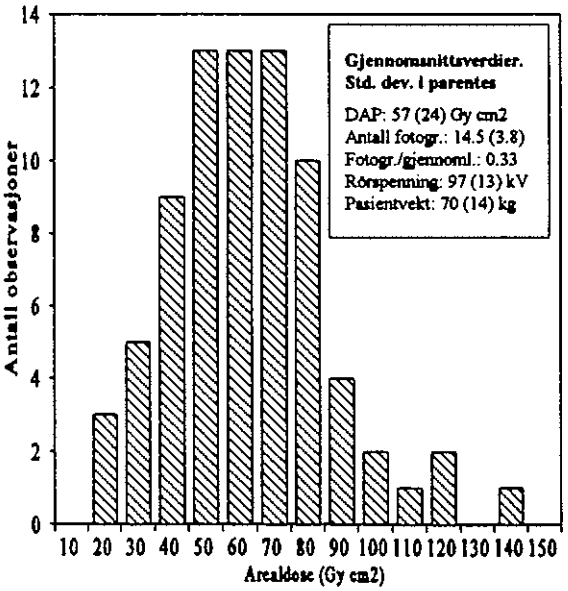
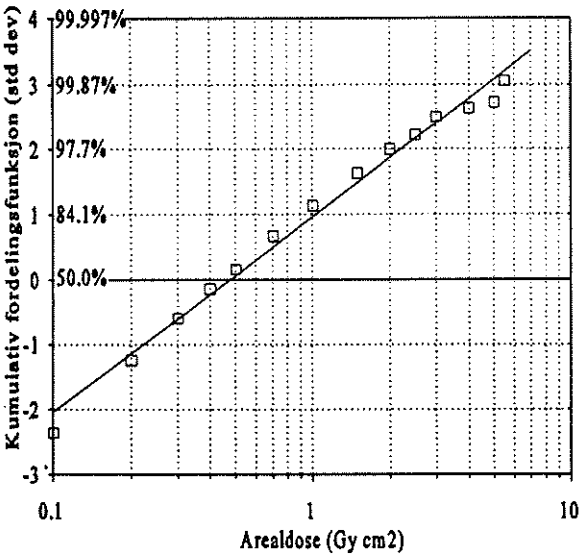
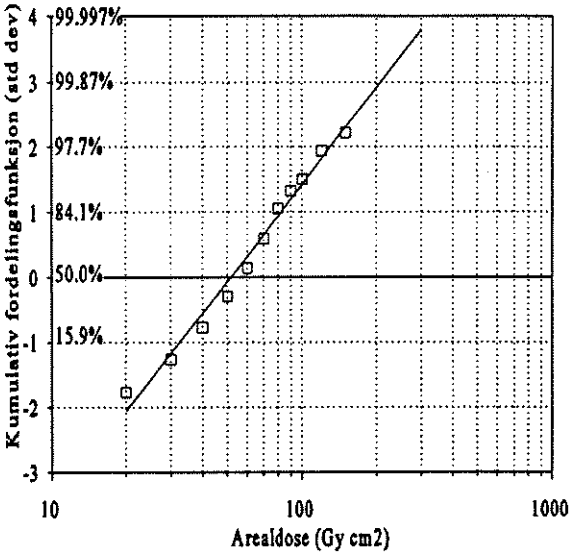


Figure 7. Fordeling av colon dobbeltkontrast undersökelser, 76 observasjoner fra alle hospitaler.

Ikke uventet er fordelingene log-normale, dvs normalfordelt på en logaritmisk skala. Ved å plotte den kumulative frekvensen med logaritmisk skala på abscisse og med normalfordelt skala på ordinat, fremkommer en rett linje, se figur 8 og 9. En slik presentasjon av data kan benyttes til å estimere sannsynligheten for unormalt höye arealdoser. Fra figur 9 kan f. eks. sees at sannsynligheten for at en pasient skal få en arealdose som overstiger 200 Gy·cm^2, som tilsvarer ca. 50-60 mSv effektiv dose, er 0.13%, dvs. at teoretisk vil kun 1 til 2 pasienter av 1000 bli utsatt for en dose av denne störrelsesorden.



Figur 8. Lognormalt plott av kumulativ fordelingsfunksjon av arealdoser fra thorax undersökelser, 774 observasjoner.



Figur 9. Lognormalt plott av kumulativ fordelingsfunksjon av arealdoser fra colon dobbeltkontrast undersökelser, 76 observasjoner.

Konklusjon

Introduksjon av arealdosemetre med grensesnitt til dator har åpnet muligheten for en grundig analyse av röntgenundersökelser med hensyn på teknikk og stråledose. Utstyret benyttet i denne undersökelse er imidlertid ikke godt nok til en tilfredsstillende analyse av mere avanserte röntgenundersökelser som benytter andre radiografiske teknikker, så som 100 mm kamera og digitale teknikker. Til det trengs en atskillelig höyere transmisjonsfrekvens av data. Når dette skrives har imidlertid et nytt produkt med en transmisjonsfrekvens på opptil 200 målinger i sekundet blitt lansert, og en regner derfor med at analyse av også de mere avanserte radiografiske teknikker er mulig i nærmeste fremtid.

Metoden med kumulative log-normale plott av arealdosefordelingene, er av spesiell interesse å benytte i intervensjonell neuroradiologi. For disse undersökelser har en i de senere år observert strålingsdoser som overstiger terskelen hvor deterministiske skader oppstår. Ved å foreta målinger av inngangsdose, f. eks. med TLD, av en gruppe pasienter som gjennomgår en slik undersökelse, kan sannsynligheten for at terskeldosen overstiges anslåes. En kan derfor utfra et begrenset antall målinger ved en avdeling avgjöre om spesielle foranstaltninger for å unngå for höye stråledoser er nödvendig.

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BEHANDLING AF GODARTEDE THYREOIDEASYGDOMME MED I-131 NATRIUMIODID I DANMARK 1990 - 1995

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ABSTRACT

Antallet af indgivne doser I-131 natriumiodid til behandling af godartede thyreoideasygdomme i Danmark er godt og vel fordoblet i perioden 1990 - 1995. Den gennemsnitligt indgivne dosis var i 1995 ca 425 MBq - i 1990 var den gennemsnitligt indgivne dosis 390 MBq. Incidensen af godartede thyreoideasygdomme i Danmark har i perioden været relativt konstant (ca 4000 nye tilfælde/år). Stigningen i antallet af indgivne doser forklares dels ved, at denne behandlingsform vinder stigende udbredelse på bekostning af (især) kirurgisk behandling, og dels ved, at man i Danmark i stigende udstrækning også behandler atoksiske strumaer med I-131.

Foredraget redegør for udviklingen i perioden og diskuterer de problemstillinger, stigningen giver anledning til. Foredraget vil desuden komme ind på de forhandlinger, der p.t. føres i EU med henblik på fælles regler omkring strålebeskyttelsen ved denne behandlingsform, herunder fælles tilladte grænser for restaktivitet/administreret aktivitet i ambulante patienter.

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Apoptosis: Effects of ceramide and ionizing radiation

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Recent evidence suggests that ionizing radiation and various other stimuli such as tumor necrosis factor-a and the Fas antigen signal through the sphingomyelin pathway to induce apoptosis. The sphingomyelin pathway is initiated in the plasma membrane by hydrolysis of the phospholipid sphingomyelin to generate ceramide by the action of sphingomyelinase. Ceramide then acts as a second messenger to stimulate a cascade of protein kinases, protein phosphatases and transcription factors like NF-kB to activate a common programmed cell death pathway.

To learn more about cell signalling in apoptosis and the cellular response to ionizing radiation we compared the effects of the cell membrane-permeable ceramide analogue N-acetylsphingosine with the response of the cells when exposed to ionizing radiation. Flow cytometry and agarose gel electrophoresis were used to detect cell-cycle effects and apoptosis. A brief review of the field will be given and a few early results will be presented.

STUDIES OF RADIOACTIVITY IN DIALYSIS PATIENTS

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Abstract

Dialysis could be an effective method for removing radionuclides from the blood-pool. Verzijl et al[4] showed in simulations that extracoporeal clearance could diminish the biological half-life of ^{137}Cs by more than 50%. The traditionally used Preussian Blue salts reduce the biological half-life of ^{137}Cs by approximately 30%.

The objective of this project was to see how fast and effective a dialysis patient could get rid of radiocaesium after intake and investigate other radiologically important radionuclides at dialysis.

The dialysis fluid was collected fractionally at every occation of dialysis. One could observe a fast accumulation of radiocaesium at the beginning and a slower at the end of each dialysis. The amount of accumulated radiocaesium decreased for each occation of dialysis both fractionally and totally.

The effective mean residental time in this study was $\tau = 17\pm3$ days for dialysis and $\tau = 143$ days for a ICRP standard man. The biological halflife for dialysis was determined to 12±2 days compared with 70 - 100 days for normally functioning clearance.

Introduction

From radiation protection point of view, it is important to identify critical groups. Patients with end-stage renal disease might constitute a critical group due to intake of radiologically important nuclides. Dialysis could be an effective method of removing radionuclides from the blood-pool. Up to date there has been very few publications concerning this field.

D. Josefsson et al[1] discussed the effect of dialysis on radiocaesium in man. Josefsson showed that there was a considerable reduction of ^{137}Cs among the haemodialysis patients compared with peritoneal dialysis patients and healthy persons. R. M. Culpepper[2], however, focused on the reduction of ^{131}I by haemodialysis. He showed that the maximum clearance of ^{131}I was achieved 24-29 hours after intake. J. M. Verzijl et al[4] showed in simulations that extracorporeal clearance could diminish the biological half-life by 50%. The traditionally used Prussian Blue salts, $Fe_4[Fe(CN)_6]_3$, from Heyl Chemischpharmazentische Fabrik in Berlin, reduce the biological half-life by approximately 32%. Furthermore, Culpepper and Chang[3], have studied the aspects of radiation protection for personnel categories working with dialysis, when treating patients who have received treatment with ^{131}I shortly before dialysis.

The objective of this project was to see how fast and effective a dialysis patient could get rid of radiocaesium after intake and investigate other radiologically important radionuclides at dialysis.

Dialysis

Dialysis is a method of cleaning the blood of the patients with end-stage renal disease and can also be used in emergency situations when dealing with poisoned patients. There are three types of dialysis that must be discussed. The types in question are haemodialysis (HD), haemofiltration (HF) and peritoneal dialysis (PD). They differ from each other with respect to how the blood is cleaned. Waste products (e.g. urea) and electrolytes, such as sodium and potassium, are separated

from the blood by the use of semipermeable membranes.

In PD the peritoneum acts as a semipermeable membrane with an effective area of 1,5 m^2. A dialysis concentrate that consists of salts is added to the peritoneal cavity and a gradient of concentration arises. There is a diffusion over the membrane where small molecules are diffusing faster than big molecules. There is also a transport of water over the membrane due to osmotic active substances within the dialysis concentrate. Therefore the concentration of molecules is the same on both sides of the membrane. There is an osmotic pressure when the water is transported over the membrane to where the osmotic active substance is situated. After five hours the dialysis fluid (now with water, waste products and electrolytes) is exchanged with fresh dialysis concentrate.

The HD also uses diffusion and osmotic pressure over a membrane to clean the blood. Instead, the membrane is outside the human body. This dead, semipermeable membrane is attached to a machine. The membrane with an area of 1 - 1,5 m^2 is folded inside a tube to make it space-saving. There is a constantly exchanged dialysis concentrate on one side of the membrane, on the other side the blood flows. The dialysis concentrate consists of salts and an osmotic substance, glucose. The HD is performed approximately five hours, three times a week.

HF uses the ultrafiltration which means that there is a difference in osmotic or hydrostatic pressure over a semipermeable membrane outside a human body. Variations in the osmotic pressure acts in the opposite direction of hydrostatic pressure. Therefore the molecules will be pressed and sucked over the membrane. No dialysis concentrate is used in the HF so the filtrated solution is replaced by a special concentrate of salts. The exact amount needed is measured by the dialysis machine. The HF procedure reduces the dialysis time by one hour at each treatment.

Membrane

The membranes in HD and HF are different from each other. In HD the blood flows on one side of the membrane and the dialysis concentrate/fluid on the other side. In HF the blood is led through a capillary tube and the waste products are pressurised through the tube.

The clearance of the blood depends on the reduction of the concentrating of waste products and electrolytes in the blood as a function of time. Dialysis is most effective in the beginning of the treatment. In HD the normal clearance velocity is 200 ml/minute and in HF 400 ml/minute. The water velocity is 500 ml/minute.

The membranes have different degrees of perviousness which is determined by a Sieving coefficient, $S = C_F/C_B$ where C_F is the concentration in the dialysis fluid and C_B the mean concentrate in the blood.

Retention

The retention[5] at a time t after intake is

$$r(t) = a_1 * e^{-\ln 2 * t/T1} + a_2 * e^{-\ln 2 * t/T2}$$

where $T_{1/2}$(eff) consists of two components, T1 and T2, and $a_1 + a_2 = 1$. The standard man used by ICRP has $a_1 = 0.1$, $a_2 = 0.9$, T1 = 2 days and finally T2 = 110 days for radiocaesium.

One can calculate the effective mean residential time, τ, in the human body

$$\tau_{dialysis} = (a_1 * T1 + a_2 * T2)/\ln 2$$

The biological half-life, $Te = \tau * \ln 2$.

For an ICRP standard man the effective mean resindetial time is $\tau = (0.1*2 + 0.9*110)/\ln 2$ days = 143 days and the biological half-life is Te = 0.1*2 + 0.9*110 days = 99 days componated to the physical half-life T_{ph} = 30 years.

Material and methods

The objective was to determine the biological half-life of ^{137}Cs during dialysis. Since persons who need dialysis in a case of emergency use the artery, dialysis patients with very high veinpressure (haemofiltration) were selected. The patients were given reindeer meat to eat with an activity determined at the laboratory. The dialysis fluids from the patients during dialysis were collected in intervals of 20 or 40 minutes. This procedure was done before as well as several times after the meal following the normal dialysis scheme. The fluids were also collected from the total dialysis after some weeks. The amount of dialysis fluid after each treatment was approximately 130 litres. The patients were also measured with the whole body counter before and after the meal.

A yield determinant, ^{134}Cs, was added to the fluids. The fluids which had been collected during the entire dialysis of 4.5 hours which was passed through cotton wound cartridge filters impregnated with copper ferrocyanide, $Cu_2Fe(CN)_6$, that absorbed the caesium. The filters were dried and ashed at 450^0C. The caesium in the fluids collected in intervals was coprecipitated with copper ferrocyanide. This was done by adding equal amounts of $K_4Fe(CN)_6$ and $Cu(NO_3)_2$ to the dialysis fluids. The precipitates were dried and like the ashed filters measured on a coaxial Ge(Li)-detector with a resolution of 2 keV at 1332 keV.

Result

One could observe a fast accumulation of radiocaesium at the beginning and a slower at the end of each dialysis for both male patients (see diagram 1 and 3). The amount of accumulated radiocaesium decreased for each occasion of dialysis both fractionally and totally (see diagram 2 and 4). The total amount of ^{137}Cs at each time of treatment is shown in diagram 5.

As discussed in the introduction the retention at a time t for an ICRP standard man is

$$r(t) = 0.1 * e^{-\ln 2*t/2} + 0.9 * e^{-\ln 2*t/110}$$

where t should be expressed in terms of days.

There are two components of the half-life of ^{137}Cs during dialysis. The fast one is calculated to be 4.6±0.9 days for both patients. The slower one is 40±8 days. This gives us T1 = 4.6±0.9 days and T2 = 40±8 days. The retention in our study is therefore

$$r(t)_{dialysis} = 0.8 * e^{-\ln 2*t/4.6} + 0.2 * e^{-\ln 2*t/40}$$

which showes that $r(t)_{dialysis}$ is determined mainly by its first term.

The effective mean residential time, $\tau_{dialysis}$, in this study becomes

$$\tau_{dialysis} = (a_1 * T1 + a_2 * T2)/\ln 2 = 0.8*4.6 + 0.2*40)/\ln 2 = 16.9 \text{ days}$$

and the total effective biological half-life, $Te_{dialysis} = \tau_{dialysis} * \ln 2 = 11.7$ days.

The error of the measuring of radiocaesium is determined to be 20%.

Discussion

The dialysis removes the cesium rapidly. The amount of cesium removed decreases from dialysis to dialysis. The cesium in the dialysis liquid decreases slowly throughout every

particulary dialysis and the dialysis is therefore more effective in the beginning. The length of the dialysis is limited by the patient's well-being. In intervals of 20 minutes the reduction of cesium is not particularly effective after about 200 minutes. After five dialysis occasions the clearance of radiocaesium is reduced to a fifth of the first one, and therefore it is not necessary to continue with the dialysis after that.

The retention of ^{137}Cs during dialysis was determined to be

$$r(t) = 0.1 * e^{-\ln 2 * t/2} + 0.9 * e^{-\ln 2 * t/110}$$

Compared to the retention for an ICRP standard man the retention at dialysis is depending more on the first term with shorter biological half-life and dialysis retention is therefore larger than for the ICRP standard man.

The effective mean residential time in this study was $\tau_{dialysis} = 17 \pm 3$ days. The ICRP standard man has an effective mean residential time of $\tau = 143$ days. This is a difference of 126 days and a considerably faster reduction of ^{137}Cs by dialysis.

The biological half-life Te, in this study is 12±2 days compared with 99 days for the ICRP standard man. This is a reduction by 87 days.

Dialysis is therefor an effective method of removing at least ^{137}Cs from the blood-pool.

The fifth dialysis on patient EK (see diagram 4) is changed from HF to HDF (a modification of HF) and therefore the activity from this dialysis differs from earlier ones. The HDF treatment appears more effective than the HF.

The total estimation error in the analysis of radiocaesium is 20%.

Acknowledgements

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Most of all I thank my supervisor Ph.D. Elis Holm who made my project possible.

I thank Jörgen Hegbrant at ParkDialys, Ideonparken in Lund and Hans Thysell at the Department of Nephrology at the University Hospital in Lund, Sweden for their co-operation and help.

The patients at the ParkDialys who were a remarkable bunch of people are wormly praised.

I also thank all the employees at the Institute of Radiation Physics, Lund University and at ParkDialys to make my work possible.

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Diagram 1 - The activity of ^{137}Cs in dialysis liquids per 20 minutes at six different occations for patient TO. The first dialysis (I) was performed two days after intake. Dialysis II took place five days after intake, dialysis III, IV, VI and VII took place seven, nine, fourteen and sixteen days after intake, respectively.

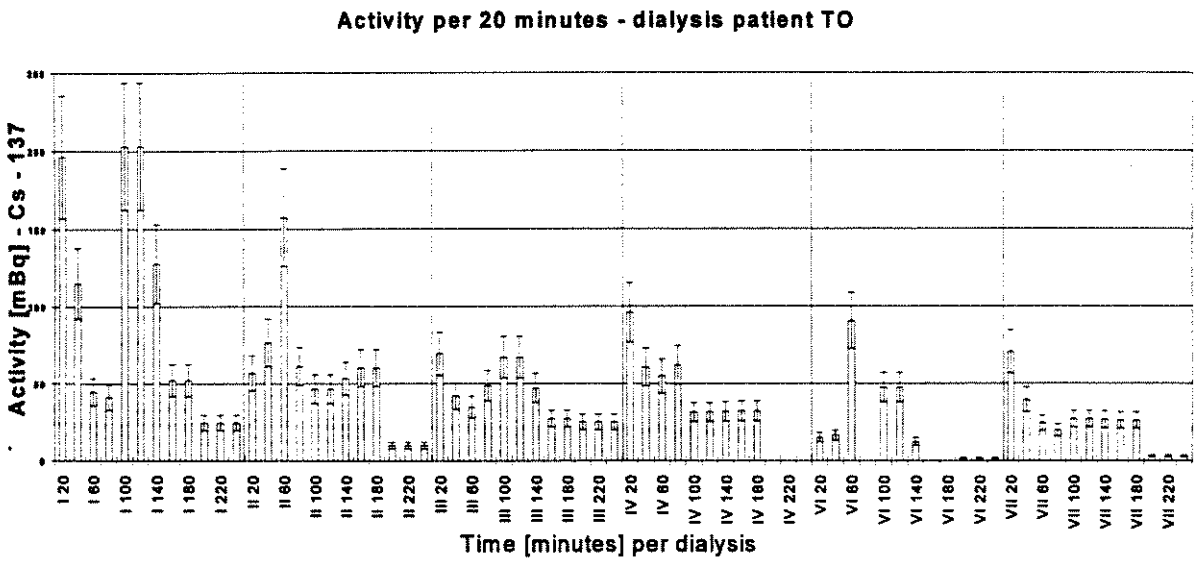

Diagram 1 - The activity of ^{137}Cs in dialysis liquids per 20 minutes at six different occations for patient TO. The first dialysis (I) was performed two days after intake. Dialysis II took place five days after intake, dialysis III, IV, VI and VII took place seven, nine, fourteen and sixteen days after intake, respectively.

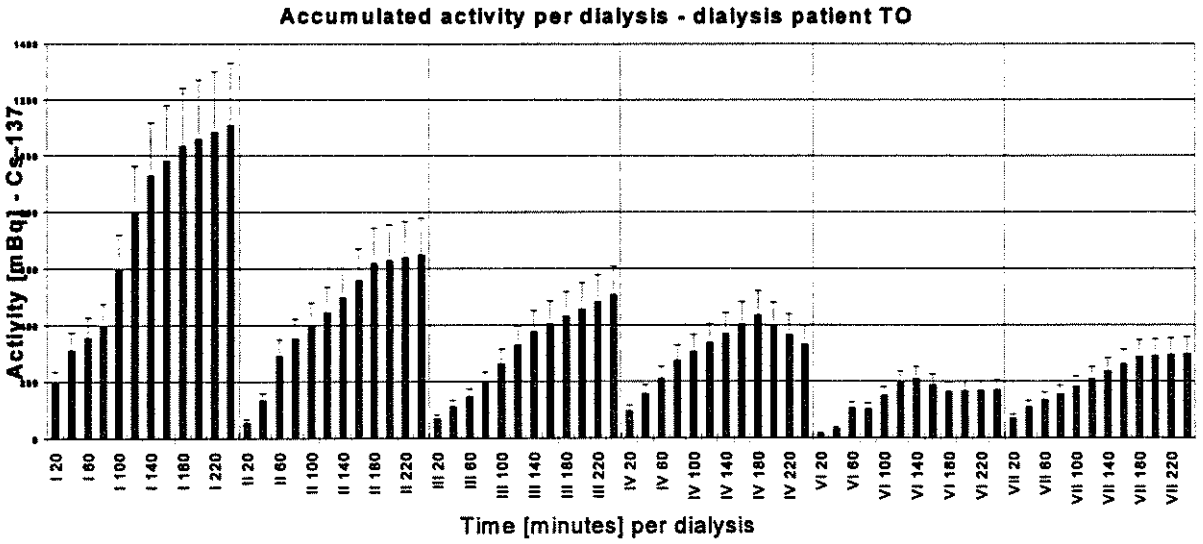

Diagram 2. The accumulated activity in mBq at six different occations of dialysis for patient TO.

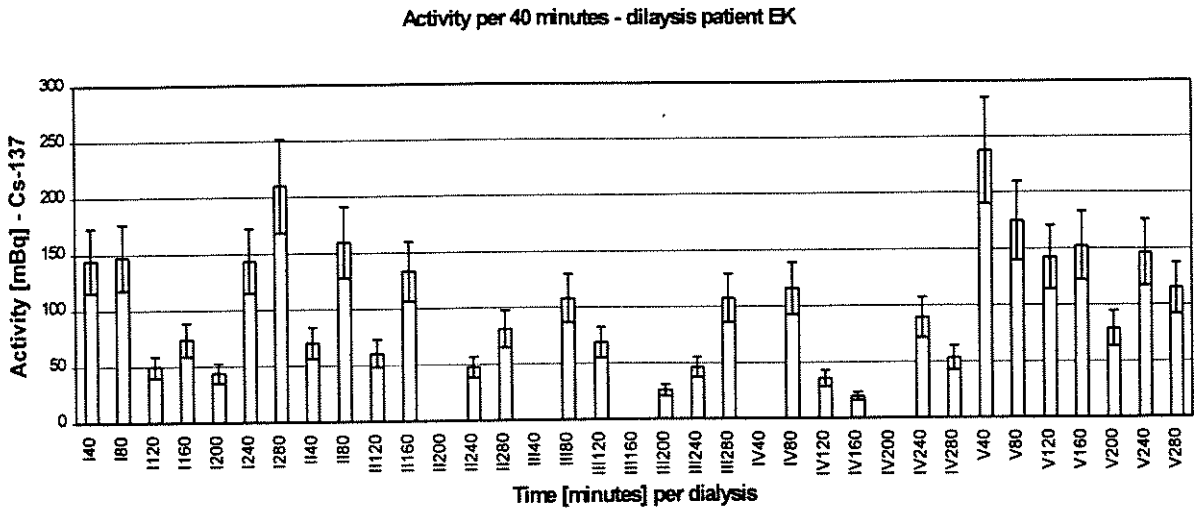

Diagram 3 - The activity of ^{137}Cs in dialysis liquids per 40 minutes at five different occations for patient TO. The first dialysis (I) was performed two days after intake. Dialysis II took place five days after intake, dialysis III, IV and V took place seven, nine and twelve days after intake, respectively.

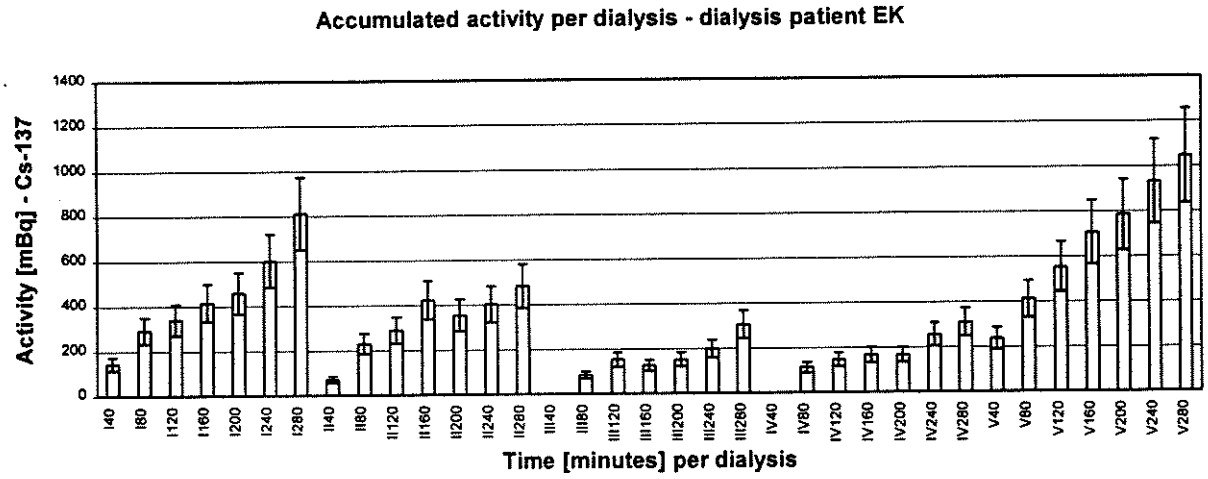

Diagram 4. The accumulated activity in mBq at five different occations of dialysis for patient EK.

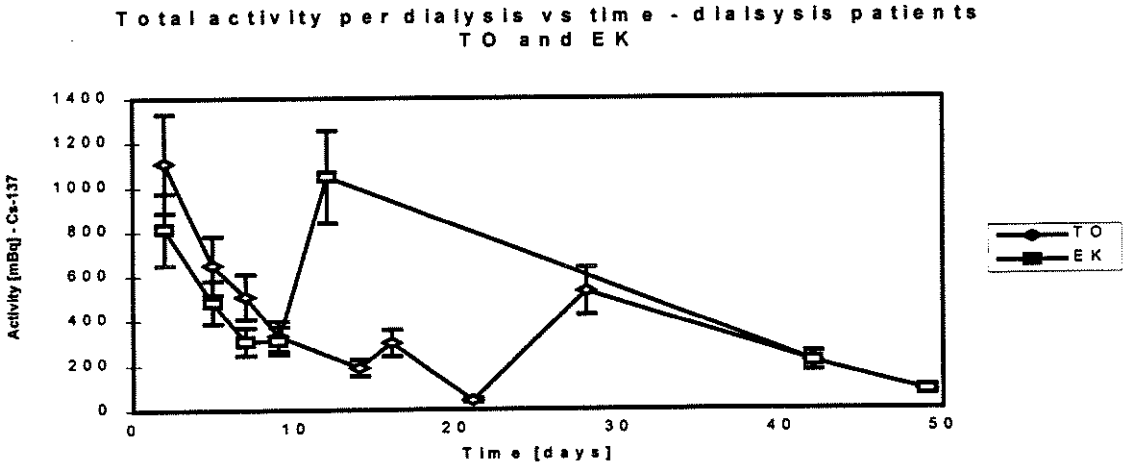

Diagram 5. Total activity of ^{137}Cs per dialysis as a function of time for the patients TO and EK.

An inexpensive phantom for quality control in dental radiology

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Abstract. A postal package containing a phantom and TLD chips, for routine quality control assessment of dental radiographic equipment and film processing, has been developed. The phantom is made of PMMA material, with details for visual and quantitative estimation of image quality of the imaging system. The phantom construction and intended use is presented, as well as a software for evaluation and registration of phantom images. Examples from measurements are given.

Introduction

The Icelandic Radiation Protection Institute has developed a postal package containing a phantom with TLD chips, for routine quality control assessment of dental radiographic equipment and film processing.

The package is based on an earlier postal survey of the dental x-ray units in Iceland, with a package consisting of an industrial radiographic film, TLD chips and a questionnaire on the use of each x-ray unit. This survey gave a fairly accurate picture of the use and of the absorbed radiation doses, concluding that a wide range of doses was used for similar examinations and equipment. Additional information of the quality of the imaging chain at each practice is therefore of interest, and may together with information of the radiation absorbed doses be used in optimising the radiographic procedure.

Materials and methods

The dental phantom is designed with the following criteria in mind:

1. Low production cost
2. Robust and light weight for easy transport by mail
3. Shall give information on contrast and resolution of the imaging chain
4. Can carry TLD ribbons for dose measurements
5. Be used with the most common size dental radiographic film (31 mm x 41 mm)
6. Easy to use for routine QC assessment by the dentist or his staff

The phantom is made of PMMA material (see *figure 1 and 2*). A 0.3 mm Cu foil is fitted into the phantom, for beam hardening and to simulate adequate tissue thickness. It contains an aluminium step wedge (seven 1 mm steps) for visual and quantitative estimation of density and contrast. A section for low contrast detail evaluation, consists of drilled holes of different sizes and depths. There are four groups of holes with 3, 2, 1 and 0.5mm diameter. There are six holes in each group with different depths (4, 3, 2, 1, 0.5 and 0.25 mm). Three wire meshes (with wire thickness of 0.4, 0.2 and 0.16 mm) make up the section for high contrast resolution. The phantom has also a lead beam stopper that makes it possible to measure the background density of the film.

The package is sent by mail to dentists with instruction on its use. The dentist makes two exposures, using his own films and the same exposure time. He then develops one of the films and sends the package back to the institute, with both films. At the institute the TLD chips is

measured and the second film developed in a standard way. Then the films are evaluated and a report is printed.

Dental films that are exposed with the phantom are evaluated by measuring the densities under the step wedge and lead beam stopper. Density in the middle of the step wedge is used as an indicator of the film speed and density differences of steps 4 and 7 as an indicator of the film contrast. By visual inspection of the section for low contrast detail, the number of holes visualized in each group can be found. The number seen is dependent on contrast, spatial resolution and noise in the imaging system, and as such it is a good indicator of the total image quality of the system. The films are also inspected for development or handling artifacts.

A software for registration and reporting, has been developed, which prints out a report for each dental unit. In this report, a contrast detail curve, based on visualized holes, is plotted, which places the curves for films developed by the dentist in comparison with a film of recommended minimum quality (see *figure 3*). The recommended minimum quality is based on more than 100 measurements made with the phantom. In *figure 4* two contrast detail curves are plotted, for a film with good image quality (line 1) and a film with inferior image quality (line 2).

Discussion and Conclusion

The low contrast objects in the phantom have large thickness compared to the widh, at least in the higher contrast region. The angle under which the x-ray beam enters the phantom may therefore influence the low contrast evaluation. Measurements with different angels show however only a slight degration for the thinnest holes (1 and 0.5 mm). This degration is not detectable for angels less than 5°. In the instructions for use the importance of correct positioning is therefore emphasized.

The phantom was developed for the institute's postal-inspection of dental x-ray practices. It is also very well suited for routine quality control of radiographic procedures in dental practice, since it is inexpensive and easy to use . The quality of a dental imaging system can be routinely monitored by visual comparison of phantom images with a reference image. The reference image can be made when the x-ray unit and development equipment and procedures are in optimal condition. The phantom can help dentists to spot and identify exposure and imaging problems. The phantoms production cost is about ~ US$100.

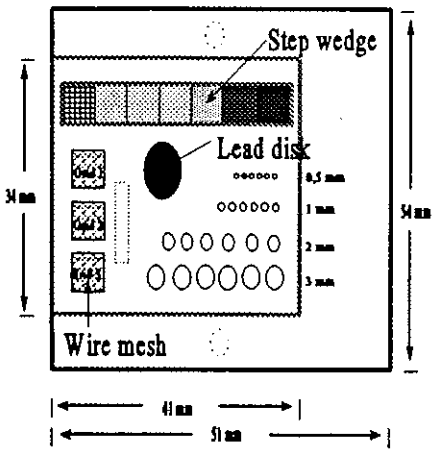

Figure 1. Layout of the phantom

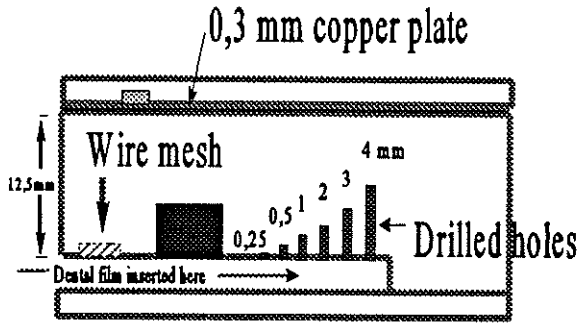

Figure 2. Cross section of the phantom

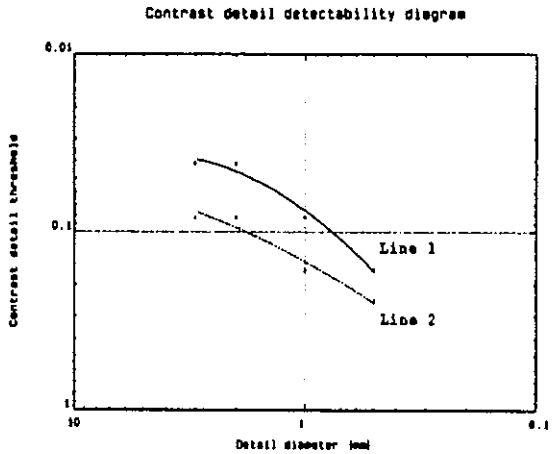

Figure 3. A film from a dental practice (1) compared to recommended minimum image quality (2).

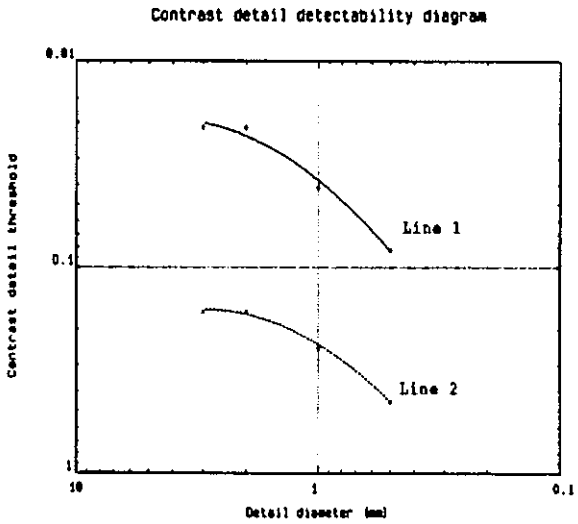

Figure 4. Example of curves for good (1) and inferior image quality (2).

Doses to Patients from Nuclear Medicine Examinations in Finland in 1994

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Abstract A country-wide survey on the use of radiopharmaceuticals in diagnostics and therapy has been made in Finland. A questionnaire was sent to all hospitals and institutes using unsealed sources in nuclear medicine examinations and therapeutic treatments. For each procedure, the radiopharmaceutical used, the number of procedures performed and the typical administered activity were queried. The survey gave detailed information on the types and frequencies of nuclear medicine procedures performed in Finland in 1994. The number of nuclear medicine examinations was 50,900 and that of the therapeutic treatments was 2,150. The frequency of nuclear medicine examinations was 10 examinations per 1,000 inhabitants in Finland in 1994. ^{99m}Tc was used in 85% of nuclear medicine examinations. The effective dose from nuclear medicine examinations to the patients and to the population was calculated. To calculate the collective effective dose, the dose factors (mSv/MBq administered radiopharmaceutical) given in ICRP Publication 62 were used. The collective effective dose to the patients was 220 manSv and the mean effective dose to the population was 0.04 mSv per person in Finland in 1994.

Introduction

New radiopharmaceuticals, new applications of existing radiopharmaceuticals, and new alternative methods continue to become available. Therefore a new country-wide survey on the use of radiopharmaceuticals in diagnostics and therapy was made in Finland. The aim of the survey was to evaluate the collective effective dose to the patients and the mean effective dose to the population from the nuclear medicine examinations. The survey also gave detailed information on the types and frequencies of nuclear medicine examinations performed in Finland in 1994.

Materials and methods

A questionnaire was sent to all hospitals and institutes in Finland using unsealed sources in nuclear medicine examinations and therapeutic treatments. The questionnaire was divided into three sections: (i) nuclear medicine examinations of adults, (ii) nuclear medicine examinations of children, and (iii) therapeutic procedures. For each procedure, the radiopharmaceutical used, the number of procedures performed and the typical administered activity were queried.

To calculate the effective dose, the dose factors (mSv/MBq administered radiopharmaceutical) given in ICRP Publication 62 were used [1]. The dose factors given in ICRP Publication 62 are based on the new tissue weighting factors given in ICRP Publication 60 [2].

Results and discussion

The number of nuclear medicine examinations was 50,900 and that of therapeutic treatments was 2,150. The number of nuclear medicine examinations decreased by 40% between 1982 and 1994. The main reason for this is perhaps, that nuclear medicine procedures are replaced by new alternative methods like ultrasonography, computed tomography and magnetic resonance imaging. The liver and the thyroid scintigraphy are replaced by ultrasonography. Computed tomography and magnetic resonance imaging are reducing the number of brain scans. The numbers of nuclear medicine examinations and the radionuclides used in 1975, 1982 and 1994 are shown in Fig. 1 [3,4,5].The frequency of nuclear medicine examinations was 10 examinations per 1,000 inhabitants in Finland in 1994. The information obtained related to total of 16 radionuclides used for 86 different types of examinations, although 20 of these accounted for over 85% of all examinations. Table 1 shows the nuclear medicine examinations of different organs or functions.

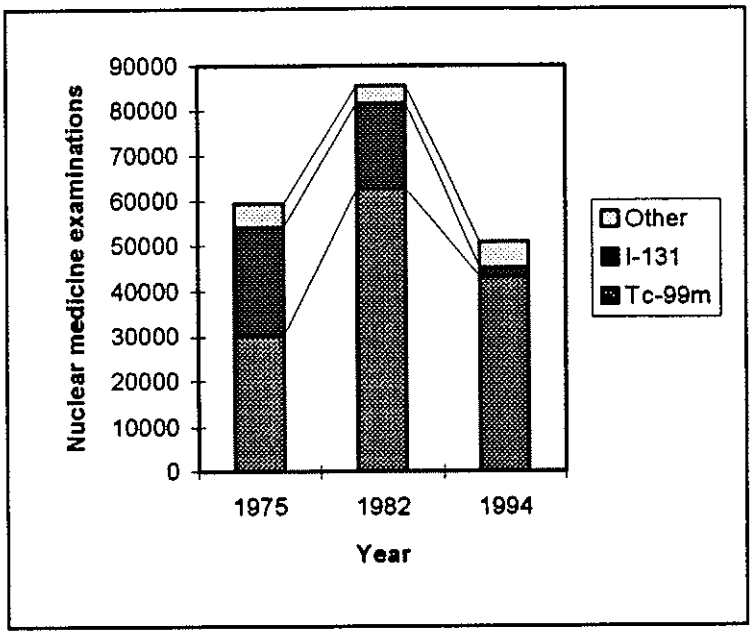

Figure 1. Nuclear medicine examinations in Finland in 1975, 1982 and 1994.

Table 1. Nuclear medicine examinations of different organs in Finland in 1994.

FUNCTION OR ORGAN EXAMINED	NUMBER OF EXAMINATIONS			% OF TOTAL
	Adults	Children	Total	
Bone	20208	704	20908	41.1
Lung	7184	14	7198	14.1
Stomach, G.I. tract	662	167	829	1.6
Kidney and urinary system	6927	2245	9172	18.0
Cardiovascular	6874	120	6994	13.7
Nervous system	1579	165	1744	3.4
Thyroid	2548	24	2572	5.0
Haematology	726	19	745	1.5
Additional imaging examinations	722	48	770	1.5
TOTAL	47430	3506	50932	100

The frequency of different examinations have changed with time (Fig. 2). From all examinations 41% were bone, 14% lung and 14% cardiovascular procedures. ^{99m}Tc was the radionuclide of choice for 85% and ^{131}I for 4% of all examinations.

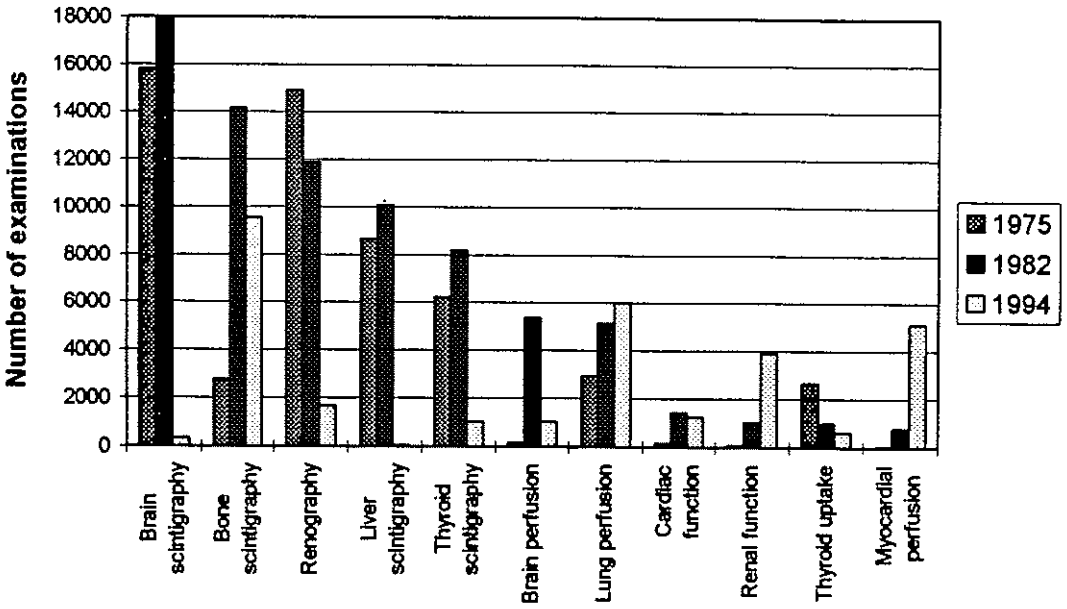

Figure 2. The numbers of different nuclear medicine examinations in 1975, 1982 and 1994.

The collective effective dose to the patients was 220 manSv and the effective dose to the population was 0.04 mSv per person in 1994. The mean effective dose per examination was 4.2 mSv. The numbers of nuclear medicine examinations and the collective effective doses to the patients from different radionuclides are shown in Table 2. The use of the radionuclides ^{99m}Tc, ^{131}I and ^{201}Tl accounted for 95.6% of the collective effective dose.

Table 2. Use of radionuclides in nuclear medicine examinations and the collective effective dose. *Other radionuclides are ^{14}C, ^{67}Ga, ^{75}Se, ^{125}I and ^{133}Xe.

RADIONUCLIDE	NUMBER OF EXAMINATIONS	TOTAL ACTIVITY (GBq)	COLLECTIVE DOSE (manSv)	% OF TOTAL DOSE
^{99m}Tc	43071	19582	136	61.8
^{131}I	2059	198	33	14.9
^{201}Tl	1801	182	42	19.0
^{123}I	914	57	2.3	1.0
^{51}Cr	724	2.3	0.04	0.0
^{57}Co/^{58}Co	560	0.2	0.09	0.0
^{111}In	450	20	4.3	2.0
^{15}O	402	1033	0.5	0.2
^{18}F	282	63	1.3	0.6
^{11}C	185	61	0.3	0.2
Other radionuclides*	446	14	0.5	0.2
		Total	220	100

The collective doses from the use of ^{131}I in 1975, 1982 and 1994 are compared in Table 3. Ten examinations with the largest contributions to the collective effective dose are shown in Table 4. The large figure for the bone scanning is due to the large number of the examinations (41% of all examinations), but that for myocardial scintigraphy is mainly due to the relatively large radiation dose per examination (only 3% of all examinations).

Table 3. The collective doses from the use of ^{131}I in 1975, 1982 and 1994 (figures marked with * have been calculated using the dose factors given by SSI [6]).

YEAR	NUCLEAR MEDICINE EXAMINATIONS	TOTAL COLLECTIVE DOSE (manSv)	COLLECTIVE DOSE FROM ^{131}I (manSv)	% OF TOTAL COLLECTIVE DOSE	COLLECTIVE DOSE FROM ^{131}I USING THE DOSE FACTORS OF ICRP 62 (manSv)
1975	59350	344*	233*	67	350
1982	85340	434*	202*	46	303
1994	50920	220	33	15	33

Table 4. Nuclear medicine examinations with the largest contributions to collective effective dose.

NUCLEAR MEDICINE EXAMINATION	RADIO-NUCLIDE	RADIOPHARMA-CEUTICAL USED	EFFECTIVE DOSE MEAN (mSv)	EFFECTIVE DOSE COLLECTIVE (manSv)	% OF TOTAL
Myocardial perfusion	^{201}Tl	ion	23	40,5	18
Bone scintigraphy	^{99m}Tc	fosfates/fosfonates	3,5	33,7	15
Bone scintigraphy, whole body	^{99m}Tc	fosfates/fosfonates	3,7	28,9	13
Myocardial perfusion, SPECT	^{99m}Tc	MIBI	8,4	25,5	12
Thyroid uptake	^{131}I	NaI	65	18,1	8
Lung perfusion	^{99m}Tc	MAA	1,2	6,9	3
Brain perfusion, SPECT	^{99m}Tc	HMPAO	6,2	6,0	3
Lung ventilation	^{99m}Tc	Venticoll	10	4,5	2
Cardiac function, equilibrium	^{99m}Tc	HSA	4,5	3,2	2
Lung ventilation	^{99m}Tc	Technegas	5,7	3,1	1
			TOTAL	170,5	77

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Patient doses from CT examinations in Norway, with analysis of factors affecting the effective dose, including specific clinical indications.

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Abstract

The patient doses from seven CT examinations were assessed based on the reported scan techniques from 44 hospitals (49 scanners) in Norway. The dose assessments were done according to the data and method published by the National Radiological Protection Board. The mean effective doses were 2 mSv for examinations of the head, 10 - 13 mSv for examinations in the trunk region and 4.5 mSv for examinations of the lumbar spine. Factors of 8 - 20 were observed for the range of values of effective dose in a given type of examination over all scanners in survey. Variations in reported scan volumes and differences between scanner models in survey might each explain about 30% of this total spread in dose for a given examination type. A need for image quality assessments in CT was seen, in order to compare scanner performance.

The survey also included dose calculations for examinations done on twelve specific clinical indications. However, the reported scan techniques and resulting doses for a given clinical indication still varied considerably over all scanners in survey. This emphasises the need for a discussion on scan technique, both with respect to the number of slices, coach increment, tube load, and the use of contrast. It was also seen a need to review the scan technique in order to reduce the dose to radiation sensitive organs like the eye lenses and gonads, and for an overall discussion on the reasons for requesting CT examinations in preference to conventional radiological examinations or examinations not involving ionising radiation. The CT doses in Norway were about 50% higher compared to a similar British survey. This might possibly be explained by a larger part of scanners with shorter focus to axis distances presented in Norway.

There were about 70 CT scanners in Norway in 1994 (16.1 scanners per million inhabitants). Based on the frequency of CT examinations in 1993 (50 examinations per 1000 inhabitants per year) the contribution to collective effective dose from CT was estimated to 1000 manSv (0.2 mSv/inhabitant/year), it means about 30% of the contribution from diagnostic use of x-rays.

Reference : Olerud, H.M. and Finne, I.E. *Computer-tomografi ved norske sykehus. Undersøkelsesteknikk og stråledose til pasient.* StrålevernRapport 1995:1 l. Østeras: Norwegian Radiation Protection Authority (1995). (Norwegian text).

Individual radon exposure history measured by an alpha track detector technique.

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ABSTRACT

The lung cancer risk from exposure to indoor radon is assessed in epidemiological studies where the radon exposure of the past is of considerable importance. The measurement of the long-lived decay products from ^{222}Rn remaining on indoor surfaces is an alternative or complementary method to the traditional measurement of the radon concentration currently used in assessing the previous radon exposure. A part of the short lived airborne radon progeny will plate-out on surfaces in a room and a part of these decay products will be implanted into the surfaces due to the recoil effect from an alpha decay.

An autoradiographic alpha-track method to assess the ^{210}Pb activity implanted in glass surfaces by measurement of ^{210}Po alpha activity has been developed and tested under field conditions. One limiting factor at low exposure levels is the alpha background activity in the sub-surface material. In the search for a practical field method, the use of two different alpha track detector materials has been found successful.

The KODAK LR-115 cellulose-nitrate film is sensitive to alpha particles with energies approximately in the range 1.2 - 4.8 MeV, while the CR-39 detector material is sensitive for alpha-particles up to higher energies. The alpha-particles from ^{210}Po have an energy of 5.3 MeV, and thus will not be detected in the LR-115 detector. By exposing one LR-115 and one CR-39 detector side by side on glass panes, the background of the glass is measured with the LR-115, and both the background and the signal by the CR-39 detector.

This combination of alpha track detectors, called RETRO-detector, can measure ^{210}Po surface activity below 1 $Bq \cdot m^{-2}$ during field conditions. Measurements in dwellings show that a radon concentration of 50 Bq/m^3 will give a surface activity of about 1 Bq/m^2 on a plate glass exposed to that radon concentration during 20 years. The RETRO-detectors will be extensively used to support the exposure assessment in the planned radon epidemiological study on non-smokers in Sweden starting in 1996.

Introduction

The statistical power and validity of studies of lung cancer and residential radon exposure depend critically on the quality and reliability of exposure data. Due to the long latency period between exposure and tumour manifestation the cancer incidence reflects radon exposure years and decades backwards in time. The assessment of the cumulative past radon exposure for each person is generally done by measurements of the radon concentration in the dwellings where the person has been living. The results are then time-weighted by length of occupancy in each home to obtain a cumulative estimate of each person's past exposure. The calculated exposure is uncertain due to many reasons. The radon concentration measured today in the dwellings may not be representative for past situations. Data for some past dwellings will be lacking due to failure to find them or refusal to permit measurement.

The amount of long-lived decay products of radon, starting with ^{210}Pb ($T_{½}$=22 y) and remaining in dwellings, depend on past radon concentration. It has been shown (Samuelsson, 1988) that past radon levels in dwellings can be measured via long-lived decay products implanted by alpha particle recoil in glass or other hard surfaces.

Methods

The recoil range of the ^{210}Pb atom upon the alpha decay of ^{214}Po is only about 0.1 μm in glass, and an almost ideally thin alpha source is formed upon the subsequent alpha decay of ^{210}Po. Laboratory measurements with a pulse ionisation chamber of glass samples show that the self absorption in the glass is low and nearly all the alpha particles emitted have an energy close to 5.3 MeV.

In cooperation with colleagues in Lund, Dublin, Copenhagen, Gent and Mol, a field measurement method has been developed. Alpha track detectors were chosen for the purpose. These detectors have a high sensitivity to alpha radiation, insensitive to beta- and gamma radiation, have a low background, are small and relatively cheap. When the ^{210}Po activity on the glass surface is low, the intrinsic background alpha activity in the glass will limit the detectable ^{210}Po activity.
Two different alpha track detectors, (CR-39 and KODAK LR-115), are used simultaneously to separate the ^{210}Po alphas from the background alphas. The KODAK LR-115 (cellulose nitrate) detector is sensitive for alpha particles with energies within the energy range 1.2-4.8 MeV, while the CR-39 (polyallydiglycol carbonate) is sensitive to alpha particles with energies from 1 MeV up to more than 20 MeV. As the alpha particles from ^{210}Po emitted from the glass surface has an energy close to 5.3 MeV they will not be detected by the LR-115 detector, while they will be detected by the CR-39 detector. By placing these two detectors side by side directly in contact with the glass surface the LR-115 will have tracks from background alphas only and the CR-39 will have tracks from both background and ^{210} Po alphas. This combination of detectors is called a RETRO-detector.

Calibration and field test

Glass panes with known ^{210}Po surface activity has been used to calibrate the two detectors. These glass panes have been measured with a well calibrated large pulse ionisation chamber at Lund University (Johansson et al., 1992). Measurements on several different glass panes gave the following relations

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$$^{210}\text{Po surface activity } (\text{Bq}\cdot\text{m}^{-2}) = (CR_n - 1.97\cdot LR_n)/0.081$$

where $\mathbf{CR_n}$ and $\mathbf{LR_n}$ are the net tracks/cm^2 per hour of exposure of the detectors to the glass panes. An exposure to an inactive surface of Plexiglas is used to assess the intrinsic background of the two detector materials.
Applying this relation to repeated measurements of these panes both on the active side and background side gave a correct value with a standard deviation of less than 1 Bq·m^{-2}, independent of glass background activity. More detailed information of the RETRO-detector is reported elsewhere (Falk et al 1996).

To investigate the field characteristics of the RETRO-detector and the correlation between today's radon concentration and ^{210}Po surface activity on indoor surfaces 31 dwellings were selected. A part of the dwellings was inhabited by ssi-staff. Another part was dwellings investigated for radon of other reason. All these dwellings had been monitored for radon previously. The surfaces to be selected for a RETRO-measurement had to be in the dwelling

for more than 20 years. With other constrains the most suitable items was a mirror in the bedroom, the glass pane in a display cabinet, plate glass, the pane in a door between to rooms and glass panes covering photos. The surfaces of the selected item were cleaned with water and detergent. A piece of paper was applied to the surface for about one hour to let short-lived radon daughter decay and protect the surface from fresh plate-out, before the RETRO-detector was mounted. About three months later the RETRO-detector was removed from the surface and attached to a clean Plexiglas pane for transportation and storage until they could be etched in the laboratory. Two items in each dwelling were measured with a RETRO-detector and the radon concentration in the same room were measured during the same period.

Result and discussion

Assuming that the RETRO-measured item had been in a constant radon concentration during its whole existence in the dwelling with a constant plate-out rate, the measured surface activities of ^{210}Po were normalised to 20 years of exposure of the item, taken into account build up and decay. With this normalisation a comparison between the current measured radon concentration and measured surface activity can be done in the same diagram. Figure 1, show the correlation between the current radon concentration and ^{210}Po surface activity measured by the RETRO-detector. It can be seen that there is a poor linear correlation ($r = 0.56$). In fig 2 the mean value of current and previous radon measurement is plotted as a function of measured ^{210}Po surface activity. Obviously, the correlation is stronger when a better radon exposure is used.

The surface activity on glass panes today depends on many factors. The most important factors are:

- Radon concentration and its variation with time for at least the last 20 years.
- Different plate-out rates for the actual surface, which depend on aerosol concentration, air movements, ventilation rate and the place of the surface in the room.

Despite the simple assumptions made in this study it is clearly demonstrated that a measurement with the RETRO-detector probably is as good as or better than a current radon measurement when the aim is to estimate the previous radon concentration in a dwelling.

In an radon epidemiological study the assessment of the cumulative past radon exposure for each person is generally done from radon measurements in the dwellings where the person has been living. The cumulative exposure is then calculated as the sum of radon concentrations weighted by length of occupancy in *each* home. The calculated exposure is obvious uncertain due to many circumstances. An alternative to assess a person's past exposure could be done by *one single* measurement of ^{210}Po activity on an object that has been in the persons possession during most of his or her life. This approach will be used in the radon epidemiological study on non-smokers in Sweden starting autumn 1996, with the aim to use one RETRO-measurement for each person in addition to the common radon measurements in dwellings. Although the ^{210}Po surface activity measured will give a large uncertainty in estimated cumulative radon exposure, the data can be used for an independent classification in exposure levels.

Acknowledgement

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Fig 1. Relationship between **current** radon concentration and Po-210 surface activity

Data normalised to 20 years of exposure at same radon level

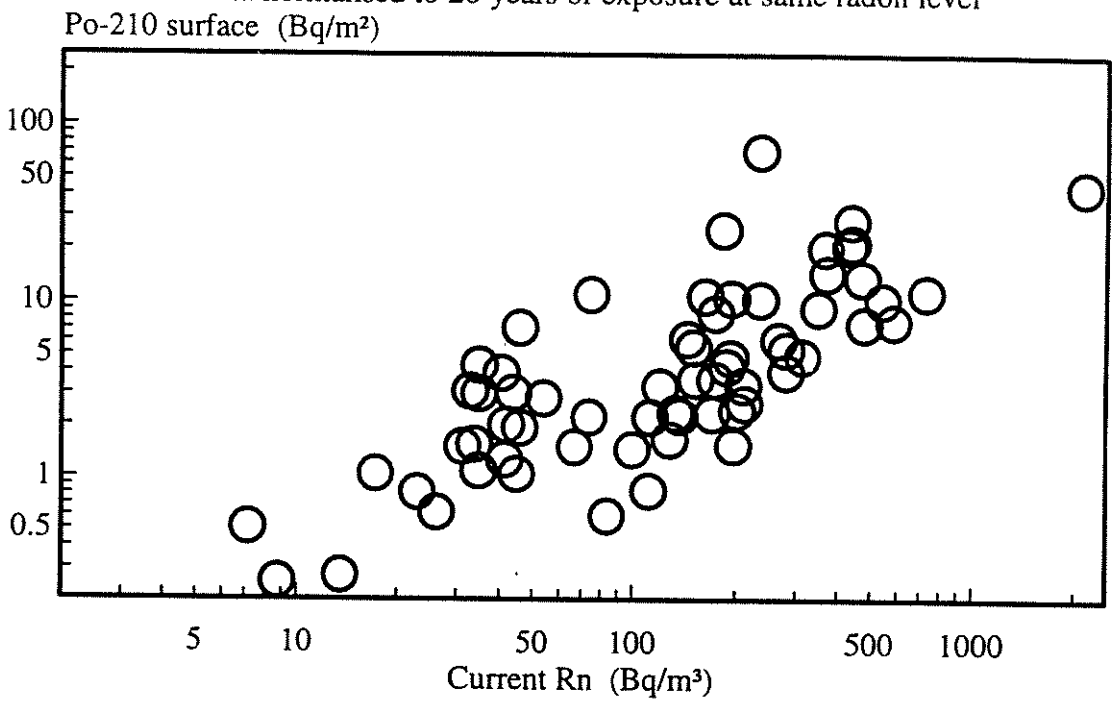
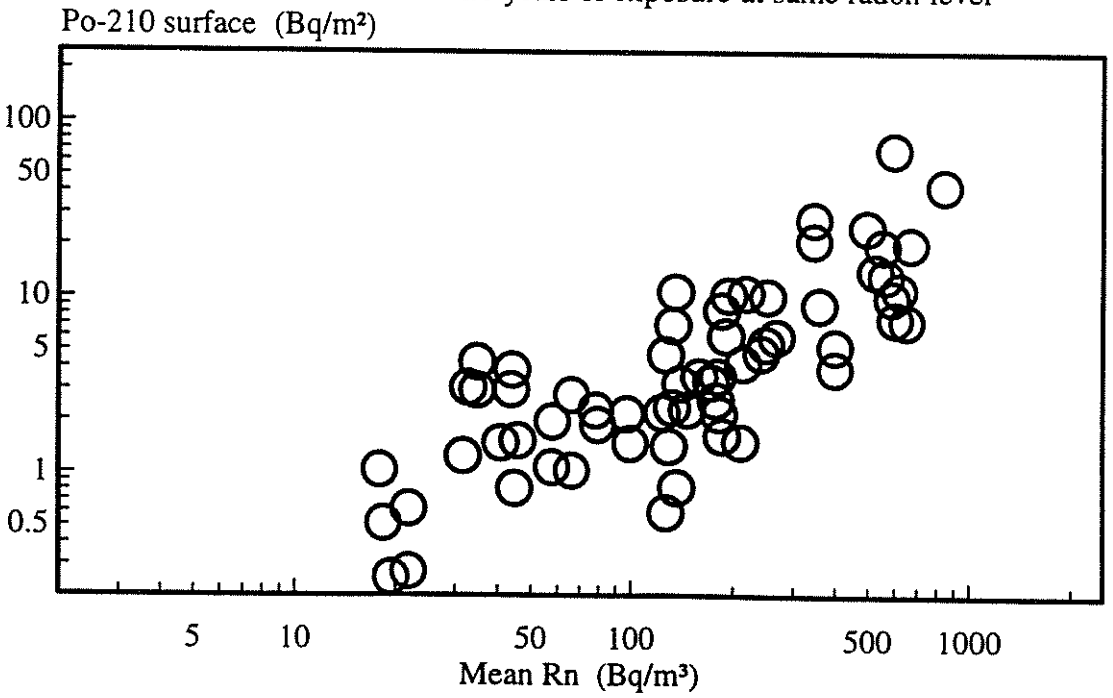


Fig 2. Relationship between **mean** radon concentration and Po-210 surface activity

Data normalised to 20 years of exposure at same radon level



Radon concentrations in new dwellings in Norway

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Results of a nation-wide survey of radon concentrations in Norwegian dwellings conducted from 1987 to 1989 have shown that the year average is between 55 and 65 Bq/m^3 All 7500 dwellings measured in this survey were built before 1980. In the period 1980 - 1995 approximately 440000 new houses were built.

Some of the 2500 measurements of radon after 1990 conducted mostly in detached and semi-detached houses included also these new dwellings. The results of these measurements have shown clear tendency of increased radon levels in the new compared to the old dwellings. In this presentation, an attempt is made to use information from the new data set, and in particular the dependence of the radon concentration on the age of the house to correct the average radon concentration found in the 87-89 survey. The present average radon concentration in Norwegian detached and semi-detached houses has been estimated to be in the range: 68 - 74 Bq/m^3.

The largest radon survey in Norwegian dwellings was conducted between 1987 and 1989. The results of the survey have been reported by Strand et al. [1]. In the following we extract some of the results in the survey relevant for further discussion.

The 87-89 survey was based on radon measurements in approximately 7500 randomly selected dwellings with at least one dwelling in each Norwegian municipality. The number of measurements per municipality was proportional to the population in the municipality. All the measured dwellings were built before 1980. Alpha track detectors with at least two months integration time were used in the survey. Radon concentration was measured in bedrooms only with one alpha track detector per house.

The distribution of radon concentrations was found to follow log-normal distribution. The average radon concentration in bedrooms was found to be 51 Bq/m^3 and the median 26 Bq/m^3. Starting with bedroom data and using information on building construction the average radon concentration for the whole dwellings was estimated to be 56 Bq/m^3. Further corrections were necessary due to unusually mild winters in the measurement period. Inclusion of these corrections gave the average radon concentration is most probably found in the range 55 - 65 Bq/m^3.

After 1990 a number of Norwegian municipalities conducted their own local surveys. In 1995 the municipalities reported approximately 3500 results of radon measurements to Norwegian Radiation Protection Authority (NRPA). All the measurements were conducted using alpha-track detectors over a period of at least 1 month, and in the cold season between September and May. It is a part of these new data that we focus upon in this paper in an attempt to estimate present radon levels.

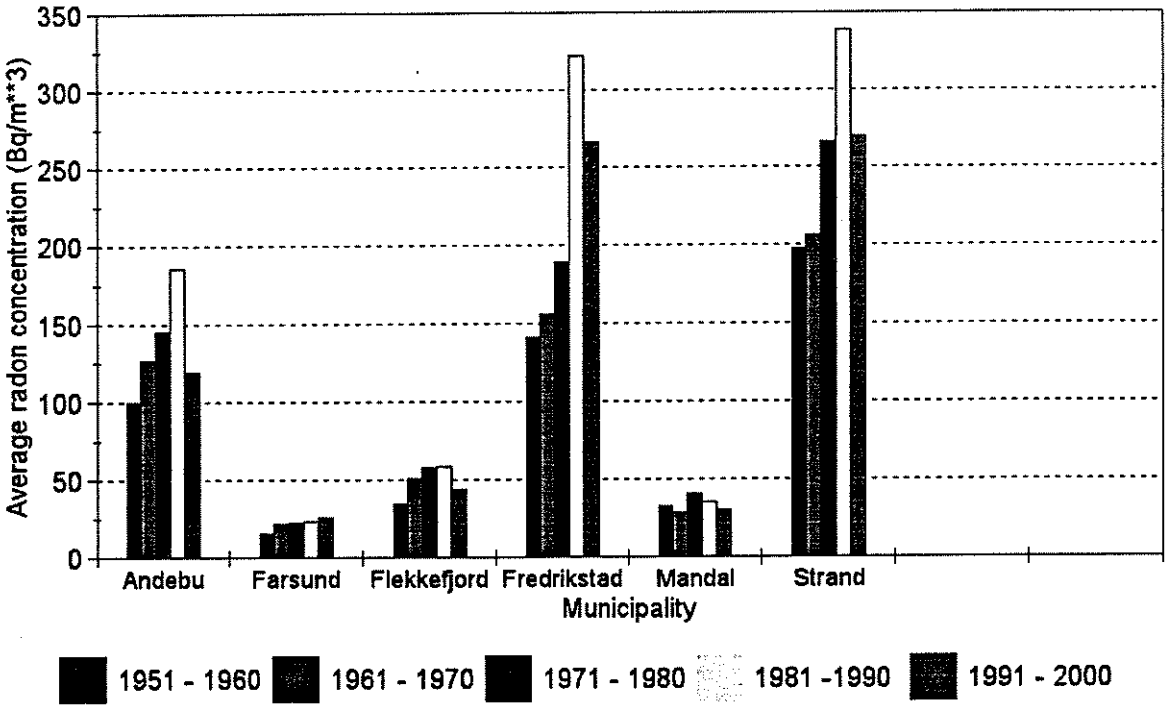

Figure 1. Average radon concentrations in six Norwegian municipalities grouped by the year the houses were built.

The most common types of dwellings in Norway are detached, semi-detached, row or terraced houses. Close to ¾ of the Norwegian population live in such houses. Incidentally radon concentration in these types of dwellings is on average higher than in flats in multi-storey buildings or smaller horizontally divided dwellings. About 92 % of the new data is from detached, terraced, semi-detached or row houses. To compare the new data set with the 87 - 89 survey data set it is necessary to match the type of data considered. We shall therefore narrow our focus and consider only detached, semi-detached, row and terraced houses in both data sets. In addition we shall consider only bedroom measurements in the new data, since only such measurements are available in the 87 - 89 data set.

In the new data set approximately 2000 measured dwellings were detached, semi-detached, row or terraced houses. A subset of 1049 of these houses had a measurement in a bedroom. Measurements in these 1049 houses are the basic new data set considered in further discussion. It is assumed that the selected houses are close to randomly distributed within the municipalities. But the municipalities themselves were not chosen randomly since they conducted their local surveys at their own initiative. The municipalities in which high radon levels were discovered in 1987 - 89 were more likely to conduct follow-up surveys. This gives rise to a considerable bias in the new data set which is reflected in the overweight of very high radon levels. In the new data set from six Norwegian municipalities (see Figure 1) the average radon concentration ranges from 22 Bq/m³ to 249 Bq/m³ with the average of 142 Bq/m³ and the median of 94 Bq/m³. Table 1. shows the number of measurements, average radon concentration and standard deviation for each municipality. If used directly, the average radon concentration calculated from this biased data would most probably overestimate the radon average concentration for the whole country.

Municipality	# of measurements	Average radon conc. (Bq/m³)	Standard deviation (Bq/m³)
Andebu	85	134	101
Farsund	199	22	20
Flekkefjord	50	69	194
Fredrikstad	549	189	226
Mandal	73	33	27
Strand	93	249	251

Table 1. The table shows number of measurements, average radon concentration and standard deviation in each municipality.

An indirect way to use the information contained in the new data set would be possible if there exist common parameters on which both unbiased (survey 87 - 89) and biased (new 1995 data) depend in the same way. If such parameter can explain the variation in the biased data then it can also be used to indicate variation in the unbiased data.

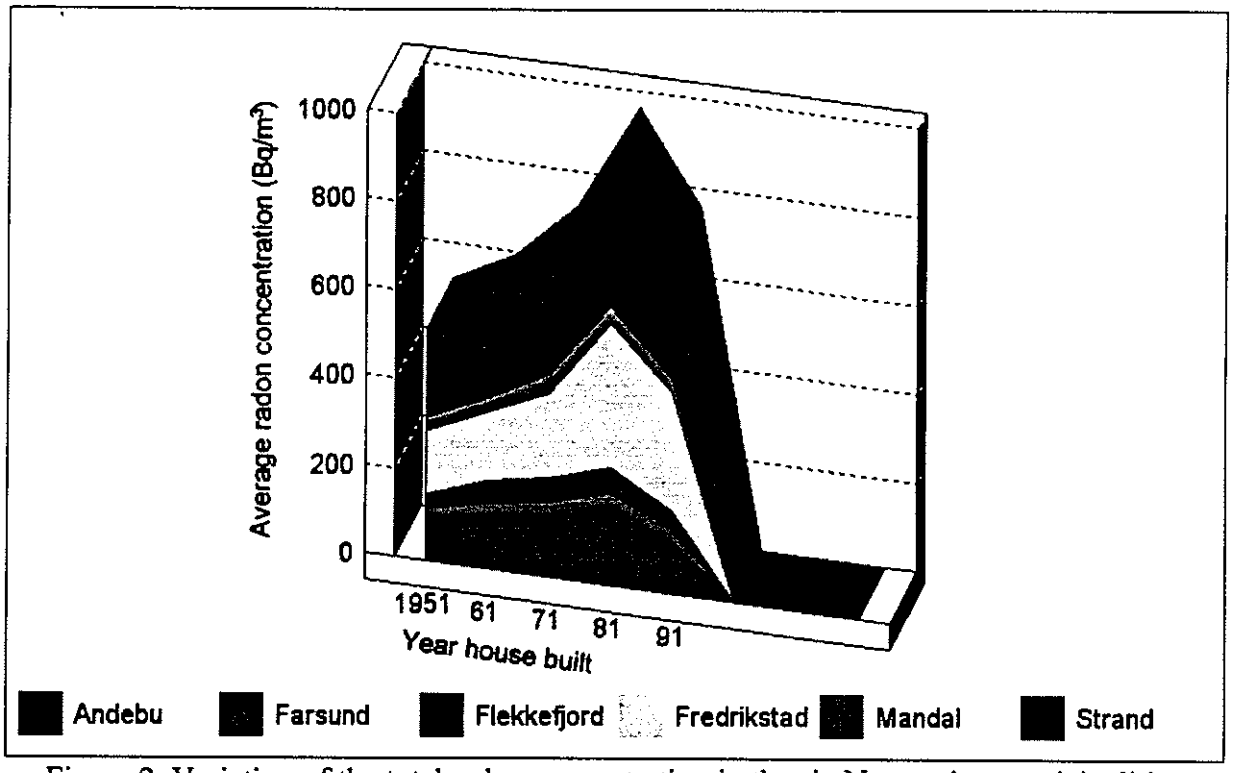

Figure 2. Variation of the total radon concentration in the six Norwegian municipalities as a function of the year the houses were built.

It appears that such a parameter is the age of the house. In Figure 1 the average radon concentration is plotted against the year the house was built for the 6 Norwegian municipalities that conducted radon measurements in 1995. The data show increased radon levels in newer houses. This effect is most clearly seen in the municipalities Andebu, Fredrikstad and Strand which all have radon levels above 100 Bq/m³. Similar trend can be observed in municipalities with lower radon levels (< 70 Bq/m³): Farsund, Flekkefjord and

Mandal, but the magnitude of the variation is less. Such patterns in the data have been observed before in both Norway [2] and Sweden [3] and is usually ascribed to excessively gas-tight building technique and insufficient ventilation in the houses built in seventies and eighties.

Houses built in the nineties though have lower radon levels than houses built in the eighties. It is believed that the very new houses which commonly have concrete floors are airtight with respect too inflow of soil gas immediately after they are built, and perhaps a few years after. However the concrete floor construction will gradually, over the years crack and the leakage of the soil gas will increase. It is also assumed that the use of balanced ventilation systems in some of the very new houses could account for the observed reduction in radon levels. In Figure 2 the total radon concentration for the six municipalities in the 1995 data set is plotted against the year the houses were built.

Data presented in Figure 1 can be, for the sake of simplicity grouped into two groups: before and after 1980. This is shown in Figure 3a. In Figure 3b relative change in radon concentration for the two groups of houses in each municipality are given. For the municipalities with low radon levels this trend represents on average an increase factor of 1.1 (or 10%). High level municipalities on the other hand have trend of 1.5 (or 50%). These factors will later be used to estimate the correction to the unbiased average radon concentration found in the 87-89 survey.

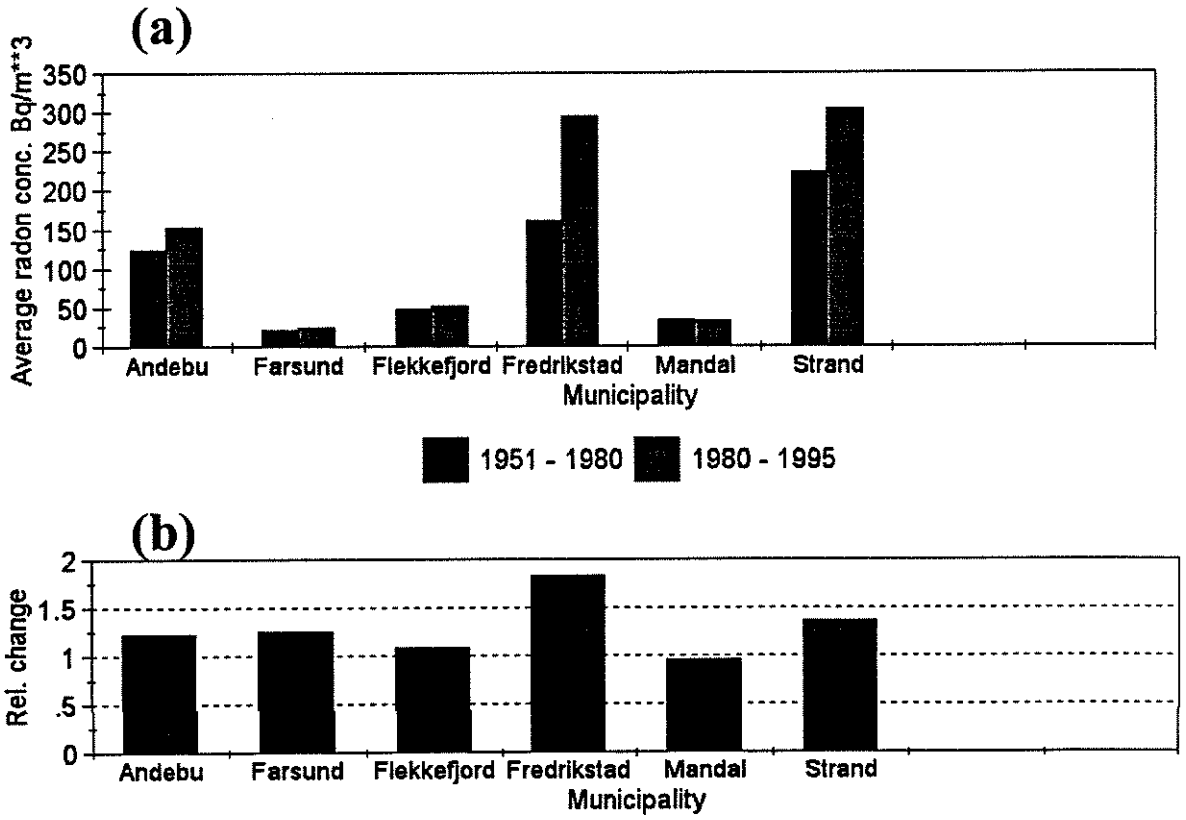

Figure 3. (a) Average radon concentrations before and after 1980. (b) Relative increase in radon concentration for the houses built after 1980 compared to those built before 1980 in the same municipalities.

According to Central Bureau of Statistics [4] the following data concerning the numbers of houses in Norway until year 1995 are available. Of approximately 1.336 million detached and semi-detached houses in Norway, 0.288 millions have been built after 1980. The number of detached and semi-detached houses in 1980 was approximately 1.048 millions.

Some assumptions have to be adopted in order to employ the above mentioned parameters to estimate the correction. Firstly, it is assumed that there are different trends for municipalities with low average radon concentration (RLL = 1.1) and high average radon concentration (RHL = 1.5). Secondly, the average radon concentration in 1.064 mill houses built before 1980 is assumed to be unchanged in 1995.

The 87-89 survey gives 56 Bq/m³ as the average for detached and semi-detached houses with radon measurements conducted in bedrooms. The average concentration for the whole house is then 61 Bq/m³, and with the estimated 10% correction due to mild winters 67 Bq/m³ is obtained. This value 67 Bq/m³ is the unbiased average radon concentration for the whole country in the detached and semi-detached houses and the starting point for correction calculation.

Instead of calculating a single value we may rather try to estimate the expected range within which the present average radon concentration is expected to be found. The basis for the correction is the dependence of radon level on the age of the house and there the two relative trends RLL=1.1 and RHL=1.5 come in. For calculation of the lower boundary of the expected range we may apply RLL, and for the upper boundary we use RHL. Thus we have the following weighted average:

$$\frac{(67\ \mathrm{Bq}/\mathrm{m}^3 \times 1.048) + (67\ \mathrm{Bq}/\mathrm{m}^3 \times \mathrm{RLL}) \times 0.288}{1.048 + 0.288} = 68\ \mathrm{Bq}/\mathrm{m}^3$$

for the lower boundary, and

$$\frac{(67\ \mathrm{Bq}/\mathrm{m}^3 \times 1.048) + (67\ \mathrm{Bq}/\mathrm{m}^3 \times \mathrm{RHL}) \times 0.288}{1.048 + 0.288} = 74\ \mathrm{Bq}/\mathrm{m}^3$$

for the upper boundary. The weighing factors in these two equations are the 0.288 million houses built in the period 1980 - 1995, and the 1.048 million houses that existed in 1980.

The average radon concentration in Norwegian detached, semi-detached, row and terraced houses at present is estimated to be in the range 68 - 74 Bq/m³. The actual uncertainty is larger than the estimated range implies, and the main contribution is assumed to be the use of single parameter: the age of the house and the indirect method of correction. Similar procedure can be applied to calculate other average quantities in the unbiased sample.

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Radonmätning på arbetsplatser i Sverige

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Abstract

Förekomst av radon och radondöttrar i inandningsluft utgör en riskfaktor för lungcancer. Radon bildas genom sönderfall av radioaktiva ämnen i berggrunden och i byggnadsmaterial.

För bostäder och lokaler för allmänheten i Sverige har Socialstyrelsen fastställt gränsvärdet 400 Bq/m^3 som årsmedelvärde för radongas. För arbetslokaler kommer enligt Arbetarskyddsstyrelsen att gälla för underjordsarbete en högsta årsdos på 2,5 MBq/m^3, vilket motsvarar ett gränsvärde på 1500 Bq/m^3 som årsmedelvärde. För övriga arbetsplatser ska 400 Bq/m^3 gälla som gränsvärde. I båda fallen har man samrått med Statens Strålskyddsinstitut.

För gruvor och annat bergarbete har gränsvärde för radonexponering funnits länge i Sverige, liksom även krav på mätning av halten radon eller radondöttrar. För övriga arbetsplatser finns inget mätkrav. Det har därför gjorts ganska få sådana mätningar. Däremot har man gjort omfattande mätningar av radon i bostäder. De senaste åren har dock mätningar av radon i lokaler av typ skolor och daghem genomförts i vissa kommuner, men långt ifrån alla.

För att få en bättre uppfattning om radonsituationen på vanliga arbetsplatser har Arbetarskyddsstyrelsen och yrkesinspektionen genomfört ett mätprojekt. Yrkesinspektionen är Arbetarskyddsstyrelsens regionala organ, som bedriver tillsyn på arbetsplatser och kontrollerar att de följer arbetsmiljölagen och de föreskrifter som styrelsen har utfärdat med stöd av lagen.

För att begränsa omfattningen av projektet väljs främst arbetsplatser där man kan förvänta förhöjda radonhalter. Exempel på sådana är

1. lokaler i bergrum
2. lokaler i källar-, souterräng- eller bottenplan av byggnader på radonriskmark
3. lokaler i hus som är byggda i blå lättbetong, främst om mekanisk ventilation saknas eller misstänks vara otillräcklig

För arbetsplatser på lermark eller liknande utan särskild radonrisk finns ingen anledning att förvänta några höga halter av radon. Det finns då ingen anledning att lägga mätresurser där.

Om höga radonhalter påträffas kan yrkesinspektionen förelägga arbetsgivaren att vidta åtgärder mot detta. Radonbrunn och mekanisk ventilation är vanliga åtgärder mot höga radonhalter.

För projektets genomförande har mätinstrument inköps, dels ett registrerande instrument (Alfaguard) med datorprogram för utvärdering och behandling av mätdata, dels av typ elektret (E-perm). E-perm ger bruttohalt under mätperioden, vanligen några dygn eller en vecka. Alfaguard kan ge diagram med radonhaltens variation under dygnet, max-, min- och medelvärden.

Bakgrund

Förekomst av radon och radondöttrar i inandningsluft utgör en riskfaktor för lungcancer. I Sverige har risken med radon har fått ökad uppmärksamhet på senare år, och gränsvärdet för radon i bostäder sänktes 1990. Statens strålskyddsinstitut har föreslagit åtgärder från kommuner och centrala myndigheter för att kartlägga radonsituationen och åtgärda byggnader med höga radonhalter.

Arbetarskyddsstyrelsen har åtagit sig att inom sitt ansvarsområde se över gränsvärdet för radon, överväga krav på radonkontroll på arbetsplatser och utarbeta information om radon i arbetslokaler. Frågan om gränsvärde för radon på arbetsplatser har remissbehandlats under våren 1996. I augusti 1996 fastställdes **400 Bq/m^3** som gränsvärde för arbetsplatser, utom där berg bearbetas under jord. För dessa gäller en årsdos för exponering för radon på **2,5 MBqh/m^3**. Med en arbetstid under jord på ca 1600 h/år (36 h/vecka under ca 45 veckor/år) motsvarar det **ca 1500 Bq/m^3.**

Gränsvärden och uppmätta halter av radon uttrycks numera vanligen som halt radon (dvs radongas), vilket konsekvent tillämpas här. Förhållandet mellan halt av radondöttrar och radon uttrycks genom den s k jämviktsfaktorn, som ofta har satts till 0,5. Statens Strålskyddsinstitut har dock nu beslutat att den ska anses vara 0,4, om man inte genom särskilda mätningar har bestämt förhållandet. För gruvor kan förhållandet dock variera ganska mycket, och Strålskyddsinstitutet räknar här ännu med 0,5.

Syftet med detta projekt är att få ett mätunderlag som belyser radonsituationen på våra arbetsplatser. Projektet ska ge svar på frågan om hur det nya radongränsvärdet kan efterlevas. Om ett krav på radonmätningar på arbetsplatser ska övervägas, behövs också ett gott underlag som medger uppskattning av kostnader.

Omfattning av projektet

I ett centralt projekt om radon har fem distrikt inom yrkesinspektionen deltagit: Falun, Linköping, Luleå, Stockholm och Örebro. Yrkesinspektionen (YI) i Sverige består av 11 distrikt.

Val av mätplatser och omfattning av mätningarna har distrikten fått göra själva, med beaktande av de råd som projektledningen vid Arbetarskyddsstyrelsen har lämnat. Distrikten har redovisat resultat i form av en kortfattad rapport från varje arbetsställe där man gjort en mätning samt vissa sammanställningar och kommentarer.

Metod och mätinstrument

Radon kan mätas på två olika sätt, med direktvisande instrument eller dosimeter. För kontroll av radonhalt i bostäder är dosimeter den vanligaste metoden. Man hänger då upp en dosimeter, en liten dosa som innehåller en filmremsa i bostaden under en viss tid, vanligen 1-3 månader. Sedan utvärderas filmremsan på laboratorium. Resultatet ger medelhalten under hela denna tid.

Filmdosimetrar är enkla och billiga, men de har nackdelen att mätning måste pågå under ganska lång tid. Dessutom kan de inte ge några upplysningar om hur radonhalten har varierat under mättiden.

Direktvisande instrument är dyrare i inköp och även i användning, eftersom de kräver manuell hantering av utbildad personal. Men de ger snabba upplysningar om radonhalten och kan även kartlägga hur halten varierar under veckan och dygnet.

I detta projekt har använts dels ett direktvisande instrument, Alfaguard, dels ett instrument, E-perm, som fungerar som en dosimeter men kan användas under kortare tid än filmdosimetrar. Med E-perm kan man mäta under ett par dagar eller någon vecka. E-perm bygger på en elektret, en laddad platta som laddas ur av alfapartiklar från sönderfallande radon. Avläsning sker med en voltmeter direkt efter mätperiodens slut. Man får därför svar på arbetsplatsen utan att behöva skicka något till ett laboratorium för utvärdering.

Alfaguard är ett avancerat registrerande instrument som innehåller programvara för utvärdering i dator. Man får medelhalt, max- och minhalt av radon samt diagram över haltens variation med tiden. Detta är en stor fördel vid mätning av radon på arbetsplatser, där man ofta stänger av ventilationen nattetid för att spara energi. I så fall stiger radonhalten under natten, och E-perm kommer att mäta en medelhalt som är högre än den personalen utsätts för på dagen.

I projektet har E-perm ibland använts som ett screening-instrument för att sortera ut platser med höga radonhalter, där man då kan göra närmare mätning med Alfaguard. Om E-perm ger en låg halt behöver man inte mäta mer. Har ventilationen varit avstängd på natten så vet man att exponeringen under dagen är ännu lägre än den uppmätta medelhalten, och det räcker.

Resultat

Av utrymmesskäl redovisas endast sammandrag av resultat från de fem distrikten.

YI Falun: Radonhalter mellan 50 och 960 Bq/m^3 har uppmätts med E-perm, 28 mätvärden från 18 olika arbetsställen. Medelvärde: 243 Bq/m^3, median 125 Bq/m^3.

Vid fyra företag gjordes kompletterande mätningar med skrivande instrument, Alfaguard. Högsta förekommande radonhalt vid någon tidpunkt var ca 5000 Bq/m^3 i ett kontor där ventilationen inte var i drift.

YI Linköping: Mätningar har utförts januari - mars 1996. Radonhalter mellan 34 och 1226 Bq/m^3 uppmättes med E-perm, 21 mätvärden från 15 arbetsställen. Medelvärde: 165 Bq/m^3, median 104 Bq/m^3.

YI Luleå: Mätningar har genomförts under april och maj 1996. Mätobjekt har valts utifrån markradonkartor och omfattar främst arbetsplatser i högriskområden för markradon. Dessutom har några mätplatser valts i hus byggda i blåbetong. Radonhalter mellan <40 och 150 Bq/m^3 har uppmätts med E-perm, 17 mät-värden från 13 arbetsställen. Medianvärde: <40 Bq/m^3. Mätningar med registrerande instrument Alfaguard visade eller bekräftade att halterna under arbetstid var låga, <40 Bq/m^3.

YI Stockholm: I Stockholms distrikt har mätningar utförts vid företag i fyra kommuner (Stockholm, Sollentuna, Vallentuna och Uppsala). Radonhalten har mätts med E-perm och i nio fall även med Alfaguard, vanligen i flera mätpunkter vid företaget. Mätplatser har valts med hänsyn till berggrunden. Radon-halter mellan ca 0 och 1000 Bq/m^3 har uppmätts med E-perm, 40 mätvärden från 17 olika arbets-ställen, mätning 5-8 dygn under tiden 29/1 - 28/2 1996:. Medelvärde: 197 Bq/m^3, median 68 Bq/m^3.

Vid nio företag gjordes kompletterande mätningar med skrivande instrument, Alfaguard. Instrumentet kan ställas in på en mätperiod av t.ex. en timme och ger då medelhalt under denna timme. Det ger även osäkerheten i mätvärdet samt summerar och beräknar medelhalter samt anger högsta och lägsta halt under mätperioden. Mätningarna utfördes under tiden 29/1 - 22/2 1996. Diagram 1-6 återger radonhaltens variation under mätperioden.

YI Örebro: I Örebro-distriktet genomfördes mätningar på ca 30 arbetsplatser, belägna i Västmanlands län och Örebro län. Mätningar utförda med E-perm under 3-14 dygn i mars 1996. Alla resultat har ännu inte inrapporterats, men några högra halter föreligger (>1000 Bq/m^3), bl.a. från kyrkor.

Sammanfattning

Projektet visar att radonhalten kan variera ganska avsevärt mellan olika arbetsplatser, även mellan sådana där man tror att den bör vara ungefär densamma. Markförhållanden och ventilation betyder mycket för radonhalten i lokalerna. Mätning på arbetsplatser där det fanns blå lättbetong i väggarna gav ganska låga halter. Det får tolkas som att markradon betyder mer än radon från byggnadsmaterial. Detta har iakttagits tidigare också. På arbetsplatser har man vanligen mekanisk ventilation och alltså inte de problem som kan finnas i villor av blå lättbetong med bara självdragsventilation.

Mätning med E-perm kan användas som screeningmetod för att sortera ut arbetsplatser där radonhalten är klart förhöjd. Eftersom E-perm mäter medelhalten dygnet runt under hela mätperioden, mäter den sällan den faktiska exponeringen för radon. De flesta arbetsplatser är obemannade nattetid, och då är det vanligt att ventilationen stängs av eller minskas påtagligt för att spara energi. Bruttohalten ger då en övre gräns för vad halten kan vara under dagen: man vet att den är klart lägre men inte hur mycket lägre. Mätning med Alfaguard har i en del fall gett den upplysningen i projektet.

Hur ventilationen inverkar och hur lång tid det tar att vädra ut en hög radonhalt som har byggts upp under en natt eller en helg kan man studera i diagrammen. Observera att mätperiodens längd är olika i olika diagram.

Diagram Företag A visar haltens variation under en hel vecka med början en måndag. Man kan se hur halten vanligen stiger under natten och sjunker under dagen. Halten är lägre under helgen, vilket kan förvåna. Det kan ha samband med att processventilation som är igång bara ibland skapar ett undertryck som får mer markradon att läcka in, och under helgen finns inget sådant undertryck.

Nästa diagram Företag B omfattar bara två hela dygn och visar en typisk stegring av radonhalten nattetid när ventilationen är avstängd. Företag F, mätning en hel vecka, visar samma regelbundna dygnsvariation under veckan men ett annat utseende under söndagsdygnet, då ingen ventilation sänker radonhalten på dagen.

Diagram Företag L visar tydligt effekten av nattens uppbyggnad av radonhalten och den snabba sänkningen när ventilationen sätts igång på morgonen. Ventilationen har uppenbarligen mycket god luftomsättning. Företag M (två hela dygn) visar något liknande.

Det sista diagrammet Företag P visar hur radonhalten stiger under en helg och är särskilt upplysande med mycket höga halter när ventilationen sätts igång måndag morgon. Mätningen omfattar torsdag morgon - tisdag morgon. Vi ser den relativt måttliga uppbyggnaden av radon under tre nätter mellan arbetsdagar och den mycket kraftiga stegring som börjar fredag kväll och når ca 2500 Bq/m^3 under söndagen. Arbetsplatsen låg liksom nästan alla de uppmätta i distriktet på en grusås. Ventilationen är fullt tillräcklig för att sänka radonhalten till låga nivåer när den väl sätts igång.

Man bör ta hänsyn till tiden som krävs för att bygga upp resp vädra ut en viss radonhalt när man väljer tiden för att slå av resp sätta på ventilationen. Man bör inte välja den bara efter arbetstiden utan hellre slå av och på en eller ett par timmar tidigare.

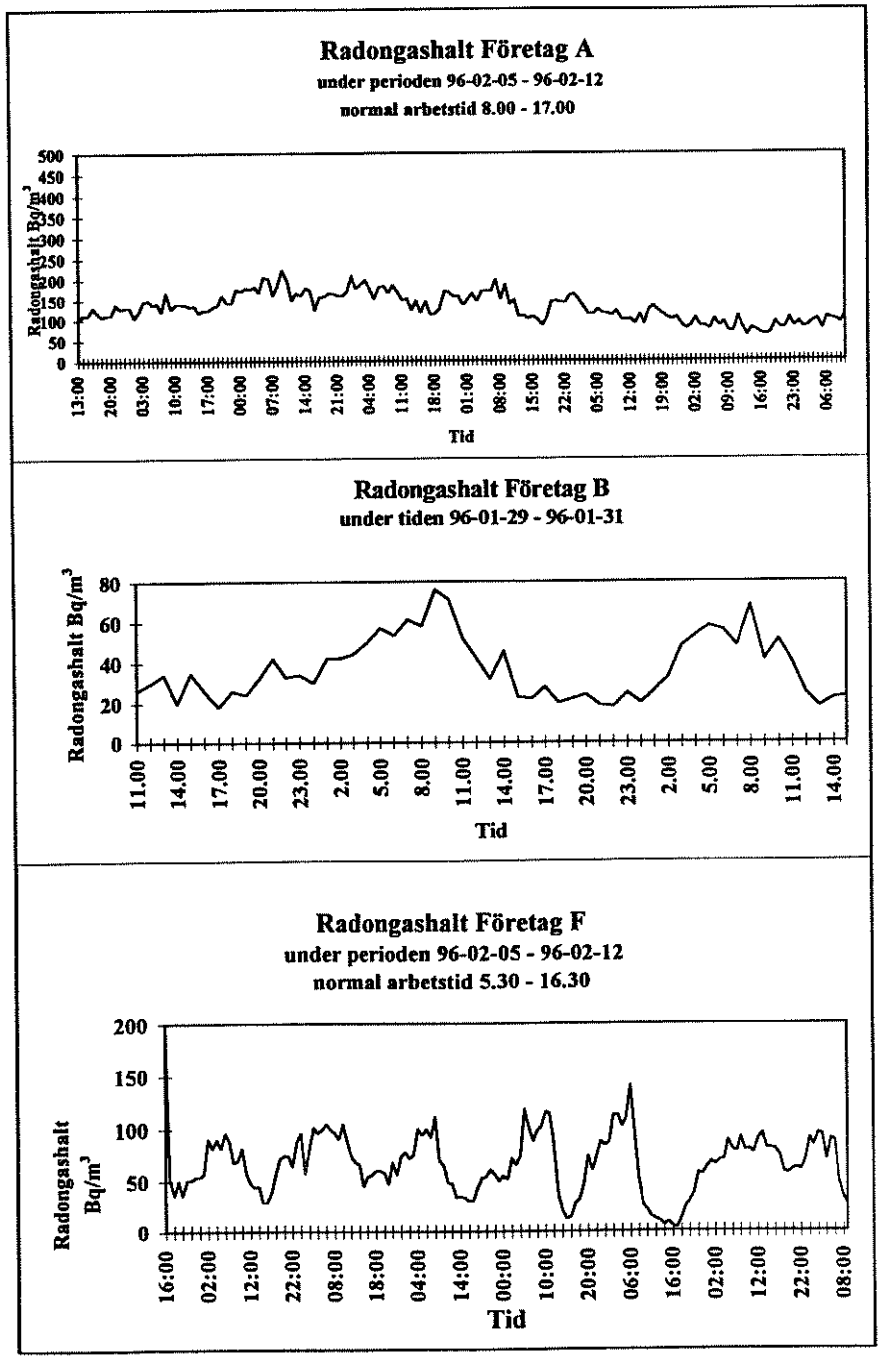
Genomgående visar de flesta arbetsplatser ganska låga radonhalter, trots att de har valts med hänsyn till markförhållanden m m, alltså efter var man kunde befara förhöjda halter. Även t. ex. en arbetsplats i Stockholm som är helt insprängd i berg hade låg radonhalt (ej mätbar med E-perm). Förklaringen är att det är en bilverkstad med tillräckligt dimensionerad ventilation. Dessutom hade man där ventilationen igång dygnet runt.

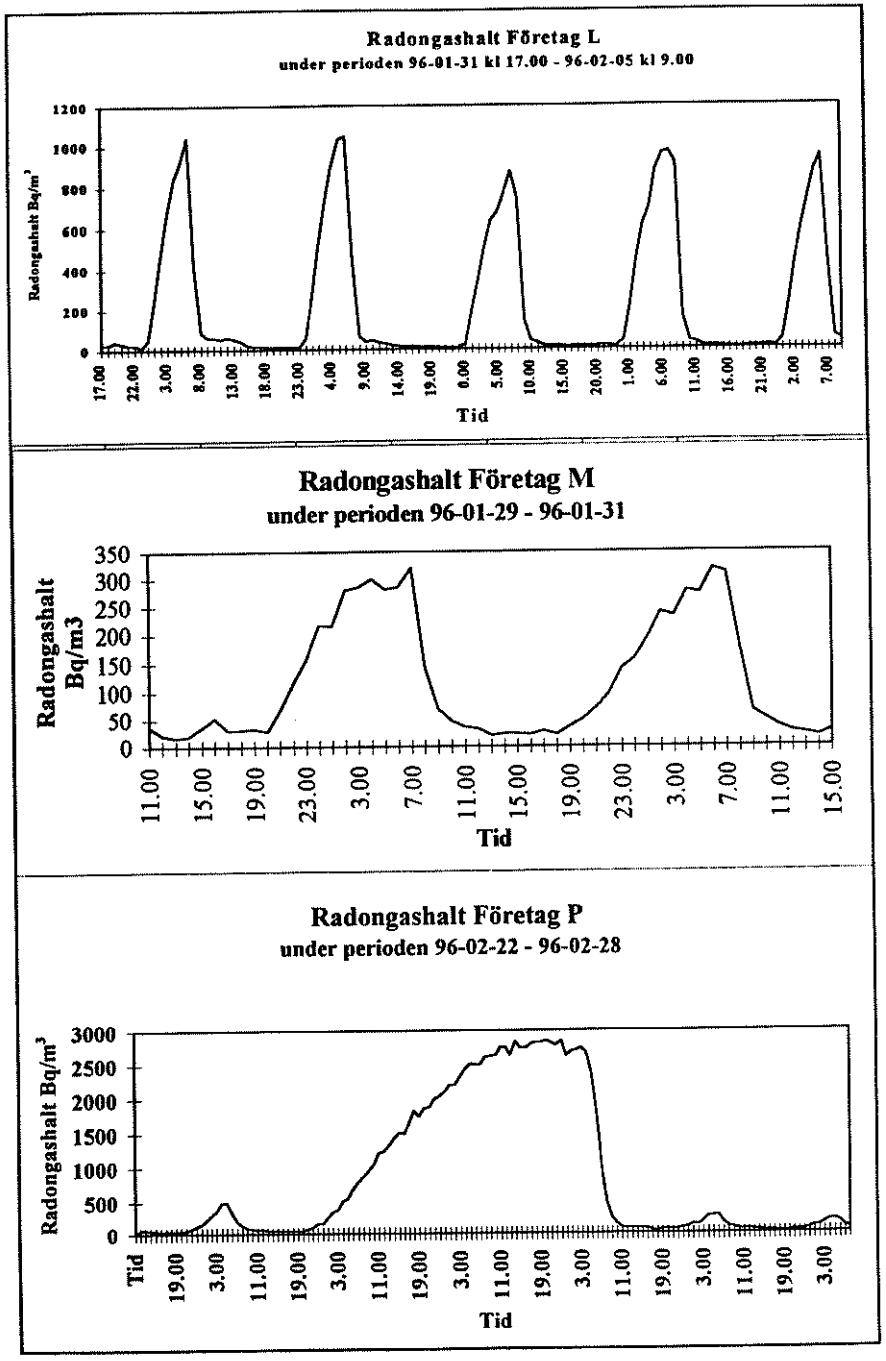
Ett resultat av projektet är att radonsituationen på landets arbetsplatser inte är alarmerande utan att man som regel klarar det nya gränsvärdet.

Arbetarskyddsstyrelsen har nu fastställt de gränsvärden för radon som redan nämnts. I samband med detta har övervägts om något krav bör ställas på arbetsplatserna att kontrollera radonhalten, dvs visa att man klarar gränsvärdet. Resultatet av projektet indikerar dock inte att ett sådant krav behövs. Enligt föreskrifterna om hygieniska gränsvärden gäller redan att om det finns grundad anledning misstänka att ämne finns i sådan halt att tillämpligt hygieniskt gränsvärde överskrids, skall förhållandet snarast utredas. Något krav utöver detta är knappast motiverat när det gäller radon.

För arbetsplatser under jord och en del arbetsplatser i källarplan och liknande på radonriskmark kan sådana misstankar finnas, och då bör man mäta. Men för de flesta arbetsplatser finns inte anledning till sådana misstankar, och då ger det inte så mycket att lägga mätresurser på radon.

Arbetarskyddsstyrelsen har publicerat mer fullständiga resultat i en rapport, Radonmätning på arbetsplatser, Rapport 1997:2.


Radongashalt Företag A
under perioden 96-02-05 - 96-02-12
normal arbetstid 8.00 - 17.00
Radongashalt Bq/m³
Tid
Radongashalt Företag B
under tiden 96-01-29 - 96-01-31
Radongashalt Bq/m³
Tid
Radongashalt Företag F
under perioden 96-02-05 - 96-02-12
normal arbetstid 5.30 - 16.30
Radongashalt Bq/m³
Tid


Radongashalt Företag L
under perioden 96-01-31 kl 17.00 - 96-02-05 kl 9.00
Radongashalt Bq/m³
Tid
Radongashalt Företag M
under perioden 96-01-29 - 96-01-31
Radongashalt Bq/m3
Tid
Radongashalt Företag P
under perioden 96-02-22 - 96-02-28
Radongashalt Bq/m³
Tid

Criteria's in evaluating thorax counting results of uranium fuel fabrication operators.

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Abstract

During the manufacturing in a nuclear fuel fabrication plant the employees are exposed to uranium materials of various characteristics.
Due to the low specific activity of commercial uranium (60 kBq/g) and most decays without significant gamma/beta contribution the Health Physics interest is mainly to minimise surface- and airborne contamination. All operators at ABB Atom are controlled at predetermined intervals by a specific thorax counting on determining inhaled uranium contamination. In determining the correct uranium lung burden chestwallthickness (CWT) has to be corrected for. If no correction is made the average ABB Atom employee CWT (24,5 mm) will cause an underestimation of 8,6 %.
A very high Cs-137 body burden sometimes could mean there is a problem in accurately determining the true U-235 lung burden. This paper explains the decision taking for some of the object characteristics determining the thorax counting results at ABB Atom.

Introduction

Assessment of uranium contamination due to inhalation comprises several parameters in determining a true lung burden.
Some of these parameters influencing the result are:

1. **Sources**
   Enrichment, chemical characteristics and particle (aerosol) size.
2. **Contamination route**
   Acute or continuos intake (inhalation and/ or ingestion).
   Matrix or non matrix distribution of the lung burden.
3. **Timeliness**
   Equilibrium uranium/uranium daughters.
4. **Measurement equipment and method**
   High or low resolution gamma spectroscopy, calibration practice and background subtraction.
5. **Object characteristics**
   Male or female, fat or thin, tall or small, skin contamination on chest surface or other parts of the body, ingestion of food containing Cs-137 food prior to measurement.

Background.

The ABB Atom fuel fabrication plant has been in operation for more than 25 years. The way of handling the uranium materials comprises several steps from conversion of UF_6, blending of powder, pelletising, pellet loading into rods and assembling as well as recycling of various uranium compounds by the use of dissolution and extraction techniques.
Commercial uranium on average has an isotopic mass enrichment of 3 % U-235. This means for each gram of uranium a total alpha activity of approximately 60 kBq.
Due to the decay and the various chemical processes the daughter products might be in equilibrium or disequilibrium with the uranium isotopes. In the different workshops

both male and female operators are exposed to uranium containing materials. All operators working with unsealed uranium material are regularly assessed upon U-235 lung burden by the use of a high resolution gamma spectrometric analysis on the U-235 gamma's penetrating the chest wall.

Material and methods.

The ABB Atom lung burden analysis is performed in a special steelroom having a reduced background radiation. In total 3 individual 70 mm, 2 cm thick HPGe detectors are used. The array of these detectors is shown in figure 1.

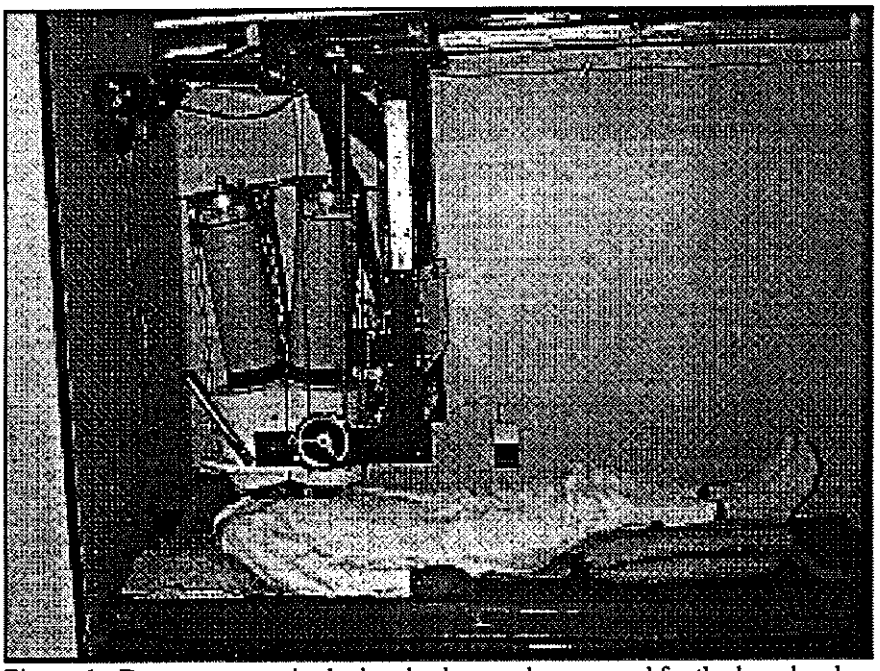

Figure 1. Detector set-up in the low background room used for the lung burden measurement of fuel fabrication operators.

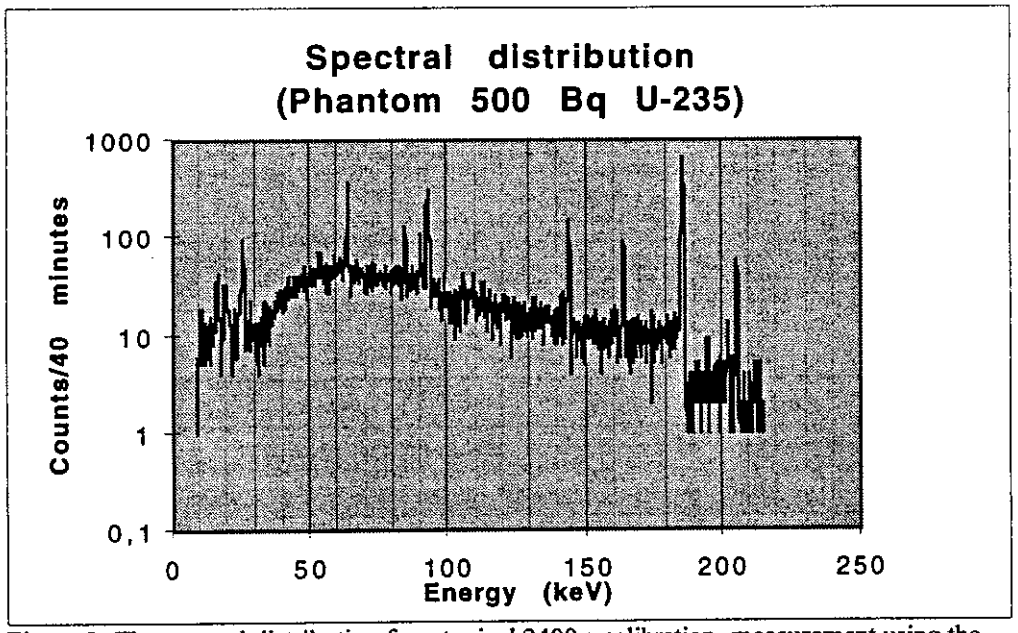

Figure 2. The spectral distribution for a typical 2400 s calibration- measurement using the livermore phantom labelled with 500 Bq U-235 (3 % enrichment).

Analysis of the pulse height distribution is based on predetermined energy regions. See table 1. For routine measurements on fuelfabrication operators the count rate is very much lower. Automatic peak search routines usually fail in finding the appropriate uranium peaks.
For calibration purposes a special Livermore thorax phantom is used (*Ref 1*). This phantom enables activity calibration by the use of uranium labelled lungs as well as background correction. In figure 2 , a typical phantom calibration spectra is seen.
During routine measurements all measured objects are assumed having the same physical shape, ed. chest wall thickness and lung size.
However, if significant uranium contamination is expected determination of the true U-235 retention from an intake needs a more thorough analysis of the actual object characteristics.

From the printout of a lung burden measurement the result is presented as:

1. Eq U-235 (U-daughters at 63, and 94 keV)
2. U-235 (143 and 185 keV)
3. Average U-235 (weighed)
4. Cs-137
5. K-40
6. Misc. (Co-60, etc.)

The Eq U-235 and the U-235 results could be further resolved in the various regions adequate for the analysis.
If the average result shows a U-235 lung burden of more than 5 Bq U-235 the result is always followed up by a new measurement a few days later.

Absorption in the chest wall might be significant and has to be corrected for. Table 1 shows some different situations of interest where the CWT- influence is considered. As shown in table 1 the interpretation of the average U-235 result might be over- or underestimated due to CWT.

Table 1. U-235 lung burden results (Bq-U-235) for various CWT.
(CWT_{ref} is the phantom chestwall thickness)

	keV	CWT=CWT_{ref}	CWT<CWT_{ref}	CWT>CWT_{ref}
	63	5	9	3
	94	5	7	4
Eq U-235	-	**5**	**8**	**3**
	143	5	7	4
	185	5	6	5
U-235	-	**5**	**6**	**5**
Average U-235	-	**5**	**7**	**4**
Cs-137		200	250	190

Results

The individual and median CWT (24,5 mm) for the fuel fabrication operators at ABB Atom is shown in figure 3. In the figure the area +/- 10 % deviation from the true value, the typical ABB CWT and the reference Phantom-CWT used for activity calibration purposes is shown.

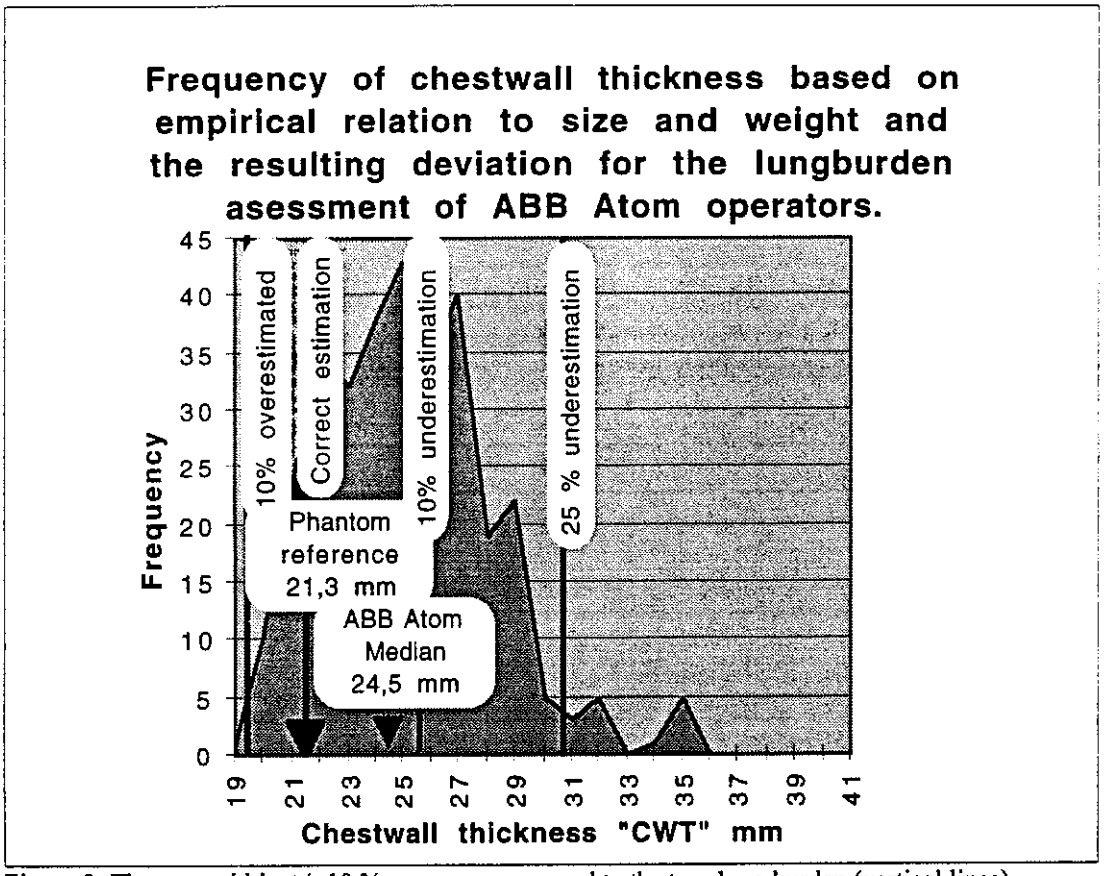

Figure 3. The area within +/- 10 % accuracy compared to the true lung burden (vertical lines). The corresponding distribution of the actual employee CWT at ABB Atom (shaded).

For calibration purposes the livermore phantom might be used with chest plates corresponding to a CWT of: 15.33, 21.3, 27.7, 32,4 or 39,8 mm. The phantom reference CWT is based on direct thickness measurements of the plates. All employee CWTs are based on the empirical formula:

CWT (mm)=45,7*W/(H)+4,4
where: W is the weight in kg and
H is the height in cm.

This algorithm has been used elsewhere (*Ref 2*) for lungburden analysis and shown to be accurate enough for most thorax measurements on males. Roughly 60 % of the ABB Atom employees has a CWT between 19.4 and 25.2 mm. This is in accordance with data on other employees working in fuel fabrication industries. However it is recognised that other techniques exists (ultrasonic devises) and various algorithms taking also body parameters as chest circumference and fat content into account ,the W and H parameters are always measured at ABB Atom.

In selecting the adequate chest wall plate the one which amounts to 21.3 mm was chosen at ABB Atom, although the average employee CWT is 24,5 mm which means an underestimation of 8,6 % or a false low result of 4,5 Bq U-235 when the accurate value would be 5 Bq U-235.

During routine lung burden analysis a diagram as shown in figure 4 is used in determining the need for special CWT correction.

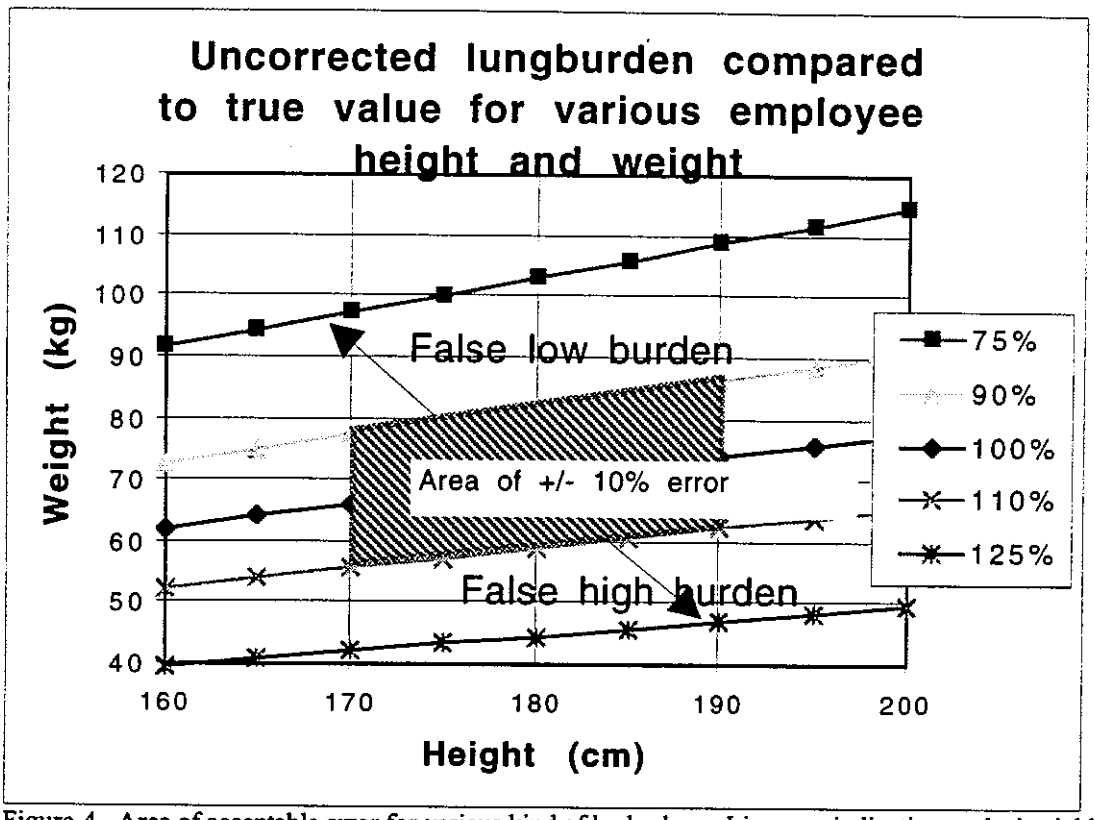

Figure 4. Area of acceptable error for various kind of body shape. Lines are indicating analysis yield compared to true value.

Besides chest wall thickness also the body burden of K-40 and Cs-137 is influencing the lung burden analysis of inhaled uranium.
As could be seen in figure 5 the body burden of ingested and retended Cs-137 has a strong influence on the U-235 MDA. Employees ingesting a lot of wild animal meet do normally have a higher MDA. A very high Cs-137 body burden sometimes could mean there is a problem in accurately determining the true U-235 lung burden. Some investigations on measurements of ABB Atom employees shows no correlation between Cs-137-K-40 body burden. The influence of Cs-137 is nowadays not as strong as it was during the first 5 years after the Chernobyl fallout.

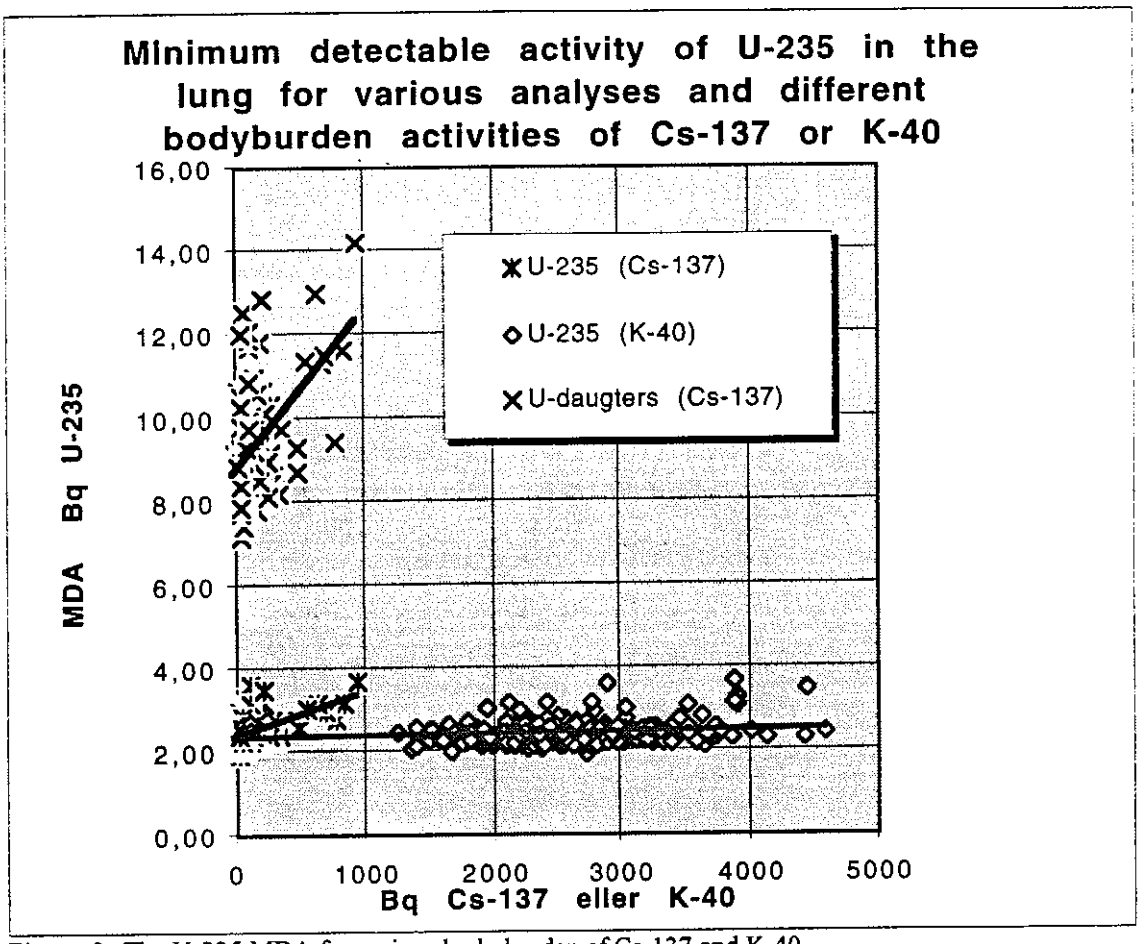

Figure 5. The U-235 MDA for various body burden of Cs-137 and K-40.

Conclusion

In determining the true lung burden of inhaled uranium compounds the CWT correction might be important. Based on weight and height a simple empirical calculation formula will give a CWT approximation.

At ABB Atom this approximation shows that lung burden analysis will be accurate within +/- 10 % for most of the employees.

References

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NORM in Oil Production - Occupational Doses and Environmental Aspects

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Abstract

Radioactive deposits, often referred to as LSA (Low Spesific Activity) scale, can under certain circumstances be formed inside production equipment in oil production. These deposits contain elevated levels of radioactivity, mainly ^{226}Ra, ^{228}Ra and their daughter products, and represents a waste problem for the oil industry.

Most of the radioactivity from the reservoir is disposed with produced water in to the sea. Extensive measurements of levels of radioactivity in produced water and LSA scale have been performed for several North Sea installations, and some of the results will be presented.

External exposure to workers has been measured during typical operations both off shore and on shore involving handling of contaminated equipment. Because of the low ALI values of the radionuclides in question, the potential internal exposure to workers is of particular importance. Release of dust from contaminated items is not found to be significant during normal operations, but internal hazards are still considered to be of major importance when discussing the potential exposures.

Several alternatives for waste disposal are under advisement by the Norwegian authorities, and these alternatives will be discussed.

Future work will consist of development of reliable methods for classification of equipment as above the exemption level or not, harmonisation of legislations and guidelines among the North Sea countries, and development of standard routines for handling contaminated equipment.

Introduction

Enhanced levels of radioactivity in deposits in North Sea oil production were first discovered in 1981, and are now found in the production system of several of the North Sea oil fields. The activity concentration in the deposits range from background level to several hundred Bq/g of ^{226}Ra (Smith 1987).

Deposits can be divided into two main categories; sulphate- and carbonate deposits. Sulphate deposits consists mainly of barium sulphate ($BaSO_4$), while carbonate deposits consist of calcium carbonate ($CaCO_3$).

Sea water is injected into the reservoir to maintain the pressure as the oil is removed. Mixing of sea water and formation water creates an incompatible solutions, and sulphates (mainly $BaSO_4$ and $SrSO_4$) are precipitated. The dominating radioactive elements in geologic formations in the sea floor is potassium and the elements in the uranium- and thorium series. Radium exists in both series as ^{226}Ra and ^{228}Ra. Radium will under certain circumstances leak from the formation and be dissolved in the formation water. Like barium and strontium, radium is part of group IIA in the periodic chart and have similar chemical properties. Consequently radium is coprecipitated with barium and strontium as radium sulphate ($RaSO_4$), even though its solubility product is not exceeded. Turbulent flow in the production system will then cause the precipitated sulphate salts to attach to the walls and form deposits with enhanced levels of radioactivity. These deposits are often referred to as *LSA (Low Spesific Activity) scale*.

Carbonate deposits is formed as a result of both pressure and temperature in production strings decreasing with distance from the reservoir. The solubility product of carbonate decreases and calcium carbonate is precipitated. These deposits do not contain enhanced levels of radioactivity.

Materials and methods

Gamma spectroscopic measurements on samples of LSA scale and produced water were carried out at the Low Level Gamma Laboratory at the Norwegian Radiation Protection Authority, using High Purity Germanium (HPGe) detectors. The activity concentration of each radionuclide, related to a specified reference time and the density of the sample, were calculated.

Measurements of external exposure to workers were carried out using Harshaw thermoluminicense dosimeters, of CaF_2:Dy (TLD-200). The detection limit is approximately 2,5 μGy for ^{60}Co. During field measurements, four TLD's are installed in a plastic badge, of which two of them are shielded on both sides by circular filters of 2 mm brass. The filter smoothes out the energy response in the range of approximately 60 - 200 keV. The TLD`s are individually calibrated before exposure, and the individual uncertainty is reduced to approximately 1%. For further details on this method of TLD measurements, the readers are referred to Wøhni (1993).

Activity in produced water and deposits

Produced water

The samples were taken during normal operation and analysed at the NRPA. The results are shown in table 1.

Table 1. Activity concentration (in Bq/l) of ^{226}Ra and ^{228}Ra in produced water, and amount of water per day.

Platform no.	Sample place	^{226}Ra	^{228}Ra	Amount, m^3/day
1	Flotation cell	n.d.	0,3 - 0,6	n.m.
1	Degassing tank	n.d.	0,5 -0,7	n.m.
2	Flotation cell	3,5	1,5 - 4,2	2229 - 3020
2	Degassing tank	0,7 - 7,6	0,5 - 1,4	15661 - 17545
3	Flotation cell	1,9 - 2,5	1,7 - 2,1	2078 - 2168
3	Degassing tank	2,2	2,4 - 2,7	n.m.
4	Flotation cell	n.d.	1,3 - 2,0	11602 - 11979
5	Flotation cell	n.d.	1,7 - 4,3	14724 - 18000
6	Flotation cell	1,3	0,5 - 1,8	12675 - 12765
7	Degassing tank	n.d.	n.d.	2385 - 2477
8	Degassing tank	2,5	1,1 - 3,3	n.m.
9	Degassing tank	10,4	10,0	n.m.
10	Degassing tank	7,7	7,8	n.m.
11	Valve	3,0 - 5,6	0,8 - 1,1	n.m.

n.d.: not detected. n.m.: not measured.

With reference to table 1, ^{226}Ra and ^{228}Ra activity concentrations appear to be in a narrow range, 0,7 - 10,4 Bq/l for ^{226}Ra and 0,3 - 10,0 Bq/l for ^{228}Ra. The concentrations are 100 - 1000 times higher than the usual concentrations in the North Sea (IAEA, 1990). The measurements lie in the range of measurements of activity concentrations in produced water in USA, Russia and Germany (Snavely, 1989 and API, 1991).

Deposits

Samples of LSA scale, sludge and sand was taken from equipment like pumps, valves and separators during revision stop summer 1995. The samples was analysed at the NRPA. The results are shows in table 2.

Table 2. Activity concentration (in Bq/g) of ^{226}Ra, ^{228}Ra and ^{210}Pb in solid samples.

Sample type	Number of samples	^{226}Ra	^{228}Ra	^{210}Pb (# measurements)
Hard LSA scale	15	4,0 - 32,3	2,6 - 33,5	0,1 - 0,3 (5)
Porous LSA scale	13	0,3 - 24,2	0,3 - 18,8	0,1 - 0,2 (3)
Sand	26	0,1 - 21,9	0,1- 12,7	0,0 - 0,5 (8)
Sludge	15	0,1 - 4,7	0,1 - 4,6	0,3 - 0,7 (4)

Table 2 shows that hard LSA scale have higher activity than porous LSA scale. The raw data shows that 11 of 15 samples of hard LSA scale and only 3 of 13 samples of porous LSA scale have activity above the Norwegian exemption level of 10 Bq/g of ^{226}Ra. Hard and porous LSA scale are suspected to consist mainly of sulphate and carbonate salts, respectively, and further analysis will be conducted to verify this. The sand samples show a wide range of activity. These samples are mainly taken from separators decontaminated with high pressure sand jetting, and the samples contain varying amounts of sand. The activity concentration in sludge appear to be in a narrow range. The activity of ^{210}Pb are generally low in all samples, opposite to what have been found in for example the Netherlands.

Occupational exposure

External exposure

A total of 20 measurements of external exposure to workers during cleaning and handling of contaminated equipment have been carried out by the NRPA. Only one of the measurements showed a doserate above background. This measurement was carried out on shore during cleaning of contaminated tubulars. The doserate was calculated to 0,024 mSv/y, taken into account that on shore workers in Norway are estimated to be in contact with contaminated equipment 10% of total working time (Ørgersen, 1996). These estimates are in accordance with results from England and USA where doses are reported to be 0,027 and 0,032 mSv/y, respectively (Kristensen, 1994).

Internal exposure

The NRPA have not carried out measurements of internal exposure off shore. Two measurement of dust generation was carried out during different cleaning operations on shore. The first operation was cleaning of pall rings, using high pressure water jetting inside a closed compartment. No dust generation was measured. The second operation was cleaning of contaminated tubulars, also using high pressure water jetting. The dust generation was 0,2 mg/m^3, and the activity in the dust was 30 Bq/g of ^{226}Ra and 20 Bq/g of ^{228}Ra. Assuming a 3:1 relation between ^{226}Ra and ^{228}Ra, and an AMAD of 1 μm (Kristensen, 1994), the dose was calculated to be 0,046 mSv/y, using LUDEP 2.0. (Jarvis 1996).

Waste disposal

Radioactive deposits represent a significant waste problem for the oil industry. Since the problem was discovered in 1981, waste from cleaning of contaminated equipment has been sent to a temporary storage facility at Kjeller, north of Oslo. The total amount of LSA scale waste stored at Kjeller is at present about 80 tonnes, with a mean activity of 25 Bq/g of ^{226}Ra. The expected amount in the future is 20 tonnes per year.

In 1995, a temporary exemption level of 10 Bq/g of ^{226}Ra was introduced by the NRPA. All waste from the oil industry with activity above this level is defined as «radioactive waste». Originally, this waste was to be sent to the national depository for final storage. It has now become clear that the amounts of waste will exceed the amount expected when planning the

depository, and the authorities have decided that the oil industry is responsible for finding other solutions. A working group representing the oil companies, headed by the OLF (the Norwegian Oil Industry Association), suggests four alternatives:

- Reinjection into abandoned wells or by hydraulic fraction. Most equipment is taken on shore for decontamination. Whether or not the London and OSPAR convention allows that waste is taken off shore after cleaning needs to be clarified.
- Storage at Langøya. Langøya is an island in the Oslo fjord used for storage of inorganic waste. Langøya does not have a licence to store radioactive waste, and a new licence is needed before this storage option is available for the oil industry.
- Export of contaminated equipment. Norwegian cleaning companies does not have the necessary competence to decontaminate complicated pumps and valves. The equipment will be sent to cleaning companies abroad for decontamination and recycling.
- Release of waste into the sea. Cleaning of separators, tanks and some pumps and valves takes place offshore. According to the London and OSPAR convention, the waste from decontamination off shore can be released into the sea.

These solutions are currently under advisement by the Norwegian authorities.

Conclusions

When handling equipment contaminated with LSA scale, it is important to take the correct safety and protective measures. These measures should concentrate on limiting the inhalation of dust, and includes:

- Creating controlled areas for cleaning of contaminated equipment
- Keeping the LSA scale wet during cleaning
- Use dusts masks and protective clothing

Using these protective measures, the doses to workers will be low. Internal doses are about two times the external doses.Deposits with enhanced levels of radioactivity represents a waste and storage for the oil industry. Solutions for storage must be in accordance with international regulations.

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INDOOR RADON AND LUNG CANCER IN FINLAND

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Abstract Two Finnish epidemiological studies on indoor radon and lung cancer were completed in 1996. The nation-wide study was a nested case-control study, and it gave non-significant OR of 1% per 100 Bqm^{-3}, with an upper 95% confidence limit of 8%. The Eastern-Uusimaa study was also a case-control study, and it gave a non-significant trend, the OR of the highest tertile and its 95% CI:s being 1.5 (0.8,2,9). Both these studies failed to establish a significant risk of lung cancer associated with indoor radon exposure. Four indoor radon studies with more than 500 cases completed to date are presented. Their results are compared with the risk assessments from studies of underground miners. When looking at the OR:s and their 95% CI:s in the highest radon exposure categories, the results of these studies do not differ significantly from each other and from the risks assessed from the miner studies.

Introduction

Indoor radon is the greatest individual source of radiation exposure to the public. Experimental animal studies and human studies of miners have clearly demonstrated an excess risk of lung cancer with radon exposure. However, results of recent epidemiological studies of residential radon are inconclusive. We faced the same situation with the two radon studies completed recently in Finland. Both these studies failed to detect the risk of lung cancer associated with indoor radon exposure. Only one large-scale study dealing with residential radon and lung cancer (Pershagen et al. 1994) clearly demonstrated the risk due to indoor radon. This paper will deal with the controversy.

The nation-wide study

The study by Anssi Auvinen and his co-workers (Auvinen et al. 1996) is a nested case-control study. A base population was selected of all Finns living in the same single-family house from 1967 or earlier until the end of 1985. Using this cohort of 892,939 persons made possible the radon exposure assessments by measuring a single dwelling (so-called index dwelling) for each study subject.

In this cohort, 1,973 lung cancer cases between the years 1986-1992 were identified from the Finnish Cancer Registry. One control subject matched by year of birth and sex was identified for each case subject. Further controls were selected to be used if no information was available for primary control subjects, so the total number of control subjects was 2,885.

Information concerning smoking and other relevant types of exposure was collected from study subjects or their next of kin using questionnaires. The dates of starting and giving up smoking, and the amount typically smoked in the sixties and seventies were ascertained. So, the duration, intensity and the age at start of smoking were assessed. As to other types of exposure, information concerning relevant occupational exposure including asbestos was requested. Information concerning the average hours per day spent indoors in the index dwelling was also collected. An integrating radon measurement of 12 months was carried out in all index dwellings.

After the radon measurements and questionnaire survey, valid information from both the questionnaire survey and radon exposure was available for 517 matched pairs, and for the unmatched analysis for 1,055 cases and 1,544 controls. Conditional and unconditional logistic regression methods were used to analyse the data. After adjustment for smoking and occupational exposure to asbestos, the matched analysis gave an excess relative risk of 1% per 100 Bq/m^3. The 95% confidence interval ranged from the negative side to the upper limit of eight per cent. For five categories of indoor radon concentrations the matched odds ratios and their 95% confidence intervals are shown in Table 1. All confidence intervals include unity. The unconditional study gave almost identical results. No

statistically significant risk from indoor radon exposure was detected in any of the subgroups defined by age, sex, smoking, histological type of lung cancer, or asbestos exposure.

Table 1. Odds ratios and their 95% confidence intervals for five categories of radon exposure in the nation-wide study.

Exposure Bqm^{-3}	OR	95% confidence interval
0–49	1	
50–99	1.03	(0.84–1.26)
100–199	1.00	(0.78–1.29)
200–399	0.91	(0.66–1.35)
400–	1.15	(0.69–1.93)

The Eastern-Uusimaa study

The study by Ruosteenoja and her co-workers (Ruosteenoja et al. 1996) is a re-analysis of her previous case-control study (Ruosteenoja 1991). Nineteen rural municipalities in southern Finland with high levels of radon exposure were included in the study. All 238 cases of male lung cancer diagnosed during 1980-85 were identified. Controls were selected from a random sample of 1,500 men of the same age distribution as the cases. After a smoking inquiry, all 395 smokers and a random sample of 50 ex-smokers and 50 non-smokers were selected.

Two-month measurements during the winter were used to determine the radon exposure. They were aimed to be carried out in all dwellings where cases and controls lived from 1950 until 1975. The logistic regression method with two-stage analysis was employed to take fully into account the sampling design balanced with respect to smoking. The final number of cases and controls was 164 and 331. After adjustment for smoking intensity, giving up smoking, and age, the odds ratios and the 95% confidence intervals for the tertiles of indoor radon were 1.8 (0.9,3.5) and 1.5 (0.8,2.9), relative to the lowest tertile of radon concentration below 95 Bq/m^3.

Comparison of radon studies in mines and homes

As to the mining environment, the pooled data from eleven cohort studies of radon-exposed underground miners analysed by Lubin and his co-workers (Lubin et al. 1995) was large enough (65,000 men and more than 2,700 cases of lung cancer) to obtain risk assessments with more accurate estimates. A statistically significant trend was found. Statistically significant risk ratios were also found for all categories above 50 WLM, corresponding to living for 50 years in the concentration of 200 Bq/m^3. They also found that the excess relative risk per WLM varied significantly with the age attained, duration of exposure, and the time elapsed since exposure. They used both additive and multiplicative models to assess the joint effect of smoking and radon exposure. Their analyses showed that the joint association was most consistent with a model intermediate between an additive and multiplicative relationship. The excess relative risk per WLM for non-smokers was about three times higher than the risk estimated for smokers.

Both of the Finnish studies failed to detect the risk of lung cancer associated with indoor radon exposure. On the other hand, the results of both these studies did not contradict risk assessments from occupational radon exposure, either ICRP (International Commission on Radiological Protection 1987), or the pooled study of eleven underground miners' cohorts by Lubin and co-workers. This is the situation we have been facing in many of the epidemiological studies carried out in a home environment. Up to now, four studies of more than five hundred cases have been completed. In Table 2 a short overview of the principal results of these studies is shown.

Table 2. Principal results of four epidemiological studies of over 500 cases.

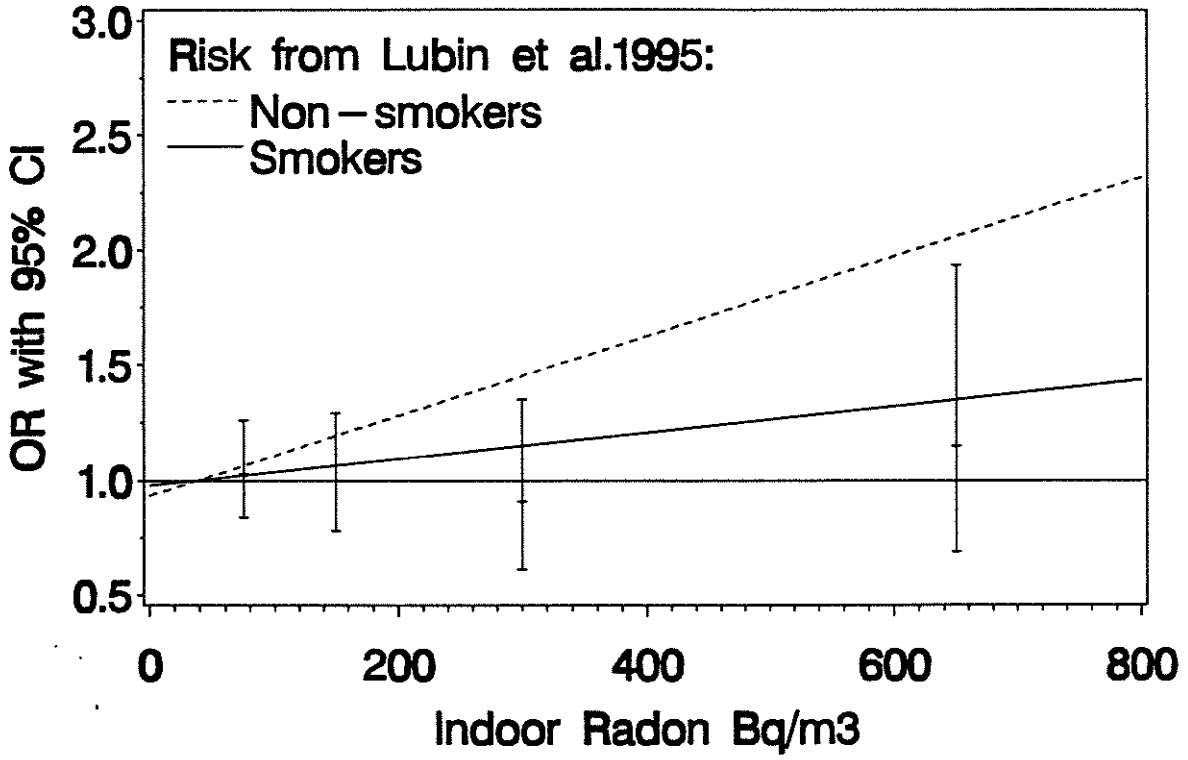
Study	Number of Case/Control	Exposure (mean concentration and measured residency)	OR (95%CI) in the highest category	Trend /100 Bq/m^3
Létourneau et al.1994, Winnipeg, Canada	250/250 female 488/488 male	118 Bq/m^3 17 years	1.0 (0.69-1.46)	NS
Alavanja et al. 1994, Missouri, USA	538/1183 female nonsmokers	67 Bq/m^3 20 years	1.2 (0.9,1.7)	NS
Pershagen et al. 1994, Sweden	1360/2847	107 Bq/m^3 24 years	1.8 (1.1,2.9)	10%
Auvinen et al., 1996 Finland	517/517	106 Bq/m^3 38 years	1.15 (0.69,1.93)	1% NS

The mean exposure level was more than 100 Bq/m^3 in all but the Alavanja study. Only the study by Pershagen et al. gave a significant trend for the whole of the data. The trend was significant for adenocarcinoma, men, and subjects over 64 years of age as well. All other studies gave non-significant trends for the whole of the data. As to subgroups, Alavanja's study gave both a significant trend and odds ratio of the highest exposure category, when cases were restricted to adenocarcinoma and results were adjusted for the amount of saturated fat intake. The odds ratios in the highest category were above unity in all studies except Létourneau's, where it was 1.0. Only the Swedish study had a 95% confidence interval that did not include unity.

Studies of home environment can be compared with the studies of miners, though this procedure is not straightforward. In figures 1 and 2 we are able to compare results of the four home studies with Lubin's study of miners. Odds ratios of different studies are shown, and the two lines are the risks of lung cancer, according to Lubin, for non-smokers and smokers. Excess relative risks for these groups are 0.0103 and 0.0034 per WLM, respectively. The exposures were calculated using the mean years of measured residency, and a conversion factor of 0.0044 WLM/Bqm^{-3}. The midpoint of the lowest category in each study was used as a reference level. The ICRP-estimate 0.0064 per WLM lies approximately in the midway between these two lines.

We can see that the nation-wide Finnish study yields point estimates less than half of those of the study miners, when, on the contrary, the study of Pershagen yields almost threefold point estimates. The study of non-smokers by Alavanja yields the estimate closest to the miners' study. The results of Letourneau are not unexpected, either.

Auvinen et al. 1996



Pershagen et al. 1994

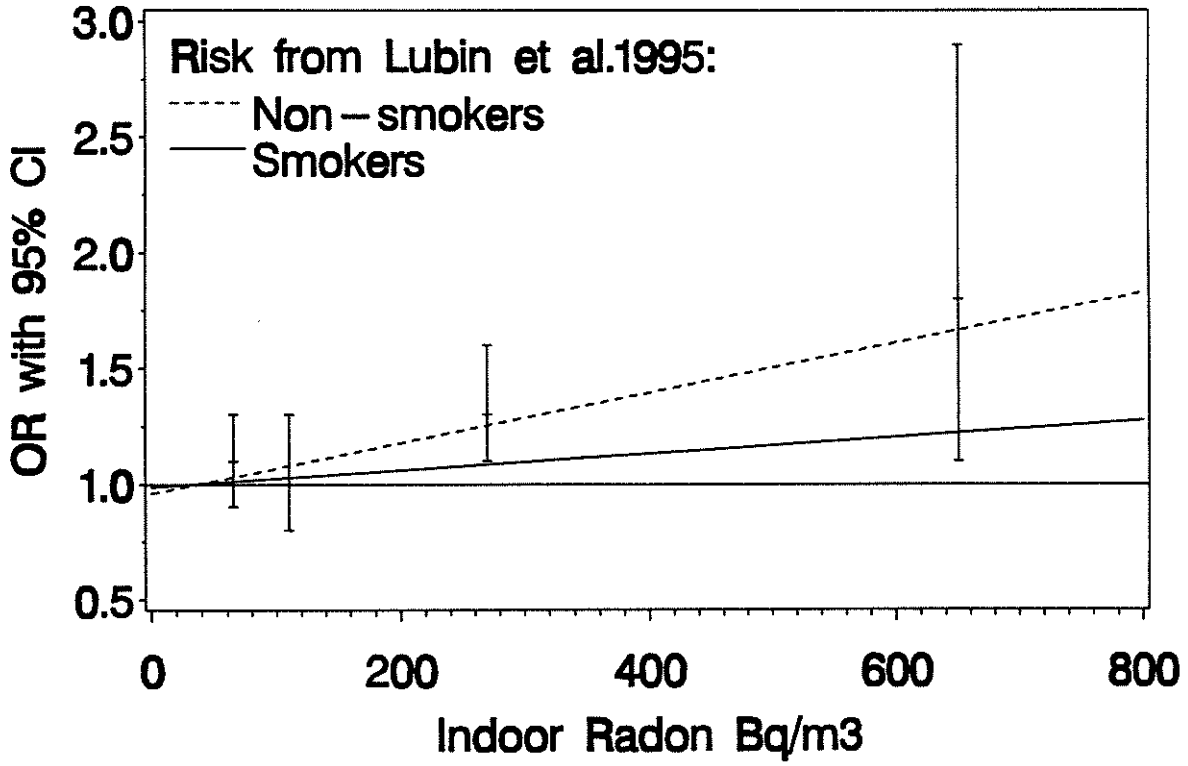
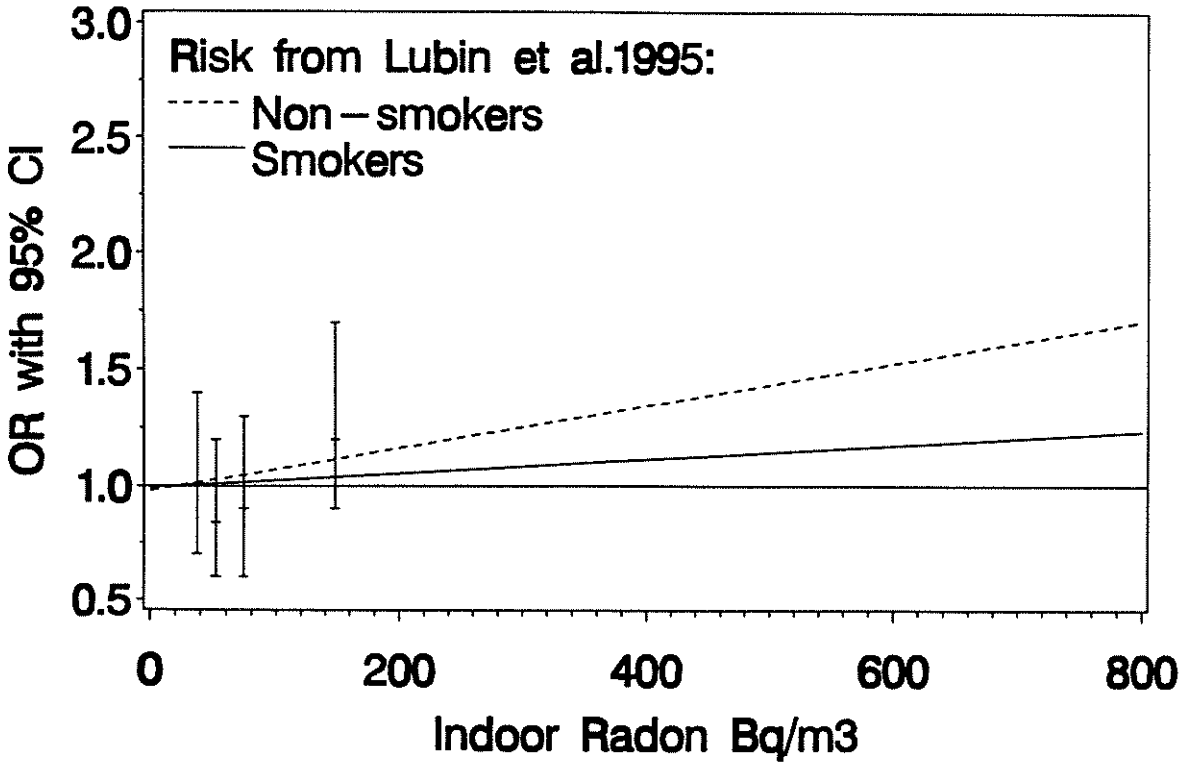


Figure 1. Odds radios (OR) of studies by Auvinen et al. 1996 and Pershagen et al 1994 . Risks according to miner studies (Lubin et al. 1995a) are indicated with a dashed line (non-smokers) and a solid line (smokers).

Alavanja et al. 1994



Letourneau et al. 1994

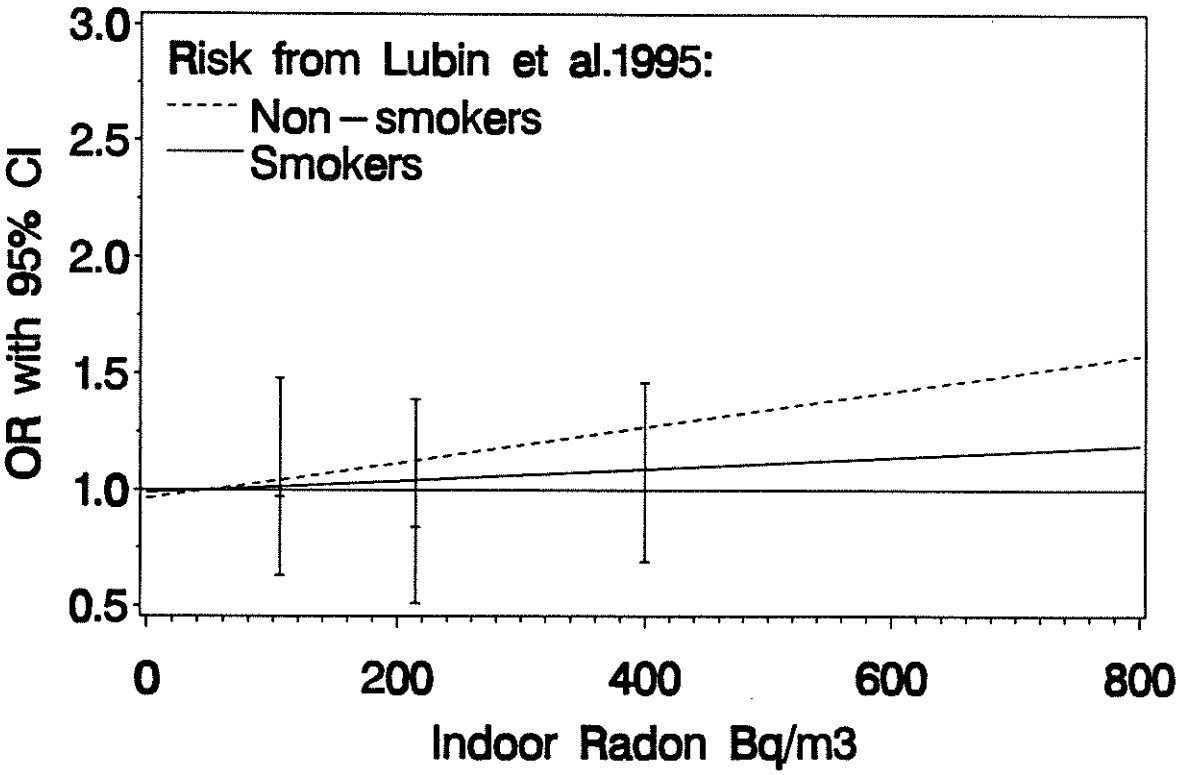


Figure 2. Odds radios (OR) of studies by Alavanja et al 1994 and Létourneau et al 1994. Risks according to miner studies (Lubin et al. 1995) are indicated with a dashed line (non-smokers) and a solid line (smokers).

Discussion

Mining exposures usually cover quite a short part of the study subjects' life spans (on average six years), whereas exposures during the rest of their lives are low when compared with occupational exposures. In a home environment the exposure levels are much lower than in mines, and exposures may continue for the whole of a person's life, but, on the contrary, exposures that are not measured are typically of the same level as the assessed exposures. This gives rise to a random error that might be substantial. This relative error is great especially when the exposure is low. The study by Auvinen and his co-workers has the advantage of exposure assessed over a relatively long period, though results from miners suggest that old exposures carry less weight.

When comparing the results of large residential studies with the results from miner studies, it is evident that the residential studies of 500-1,000 cases and controls and mean exposure levels of 100 Bq/m^3 are not able to give unambiguous risk estimates. Lubin and his co-workers had the same conclusion (Lubin et al 1995b). This is partly because of the errors in exposure estimates, partly because of real differences in the risks per exposure unit. New epidemiological studies in a home environment are going on in a number of countries, and after they are completed it will be time for a meta-analysis. Furthermore, new biological and biomathematical studies and model development relating to carcinogenesis are under way. They may help us to understand the mechanisms that cause the excess relative risk per WLM to vary due to many factors.

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^{137}CS IN A LARGE FRESHWATER BASIN - A DYNAMIC MODEL

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Abstract

A dynamic environmental model was made for the Asikkala water basin in Lake Päijänne. The Asikkala basin is of special interest both because of its importance to fishing and because it supplies freshwater for the whole Helsinki region. The model predicts the long-term behaviour of fallout ^{137}Cs in water, sediment and fish.

The model takes into account the seasonal stratification and mixing characteristics of a large basin by dividing it into the epilimnion and hypolimnion. For the runoff calculations, the drainage area is divided into arable and forest land. Seasonal variations in flow and transfer parameters are included.

The substantial volume of radioactivity and hydrological data available on Asikkalanselkä provides a good base for model construction. The calculated results are in good agreement with post Chernobyl measurements. Comparisons between model calculations and measured data are presented.

Introduction

Asikkalanselkä is the southernmost basin of the Päijänne catchment and is of special importance because it is the main freshwater supply for the Helsinki region. Water is lead directly from the Asikkala basin to the Helsinki waterworks through a bedrock tunnel. With a total area of 26 460 km^2, Päijänne is one of Finland's largest catchments. It lies in the area where the radioactive fallout was highest in Finland after the Chernobyl accident. The water outlet of Päijänne is the Kymijoki and the mean discharge rate is 209 m^{3}s^{-1} /1/. The Asikkala basin covers an area of 52.8 km^2 and its total volume is 805.2×10^6 m^3 /2/. Asikkalanselkä is also an important fishing ground.

Since Asikkala is used as a fresh water basin it is important to be able to accurately estimate the contents of radionuclides in the lake water in the early stages of a fallout situation. On the other hand, consumption of freshwater fish is a major source of radiocaesium in the human diet especially in later stages. Our aim has therefore been to establish a model able to provide accurate short and long term predictions. To this end we included the seasonal variation in the model parameters and divided the basin into two vertical layers, the epilimnion and the hypolimnion. This division is of particular importance for the short term predictions of radionuclides in the lake water.

Measurements of environmental radioactivity have been conducted at Asikkala since the beginning of nuclear weapons tests fallout, and extensive studies of radiocaesium in water, sediment and fish have been made since the Chernobyl accident. We also have good knowledge of hydrological data and the biological activity in the Päijänne area.

Model structure and parameters

Models based on linear differential equations have been widely used in dynamic calculations of radionuclide transport in ecosystems. The present model is implemented using the Stella II™ modelling environment.

The model is built so that the time of year, the amount of fallout and the distribution coefficient of the radionuclide between the solid and liquid phases (in the form of the K_d constant) are easily changeable by the end user. For the test runs we used runoff calculated from long-term averages of meteorological data. It would not however, be difficult to replace this runoff with actual measured or predicted yearly rainfall and runoff data.

The model divides the ecosystem into compartments and calculates the transfer between them according to the equation:

$$\frac{\partial A_i}{\partial t} = \sum_i (In_i - Out_i) \qquad (1)$$

where A is the amount of a resource in one compartment, In_i is an inflow of the resource to the compartment and Out_i correspondingly an outflow. Any flow can take place between two compartments within the system or it can be into or out of the system.

The Asikkala model can be seen as divided into three parts: catchment area, water basin and fish. The catchment and water basin parts of the model are based on the transfer of material (water and solid) and on the distribution of radiocaesium between the solid and liquid phases as described by the K_d constant:

$$K_d = \frac{C_s}{C_w} \left[\frac{Bq\,kg^{-1}}{Bq\,l^{-1}} \right] \qquad (2)$$

where C_s is the concentration in the solid phase and C_w is that in the liquid phase.

In the modelling of the transfer of radiocaesium in fish the metabolic effects are such that the use of mass transfer is impractical. As the total uptake of radiocaesium in fish is small compared with the amount in the lake, the radioactivity ingested by fish is omitted from the model of the basin. Radiocaesium accumulates in fish when they feed on contaminated plankton, benthic animals or smaller fish, and it leaves the fish according to the biological half-lives of radiocaesium in different types of fish.

The nearby catchment of the Asikkala basin covers an area of 155 km^2 /3/ of which 13% is arable land /1/. Wetland is not included in this model because of the very low occurance (<1%) of mires in the area. Forest land is vertically divided into three sublayers: top soil (2 cm), which is the active layer in respect to runoff; subsoil (2-7 cm), where there is some change of activity with the top soil layer; and bottom soil, which acts as an activity sink. For the arable land the 5 cm intermediate layer is replaced by a 28 cm plough layer. In our model the impact of the differences between forest and agricultural soils is almost negligible.

The compartmental structure of the catchment is depicted in Fig. 1. The transfer of radiocaesium is driven by rainfall, and is due either to radiocaesium in the liquid phase, which can move either downwards in the soil or into the lake as surface runoff, or to radiocaesium in the solid phase in the surface runoff (e.g. soil erosion, litter). The mean annual rainfall in the area is 600 mm /4/.

The amount of solid matter washed into the lake has been calculated from the amount of net sedimentation in the lake and the amount of dry matter transferred from the lake through the Kymijoki outlet. This yields a total of 124 g m^{-2} y^{-1}.

Because Asikkalanselkä is a deep basin (maximum depth 53 m), it would not be realistic to model it as a single compartment. Our solution was to divide it into two compartments, representing the epilimnion (0~15 m) and the hypolimnion (~15-53 m). The thermocline, which in Asikkalanselkä normally resides at 13-25m depth /2/, was left out for simplicity. With this division, 75% of the water volume resides in the epilimnion.

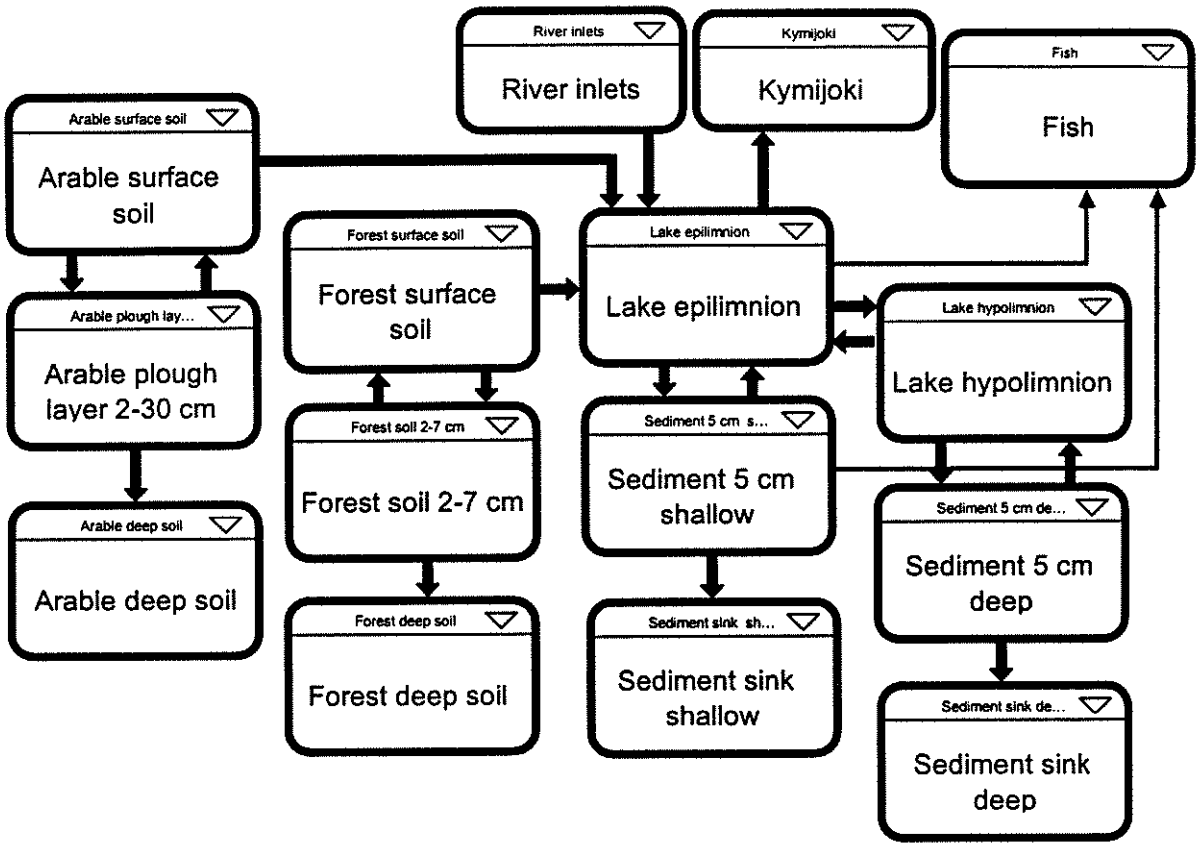

Fig. 1. *Compartmental structure for the hydrological model of the lake. Thick arrows represent the transfer of radiocaesium with soil and water, thin arrows represent the transfer of radiocaesium from the lake basin to the fish chain.*

Owing to the partial through-flow of water from northern Lake Päijänne, the water residence time in the Asikkala basin is difficult to estimate. Here however, the residence time is estimated to be of the same magnitude as the theoretical residence time in Päijänne, i.e. 2.7 a. Asikkalanselkä has a relatively small runoff area, and most of the water running through the basin originates from northern Päijänne or Lake Vesijärvi. Without these large flows, the residence time would be about ten times longer. Thus the impact of the modelled runoff area is comparatively small.

The seasonal mixing of the water layers is modelled as a complete turnover twice a year: in spring just after the break-up of the ice cover (week 18) and in autumn (week 44). At other times there is little mixing of water between the layers, and the seston sinks down to the hypolimnion at the sedimentation rate.

The lake bottom is divided laterally into a deeper bottom (>15 m) and a shallow bottom. The sediment is further divided vertically into two layers, 0-5 cm and the sediment sink. This division, although not sufficient for accurate estimations of sediment mixing, is good enough for reasonable estimates of the water-sediment interaction (sedimentation and resuspension) and for a base for the estimation of radiocaesium concentrations in fish food.

The transfer of radiocaesium to the lake bottom is mainly due to sedimentation. Thus an important factor affecting the flux is the K_d constant. Estimated from measurements made by Kansanen et al. in the years 1986-1988, the K_d constant was found to fluctuate between 45 000 and a little over 90 000, with a mean of 69 300 /2/. This mean value was used in the case study in this paper, but the K_d value is not a fixed parameter of the model.

The sedimentation rate in Asikkalanselkä was measured by Kansanen et al. using sedimentation traps and bottom sediment sampling. They obtained a mean rate of 1.5 g m^{-2} d^{-1}, with only slight spatial variation. The rate was however significantly lower in winter, when the lake was covered with ice /2/. This mean value was used here, but the sedimentation rate was allowed to vary with the amount of seston in the water column, resulting in the same type of variation as measured.

Resuspension was found by Kansanen et al. to be a significant factor for the behaviour of radiocaesium in Asikkalanselkä. On the basis of their work we used the value 1 g m^{-2} d^{-1} for the mean resuspension.

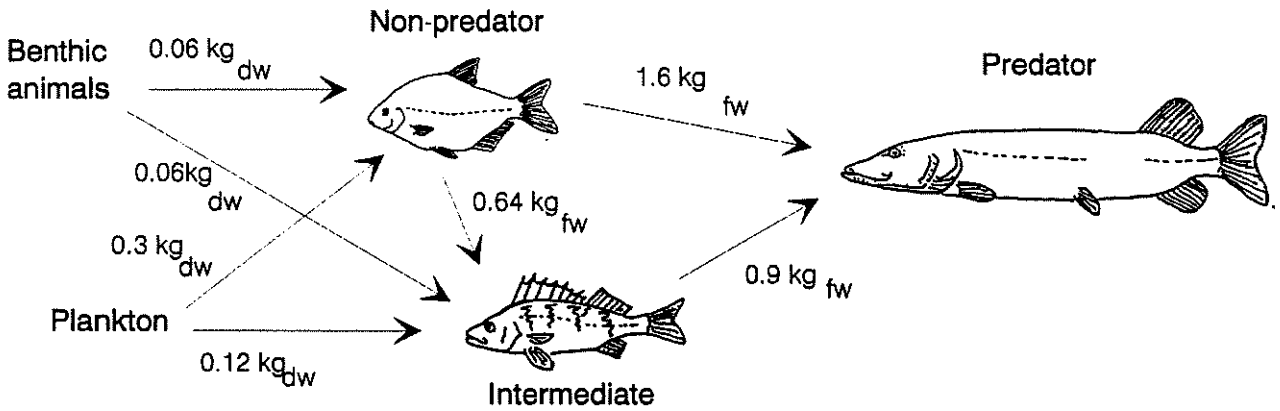

Fig. 2. *Structure of the fish model. The indicated feed intakes are kilograms per kg_{fish} per year. The intakes were estimated using local data /5/, Finnish literature on fish /6/ and previous modelling work /7,8/.*

Different fish species are lumped together in three categories: predators, non predators, and intermediate. Non-predators feed on a mix dominated by plankton with some benthic animals. Intermediate fish feed on a mix of non-predators and plankton, with a small amount of benthic animals. Predators feed on a mix of non-predators and intermediate fish. The feed amounts are indicated in Fig. 2.

The radiocaesium contents in plankton are presumed to follow the variation in radiocaesium contents in water, and those in the benthic animals the contents in sediment. Thus benthic animals becomes a very important source of radiocaesium for the fish chain some years after the fallout. As we lack proper local data on the transfer factors for radiocaesium from the water and sediment to the feed of fish in Asikkalanselkä, these have been replaced by the concentration ratios, 5000 Bq kg_{dw}^{-1} / Bq l_{water}^{-1} for plankton and 8 Bq kg_{dw}^{-1} / Bq $kg_{sediment}^{-1}$ for benthic animals, derived from measured contents in fish.

The seasonal dependency of fish metabolism is modelled by simply letting the fish feed twice as much in summer (3 months) and one half as much in winter (3 months), as in spring and autumn. The average biological half-life of the fish types chosen is 200 days /8,9/, varying in the same way as the feed intake.

Model calculations

Chernobyl data can be used to test the model for ^{137}Cs in water. For the fish model post-Chernobyl measurements were utilised in estimating the activity level of fish food. This gives an autocorrelation between the model and the test data regarding the overall level of fish activity.

Total Chernobyl deposition in the area ranged from 60 900 Bq/m^2 to 66 700 Bq/m^2 according to the nation-wide survey of external radiation /10/. Lichen samples taken in 1986 gave an estimated deposition of 64 300 Bq/m^2 /2/. In our calculations the value 65 000 Bq/m^2 was used. The local variation in deposition is considerable, with soil samples taken in the area showing values ranging from 40 kBq m^{-2} to 100 kBq m^{-2}.

For model testing we used data of STUK and the Radiochemical Laboratory at the University of Helsinki reported in /2,11,12,13,14,15,16/

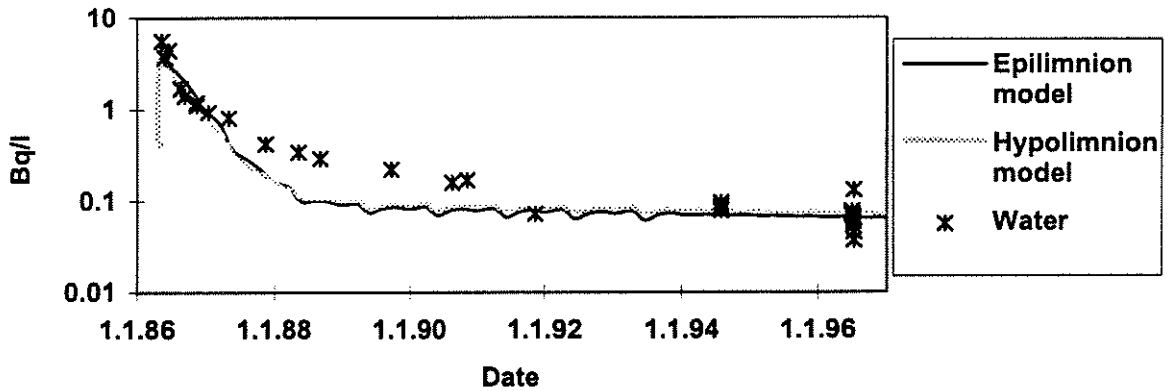

Fig. 3. *Calculated ^{137}Cs in Asikkalanselkä, compared with measured data. The measured values are from different areas and depths in the water basin.*

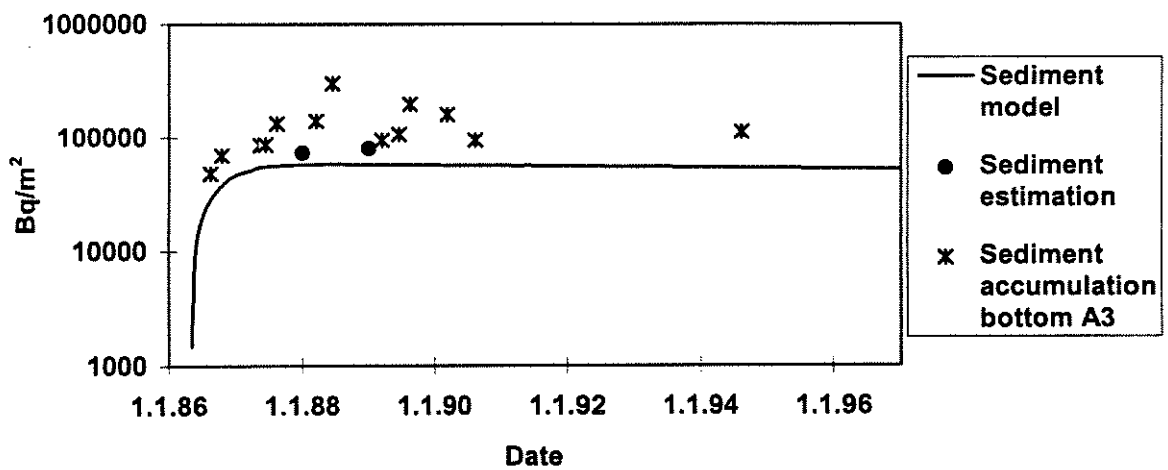

Fig. 4. *Calculated ^{137}Cs in Asikkala sediment. Estimated average sediment activity for the whole basin (filled dots) /2/, and measured activity at accumulation bottom A3. Owing to focusing effects, the accumulation bottom is expected to show activity values higher than the sediment average.*

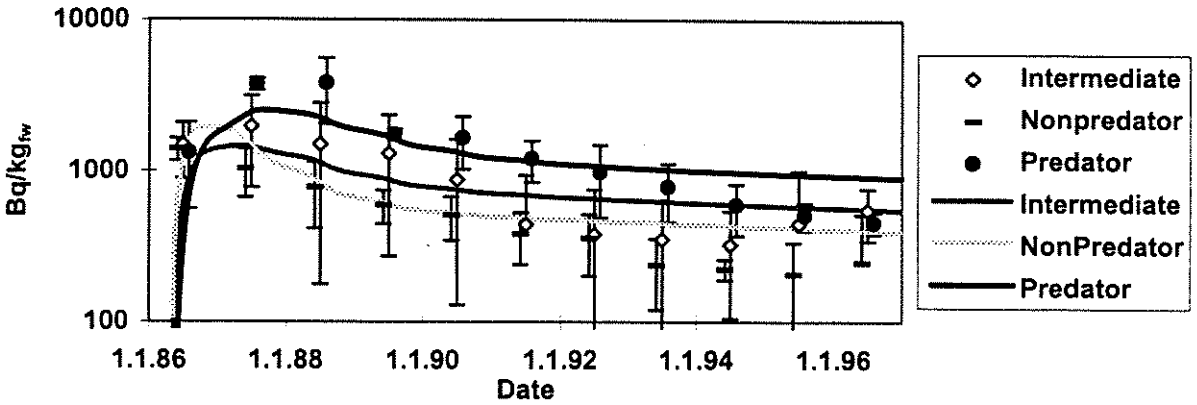

Fig. 5. *Calculated values for ^{137}Cs in fish compared with yearly means of measured data (STUK). The error bars represent one standard deviation.*

The calculated values for water (Fig. 3) are in good agreement with the measured data, even if the model tends to predict a somewhat faster decline in water activity after the first year, reaching an equilibrium state faster than that shown by the measurements. Both the calculated values for sediment and the measured data (Fig. 4) show a fast rise in activity after the accident, reaching values equal to or even exceeding the total fallout into the basin.

The calculations for different fish types are consistent with the measured data (Fig. 5). The predictions for predators in 1987-1988 seem to be slightly low and the decline in activity for predators and nonpredators is somewhat too slow. The discrepancy is small though.

Discussion

Lake Asikkala with its deep accumulation bottoms and large shallow eastern parts is a very variable environment, and the modeller has to make rather harsh averaging assumptions. The behaviour of the calculated water activity may well be a result of such oversimplification. One of the future goals of this project will be to test whether elaboration of the sediment part of the model could improve the outcome. Another obvious factor causing uncertainty in the Asikkala model is the residence time in the water basin.

In recent years a great number of ecological transfer models, the present one among them, have been constructed and/or validated using Chernobyl data. When estimating the reliability of these models for the prediction of possible future fallout situations, one should always bear in mind that a certain bias within these models is to be expected.

When building the Asikkala model we were able to use data from site specific radioactivity measurements conducted over a long period. The sustained collection of data has enabled us to construct a model that gives realistic long term estimates of ^{137}Cs activity in water, sediment and fish in a fallout situation, where the activity is reasonably evenly distributed throughout the Päijänne area.

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Radiocaesium in grazing sheep - A statistical analysis of variability, survey methodology and long term behaviour

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Abstract- Data from two grazing units in the county of Hedmark, Norway, where live monitoring of sheep has taken place each autumn since 1987, formed the basis for a study of the importance of fungi for radiocaesium levels in sheep. The results show that the ^{137}Cs concentration level due to ingested vegetation has decreased by an effective half life of 7.8 ± 2.6 years, and that the contribution from ingested fungi to the radiocaesium concentrations in the sheep was up to 75-80 % in 1988 and 1991, during which years fungi were most abundant.

Observations from another grazing unit, where a whole herd was monitored in 1987, formed the basis for a study of sampling strategy and variability of radiocaesium concentrations within a herd. Analysis of variance showed that lambs had approximately 20-40 % higher levels than ewes. The estimated average radiocaesium concentration in a grazing unit is based on monitoring a sample of 8-10 animals. A study of the variability indicated that more than twice as many animals should be monitored if an estimate of the mean radiocaesium concentration should be obtained with uncertainty less than 10 %.

Introduction

Mountain areas in Norway received considerable amounts of Chernobyl fallout in 1986 [1] and consequently the accident gave specially large consequences for agriculture involving animals grazing on unimproved mountain pastures. The LORAKON system has since 1987 been responsible for annual live monitoring of sheep. Median radiocaesium concentrations in grazing units (sheep from several farmers grazing in the same area) has been estimated from monitoring a selection of 8-10 animals, and this estimated value determined the requirements for countermeasures in the whole grazing unit.

In 1988 an unexpected high radiocaesium concentration level in sheep was observed, and this coincided with the early appearance of large amounts of fungal fruit bodies[2]. The observations in 1988 showed that intake of fruit bodies affects the radiocaesium concentration in grazing animals, and that the Chernobyl fallout could have more long term consequences than was initially assumed. However, the absolute contribution by radiocaesium in ingested fungi to the radiocaesium concentrations in animals is not known.

In order to study the different factors (with emphasis on fruit bodies) influencing radiocaesium concentrations in animals grazing on unimproved pasture, as well as the variability within animals in a grazing unit, we used live monitoring records from some of the most contaminated areas in Norway.

Materials and methods

Long term behaviour

The observations from the grazing units of Fåset (Tynset) and Fonnåsfjellet (Rendalen) in Hedmark were used to study the long term behaviour of radiocaesium in sheep, and also the importance of ingestion of radiocaesium in fungi to the radiocaesium concentrations in animals. In both areas animals had been monitored every year since 1987, and no countermeasures had been implemented at any time. As a study of fungi is carried out in the nearby area of Elgvasshø, information about fungal appearance for the full period was available. The two data sets from Fåset and Fonnåsfjellet consist of altogether 209 measurements from 9 years, 120

lambs and 89 ewes. From these data we developed a regression model for the radiocaesium concentrations in sheep during the grazing season.

Model development

Winter feeds like hay and silage contain relatively low radiocaesium concentrations. Combined with a biological half life in sheep of about 20 days [3] the winter feeding makes the animals low in radiocaesium content at the beginning of the grazing season. Thus, when released on pasture the activity concentrations in meat will theoretically rise from about 0 Bq/kg.

The first 4-6 weeks on pasture the animals will consume contaminated vegetation and the radiocaesium concentrations will increase. Fungi, with high radiocaesium concentrations, appear in late summer and will cause higher radiocaesium levels as animals consume them. If, for simplicity, the radiocaesium accumulating fruit bodies are assumed to arrive at approximately the same time every year (and the same species arrive in the same amounts), the important factor for radiocaesium concentrations in animals will be how long they stay on pasture, or more specifically, time on pasture with fungi available as feed for the animals. The radiocaesium levels in animals during a grazing season can then be approximated by :

$$RC\,(t_c, t_p, F) = C \cdot e^{-\lambda_{eff} \cdot t_c} + B \cdot F \cdot e^{-\lambda_{fungi} \cdot t_c} \cdot (1 - e^{-\lambda_{bio} \cdot t_p}) \qquad (1)$$

where t_c is time after the Chernobyl accident in years while t_p is time on pasture with fungi available as feed. λ_{eff} is the effective elimination rate for radiocaesium in vegetation, λ_{fungi} describes the ecological decrease in radiocaesium concentration in fungi, and λ_{bio} is the biological elimination rate of radiocaesium in sheep, 0.03 d^{-1}. C is a constant and the parameter B gives the equilibrium radiocaesium level and will for instance depend on radiocaesium concentration in vegetation and intake of and radiocaesium concentration in fungi. The factor F estimates the amount of fungi every year. The first term estimates the activity concentration due to intake of vegetation and the second term the additional radiocaesium level due to consumption of fungi.

The available information about amount of fungi is recorded simply as relative units «bad», «medium» and «good» year for fungi (and refer to the amount of fruit bodies only). The relationship between these units can be estimated by dividing B·F in three factors; $D \cdot F_1$, $E \cdot F_2$ and $G \cdot F_3$ (F_1, F_2 and F_3 are indicator variables where $F_1 = 1$ for a «good year», 0 otherwise, $F_2 = 1$ for a «medium year», 0 otherwise, and $F_3 = 1$ for a «bad year», 0 otherwise). D, E and G are equivalents to B in Eq. 1. Hence, the radiocaesium concentration can be described as a function of t_c, t_p and the indicator factors F_1, F_2, and F_3 :

$$RC\,(t_c, t_p, F_1, F_2, F_3) = C \cdot e^{-\lambda_{eff} \cdot t_c} + (D \cdot F_1 + E \cdot F_2 + G \cdot F_3) \cdot e^{-\lambda_{fungi} \cdot t_c} \cdot (1 - e^{-\lambda_{bio} \cdot t_p}) \qquad (2)$$

The ratio D : E : G can then be used as an estimate of the relative importance of the different amounts of fungi.

Variability

In the grazing unit of Baklia the radiocaesium concentrations in 212 animals were measured in September 1987. Of these animals 81 were ewes (born in 1986 and earlier) and 131 lambs. This dataset was used to study the distributions of radiocaesium concentrations in sheep in a grazing unit and was also implemented in a study of sampling statistics.

For the grazing unit the estimated mean $\overline{X}$ and the estimated variance s^2 can be calculated, and it may therefore be interesting to look at the sample sizes needed to attain a certain degree of precision in the estimations of the means of the populations. A general sampling strategy may be evolved by implementing the coefficient of variation CV (σ/μ). Studies have indicated that despite the movement in the average activity in a flock, and the wide ranges in the activity of individuals, the distribution of the coefficient of variation remained essentially constant, which means that the error is proportional to the value of the results [4]. An expression for the required sample size n is:

$$n \geq \frac{t^2_{\alpha/2,n-1} \cdot CV^2 \cdot N^2}{t^2_{\alpha/2,n-1} \cdot CV^2 + E^2 \cdot (N-1)} \qquad (3)$$

According to this expression the required sample size will depend on the confidence level α, the accepted error E in the estimate of mean, and the size of the total population, N.

Results and discussion

Long term behaviour of radiocaesium in sheep

The model in Eq. 1 allowed for decreasing radiocaesium concentrations in fungal fruit bodies through the term $e^{-\lambda_{fungi} \cdot t_c}$, although different studies have found this to be insignificant after the Chernobyl accident [5], [6]. The regression analysis showed that λ_{fungi} also is non-significant in the present data set. All further analysis is therefore performed without this term, thus reducing the model complexity to:

$$RC\ (t_c, t_p, F) = C \cdot e^{-\lambda_{eff} \cdot t_c} + B \cdot F \cdot (1 - e^{-\lambda_{bio} \cdot t_p}) \qquad (4)$$

August is generally thought to be the main season for fungal fruit bodies, and the largest quantities of fruit bodies seemed to be found at the end of August and beginning of September in the nearby study area. This indicates that it is difficult to estimate the exact time for fruit body appearance as input to the model. Mid-august (August 15) was chosen as the average fruiting date in this work. The ratio between «good, medium and bad» fungi abundance, D : E : G, was estimated to 1 : 4 : 9.

Table 1: Parameters in the regression model for long term behaviour of ^{137}Cs in sheep (with the adopted fungi ratios 1:4:9 estimating low, medium and high abundance).

fruiting date	parameter	estimated value [y^{-1}]	95% conf.int.	R^2	estimated $T_{eff} \pm$ std.error
	λ_{eff}	0.09	(0.03, 0.15)		7.8 ± 2.6
15.08	C	370	(260, 480)	0.60	years
	B	190	(160, 210)		

The resulting model for describing the ^{137}Cs concentration in sheep is :

$$RC\ (t_c, t_p, F) = 370 \cdot e^{-0.09 \cdot t_c} + 190 \cdot F\ (1 - e^{-0.03 \cdot t_p}) \qquad (5)$$

which gives an effective half life for ^{137}Cs in vegetation of 7.8 ± 2.6 years.

There are several factors not taken into account in model 4. The fact that the data are derived from two separate areas increases the uncertainty, and the difference in biological half life for

radiocaesium between lambs and ewes is not considered in the model. Furthermore, it is assumed that the fungal fruit bodies arrive at the same time every year. The radiocaesium concentration in vegetation is assumed to follow an exponential decrease over years, while for sheep on pasture it has been assumed that the radiocaesium concentration increases and then stabilize at an equilibrium level determined by the radiocaesium concentration of the vegetation.

Despite many assumptions and sources of uncertainty, the estimated values for radiocaesium concentration in sheep obtained by model 4 are in accordance with the observed radiocaesium concentrations (Fig. 3). The difference between the ^{137}Cs level according to vegetation and estimated ^{137}Cs concentration in sheep is an indication of the importance of fungi for the ^{137}Cs concentration in the monitored sheep.

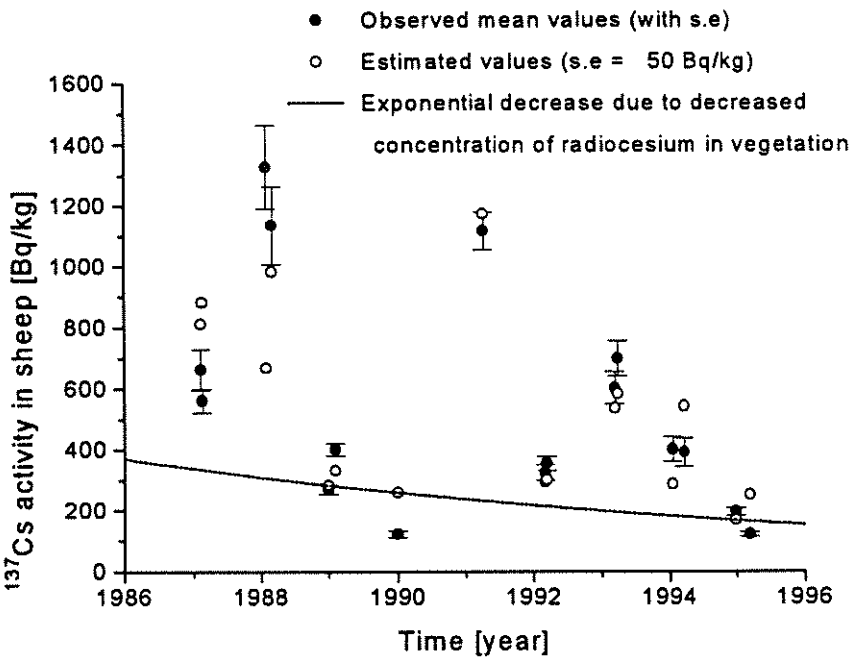

Figure 1: Estimated values for ^{137}Cs concentration in sheep are compared with the mean of the observations for each year ± std. error of mean. The uncertainty associated with the estimated values is not shown, but is about ± 50 Bq/kg. The dot-and-dash line illustrates the underlying exponential decrease in ^{137}Cs concentration in sheep due to decrease in contamination level in vegetation. Time for appearance of fruit bodies is set to 15.08, and the relative importance between the different levels of fungi abundance is 1:4:9.

The estimates of effective half life for ^{137}Cs are shorter than the 20 years suggested earlier [7]. However, the rate of decrease in ^{137}Cs levels will differ between environments and areas of the country, and a summary of half lives in Norway gave values in the range 3-20 years [8]. The model with its estimated parameters is only valid for the areas studied and the observed period, and can not without reservations be transferred to other areas or later years.

Variability within herds

Figure 1 shows the distribution of the data for the whole herd and the two subgroups, ewes and lambs. There was a relatively clear trend of two subgroups in the data material, where lambs seemed to have generally higher radiocaesium activity than ewes. From the figure it appears that dividing the data material into two subgroups may enhance the approximation to

normal distribution. The standard deviation decreased for both groups compared to the standard deviation for the whole herd.

An analysis of variance of the observations gave the results in Table 2. The analysis showed a significant difference between ewes and lambs ($p < 0.001$).

Table 2: Data from Baklia (1987)

	cases	mean ($\bar{x}$)	std.error of mean	95% conf.int. for mean	median
ewes	81	820	29	(760 , 880)	780
lambs	131	1170	29	(1110 , 1220)	1160
whole herd	212	1030	24	(990 , 1080)	1000

The observations were checked for normal and lognormal distribution. The data did not seem to fit a lognormal distribution any better than a normal distribution. Analysis has therefore been undertaken with untransformed data even though the deviation from normal distribution will increase the uncertainty in the results.

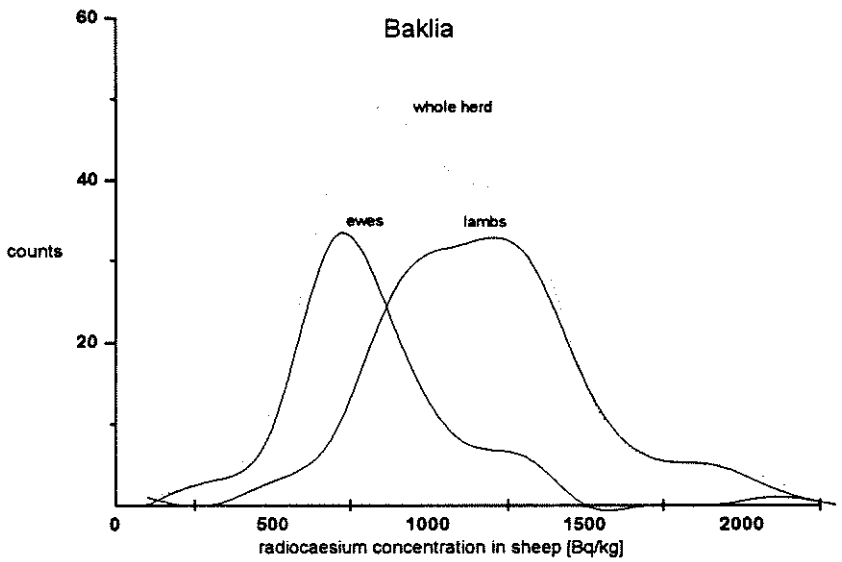

Figure 1: Frequency distribution for Radiocaesium concentrations in 212 animals monitored in Baklia, Valdres (1987). The figure shows the distribution of ewes and lambs and the total herd.

From the Baklia data set a general variation coefficient valid for other sheep herds of 0.3 (30 %) was estimated.

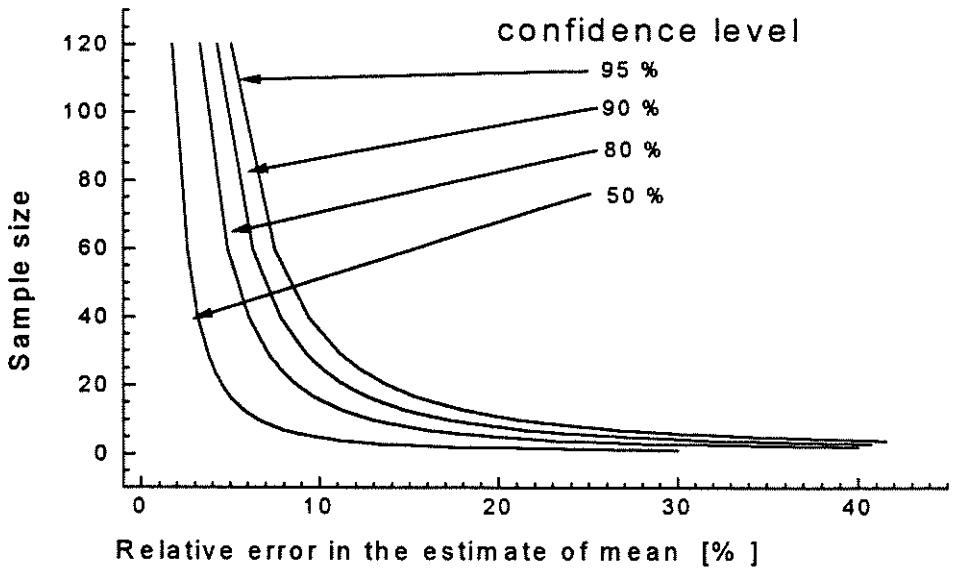

Figure 2: Required sample size n as a function of relative error in the estimate of mean (in % of mean) at different confidence level. The population size is 1000 animals.

In Figure 2 the population size is set to 1000 animals, and the sample size is shown as a function of error in estimation of mean (for different confidence levels). The figure indicates that if we want a confidence of 90 % for keeping the error in the estimate of mean under 10 % of mean the sample size should be more than 25 animals.

This analysis shows that the present live monitoring procedure with selection of 8-10 animals, will at the 90 % confidence level estimate a mean value within approximately 20% of the true mean (assuming that the median is close to the mean, which is the case for normal distributed data).

Conclusions

The results support that fungi is an important source of radiocaesium. The contribution from ingested fruit bodies is estimated to 75-80 % of the total radiocaesium concentration in sheep in 1988 and 1991, which both were years with large abundance of fruit bodies in the pasture.

The estimated effective half life for ^{137}Cs in vegetation in the grazing units of Fåset and Fonnåsfjellet using the general model was 7.8 ± 2.6 years.

The observations from Vuludalen and Baklia in 1987 indicated that the radiocaesium concentrations within the flocks were close to normal distributed. The results showed a significant higher radiocaesium concentration in lambs than in ewes, even at time of slaughter when lambs are assumed to have the same size as adult animals. The variability within lambs and ewes was lower than the variability within the whole herd.

One reason for monitoring live sheep is to check whether the mean radiocaesium concentration in animals from a grazing unit is below the accepted limit of 600 Bq/kg, and thus may be slaughtered without any form of countermeasures. It might therefore seem useful to monitor samples of only lambs, both because of the higher radiocaesium compared to ewes, and because the lower variability within the group decreases the uncertainty in the estimated mean radiocaesium concentration.

The uncertainty in estimated mean radiocaesium concentration in a grazing unit was found to be about 20 % with the present sampling method of monitoring 8-10 animals. If a probability of 90 % is desired for estimating the mean radiocaesium concentration with an error less than 10 % (uncertainty in the monitoring method not taken into account), the required sample size is ca. 25 animals.

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Transfer and long term behaviour of ^{137}Cs in sheep at Tjøtta, Norway.

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Abstract - Norwegian agricultural production was heavily affected by the Chernobyl accident. Large effort has been put into countermeasures and restrictions to reduce the radiocaesium levels in animals and animal products originating from areas with high deposition. Research at Tjøtta, Nordland, shows that the effective ecological half-life for transfer to sheep is long; 9 years for lambs and 19 years for ewes. Only in 1994 and 1995 a significant decrease in activity levels in sheep and transfer to vegetation have been observed.

Introduction

It is estimated that 3-5% of the radiocaesium released from the Chernobyl reactor have been deposited in Norway [1]. The most contaminated areas are grazing areas for sheep, reindeer and cattle which extensively are utilising these natural pastures. Special feeding of the animals, use of caesiumbinders and changing of slaughter time has been used in large scale, and are still used today, for lowering the radiocaesium levels in reindeer and sheep [2]. During the first decade the total cost for implementing countermeasures, research, monitoring and interdiction of sheep meet was 370 million NOK (57 million US$). The corresponding costs for the reindeer-breeding was 110 million NOK (17 million US$).

Many factors such as soil characteristics, vegetation type and grazing habits are shown to influence the uptake of radiocaesium from soil to grazing ruminants. The amount of mushrooms in pasture play an important role in determining activity levels of radiocaesium in animal products [3]. Some of the Norwegian mushrooms accumulate radiocaesium very efficiently, and are shown to have a long effective ecological half-life [4].

Knowledge concerning the long term behaviour of radiocaesium in the pasture-sheep ecosystem is essential to control the activity levels and to choose the most cost effective countermeasures. This study is conducted at Tjøtta Rural Development Centre located on the coastline in Nordland county (66.6°N). It started in 1988 and has partly been funded by NKS and by the Norwegian Research Council.

Materials and methods

The study site is a fenced area of 0.4 ha of a natural pasture with highly organic peaty soil and a vegetation dominated by *Poa spp*, *Festuca spp*, *Trifolium spp* and *Ranunculus spp*. Since 1988 5 to 10 ewes and 10 to 20 lambs have grazed this area from the beginning of June till the beginning of September. The animals were live monitored every week by use of a Canberra Serie 10 multichannel analyser equipped with a 3*3 inch NaI(Tl) detector [5]. A

time period of 100 seconds were used for measurements. Total radiocaesium (^{137}Cs + ^{134}Cs) were measured in Bq/kg. The Chernobyl radiocaesium is dominating the old fallout, thus data presented as ^{137}Cs were obtained by subtracting ^{134}Cs in accordance with the reported Chernobyl ratio ^{137}Cs : ^{134}Cs = 2.0 by 1. July 1986 [1].

Exclusion cages were placed on the pasture each year before the animals started grazing. Soil and vegetation samples were taken in the beginning of September each year. In 1992 vegetation samples from the cages were taken three times in July, August and September. The cages were turned 30-45° each year to avoid to have the hole from previous years soil sampling in the cages. In 1995 six additional cages were put out to estimate the biomass-production and for collection of mushrooms. Two soil samples were taken in six of the cages. A plastic core with inner diameter of 10.3 cm and a depth of 10 cm were used. One of the corers were sliced in 1 and 2.5 cm slices for determining the vertical distribution of radiocaesium down the soil profile. The soil samples were dried, homogenised and measured for content of ^{137}Cs.

Percentage cover of the major plant genus/species in the different cages was visually estimated. Mixed species samples were taken from a defined area each year. In addition single genus samples were collected from the pasture at the same time as the mixed species samples. All samples were dried at 65° C, ground and measured for content of ^{137}Cs.
The data on precipitation and temperature are taken from the weather station located at Tjøtta only 4-500 m from the experimental field.

The aggregated transfer factor (T_{ag}), the transfer factor (TF) and the concentration factor (CF) was calculated as follows: T_{ag} soil-meat = Bq/kg FW (fresh weight) meat per kBq/m^2 soil, TF soil-gras = Bq/kg DW (dry weight) grass per kBq/m^2 soil, CF grass-meat = Bq/kg FW meat per Bq/kg DW grass. The activity in lambs are based on live monitoring of lambs in the period 8-12 weeks of grazing. The effective ecological half-life of ^{137}Cs in pasture-sheep is calculated by taken the average activity levels measured by live monitoring in the period 8-12 weeks of grazing. An exponential regression curve was fitted to the data to estimate the effective ecological half-life.

Results and discussion

The mean deposition of ^{137}Cs in the paddock was about 35 kBq/m^2 and was relatively homogeneous distributed throughout the grazing area. In 1995 70 % of the ^{137}Cs remain in the top 5 cm of the soil. There was a tendency of ^{137}Cs to move downwards in the period 1992-1995 (Fig. 1)

The percentage of cover of vegetation in the exclusion cages was mainly dominated by *Trifolium spp*, *Poa spp*, *Deschampsia caespitosa* and *Ranunculus spp* estimated at the end of the grazing period.

The content of ^{137}Cs in vegetation has been relatively constant in the period 1990-1993 with levels of approximately 2000 Bq/kg DW. Up to 1993 the ^{137}Cs levels in mixed species samples from the cages varied with a factor of 10 from 400 Bq/kg DW to 4500 Bq/kg DW (table 1). In 1995 a marked decrease in activity levels in vegetation from five of the six cages was found compared to levels from 1994. This decrease was confirmed by the live monitoring data from the sheep. July and especially August 1995 were very wet and cool months. This may have influenced the transfer from soil to vegetation.

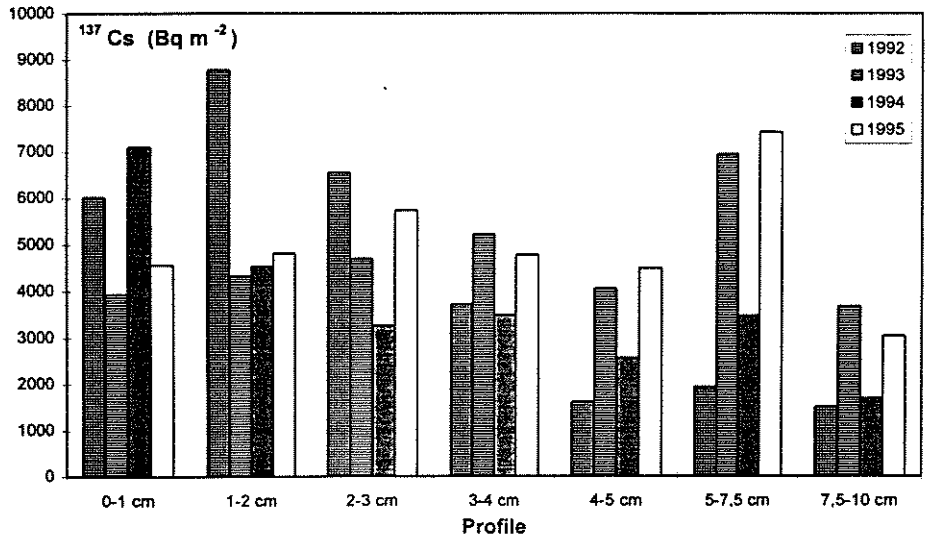

Figure 1. ^{137}Cs in soil profiles (Bq/m^2) from the years 1992-1995

Activity levels of radiocaesium in vegetation are shown in table 2. *Festuca spp* and *Rumex acetosa* accumulate more ^{137}Cs than the other species. In the time period from 1990-1994 *Trifolium spp*, *Deschampsia cespitosa* and *Festuca spp* showed a decrease in activity levels which was not observed in the other four species studied in this period. However a significant decrease in activity levels was observed in 1995 (except for *Trifolium spp* which was only found in one cage).

There are reported to be very little mushrooms at this pasture during the time period of the experiment. However, only in 1995 it was studied specifically. No mushrooms were observed in any of the cages. A few species where found outside the cages. The following species were observed: *Leccinum versipelle*, *Leccinum scabrum*, *Russula vesca* and *Hypholoma capnoides*.

The production of biomass was measured in 1995. The production decreased during the grazing period. In July the production ranged from 41-119 g/m^2, in August 23-67 g/m^2 and little production was measured in the first half of September (< 21 g/m^2).

The aggregated transfer factor soil-meat and the concentration factor grass-meat has not changed significantly during the period 1990-1994 (table 3). However in 1995 these factors decreased significantly. Also the CF grass-meat changed markedly and increased by a factor of two compared to previous years.

Figure 2 shows the average activity levels of ^{137}Cs in lambs from 1989-1996 obtained by live monitoring. A trend of decreasing levels of radiocaesium was observed. This was particularly clear for the years 1992-1995 in the time period before the equilibrium state was reached (week 1 to 7). The equilibrium state seem to have changed from approximately five weeks the first three years, to 9 or 10 weeks for the later years. Not surprisingly the ewes and lambs data corresponds (data not shown). The 1996 activity levels are approximately similar to the levels measured in 1995, indicating a marked decrease compared to the results in 1993 and 1994. However, this years results do not show a further decrease from 1995, maybe indicating a more slow decrease or threshold in the years to come.

Table 1. Activity concentration of ^{137}Cs in mixed species samples (Bq/ kg DW) from the exclusion cages.

Date:	Sept. 1990	Aug. 1991	July 1992	Aug. 1992	Sept. 1992	Sept.93	Sept 94	Sept 95
Cage	^{137}Cs	^{137}Cs	^{137}Cs	^{137}Cs	^{137}Cs	^{137}Cs	^{137}Cs	^{137}Cs
	Bq kg^{-1}	Bq kg^{-1}	Bq kg^{-1}	Bq kg^{-1}	Bq kg^{-1}	Bq kg^{-1}	Bq kg^{-1}	Bq kg^{-1}
1	2460	1460	1158	1771	1432	714	609	208
2	497	530	369	434	484	402	421	74
3		3680	3158	4512	4610	4421	554	163
4	1008	1950	1389	1173	1874	2899	2195	402
5	3650	4200	2044	1969	1289	2426	2002	788
6		2900	2148	2753	3690	1032	1456	1285
Mean ± Standard error								
Bq kg^{-1} DW	1900±720	2450±570	1710±390	2100±580	2230±650	1980±630	1210±320	490±190

Table 2. ^{137}Cs activity (Bq/kg DW) in the major plant genus as an average from six cages (* only from one cage).

PLANT	1990	1991	1992	1993	1994	1995
	Cs-137 (Bq/kg DW)					
Trifolium spp		2380	1244	1035	911	2194*
Poa spp	623	1210	133	878	950	210
Agrostis spp	2045	1720	256	2557	2215	260
Deschampsia cespitosa	1165	1700	185	159	514	180
Festuca spp	4201	2340	619	1648	636	255
Rumex acetosa	1620	6210		948	6074	
Ranunculus spp	736	1480	819	1235	1176	463
Carex spp				857	3210	558

Table 3. Aggregated transfer factor; T_{ag} (m^2 kg^{-1}), transfer factor; TF (m^2 kg^{-1}) and concentration factor; CF.

Year Coefficient	1988	1990	1991	1992	1993	1994	1995
T_{ag} soil-meat	31	37	48	40	37	37	19
TF soil-gras		61	72	77	60	46	14
CF grass-meat		0.6	0.7	0.5	0.6	0.8	1.4

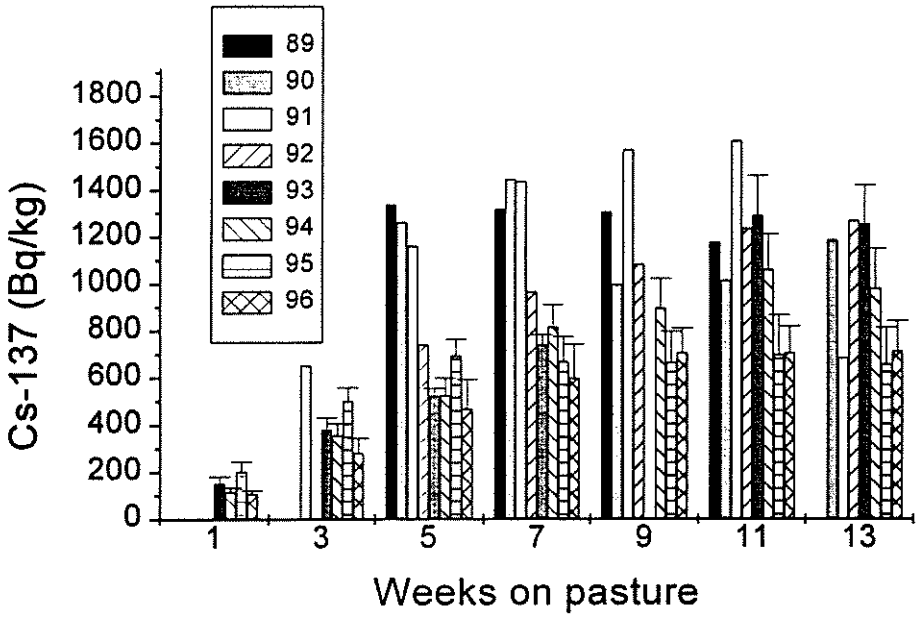

Figure 2. Activity concentration of ^{137}Cs in lambs meat (Bq/kg) at different times after the animals were released on the pasture. Standard deviation are shown for the latest years.

When estimating the effective ecological half-life of ^{137}Cs the contribution of ^{134}Cs is subtracted according to the reported ratio of ^{134}Cs and ^{137}Cs in the fallout in Norway. Figure 3

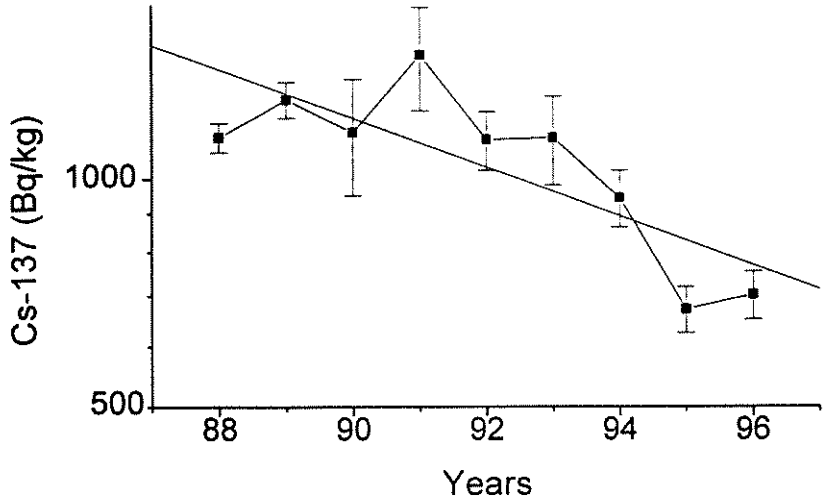

Figure 3. ^{137}Cs in lambs as average values in the period 8-12 weeks of grazing (years 1988-1996), plotted on a logarithmic Y-axes. The standard deviation of the single measurements are also shown. An exponential regression curve is fitted to the data.

shows the average levels of ^{137}Cs in lambs from 1988-1996 at a time period of 8 to 12 weeks on pasture when the equilibrium stage has been reached. An exponential regression curve was fitted to the data, indicating a relatively good fit with an R^2 of 0,62 and an effective ecological half-life of 9 years. The P-value was 0.012. Similar calculations were done for the ewes in the period 1988-1995. These gave a poor fit to the exponential regression curve with an R^2 of 0.14. The effective ecological half-life for ewes was estimated to be 19 years.

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Transfer of radiocaesium via reindeer to man - the effect of countermeasures used in Sweden after the Chernobyl accident

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Large areas of reindeer pasture in Sweden were contaminated with radioactive material after the accident at the Chernobyl nuclear power plant in Ukraine in April 1986. Reindeer from these areas are still contaminated with radiocaesium. In Sweden, the upper limit for ^{137}Cs in reindeer meat sold for human consumption is 1 500 Bq kg^{-1} (fresh weight). This limit has greatly reduced the transfer of radiocaesium via reindeer meat to man. Counter measures, as altering the time of slaughter and feeding reindeer uncontaminated feed, has also reduced the transfer.

In this paper I present an estimation of the actual intake of radiocaesium via reindeer meat, and the corresponding collective radiation dose, to humans in Sweden during the first ten years after the Chernobyl fallout. The transfer of radiocaesium and the collective dose is compared with the potential transfer, and corresponding dose to man, that would have occurred in the absence of activity limits and countermeasures.

Transfer of Cs-137 from Soil to Vegetation and to Grazing Lambs in a Mountain Area in Northern Sweden. Ecological Half-life of the Nuclide

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ABSTRACT

Within the research programme of NKS (Nordic Nuclear Safety Research) radioecological studies in the chain soil-grass-lamb were carried out in natural or seminatural areas in the Nordic countries in 1990-1993. Further studies are made in the same place during another 4-year period in order to i.a. get information on the ecological and the effective ecological half-life of Cs-137. The possible activity contribution by fungi and soil ingestion of the lambs will be studied. In Sweden the studies are located to a mountain farm in the Chernobyl fallout area in the country of Jämtland. The study lambs are grazing freely in a mountain area covering about 10 km^2. In the present paper the main study results obtained in Sweden in 1994 and 1995 are given. Comparisons are made with the published results of the previous study years.

Time-corrected (corrected to August 1990) means of Cs-137 concentrations were as follows: in soil (average of samples taken to a depth of 10 cm in the soil layer in five fixed localities distributed over the grazing area) 17.62, 14.56, 16.66, 14.07, and 13.69 kBq/m^2 in 1990-94, respectively; in herbage of the mentioned localities 1175, 1125, 960, 900, 879, and 779 Bq/kg dry weight (1990-95); in abdomen wall muscle of lamb carcasses 1090, 684, 538, 640, 509, and 684 Bq/kg wet weight (1990-95). The results indicated an ecologcal half-life of Cs-137 in herbage of 8.5 years and in lamb muscle of 8.0 years. During the period 1991-95 the concentrations in lamb mucle were, however, nearly unaltered, indicating a very long ecological half-life, >100 years, with a corresponding effective ecological half-life of 24.1 years (thus approaching the physical half-life of the nuclide). In all years all individual lamb carcasses exceeded the Swedish intervention level of Cs-137 for human consumption.

Values of transfer factors are given for the respective years: *TFsoil to plant* (Bq/kg d.w. plant per KBq/m^2 soil) 74.3, 72.3, 64.7, 57.2 and 68.6 ; *TFplant to muscle* (Bq/kg w.w. muscle per Bq/kg d.w. plant) 0.93, 0.61, 0.56, 0.71, 0.58 and 0.88; *TFsoil to muscle* (Bq/kg w.w. muscle per kBq/m^2 soil) 61.9, 47.0, 32.3, 45.5 and 37.2. The transfer factors except the *TFplant to muscle*, showed minor decreases. Further studies until 1997 will give more information on possible trends.

The effect of biotite and other minerals on the transfer of radiocesium to grass

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Abstract

Short-term pot experiments with peat soil were carried out to investigate the effect of biotite, heavy (illitic) clay soil, apatite, bentonite and zeolite on the uptake of ^{134}Cs by ryegrass. Further, a long-term effect of biotite on ^{137}Cs concentration of ryegrass was followed up in a five years field experiment on peat soil. Biotite and apatite are recommended as low soluble potassium and phosphorus fertilizers, respectively, for organic farming. Biotite is a byproduct of phosphorus mining and apatite is a raw material for phosphorus. The relative ^{134}Cs concentrations of plants are expressed as percentages of those of the control plants grown on pure peat soil.

The plant uptake of radiocesium was decreased most by zeolite. The average concentration of ^{134}Cs of the plant was in the case of zeolite 5 %, in heavy clay 30 %, in bentonite 70 %, in biotite 80 % and in apatite over 90 %. (The precentages are means of three cuts and the three applications of the minerals: 10, 20 and 40 g/litre of soil.) The effect of zeolite was not as persistent as that of biotite. When zeolite (1-8 g/l) was mixed in biotite (40 g/l), the plant concentration of ^{134}Cs of the three cuts ware fairly low decreasing with cuts as follows: 30, 13 and 5 %. When the same amounts of zeolite were used without biotite the plant cesium was increased with cuts: 34, 67 and 50 %. Further, in the same experiment, the addition of biotite (40 g/l) without zeolite decreased the plant cesium with cuts as follows: 76, 20 and 6 %. In a field experiment on peat soil, application of biotite of about 30 tons per hectare decreased the concentration of the fallout ^{137}Cs of plants by factor of about 2.5. The average plant/soil concentration ratio of ^{137}Cs for the biotite field was 0.05 and that for the control field 0.12. The effect of biotite on the cesium concentration of plants was observable after five years since application.

A foodbasket investigation to assess the average intake of ^{137}Cs by the population in Sweden in 1994.

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Abstract An investigation was done in 1994 to assess of the intake of ^{137}Cs from foodstuffs by Swedes. Foodbaskets representative of the yearly total food consumption were collected.
The population weighted average of the intake for the year 1994 was estimated to be 270 ± 50 Bq.
This gives a committed effective dose equivalent of 3.6 μSv.
The body burden was calculated to be 1.3 Bq kg^{-1}, compared to 2.0 Bq kg^{-1} obtained from whole body measurements.

Materials and methods

The fallout of ^{137}Cs in Sweden after the Chernobyl accident had a geographical distribution according to figure 1.

Representative samples of foodstuffs were collected in standardised "food baskets" to estimate the intake of ^{137}Cs by the average Swede. The baskets reflect about 92 % of the total food consumed in a year, another 5 % consists of spirits and wine which are unlikely to contribute any ^{137}Cs. In all, 97 % of the relevant food consumption is covered by the baskets. Two grocers in 10 localities were chosen, 26 foodbaskets were collected. Collection was performed between late September and early November. The sampling localities are shown in figure 1. The localities were chosen to represent the fallout situation in different areas and for the whole country.

The foodbaskets were divided into local baskets and base baskets. The local baskets comprised foodstuffs of local origin, e.g., dairy produce and game. Local baskets were collected in all localities. The base baskets included foodstuffs that are distributed over the whole country, sampling was done in Malmö, Gävle and Sundsvall. For further description of foodstuffs included see the reference.

The samples were mixed in proportions to their part of the total consumption and were measured on a HPGe detector at SSI. The selection of the items to be included in and the collection of the baskets were coordinated by the National Food Administration, they also processed them into one litre samples.

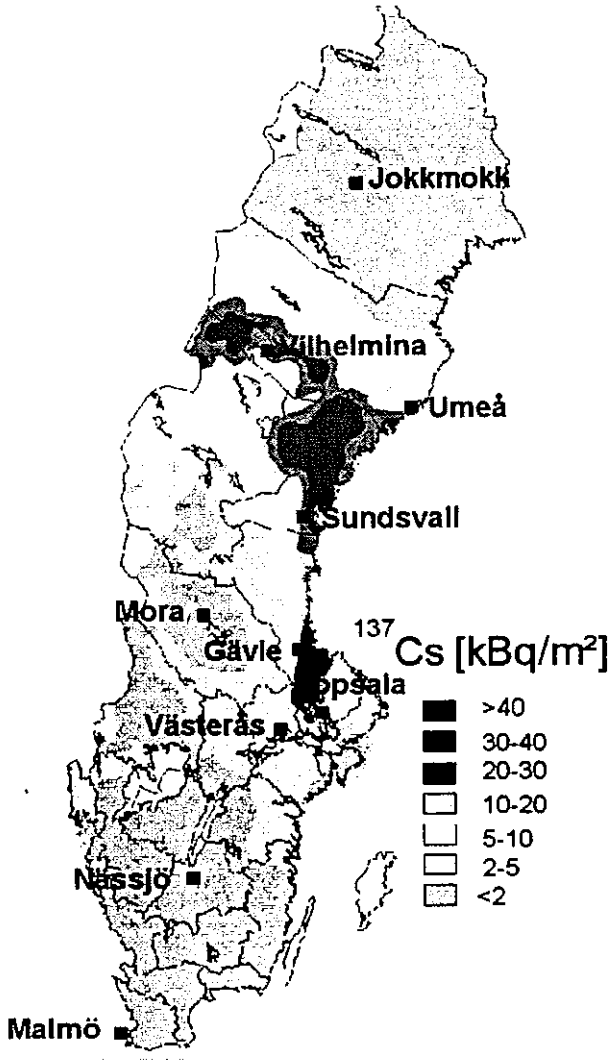

Fig 1. Distribution of ^{137}Cs fallout after the Chernobyl accident and sampling localities for the foodbaskets

Results

The baskets were subdivided into a liquid and a solid phase.
The beverages, i.e., juices, soft drinks and light beers, of the base baskets contributed less than 10 Bq to the intake. The average total base basket resulted in an estimated intake of 140 Bq.

In figure 2 the estimated population weighted mean values for the total intake of ^{137}Cs in 1994 for different areas are shown. The inland county of Västerbotten (Vilhelmina) received an intake of 820 Bq. The five counties most affected by fallout acquired 440 Bq and the rest of the country got 240 Bq. The box to the right represents the estimated population weighted mean value for the whole country. This is 270 Bq (0.75 Bq day^{-1}) with an estimated uncertainty of 50 Bq.

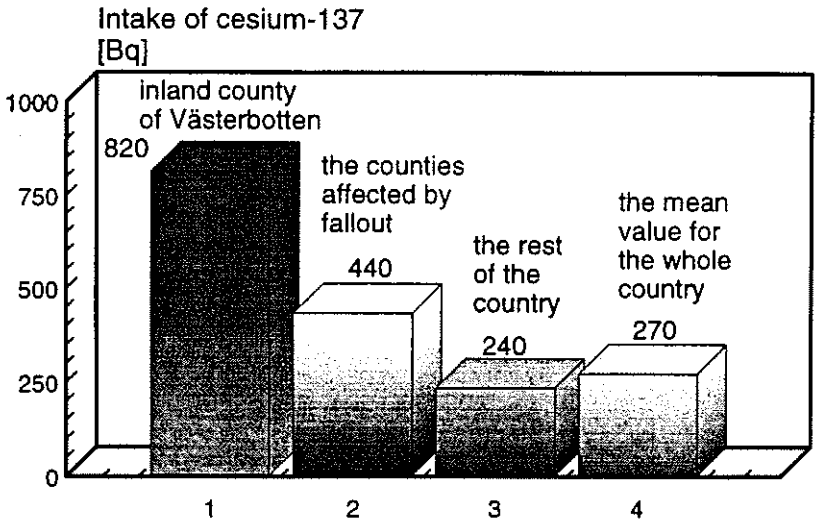

Fig 2. Population weighted mean values for intake of cesium-137 in 1994

Discussion

The committed effective dose equivalent is calculated to be 3.6 µSv from a discrete intake of 270 Bq. Committed dose means an integration over 50 years of the dose contribution from the intake.

The equilibrium level of the body burden resulting from the average intake, 0.75 Bq day^{-1}, can be calculated to be 1.3 Bq kg^{-1}. This can be compared to 2.0 Bq kg^{-1} obtained from whole body measurements in 1994 on a random sample of 179 persons from the whole population. The calculated value is based on an excretion time of 81 days for ^{137}Cs, which is a mean value weighted for ages above 7 years acquired from 379 whole body measurements at SSI 1986 - 87. The weight was 68 kg according to the 1994 random sample. When considering the accuracy of the calculated body burden one should bear in mind that the excretion times vary considerably between children, men and women and that there is a large variation within the groups.

The difference between calculated and measured body burden can have several sources. One explanation is that moose meat and freshwater fish were missing in 2/3 of the local foodbaskets, and reindeer and mushrooms totally. These foodstuffs generally have a higher activity concentration, thus possibly leading to an underestimation of the average intake.

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Reykjavík, August 26-30, 1996

MEASUREMENTS OF AIRBORNE BERYLLIUM-7 IN NORTHERN FINLAND

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Abstract - Daily high-volume aerosol samples (3500 m^3/d) have been collected at the Sodankylä meteorological observatory in northern Finland. The ^{7}Be contents of the samples were determined with semiconductor gamma spectrometry. The long-lived beta activity, i.e. total beta activity measured five days after the end of sampling (after the decay of ^{220}Rn progeny and short-lived ^{222}Rn progeny), was assessed with an automatic alpha/beta analyser. The long-lived beta activity consists mainly of ^{210}Bi, the daughter nuclide of ^{210}Pb, as the amount of artificial radionuclides in the air is negligible. The main goal of this project is to find the relative influence of stratospheric and upper tropospheric ozone versus the part from the continental boundary layer by the use of a ^{7}Be/^{210}Pb ratio index.

The air concentrations of both the beryllium-7 and the long-lived beta activity (≈ ^{210}Bi) displayed a lognormal distribution during the study period July-December 1995. The arithmetic mean of the daily ^{7}Be concentrations was 2360 µBq/m^3, range 230-7600 µBq/m^3 and the geometric mean 1950 µBq/m^3. Corresponding values for long-lived beta activity were 280, 9-890 and 240 µBq/m^3, respectively.

Despite the different origins the concentrations of both the beryllium-7 and the long-lived beta activity are partially correlated. Both have maximum concentrations during stable high-pressure situations. In Finland high-pressure situations often mean continental air masses with high ^{210}Pb contents. On the other hand the descending movement of air during high-pressure situations brings beryllium-7 downwards. Minimum concentrations are associated with cyclones travelling from the North Atlantic Ocean over Scandinavia to east. Within these low-pressure areas the subsidence rate is low and the maritime air masses contain very low ^{210}Pb concentrations.

Introduction

The Earth's upper atmosphere is continuously impacted by energetic charged particles, mainly electrons, protons and alpha particles. Most of this cosmic radiation arises from sources outside the Solar System, but Sun also contributes to this particle flux (REITZ, 1993). In addition to ions and radicals the cosmic rays produce also radionuclides ("cosmogenic" radionuclides) in the upper troposphere and stratosphere by inducing nuclear reactions in atmospheric oxygen and nitrogen atoms. One of the most abundant cosmogenic radionuclides is beryllium-7 ($t_{1/2}$ = 53 d). About 75 % of ^{7}Be is produced in the stratosphere and 25 % in upper troposphere (JOHNSON and VIEZEE, 1981).

Compared to ^{7}Be airborne lead-210 has a totally different source. It is formed in the atmosphere from the radioactive noble gas radon-222, a decay product of uranium-238, emanating from soil. High ^{210}Pb concentrations are found in continental air masses since ^{222}Rn sources at sea are practically non-existing. Lead-210 has been used as a tracer for

particle-bound sulphate because they both are secondary aerosols, i.e. produced in the atmosphere from their gaseous precursors ^{222}Rn and SO_2 (TUREKIAN ET AL., 1983). No increases of ^{210}Pb concentrations in surface air due to anthropogenic sources have been found (HÖTZL and WINKLER, 1987).

During recent years much attention has been paid to the biological effects of atmospheric ozone to man, animals and plants. The decrease in the amount of stratospheric ozone caused by e.g. chlorofluorocarbons increases the intensity of solar UV radiation on earth. On the other hand, the increase in the amount of tropospheric ozone is also hazardous (HOUGHTON ET AL., 1994). In high northern latitudes the tropospheric ozone has several sources: net import from continental industrialized regions, *in situ* production, lightnings and forest fires and influx from stratosphere (MAUZERALL ET AL., 1996). The ozone influx from stratosphere can be assessed using $^{7}Be/^{210}Pb$ index as a tracer for stratospheric and upper tropospheric air masses, although the fluctuations of ^{7}Be activity concentrations in the surface air have several causes: the variations in the rate of exchange of air through the tropopause, the variations in vertical mixing of the troposphere, variations in horizontal transport of air masses from middle latitudes towards polar regions and the effect of precipitation (GRAUSTEIN and TUREKIAN, 1996; FEELY ET AL.,1989).

Materials and methods

Daily aerosol samples have been collected with a high-volume sampler onto Munktell MGA glass fibre filters (∅ 24 cm) at the Sodankylä Meteorological Observatory (67°22'N, 26°39'E) in Finnish Lapland. The sampler has a capacity of 3500 m^3 per day and it collects particles with an aerodynamic diameter less than 10-15 µm depending on the wind speed.

The filters were measured in the laboratory with an automatic alpha/beta analyser (MATTSSON ET AL., 1996). The detector arrangement of the instrument consists of five gas flow proportional counters with 25 cm x 25 cm windows. The flow gas is a mixture of Ar (90 %) and CH_4 (10 %). The counting efficiency for alpha particles (^{210}Po) is 42 % and for beta particles (^{210}Bi) 76 %. The sample changer unit can handle 25 samples per measuring sequence. A background sample (an unexposed filter) is measured at least twice during a measuring sequence. Similarly three reference samples (^{242}Pu, ^{90}Sr and ^{55}Fe) are measured at least once per measuring sequence to monitor the stability of the instrument.

The total beta activities of the aerosol samples were measured five days after the end of sampling, when the short-lived ^{222}Rn progeny had decayed into ^{210}Pb and the ^{220}Rn progeny had decayed into stable lead. The measured total beta activity, "long-lived beta activity", consists mainly of ^{210}Bi (usually not yet in equilibrium with ^{210}Pb) and possible artificial beta emitters. The ^{7}Be activity concentrations were determined with semiconductor gamma spectrometry.

Results and discussion

The concentrations of ^{7}Be and long-lived beta activity in the surface air during the study period July-December 1995 are presented in Figures 1 and 2. The concentrations of both the beryllium-7 and the long-lived beta activity (≈ ^{210}Bi) displayed a lognormal distribution

(Figures 3 and 4). The arithmetic mean of the daily ^{7}Be concentrations was 2360 μBq/m^3, range 230-7600 μBq/m^3 and the geometric mean 1950 μBq/m^3. Corresponding values for long-lived beta activity were 280, 9-890 and 240 μBq/m^3, respectively.

Despite the different origins the concentrations of both the beryllium-7 and the ^{210}Pb (long-lived beta activity) are partially correlated. Both have maximum concentrations during stable high-pressure situations (Figure 5). In Finland high-pressure situations often mean continental air masses with high ^{210}Pb contents. On the other hand the descending movement of air during high-pressure situations brings beryllium-7 downwards. Minimum concentrations are associated with cyclones travelling from the North Atlantic Ocean over Scandinavia to east (Figure 6). Within these low-pressure areas the subsidence rate is low and the maritime air masses contain very low ^{210}Pb concentrations. Situations with high (≈49) and low (≈2.4) ratios of ^{7}Be to long-lived beta activity are presented in Figures 7 and 8. In these cases the ^{7}Be concentrations differ only by a factor of two while long-lived beta activity concentrations differ by a factor of 9. In the first case (Fig. 7) the ^{210}Pb concentration is low because the air mass is coming from Arctic regions where practically no radon sources exist. In the latter case the wind is blowing from southerly direction bringing continental air masses with high ^{210}Pb concentration to northern Finland.

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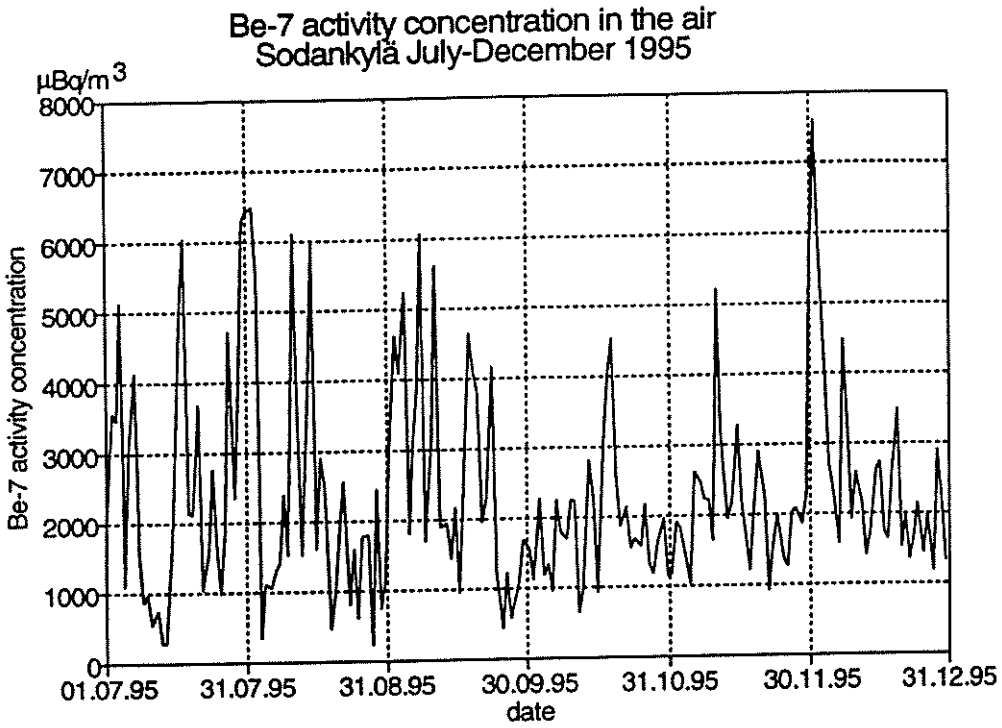

Figure 1. Daily ^{7}Be concentrations in the surface air (µBq/m^3), Sodankylä July-December 1995.

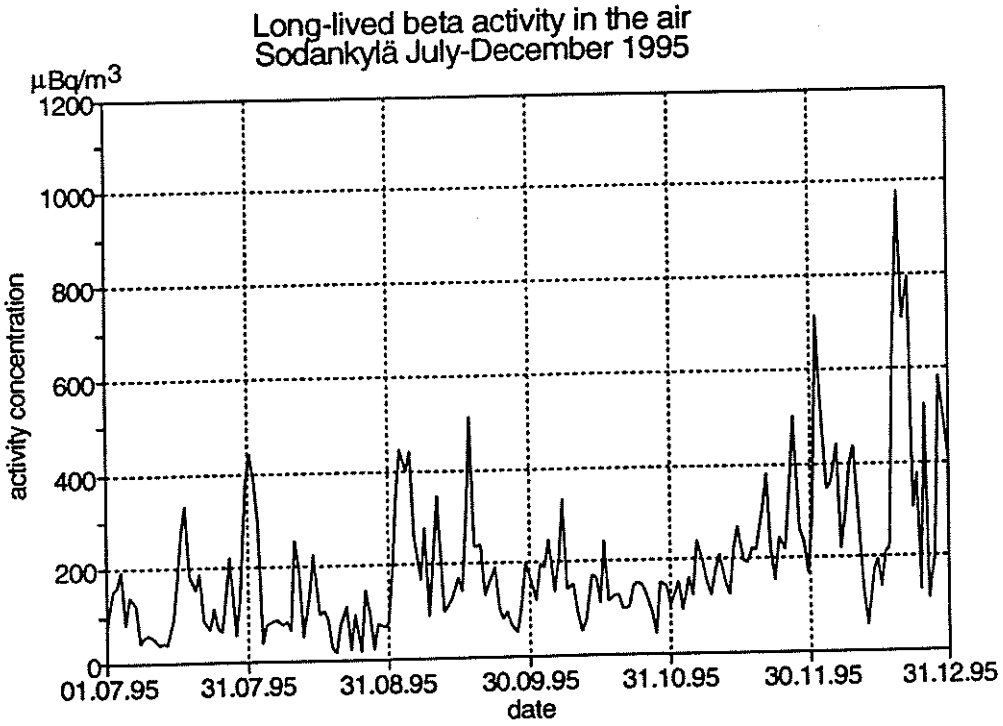

Figure 2. Daily concentrations of long-lived beta activity in the surface air (µBq/m^3), Sodankylä July-December 1995.

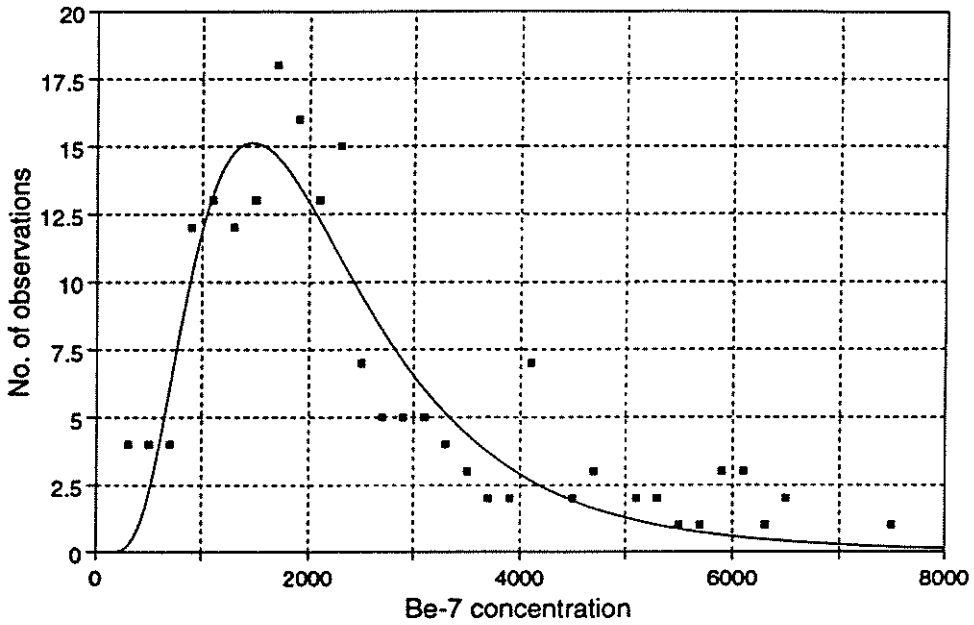

Figure 3. Distribution of daily ^{7}Be concentrations in the surface air, Sodankylä July-December 1995.

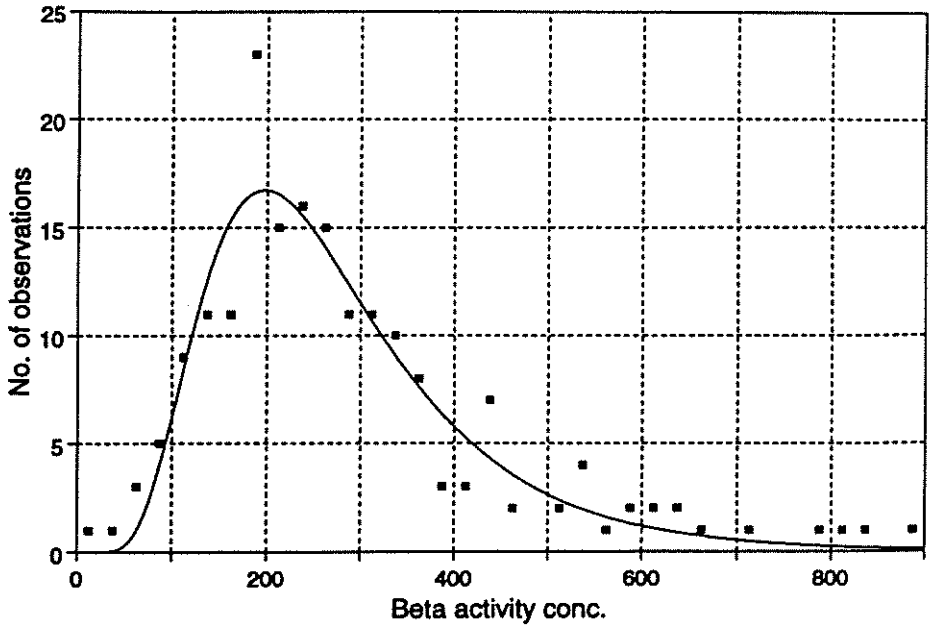

Figure 4. Distribution of daily concentrations of long-lived beta activity in the surface air, Sodankylä July-December 1995.

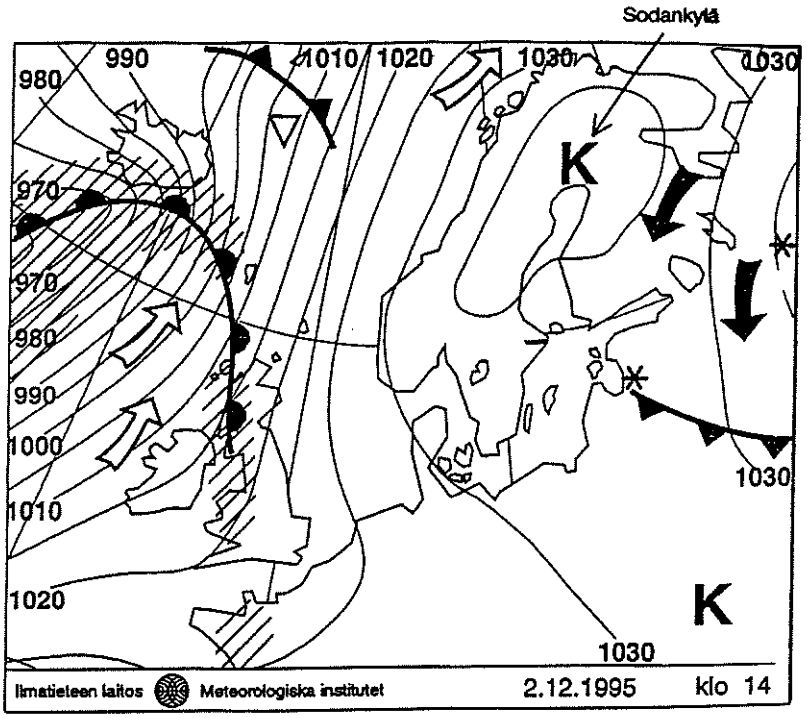

Figure 5. Weather situation on 2 December 1995 12 UTC, a day with high ^{7}Be and long-lived beta activity concentrations at Sodankylä (K = high-pressure, M = low-pressure). The location of Sodankylä is marked with an arrow.

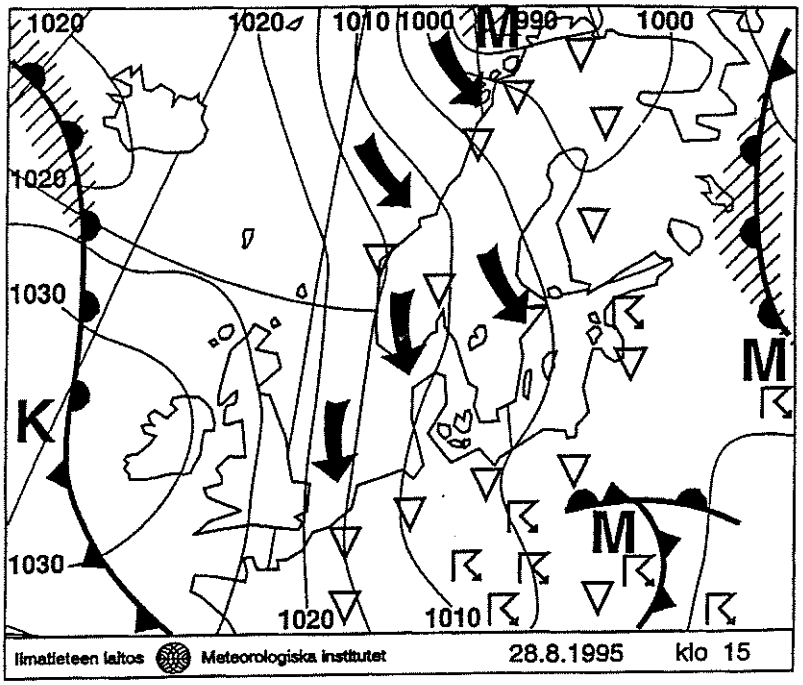

Figure 6. Weather situation on 28 August 1995 12 UTC, a day with low ^{7}Be and long-lived beta activity concentrations at Sodankylä (K = high-pressure, M = low-pressure).

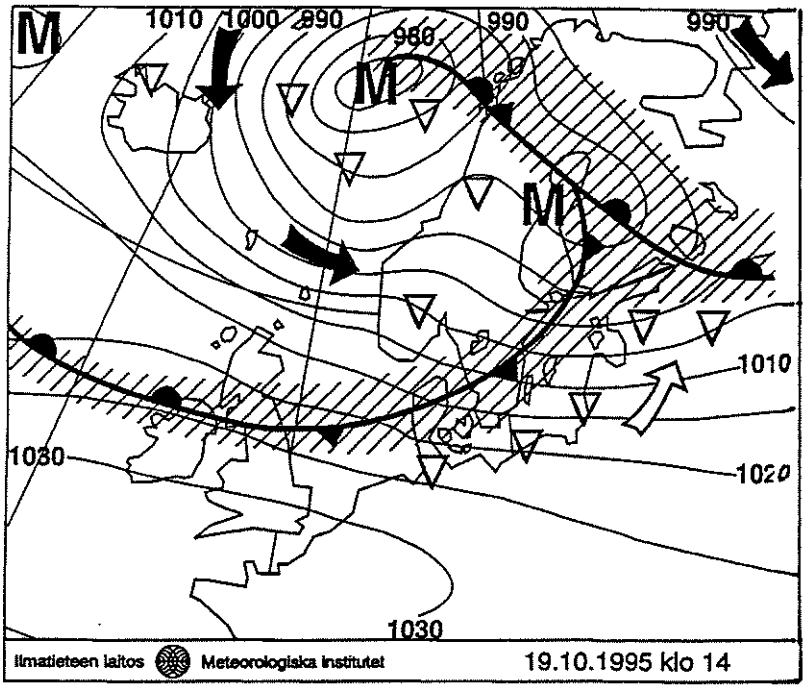

Figure 7. Weather situation on 19 October 1995 12 UTC, a day with a high ^{7}Be/long-lived beta activity concentration ratio at Sodankylä (K = high-pressure, M = low-pressure).

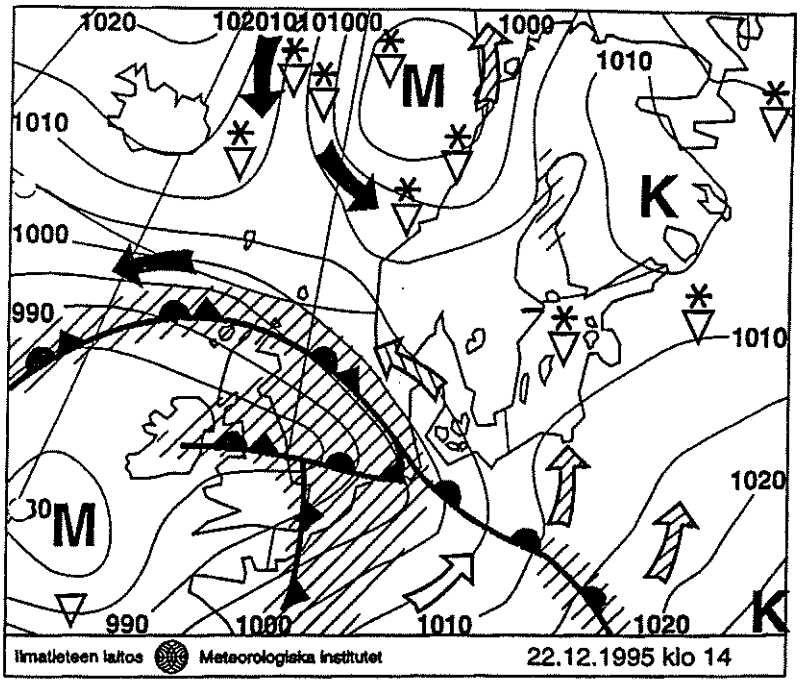

Figure 8. Weather situation on 22 December 1995 12 UTC, a day with a low ^{7}Be/long-lived beta activity concentration ratio at Sodankylä (K = high-pressure, M = low-pressure).

User Specific Environmental Transport Analyses Code DETRA

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A fully modernized version of the earlier presented computer code DETRA (Doses via Environmental Transport of Radionuclides) is developed. The present version of the code permit for the user to perform analyses in Windows interface environment, and new features for the calculation possibilities of the code are added: e.g. applications of multiple calculation sequences for different compartment model structures are now also available for the user.

DETRA is a suitable tool for various types of environmental analyses problems. Radionuclide transfer models for aquatic fresh water as well as for marine environment, soil layers, domestic animals products and several other applications of consequences of radioactive depositions in general have been analyzed with the code. The new interface of the code makes it also easy for the user to save and utilize afterwards ready input files in subdirectories of relevant research areas. The recently developed Windows interface of DETRA include own displays for each of the segments of input file: nuclide chains, compartments, time grid, source term, initial values and formats for printout specifications. The numerical data and graphics of calculation results are shown in their own displays, correspondingly. Examples of the input and output displays of DETRA are shown below in the figure.

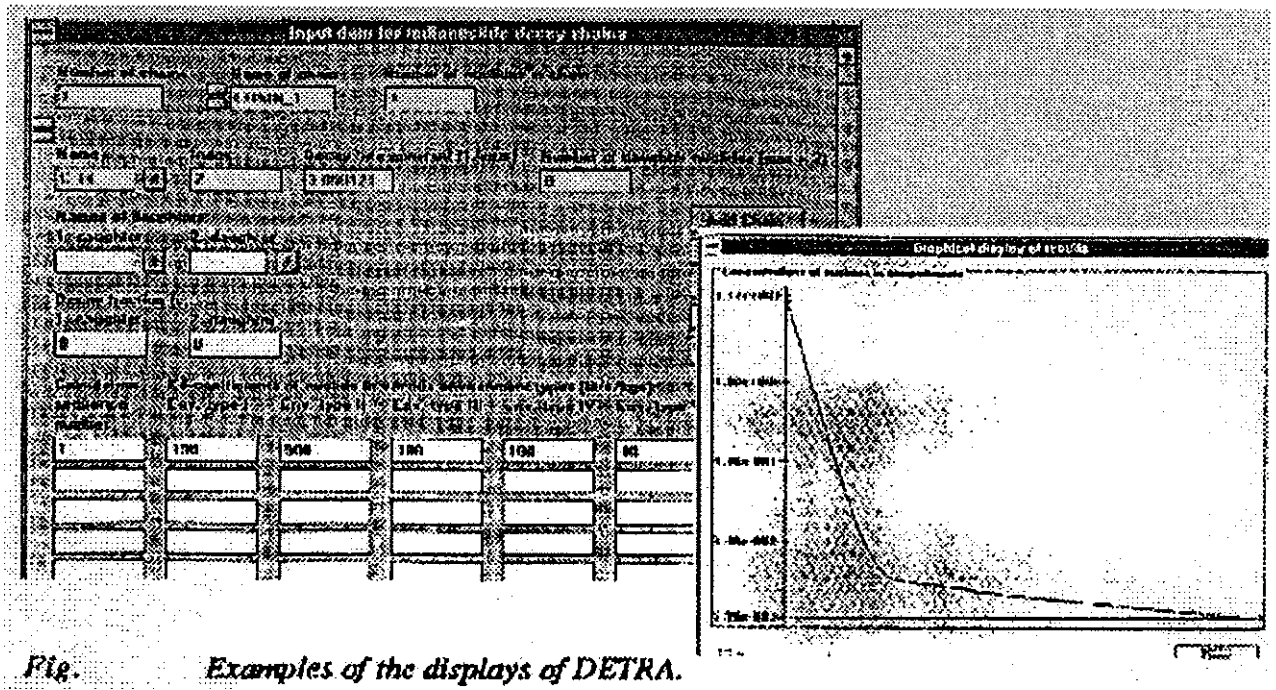

Fig. Examples of the displays of DETRA.

RETROSPECTIVE RADIATION DOSIMETRY USING OPTICALLY STIMULATED LUMINESCENCE ON NATURAL MATERIALS AND CERAMICS FOR ASSESSING POPULATION DOSES IN NUCLEAR ACCIDENTAL AREAS

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Abstract

Optically stimulated luminescence (OSL) techniques especially aimed at using natural materials for retrospective reconstruction of accidental radiation doses in populated areas were developed and studied at Risø as part of an EU research project. Quartz and feldspars separated from building materials, such as bricks and tiles, in addition to porcelain from toilet tanks had their OSL signals measured using different light sources for stimulation to assess radiation doses received by the material. Radiation doses were also evaluated from OSL measured directly on unseparated samples i.e. directly from the surface of brick and tile materials. The lower detection level for e.g. quartz extracted from a modern brick measured using a green light wavelength band as the stimulation light source was found to be less than 1 mGy. The techniques developed and applied are described and results from measurements carried out on a variety of materials are presented.

Introduction

Optically stimulated luminescence (OSL) arises from recombination of charge which has been transferred into metastable locations in the lattice of materials as a result of irradiation, and thus is related to the radiation dose which the materials has received. During the exposure to the stimulation light the OSL signal decreases down to a low level.

Application of OSL for dating of sediments was first demonstrated by Huntley et al. (1985) who used the green light from an argon laser (514.5 nm) to stimulate luminescence from quartz. Luminescence emitted during recombination of the detrapped charges is measured in a spectral region different from that of the exciting photons and is proportional to the radiation dose absorbed in the material. The physical principles of this technique are closely related to those associated with the well known thermoluminescence (TL) technique where heating is used for excitation. However, using light instead of heat to induce recombination clearly simplifies the instrumentation, since there is no need for accurate thermal controls.

The instrumental simplicity of optical stimulation also makes this technique attractive for radiation dosimetry using natural materials collected in the environment. Retrospective dose reconstruction after radiation accidents can be accomplished on the basis of environmental materials which were exposed to radiation during the accident and possess the ability of producing recombination luminescence. Natural dosemeters meeting this requirement are quartz and feldspar inclusions found in fired materials such as bricks, tiles and porcelain where the firing has erased any previously absorbed dose. The cumulative dose accrued from natural sources as well as the accidental dose can be measured either by thermoluminescence (Bailiff and Haskell 1984, Haskell 1993, Stoneham 1985) or by optically stimulated luminescence (Godfrey-Smith and Haskell 1994).

In an effort to apply optical stimulation for retrospective radiation dosimetry a number of basic studies of OSL techniques were undertaken at Risø. The materials investigated include quartz and feldspars extracted from bricks, outdoor tiles and porcelain from toilet tanks.

Apparatus and techniques

The apparatus used for the experimental work were mainly OSL units developed as attachments to the automated Risø TL reader (Bøtter-Jensen 1987) and include a scanning monochromator for wavelength resolved luminescence measurements. An instrument for continuous OSL scanning of sediment cores was also developed. The latter technique naturally lends itself to the continuous scanning measurements of brick cross-sections, allowing radiation depth dose profiles to be measured directly.

The basic OSL unit, containing light sources for both green light and infrared stimulation, enables measurements of OSL signals from both quartz and feldspar samples (Bøtter-Jensen and Duller 1992). Green light stimulated luminescence (GLSL) is achieved by illumination with a filtered light spectrum from a halogen lamp using exchangeable excitation and detection filter packs. The GLSL unit is designed to select a green light stimulation wavelength band using excitation filters extending to as low a wavelength as possible while still being sufficiently separated from the luminescence emission spectrum. The infrared stimulated luminescence (IRSL) is generated by an infrared diode array (peak emission of 875 $\pm$ 80 nm) placed close to the sample. While GLSL can be used with both quartz and feldspars, IRSL seems only to work with feldspars.

Ideally, however, the spectral excitation and emission characteristics of quartz and feldspar materials prepared for dosimetric evaluation would be routinely scanned since this would also allow the possibility of choosing the most suitable energy windows in which to carry out the measurements. A compact module was developed that allows for the monochromatic illumination of samples in the wavelength range 380 to 1020 nm, enabling the measurement of energy resolved OSL (Bøtter-Jensen et al 1994). The unit can be directly coupled to the existing automated Risø TL/OSL system. The unit can also be used for recording wavelength resolved emission spectra, whether photo excited or thermally stimulated. A schematic diagram of the combined OSL attachment is shown in Fig. 1.

The continuous OSL core scanner system allows the optical sensors to be moved across either sediment or brick cores. A stepper motor drive ensures constant scan rates and accuracy in positioning to better than 0.1 mm. The optical sensor system was developed at Risø and consists of a photo-excitation and detector module together with lamps for bleaching and regenerating OSL. Photo excitation is made using a filtered halogen lamp generating a green wavelength band (420-550 nm) and the brick core is scanned using an excitation slit beam of 10 mm x 1 mm which determines the resolution of the system. Photo detection is made through a 6 mm standard Hoya U-340 filter (peak emission at 340 nm). OSL dose normalization is made either by using short wave UV light from a 20 W low pressure HG lamp or exposing the brick cores to a Cs-137 gamma field and afterwards scanning the OSL sensitivity across the brick profile. A schematic diagram of the OSL scanner system is shown in Fig. 2.

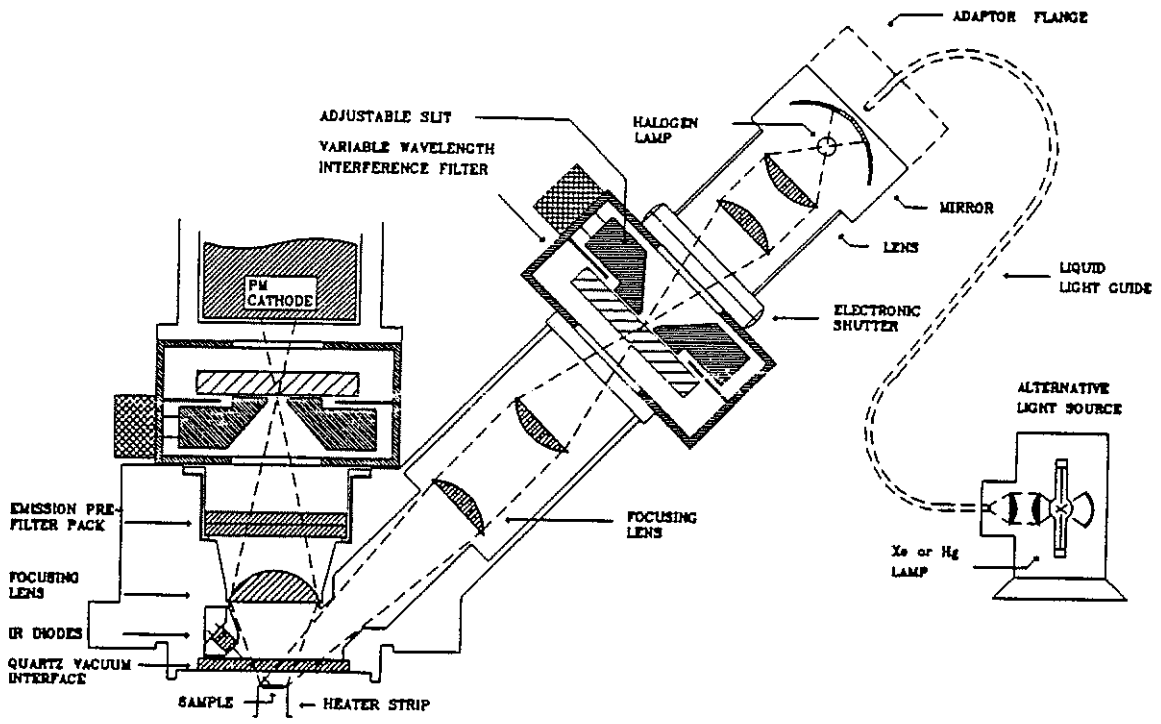

Fig. 1. Schematic diagram of the OSL attachment showing the excitation lamp system with monochromators mounted both on the excitation side and the detection side.

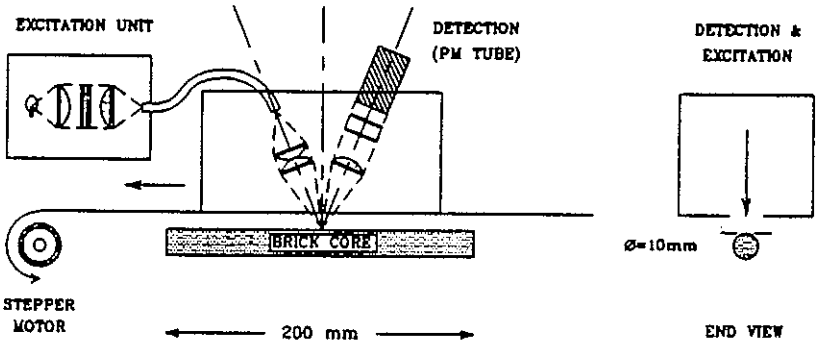

Fig. 2. Schematic diagram of the automatic OSL brick core scanning system.

Results

OSL sensitivity of quartz depends very much on the thermal history of the material: fired samples giving up to an order of magnitude larger signals than non-heated samples. An attempt to determine the lower detection levels using green light stimulated luminescence on fired quartz was made by obtaining dose versus GLSL response curves for a variety of quartz samples extracted from archaeological specimens such as bricks, burnt stones and clay. The GLSL response curve for a sensitive quartz extracted from burnt clay obtained using the multiple aliquot method is shown in Fig. 3A. As seen, the lowest detectable dose for this material is well below 1 mGy.

A single aliquot method was also tested on the same quartz sample as above where the dose response curve was obtained using the regeneration technique. Beta irradiations were carried out using the Sr-90 source of the Risø TL/OSL reader and GLSL measurements were carried out as described above. The results obtained show very little scatter because of the single

aliquot technique used where no normalisation is needed. Fig.3B presents the dose versus GLSL response curve in the dose range 0.1 to 5 mGy compared to that of the TL response. The lower detection level using GLSL on this particular sample is seen to be well below 0.5 mGy.

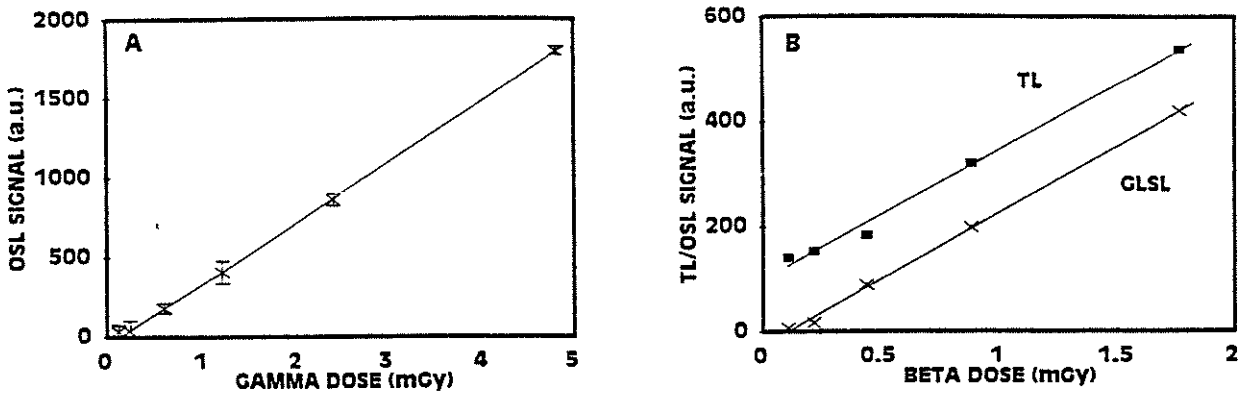

Fig. 3. (A) OSL versus Co-60 gamma dose (multi sample technique) for quartz extracted from a burnt stone. (B) TL and OSL versus beta dose (single aliquot method) for the same quartz sample.

The accumulated "natural" dose induced in quartz by radionuclides contained in modern brick materials and the environmental gamma radiation was measured using GLSL on extracted quartz samples. This experiment was aimed at determining the lower detection limit for an additional dose received by a brick as a result of radioactive release from a nuclear accident taking into account the GLSL contribution from the natural background dose. The quartz grains were extracted from the material and the absorbed dose was determined by GLSL using the additive dose technique. The dose was estimated to be about 200 mGy which is in very good agrement with the expected value based on an annual dose rate of about 5 mGy/y from the environmental radiation and the natural radioactivity in the brick. For this particular brick a lower detection limit for an additional accidental dose would be in the order of 20 mGy (10% above the background).

Depth dose profiles in bricks can be determined by measuring the OSL signals directly from the unseparated material across the brick. Modern bricks were annealed at 500°C to remove any previously acquired TL/OSL signal and then exposed to Co-60 and Cs-137 photon radiation fields, respectively. After irradiation, 8 mm diameter cores were drilled out of the brick and sliced into 1 mm thick circular discs using a diamond saw. Each disc, representing a particular depth in the brick, had their GLSL measured directly from the surface of the unseparated material and as an example the OSL versus depth for an ancient brick irradiated with Co-60 radiation is shown in Fig. 4A. As seen, the half value layer value is about 75 mm which corresponds reasonably well with the expected attenuation of Co-60 photons in brick material. A brick collected from a house in the town Berezyaki in the Chernobyl area was further measured using the same procedure and the GLSL signal corresponding to the natural plus accidental dose versus depth is shown in Fig. 4B. The more rapid attenuation seen here is in contrast to that obtained from the Co-60 irradiation and demonstrates that the Chernobyl brick was exposed to a gamma spectrum with a much higher content of low-energy photons.

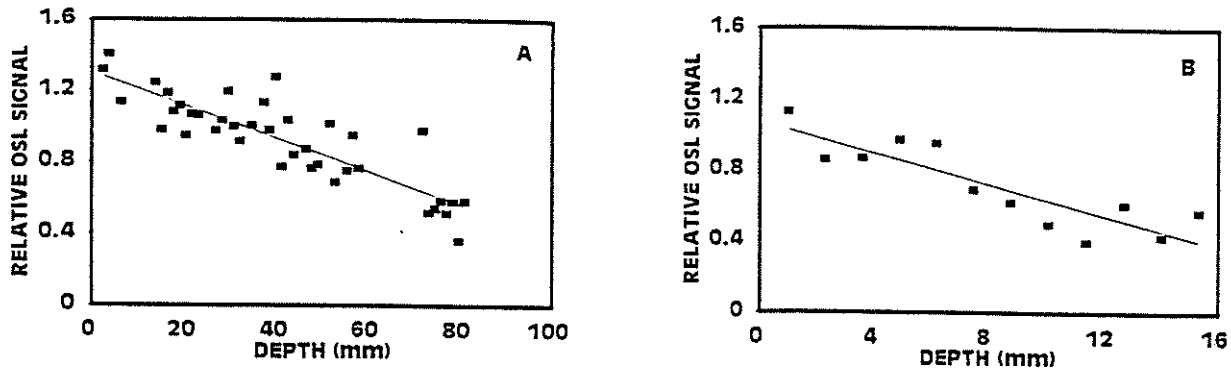

Fig. 4. OSL versus depth into an ancient brick measured with green light stimulation on 1 mm discs drilled and cut through a brick that had been irradiated to 5 Gy Co-60 radiation from one side. (B) OSL versus depth into a brick collected at Chernobyl that had been exposed to the "accidental" dose. Same technique was used as in (A).

The automatic core scanner system was used to measure the dept dose profiles across the 200 mm length of a modern brick exposed in the laboratory to Co-60 and Cs-137 gamma radiation doses, respectively. 10 mm cores were drilled from the brick after irradiation and mounted directly under the light sensing head of the core scanner. The cores were scanned with a speed of 1 mm per sec. using an excitation light density of 20 mW/cm2. Normalisation was made after bleaching the cores either by exposing the cores by UV light produced by the attached low-pressure Hg lamp or by exposing the cores perpendicularly to a Cs-137 gamma field. Figs 5A and 5B show the dose depth profiles obtained from OSL scanning of cores from a brick that had been exposed from one direction in the laboratory to Cs-137 and Co-60 radiation, respectively (20 Gy). Monte Carlo calculated attenuation curves for the same irradiation geometries are shown as well and as seen they compare well with the experimentally obtained curves.

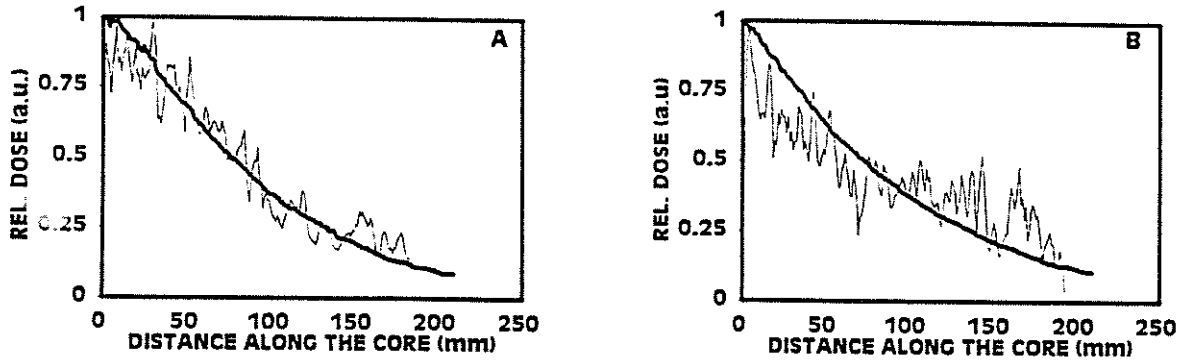

Fig. 5. (A) Relative depth dose profile measured with the automatic OSL scanner on a core from a brick that had been irradiated to 20 Gy Cs-137 gamma radiation from one side. The bold line shows the Monte Carlo calculated attenuation curve for comparison. (B) Same as (A) but for 20 Gy Co-60 gamma radiation.

Finally, an assessment was made as to whether the OSL technique can be used as an alternative method for the established thermoluminescence technique for measuring accrued dose levels in porcelain: since no heating is required, problems of heat induced sensitivity changes during measurement are overcome. The OSL versus beta dose for a porcelain sample was found to increase linearly with dose in the range 0-16 Gy and shows a further sublinear increase of OSL signal up to at least 200 Gy. These preliminary measurements indicate that dose levels of about 0.1 Gy can be determined using OSL techniques on porcelain. Porcelain

samples extracted from outdoor electrical insulators and lamp fittings collected at different locations near the Chernobyl Nuclear Power Plant had their total absorbed doses assessed by OSL. The OSL signals were measured on 8 mm diameter x 1 mm discs made from the samples and using green light stimulation. The total doses determined from 6 different specimens varied from about 1 to 3,9 Gy: the highest doses being measured from electrical insulators collected in the Red Forest hot area near the power plant.

Conclusions

Different OSL techniques developed recently at Risø for determining accrued doses in natural materials have been described. These include scanning techniques for assessing depth dose profiles in bricks by measuring OSL directly on the surface of the raw material. Quartz extracted from fired archaeological specimens showed high OSL sensitivity and the lower detection limit for these materials was found to be less than 1 mGy. Porcelain showed interesting OSL properties and the high OSL sensitivity found especially for the glazing layer of e.g. toilet tanks suggests that this particular material can be used with advantage for assessing low accidental doses.

Acknowledgement

This work is partly funded by the European Union under the research contract FI4P-CT95-0011 "Dose Reconstruction".

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Dosimetry of Beta and Low-Energy Photon Radiation: Standardisation, Individual Dosimetry and Monitoring of the Workplace

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Abstract-The paper discusses recent progress achieved in the field of dosimetry of low-penetrating radiation, in particular that obtained from joint European studies. New developments of calibration facilities including both radiation fields from beta particles and low-energy photons with photon energies from 6-15 keV are described. Dosemeters intended for individual monitoring for skin doses from low-penetrating radiations should contain detectors with a small effective thickness in order to give responses that are independent of energy and angle of incidence of the radiation. The paper presents performance data of new thin highly sensitive solid state detectors and furthermore new developments of electronic dosemeters particularly suited for personal dosimetry of weakly penetrating radiation are discussed. In the field of area monitoring of workplaces the limitation of presently used survey instruments for monitoring for low-penetrating radiations is discussed. A more precise characterisation of beta radiation fields can be obtained by use of beta spectrometers and a new design of spectrometer consisting of three different silicon detectors arranged in a telescope unit is discussed.

Introduction

Weakly penetrating radiation consisting of beta rays and low-energy photons, i.e. photons with energies below 15 keV, is part of the radiation environment of the radiation worker and may, if not properly protected against, contribute with significant doses to the skin. At workplaces in German nuclear plants dose equivalent rates up to 1-2 Svh^{-1} were measured [1]. The ratio of dose rates, weakly to strongly penetrating radiation, varies significantly from one working place to another. Typically, ratios from 1:1 to 10:1 were measured in German nuclear power plants; however in some cases values of up to 40:1 were found [1]. In most cases weakly penetrating radiation, present at the nuclear power plants will be satisfactory shielded against by use of protective clothings and thick rubber gloves with a density thickness of about 50 $mgcm^{-2}$ [1,2]. However, in some cases where high-energy beta emitters are present protective clothings and gloves will not provide adequate protection [1]. In nuclear medicin high dose rates may occur in handling solutions with high-energy beta emitters, e.g. ^{90}Y and ^{32}P. Exposure to low-energy photon radiation is generally not considered to be a serious radiation problem in reactor areas, although dose contributions from low-energy photons due to K radiations and scattered photons occur. In recent years low-energy photon radiation from synchroton radiation facilities has found increasing application in various research fields, e.g. physics, biology, chemistry, medicine and technology. There are currently over thirty synchroton radiation facilities in operation around the world and approximately another thirty are in the construction, design or proposal stage [3]. With the rapidly increasing number of persons engaged in experiments with low-energy photon radiation there is a growing interest in performing accurate dosimetry in this field.
In particular, in connection with nuclear accidents and the following clean-up phases there exist potential hazards from exposure of the skin to large doses from weakly penetrating radiation. During the accident at the Chernobyl nuclear power plant in 1986 and in the post-accident period workers received significant doses, of the order of up to tens of Gy, to the skin from beta and low-energy photon radiation [4]. As a consequence, skin burns due to exposure from beta radiation were observed for a great number of workers. These burns significantly complicated the clinical prognosis of the irradiated persons and often contributed to or caused their death.
It was recognised, specificially during the work concerned with the clean-up phase of the Three Mile Island accident and further confirmed by the problems involved with the Chernobyl accident

that there was a need for improving the dosimetry techniques used both for individuel as well as workplace monitoring of weakly penetrating radiation. The dosemeters used for routine personnel monitoring had essentially been developed aiming at monitoring for photon exposures with the result that the thickness of the front cover as well as that of the detector itself was unacceptably large for beta dose measurements, implying a considerable underestimation of the dose to be measured. Also the survey instruments used for workplace monitoring were not adequate for monitoring for weakly penetrating radiations. Most frequently survey instruments based on ionisation chambers with a geometry unsuited for beta dose rate measurements were used. Therefore, during the last 15 years significant efforts have been devoted to studies and developments in the field of dosimetry of low-penetrating radiations. Several US and European conferences, workshops, seminars and intercomparisons have been devoted exclusively to this field and considerable financial supports have been given to research work in the field both in USA, Europe and other countries.

This paper discusses recent progress obtained in the field of dosimetry of weakly penetrating radiation, in particular that obtained from joint European studies covering problems involved with standardisation, individuel dosimetry and area monitoring of the workplace.

Standardisation

Dose limits and quantities to be measured

According to ICRP [5] a weighing factor of 0.01 for exposures of the skin is adequate for estimating the risk of skin cancer; however to prevent undesirable deterministic effects from localised exposures the occupational dose equivalent to the skin, including hands and feet, should be limited to 500 mSv per year. The skin dose equivalent should be determined at a nominal depth of 0.07 mm (7mgcm^{-2}) and averaged over an area of 1 cm^2, regardless of the area exposed. Determination of this quantity is the most common objective of beta dosimetry. In a few cases where high-energy beta rays are involved it will also be necessary to control the dose to the the eye lens. The ICRP has recommended a limit of 150 mSv per year for the lens of the eye which should be determined at a nominal depth of 3 mm (300 mgcm^{-2}).
As operationel quantities for radiation protection dosimetry of weakly penetrating radiations the ICRU recommends the directional dose equivalent $H'(d,\alpha°)$ related to the ICRU sphere for area monitoring and the personal dose equivalent $H_p(d)$ in a phantom having a composition of ICRU tissue for individuel monitoring [6]. For skin dosimetry a value of the depth, d = 0.07 mm should be used. It has been proposed to define the skin dose equivalent quantity for three phantoms made of ICRU soft tissue [7]:

- slab phantom, 300 mm x 300 mm x 150 mm, to approximate the human torso.
- pillar phantom, a cylinder 73 mm in diameter and 300 mm in length, to approximate a lower arm and leg.
- rod phantom, a cylinder 19 mm in diameter and 300 mm in length, to approximate a finger.

In addition, also proposals for practical phantom designs for calibration of personal dosemeters are given [7]

The Extrapolation Chamber

The extrapolation chamber measurement method is the basic method for determination of absorbed dose rates to tissue from exposure in beta radiation fields. The extrapolation chamber is an ionisation chamber operating with a variable chamber volume. By measuring the ionisation current at

decreasing chamber volumes the absorbed dose rate at zero-volume can be determined. Depth-dose curves can be obtained by using different absorbers placed in front of the entrance window.
Calibration facilities for weakly penetrating radiation should be able to provide the quantities $H'(d,\alpha°)$ and $H_p(d)$, the latter given for the three different phantoms mentioned above. In praxis the absorbed dose rate to tissue at tissue depth, d, and irradiation angle, α, $\dot{D}_t(d,\alpha°)$, measured by the extrapolation chamber will be good approximations of $\dot{H}'(d,\alpha°)$ and $\dot{H}_p(d)$ and the extrapolation chamber is usually used to establish the quantities needed for beta reference radiations. Similar considerations as those given above for beta radiation will be valid also for low-energy photon radiation (photons with energies below 15 keV).
The extrapolation chamber measurement method is sophisticated requiring a number of correction factors to be applied. Joint European studies of problems connected to the extrapolation chamber measurement method have resulted in improved standardised measurement and evaluation procedures [8]. A complete protocol for the extrapolation chamber measurement method is being developed as one of the task of a new research project concerning dosimetry of low-penetrating radiation within the EU Radiation Protection 4th Framework Programme 1994-1999.

Beta Calibration Facilities

Reference beta radiations for calibrating dosemeters are specified in the international standard, ISO 6980 [9]. The beta emitting radionuclides ^{14}C, ^{147}Pm, ^{204}Tl, $^{90}Sr/^{90}Y$, and $^{206}Ru/^{206}Rh$ are explicitly recommended. Set-ups of secondary standard beta calibration sources calibrated at PTB, Germany, have been commercially available from the Buchler company for about 20 years [10]. The sources are delivered with a calibration certificate providing data on absorbed dose rates at different tissue depths obtained by use of the extrapolation chamber. Other characteristics of the standard radiation fields have been reported by various users [10]. Table 2 presents typical dose rates available from the PTB/Buchler sources. A new upgraded version of the standard beta calibration facility with higher dose rates for the two lower beta energies and with computer-controlled irradiation procedures including irradiation at different angles of incidence of the radiation, is under development at PTB and is expected to be commercially available in 1997.

Table 2. Characteristics of the PTB/Buchler secondary standard beta sources.

Radionuclide	$^{90}Sr/^{90}Y$	$^{90}Sr/^{90}Y$	^{204}Tl	^{147}Pm
Nominal Activity (MBq)	1850	74	18.5	518
Beam-flattening filter	No	Yes	Yes	Yes
Irradiation distance (cm)	30	30	30	20
Approximate value of $D_t(0.07,0°)$ ($mGyh^{-1}$)	300	10	1	0.3

In addition to the beta calibration beams provided by the PTB/Buchler sources, there is for various dosimetry purposes a need also for beta calibration beams with other energies and dose rates. The establishment of such calibration beams implies the characterisation of the radiation fields in terms of $\dot{D}_t(d,\alpha°)$ by use of extrapolation chamber. In addition a number of other characteristics of the fields, e.g. dose rate homogeneity, maximum residual energy, etc. should be studied and compared with the ISO specifications. In recent joint European research projects data have been obtained for a great number of beta radiation fields from extended area sources [8]. At Risø, in particular, work has focused on the characterisation of the radiation fields from the two low-energy beta radiation sources, ^{14}C with E_{max}= 156 keV and ^{63}Ni with E_{max}= 67 keV. Due to the strong absorption in air of the

beta particles from ^{63}Ni the source was placed in a container that could be evacuated. In this way it was possible to achieve a dose rate of 235 $mGyh^{-1}$ at a distance of 20 cm from the source.

Low-energy Photon Calibration Facilities

An X-ray facility which can provide filtered X-rays with energies below 15 keV was established at CEC, Fontenay-aux-Roses, France in connection with a joint European research project on dosimetry of low-penetrating radiations [8]. Reference beams with average energies of 6.5, 8.5 and 12.2 keV were calibrated in terms of $\dot{D}_t(0.07;\alpha°)$ using an extrapolation chamber. Also air kerma rates were determined with a free air ionisation chamber. In addition, a 6 keV, 37 GBq ^{55}Fe, photon calibration facility has been established at Risø and characterised in terms absorbed dose rates to air and tissue by use of the extrapolation chamber. Characteristics of the low-energy photon calibration beams are given in Table 3.

Table 3. Characteristics of four low-energy photon calibration beams.

Irradiation beam	$E_{average}$ (keV)	Calibration distance (cm)	Air kerma rate[1)] ($mGyh^{-1}$)
7.5 kV X-ray; No filter[2)]	6.5	139	18.8
10 kV X-ray; 0.1 mm Al filter[2)]	8.5	139	38.4
15 kV X-ray; 0.5 mm Al filter[2)]	12.2	139	46.5
^{55}Fe, 5.9 keV(80%)+6.5 keV (20%)	6.0	40	3.30

[1)] Measured with extrapolation chamber
[2)] Inherent filtration: 1 mm Be

Individual Monitoring

The basic requirement of a personal dosemeter for routine monitoring for doses from weakly penetrating radiation is to measure $H_p(0.07)$ from beta and photon radiation with satisfactory accuracy independently of energy and angle of incidence of the radiation. Thin detectors with a mass thickness below 10 $mgcm^{-2}$ are preferable to obtain satisfactory energy and angular response and furthermore they must show a sensitivity high enough to measure a skin dose below 0.5 mSv and preferably below 0.2 mSv. The dosemeter response should not be significantly influenced by environmental parameters such as temperature, atmospheric humidity and light. In addition to the dosimetric requirements the dosemeter should be designed for practical handling.
In recent years various types of thin detectors have been prepared from highly sensitive TL phophors,e.g. MgB_4O_7:Dy, $CaSO_4$:Dy, Al_2O_3:C and LiF:Mg,Cu,P, which have all shown good energy and angular responses for measurement of beta doses. The LiF:Mg,Cu,P material also shows a good energy response for exposures to photons with energies above 10 keV and therefore this material is in particular attractive for monitoring in mixed beta/photon radiation fields. Figure 1 shows the energy and angular response to beta ray exposures of two types of thin detectors prepared from the LiF:Mg,Cu,P phosphor [11]. It can be seen that by selecting a suitable thickness of the detector cover it is possible for detector thicknesses below 10 $mgcm^{-2}$ to obtain a satisfactory energy and angular response for exposure to beta radiation with maximum energies of the range 0.23 to 2.27 MeV. In Figure 2 energy response curves for measurement of $H_p(0.07)$ in photon fields are shown for different TL dosemeters with thin detectors for the energy range 6 to 48 keV. The irradiation of the dosemeters was carried out at the Danish National Institute of Hygiene (25,

33 and 48 keV, ISO, Narrow Spectrum Series) at CEA, Fontenay-aux-Roses, France (6.5, 8.5 and 12.2 keV from low-energy X-ray facility) and at Risø (6 keV from ^{55}Fe beam). It can be seen that detectors prepared from LiF:Mg,Cu,P significantly understimate $H_p(0.07)$ doses from photons with energies below 10 keV, in contrast to LiF TLD-700 and MgB_4O_7:Dy detectors showing an overestimation in this energy range. Recently a new type of electronic dosemeter based on the DIS (Direct Ion Storage) principle has been developed by the Rados Technology OY, Finland [12]. This device shows an excellent energy response for measurement of $H_p(0.07)$ from photons with energies down to at least 6 keV.

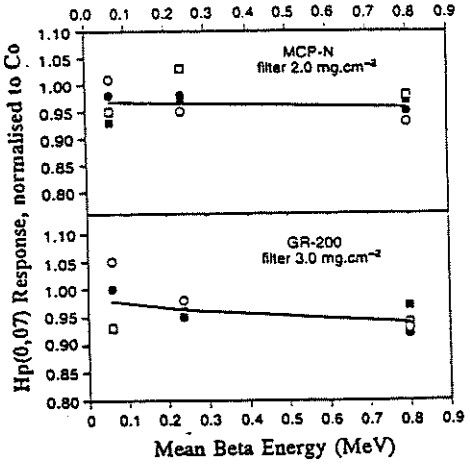

Figure 1. Beta energy and angular response of Polish (MCP-N, 2 $mgcm^{-2}$ cover, 8 $mgcm^{-2}$ thick) and Chinese (GR-200, 3 $mgcm^{-2}$ cover, 5 $mgcm^{-2}$ thick) LiF:Mg,Cu,P TL detectors. (■)0°,(□)20°,(●)40°,(○)60°.

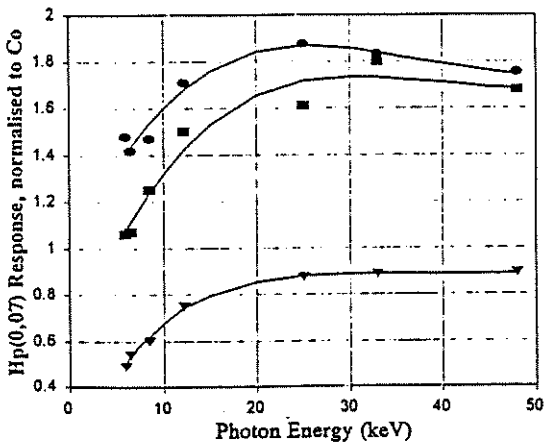

Figure 2. Photon energy response of thin TL detectors covered by 6 $mgcm^{-2}$ aluminised Mylar foil. (▼)LiF:Mg,Cu,P (GR200, 5 $mgcm^{-2}$ thick).(■)LiF,TLD-700 (20 $mgcm^{-2}$ thick). (●)MgB_4O_7:Dy (2 $mgcm^{-2}$ thick).

Monitoring of the Workplace.

Survey instruments used for monitoring in workplaces with mixed beta/gamma radiation should be able to measure $H'(0.07,\alpha^{o})$ for determination of the skin dose equivalent and $H^{*}(10)$ for making an estimate of effective dose. Ionisation chambers are the type of instruments most widely used for measuring beta-ray dose rates. GM counters are not suitable for direct dose rate measurements due to a very strong energy dependent response; however if some knowledge of the energy of the radiation is present, GM detectors may be useful for measurement of low beta-ray dose rates[13]. Ionisation chambers may be well-suited for measurement of skin dose rates of radiation protection levels in beta/photon fields if the measurement is made at some distance from the source, e.g. above about 10 cm. However, many commercial beta/gamma survey meters show responses which significantly depend on energy and incident angle for exposure to beta radiation implying that correction factors must be used [14]. At shorter distances the requirement for a small detector size to obtain satisfactory geometrical conditions implies an unsatisfactory sensitivity for measurement in radiation fields of low dose rates. An interesting small-size ionisation chamber instrument which can be used at very short distances is the DL1 type survey meter with a chamber volume of 15 cm^3 developed at Research Centre Jülich, Germany [15]. Alternative instruments with a higher sensitivity may include scintillation counters with a thin tissue-equivalent detector or the new Rados electronic device mentioned above. More detailed knowledge of beta radiation fields can be obtained by use of a beta spectrometer. In addition to dose they also give possibilities for

identification of the radioisotope as well as estimation of radiation shielding needs. By using two or three Si detectors arranged in a telescope unit it is possible to measure beta spectra in mixed beta/gamma fjelds [16]. The development of a small beta spectrometer-dosemeter unit consisting of three Si detectors is one of the task of a new research project concerning dosimetry of low-penetrating radiation within the EU Radiation Protection 4th Framework Programme 1994-1999.

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Thirty-eight Years of Dosimetry at Studsvik

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Abstract – Personnel dosimetry has been performed since 1958 at Studsvik 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 burn-up fuel testing, radioisotope production for medical and industrial purposes and neutron transmutation doping of silicon are performed. Plans for Boron Neutron Capture Therapy, BNCT, for medical treatment at the R2-0-reactor have recently been discussed. University of Uppsala has a department of neutron research at the reactor. 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.

Radiation doses are normally much smaller than the limit set by the authority. Since 1958 five persons have received more than 50 mSv in a year. Highest whole body dose for a single event was 115 mSv in 1982. These incidents will be described in the paper. The only skin damage, still visible, occurred in 1978 when a worker received more than 10 Sv to his hand.

The limit for workers since 1989, set up by the Swedish Radiation Protection Institute, for the maximum lifetime radiation dose in Sweden is 700 mSv. At Studsvik no one has reached that level. Maximum dose is at the moment 575 mSv for a 62 year old man. Two persons have retired with 503 and 505 mSv respectively. In total 17 persons have received doses above 300 mSv. The natural radiation in Swedish daily life will on average result in a lifetime dose of more than 300 mSv, half of the dose due to indoor radon. Living in dwellings at the upper limit for radon concentration before taking measures, will result in almost 1000 mSv during a lifetime.

Introduction

Personnel dosimetry has been performed since 1958 at Studsvik when the site was built. Before that, Studsvik AB, the former AB Atomenergi established in 1947, was situated in Stockholm. Today more than 20 companies are situated at Studsvik, 6 of them working with radionuclides. Since 1958 only five persons have received more than 50 mSv in a year. Highest whole body dose for a single event was 115 mSv in 1982.

The limit since 1989 for the maximum lifetime radiation dose in Sweden is 700 mSv. At Studsvik no one has reached that level. Maximum individual dose is at the moment 575 mSv for a 62 year old man. Two persons have retired with 503 and 505 mSv respectively. In total only 17 persons are registered for doses above 300 mSv.

Annually radiation doses exceeding 50 mSv will be described in this paper. Of the 5 people having received more than 50 mSv only 3 persons had whole body doses due to a single incident.

Collective doses at Studsvik since 1958 are shown in Figure 1. Collective doses for the staff at both Stockholm and Studsvik (AE) are also presented. The lower limit of registration has changed from 1 mSv per quarter to 0.3 mSv per month but is now 0.1 mSv per month.

Early Dosimetry before the Studsvik Era

Records of dosimetry from the Stockholm period have been kept since 1952. At that time the exposure was measured in röntgen with film dosimeters on the chest and sometimes around the wrist. Also pen type quartz–fibre electrometers were used. Early records only show mean values from 3 months of measurements on a scale 1 to 5, where 1 means 0–30 mR (zero to approximately 0.3 mSv) per week and 5 means above 100 mR (approximately >1 mSv) per week. Urine and blood (lymphocytes) samples were also taken. Between 1953 and 1956 only one person (a neutron physicist) has received a dose in the interval 2–3 R per year. In 1957 only one person received more than 1 mSv (>100 mR) in a week.

Dosimeter Types used at Studsvik

Film dosimeters were used at Studsvik up to 1978. But already in 1974 thermoluminescense dosimeters (TLD) were used when a national dose centre was established at Studsvik with dose records of 18 000 people. Between 1975 and 1987 also the Swedish nuclear power plants joined the national dose centre.

In 1960 the film badge (Du Pont, type 558) was changed each 3 months for non radiological personnel and once every 14 days for others. Pen dosimeters used were Landis & Gyr or Bendix. Also dosimeters for thermal neutrons (Bendix) and foils for high dose neutrons were used. In 1971 the film type was changed to Kodak PM–2 with four filters; plastic, Cd, Cu, and Pb.

From 1974 TLDs have been used at Studsvik and after 1978 no more films were used. LiB_2O_4–powder from Risö in Denmark was first used in the tablets but now three $Li_2B_4O_7$: Mn, Si–chips (from the start one 7 mg/cm^2 and two 300 mg/cm^2) and one LiF chip are in use, delivered by Studsvik Instrument AB. Detection level was first 0.25 mSv. Today monthly doses above 0.1 mSv excluding background are detected.

Up to 1987 all employed at the Studsvik site had to be monitored. Approximately 1000 people had to use TLD. Since 1987 approximately 300 employed and 200 external workers are monitored annually.

Neutrons can also be detected. Since 1993 Neutrak 144 is used for fast neutrons. The two LiB chips in the standard TLD holder are used for thermal neutrons after correction for gamma dose.

Personnel Annual Radiation Doses Exceeding 50 mSv

In 1967 one person received a total dose of 56.6 mSv working at the Isotope Laboratory. The dose was evenly distributed over the year. The collective dose in general had increased rapidly at the R2 and Isotope lab between 1966 and 1970. The dose changed from 140 to 700 mpersonSv for the staff at the R2–reactor and from 130 to 620 mpersonSv for the staff at the laboratory. This was mainly due to a larger production of radioisotopes, changing of in–pile loops, isotopes stuck in the transportation tubes and a reduction of radiation protection officers from 19 to 10 people. In 1969 the R2 reactor increased the thermal effect from 30 to 50 MW.

Tubes are used to transport radionuclide sources between the R2 reactor and the reception cell after irradiation. The collective dose after one serious stuck in the tubes in 1971 with ^{60}Co was 170 mpersonSv. When repairing the tubes in the tunnel under the cell after a malfunction, one of the workers did not know that the cell contained a ^{192}Ir source. The worker received 70 mSv. Measurements after the incident showed that the dose rate had been almost 1 Sv per hour. During the whole year the total dose was 84.2 mSv for this person. Another person at the Isotope laboratory also exceeded the limit 50 mSv in that year. The reason for that was due to the high working load at the time and problems with several stuck sources, often ^{60}Co, in the tubes transporting radioactive materials from the reactor. The monthly maximum was 16 mSv and the total annual dose was 50.5 mSv.

In 1982 a control of a valve in the D_2O ion exchanger at the R2 reactor caused the highest personnel dose ever at one time in Studsvik, 113.5 mSv. The surface dose rate of the ion exchanger was approximately 1 Sv per hour and less than 30 mSv per hour at a distance of 0.5 m. In other parts of the room the dose rate was 10 mSv per hour. The person involved had two TLDs and a pen dosimeter but did never observe the sign at the entrance telling the dose rate was above 1 Sv per hour. The work took only 5 to 10 minutes but the valves were difficult to reach. An electrician changed a light bulb in the same area before the work with the valve started. His dose was only 0.5 mSv after 3 minutes.

Staff from the university of Uppsala also use the neutron beams from the two reactors. In 1995 one person equipped with TLD but without an electronic alarm dosimeter, reached 49.7 mSv. It was during a half hour work adjusting a lead collimator with, what was believed, non radioactive ions the dose was received. Earlier work that year resulted in a total dose of 72.5 mSv for the person. The worker was not allowed to do more radiological work for the rest of that year. After the accident the department is now better equipped with electronic detectors.

Hand Radiation Doses Exceeding 500 mSv

The R2 reactor was used by AB Kabi Diagnostica in 1978 to make ^{32}P for medical use. After an accident with an oven for destillation inside a cell, decontamination started. The dose rate from gamma radiation was only 1 mSv per hour but the beta dose rate was above 1 Sv per hour. Using manipulators from the outside, the cell was cleaned and instruments were removed. Unfortunately some work had to be done by hand. When finishing the work only a small contamination could be observed to one of the involved workers. The day after this, the right hand became swollen and a red spot could be seen. The dosimeter of the hand showed 2.9 Sv, but after an medical examination the dose was estimated to approximately 5–7 times more.

Summary

Radiation doses at Studsvik are low. Average dose for all holders of a TLD is close to 1 mSv annually. In spite of several incidents with radioactive materials no serious radiation doses have occurred at Studsvik. Only 5 incidents have resulted in annually whole body doses exceeding 50 mSv.

Studsvik has a long tradition in radiation protection and a good knowledge of dosimetry. Yearly calibrations of our detector system show excellent agreements with the secondary standards at the Swedish Radiation Protection Institute. Plans for the first Swedish accreditation of a dosimetry laboratory will make the system even more interesting for the customers both at our own companies and outside Studsvik.

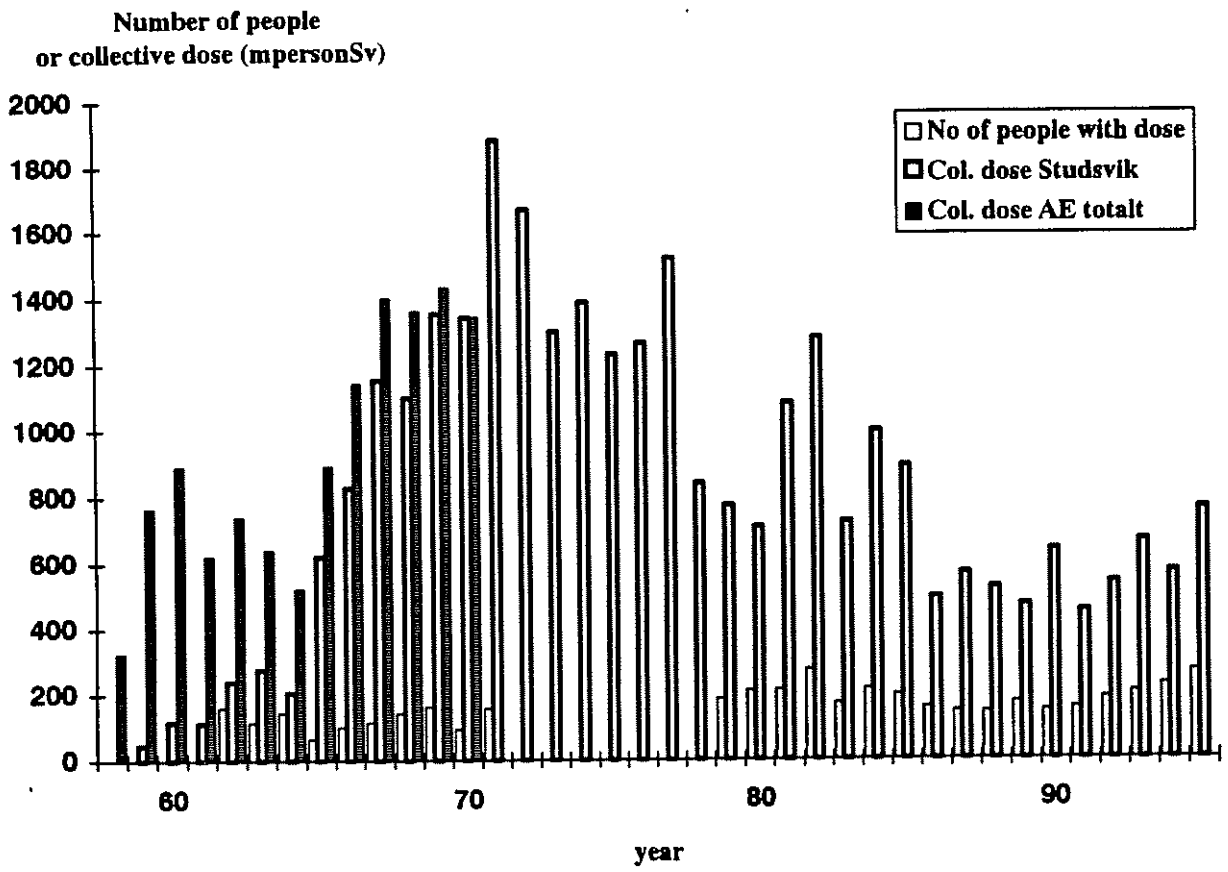

Figure 1. Collective doses at Studsvik from 1958 to 1995. Since the 1970s the collective dose in Studsvik has decreased and is from 1986 approximately 0.5–0.7 personSv per year or close to 1 mSv per person as an average for all TLD-holders. AE includes the staff at both Stockholm and Studsvik.

Improving quality control and assessment of systems for direct measurement of radionuclides in people

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Abstract - Intercalibration and intercomparison measurements with different kinds of sample materials have been done since many years at both national and international level. For direct measurements of man the possibilities for this type of quality control have been limited. Scientists visiting laboratories in other countries have been whole-body counted abroad as well as at home for intercomparison purposes.
In the Nordic countries two projects within the framework of NKA and NKS programs have been performed. The Commission of the European Communities (CEC) produced instructions on the calibration of whole-body counters and the interpretation of the measured results in 1994. This work was followed by an intercomparison project started in 1995. After Finland and Sweden joined the European Union all the Nordic countries with whole-body counting systems could participate. The phantom that is used is produced in St.Petersburg, Russia. It will be measured in about 40 different European whole-body counting laboratories. For international intercomparison a phantom representing a 4-year child was made in Canada some years ago. Many European including some Nordic laboratories could participate in this intercomparison exercise. In both these exercises the nuclide composition and activity concentrations were unknown. The results show that the quality level in most laboratories is good. In these international projects only a few laboratories from each country could participate. Nordic projects are needed to give the possibility for all Nordic laboratories to take part.

Introduction

With direct measurements of radionuclides in people we understand mainly the measurement of the total body called the whole-body counting. Also organs or spesific tissues can be measured directly. The efficiency calibration of such measurement systems is much more complicated than the corresponding calibration for sample measurements. Samples can be homogenized, reduced in volume by drying or ashing and put into containers with standard geometries. That is not possible for people. You have to accept different sizes, shapes, distributions of radionuclides etc. For calibration you need to create a 'model' usually called a phantom, resembling the person you want to measure in all aspects. A phantom is commonly made of plastic bottles filled with radionuclide solution. A more tedious method is using point sources. Earlier whole-body counting systems were also calibrated using people to whom appropriate amounts of radionuclides were administered beforehand. This method is more or less abandoned because of radiation protection considerations.

During outage periods at nuclear power plants also radiation workers from foreign countries are whole-body counted for determination of body burdens of contamination radionuclides. The results of these measurements are used for calculation of internal radiation doses. For surveillance purposes the doses are reported to the dose register in the worker's home country. It is essential that the quality assurance in all whole-body counting laboratories is good in order to have reliable and comparable doses.

Quality assurance

With quality we understand the totality of features and characteristics of a product or service that bear on its ability to satisfy started or implied needs (SFS-ISO 8402). Quality assurance includes all those planned and systematic actions necessary to provide adequate confidence that a product or service will satisfy given requirements for quality. A proper quality assurance program includes both quality control and quality assessment (ISO 9000).

The quality control of a program for internal dosimetry using direct methods of measurements can be designed in accordance with corresponding programs for gammaspectrometric sample measurements. The main difference and difficulty is the effieciency calibration of the detection system. The quality control also includes specially designed procedures and protocols for the internal dosimetry program as well as systems for data recording and record keeping. Quality assessment includes audits and accreditation as well as intercomparisons.

Intercalibrations and intercomparisons

In the Nordic as well as in other countries many whole-body counting laboratories have poor possibilities to use real phantoms for regulary checking their effieciency calibrations. Phantoms are expensive and those filled with liquid solutions easily start leaking and create a contamination risk. Within the framework of NKA an intercomparison program for whole-body counting laboratories in the Nordic countries was designed and the measurements done 1984-1985 just before the accident at Chernobyl. This project was followed up by an intercalibration program within the NKS in 1990 to 1993 in which 20 Nordic laboratories participated. The results of both projects were good and were published in 1994 (1). The phantom of modular type was constructed using flat plastic bottles with a capacity of 550 ml Fig. 1 (1). The bottles were filled with calibrated solutions of Co-60, Cs-134 and Cs-137.

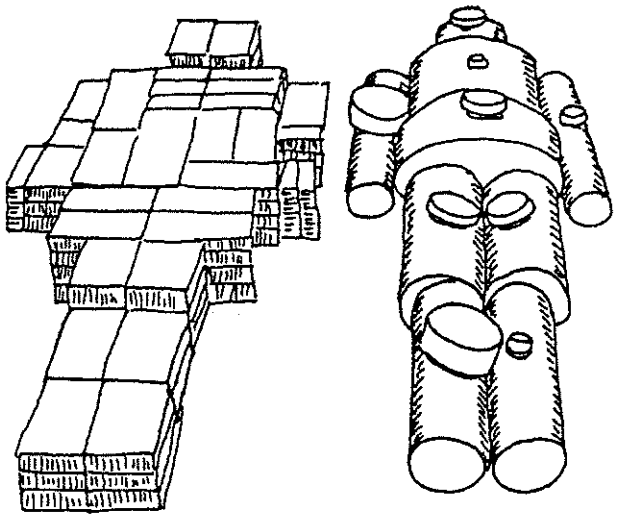

Figure 1. The phantom to the right is the commercially available, often referred to as Bush (2) or Bomab phantom and to the left a phantom assembled with ordinary rectangular shaped plastic bottles. Both phantoms are filled with liquid solutions.

In 1994 guidance notes on the calibration of whole-body counters was published by the European Comission (3) followed in 1996 by an IAEA Safety Series publication on direct methods for measuring radionuclides in the human body (4).

Lately different types of phantoms of solid material have been developed. This makes them much easier to transport. One of these was the phantom simulating a four-year-old child prepared in collaboration by the Canadian National Calibration Reference Centre for In-Vivo Monitoring and the United States Department of Energy (DOE) (5). Another is the phantom

developed by the Research Institute for Industrial and Sea Hygiene and produced by the 'Scientific and Technical Centre -Protection Ltd.' both in St. Petersburg, Russia (6). The phantom consists of 1 kg and 0,5 kg plastic bricks. The bricks are held together with aluminium connection pieces. Different sized phantoms are easy to build. The radionuclides are introduced into the phantom with rods containing known amounts of radionuclides. The rods are inserted into holes drilled through the lenght of each brick. The advantage with this phantom is that it is possible to use rods with different radionuclides either one at a time or a combination of radionuclides as for example K-40, Cs-134 and Cs-137. This shortens the overall measurement times. If the phantom is to be transported from one laboratory to another the major part of the phantom is non-radioactive. The radioactive rods are easily transported in special small containers. After the Chernobyl accident the lack of calibration factors for children was clearly noted. This resulted in a growing interest for flexible phantoms that can be used in different sizes.

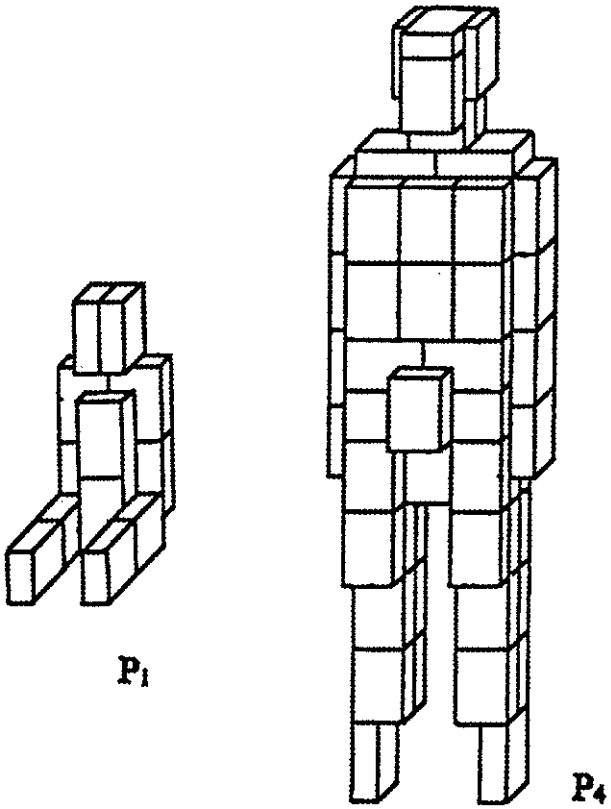

Figure 2. The Russian made phantom consists of bricks of tissue equivalent material. This phantom can be used in sizes from small children (P1, sitting) to adult men (P4, standing) and can be assembled in different positions.

The latest method not yet in routine use is the computational calibration (7). The availability of computers makes this an attractive way to calibrate in the future. This computational method, however, must be controlled by measuring realistic phantoms.

Conclusions

The different types of phantoms now available makes it possible to do better quality assessment than before. Radiation workers move from one country to another and the intercomparison of systems, used for whole-body counting in different laboratories is essential if the radiation doses calculated based on whole-body counting measurements are to be

comparable. The next Nordic project in the field of internal radiation protection could well be focused on dose calculations.

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Installation of two whole-body counters in new laboratory premises

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Abstract: Two whole-body counters with massive background shielding are in use in Finland. Both installations were moved in 1994 to new laboratory premises, planned for low-activity measurements. Both steel rooms were also enlarged, using old steel, to improve measuring geometry.

Concrete constructions adjacent to the steel rooms were made using olivine as the ballast material. Gamma-spectrometric measurements of special concrete showed that the activity concentrations of both potassium-40 and of the uranium and thorium decay series were approximately one eighth of those in normal concrete.

Radon isotopes and their decay products are especially troublesome in measuring rooms for low activities. Radon concentrations vary with the season and with the weather, giving rise to unpredictable variations in the background level. The whole-body counting laboratories were constructed with double walls, floors and ceilings. Fresh filtered air is supplied to the measuring rooms, and the exhaust flushes the space between the double walls.

Both steel rooms were also lengthened. In order to be sure that the steel contained no ^{60}Co, old steel was jointly obtained from a scrapped gasometer, built before 1930.

Introduction

'Whole-body counting' is a colloquial term for the measurements of X- and gamma-radiation emitted from radionuclides in the human body. A typical whole-body counting facility consists of shielding, detectors and analyzers. In Finland there are two systems with massive steel background shielding. The first is operated by the Finnish Centre for Radiation and Nuclear Safety (STUK), the Finnish national radiation protection authority. The other system belongs to the Laboratory of Radiochemistry (LoR), Department of Chemistry, University of Helsinki (formerly Department of Radiochemistry). Both institutions moved to new laboratory premises in 1994. It was possible to design the whole-body counting environment for the two systems. Within the overall purpose of reducing the gamma-spectrometric background there are four separate design targets:

- to reduce the activity concentrations (of ^{40}K and the uranium and thorium decay series) of all construction materials adjacent to the steel rooms;
- to reduce the concentrations of radon isotopes and their decay products in the air entering the steel room;
- to reduce the danger of contamination in an acute fallout situation, and
- to minimize the influence of radiation sources, as it was not possible to place the whole-body counters in buildings dedicated to low-activity measurement.

Construction materials

In a modern building concrete is the material most likely to affect background pulse rates, since concrete is used in large quantities. Finnish concrete may contain quite high amounts of uranium, thorium and potassium, because the ballast gravel is mostly of granitic origin. Mustonen[1] reports mean activity concentrations in ready concrete from the Helsinki area: ^{226}Ra 53, ^{232}Th 38 and ^{40}K 838 Bq kg^{-1}. In the surrounding Uusimaa province the mean activity concentrations in concrete were 40, 39 and 840 Bq kg^{-1}, respectively[2]. Erlenbach[3] reports total gamma activity concentrations in different building and shielding materials, which he checked before moving a whole-body counter to a new location. The activity concentration of sand, for instance, varied from 5 to 100 Bq kg^{-1}, which shows how necessary it is to check building materials.

A special concrete was selected for all construction details adjacent to the whole-body counters. It was used both for pouring on site and for structural elements manufactured elsewhere. In this

special mixture crushed olivine was used instead of the granitic gravel ballast of normal concrete. For the LoR building white cement was also substituted for standard cement. Samples of the olivine concrete intended for the LoR low-activity area and of normal concrete were measured with a semiconductor detector. The results are given in Table 1.

Table 1. Gamma photopeak intensities of special and normal concrete

Nuclide	Net photopeak pulse rate $s^{-1} \bullet 10^3$		Improvement
	Special concrete	Normal concrete	$R_{normal}/R_{special}$
Ra-226	1.45	15.8	11
Pb-212	8.0	73	9.1
Pb-214	6.1	62	10
Bi-214	5.3	50	9.4
Cs-137	0.14	1.3	9.3
K-40	13.6	111	8.2
Tl-208	1.4	11.5	8.2

Note that no efficiency calibration was available for this type of sample; the reported values are net photopeak pulse rates for the most intensive lines of the listed nuclides. The results for the special concrete are means of three samples; only one sample of normal concrete was available. The activity concentrations in olivine concrete are one tenth to one eighth of those in normal concrete.

Radon and radon decay products

Radon isotopes originating from the uranium and thorium decay series present both in building materials and in the ground beneath buildings, are exhaled into the indoor air. Gamma emitting decay products of radon isotopes contribute to the background, and are especially troublesome for low-activity measurement since their contribution varies with the seasons, climatic conditions and the rate of air exchange. Sugiura et al.[4] have studied the influence of radon on the background in whole-body counting.

In the installations described here the main countermeasure against radon was the equipping of low-activity measuring rooms with double walls and ceilings. The exhaust air is taken from the space between the walls, and is replaced from the measuring room. Fresh air, taken from a height of 25 metres above ground, is supplied to the measuring room through a HEPA filter. Radon isotopes and their decay products are thus flushed from the wall space.

The STUK installation is situated on the first floor, and radon exhaled from the ground below the building is flushed away by the ventilation in the basement. The LoR low-activity counting room is placed in the basement, and as an additional measure against radon a double floor was constructed here. Both floors were poured with the special olivine concrete, and an additional radon barrier of galvanized sheet steel was also incorporated in the construction.

Other background sources

It was not possible to place the whole-body counters in buildings dedicated to low-activity counting. In order to decrease the influence of cosmic radiation the measuring rooms could be underground, but that was not possible for these installations. The next best solution was to place the counters in a low-activity environment as described above, and to make sure that sources of penetrative radiation are located as far as possible from the low-activity areas.

Sources of electromagnetic fields, which might affect the detectors and induce noise, should also be avoided.

Another hazard is that radioactivity from open sources used elsewhere in the building may be transported to the low-activity area. A related risk is contamination in a fallout situation, which was a real problem after the Chernobyl disaster. Countermeasures against contamination include the slight overpressurizing of low-activity rooms, air filters, an airlock in the entry passage and provision of showers and changing rooms. The schematic layout of the whole-body counting facility at STUK, showing these features, is presented in Figure 1. The LoR low-activity room also has an airlock, but not the changing and shower rooms of the STUK facility.

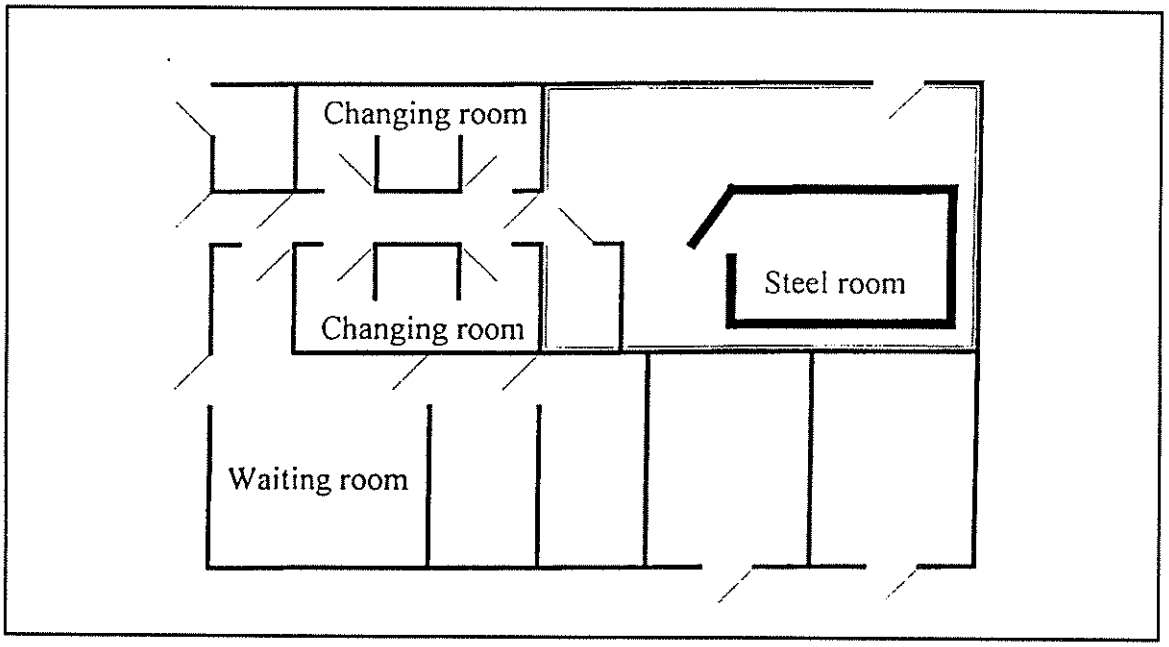

Figure 1. The schematic layout of the whole-body counting installation at the Finnish Centre for Radiation and Nuclear Safety (STUK) showing the steel room for monitoring, two changing room lines with showers and the waiting room. The straight passage to the low-activity area can be closed if contamination is suspected. The dimensions of the steel room are given in Table 2.

Steel rooms

The steel rooms are made from old naval steel. Steel manufactured after 1945 may contain fallout radionuclides and also ^{60}Co, used to monitor the condition of blast furnaces. Data for the steel rooms is given in Table 2.

Since moving the steel rooms actually meant dismantling them into movable pieces and then reconstructing them in the new locations the opportunity was taken to lengthen the rooms. In the STUK installation the geometry is of the scanning-bed type. Before moving, the detector ring travelled along the stationary subject to be measured[5]. With a longer steel room the detectors can remain stationary while the subject bed moves.

In the LoR installation it was only possible to use the chair geometry before moving, but now bed geometry is also possible. The measuring chair moves on rails out of the steel room, which makes entry and exit easy for the subjects, and loading of phantoms much easier than the previous arrangement. The chair is surrounded by 40 millimetres of extra lead shielding.

During re-erection of the steel rooms precautions against introducing radioactive substances were taken. The steel components were scrupulously cleaned from dust etc. All materials intended for the steel rooms were checked for gamma-activities. Since ordinary welding rods usually contain thorium compounds, MIG welding was employed instead of normal arc welding.

Table 2. Dimensions of the steel rooms.

	STUK installation	LoR installation
Inside dimensions, metres		
Length (before moving)	2.45	1.8
Length (after moving)	4.1	2.1
Width	2.4	1.1
Height	2.3	1.8
Thickness of shielding, millimetres		
Steel	150	150
Lead	3	3
Copper	2	1

Results

In the LoR facility three gamma spectra were obtained with the same high purity germanium (HPGe) detector (EG&G Ortec GEM-25195-P-S, relative efficiency 27%): inside the steel room, inside the low-activity room, i.e. adjacent to the steel room, and in an ordinary counting room next to the low-activity area. In the STUK installation similar measurements were made inside the steel room and immediately outside it, i.e. in the low-activity area. The detector was, in this case, a Gamma-X GMX-25200-S HPGe detector, of relative efficiency 27%. The results are presented in Table 3, showing the actual photopeak counting rates for the gamma peaks detected inside steel rooms and the corresponding rates outside the room.

Table 3. Pulse rates for gamma photopeaks observed in the steel rooms.

Nuclide	Energy, keV	Photopeak pulse rates, pulses per 1000 seconds				
		A	B	C	D	E
Ra-226	185.9	5.65	62	77	8.74	68
Pb-212	238.6	3.27	288	537	5.05	258
Pb-214	241.9	3.59	67	92	3.08	99
	295.2	8.77	168	233	2.59	167
	351.9	13.75	302	423	6.83	298
Bi-214	609.3	9.68	305	455	5.44	296
	1120.3	1.81	90	133	1.32	82
	1764.5	1.74	88	126	0.81	81
Tl-208	2614.5	1.53	110	258	not measured	
K-40	1460.8	5.48	570	1967	2.7	645

A Pulse rates inside the LoR steel room
B Pulse rates in the LoR low-activity room, outside the steel room
C Pulse rates in the LoR ordinary counting room
D Pulse rates in the STUK steel room
E Pulse rates in the STUK low-activity area, outside the steel room

The results in Table 3 show that the lowest backgrounds have been achieved in the STUK installation. This is possibly because it is located on the first floor, further away from the granitic bedrock than the LoR counter. It must also be pointed out that the lead-copper shield grading and the interior lead shielding were not yet installed in the LoR steel room when the measurements were taken.

By comparing columns A and B or D and E the reduction in gamma-ray intensity can be calculated. This is not simply a measure of attenuation in the shielding, since radon decay products are transported into the steel room and decay there. For this reason the reduction factor is not energy dependent. By comparing columns B and C one can see the effect of using selected building materials. This effect is especially dramatic for potassium.

A comparison of background pulse rates in the ^{137}Cs and ^{40}K energy bands, using NaI(Tl) scintillation crystals, is presented in Table 4. The STUK installation has four detectors, each 127 mm in diameter and 102 mm thick, while the LoR counter has one detector, 204 mm in diameter and 102 mm thick.

Table 4. Comparison of NaI(Tl) backgrounds before and after moving.

	Cs-137	K-40
STUK		
Energy band, keV	585-735	1335-1585
Before moving, min^{-1}	170	88
After moving, min^{-1}	130	60
LoR		
Energy band, keV	600-720	1360-1650
Before moving[5], min^{-1}	71	48
After moving, min^{-1}	72	43

The background counting rates show marked improvement for the STUK installation, but not for the LoR counter, which was previously situated in a building erected in 1837. Apparently this old building was constructed from materials with a low level of natural radioactivity.

References

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3. Erlenbach, H.R., *Investigations into moderate-cost shielding improvements to a standard steel room using an approximate scale model.* Assessment of radioactivity in man, 153. IAEA, Vienna, 1972
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Urinary Excretion of Cesium-137 for Estimation of Body Content and Internal Absorbed Dose

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Abstract

Since 1987 we have performed regular whole body measurements on a group of about fifteen individuals living in the Malmö-Lund region in south-west of Sweden. At the same time collection of 24 hour urine has been performed on most of these persons. The aim of this study was to determine the ratio between the daily excretion in urine and the whole body content of cesium-137 for a south-Swedish population and later on use this ratio to estimate the Cs-137 whole body burden in other population groups.

Assuming that 80 % of the Cs-excretion takes part through urine, the ratio obtained yielded an equivalent biological half-time of 95 ± 4 d (range, 67 - 121 d) for females and 109 ± 4 d (range, 89 - 133 d) for males which is within the ranges established in earlier work. For the potassium excretion we assumed a fraction of 85 % passing via the urine and the corresponding values, 32 ± 2 d for females and 40 ± 2 d for males, agreed well with ICRP 23 ranges for the reference man, however large spread within the study group was found.

Taking the ratio between the potassium normalised body burden of Cs-137 and the potassium normalised daily Cs-137 urinary excretion we obtained a value of 3.6 ± 1.3 (averaged over both sexes). Applying this ratio from the Lund-Malmö group on urine samples from a group of adults from the Briansk area, Russia, we found that in 1995 the estimated average whole body content of Cs-137 was about 19 kBq (range, 4 - 60 kBq), which would result in an effective dose rate of 0.67 mSv y^{-1} per individual. The corresponding figures for 1994 are 26 kBq (range, 5 - 72 kBq) and 0.90 mSv y^{-1} respectively.

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Whole-body Measurements at a Research Facility

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Abstract

Whole-body measurements were performed 3 times a year for a 3-year period on a group of 20 workers at Risø National Laboratory.

One of the aims of the project was to establish the "background" content of radionuclides in persons working at Risø. Persons working at the different nuclear facilities as well as persons working in non-nuclear departments were included in the group.

Another purpose was to "check" the normal procedure for detecting internal contamination, namely measurements on urine samples. One of the whole-body measurements revealed an incidence with uptake of ^{131}I. In this case the normal procedure of giving a urine sample for measurement after an operation, which turned out to be dubious, had not been followed.

Apart from this incidental uptake only small amounts of activity from radionuclides such as ^{60}Co, ^{65}Zn, etc. were detected in the persons working at the nuclear facilities.

1 Introduction

At Risø National Laboratory a fifth of the workforce is classified as radiation workers. They work at the nuclear facilities: two research reactors, a main isotope laboratory, a waste treatment plant and at laboratories using radionuclides for research purposes.

The work involving radioactivity and ionising radiation is regulated by law plus a set of local rules and is supervised by the section of applied health physics which also participate in the work planning.

Areas used for working with radioactivity and ionising radiation are classified and monitored for radiation and contamination both by routine measurements and by measurements during operations which can lead to elevated levels of radiation and/or contamination.

It is a general aim of the radiation protection of the workers, that committed effective doses from intakes of radionuclides should be negligible and preferably avoided. This however is not always possible for intakes of tritium which is present in the air in the containment of the heavy water moderated DR 3 reactor.

Measurements of the beta-activity in urine samples are used to determine the intakes of radionuclides. Monthly or quarterly samples are collected from persons working in areas where

contamination is possible. Special urine samples must be delivered some hours after doing work where an intake is known to have taken place or is a possibility.

The whole-body counter at Risø has been used to measure the content of radionuclides in a selected group of workers primarily among the radiation workers. This has given information about intakes of γ-emitting radionuclides as a result of working conditions, about "normal levels" and has acted as a simple check of the normal procedure for detecting internal contamination.

2 Selected persons

A total of 20 persons was selected for whole-body measurements. Sixteen of the persons are classified as radiation workers and 3 are non-radiation workers. A girl of age 13 who had participated in former whole-body measurements was also included. Table 1 lists the different work categories present in the selected group and the number of persons in each category.

The group has been followed at 4 month intervals over a period of 3 years, in order to have a dynamic picture of the contents of radionuclides.

Table 1. Measured group divided into work categories.

Category of work	Number of persons
Reactor DR 3 (staff)	4
Reactor DR 3 health physics technicians	2
Main isotope laboratory	2
Waste treatment plant	4
Reactor DR 1	1
Risø health physicists	3
Risø non-radiation workers	3
Non Risø	1
total	20

The selected group does not represent all kinds of "radiation work" at Risø.

3 Measurements and determination of activity

The whole-body counter is a shielded underground room (old navy gun tower) equipped with a high purity 52% Ge-detector. The detector is suspended in a frame construction and can be positioned over the person to be measured. The person is placed in a chair in the middle of the room. The room is ventilated by a filtered air-stream. The set-up is shown in figure 1. The wide angle picture shows the entrance (left) and the counting room (right).

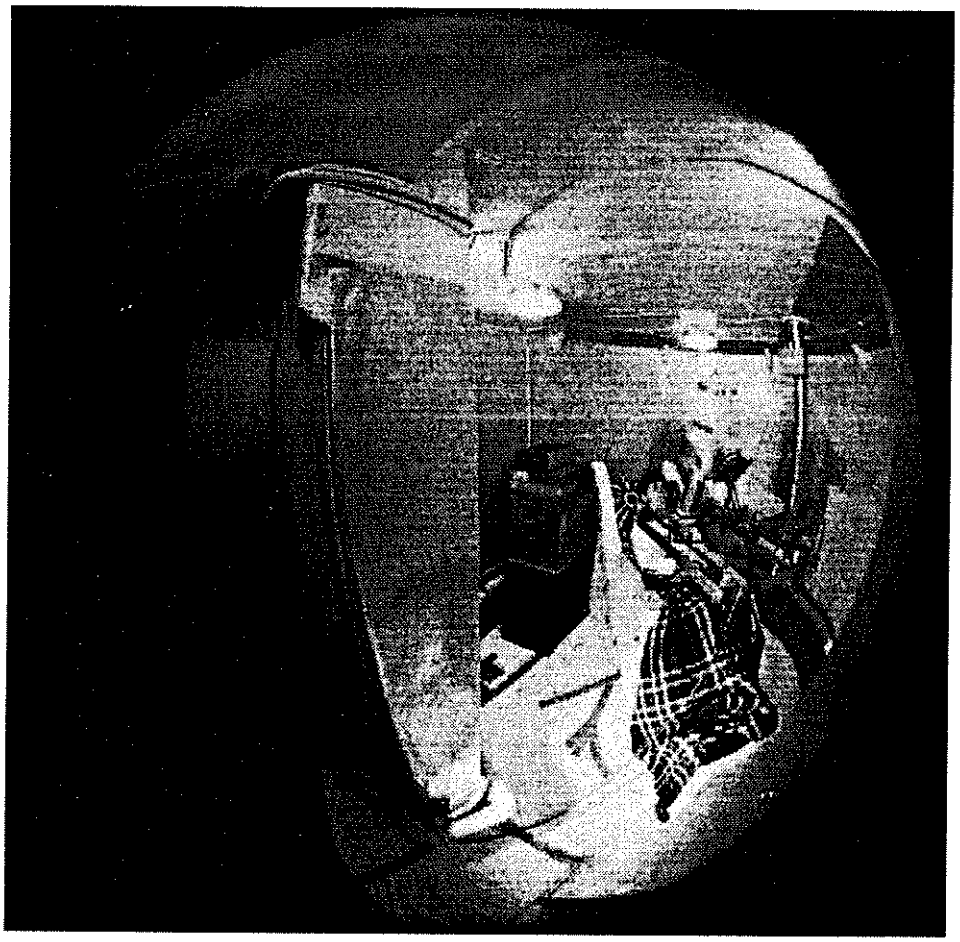


Figure 1: The Risø whole-body counter.

A standard measuring time of one hour is used. Persons known to have been in a contaminated area prior to a measurement are asked to wash and change to "clean clothes" before they are measured. This should reduce possible external contamination, but from successive measurements it was clear, that external contamination could not totally be avoided.

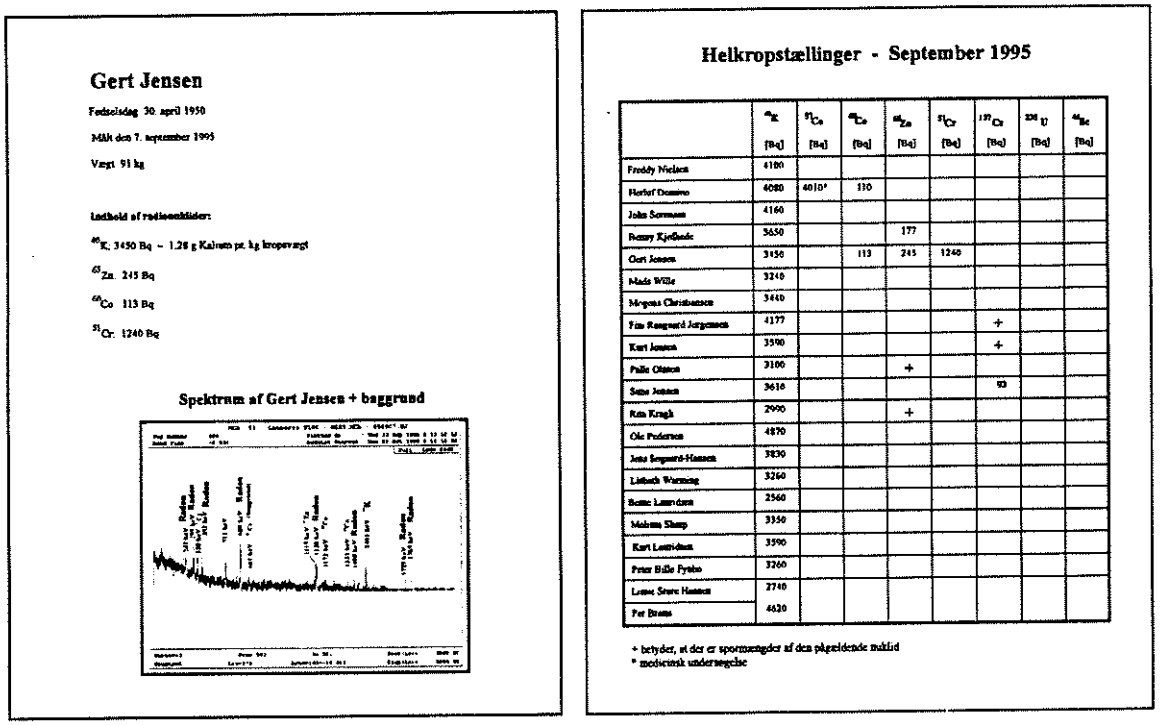
The detector has been calibrated using a 25 litre trunk with an ^{152}Eu-solution.

The minimum detectable activity is for ^{137}Cs of the order of 50 Bq.

Each measurement generates a 4096 channel spectrum covering γ-energies from 80 to

2000 keV (higher energies can be measured if needed). The gamma-spectra are analysed with standard Canberra software, which also include subtraction of a background spectrum.

After each series of measurements, each person gets the results from the measurements. One sheet of paper contains the content of radionuclides in the person together with a drawing of the γ-spectrum (person + background), and a sheet of paper with the body content of radionuclides for the whole group. In figure 2 an example of the two sheets is shown (in Danish).



Gert Jensen

Fødselsdag 30. april 1950

Målt den 7. september 1995

Vægt 91 kg

Indhold af radionuklider:

^{40}K, 3450 Bq ~ 1.28 g Kalium pr. kg kropsvægt

^{65}Zn 245 Bq

^{60}Co 113 Bq

^{51}Cr 1240 Bq

Spektrum af Gert Jensen + baggrund

Helkropstællinger - September 1995

	^{40}K [Bq]	^{57}Co [Bq]	^{60}Co [Bq]	^{65}Zn [Bq]	^{51}Cr [Bq]	^{137}Cs [Bq]	^{238}U [Bq]	^{40}K [Bq]
Freddy Nielsen	4180							
Herluf Dommo	4080	4010*	110					
John Sørensen	4160							
Benny Kjellende	3650			177				
Gert Jensen	3450		113	245	1240			
Mads Wille	3240							
Mogens Christiansen	3440							
Finn Rosgaard Jørgensen	4177					+		
Kurt Jensen	3500					+		
Palle Olsen	3100			+				
Sune Jensen	3610					93		
Ken Krogh	2990			+				
Ole Pedersen	4870							
Jens Sørgaard-Hansen	3830							
Lisbeth Warming	3260							
Bente Lauridsen	2560							
Malene Sharp	3350							
Kurt Lestrøhm	3590							
Peter Bille Pyrko	3260							
Lasse Store Hansen	2740							
Per Brams	4620							

+ betyder, at der er spormængder af den pågældende nuklid
* medicinsk undersøgelse

Figure 2. An example of the results of the measurements (in Danish).

4 Results and discussion

The measurements show that small amounts of radioactivity can be found in DR 3 staff, health physics technicians working at DR 3, workers at the main isotope laboratory and workers at the waste treatment plant.

Table 2 shows the detected radionuclides (^{40}K not included), the number of persons in which a nuclide was found, and the maximum contents measured in the 3-year period. For each person the average content of each of the found nuclides was calculated, and the maximum values of these are also shown in table 2. The uncertainty on the contents is estimated to be less than 30%.

Table. 2. Detected radionuclides, number of persons in which a radionuclide was found, the maximum measured individual body contents and the maximum average of the measured individual body contents in the measuring period.

Radionuclide	Number of persons	Maximum content [Bq]	Maximum average content [Bq]
^{51}Cr	2	2440	1010
^{60}Co	8	680	305
^{65}Zn	6	1890	405
^{124}Sb	1	110*	–
^{131}I	1	1850*	–
^{137}Cs	8	219	67
^{203}Hg	1	625*	–

* Radionuclides detected only once.

Except for the ^{131}I, all the detected radionuclides could be directly related to known sources at the workplace. Part of the measured contents of ^{137}Cs is of course suspected to be of Chernobyl origin. The measured maximum contents are believed to be an upper limit for the "normal levels" of the body content in persons working in the same work categories. In the case of ^{131}I, further investigation was initiated.

The content of ^{131}I was found in a health physics technician. An investigation concluded, that the intake must have occurred during a test of a filter using an ^{131}I compound. An injection apparatus was later found to be defect allowing ^{131}I to be flushed back at the operator and the bystanders. The operator was consequently measured, a content of ^{131}I of 184 Bq was determined 69 days after the filter test and a committed effective dose of 8 mSv was assigned. The committed effective dose to the health physics technicians was determined to be 0.3 mSv. If the operator had followed the normal procedure and had delivered a urine sample after the operation, the intakes would have been detected shortly after the operation.

Based on the measured activity contents in the whole measuring period, crude estimates of the maximum intakes in a year for the detected nuclides were made. In the estimations intakes by inhalation of particles with a size of 1 to 10 microns particles were assumed. For a given measured body content (from the second measurement and onwards)

time of intake was taken to be just after the previous measurement, to maximize the calculated intake. Estimates for ^{131}I and ^{203}Hg were not made, as intakes from these nuclides were not considered to be "chronic". The committed effective doses from the estimated intakes were calculated using the most conservative dose factors from reference 1. The estimated maximum yearly intakes and resulting committed effective doses are given in table 3.

Table 3. Crude estimates of maximum yearly intakes of radionuclides and corresponding doses.

Radionuclide	Maximum possible yearly intake [Bq]	Maximum possible committed effective dose [μSv/y]
^{51}Cr	$3 \cdot 10^5$	10
^{60}Co	$2 \cdot 10^3$	60
^{65}Zn	10^3	3
^{124}Sb	$5 \cdot 10^3$	30
^{137}Cs	$5 \cdot 10^2$	4

The calculated maximum intakes and thus all the measured body contents are in agreement with the general aim of trying to avoid internal doses. They are also consistent with the (almost always) "negative" results of the measurements on urine samples, as they give rise to urine concentrations below the lower limit of detection, which is equivalent to approximately 50 Bq/l urine, when standard procedures are used.

References

Ref.1., ICRP 68. Dose Coefficients for Intakes of Radionuclides by Workers.

Studier av ^{137}Cs i befolkningsgrupper bosatta i Norrland, en uppföljning.

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**FOA, S-901 82 Umeå, Sverige

Abstrakt - Efter Tjernobylolyckan 1986 har upptaget av ^{137}Cs studerats i ett flertal olika befolkningsgrupper i norra Sverige.
Under 1988 - 1996 har undersökningar av ^{137}Cs -halten genomförts genom mätningar av psoasmuskel. Ur denna studie har vi beräknat en halveringstid för ^{137}Cs i populationen på 5.0 år. Ur dessa mätningar kan man dock se en längre halveringstid vid de senare mätningarna jämfört med de tidigare.
Helkroppsmätningar har även utförts på olika befolkningsgrupper från samma region. Mätningarna startade 1986 med en grupp samer i vilka ^{137}Cs -halten därefter regelbundet har kontrollerats. År 1991 startades en studie baserad på ett slumpmässigt urval av den allmänna befolkningen respektive samer boende i tre områden i norra Sverige med olika markdeposition av ^{137}Cs. Resultatet av de tidigare helkroppsmätningarna har bland annat visat på en halveringstid av ^{137}Cs hos samerna på cirka 5 år. Från år 1991 till 1992 fann vi dock en ökning av ^{137}Cs -halten i ett flertal av de studerade grupperna. Mätningarna 1996 visar dock på en fortsatt nedgång i ^{137}Cs -halten för samtliga grupper. Resultaten från denna studie visar på halveringstider som ligger mellan 4 och 12 år i de olika populationerna.

Inledning

Mätningarna som detta arbete bygger på är utförda på befolkningsgrupper bosatta i de fyra nordligaste länen i Sverige: Jämtland, Västernorrland, Västerbotten och Norrbotten. Dessa län upptar ca 55% av Sveriges yta men i denna region bor endast 10% av landets befolkning. Stora delar av de län som omfattas av undersökningen är glest befolkade och en stor del av födan tas från skogsekosystemen i form av viltkött, bär och svamp m.m. I regionen bedrivs även renskötsel och i den samiska kulturen är av förklarliga skäl produkter från renen, främst kött, en mycket stor del i kosthållningen. De näringskedjor som börjar i skogsekosystemen och som slutar vid människan är i många fall effektiva transportvägar av cesium under en längre tid efter ett nedfall och i vissa ekosystem har den effektiva halveringstiden för cesium visat sig vara i stort sett lika med den fysikaliska halveringstiden [ref. 1]. En av de näringskedjor, av betydelse för människan, med den effektivaste överföringen av cesium en tid efter ett nedfall är lav-ren-människa. Detta tillsammans med ett stort intag av renkött leder till att samer har ett mycket högre helkroppsinnehåll än den övriga befolkningen från samma region. Då livsstil och kultur i denna region starkt påverkar valet av föda leder det till stora variationer i cesiumintag mellan olika individer vilket i sin tur leder till stora variationer i helkroppsinnehåll.

Mätningar av ^{137}Cs i psoasmuskel

Provtagningen av psoasmuskel utfördes på Rättsmedicinska institutionen vid Norrlands universitetssjukhus i Umeå. Huvudsyftet med mätningarna av psoasmuskel var att uppskatta medelinnehållet av cesium i den norrländska befolkningen och ur dessa data kunna göra riskuppskattningar av följderna efter en händelse som Tjernobylolyckan. En av de parametrar som krävs för en sådan uppskattning är halveringstiden för den aktuella nukliden i populationen. De halveringstider som rapporterats rör sig mellan 1,7 och 7 år beroende på mätgrupp och tidsperiod [ref. 2, 3, 4]. Av resultaten från dessa studier kan man misstänka att halveringstiderna ökar ju längre tiden gått efter en olycka. Detta kunde även skönjas i vår tidigare studie [ref. 5].

Provtagning

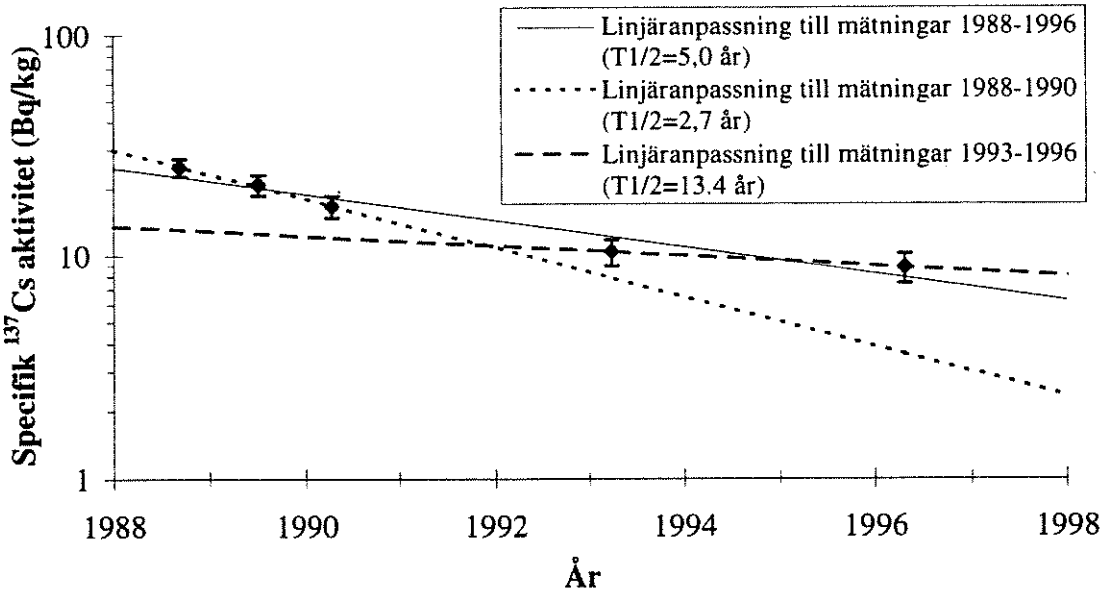
Anledningen till att psoasmuskeln valdes för mätning är att den har en relativt homogen sammansättning med liten variation mellan olika individer samt ett lågt fettinnehåll. I de obduktionsfall som valts ut har dödsorsaken oftast varit olycka, självmord eller naturlig död. För att erhålla en så representativ mätgrupp som möjligt uteslöts fall, där missbruk eller av andra orsaker stora avvikelser från normal kostsammansättning kunde misstänkas. Anledningen till detta var främst att gruppen missbrukare är kraftigt överrepresenterad bland de fall som kommer in till den Rättsmedicinska institutionen.

Statistiska metoder

Det urval som undersökningen bygger på är inte representativt för den befolkningsgrupp som den är ämnad att beskriva. De främsta och ur resultatmässigt hänseende viktigaste avvikelserna ligger i könsfördelningen och åldersfördelningen i provurvalet. För att komma till rätta med detta användes en viktning av mätresultaten för de olika grupperna. Den gruppindelning som användes bygger på kön och ålder och har viktats mot de fördelningar som gäller för regionen. Män respektive kvinnor delades in efter ålder på följande sätt: 0-29, 30-49, 50-69, 70- d.v.s. totalt 8 grupper. Medelvärdena i varje grupp viktades sedan enligt deras frekvens i undersökningen relativt frekvensen i regionen. Markbeläggningen av ^{137}Cs delades in enligt: låg < 10 Bq/m^2 < mellan > 50 Bq/m^2 >hög. Indelningen efter urbanisering gjordes efter kommun, större eller mindre än 25 000 invånare. I fråga om bostadsortens grad av urbanisering samt markbeläggningen i och kring bostadsorten befanns provurvalet ha en representativ fördelning.

Resultat

Med hjälp av multipel linjär regressionsanalys utfördes undersökningar av följande parametrars inverkan på cesiumkoncentrationen i provurvalet. Dessa parametrar var: kön, ålder, bostadsortens grad av urbanisering, samt markbeläggningen i och kring bostadsorten. Den beroende variabeln är i detta fall logaritmen av cesiumkoncentrationen då denna typ av provurval visat sig ha en log-normalfördelad frekvensfördelning [ref. 5]. Avtagandet av specifika aktiviteten för ^{137}Cs ser ut att bestå av en kombination av två halveringstider. Att försöka anpassa en dubbelexponentialfunktion till dessa punkter är svårt då punkterna ligger för tätt i tiden och den erhållna ekvationen blir känslig för variationer i införda gränsvillkor. Gränsvillkoret att aktiviteten ska gå mot noll då en längre tid förflutit efter depositionen krävs, i det här fallet, för att få en konvergerande funktion. Halveringstiderna som beräknats bygger därför endast på två enkla exponentialanpassningar till tidiga (1988-1990) respektive sena (1993-1996) mätdata, se Figur 1.



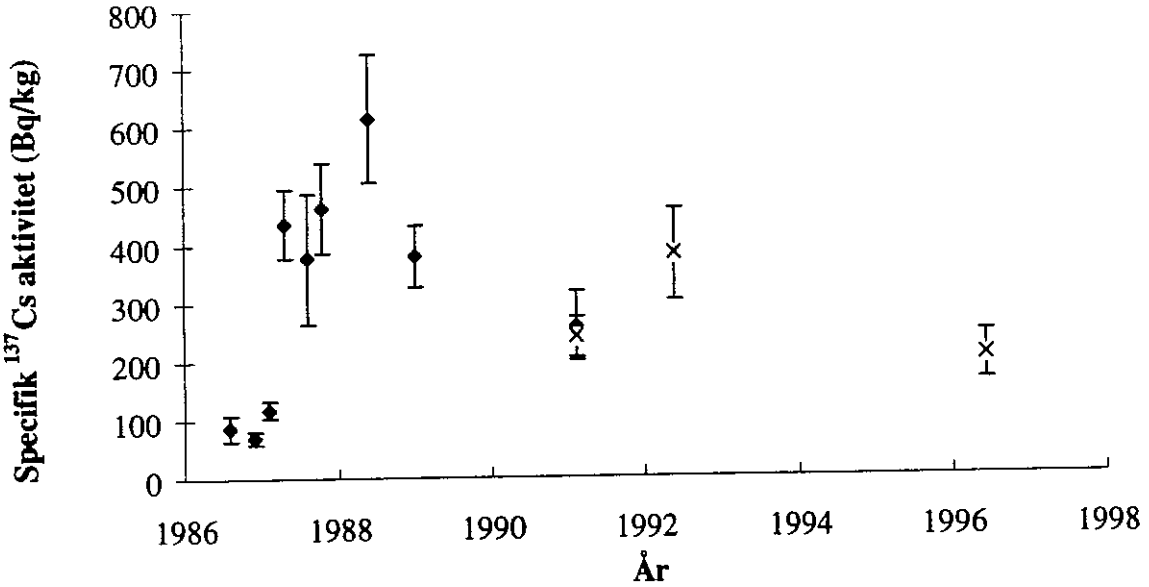
Figur 1. Resultat av mätningar på psoasmuskel under tiden 1988-1996. Aktivitetsinnehållet är per kg muskel. Halveringstider beräknade för de tidiga samt de senare mätningarna framgår av figurtext. Osäkerheterna med 95 % konfidensintervall är angivna.

Helkroppsmätningar

Sedan Tjernobylolyckan 1986 har helkroppsinnehållet av radioaktivt cesium i en grupp samer regelbundet mätts av institutionen för Radiofysik i Umeå samt FOA. Denna grupp består av 30 stycken utvalda samer från södra Norrlands fjälltrakter. För att få ett representativt urval mättes under 1991 och 1992 slumpmässigt utvalda grupper av samer samt allmän befolkning från Dorotea, Arvidsjaur samt Karesuando. Gruppen av samer i Dorotea kommer från samma område som de samer som ingår i den tidigare gruppen om 30 samer och markbeläggningen av ^{137}Cs är här ungefär 40 kBq/m^2. I Arvidsjaur är markbeläggningen ungefär en tiondel av den i Dorotea (4 kBq/m^2) och i Karesuando ytterligare 10 gånger lägre (0.4 kBq/m^2). Mätningarna på dessa grupper har under 1996 upprepats, se Tabell 1.
För samerna i södra Norrland visar mätningarna 1996 på en fortsatt minskning av ^{137}Cs-halterna, se Figur 2. Den halveringstid som mätningarna 1991-1996 ger för gruppen samer i Dorotea, 12 år, är jämförelsevis lång, se Tabell 2. Den längre halveringstiden i denna grupp kan bero på vidtagna motåtgärder för att minska halterna av cesium i renkött. Med dessa motåtgärder sker en relativt sett kraftigare sänkning av cesiumintaget tidigt efter ett nedfall. Då halterna med tiden naturligt börjar sjunka minskas de vidtagna motåtgärderna. I gruppen samer i Karesuando, som är den av samegrupperna som lever mest opåverkade av Tjernobylolyckan, är halveringstiden 5.9 år vilket stämmer väl med flera av de övriga grupperna.

Tabell 1. Observerade medelvärden och standardavvikelser av ^{137}Cs helkroppsinnehåll i studerade befolkningsgrupper.

Grupp	*Kön*	*Helkroppsinnehåll av ^{137}Cs*					
		Medelvärde (kBq)			*Observerat maximum (kBq)*		
		1991	1992	1996	1991	1992	1996
Dorotea Vilhelmina							
Samer	M	20 ±2	29 ±1,6	16,6 ±3,4	74	110	60.3
	K	6,6 ±3,0	18 ±6,8	7,3 ±3,1	23	66	30,4
Allmän befolkning	M	2,4 ±2,5	2,3 ±0,3	1,4 ±0,2	8.5	4.8	4.7
	K	1,5 ±0,2	1,4 ±0,2	0,7 ±0,1	2.9	3	2.1
Arvidsjaur							
Samer	M	7,4 ±1,3	9,9 ±2,7	4,2 ±1,3	31	58	21.3
	K	2,1 ±0,3	1,8 ±0,4	1,3 ±0,5	8.1	8.9	6.7
Allmän befolkning	M	1,3 ±0,2	2,0 ±0,6	0,7 ±0,2	3.5	12	3.9
	K	0,8 ±0,2	0,6 ±0,1	0,4 ±0,1	3.6	3.4	2.4
Karesuando							
Samer	M	4,3 ±0,3		2,6 ±0,3	7.1		1.8
	K	2,3 ±0,2		1,1 ±0,1	4.8		5.8



Figur 2. Specifik helkroppsaktivitet av ^{137}Cs i en referensgrupp samer (◆) samt samer bosatta i Dorotea-Vilhelmina regionen (×). Felstaplarna representerar medelvärdets standardavvikelse.

Tabell 2. Observerade halveringstider för ^{137}Cs i olika befolkningsgrupper efter atmosfäriska kärnvapentester och Tjernobylolyckan.

Befolkningsgrupp, och tidsperiod	$T_{(1/2)}$ *eff*	*Ref.*
Svenska samer 1965 - 1970	4.7 år	[4]
Svenska samer 1987 - 1991	5 år	[2]
Samer Dorotea 1991 - 1996	12 år	
Samer Arvidsjaur 1991 - 1996	5.9 år	
Samer Karesuando 1991 - 1996	5.9 år	
Allmän befolkning Dorotea 1991 - 1996	5.6 år	
Allmän befolkning Arvidsjaur 1991 - 1996	4.4 år	
Allmän befolkning i norra Sverige 1988 - 1996	5,0 år	

Överföringsfaktorer mor-barn via modersmjölk

Spädbarn är en av de mest strålningskänsliga grupperna av människor och relativt lite material finns att tillgå vid exempelvis en bedömning av hur kostråd till ammande mödrar efter ett nedfall av radionuklider skall utformas. Spädbarn är även svåra att mäta på grund av lågt aktivitetsinnehåll samt att det är svårt att uppnå tillfredsställande mättider. I en eventuell situation, där man har ett nedfall, är det därför lättare att mäta endast modern och därefter bedöma halterna i barnet. För denna bedömning krävs då att man känner överföringsfaktorn mellan mor och barn.

Metoder

I syfte att undersöka till vilken grad radionukliderna ^{137}Cs och ^{40}K transporteras till spädbarn via modersmjölk har vi studerat 12 mödrar med spädbarn, som endast ammats. Studien har utförts på två ställen i norra Sverige, Kiruna och Dorotea. I området kring Kiruna är markbeläggningen låg och homogen, ca. 400 Bq/m^2. I Kiruna undersöktes 9 par (mödrar med barn) och i Dorotea undersöktes 3 par. Från samtliga mödrar samlades prover av bröstmjölk in för aktivitetsanalys. Mätningarna av helkroppsinnehållet i mödrar och barn gjordes med en mobil helkroppsmätningsutrustning där en vanlig bilbarnsstol användes för spädbarnsgeometrin. Spädbarnsgeometrin hade ca 3 ggr högre effektivitet än vuxengeometrin då personen som mäts befinner sig närmare detektorn. De mjölkprover som samlades in mättes i provburkar (100 ml) av standardtyp. Då de flesta mjölkproverna innehöll liten ^{137}Cs aktivitet krävdes långa mättider, därför tillsattes i varje prov 3 ml 5 % natriumazid (NaN_3), för att minska inhomogeniteter i provet på grund av bakterietillväxt. Vidare tillsattes en antioxidant, butylerad hydroxytoluen (BHT), för att ytterligare minska inhomogeniseringen av provet under mättiden. Antioxidantens effekt är att motverka härskning och därmed sammanklumpning av lipidkedjorna vilket minskar den skiktning av proverna som annars uppstår. Innan varje mätning skakades proverna kraftigt i en Vortex-skak under ca 5 minuter.

Resultat

Halterna av ^{137}Cs hos mödrarna i Kirunagruppen var betydligt lägre än förväntat vilket leder till låg aktivitet även i barn och i modersmjölk. Följden av detta blir stora osäkerheter i individuella mätresultat och därigenom stor spridning inom gruppen. I Doroteagruppen erhölls dock högre värden för samtliga tre prover som togs på varje par. I denna grupp ingick en mor med relativt högt helkroppsinnehåll och överföringsfaktorerna hos detta par, stämmer väl överens med de medelvärden som beräknats för övriga par i undersökningen. Överföringsfaktorerna bygger på specifik aktivitet (Bq/kg) i respektive mätobjekt, se Tabell 3. Spridning av mätresultaten för ^{137}Cs är större än för ^{40}K vilket är att förvänta då helkroppsinnehållet av ^{40}K ej är lika beroende av individuella kostvanor som ^{137}Cs. Resultaten för överföringen från mor till mjölk stämmer väl överens med andra publicerade resultat [ref. 6,7].

Tabell 3. Beräknade överföringsfaktorer

	^{137}Cs (%)	*^{40}K (%)*
Mor-Mjölk	14 ± 12	44 ± 6
Mjölk-Barn	300 ± 300	300 ± 40
Mor-Barn	30 ± 150	120 ± 14

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Stråldosen till svenska befolkningen från cesium i kroppen, bestämd från helkroppsmätningar av en statistiskt representativ grupp.

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Abstract

The fallout from the Chernobyl accident contributes to the collective dose to the Swedish population through external and internal irradiation. The radionuclides giving the main part of the internal dose are radioactive caesium, ^{134}Cs and ^{137}Cs. In Sweden several investigations have been performed to study the behaviour of caesium in the body. As a part of these studies 179 persons, representing a random sample of the Swedish population, were whole-body measured during the autumn of 1994.

In co-operation with SCB, Statistics Sweden, a statistically representative sample of 1000 persons from the Swedish population was selected to be interviewed regarding their eating habits after the Chernobyl accident. From that group 200 persons were selected for whole-body measurements. They were divided into two groups, one consisting of persons from the five counties with the highest fallout and the other consisting of persons from the rest of the country. Each of the two groups were subdivided into three subgroups, consisting of men, women and children. When the number of persons in each group to be measured was decided consideration was taken to the expected standard deviation in the activity concentration within that group, resulting in an over-representation of persons living in areas with higher fallout.

163 persons were whole-body measured at SSI in Stockholm, and 16 persons at FOA, the National Defence Research Establishment, in Umeå. The statistically weighted mean of the dose to the Swedish population is calculated to about 5 µSv/year, 1994, and the activity concentration of ^{137}Cs is about 2 Bq/kg. The dose to men is about four times higher in the areas with higher fallout, and for women and children about twice compared to the rest of the country.

The results are compared to two similar investigations carried out in 1987 and 1988. Combining these results with measurements of a reference group at SSI and measurements of other groups makes it possible to estimate the collective dose to the Swedish population. Over a 50 year period the collective dose to the Swedish population is estimated to be about 1100 manSv from radioactivity in the food as a consequence of the Chernobyl accident. The results from the investigation are also compared to a national survey of the contents of caesium activity in Swedish food, carried out during the same time.

Inledning

Stråldosen till den svenska befolkningen från Tjernobylolyckan erhålls huvudsakligen från de radioaktiva ämnen som förts med vindarna till Sverige och deponerats här. Dosen härrör huvudsakligen från de båda cesiumisotoperna ^{134}Cs och ^{137}Cs. Detta gäller både för den externa bestrålningen, då det radioaktiva ämnet befinner sig utanför kroppen, t ex på marken, och för den interna bestrålningen då det radioaktiva ämnet befinner sig i människans kropp. För beräkning av stråldosen från den interna bestrålningen måste kroppens aktivitetsinnehåll och dess variation med tiden uppskattas. Detta kan göras direkt genom helkroppsmätningar som upprepas i tiden. Indirekt kan stråldosen beräknas med utgångspunkt från aktivitetsmätningar av livsmedel och med kunskaper om de radioaktiva ämnenas upptag och omsättning i människan.

I denna studie redovisas hur cesiumaktiviteten i Sveriges befolkning 1994 har bestämts genom helkroppsmätningar av ett slumpmässigt urval av personer från hela landet. Resultatet från denna undersökning jämförs med två liknande undersökningar från 1987 och 1988.

Urvalet

Ett slumpmässigt urval av 1000 personer i åldrarna 7-75 år drogs från "Registret över totalbefolkningen i Sverige" i augusti 1994. En första uppdelning i två grupper gjordes. I den ena gruppen ingick personer från de fem län (Uppsala, Västmanlands, Gävleborgs, Västernorrlands och Västerbottens län) som hade den högsta markdeponeringen av radioaktivitet efter Tjernobylolyckan, här kallade 5-län. Den andra gruppen bestod av personer från landets övriga 19 län, här kallade 19-län. Dessa två grupper delades i sin tur in i tre undergrupper bestående av barn (7-17 år), kvinnor (18-75 år) och män (18-75 år). För att minimera felet i stråldosberäkningen och eftersom den förväntade standardavvikelsen av cesiuminnehållet i de olika grupperna var olika gjordes en viktning av grupperna, vilket innebar att man erhöll en överrepresentation i urvalet av personer från de fem länen med högst markdeponering och av män. Antalet personer i hela landet i de olika grupperna och andelen som mättes visas i Tabell 1.

Tabell 1. Antal personer i hela landet i de olika grupperna, dess andel av hela populationen och andelen helkroppsmätta.

Grupp	Antal personer i gruppen	Andel av hela populationen	Antal mätningar i gruppen	Andel av alla mätningar
5-län, barn	174 146	0,024	10	0,056
5-län, kvinnor	475 133	0,065	28	0,156
5-län, män	483 543	0,066	35	0,196
19-län, barn	934 345	0,128	17	0,095
19-län, kvinnor	2 615 245	0,358	41	0,229
19-län, män	2 625 733	0,359	48	0,268
Alla	7 308 145	1,000	179	1,000

Statistiska centralbyrån genomförde en inledande intervju i det första urvalet om 1000 personer. Intervjuerna hade två syften, dels att undersöka förändringar i svenska folkets matvanor till följd av Tjernobylolyckan, dels att finna 200 personer som var villiga att delta i helkroppsmätningar vid SSI i Stockholm. För att minska bortfallet erbjöds personer från de norra länen möjligheten att genomföra mätningen vid FOA i Umeå.

Av de 200 personer som valdes ut för mätning mättes 179 personer från oktober till december 1994. Bortfallet var således ca 10%, Tabell 2.

Tabell 2. Antal personer i urvalet från respektive grupp och andel som deltog i hellkroppsmätningarna.

Grupp	Antal i urval	Antal mätningar	Andel mätningar
5-län, barn	12	10	0,83
5-län, kvinnor	30	28	0,93
5-län, män	38	35	0,92
19-län, barn	18	17	0,94
19-län, kvinnor	46	41	0,89
19-län, män	56	48	0,86
Alla	200	179	0,90

Helkroppsmätningar

Helkroppsmätningarna utfördes i första hand med SSIs helkroppsscanner (Ref 1) vilken har tre stycken NaI(Tl)-detektorer med en diameter av 12,7 cm och en höjd av 10,2 cm. För barn under ca 10 år gjordes mätningarna i "stol-geometri" med en NaI(Tl)-detektor av samma storlek. Mättiderna var ca 40 minuter i helkroppsscannern och 20 minuter för barnmätningarna. Båda helkroppsmätarna har kalibrerats genom mätningar av tre olika fantom innehållande kända aktiviteter av ^{134}Cs, ^{137}Cs och ^{40}K. För helkroppsscannern har sju olika fantomvikter använts (30, 40, 50, 60, 70, 80 och 90 kg) och för stol-geometrin fem olika fantomvikter (5, 10, 20, 30 och 40 kg).

Beräkningarna av ^{134}Cs-, ^{137}Cs- och ^{40}K-aktiviteten vid en helkroppsmätning har gjorts genom en linjär minsta kvadratanpassning av referensspektra av de tre nukliderna och ett bakgrundsspektrum till det spektrum som erhållits vid helkroppsmätningen. Metoden ger en separat aktivitetsbestämning av de båda cesiumisotoperna trots den relativt dåliga energiupplösning som NaI(Tl)-detektorer har.

FOAs helkroppsmätare har en högren germaniumdetektor med 50% effektivitet (relativt en ⌀ 3" x 3" NaI(Tl)-detektor) monterad i en modifierad stol-geometri. Geometrin är en kombination av en stol-geometri och en båg-geometri, där stolen har formen av en vagga, (Ref 2). Fördelen med denna geometri jämfört med en ordinär stol-geometri är ett mindre uttalat massberoende för effektiviteten. Detektorsystemets effektivitet har kalibrerats med fantom uppbyggda av kylklampar, där varje kylklamp har en vikt av cirka 0,5 kg. Tre fantom med kända mängder av ^{134}Cs, ^{137}Cs respektive ^{60}Co har använts för kalibrering. De tre fantomens uppbyggnad gör det möjligt att konstruera fantom i varierande massor och helkroppsmätaren har kalibrerats för massor mellan 10-90 kg. Massberoendet kan för dessa nukliders gammaenergier anses vara konstant för alla massor över 45 kg medan en korrigering av effektiviteten måste göras för mätning av personer med massor lägre än 45 kg.

Resultat och diskussion

Sammanlagt utfördes helkroppsmätningar på 179 personer, 163 vid SSI och 16 vid FOA i Umeå. Kroppsinnehållet av ^{137}Cs redovisas i Tabell 3 och innehållet av ^{137}Cs per kg kroppsvikt redovisas i Tabell 4 och Figur 1. Det sammanvägda medelvärdet för hela landet är 2,0 Bq/kg. Vid de två tidigare landsomfattande mätningarna 1987 och 1988 erhölls 7,7 respektive 5,6 Bq/kg (Ref 3 och 4). Aktiviteten av ^{134}Cs låg 1994 i de flesta fall under detektionsgränsen. Genom att jämföra tidigare uppmätta förhållanden mellan ^{134}Cs- och ^{137}Cs-aktiviteten och korrigera för de olika halveringstiderna har ^{134}Cs-aktiviteten uppskattats till ca 3% av ^{137}Cs-aktiviteten i slutet av 1994.

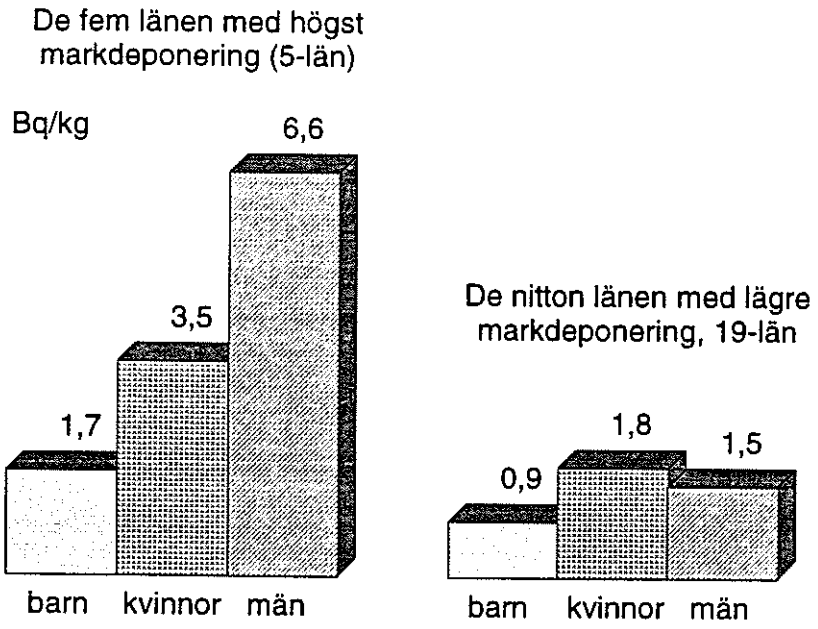
Tabell 3. Helkroppsinnehållet av ^{137}Cs i de olika grupperna.

Grupp	Antal	Medelvärde Bq	Median Bq	Högsta värde Bq
5-län, barn	10	79	74	180
5-län, kvinnor	28	222	135	867
5-län, män	35	543	247	4477
19-län, barn	17	39	34	123
19-län, kvinnor	41	117	69	529
19-län, män	48	119	89	725

Tabell 4. Innehållet av ^{137}Cs per kg kroppsvikt i de olika grupperna.

Grupp	Antal	Medelvärde Bq/kg	Median Bq/kg	Högsta värde Bq/kg
5-län, barn	10	1,7	1,5	2,8
5-län, kvinnor	28	3,5	2,3	15,5
5-län, män	35	6,6	3,2	52,2
19-län, barn	17	0,9	0,8	1,9
19-län, kvinnor	41	1,8	1,1	8,2
19-län, män	48	1,5	1,1	9,2

Osäkerheten i de angivna medelvärdena av ^{137}Cs-aktiviteten har uppskattas till mellan 15 och 30%, för de enstaka grupperna.



Figur 1. Medelvärdet av kroppsinnehållet, Bq/kg, av ^{137}Cs i olika grupper.

De skillnader i kroppsinnehållet av ^{137}Cs, Figur 1, som finns mellan de två olika länsgrupperna ger att medelvärdet av kroppsinnehållet av cesium i de fem länen med högsta markdeponeringen är ca 3 gånger högre än för övriga län, 4,5 respektive 1,5 Bq/kg. Detta överensstämmer med de skillnader som erhölls vid mätningarna 1987 och 1988. De skillnader som finns i 1994 års undersökning mellan kvinnor och män i 5-länsgruppen kan eventuellt förklaras av de skillnader i matvanor avseende vilt, insjöfisk, svamp mm som framgick av SCBs intervjuer om matvanorna. Dessa intervjuer visade också att, nio år efter olyckan, 7% av den svenska befolkningen fortfarande hade förändrade matvanor.

Parallellt med undersökningen av helkroppsinnehållet under hösten 1994 utfördes en studie vid SSI, i samarbete med Statens Livsmedelsverk, av cesiuminnehållet i livsmedel i Sverige (Ref 5). En uppskattning av kroppsinnehållet av ^{137}Cs från denna undersökning visar ett medelvärde till den svenska befolkningen av ca 1,3 Bq/kg. I denna studie ingår dock endast livsmedel från butiker, där livsmedel som vilt, insjöfisk, svamp mm till en stor del inte ingår, vilket kan förklara att uppskattningen av kroppsinnehållet av cesium blir något lägre.

Den effektiva dosen till en individ har beräknats som summan av dosen från ^{134}Cs och ^{137}Cs. Den dosmodell som använts (Ref 6) tar hänsyn till kroppsvikten och anger föjande samband mellan dos, cesiumaktivitet och kroppsvikt.

Effektiv dosrat (µSv/år) = $1{,}64 \cdot A \cdot m^{-0{,}812}$ för ^{134}Cs och
effektiv dosrat (µSv/år) = $1{,}40 \cdot A \cdot m^{-0{,}889}$ för ^{137}Cs.

där A = helkroppsinnehållet i Bq
m = kroppsvikten i kg

I Tabell 5 redovisas resultaten av dosberäkningarna. Baserade på mätningarna 1994 har det sammanvägda medelvärdet av den effektiva dosraten beräknats till ca 5 µSv/år. Detta kan jämföras med motsvarande värden vid de tidigare landsomfattande mätningarna som omräknat till årsdos gav 23 µSv/år och 27 µSv/år under första året (maj 1986 - maj 1987) respektive andra året (maj 1987 - maj 1988) efter Tjernobylolyckan (Ref 4).

Tabell 5. Den beräknade effektiva dosen per år, 1994, till de olika grupperna.

Grupp	Antal	Medelvärde µSv/år	Median µSv/år	Högsta värde µSv/år
5-län, barn	10	3,9	3,3	6,4
5-län, kvinnor	28	8,0	5,3	35,6
5-län, män	35	15,9	7,6	125,7
19-län, barn	17	1,9	1,9	4,2
19-län, kvinnor	41	4,1	2,5	19,0
19-län, män	48	3,7	2,6	22,0

Osäkerheten i de angivna medelvärdena har uppskattas till mellan 15 och 30%.

Vid beräkningen av stråldosen till följd av Tjernobylolyckan måste förutom det genomsnittliga helkroppsinnehållet av cesium också dess variation med tiden vara känd.

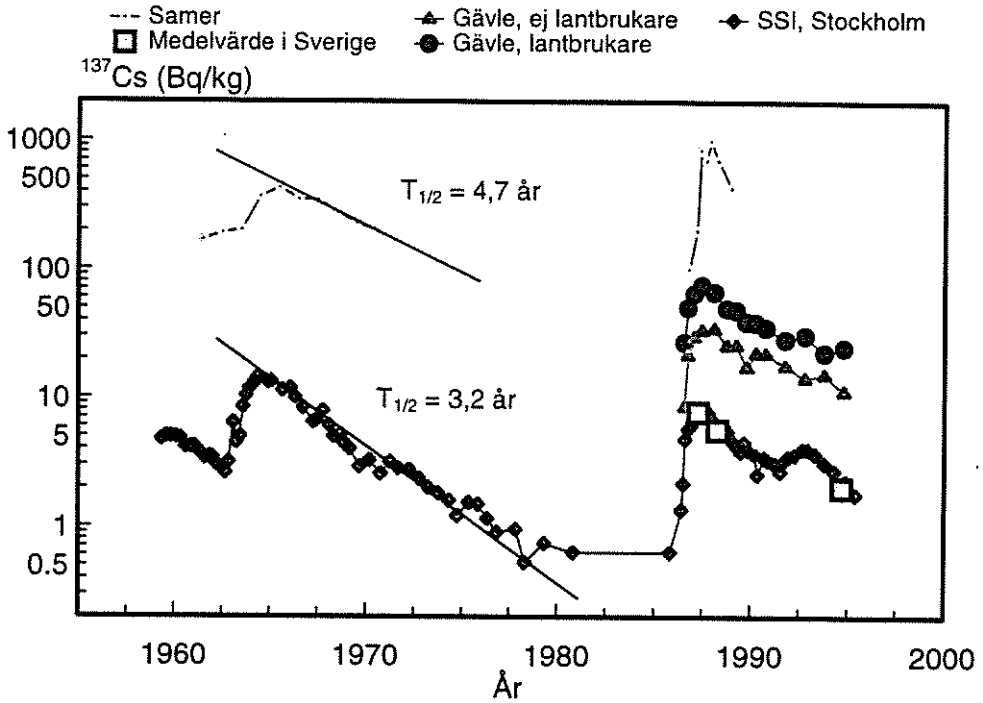
Som ett led i SSIs arbete att övervaka den allmänna strålmiljön i Sverige och som uppföljning av Tjernobylolyckans påverkan av vår strålmiljö utförs vid SSI helkroppsmätningar av olika grupper.

Sedan 1959 har upprepade mätningar utförts på en grupp personer anställda vid SSI, kontrollgruppen (Ref 7). Efter Tjernobylolyckan och till och med 1994 har 34 mätomgångar genomförts av denna grupp som består av ca 35 personer. Resultaten av dessa redovisas i Figur 2.

Upprepade helkroppsmätningar utförs också på personer från ett område nära Gävle med ett relativt stort radioaktivt nedfall. Denna grupp består av 12 lantbrukare och 12 andra personer från samma område. Till och med 1994 har 14 mätomgångar genomförts av denna grupp. Som framgår av Figur 2 är förändringarna i tiden av, cesiuminnehållet, likartade mellan kontrollgruppen och Gävlegruppen medan nivåerna är något olika.

De uppmätta landsmedelvärdena av cesiuminnehållet 1987, 1988 och 1994 överensstämmer väl med motsvarande mätningar av kontrollgruppens cesiuminnehåll. Det innebär att förändringarna i kontrollgruppens cesiumaktivitet förefaller att väl representera förändringarna hos Sveriges befolkning mellan 1987-1994.

Från mätningarna 1987 och 1988 uppskattades kollektivdosen i Sverige, under de två första åren efter olyckan, till ca 400 manSv från radioaktivt cesium i kroppen (Ref 8). Tillsammans med mätningar av olika grupper, Figur 2, och de tre landsomfattande undersökningarna som väl överensstämmer med förändringarna och värdena för SSIs kontrollgrupp, har kollektivdosen uppskattats ur aktivitetskurvan för SSI under åren 1988-1994. Under antagandet att kroppsinnehållet minskar med en halveringstid av 4 år efter 1994 erhålls en uppskattning av 50 års kollektivdos till ca 1100 manSv från cesium i kroppen med en osäkerhet av ± 200 manSv. Motsvarande uppskattning av kollektivdosen från externa strålkällor ger ca 5000 manSv (Ref 9).



Figur 2. Uppmätt kroppsinnehåll, Bq/kg, av ^{137}Cs hos några grupper i Sverige mellan 1958-1995.

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MIGRATION OF RADIOCAESIUM IN SWEDISH SOIL PROFILES AFTER THE CHERNOBYL ACCIDENT, 1987-1995

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ABSTRACT

The study covers both temporary and permanent grasslands, of which some are fertilized and used for hay production and others are semi-natural or natural pastures used for extensive grazing. Eight soils representing different soil types and soil textures were investigated; five cultivated or semi-natural minerals soils (Dystrochrept, Haploboroll, Udorthents, Cryorthent), two cultivated organic soils (Sulfihemists) and one podsolized soil (unploughed) (Haplocryod). At the selected sites, the ground deposition was 14-184 kBq m^{-2} (average 82). The ^{137}Cs-activities were measured in cm-sliced 25 cm deep soil cores as well as in the soil horizons (0.7-1.0 m). Soil cores were collected at two or three occasions between 1987 and 1994, and migration rates were calculated from the median depths. Grass samples were taken from the same locations for determination of the transfer of ^{137}Cs from soil to grass (TFg, m^2 kg d.w.$^{-1}$).

The study showed that after eight years the main part (50-92 %) of the ^{137}Cs fallout was still retained in the upper 0-5 cm (median depth 2.3-5.1 cm), although considerable amounts have migrated to deeper soil layers. This was most pronounced in the organic soils and in the podzol where ^{137}Cs was found to about 50 cm depth. In the podzol 40 % of ^{137}Cs was found below the O-horizon (0-5 cm), indicating a transport of ^{137}Cs within the soil profile. The migration rates (1987-1994) were in the range 0.2-0.6 cm year^{-1}, higest in one organic soil. The transfer of ^{137}Cs to grass was highest in the mountain region, intermediate in the organic soils, and lowest in the mineral soils. TFg decreased with time at all sites, except in the podsolized soil where the transfer was slightly higher in 1994 than in 1990.

The different distribution patterns in the studied soils, indicated differences in cation exchange capacities and in the occurrence of specific binding sites (illitic clay minerals). The results also indicate that differences in land use and management practices are of importance. ^{137}Cs seems to be more mobile in O- and Ah-horizons rich in decomposing plant residues than in Ah- or Ap-horizons dominated by well-decomposed and humified organic matter. The results can be further evaluated by means of mathematical models for transport of radionuclides as well as through additional analyses of soil and clay mineralogy.

A portable gamma spectrometry system for environmental applications

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Gamma spectrometry systems for radionuclide mapping and search have been in use for more than 30 years. Applications in the field of mineral exploration and geological mapping have resulted in large airborne systems often integrated with other geophysical equipment. Vast surveys have been performed all over the world. The usefulness of such systems for environmental applications has been proven many times, i.e. by the Swedish Geological Company in 1986 when mapping Chernobyl fallout in Sweden. The arrival of GPS and the general trend of smaller, less power consuming and more competent electronic equipment have opened new possibilities regarding system design. The Swedish Radiation Protection Institute has developed a small gamma spectrometry system for environmental applications. It is small enough to be battery operated and portable as a back-pack system and contains what is needed for both small and extensive surveys and search operations in radiation protection and environmental research. If the system is connected to a PC, software developed by SSI can be used for real time analysis and display af results. The system design principles, the hardware, calibration technique and the software will be presented. Two application examples will he given, one focused on radionuclide mapping and search in emergency situations and the other on radioecology research.

^{90}Sr in various matrices - speciation and analysis

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Abstract

Due to long half-life, high fission yield, high mobility in ecosystems and high radiological toxicity, questions concerning short and long term consequences of ^{90}Sr contamination of the environment from nuclear sources are frequently addressed.

In order to determine the degree of ^{90}Sr contamination, proper determination of the total concentrations in samples from various compartment of different ecosystems is needed. However, in order to assess the short and long term transfer within the ecosystem, the determination of physico-chemical form is essential, as information on the mobility and bioavailability of ^{90}Sr is needed. Hence, the analytical challenges include accurate determination of ^{90}Sr in various matrices, and the time-consuming radiochemical procedures should be simplified. Furthermore, fractionation techniques should be developed to categorise ^{90}Sr-species according to size (i.e. nominal molecular weight distribution) and reactivity (e.g. reversible or irreversible interactions with other components).

In the present work fractionation techniques for ^{90}Sr speciation in water, soil and sediments will be focused and the introduction of simplified procedures for ^{90}Sr in waters will be discussed.

Titrering med EDTA som alternativ metode for bestemmelse av yttrium-utbytte ved ^{90}Sr–analyser

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Sammendrag

Bestemmelse av yttrium-utbytte ved ^{90}Sr-analyser har tradisjonelt vært utført gravimetrisk etter felling av yttrium. Ved IFE på Kjeller har en alternativ metode for yttriumbestemmelse vært under utprøving. Metoden, hvor yttrium bestemmes ved titrering med EDTA i bufret løsning, viser bedre reproduserbarhet enn den tradisjonelle metoden og har nå erstattet denne ved laboratoriets rutineanalyser av ^{90}Sr.

I det følgende diskuteres tradisjonell og ny metode med basis i resultater fra over to år med bestemmelse av yttrium-utbytte med begge metoder på de samme prøvene.

Innledning

Bestemmelse av yttrium-utbytte ved ^{90}Sr-analyser har tradisjonelt vært utført ved veiing av gløderest etter måling av yttriumpreparatet i β-kammer. Det er alminnelig kjent at den gravimetriske metoden har betydelige svakheter som bidrar til redusert reproduserbarhet blant annet: i) at gløderesten også inneholder andre oksider enn Y_2O_3, og ii) måle-usikkerhet som skyldes stor vektforskjell mellom tara og gløderest. Siden analyseverdien i ^{90}Sr-analyser er avhengig av bestemmelse av yttrium-utbyttet, er det viktig å kunne redusere usikkerheten ved denne bestemmelsen.

Ved Institutt for energiteknikk (IFE) er det utviklet en alternativ metode for denne bestemmelsen. Metoden (titrimetrisk bestemmelse av yttrium) er enkel, rask og har vært brukt parallelt med den gravimetriske metoden på over 150 ^{90}Sr-analyser i de siste to år. I det følgende presenteres den nye metoden i en sammenlignende evaluering med gravimetrisk metode i lys av resultater fra utprøvningsperioden.

Metoder

Sr-separasjon

For alle prøvemedia unntatt melkeprøver utføres Sr-separasjonen etter anerkjente metoder (DOE, 1992; Harvey *et al.*, 1989) som alle innebærer felling av Sr med rykende HNO_3. Videre fellinger med for eksempel barium, jern, kromat og karbonat avhenger av prøvetype. Etter endt Sr-separasjon tilsettes yttrium-bærer til prøveløsningen som så settes til henstand i 2 uker..

For prøver av melk tilsettes yttrium-bærer til å begynne med, før yttrium separeres direkte fra prøven (DOE, 1992: HASL 300 E-Sr-05-01 fast method).

Gravimetrisk Y–bestemmelse

Etter telling måles vekt av gløderest etter forasking i porselensdigel ved 900 °C i hurtigforasker.

Titrimetrisk Y–bestemmelse (ny metode)

I det følgende er gitt en punktvis oversikt over den titrimetriske metoden:

1. Etter telling glødes filter med prøve i porselensdigel ved 900 °C i hurtigforasker.
2. Gløderest løses med kons. HNO_3 (3 ml) og dampes inn til tørrhet.
3. Prøven løses i acetat-buffer (20 ml, pH 4.4).
4. Prøven tilsettes tetra-natrium-xylenolorange (0.5 %,1 dråpe, indikator).
5. Prøven titreres med EDTA (Titriplex III, 0.01 M) til gult omslag.

For å unngå interferens fra eventuelle andre metallioner i makromengder (Fe^{3+}, Ca^{2+} og Ba^{2+}) må titreringen utføres ved svakt sur pH i bufret løsning (Reilley & Schmid, 1958). Forsøk med titrering i ubufret løsning ga ikke stabile resultater og det var også problemer med å definere omslagspunkt.

Kompleksiometrisk titrering med EDTA finnes beskrevet for eksempel i MERCK (1982).

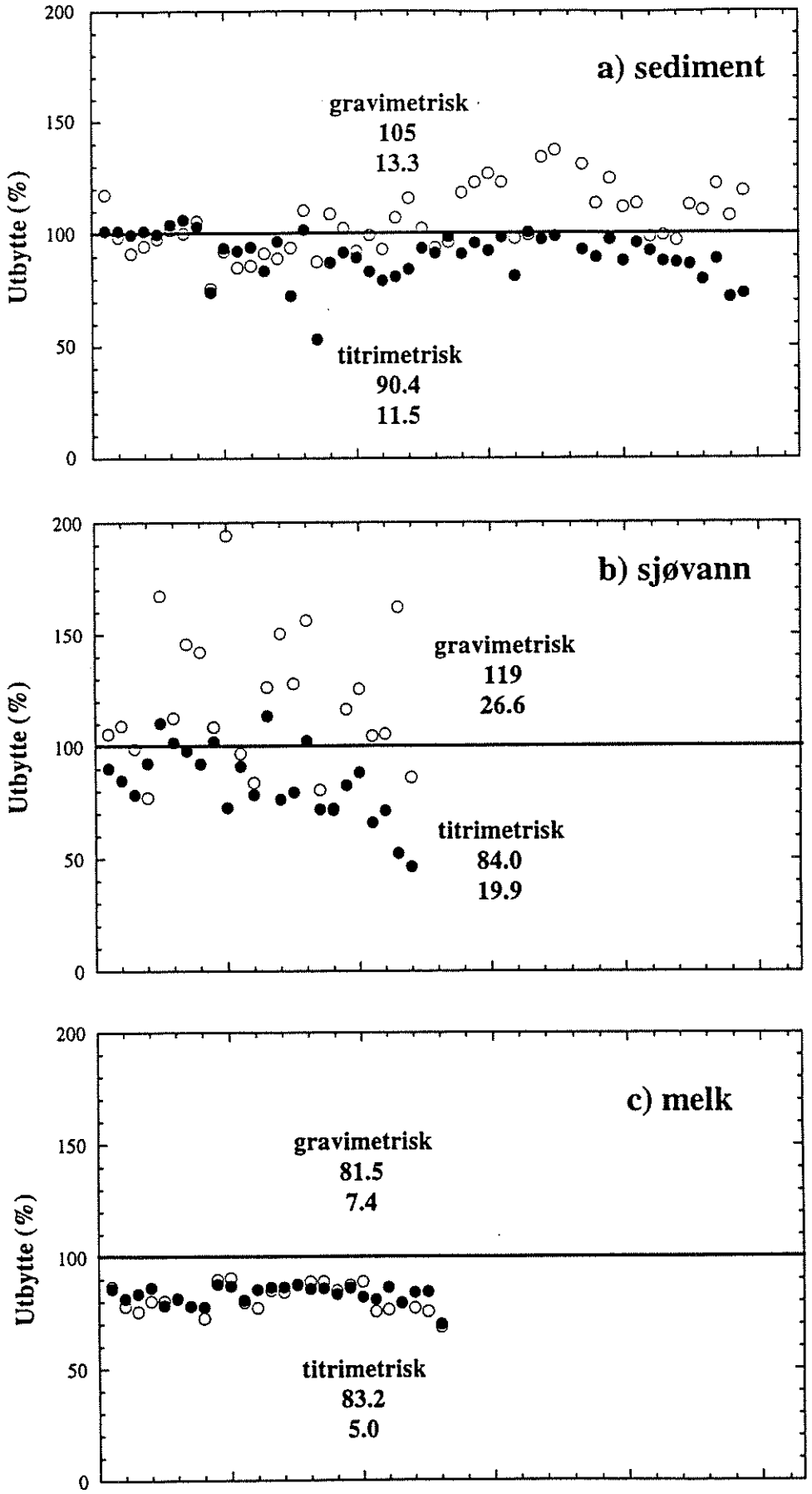

Fig. 1: Sammenligning av yttrium-utbytte for gravimetrisk (○) og titrimetrisk (●) metode for a) sedimentprøver, b) sjøvannsprøver og c) prøver av melk. For hver metode og prøvetype er det angitt gjennomsnittlig utbytte (over) og relativt standard avvik (under).

Resultater og diskusjon

Metoden har vært utprøvd på de prøver som laboratoriet rutinemessig analyserer. Dette innebærer mange ulike prøvemedia (jord, sediment, havvann, ferskvann, melk og biota som kjøtt, bein, vegetasjon og tang)og stor variasjon i ^{90}Sr-innhold (fra havvannsprøver med svært lavt innhold fra Karahavet til høyaktive prøver fra området rundt Majak, Russland). Siden separasjon av Sr i hvert fall til en viss grad antas å være avhengig av prøvematrix er resultatene ordnet etter prøvetype. Av plasshensyn er bare tre prøvetyper tatt med her, men resultater og vurderinger for andre prøvemedia kan fås ved henvendelse til forfatterne. Resultatene gir ikke grunnlag for å teste eventuelle forskjeller i yttrium-utbytte som skyldes forskjeller i ^{90}Sr-innhold.

Gravimetrisk bestemmelse av yttrium i prøver av sedimenter (Fig. 1 a) og sjøvann (Fig. 1 b) gir ofte et utbytte signifikant over 100 %. Dette representerer en situasjon som det i praksis er umulig å kompensere for, fordi det kan indikere at også andre oksider enn yttriumoksid bidrar til vekt i gløderesten. Ved titrimetrisk metode unngås dette problemet og reproduserbarheten slik den er målt her (som RSD) er også noe lavere. Basert på erfaringer gjort under utprøving av metoden, ser det ut til at variasjonen i titrimetrisk yttrium-utbytte hovedsakelig skyldes variasjoner mellom prøver (matrix). Det er nærliggende å tro at forskjeller under Sr-separasjonen (f. eks. i innholdet av kalsium i prøvene) også påvirker fellingen av yttrium senere i analyseprosedyren.

Tabell 1: Yttrium-utbytte for prøveparalleller av jord og sedimenter.

Prøve	N	Gravimetrisk bestemmelse		Titrimetrisk bestemmelse	
		Middel	RSD	Middel	RSD
Sediment 1	5	128	7.3	95.1	4.1
Sediment 3	4	100	11.6	101	0.8
Sediment 4	4	101	3.3	103	2.5
Jord	4	102	7.2	88.3	3.1

Bestemmelser av yttriumutbytte i prøveparalleller av jord og sedimentprøver (Tabell 1) viser mindre spredning og mer stabile resultater for den titrimetriske metoden. Det må presiseres at den beregnede usikkerheten er basert på yttriumbestemmelser i rutineanalyserte prøver. Eventuelle matrix-effekter inngår derfor også i verdiene for RSD.

Konklusjon

Titrimetrisk bestemmelse av yttrium-utbytte er funnet å være bedre enn gravimetrisk bestemmelse, og den titrimetriske metoden har nå erstattet gravimetrisk metode ved ^{90}Sr-analyser ved IFE–Kjeller.

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Radiocaesium redistribution in a Swedish pine forest: Dynamics of root uptake

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Abstract - Studies on radionuclide transport in forested landscapes since the Chernobyl accident have demonstrated the high mobility of radioactive caesium within the forest ecosystem. To better characterize this phenomenon, the resdistribution of radiocaesium in a Swedish pine forest is presently studied by field experiments and modelling. Data concerning the radiocaesium contents in pine trees has been collected in three different forested locations near the town of Gävle. The forests differ by their ages, with one of the locations established after the Chernobyl accident, providing a unique opportunity to study root uptake independently. The data describes the redistribution of radiocaesium over time in the differently aged forest stands, demonstrating the difference in the dynamics as a function of tree age and original contamination process. This report focuses on the portion of the data which characterizes root uptake. The data show that the concentration of Cs-137 in the new growth of Scots pine trees planted 2 years after the Chernobyl accident continues to increase as of 1995. The averaged total activity in these trees has increased by a factor of 35 between 1991 and 1995. It is concluded that the transport of Cs to new growth comes primarily from root uptake. This is part of a study which will estimate natural landscape ecosystem exposures from radioactive contamination.

Introduction

Radioactive contamination of the forest environment has clearly been demonstrated after the Chernobyl accident to have long-term radiological consequences. Large areas of forested land in Sweden were contaminated by the radioactive fallout in 1986. The only long-lived radionuclide of importance is Cs-137. Estimates indicate that the ecological half-life in the forest environment in some cases can be as along as 30 years, which is the physical half-life of Cs-137 (Bergman 1994).

This report focusses on the dynamics of root uptake of Cs-137 in a subsystem of the forest ecosystem, Scots pine (*Pinus Silvestris L.*). This is a part of an ongoing project designed to obtain a basis for forming reliable assessments of the radiation exposure to man under different time scales from radionuclides in plant and animal products of representative semi-natural ecosystems in Europe. It is known that the estimated collective dose to the Swedish population due to intake from food products from the forest during the 50 years following the Chernobyl accident is a significant part of the total internal dose.

Study areas Our long-term project is based on samples collected from an area in Sweden with a Cs-137 deposition resulting from the Chernobyl accident of between 85 and 220 kBq/m^2. Table 1 lists specific characteristics of these sites. This report focusses primarily on sites P and H.

Results and discussion

Contamination of Scots pine - direct contamination and root uptake The trees at sites H and L were exposed to direct contamination by wet deposition during the last days of April 1986. Around 80% of the radiocaesium was initially intercepted by the canopy (Melin and Wallberg 1991). While part of these radionuclides will migrate from the canopy to the forest floor, part of the intercepted activity is assumed to be assimilated by the tree and internally redistributed between its different parts (translocation). The trees also incorporate radioactive caesium from the soil through root uptake. Accordingly, the concentrations of Cs-137 in the tree samples from sites H and L are assumed to be due to a combination of both foliar and root uptake. To study the relative importance of these two mechanisms, the trees at a third site (P) have been sampled.

Table 1 Basic data on the sites Hille (site H), Långsjön (site L) and Prylen (site P)

PARAMETER	SITE H	SITE L	SITE P
Location	60°48´N 17°13´E	60°48´N 17° E	60°48´N 17° E
Trees			
Type	Pinus Sylvestris L.	Pinus Sylvestris L. (60%) Picea abies Karst (40%)	Pinus Sylvestris L. (90%)
Year of establishment	1930	1977	1988
Site index (H100)	T29	T24	T23
Stems/ha	940	2100	2300
Current annual increment (m^3/ha)	8.2	6	5.5
Understorey vegetation	grass Pleurozium schreberi Dicranium sp.	grass Pleurozium schreberi Dicranium sp.	grass Dicranium sp.
Soil			
Classification	Podsol	Podsol (watersorted moraine sandy-fine sandy-clay fraction<2%-top soil very rocky)	Podsol (watersorted moraine - sandy-clay fraction<2% top soil very rocky)
Humus type	Mor	Mor	Mor
Humus layer (cm), A_0	2-6	5-10	5-10
Bleached layer (cm), A_1	5-12	7-12	5-9
pH(H_2O), humus	4.3	4.0	4.0
Sampling date	1986,1990,1991, 1994	1991, 1994	1991, 1994, 1995, 1996
Estimated deposition (kBq/m^2) Cs-137 aerial survey Cs-137 A_{01}measurement	 >180 90	 30-40 20	 30-40 37

Root uptake only The trees at site P were not directly exposed to the Chernobyl fallout. They were still in a greenhouse at the time of deposition, and planted at the site in 1988. These young trees are used in our long term study to follow independently the dynamics of root uptake of radiocaesium. The only possible external contaminations are limited to resuspension, which is considered to be negligable a few years after deposition, and to a possible surface contamination at the time of planting.

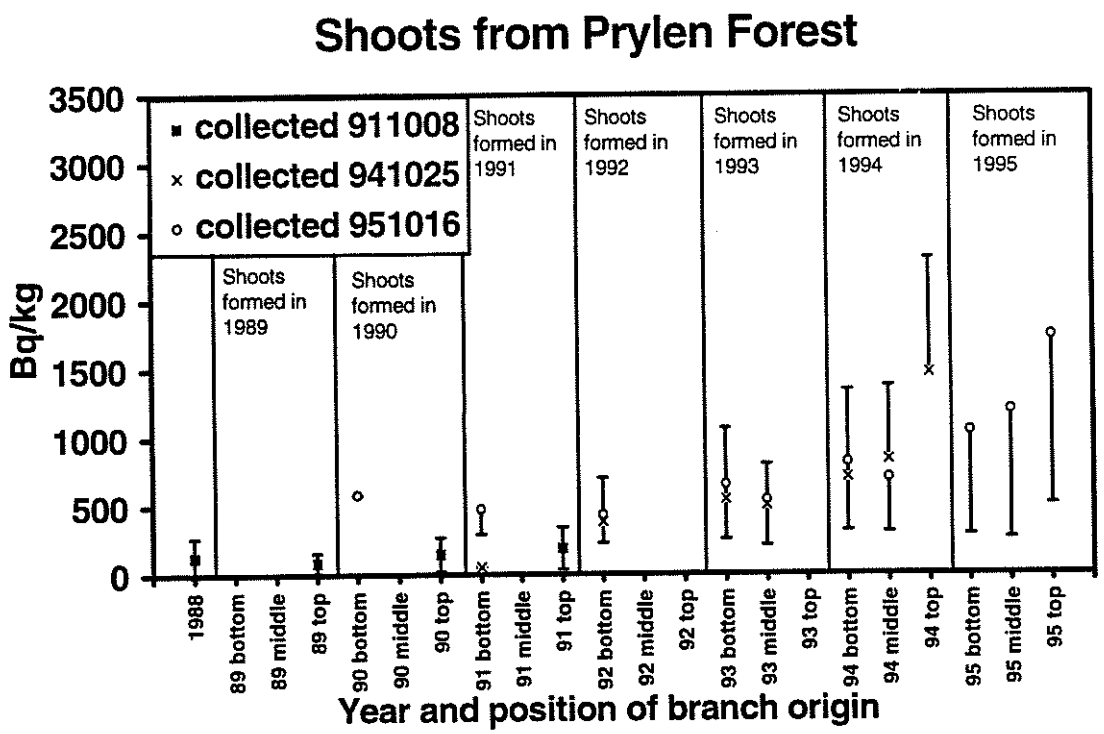

Figure 1.

Figures 1 and 2 show the Cs-137 concentrations in shoots and needles from the young Prylen forest collected in 1991, 1994, and 1995. Samples from growth of the current year (C shoots), 1 year old growth (C+1 shoots), etc., were collected from four different trees at each collection time. Each data point displayed in the figures is the average of up to four values

with the standard deviation displayed for each set of points. The standard deviation is displayed on only 1 side of each point to increase the clarity of the figure. At any given collection time, needles can usually be obtained up to 4 years old, and shoots can be obtained up to 5 years old. The data are also displayed according to location on the tree where they were collected: bottom, middle, or top. The concentration of Cs-137 in the new growth of these young Scots pine trees continues to increase, as of 1995.

The agreement between points from the same position on the tree (bottom, middle, or top) and the same year of origin, but from different collection years and different trees is suggestive. It implies not much is changing over time, and is consistent with a mechanism that root uptake is dominating. If Cs-137 was translocating out of the older needles and into the shoots and upwards to newly forming shoots and needles, it would be decreasing in concentration in the same aged needles with later collection times. This data does not suggest this mechanism. This data implies that there is increase in Cs in each new growth due primarily to root uptake. The general importance of root uptake is also strengthened by the fact that the total activity per tree is increasing with increasing biomass.

Some degree of translocation cannot be ruled out, however. There is a suggestion, especially in Figure 2, of a decrease in Cs-137 concentration in the needles with age. For example, the needles formed in 1994 and collected in 1995 are all lower in concentration than those collected in 1994, the year they were formed. Because the points fall within the limits of the standard deviations of each other, and because the data come from different trees, this observation is only a suggestion. A current investigation is concentrating on monthly sampling of needles at site P to help clarify the yearly cycling of Cs-137, e.g., whether it is transported back into the tree from the older needles just before they dry up and fall off, and therefore available for translocation to newly forming needles.

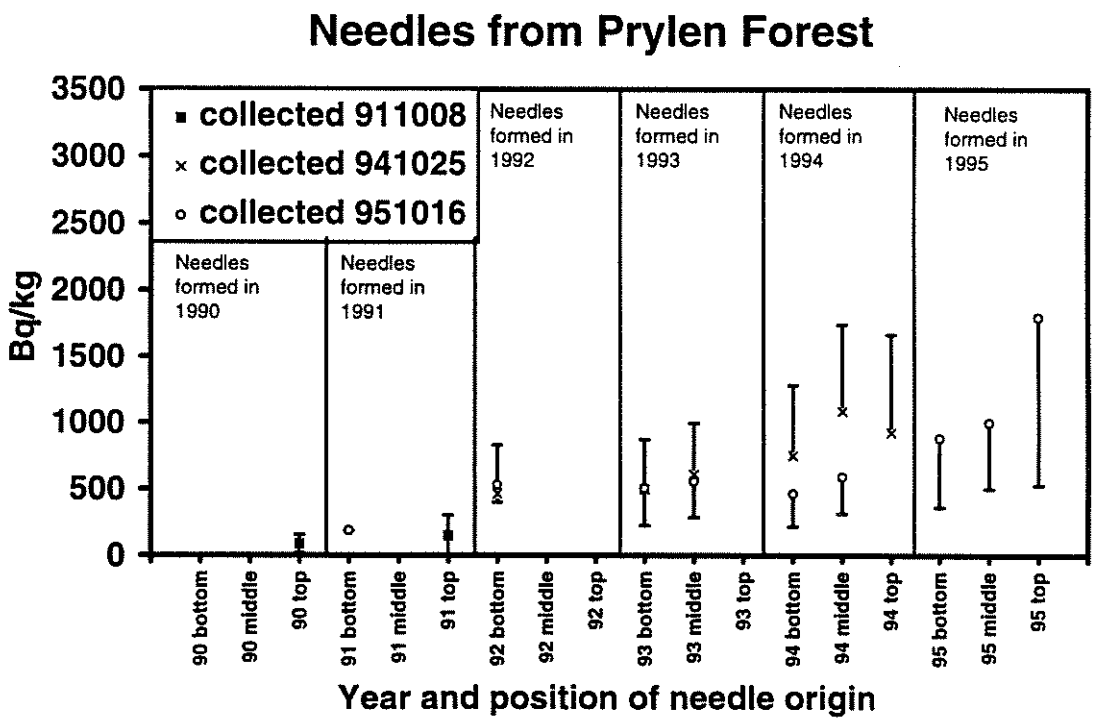

Figure 2.

The figures show the average concentration of Cs-137 in shoots and needles from 4 trees sampled in 1991, 1994 and 1995, respectively. During the four years between sampling, the biomass, roots included, has on the average increased 9 times, while the activity concentration has increased 5 times. The caesium concentrations are highest in the youngest needles, and increasing with each sampling time.

The data show a considerable uptake of radiocaesium by the actively growing tree. In the trees at site P, the activity concentration is higher than in the older trees at site H, shown in Figure 3. This is of special importance, as young pine trees comprise an important food for the moose especially during the late autumn and winter, and therefore can lead to substantial caesium concentrations in moose meat. Normally, however, the hunting season in Sweden is before the moose changes its diet and increases the intake of pine, but this has to be further studied.

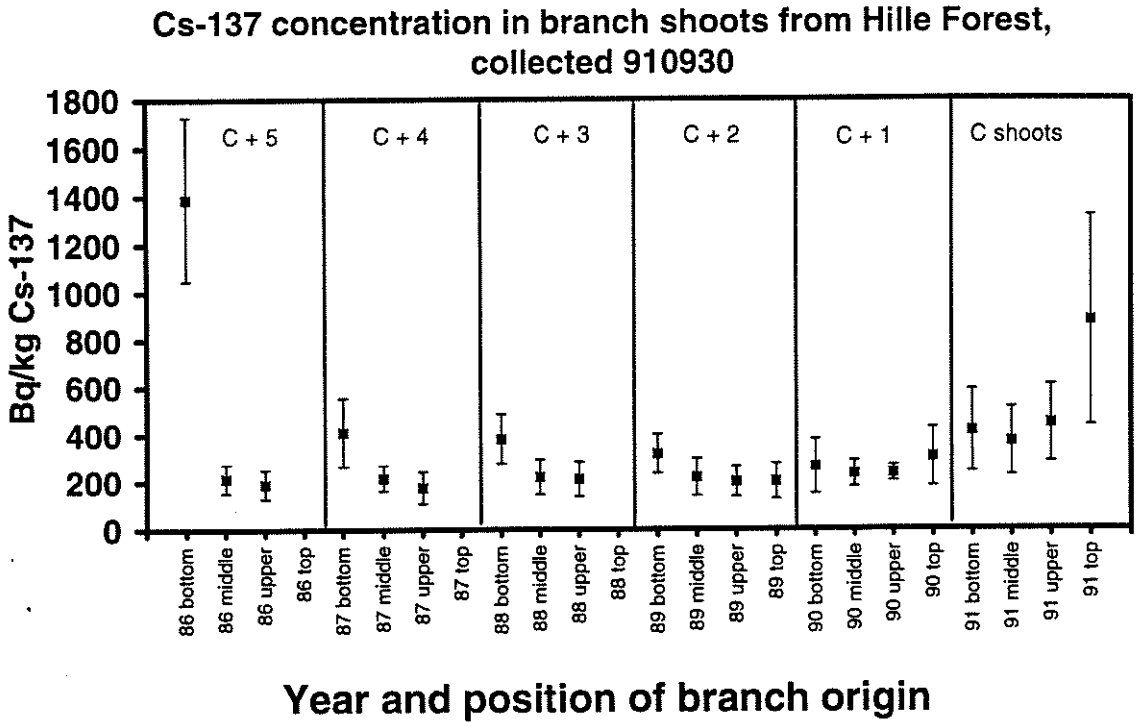

Figure 3.

Direct contamination versus root uptake. Figure 3 shows the Cs-137 concentration in shoots from site H collected in 1991 when the trees were 61 years old. Four trees were felled and sampled and each point in the figure is an average of the four trees shown with standard deviation. The highest value is for the oldest shoots at the bottom of the tree from 1986, the year of Chernobyl deposition. Otherwise there is a pattern within each years grouping, which is pronounced in the older shoots, of higher concentrations at the bottom of the tree. The data shows a tendency towards increasing Cs-137 concentration in the younger shoots, with a maximum in the new shoots at the top of the crown. A similar pattern is seen in the needles.

Melin et al (1994) determined the concentration of Cs-137 in different fractions of Scots pine on samples taken in 1980, i.e., those trees were only exposed to the fallout from the nuclear bomb test explosions in the atmosphere during the 1960's. One of the sites (site A in Melin et at, 1994) is comparable to site H in this study concerning the age of the trees, the type of forest, soil characteristics and climate. For both sites, the concentrations in the current needles and shoots are higher than in the older ones.

Increasing Cs-137 activities in the youngest pine needles have also been observed by other groups, for example in Finland (Raitio and Rantavaara 1994). However, they presumed that the proportion of radiocaesium of soil origin was still very low in conifer needles up to 1991, and that the increase in radiocaesium was due to translocation of caesium from the bark or the older needles to the growing needles during spring. This can be compared to what is reported by Erkel and Ziegler (1991) who showed that only about 5% of the radiocaesium in the higher contaminated shoots of larch originated from root uptake 2.5 years after the accident.

On the other hand, Tikhomirov and Shcheglov (1994) describe the dynamics of radionuclides in the forest based on studies after the Kyshtym and Chernobyl accidents, and show a

decreasing importance of the initially intercepted radioactivity as a source for contamination of different stand components and the increasing importance of root uptake with time. After a definite time, most of the radionuclides have migrated from the canopy into the forest litter. The first phase is 2-4 years and does not depend on soil properties. After 10-15 years an equilibrium state is reached, and the throughfall losses are being compensated by root uptake. The time to reach equilibrium is dependent on the biological availability of the radionuclide.

The distribution of Cs-137 in the older trees at site H, Figure 3, does not allow a distinction between root uptake and translocation, although it has a similar pattern to the other observations. The evidence discussed above from site P helps clarify our understanding. The data presented here strongly support root uptake as the dominant mechanism increasing the activity concentration in new growth, but assistance by translocation from older needles before they fall off may also be important. The significance of this contribution will be clarified in ongoing experiments.

Modelling of the forest ecosystem A dynamic compartment model for evaluating radionuclide distribution in forests has been proposed by Schell, *et al*, (1996), and is called FORESTPATH. One important contribution of that work is the literature review of half-times for radionuclides in forest compartments and a best guess for generic values plus literature ranges for model coefficients for both coniferous and deciduous forests. The model itself has 6 compartments including the tree, understory, organic layer, and the three soil compartments: labile, fixed, and deep soil, giving a macroscopic view of radionuclide cycling in the forest. We are using FORESTPATH as a starting point for incorporating the tree dynamics obtained from the current data, dividing the tree into more than just a single compartment.

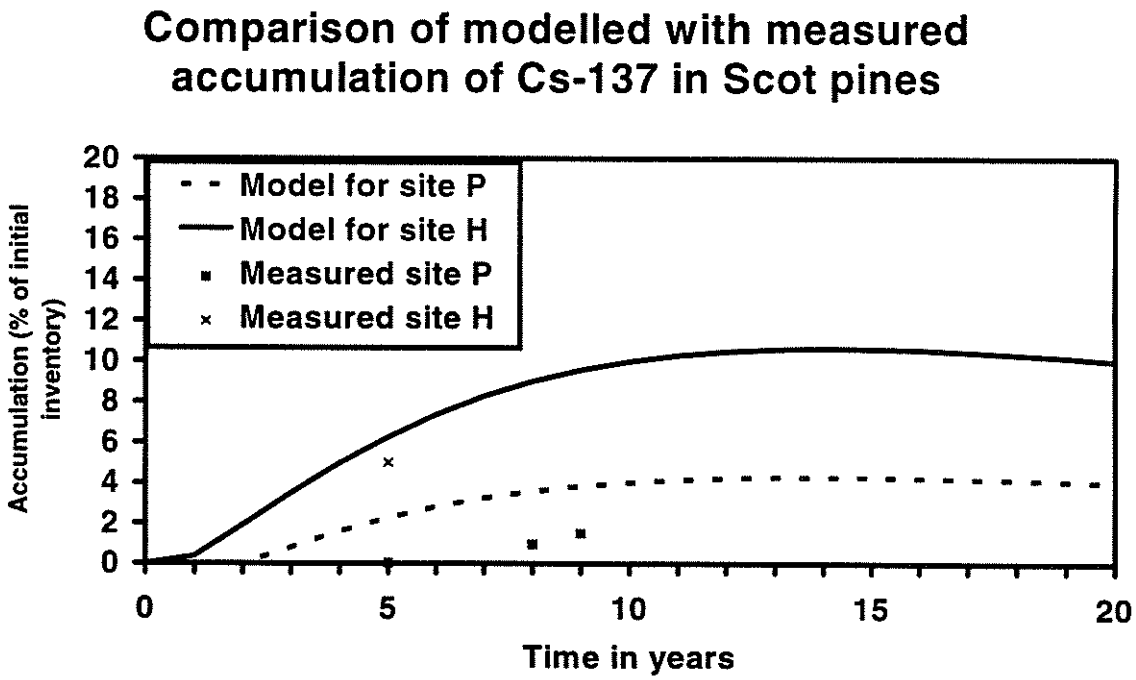

Figure 4.

For the purpose of this study we have checked the validity of the original model with data collected from site P and H. Fig. 4 shows the percent accumulation of Cs-137 in the total pine tree compartment for input data relevant to site P: no trees present until 2 years after initial deposition, and all available deposition occuring on the understory and soil. The model prediction is the dotted curve and the data collected from site P are the three points. The measured averaged total activity in the trees at site P has increased by a factor of 35 between

1991 and 1995. This can only be explained by root uptake. The model overpredicts the tree accumulation, one likely reason is because the model does not distinguish between the age of the trees. A similar model run for initial conditions appropriate to site H, which consists of 61 year old trees, is also shown in the graph. The magnitude of the single point from the measurements performed at site H in 1991 is in reasonable agreement with the magnitude of the modelled curve at that time point.

Conclusions

The concentration of Cs-137 in the new growth of Scots pine trees planted two years after the Chernobyl accident (site P) continues to increase, as of 1995. The averaged total activity in the trees at site P has increased by a factor of 35 between 1991 and 1995. These two points, along with the lack of dynamical change in Cs-137 concentration in needles and shoots, implies that the transport of Cs to new growth comes primarily from root uptake, but the statistics do not rule out some contribution from translocation.

Acknowledgement

This research was partly supported by the European Commission Contract No. FI3PCT920050, for which we are grateful.

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Nordic Society for Radiation Protection, 11th Regular Meeting, Reykjavik, Iceland, 26-29 August 1996

En enklare metod att analysera strontium-90 testad på jordprover från Tjernobyl-området

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Sammanfattning: Vid SSI:s radiokemiska laboratorium utförs regelbundet analyser av förekomsten av strontium-90 i olika provslag. Omgivningsprover såsom jord, gräs, träd, alger och sediment samt mjölk är de vanligaste provmatriserna. Dessa analyser har hittills baserats på den så kallade HDEHP-metoden, där yttrium-90 separeras från Sr-90 genom extraktion med komplexbildaren HDEHP (di(2-etyl-hexyl)fosforsyra). I syfte att förenkla och samtidigt minska tiden för den radiokemiska behandlingen av det studerade provet har en ny metod för analys av Sr-90 utvecklats. Förbehandlingen av proverna är densamma som vid gängse analysmetoder dvs våt- eller torrinaskning. Genom oxalat och hydroxidfällningar separeras yttriet. Liksom för HDEHP-metoden bestäms aktiviteten Y-90 genom mätning av Cerenkovstrålning i en vätskescintillationsräknare. Metoden har testats på jordprofiler från tremilszonen runt den havererade Tjernobylreaktorn. Analysresultaten för de bägge metoderna visar god överensstämmelse. Den förenklade metoden är kostnadseffektiv jämfört med tidigare metod och tidåtgången för den radiokemiska behandlingen är mindre än hälften av den för HDEHP-metoden.

Inledning

Strontium-90 är en fissionsprodukt som bildas vid klyvning av uran. En klyvning av uran sker både vid kärnvapenexplosioner och i härden vid drift av kärnkraftverk. Sr-90 spreds globalt över hela jorden under den tid atmosfäriska kärnvapenprov utfördes. Vi haveriet av reaktorn i Tjernobyl skedde också utsläpp av Sr-90 till omgivningen, men den större delen av strontiet stannade i närområdet till reaktorn.

Vid SSI:s radiokemiska laboratorium utförs regelbundet analyser av förekomsten av Sr-90 i olika provslag. Omgivningsprover såsom jord, gräs, träd, alger och sediment samt mjölk är de

vanligaste provmatriserna. Dessa analyser har hittills baserats på den så kallade HDEHP-metoden. I denna rapport beskrivs en ny metod för analys av Sr-90 som utvecklats i syfte att förenkla och samtidigt minska tiden för den radiokemiska behandlingen av det studerade provet. Mätningar på jordprover från Ukraina har utnyttjats för att jämföra metoderna.

Metodik

HDEHP-metoden

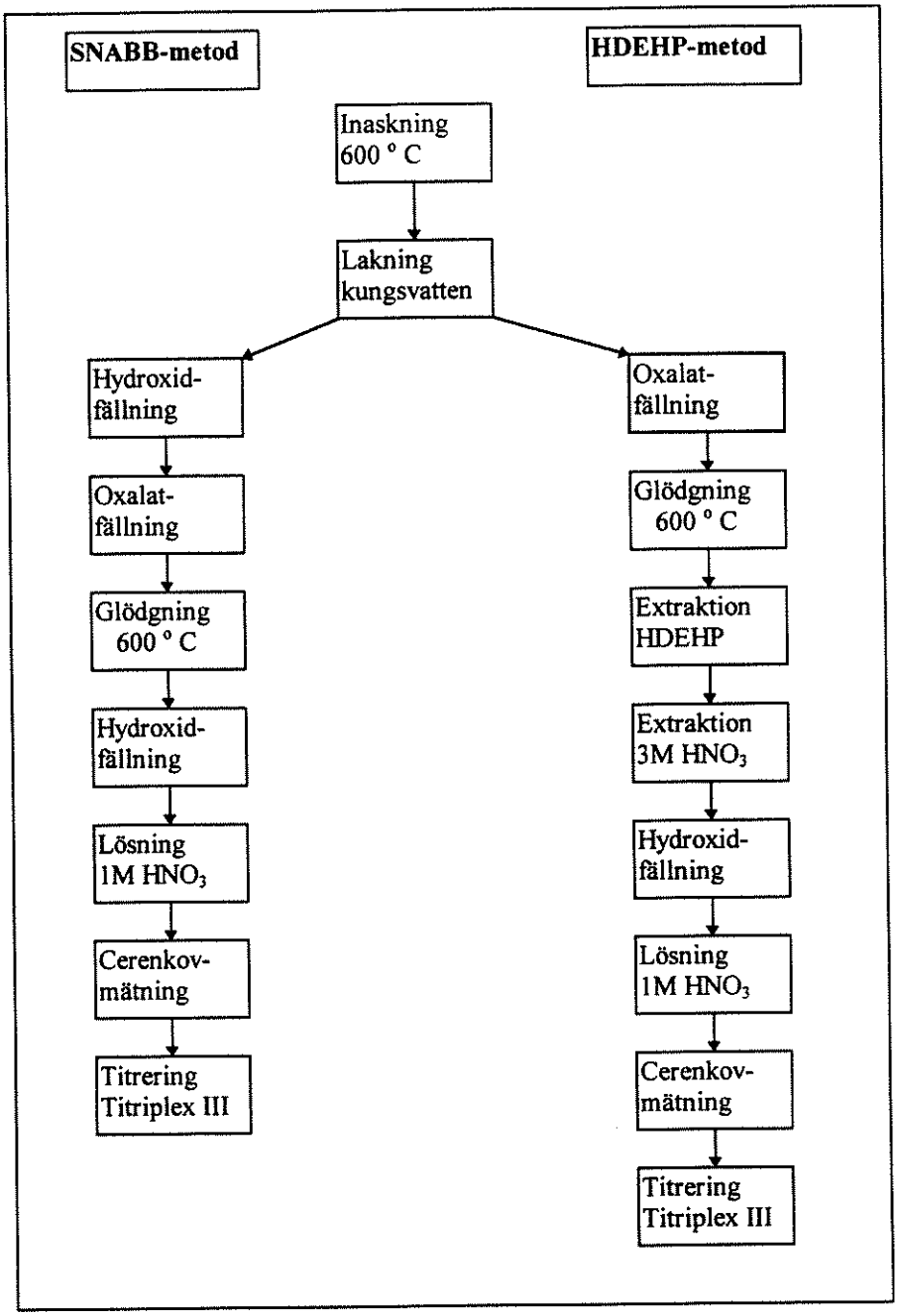
Principen för HDEHP-metoden är att HDEHP (di(2-etyl-hexyl)fosforsyra) bildar komplex med trevärda joner (Peppard et al, 1957). Jordproverna glödgas vid 600°C, glödgningsresterna lakas med kungsvatten och den olösliga inaskningsresten filtreras ifrån. Vid pH 1.0-1.2 extraheras yttriet från provlösningen med en tio procents lösning av HDEHP i toluen. Alla en och tvåvärda joner, såsom cesium, kalium, strontium och kalcium stannar kvar i vattenfasen. Yttriet återextraheras från HDEHP-fasen med tre molar salpetersyra och utfälls som hydroxid. Fällningen löses i en molar salpetersyra och lösningen förs över till en vätskescintillationsburk och Cerenkovstrålningen från Y-90 mäts i en vätskescintillationsräknare. Efter mätningen bestäms det kemiska utbytet genom titrering med Titriplex III (Suomela et al, 1993).

Analysmetoden fungerar utmärkt för de flesta provmatriser. Vid analys av mineraljordsprover innehållande aluminium och järn (Fe^{3+}). extraheras även dessa metaller av HDEHP. Vid stora mängder av dessa metalljoner blir HDEHP-fasen mättad och yttriet stannar i vattenfasen. Även mindre mängder aluminium och järn påverkar analysgången eftersom de följer yttriumfasen och stör den avslutande titreringen för bestämning av yttriumutbytet.

För att separera ifrån aluminium och järn utförs därför en oxalatfällning före extraktionen med HDEHP. Fällningen glödgas för att omvandla oxalatfällningen till en blandning av karbonater och oxider. Analysgången är därefter som beskrivits ovan.

Snabbmetoden

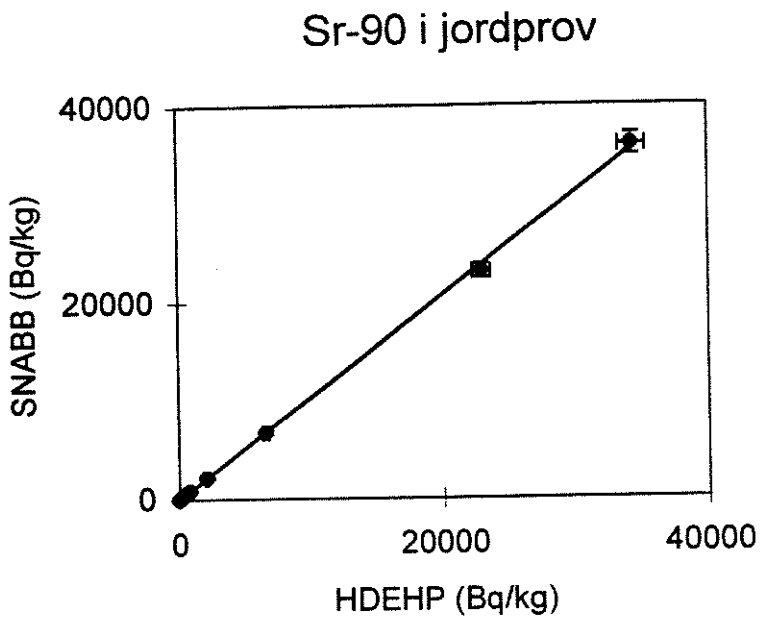
Den nu utvecklade "snabbmetoden" skiljer sig från HDEHP-metoden genom att HDEHP-extraktionen ersatts med en hydroxidfällning för att separera yttrium från strontium. Metoden utarbetades i samband med analyser av ett större antal jordprover från områden i Ukraina, Vitryssland och Ryssland som erhöll ett omfattande radioaktivt nedfall efter Tjernobylolyckan. Följande flödesshema visar stegen i de två metoderna.



Jämförelse mellan de två metoderna

Överensstämmelsen mellan de två metoderna för analys av Sr-90 har jämförts bland annat genom mätningar på jordprov tagna i två byar belägna inom tremilszonen runt den havererade reaktorn i Tjernobyl: Chistogalovka (sandig lerjord) och Kopachy (sandig podsol). Jordprofilerna, som var 25 cm långa, skiktades i 2 cm tjocklek ned till 10 cm och därefter i 5 cm skikt. För varje skikt har 2-4 mätningar genomförts varefter medelvärdet för dessa beräknats. Båda

metoderna baseras att jordproverna (7-10g) inaskas, lakas och filtreras. Provlösningen späds till 100 ml och delprover motsvarande 0.1 - 3 gram tas ut för analys i enlighet med flödesschemat ovan. I nedanstående figur visas sambandet mellan de två metoderna för Sr-90 aktiviteter upp till ca 36000 Bq/kg. I figuren ingår data (15 punkter) från båda provplatserna. En linjär anpassning ger ekvationen $y=1.04x-11.2$ (Bq/kg) ($R^2=1$, $p<0.05$). Felstaplarna anger det räknestatistiska felet.



Diskussion

För de prover som hittills undersökts är överensstämmelsen mellan de två metoderna god. Snabbmetoden är framförallt lämplig att använda då stora markytor skall mätas med avseende på Sr-90. Analysmetodiken är mycket enkel och noggrannheten i de flesta fall helt tillfredställande.

Den lägsta koncentrationen av Sr-90, 15 Bq/kg, uppmättes i jordprover från Kopachy på 15-20 cm djup. Den naturligt förekommande radionukliden Bi-210 kan förväntas ha ungefär samma aktivitetskoncentration. Vismut följer yttrium i analysgången vid snabbmetoden och har ett betasönderfall som ger upphov till cerenkovstrålning. I de betaspektra som analyserats har emellertid inte någon cerenkovstrålning från Bi-210 kunnat påvisas. Detta torde bero på att vismut inte frigjörs vid lakning med kungsvatten. Vid en fullständig uppslutning av jordprov

finns det risk att cerenkovstrålningen från Bi-210 vid låga aktivitetskoncentrationer påverkar analysresultaten vid snabbmetoden.

Den nya metoden innebär att det radiokemiska arbetet vid analys av Sr-90 kan minskas till mindre än hälften av den för HDEHP-metoden. Kostnaden minskar också till följd av att HDEHP inte behöver användas. Detta innebär också en miljövinst.

Fler tester kommer att genomföras för att ytterligare klargöra snabbmetodens för- och nackdelar.

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On the choice of mathematical transformations for visualising gamma-ray spectra

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Introduction

The human brain is very well adapted to receiving information in a graphical format. Checks can quickly be made for many different factors simultaneously and with much less effort than checking numbers. Comparable checks using numerical parameters would normally involve considerable mathematical analysis (and programming). Software for analysing gamma ray spectra from Ge detectors commonly offers the choice of viewing the spectra using either a linear or logarithmic transformation. Often it is not however systematically pointed out to the user how these transformations can be used effectively to investigate and check various properties of gamma ray spectra. Other transforms are usually also not offered.

In recent years powerful, and yet simple to use, software with flexible graphics possibilities has become common amongst users of personal computers. It has now become realistic for ordinary users to use some of this powerful software with minimum programming to view their own spectra. It is of course possible to take this approach further and use the methods mentioned when writing software for analysing gamma ray spectra. This application is however outside the scope of this paper. Using mathematical transformations in a systematic way to view gamma ray spectra can help to:

- give users a better understanding for its various properties
- provide a simple graphical tool for making quality checks

In this paper some properties of two commonly used transformations will be discussed, along with a third transformation which is not commonly used for transforming gamma ray spectra although it is often used in statistics. It is hoped that this text can be of some interest to the novice in gamma spectroscopy and possibly to others as well. The mathematics is deliberately kept on a fairly basic level and the main assumptions made are stated in the appendix. The approach presented in this paper is also valid for other types of spectra where the assumptions hold, e.g. in XAFS and syncrotron radiation induced Mössbauer spectroscopy.

Spectrum properties to be discussed

The following properties of spectra will be discussed and how these different transformations can assist the user to assess them in a spectrum:

- Spectrum shape
- Peak shape
- Relative peak size
- Level of significance of peaks

General description of the transformations used

Each transformation is given the form:

(1) $y_i = f(n_i)$ where n_i is the number of counts in channel number i

As a simplification the subscript i will be dropped

(2) $y = f(n)$

The standard deviation gives a measure of the expected spread of values. When viewing the spectrum one can also thinking of it as describing the inherit statistical "noise" in the spectrum. The standard deviation is then a measure of the bandwidth of the noise.

Assuming that the number of counts in each channel can be described by the Poisson distribution we get the following estimate for the standard deviation:

(3) $s_n = \sqrt{n}$

The standard deviation of y = f(n) can be found by:

(4) $s_{f(n)} = s_n \cdot \frac{df}{dn}$

Transformation discussed

In this paper we will discuss the three transformations mentioned:

(5) $y = n$ The *linear* transformation

(6) $y = \sqrt{n}$ The *square root* transformation

(7) $y = \ln(n)$ The *logarithmic* transformation

Each transformation could be written in a more general (and complex) form (e.g. including multiplication with a constant). We can however limit ourselves to these simple forms without loss of generality.

Let us first compare the basic properties of the transformations. The linear transformation considered leaves the argument (n) unchanged, the other two can be considered to offer various degrees of "compression" and the greater the argument the greater the compression. The degree of compression is lower for the square root transformation than for the logarithmic one. This property can be seen in the following table which lists the derivatives of each transformation function with respect to n. The table also lists the standard deviation of the transformed number of counts for each transformation. The standard deviation is computed by taking the derivative and putting it into equation (4) and using the standard deviation of number of counts from equation (3).

Table 1 Derivatives of transformation functions and standard deviation of transformed number of counts

Transformation used	Derivative of transformation function with respect to n	Standard deviation of transformed number of counts, s_y
Linear $(y = n)$	1	$\sqrt{n}$
Square root $(y = \sqrt{n})$	$\frac{1}{2} \cdot \frac{1}{\sqrt{n}}$	$\frac{1}{2}$
Logarithmic $(y = \ln(n))$	$\frac{1}{n}$	$\frac{1}{\sqrt{n}}$

Table 1 clearly shows how the square root transformation can be considered to lay between the other two. This will also be clearly seen in the examples that follow.

Spectrum shape
"Spectrum shape" refers here to the overall appearance of a spectrum. It is a good practice to know beforehand what type of spectrum shape to expect from a given sample. If something unusual shows up in the spectrum it might then not go unnoticed, as could be the case if the peak(s) in question did not belong to one of the nuclides to be analysed for. It can also help to eliminate human errors such as mixing up samples.

When assessing the spectrum shape visually, it is preferable that the spectrum plot should not be too dependant upon the activity in the sample or the counting time. It should however give an indication of the relative amounts of different radionuclides.

The logarithmic transformation has a clear advantage over the other transformations when judging a spectrum's shape. If the number of counts in all channels of a spectrum is increased in the same proportion it will show on a logarithmic graph as a lifting of the whole spectrum but without changing its shape. The shape of the spectrum will however change if the other transformations are used. The logarithmic transformation will thus give the same spectrum shape irrespective of total activity or counting time. The following conditions must be met if this is to be exactly the case, but some variations from these conditions will usually not change the spectrum drastically:

- The relative amounts of radionuclides present in the samples is the same.
- The sample geometries and densities are the same.
- The peaks of interest are not found in the background spectrum at the laboratory or are relatively small there

The logarithmic transformation offers one important advantage for assessing the spectrum shape.

Peak shape
The peaks considered here have a Gaussian shape and are sitting on a background continuum. Changes in the number of counts will change the height of the peak but not its width. The *logarithmic transformation* has the important property that it transforms a Gaussian peak to a convex *parabola*. Gaussian peaks with a fixed width are transformed to parabolas with exactly the same shape, irrespective of the size of the Gaussian peak. This makes it very easy to check using the logarithmic transformation if a certain peak has a plausible form. One can even have a reference peak on a transparency and lay it over the peak in question. One must however take care not to compare peaks of different energy in this manner, since the peak width increases with energy in a gamma spectrum.

With the *linear* and *square root* transformations the curves will stretch vertically with increased number of counts and thus change shape.

Relative peak size
The *linear transformation* is the only transformation which allows one to compare and assess the relative sizes of peaks, irrespective of whether they are sitting on a low or high background continuum. Direct comparisons can e.g. thus be made of the activity

of two samples, if other parameters are the same (counting times, sample geometry, density).

Level of significance of peaks

Let us consider the case of trying to judge visually whether a weak peak is present or not. The peak has to rise significantly above the "noise" of the background continuum. It can therefore be of help to use a transformation which keeps the noise level at a fixed constant rate. In table 1 it was shown that the *square root transformation* has this property. Using that transformation the standard deviation of the transformed number of counts is the same over the whole spectrum, irrespective of the number of counts in the channels. This property is clearly seen if one looks at the bandwidth of the "noise" in a spectrum using the different transformations. As the peaks grow higher above the background continuum the square root transformation still offers the best measure of their significance.

Conclusion

Each of the three transformation discussed has advantages over the others depending on the task. All of them deserve a place in the spectroscopist's toolbox. Table 2 summarises the conclusions of this paper concerning preferred transformation for different tasks, using the assumptions given.

Table 2 Preferred transformations for assessing visually different properties of gamma spectra.

Property of spectrum to be assessed	Preferred transformation
Spectrum shape	Logarithmic
Peak shape	Logarithmic
Relative peak size	Linear
Peak significance	Square root

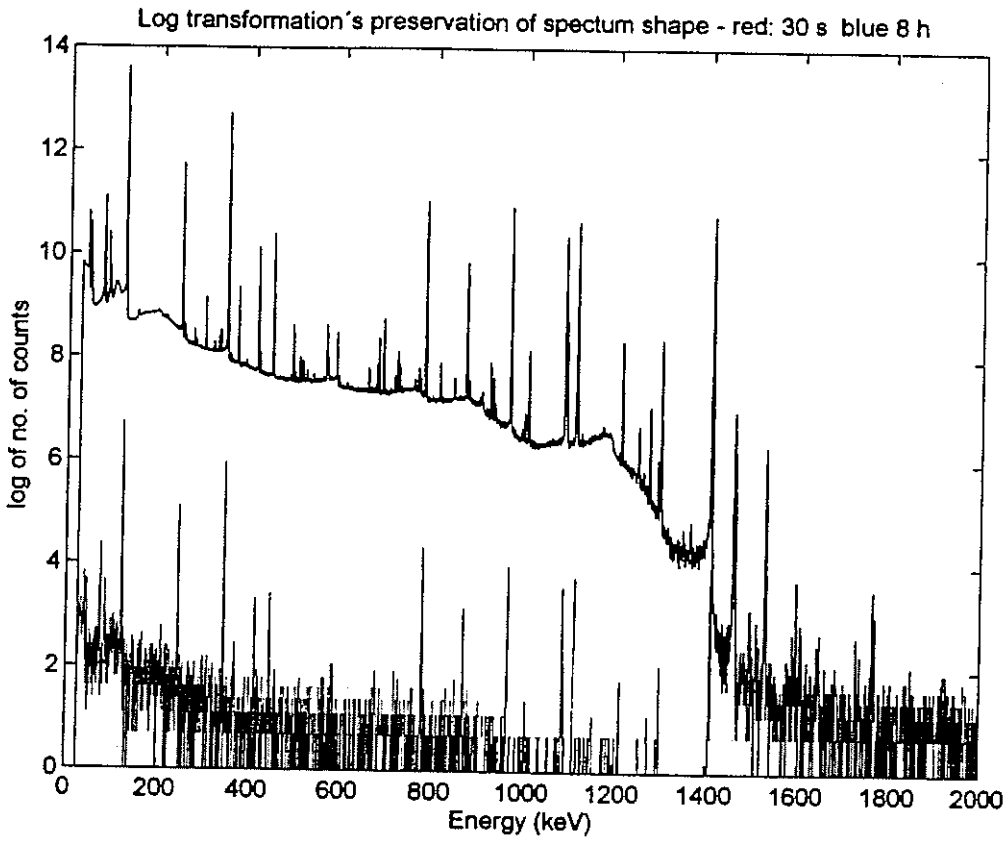
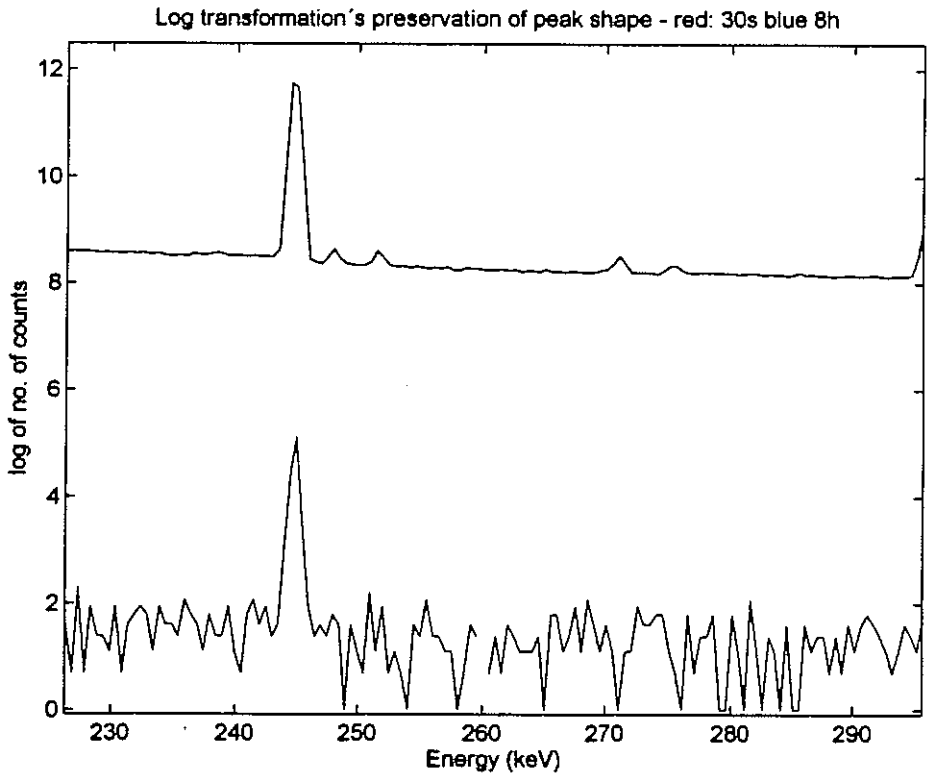


Figure 1
The log transformation's preservation of spectrum shape. The graph shows 2 spectra obtained using the same Eu-152 source, one by conting for 30 s the other by counting for 8 h.



Figur 2
The log transformation's preservation of peak shape. The graph shows a close up of an energy region in the same 2 spectra as in figure 1. Although one of the peak has poor statistcs they both show the same parabolic form and width.

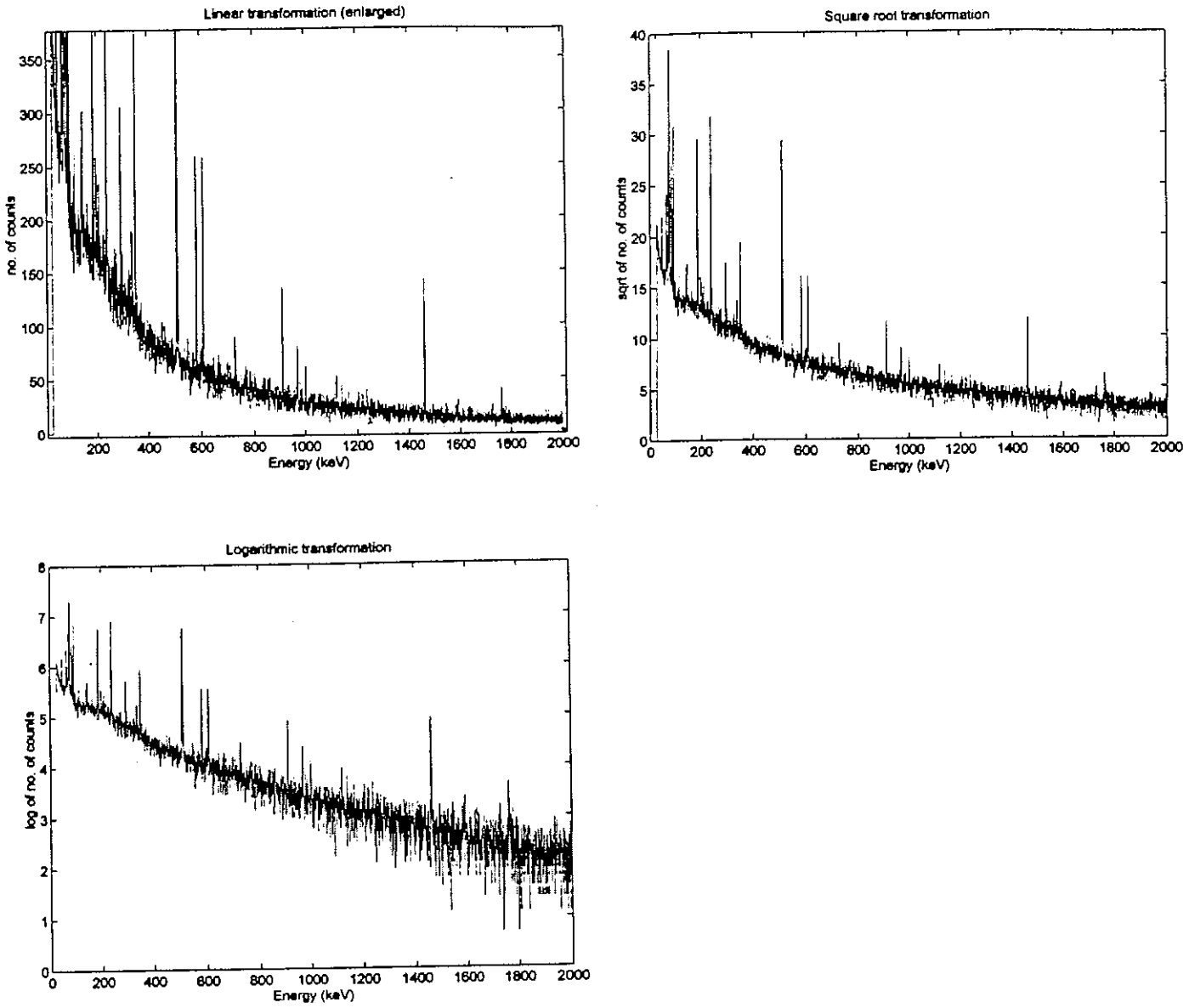


Figure 3
Shows 3 graphs of the same background spectrum using the three transformations discussed in the paper. The 3 graphs clearly demostrate the effect of each transformation upon the statistical "noise" (the bandwidth of the noise being constant in case of the square root transformation).

Appendix

The following assumptions have been made in this paper:

- The number of counts in each channel of the spectrum is governed by the Poisson statistical distribution. The number n used in the text refers actually to the *expected* number of counts in each channel, not the *observed* one.
- The shape of the peaks of interest in the spectra can be described by a Gaussian statistical distribution. In gamma spectra it is common to have some tailing effects at the bottom of the peak, but the basic shape of the peak is still Gaussian.

Radioactivity levels in foodstuffs in Finnish Lapland

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Abstract - The radioactive fallout in Finnish Lapland from the nuclear reactor accident in Chernobyl in 1986 was low, for Cs-137 in average 1000 Bqm^2. The enrichment of radionuclides in the special foodchains of Lapland is however very effective. After the accident the highest concentrations of Cs-137 were noticed in reindeer meat. The average concentration increased from 300 $Bqkg^{-1}$ in the slaughtering season 1985/1986 to 720 $Bqkg^{-1}$ in 1986-1987. A slight increase was also measured in fresh water fish, mushrooms and wild berries, but hardly any effects were noticed in milk, garden products and game. The type of soil and other environmental parameters or cultivation practise had higher influence than the low level of fallout.

In 1994/1995 the average Cs-137 concentrations of all foodstuffs produced in Finnish Lapland had decreased to the same level or even below that measured before the accident. Reindeer meat contained in average 300 $Bqkg^{-1}$, predatory fish 50, other fish species 10 - 25 but salmon from Tana river only below 1 $Bqkg^{-1}$. In mushrooms the Cs-137 levels varied from 120 in Lactarus and 75 in Russula and Suillus, to 45 $Bqkg^{-1}$ in Boletus species. Cloudberry contained 25, blueberry 10 and lingonberry 6 $Bqkg^{-1}$, elk 20 and ptarmigan 10 $Bqkg^{-1}$, root crops 0 Bqkg,$^{-1}$ cow 's milk 0.6 Bql^{-1} Cs-137. The Sr-90 concentration in cow's milk was 0.06 Bql^{-1} Sr-90. All results are presented per fresh weight.

Elevated internal radiation doses were caused in Lapland by the radiocesium in reindeer meat. Common for all Finland is that foodstuffs from nature are the dominating sources. The estimated mean internal radiation dose to the Finnish population is 0.7 mSv and to reindeer herders below 5 mSv for the the 50 year period 1986-2036 after the Chernobyl accident.

Introduction

The enrichment of radionuclides in the special foodchains of Lapland is very effective. After the nuclear reactor accident in Chernobyl in 1986 the deposition of radionuclides was smaller in Lapland than in other parts of Finland, for Cs-137 on the average 1000 Bqm^{-2} with a variation from 200 to 2000 Bqm^{-2}. After the accident an extensive sampling campaign of the locally produced or from nature collected foodstuffs was started in Finnish Lapland. Gammaspectrometric measurements of the collected samples were done. Radiochemical Sr-90 analyses have so far been performed only for milk samples. Milk samples were collected also earlier and samples of reindeer, game, fish and berries occasionally (1).

The sampling and the measurements were carried out by the Regional Laboratory in Northern Finland of the Finnish Centre for Radiation and Nuclear Safety. The laboratory has operated at Rovaniemi since 1970 and one of its main tasks is to follow the radioactivity levels of the locally produced or from nature collected foodstuffs.

Sampling, sample treatment and analyse methods

The laboratory has created a wide network with regular sample senders. Farmers, fishermen, hunters, reindeer herders, reindeer co-operatives, waterpower companies, biological and agricultural research stations and reindeer, game, fish and forest researchers, veterinarians and private persons are sending samples annually or when requested. The staff of the laboratory is catching fish from one lake near Rovaniemi and also picking berries and mushrooms.

Milk samples are obtained monthly from one farmer and twice per year from 6 other farms. The fresh milk is measured gammaspectrometrically in a Marinelli-geometry and then, evaporated to dryness under infra-red thermal lamps and ashed at 450°C before strontium analyse.

Root crops, potatoes, carrots, beetroots, turnips, sweeds, onions and herbs are obtained ammually from two farms. The vegetables are washed, peeled, sliced, dried at 105°C and homogenized before measurements.

Fish samples are at present obtained regularly from 15 lakes and 2 rivers. During the first year after the accident samples from 48 lakes and 4 rivers were measured. The weight and length of the fish samples are registered, the scales and gut are removed and the fish flesh is separeted from the bones, dried and measured. The flesh of larger fish is measured fresh, smaller samples are dried.

Reindeer samples were 1986 - 1990 received from the whole reindeer herding area via the companies buying the carcasses from the reindeer herders. Thereafter samples were obtained directly from the reindeer co-operatives, annually from the 10 northernmost Sami co-operatives and from the southernmost Halla co-operative. At present samples are requested from all the 56 co-operatives only every fifth year. Reindeer meat sample is usually taken from the neck muscle. All sinews and membranes are removed and the samples are dried, homogenized and measured.
Berries and mushrooms are obtained from several locations in Lapland but due to the weather conditions the availability varies from year to year. All litter and the lower part of the stem of mushrooms soiled with earth is removed. The samples are dried and homogenized before measurement.

Samples of game are analysed occasionally. Brown bear samples are obtained from veterinarians inspecting the carcass for the detection of trichinae or from illegally hunted bears taken into custody by police. Elks samples are obtained from animals killed in traffic accidents or from hunters. Ptarmigans are bought from hunters in Utsjoki. Other birds and hare are obtained only sporadically. All samples are cleaned, dried and homogenized before measurement.

Gammaspectrometric measurements are performed with high purity Ge detectors using either a 500 ml Marinelli geometry or the samples are packed into 35 ml cylindrical containers (1).

The radiochemical strontium-90 analyse of the milk samples is performed from an ashed sample according to the method of Bryant, Morgan and Spicer (2) and the concentrations are measured with a 5-channel low-level beta multicounter system CM-25-5 produced by Risø National Laboratory, Denmark.

Results

Only Cs-137 and Cs-134 were detected in gammaspectrometric measurements of the foodstuffs. Concentrations of Cs-137 in different foodstuffs during the periods 1986/1987, 1999/1991 and 1994/1995 are presented in Table 1.

Table 1. Cs-137 concentrations (Bq/kg^{-1} fresh weight) in foodstuffs produced in Finnish Lapland

Foodstuff	1986/1987	1990/1991	1994/1995
MEAT			
reindeer	700	400	300
cow	< 30	< 20	< 10
pork	10	< 10	< 3
FISH *			
perch	200	110	50
pike	120	120	50
burbot	90	180	63
whitefish	60	35	25
vendace	50	30	10
roach	60	40	20
arctic char	90	45	50
salmon (Tana river)	2	0.7	0.6
salmon (Tornio river)			38
GAME			
brown bear	100	150	100
elk	40	20	20
ptarmigan (Utsjoki)	35	20	10
MILK	1.8	0.6	0.6
ROOT CROPS	0.8	0	0
WILD BERRIES			
cloudberry	75	35	25
blueberry	35	10	15
lingonberry	10	6	6
MUSHROOMS			
Lactarus	140	180	120
Russula	80	100	70
Suillus, Boletus	50	85	45

* (average conc. of fish caught in several lakes)

Because we wanted to compare the effects of Chernobyl fallout to those from the atmospheric nuclear weapons tests in the 50's and 60's, radiocesium concentrations in reindeer meat before 1975 (3) are also presented in Fig.1, and the fish results from Lake Inarinjärvi 1962 -1969 (4, 5) in Fig.2.

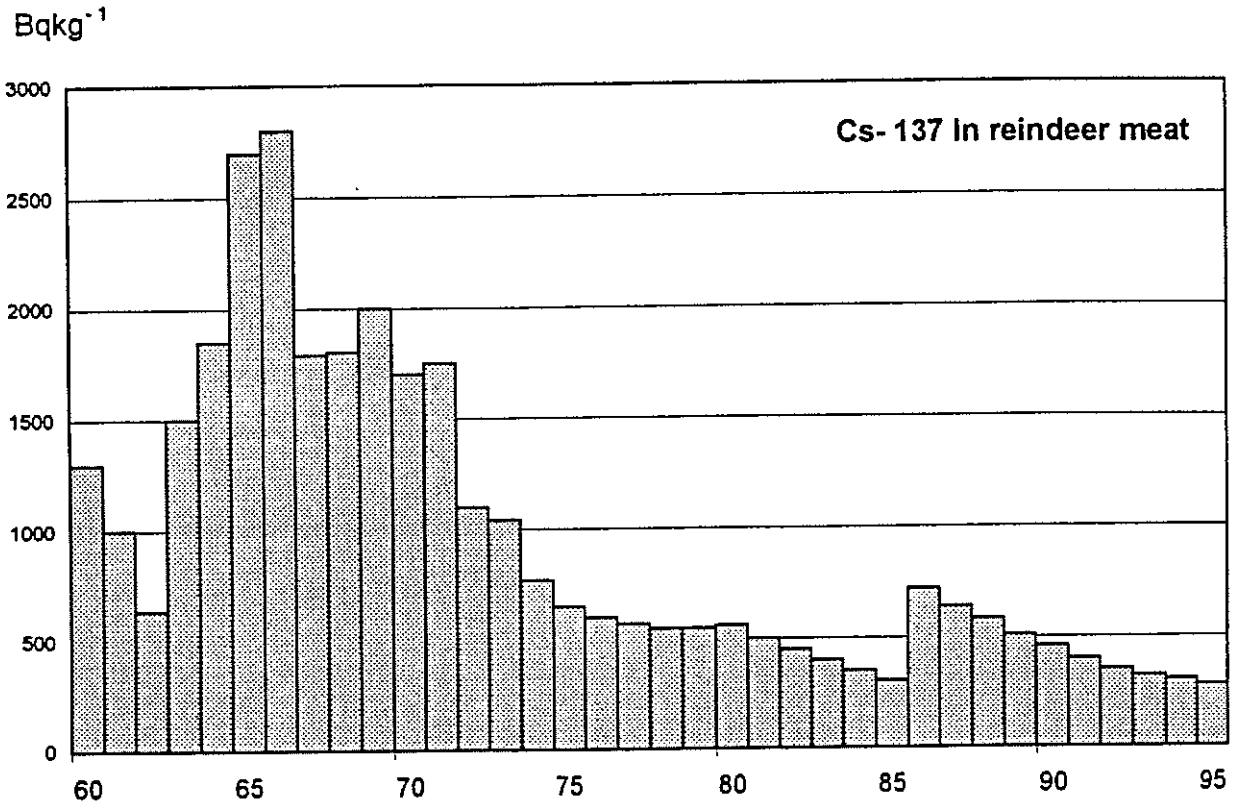

Fig. 1. Cs- 137 concentrations, Bqkg^{-1}f.w., in reindeer meat 1960 - 1995

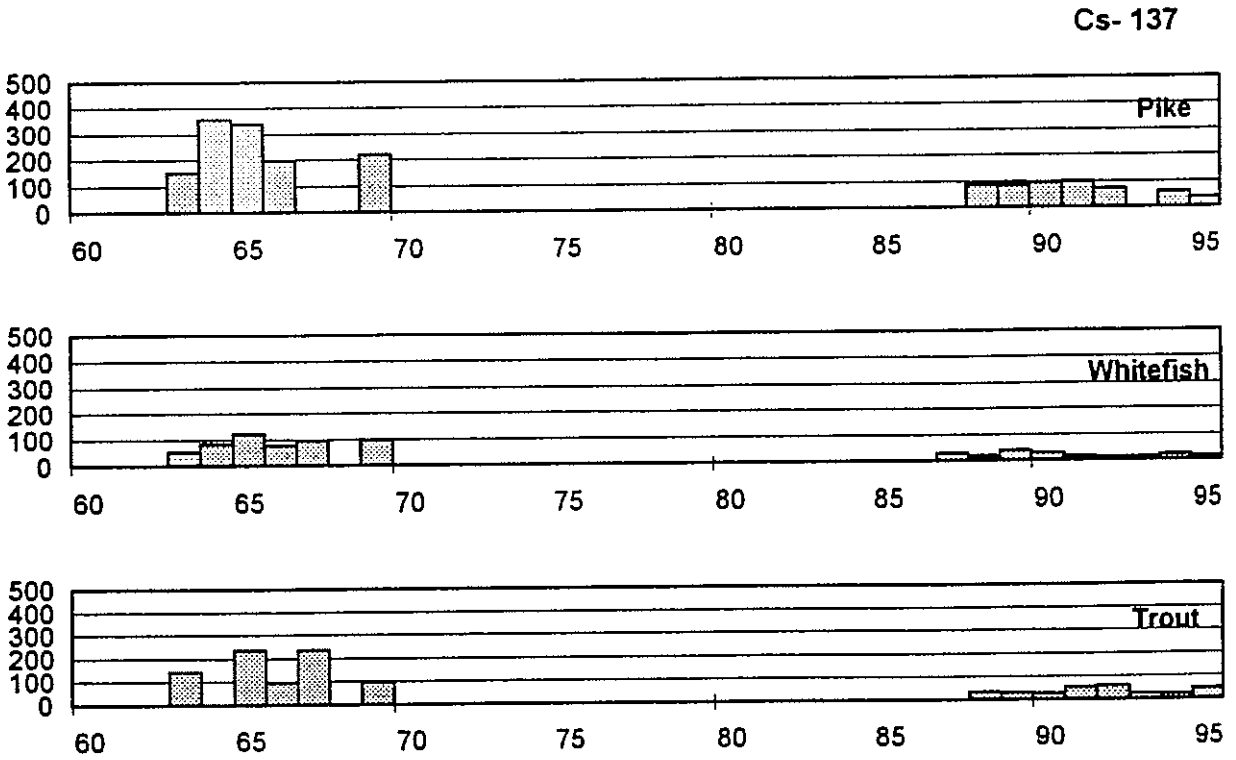

Fig. 2. Cs- 137 concentrations, Bqkg^{-1} f.w., in flesh of pike, whitefish and trout from lake Inarinjärvi 1963 - 1995.

As in the 60's the most significant increase in radiocesium concentrations after the Chernobyl accident was noticed in reindeer meat. But the average Cs-137 concentration increased in the slaughtering season following the accident only from 300 Bq/kg^{-1} in 1985/1986 to 720 Bq/kg^{-1} 1986/-1987 (6), when the highest concentration in 1966 was 2800 Bq/kg^{-1}. In 1995/1996 the cesium concentrations had decreased to the same level as before the accident.

Cs-137 ja Sr-90 concentrations measured in milk produced in Apukka (1971-1991) and in Vikajärvi (1991-1995), Rovaniemi rural commune are presented in Fig.3. The very low Sr-90 concentrations, 0.06 Bq/l^{-1}, in milk samples (Fig.3) and similar results from 6 other farms show that Lapland did not receive any detectable radiostrontium fallout from the Chernobyl accident.

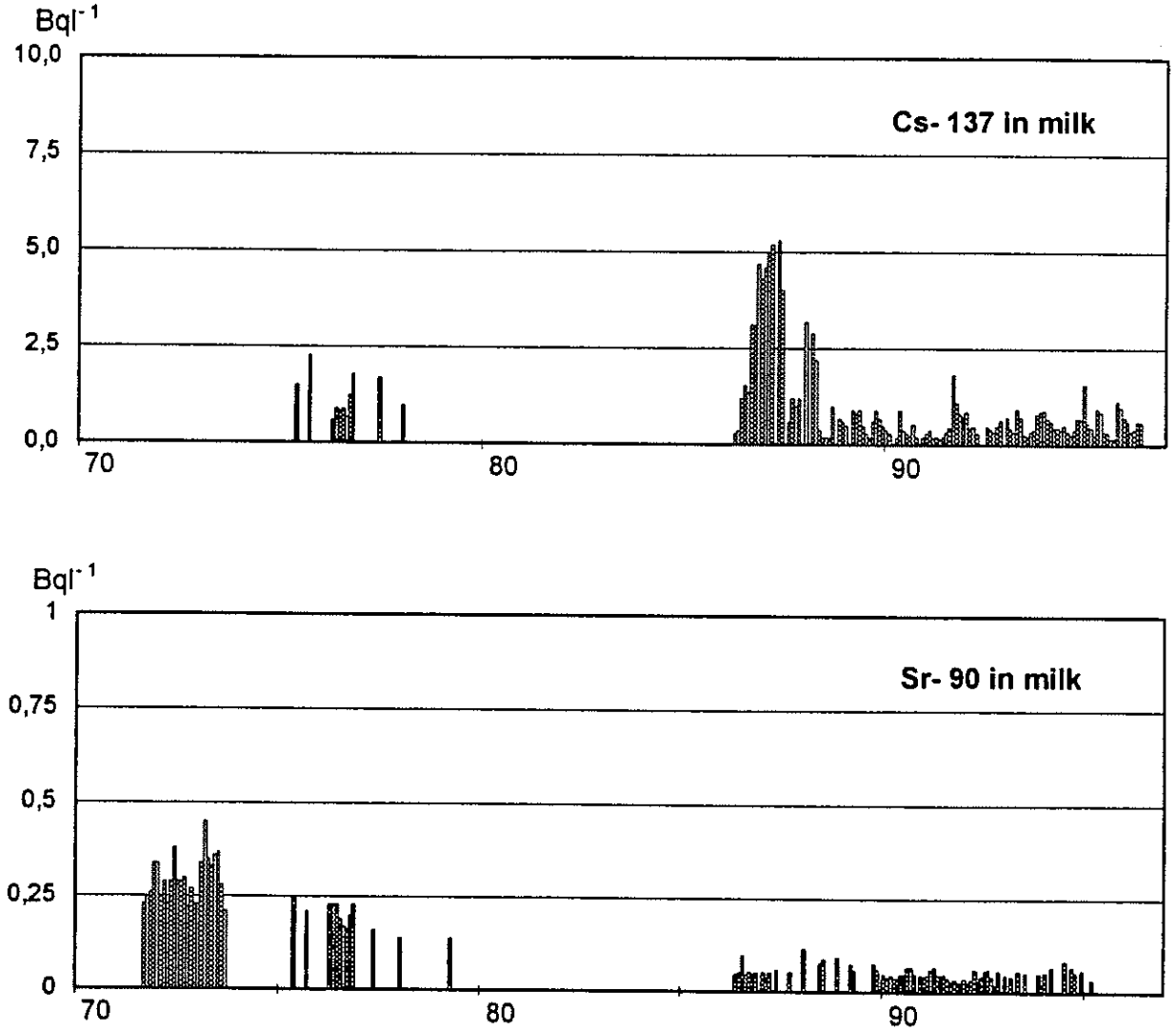

Fig. 3. Cs- 137 and Sr- 90 concentrations Bql^{-1} in milk from Rovaniemi rural commune, Apukka 1971 - 1991, Pöykkö 1991- >.

Cs-137 and Cs-134 concentrations in flesh of perch, pike and roach from lake Apukkajärvi, Rovaniemi 1982 - 1996 are presented in Fig.4.

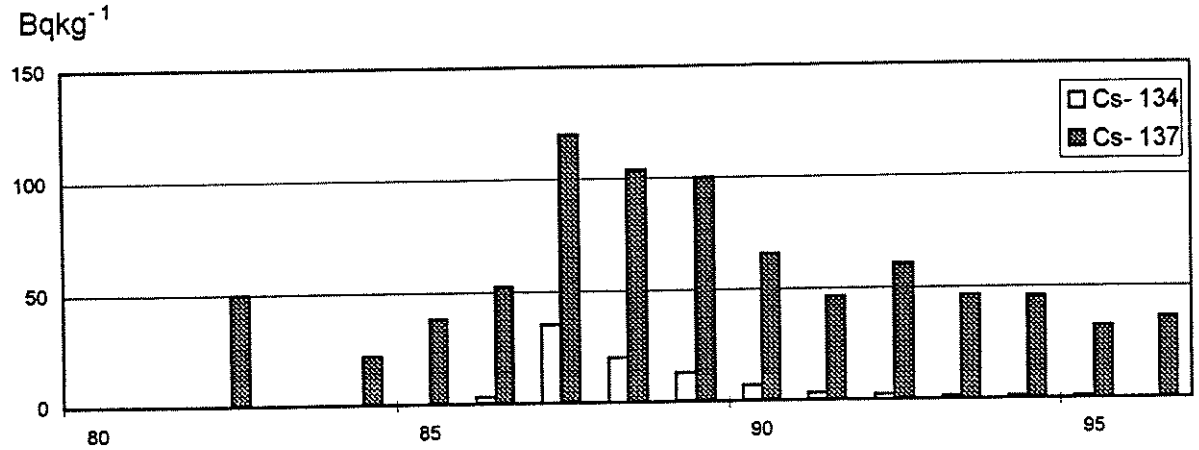

Fig. 4. Cs- 134 and Cs- 137 concentrations, $Bqkg^{-1}$, f.w., in flesh of pike from Lake Apukkajärvi, Rovaniemi rural commune 1982 - 1996.

Discussion

Except for the increase of radiocesium concentration in reindeer meat a slight increase was also measured in fresh water fish, mushrooms and wild berries, but the accident had hardly any effect on the cesium-137 concentration of milk, garden products and game (elk, birds). The type of soil, lake and other environmental parameters or cultivation practise, eg.fertilization, had higher influence than the low level of fallout. All these products contained also before the Chernobyl accident variable amounts of radiocesium originating from the nuclear weapons test in the 50's and 60's.

Today the average cesium-137 concentrations of all foodstuffs produced in Finnish Lapland have decreased to the same level or even below that measured before the Chernobyl accident.

The terrestrial environment of Northern Finland, Norway and Sweden and the Kola peninsula is very vulnerable regarding severe nuclear accidents because of the abundance of nutrient deficient soils and bog areas and of the special foodchains in these regions. In this kind of environment the foodstuffs accumulate much higher radioactivity concentrations with longer halflives of radionuclides than in southern Finland with other types of agricultural soils.

In regions with higher Cs-137 deposition than in Lapland especially radiocesium in fresh water fish (7) caused elevated internal radiation doses during the years after the accident. Compared with radiocesium the addition from radionuclides with shorter halflives was small. In Lapland it was reindeer meat that caused the most of the radiation doses. Common for all Finland is that foodstuffs from nature now are the dominating sources. The estimated mean internal radiation dose from Cs-137 to the Finnish population is 0.7 mSv for 1986-2036. The corresponding dose for reindeer herders is below 5 mSv. These doses are estimated using whole-body counting measurements. The total annual radiation dose in Finland is 4mSv. Thus the reindeer herders receive about the same internal dose from radiocesium in the 50 year period that Finnish people on the average every year.

Only reindeer meat is produced in Lapland in such quantities that it is of importance from radiation protection point of view in a fallout situation and even then the importance is biggest for the reindeer herders with a high consumption of reindeer meat.

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Thirty–nine Years of Environmental Monitoring at Studsvik–Part 2

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Abstract – Environmental monitoring of radionuclides has been performed since 1957 in the scenic surroundings of Studsvik 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 materials testing reactor (MTR) and one 1 MW low–power pool reactor are used. In the R2–reactor testing of nuclear reactor fuel, production of radioisotopes for medical purposes and neutron transmutation doping of silicon are performed. Plans for Boron Neutron Capture Therapy, BNCT, for medical treatment at the R2–0–reactor have recently been discussed. University of Uppsala has a department of neutron research at the reactor. Other types of nuclear work done at Studsvik are decontamination, incineration and recycling of low–level waste.

The most dominant source of radionuclides found in the surroundings of Studsvik comes from nuclear atmospheric tests performed between 1945 and 1980 and after the Chernobyl accident in 1986. Natural radionuclides are also easily found in all types of samples. Due to the changed background mentioned above only few samples indicate small outlets of radionuclides from Studsvik. Yearly emissions are much smaller than the limit set by the authority.

Introduction

Geologic and hydrologic explorations were performed in 1956 when it was decided to build a reactor site in Studsvik. Since 1957 environmental monitoring of radionuclides has been performed but the first emission of radionuclides from the Studsvik plant was performed first in July 1960.

The aim of the environmental sampling up to the middle of the 1960s was to find useful methods to measure the natural radioactivity, fall–out and outlets from Studsvik. The distribution of radionuclides was also investigated.

Results

Unfortunately the background measurements were first affected by the fall–out from the nuclear testing in the atmosphere and after 1986 by the Chernobyl accident. Higher, sometimes the double, radioactive concentration was found in 1958 and in some samples from 1959 compared to 1957 due to atmospheric fall–out by above ground nuclear weapon testing.

An ecological in–situ experiment in Bergösundet, at the outlet, with 390 GBq ^{137}Cs was performed in 1959 and is the highest outlet ever for a single nuclide. Highest total–β activity in air was found in October 1961, in November 1962 and at several occasions in 1963 with almost 0.4 Bq/m^3. The increased activity concentration was due to

atmospheric fall–out resulting in the highest concentration ever in fucus in 1963. Total alpha– and beta measurements on hay, straw, soil, drinking water, surface water and other types of terrestrial samples have always showed natural levels and contributions mainly from fall–out from nuclear weapons tests made since 1945 and later from the Chernobyl nuclear accident on 26 April 1986.

Due to very sensitive methods it is possible to detect very small amounts of radio–nuclides at long distances from the site. In 1975 approximately 1 MBq of ^{123m}Te was released from the incineration plant at Studsvik. The treated waste came from a hospital in Stockholm [1]. The sampling of air filters were performed 44 and 76 km north east of Studsvik. There has never been an accidental release above the limits.

Emission to Water and Air

Monthly reports from Studsvik to the Swedish Radiation Protection Institute, show small outlets of radionuclides to air and water from the site, excluding non nuclear installations which are not monitored [2]. The liquid waste is treated and monitored at the site before being released to the Baltic Sea from Tvären and Bergösundet near by. During the last couple of years the total emission to the Baltic Sea was between 100 and 1000 times lower than the limit of 0.1 mSv/year, set up by the authority [3].The radiation dose in mSv/year to the public from the emission of radionuclides sampled on filters or in water is calculated by using a model and conversion factors between radioactivity (in Bq) and the effective dose (in Sv) to the public. Emission to air was more than 1000 times lower than the limit (Figure 1).

Fucus

Fucus vesiculosus is a useful bioindicator for many nuclides [4]. For ^{137}Cs in fucus it was possible to show a correlation with outlet of ^{137}Cs after the 1960s [5] but the dominating source has normally been fall–out. Figure 2 shows measurements of ^{137}Cs in fucus since 1957. At the beginning only total beta measurements were performed. A factor of more than 10 is found between total beta and ^{137}Cs. In 1968 ^{110m}Ag was found for the first time [6] and in 1979 ^{51}Cr was found [7] immediately after an outlet of 11.5 GBq. A concentration of 1300 Bq/kg of ^{65}Zn in fucus was found in the spring of 1972 due to an emission of 28 GBq in October 1971 [8]. Other measurements [3] shows that emissions from Studsvik has a major part in the sediment concentration at least close to the outlet.

Conclusions

The most dominant source of radionuclides found in the surroundings of Studsvik comes from nuclear atmospheric tests performed between 1945 and 1980 and after the Chernobyl accident in 1986. Only few samples indicate small outlets of radionuclides from Studsvik. Small contributions of radionuclides can be seen by standard analyses only a few kilometres from the controlled waste water outlet. After 1 km from the main outlet a sharp decrease in radionuclide concentration can be seen. Yearly emissions are much smaller than the limit of 0.1 mSv set by the authority.

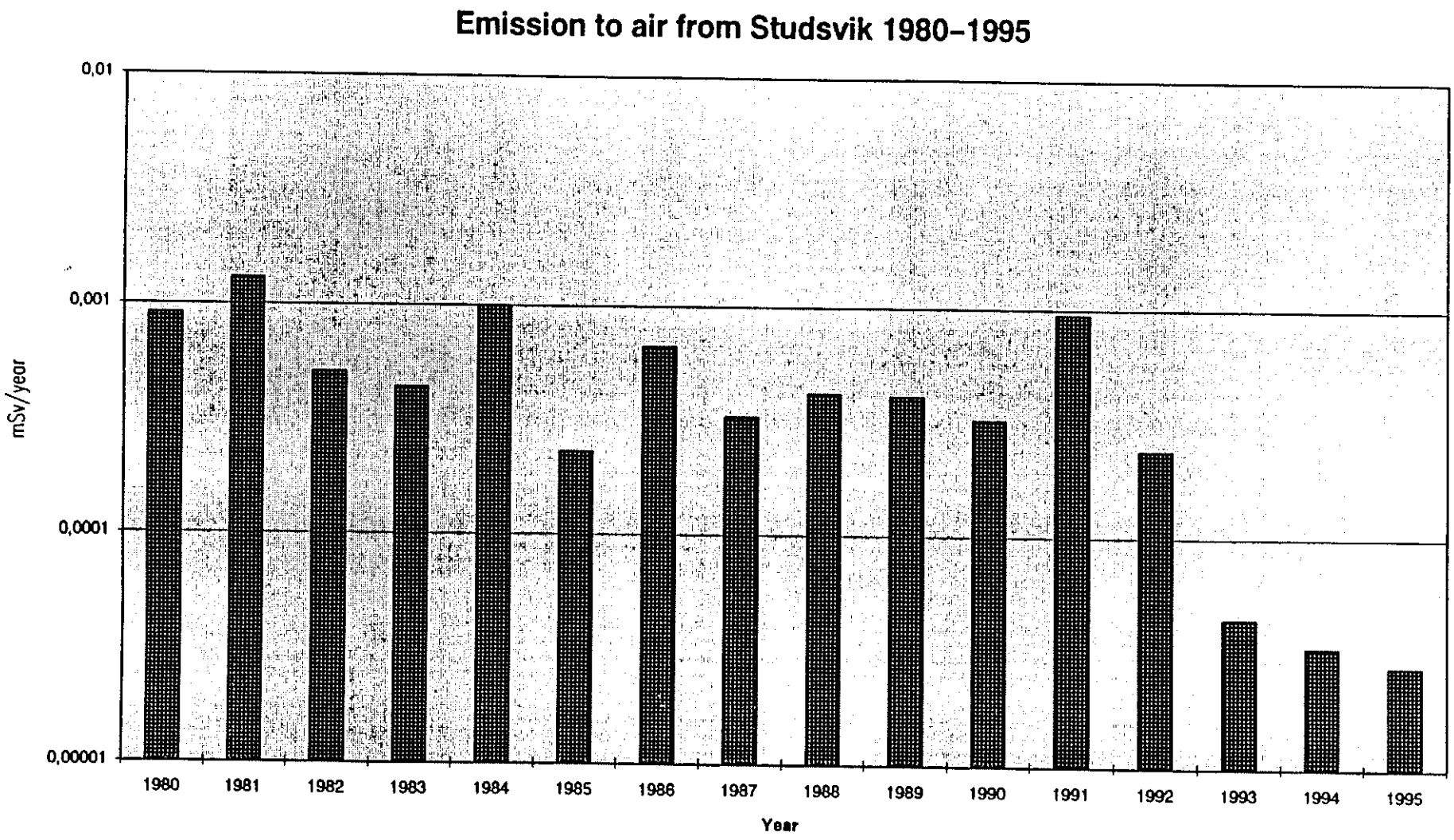

Figure 1. Emission to air of radionuclides from Studsvik.

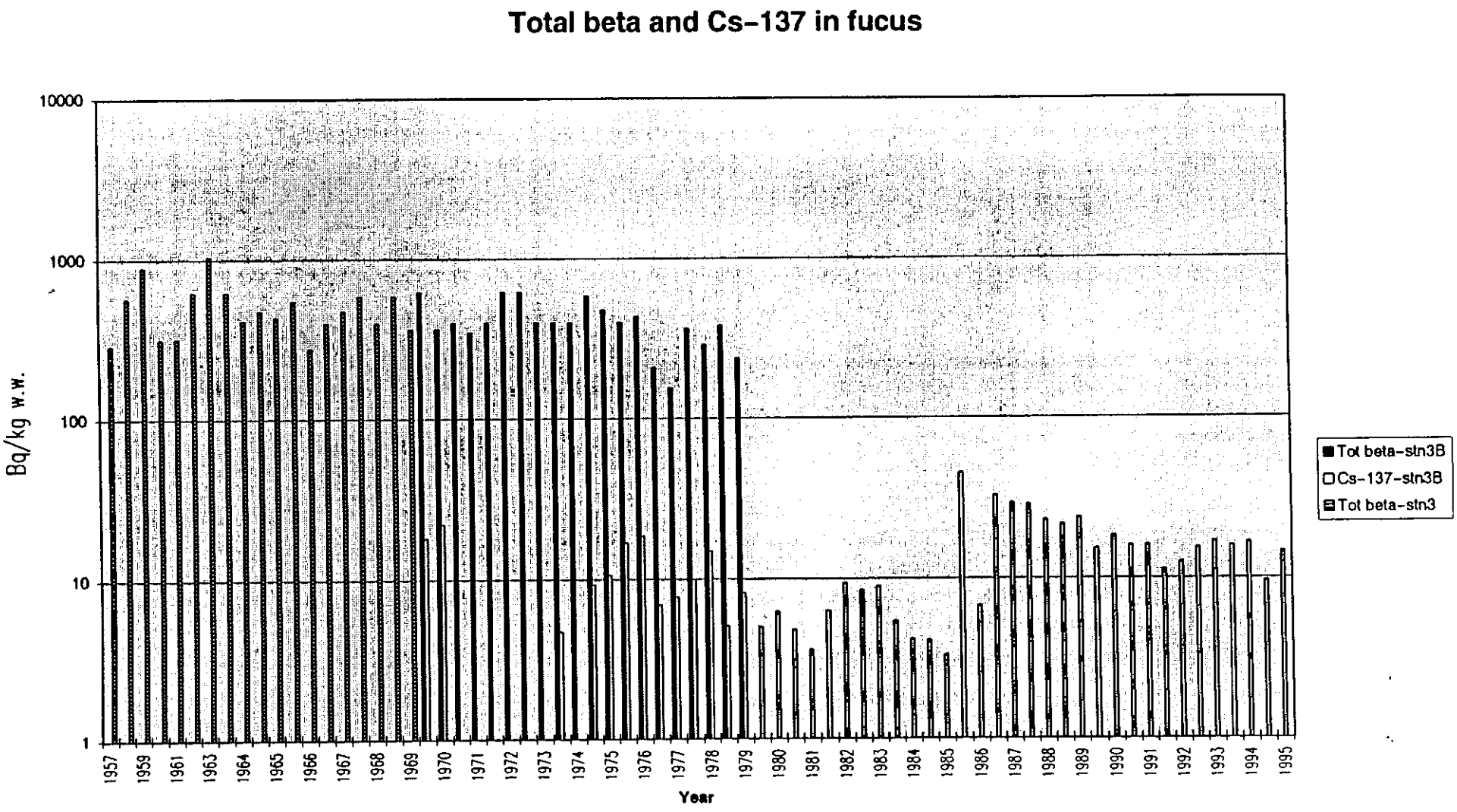

Figure 2. Radioactivity concentration of Cs-137 in fucus vesiculosus close to the outlet.

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Use of the moss *Hylocomium splendens* for the mapping of radiocaesium fallout from the atmosphere

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Abstract - Mosses are useful biomonitors of atmospheric deposition of trace elements, and are used on a regular basis to register heavy metal deposition in a number of European countries. Samples of the ground-growing species *Hylocomium splendens* collected in 1986 appeared to reflect well regional differences in the deposition pattern of ^{137}Cs from the Chernobyl accident in different parts of Norway. Later measurements of the same moss have demonstrated that annual segments formed after 1986 still show high activity in areas receiving high deposition in 1986, in spite of the fact that the post-1986 deposition has been very low. This shows that a certain translocation of caesium takes place within the moss plant, and indicates that samples collected more recently may still be used to learn more about the Chernobyl fallout pattern. Accordingly moss samples collected in 1990 for a national heavy metal survey in Norway were subjected to ^{137}Cs measurements. In this paper results based on 300 moss samples are presented. The observed geographic distribution trend is in close agreement with that observed in 1986 based on surface soil samples. This confirms that samples of *Hylocomium splendens* and similar moss species can provide useful information about the atmospheric deposition of radiocaesium.

Introduction

Primitive plants such as mosses and lichens have no root system and therefore absorb chemical substances mainly from above. Mosses as well as lichens have therefore been used extensively for the monitoring of various pollutants in the atmosphere. Heavy metals in particular have been frequently studied using this approach.In heavy metal deposition studies mosses are normally considered preferable to lichens because the annual incremental growth is more easily defined in mosses, which makes it possible to take for analysis a sample representing a defined exposure period.The feather moss *Hylocomium splendens* has shown to be particularly useful for heavy metal deposition studies in Scandinavia (1,2),because it can be found almost everywhere and the annual growth is easy to identify. This species is therefore used regularly in Norway (3-5) for monitoring the atmospheric deposition of heavy metals. So far nation-wide surveys have been carried out in 1977, 1985, 1990, and 1995, and the results are expressed as isopleths of the heavy metal concentration in moss, reflecting the relative deposition pattern. By calibrating against deposition figures as obtained from precipitation samplers (6) the moss isopleths can be transformed to metal deposition maps.

Until recently, however, the use of mosses to study atmospheric deposition of radionuclides had been seldom reported in the literature (7,8). Following the heavy deposition of radiocaesium in Norway from the Chernobyl accident, a pilot study was carried out to investigate the feasibility of different monitor plant to register the geographic distribution pattern of ^{137}Cs (9). That study showed that both *Hylocomium splendens* and the previously used reindeer lichen *Cladonia stellaris* exhibited radiocaesium levels in satisfactory agreement with deposition figures obtained from analysis of surface soil (10). This indicated that regional heavy metal deposition surveys employing *Hylocomium splendens* might be extended to include artificial radionuclides such as ^{137}Cs, if desirable.

When using mosses for heavy metal deposition studies, the best results are obtained in cases where an element is strongly fixed at the site on the moss plant where it was originally deposited. Translocation of an element within the moss would tend to disturb the temporal deposition trend stored in the different annual increments, especially if there are large fluctuations. Experience discussed below clearly showed that caesium is readily re-distributed within the plant, from the fact that high ^{137}Cs activity was still registered in moss shoots grown in 1987 and later. On the other hand, if this internal transfer occurs at the same rate at different sites, the original Chernobyl distribution would still be preserved in moss samples collected at a

later point in time.This was the basis of an attempt to use moss samples from the 1990 heavy metal survey in Norway to obtain additional information about the geographical pattern of Chernobyl radiocaesium.

Experimental

Samples of *Hylocomium splendens* for the 1990 heavy metal deposition survey were collected at 494 sites regularly distributed over Norway during the period May 20 - August 15, 1990. The samples were composites of sub-samples taken within a 50x50 m area in forests or other natural terrestrial ecosystems. Sampling under the canopy of trees was avoided if possible. Further description of the sampling and preparation of samples for analysis is given elsewhere(5). The last three annual segments, representing the growth during the period 1987-1989, were subjected to gamma-spectrometric measurements.

Samples of *Hylocomium splendens* collected in consecutive years at four field sites in central Norway were also studied.In this case the living part of the moss, which normally consisted of 3-4 annual segments, was divided in last year's and earlier years' growth before the radioactivity measurement.

The ^{137}Cs activity of the moss samples was measured with a multichannel analyzer equipped with a 3in.x3in. well-type sodium iodide detector and a sample changer.

Results and discussion

Results from the ^{137}Cs measurements in annual moss samples from the field sites are shown in Table 1:

Table 1. Distribution of ^{137}Cs between last year's and earlier years'growth of *Hylocomium splendens* from different sites in central Norway.

Site	Year of sampling	Last year (Bq/kg)	Earlier years (Bq/kg)	% activity in last year's growth
A	1990	7224	11275	39.1
	1991	6787	9752	41.0
	1992	6250	6650	48.4
	1993	2985	2951	50.3
B	1990	14218	29586	32.5
	1993	3841	4831	44.3
C	1991	3839	7831	34.4
	1992	8304	8969	48.1
D.	1988	6076	8297	42.3
	1991	2534	2394	51.4
	1992	998	1663	37.5

It is seen from the table that nearly half of the ^{137}Cs activity in the living part of the moss (Average 42.7 %) is stored in the most recent growth segment. This indicates that *Hylocomiom splendens* samples collected several years after 1986 may still be used to obtain information about the initial deposition pattern of the radiocaesium fallout from the Chernobyl accident, provided that the sampling is carried out in the same year, and before the development of the current year's growth, which normally starts in August - September. This means that the samples from the 1990 metal deposition survey should be suited for this purpose.

So far, nearly 300 samples from the 1990 collection have been analyzed, and the geographical distribution of ^{137}Cs in these samples is shown in Fig. 1. For comparison, the distribution of ^{137}Cs in topsoil measured by Backe et al. (10) in 1986 is also shown. Apart from the fact that the moss map is drawn on the basis of single points whereas the soil map was based on mean values for each municipality, the two maps are very similar. Thus there is clearly a potential to utilize moss samples collected in the present manner to obtain information about radiocaesium deposition in general as well as the Chernobyl fallout in particular.

Further development in this project will include measurement of the ^{137}Cs concentration in the remaining 200 samples from the 1990 moss collection, which may lead to improvement of the distribution map particularly in the areas that received little contribution from the Chernobyl accident. Another interesting development that will be attempted is to integrate the moss data with the soil data from 1986 in order to produce an improved deposition map for Chernobyl radiocaesium in Norway.

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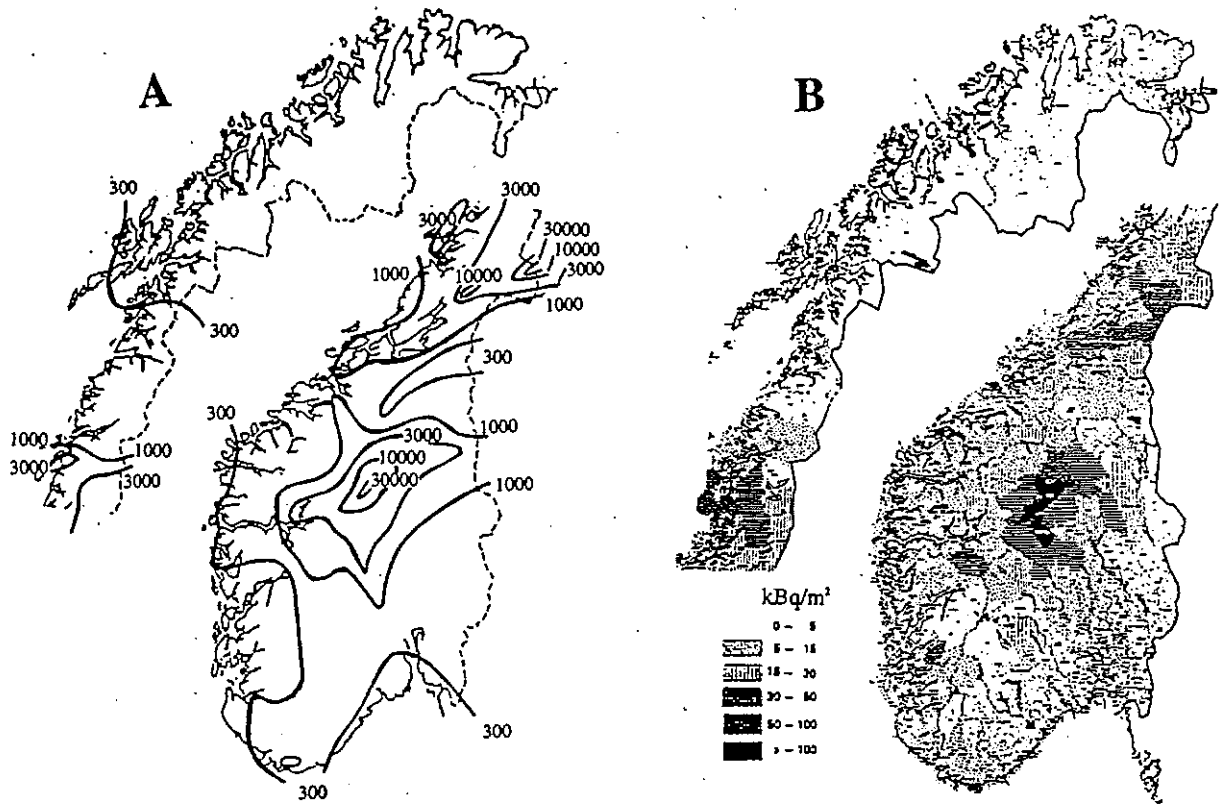

Fig. 1. Geographical distribution of radiocaesium in Norway

A. ^{137}Cs in *Hylocomium splendens*, this work (Bq/kg).

B. Areal distribution of ^{137}Cs + ^{134}Cs in 1986 (kBq/m^2) based on samples of surface soil. Data from Bache et al.(10).

Consumption of mushrooms and other wild food products in Finland

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Abstract - To improve the reliability of dietary assessments of ^{137}Cs received from foods of natural and seminatural origin a mail survey of Finnish households was carried out in spring 1996. An areally divided population weighted random sampling was used. The participants, 1500 households, were asked about their use of wild berries and mushrooms, game and reindeer meat, freshwater fish and some minor products of vegetative origin. Some questions sought to characterise the household.

The mean consumption rates (kg a^{-1}) with 95% confidence limits were as follows: wild mushrooms 1.5 (1.4-1.6), wild berries 8.3 (7.7-8.8), freshwater fish 7.9 (7.3-8.6), game 1.2 (1.1-1.3) and reindeer meat 0.52 (0.45-0.58). Use of wild food products differed notably between households of hunters and non-hunters. For both groups consumption rates also differed by subregions of the country. The households consuming the most wild food products are in fact multiple users and should have to be considered after large-scale radioactive fallout.

Key words: Consumption rate, berries, mushrooms, game, reindeer meat, herbs, sap, radiation dose assessment, mail survey

Introduction

The residence times of radiocaesium in a natural and seminatural environment are long and therefore consumers of wild food products were submitted to a survey. Such people are exposed to more internal radiation than is the rest of population. The lack of relevant consumption data on wild mushrooms in particular has increased the uncertainty of dietary radiocaesium assessments, especially since the Chernobyl accident. The substantial variations in ^{137}Cs contents of different species of mushrooms and wild animals prompted us to survey the consumption rates and patterns with a view to assessing radiation doses. Consumption data were therefore collected on species rather than groups of wild products.

For years the most representative regular dietary survey in Finland has been the Household Survey conducted by Statistics Finland [1]. It is based on quantitative book-keeping of foodstuffs delivered to the home. Wild products have been included in the survey irrespective of their source. Meals eaten outside the home are also included. However, no species-specific information about wild mushrooms has been given. The findings of a few nationwide or areal consumption surveys have been published in recent years [1, 2, 3, 4]. The aims of these studies have varied as has the importance of wild food products in the study plans. Some of them focus on certain subregions of the country. Both private households and institutional kitchens were studied for their consumption of mushrooms and willingness to increase it in 1992 [3]. Use of berries and mushrooms in Lapland was studied as part of a regional project in 1986 [4].

Mail survey

The use of wild food products by households and individual members of the households was studied in a mail survey conducted in spring 1996. The sampling frame was private households. The intake of wild products via meals eaten outside the home was omitted. A sample of 1500 households was taken from the population register as a regionally stratified random sample. The country was divided into four regions: metropolitan region, Western, Eastern and Northern Finland. The metropolitan region consists of the cities of Helsinki, Espoo, Vantaa and Kauniainen. Western Finland includes the rest of the province of Uusimaa, and the provinces Häme, Turku & Pori, Åland, Vaasa and Central Finland. Eastern Finland comprises the provinces of Kymi, Mikkeli, Kuopio and North Karelia, and

Northern Finland the provinces Oulu and Lapland. The sample was weighted by population. The ages of household members and their municipality of residence were given as entered in the population register.

The questionnaire contained 22 questions, many with clarifying subquestions. The participants were asked about their consumption of wild mushrooms, berries, game, reindeer meat, freshwater fish and other wild products of vegetative origin. The food groups were divided into groups based on the most important species in each. The species or families of mushrooms were *Russula, Lactarius, Cantharellus cibarius, Boletus* (including *Suillus* and *Leccinum* families), *Craterellus cornucopioides* and *Cantharellus tubaeformis*. All other mushrooms were in the group "other mushrooms". For berries and mushrooms the questions concerned both the actual use in harvest year 1995-96 and the potential use after a good harvest. For meat and fish the questions referred to the quantities used during the latest 12-month period. All quantities of interest were edible fractions.

Questions were asked about the cooking methods including preparatory treatment of mushrooms and the making of berry juice by steaming, the aim being to collect data for estimation of cooking losses of radionuclides. Seasonality of consumption was enquired for all types of products. Questions about consumption patterns, the way of delivery of and the motivation to use natural and seminatural food products were included in the study. The following information was collected on household members: reasons for the use of wild food products, interest in nature, municipalities where the products were acquired, availability of a car and/or summer cottage, type of residential area, occupation, education and annual income of family supporters.

An acceptable response rate (59%) was reached after two additional letters reminding people of the questionnaire and encouraging them to complete and return it. The response rate was lowest, 55%, in the metropolitan region (Table I). All reasonable answers were considered, even if not all questions had been answered. The final sample contained the 884 households which had returned an acceptable questionnaire. The number of persons in the sample was 2360. There was a slight difference in the age distribution between our sample and the whole population (Table II). The number of household members was somewhat higher in our group of respondents than nationwide (Table III) [5]. The original data were used without corrections for these differences.

Table I. Response statistics.

Area	Acceptable response % (N)	Empty or incomplete % (N)	No response % (N)	Recipient not at given address % (N)	Total N
Metropolitan region	54.7 (145)	3.8 (10)	40.4 (107)	1.1 (3)	265
Western Finland	60.9 (457)	6.4 (48)	32.0 (240)	0.8 (6)	751
Eastern Finland	56.7 (170)	5.7 (17)	36.7 (110)	1.0 (3)	300
Northern Finland	60.9 (112)	3.3 (6)	34.8 (64)	1.1 (2)	184
Whole country	58.9 (884)	5.4 (81)	34.7 (521)	0.9 (14)	1500

Table II. Age distribution of sample and whole population.

	Under 15 yr.	15-65 yr.	Over 65 yr.	N
Sample	18.3% (431)	69.6% (1 644)	12.1% (285)	2360
Whole nation [5]	19.1% (972 244)	66.8% (3 104 095)	14.1% (719 718)	5 098 754

A preliminary analysis of the responses showed that the proportion of hunting households was significantly greater than in the population as a whole. Hunters are more interested in wild products than are people on average [6]. The responses were therefore analysed separately for two groups of households: hunters, those who get bags; and non-hunters, all other people. The areal percentages of hunters who got bags (1993) were considered when the subgroup of hunters was defined [7]. The

resulting mean consumption rates for the population were weighted by the size of the two groups [5, 7, 8].

Table III. Size of households (Hh) in sample and whole population.

Region	Non-hunters			Hunters			Population in 1994		
	Hh	Persons/ Hh	Persons	Hh	Persons/ Hh	Persons	Hh	Persons/ Hh	Persons
Metropolitan region	137	2.28	313	8	2.88	23	395982	2.21	874953
Western Finland	414	2.68	1111	43	3.12	134	1070626	2.42	2594957
Eastern Finland	138	2.62	362	32	3.09	99	407648	2.40	976810
Northern Finland	77	2.60	200	35	3.37	118	245435	2.66	652034
Whole nation	766	2.59	1986	118	3.17	374	2119691	2.41	5098754

The measurement error, or, the uncertainty in the estimated amounts of foodstuffs, was increased by the difficulty of retrospectively recalled amounts, which had not always been measured at delivery. Conversion factors from volume (l) to mass (kg) were: mushrooms 0.4 kg l^{-1}, blueberries and lingonberries 0.6 kg l^{-1}, raspberries and cloudberries 0.5 kg l^{-1} and other berries 0.55 kg l^{-1} [9].

Results and discussion

Berries and fish are the most important products for the whole population and for hunters (Tables IV, V). Consumption of game by hunters is about tenfold that of non-hunters. The average consumption of the most common mushrooms differed by region (Fig. 1). Most mushrooms were consumed in Eastern Finland. The mushroom consumption pattern was similar in the metropolitan region and in Western Finland. In the North, *Craterellus* and *Cantharellus* are uncommon due to the climate, a fact which may partly explain the small consumption [10].

Table IV. Annual *per capita* consumption and 95% confidence intervals (CI) for main types of natural or seminatural food products (kg).

Product group	Whole population		Non-hunters		Hunters	
	Mean	95% CI	Mean	95% CI	Mean	95% CI
Mushrooms	1.5	1.4 - 1.6	1.5	1.4 - 1.6	1.8	1.6 - 2.1
Berries	8.3	7.7 - 8.8	8.1	7.5 - 8.7	13	11 - 14
Freshwater fish	7.9	7.3 - 8.6	7.7	7.0 - 8.4	13	12 - 14
Game	1.2	1.1 - 1.3	0.66	0.57 - 0.75	13	11 - 15
Reindeer meat	0.52	0.45 - 0.58	0.48	0.41 - 0.54	1.4	1.1 - 1.7
Other wild products	0.4	0.3 - 0.5	0.4	0.3 - 0.5	0.8	0.4 - 1.2

Previous studies suggest a 3-13.5 kg a^{-1} *per capita* consumption for berries [1, 2]. However the species included varied. For all types of fresh fish two dietary surveys showed a consumption rate of about 13 kg a^{-1} [1, 2]. Reported production of reindeer meat [11] and total annual game bag [1] give per capita consumption rates 0.7 and 1.7 kg a^{-1}. When converted to edible fractions of meat, these rates differ less than 15% from our results. In Lapland less than one kg of wild mushrooms were picked per person in 1986 [4].

Table V. The proportion (%) of users of different products.

Group	Berries	Freshwater fish	Mushrooms	Game	Reindeer meat	Other wild products
Whole nation	94	77	64	36	34	23
Non-hunters	94	77	64	33	34	22
Hunters	99	92	73	100	46	26

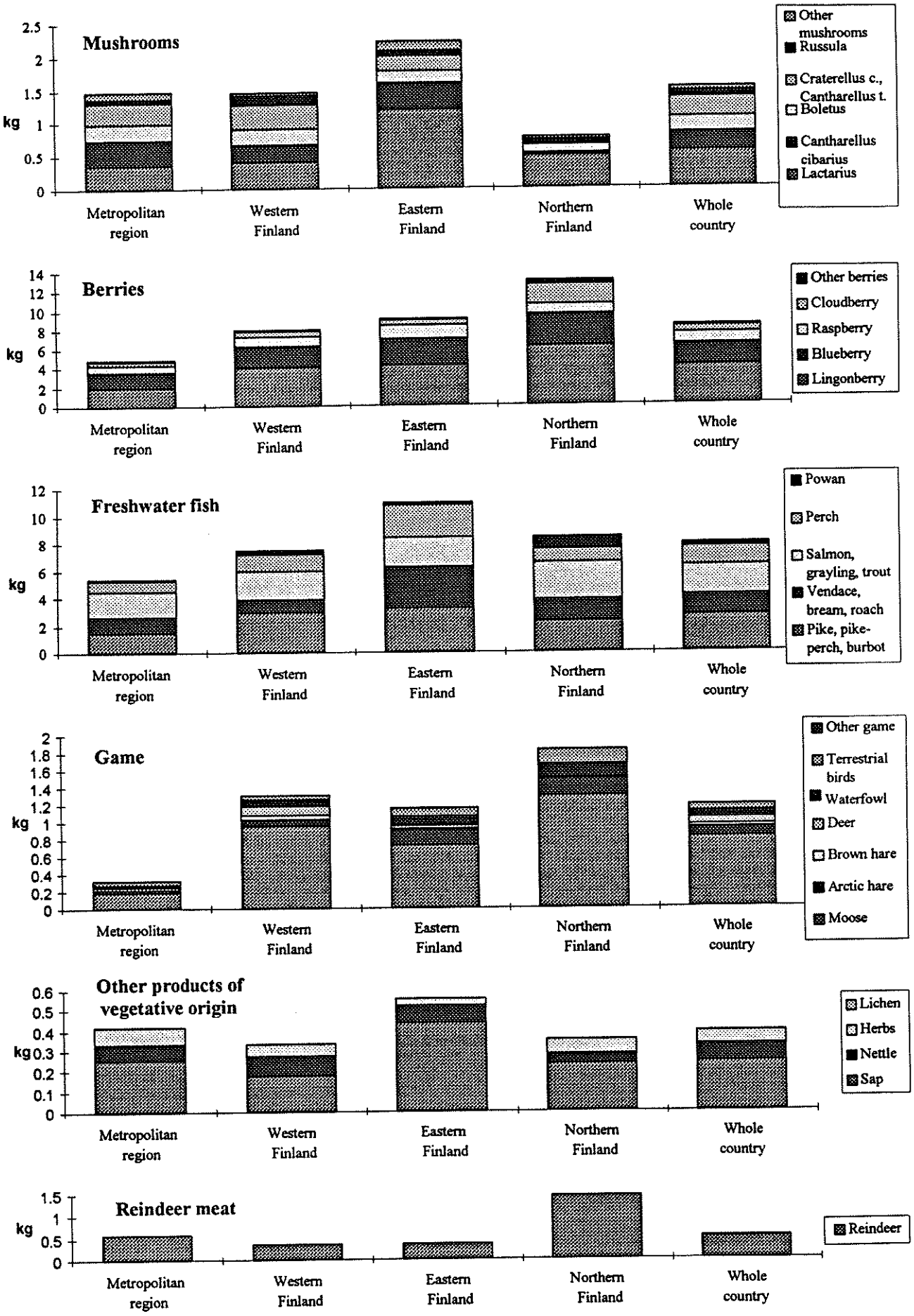

Figure 1. Annual average *per capita* consumption of foodstuffs of natural and seminatural origin in subregions of Finland and in the whole country. Note different kg scales.

Areal differences in consumption rates were obvious in all types of products (Fig. 1). The relatively few consumers of the sap explain more than half of the consumption rates of the "other wild products". Only two households reported using of lichen.

The use of several wild products by the same households is interesting in terms of maximum radionuclide intake and finding the part of population most at risk (Fig. 2, Table VI).

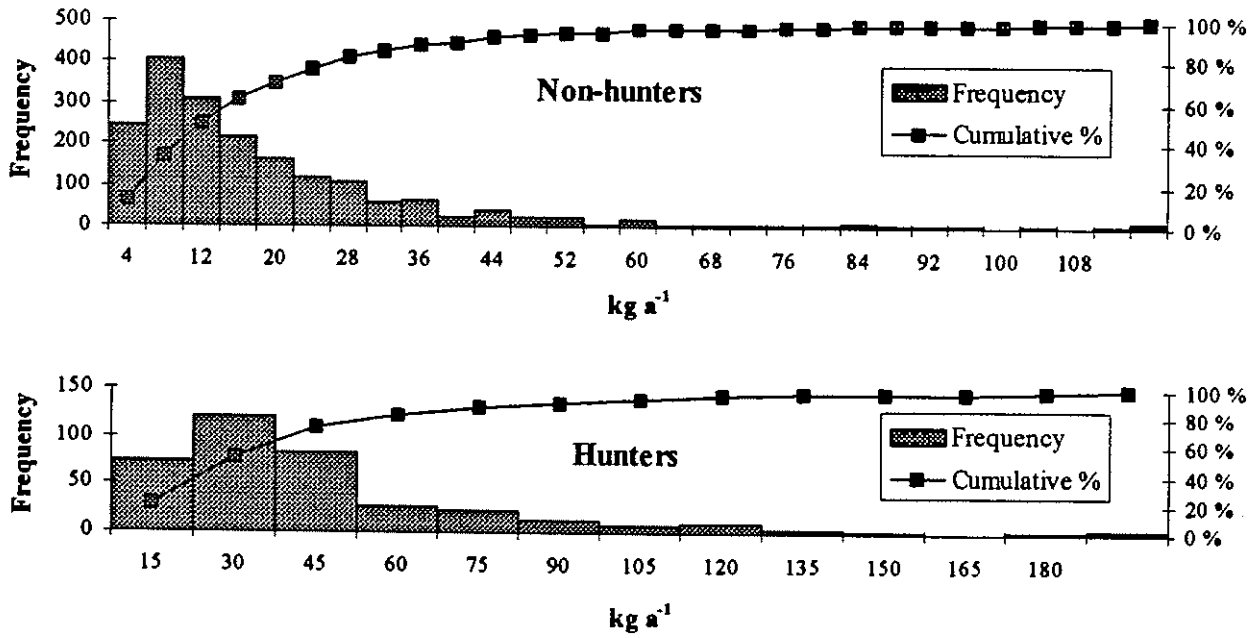

Figure 2. *Per capita* consumption of all wild food products by households of non-hunters and hunters. Frequency and cumulative distributions; upper limits of kg a^{-1} axis are given. People not using wild products (3.3%) are excluded.

Table VI. Structure of maximum *per capita* consumption based on sum of consumption rates of all wild products by the most consuming 10% of households. People not using wild products (3.3%) are excluded.

Product group	Non-hunters (N=191)		Hunters (N=37)	
	Consumption rate (kg a^{-1})	Proportion of amount consumed by non-hunters	Consumption rate (kg a^{-1})	Proportion of amount consumed by hunters
Mushrooms	5.7	37%	5.2	28%
Berries	28	34%	30	24%
Freshwater fish	34	43%	41	32%
Game meat	1.6	24%	49	37%
Reindeer meat	1.7	35%	4.8	33%
Other food products	2.0	47%	4.7	57%
All products	72	38%	130	31%

Hunters know how to take advantage of wild products, and all wild food products are consumed more in their households than in those of others, except in the metropolitan region, where non-hunters use freshwater fish more often. Almost all hunters' households use berries. In the metropolitan region and in Northern Finland about 70% of hunters' households consumed reindeer meat. In all other regions about 30-40% of all households consumed reindeer meat in 1995. A recent study on the hunter's profile revealed that most hunters are very active in fishing and in picking berries and mushrooms [6].

Self-picked mushrooms accounted for about 90% of the total in all regions except the metropolitan region, 80%. For berries the figures are 85% and 70%, for game meat 90% and 70%, and for fish

70% and 50%. In cities, the proportion of self-picked, -caught or -hunted wild food products is some percentage points lower than in the countryside.

The seasonality of consumption was most pronounced for mushrooms and fish, about a half of households used them outside the season. Berries are used throughout the year by 91% of households. There were no significant differences in seasonality between hunters and non-hunters. A good harvest would increase the consumption of mushrooms by about 60% and that of berries by about 20% compared with 1995, which was not a good mushroom year. In a good year consumption of mushrooms in Northern Finland would be twice that in 1995. Following criteria for choice of picking area of berries or mushrooms were cited: place of residence (37%), location of summer cottage (27%) and lack of contamination (23%).

Mushrooms were generally cooked using traditional methods. Almost all *Lactarius* type mushrooms were parboiled or salted, as were most *Russula*-type mushrooms (70%). Drying was most popular for *Craterellus cornicopioides* and *Cantharellus tubaeformis* (18%). As expected, most *Boletus*-type mushrooms (67%) and *Cantharellus cibarius* (69%) were cooked without any pretreatment. Roughly one tenth of these two types were parboiled. Deep frozen amounts were small. In the metropolitan region 10% of *Cantharellus cibarius* and *Boletus*-type mushrooms were frozen. Soaking was minimal except for *Boletus* in Eastern Finland (11%). Relatively small amounts of berries were used as steam juice: in Northern Finland 15%, Eastern Finland 11%, the metropolitan region only 3%, whole country 8%.

Conclusions

The consumption of wild food products was regionally unevenly distributed. The average consumption rates were in the range of, or somewhat greater than, those found in previous studies. Relevant comparisons are difficult due to the lack of representative information on entirely wild food products. The majority of people use below-average amounts of different products but maximum consumers exceed the average consumption many times over. Obviously there are dietary subgroups which should be carefully considered in the event of large-scale radioactive fallout.

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Importance of wild mushrooms for intake of radiocaesium by populations in Russia

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In a study of transfer of Chernobyl radiocaesium to man, dietary surveys and whole body monitoring were conducted at two sites in the Bryansk Region of the Russian Federation. The dietary survey confirmed that the inhabitants consume large amount of wild mushrooms. Based on the dietary survey and measured radiocaesium concentrations in mushrooms, consumption of mushrooms was estimated to account for 30-60% of the average intake of radiocaesium in the studied rural population 8-9 years after the Chernobyl accident. The correlation between dietary intake and observed body burdens also showed that wild mushrooms accounted for 20-40% of the variability of radiocaesium activity concentrations in the populations. Consumption of mushrooms was also the main reason for a 60-70% increase in radiocaesium activity concentrations in autumn. Long term dose assessments after accidental releases should therefore consider the contribution by mushrooms to ingested dose.

Fallout from atmospheric nuclear tests in the sixties
- Review of data from Iceland

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Abstract
Data on fallout in Iceland from the atmospheric nuclear tests in the sixties are relatively unknown, since little of the data collected have been published in scientific papers. There is data available in three Icelandic reports published in the sixties. The first report[1] was published in 1964 and is in English, the second[2] was published in 1965 and is in Danish and the last and most comprehensive one[3] was published in 1967 and is in Icelandic. In this paper data from the last report[3] is taken and analysed. This report has an extensive set of tables listing results of measurements and thus a statistical analysis of the data is possible. A highly significant difference in Cs-137 levels in cows milk and lamb meat is found between different production areas. The relative distribution of Cs-137 levels is similar as can be seen in recent years.

Introduction
The first measurements on environmental radioactivity in Iceland were probably done be Þorkell Þorkelsson[4] in the summer of 1906. These were measurement of radon emanating from geothermal hot springs. Monitoring of radioactive fallout was started in Iceland in 1958 by *The Physical Laboratory of the University of Iceland*, with regular measurements of total beta activity in aerosols and precipitation. The United Kingdom Atomic Energy Authority in Harwell also began measuring Sr-90 in precipitation in Iceland the same year and Cs-137 the following year. The Health and Safety Laboratory, USA, made three measurements during this period of Sr-90 in soil at Keldur in the vicinity of Reykjavík 1957 - 1959.

Measurements of radioactivity in foodstuffs in Iceland began with measurements of Sr-90 and Cs-137 in samples of milk, sheep and reindeer. The first measurements were carried out at the Risø National Laboratory in Denmark, but later on at the Physical Laboratory of the University in Iceland. In the latter part of 1964 the laboratory got a 3"x3" NaI(Tl) scintillation detector and subsequently measurement of Cs-137 were carried out on a regular basis. Samples were measured using a Marinelli beaker geometry with a single channel analyser. Due to this most of the data presented here come from the period 1965 - 1967 (1967 was the last year covered by the Icelandic reports).

Description of original data

In this paper main emphasis is put on analysing data for Cs-137 in milk and lamb. These data sets are the most complete ones described in the reports. The present analysis is based on the data given in the last report[3]. In one of the reports[2] there is also a description of whole body measurements on 9 Icelandic individuals carried out at Kernreaktor Bau- und Betrieb Gesellschaft in Karlsruhe in January 1962 and measurements on 11 individuals which were carried out at the Finsenslaboratoriet and Radiobiologisk Laboratorium in Copenhagen, October 1962.

Total beta activity in air, precipitation and fallout
Total beta activity was monitored in air, precipitation and fallout. The concentration in air reached its peak in 1962 (annual average read from graph in the last report[3] was 0,04 Bq/m^3) and the concentration in precipitation was maximum in the following year (20 Bq/litre). Thereafter the levels declined rapidly, being relatively low in 1965 and the 1966 values are almost unreadable on the graphs in the last Icelandic report[3].

Cows milk
Milk was collected from a few dairy production areas on a weekly or monthly basis with the exception of Reykjavík. There samples were originally collected on a daily basis, later the sampling frequency was reduced down to 2 per week. Figure 1 shows a time series of monthly values from all the 8 dairy producing areas. This figure was made using the data given in the last report. It shows why analysing this data can be difficult. Large variations can be seen in the data. Some of them seem to be systematic, e.g. occur at more than one place simultaneously. Missing data complicates the analysis further, especially for those places where systematic variations are expected.

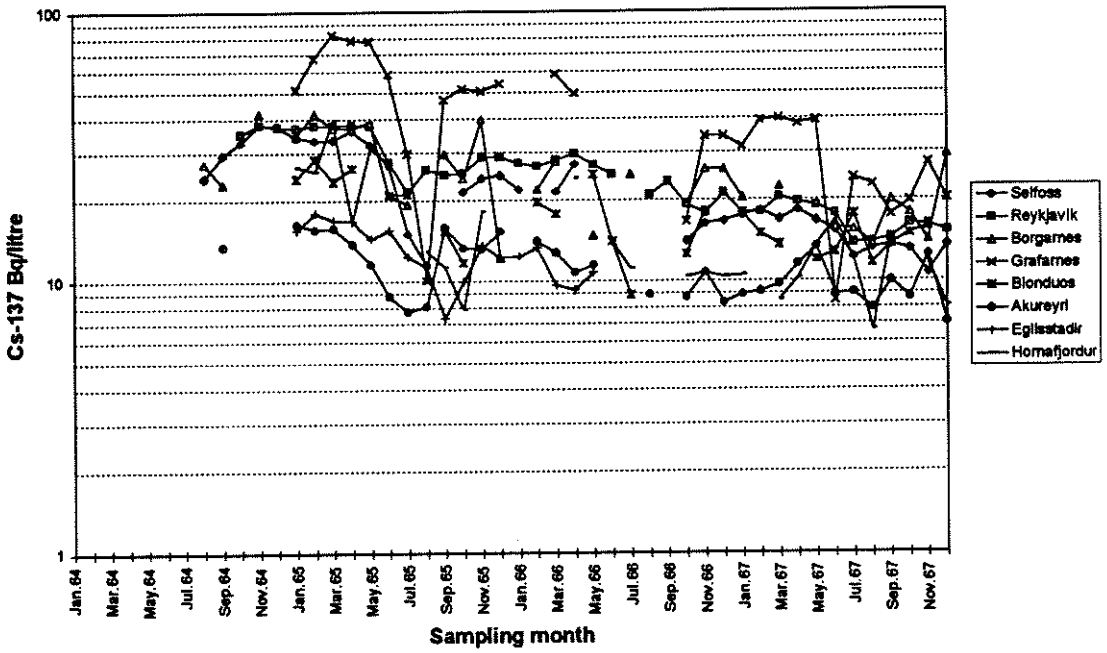

Figure 1. Cs-137 in cows milk in Iceland 1964 - 1967.

Lamb
During the years 1964 - 1967 samples of lamb meat were collected in the autumn from up to 10 slaughterhouses in different areas of Iceland. Five samples were collected from each. The geometric means for each area and year are displayed in figure 2.

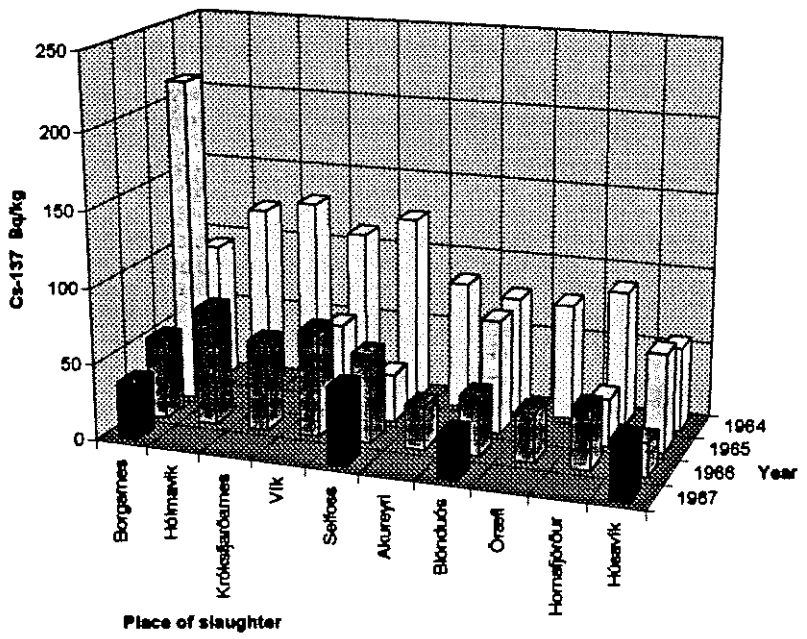

Figure 2. Cs-137 in Icelandic lamb meat 1964 - 1967, annual geometric means

Analysis of the data

The time period January 1965 - December 1967 was chosen for the analysis. Possible annual effects are minimised by using whole years and 1965 was the first year with a comprehensive data set. The measurements of total beta activity in air and precipitation indicate that the fallout in Iceland had also decreased to relatively low levels at that time. The lamb and cows milk data shows a rapid decline in Cs-137 concentration during 1965 - 1967. This does however not necessarily reflect the effective decay rate of Cs-137 (physical decay and other forms of removal from the ecosystem). The remaining fallout, although much less than previously, may have created a large enough input to affect the decay rate observed here. This decay rate will therefore in this paper be expressed as *observed half life*.

The cows milk and lamb data is analysed in this paper with the following objectives in mind:

- Determining the observed half lives for Cs-137 for the different production areas
- To check how significant the difference in the observed half lives is.
- To compare the Cs-137 levels between the different production areas and how significant the possible difference is.

An exponential curve was fitted to the data from each place versus time (more precisely a straight line was fitted to logarithms of the data). The values obtained are presented in Table 1. Observed half lives for Cs-137 in cows milk from eight areas.

Table 1. Observed half lives for Cs-137 in cows milk from eight areas

	Selfoss	Reykjavík	Borgarnes	Grafarnes	Blönduós	Akureyri	Egils-staðir	Horna-fjörður
$t_{1/2}$ (y)	2.3	2.2	3.1	1.9	3.5	4.8	1.8	2.2
S.E. (y)	0.3	0.2	0.8	0.5	0.9	1.3	0.5	0.5
n	29	35	27	29	25	32	17	27

By using one-way analysis of variance (ANOVA) one finds that the variation amongst observed decay rates is not significantly greater than can be expected by chance (P = 0.07). The conclusion is that although there might be a significant difference in the radionuclide removal processes in these 8 areas, the data presented here do not indicate strongly that such differences must exist.

Assuming that the difference in the observed half lives in these 8 areas is insignificant, all the data was combined and an average observed half life was calculated in the same way as before. This gives the results presented in Table 2. Average observed half life of Cs-137 in Icelandic cows milk, 1965 - 1967.

Table 2. Average observed half life of Cs-137 in Icelandic cows milk, 1965 - 1967

Average observed half life	Standard error	N	95% confidence interval
3.0 y	0.46 y	221	2.1 to 3.9 years

Investigation of possible differences in Cs-137 levels between areas

All milk data was decay corrected to the same reference time, middle of the year 1966. The average observed half life presented in Table 2 was used for this correction. The data was assumed to be distributed according to the lognormal distribution. Therefore logarithms of measurement data were used in the statistical analysis in place of the data itself. The average values presented here are therefore geometrical, not arithmetical.

Table 3. Geometric means of Cs-137 in milk, decay corrected to 1966

	Grafarnes	Borgarnes	Reykjavík	Selfoss	Blönduós	Horna-fjörður	Akureyri	Egils-staðir
Geometric mean	36	23	23	19	17	14	11	11
Geometric standard error	1.09	1.06	.1.02	1.04	1.05	1.07	1.04	1.05
No. samples	29	27	35	29	25	27	32	17
Median	40	25	23	20	17	13	11	11

Values referring to Cs-137 concentration are in (Bq/litre). Reference time is middle of 1966.

One way analysis of variance (ANOVA) of the logarithm of the time corrected milk data suggests that there is a highly significant difference between areas (P value less than 0,0001). Even after removing the three areas contributing most to the variance between locations the difference remains highly significant. The differences between geometric means are therefore greater than what can be expected from the variances within each area. One can also use the Kruskal-Wallis Test to compare the medians instead of means. This test is insensitive towards the presence of outliers and the form of the statistical distribution describing the data. The median values for each data set are given in table 3 and they are very similar to the geometric means. The Kruskal-Wallis Test also shows that the difference is significant ($P<0.05$) whatever form of distribution assumed.

Lamb

The observed half life of Cs-137 in Icelandic lamb meat was calculated in the same way as for the milk. Each data set has only four or fewer sampling times and figure 2 clearly shows a considerable variance within each area's data set. Using ANOVA as for the

milk data failed to show any significant difference in observed decay rates between different areas. The average observed half life of all the areas combined was calculated in the same way as was done with the milk data. The results are presented in table 4.

Table 4. Average observed half life of Cs-137 in lamb meat, 1964 - 1967

Average observed half life	Standard error	N	95% confidence interval
2.1 y	0.3 y	147	1.5 to 2.6 years

Various factors (e.g. differences in areas, plant selection of animals) can easily contribute to a difference between the observed half life in cows milk and lamb meat. There is therefore no reason to suppose that such a difference should not exist and a statistical test for significance of difference will not be made here.

Investigation of possible differences in Cs-137 levels between areas
All the lamb data was decay corrected to the same reference year 1966 using the average observed half life presented in table 4. The results for the different areas are presented in table 5.

Table 5. Cs-137 in lamb meat, decay corrected to 1966

	Borgarnes	Hólmavík	Króksfjarðar-nes	Vík	Selfoss	Blönduós	Öræfi	Húsavík	Akureyri	Horna-fjörður
Geometric mean	70	70	63	56	53	43	36	35	35	34
Geometric standard error.	9	6	9	7	8	3	4	4	6	5
No. of samples	21	10	10	15	17	19	10	20	10	15

Using one way analysis of variance to analyse the logarithmically transformed lamb meat data decay corrected to 1966 gives a highly significant difference between locations ($P < 0.0001$).

Comparison of geographical distribution of fallout (milk and lamb meat data)
Both the milk data in table 3 and the lamb meat data in table 5 show a similar pattern in fallout distribution. The western part of Iceland (e.g. Borgarnes) shows relatively high values. The southern part (Selfoss) and the northern part (Blönduós) show values near the middle of the range. Akureyri is also in northern Iceland, but the values there are amongst the lowest. This could be due to geographical conditions (the area is enclosed by mountains). Places in eastern Iceland also have relatively low values (Hornafjörður, Egilsstaðir, Húsavík).

Comparison with recent data
Iceland received a very minor fallout from the accident at the Chernobyl Nuclear Power Plant. The Cs-137 that can be found in the Icelandic terrestrial ecosystem stems almost entirely from fallout from the atmospheric nuclear weapon tests in the sixties. Results from measurements during the last decade by Geislavarnir ríkisins show a similar difference in Cs-137 levels as the data from the sixties. Western Iceland tends to show the highest values, the main agricultural area in south gives levels that are middle of the range. The lowest values can be found in areas such as Eyjafjörður (the area around Akureyri).

Comparison with milk data from the Faroes Islands

Fallout in the nearby Faroe Islands was studied and reported by the Risø National Laboratory, Denmark. These results are e.g. quoted in the 1977 UNSCEAR compilation on the world-wide distribution of fallout from the atmospheric tests. In this compilation the Faroe Islands had by far the highest concentration of Cs-137 in milk during the period quoted (1966-1975). Figure 3 shows milk data from 3 areas in the Faroe Islands (the figure is based on data published by the Risø National Laboratory[6]).

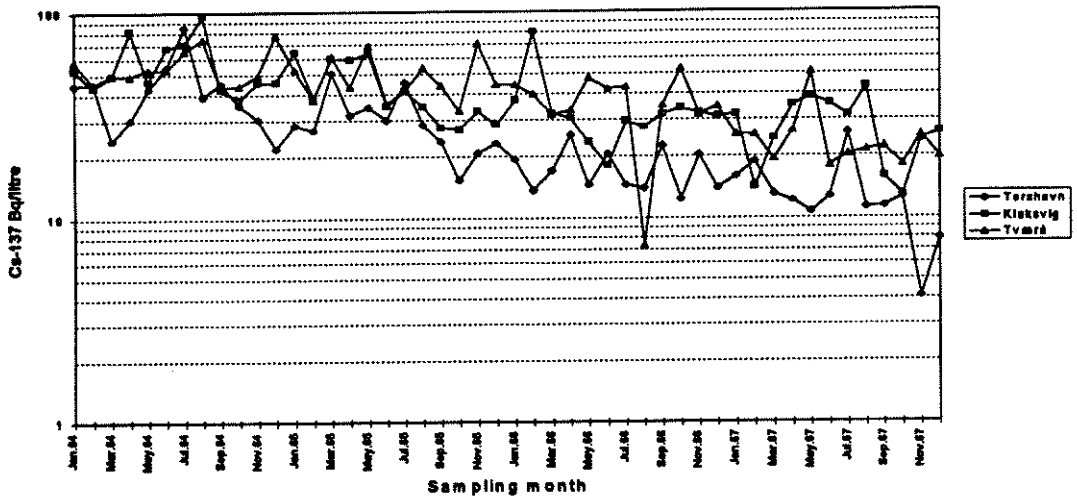

Figure 3. Cs-137 in cows milk from the Faroe Islands, 1964-1967

The results displayed in figure 3 can be compared with the corresponding Icelandic results in figure 1. The Icelandic results are generally lower, but the order of magnitude is the same.

Conclusions

The analysis of the measurements carried out 1965 - 1967 in Iceland fails to show any significant difference in the observed half life of Cs-137 between different areas for cows milk on one hand and lamb meat on the other hand. The levels of Cs-137 in these products however varied very significantly between production areas and a similar distribution has been observed in recent years. A study is under way comparing the reported results with data obtained in other countries at this period and with more recent Icelandic and international data.

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Det ellevte ordinære møte. Nordisk selskap for strålevern og Det syvende Nordiske Radioøkologiseminar, 26.-30. august 1996, Reykjavik.

Oppgraving av et 25 år gammelt deponi for radioaktivt avfall i Norge

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Oppsummering

I 1970 ble ca 1000 tønner med lavt- og middels radioaktivt avfall gravet ned på området til Institutt for energiteknikk (IFE) på Kjeller i Norge. Statens strålevern (NRPA), den gang Statens institutt for strålehygiene, SIS) ga tillatelse til denne deponeringen etter en sikkerhetsvurdering og med krav knyttet til deponiet. Avfallet består av ionebyttermasse, avfall fra laboratorievirksomhet, flytende avfall og metallskrap som etter avfallsbehandling ble lagt i 210 liters ståltønner med variabel innvendig utforming avhengig av avfallstypen og aktivitetsnivået.

Arbeidet med å etablere et deponi for lavt- og middels aktivt avfall i Norge startet i 1989. I april 1994 besluttet Stortinget at det skulle bygges et kombinert lager og deponi for lavt- og middels radioaktivt avfall i Himdalen i Aurskog-Høland kommune. I forbindelse med dette ble det bestemt at de nedgravde tønnene skulle fjernes og plasseres i det nye anlegget. Ca 230 av de nedgravde tønnene inneholder plutonium. Det ble bestemt at disse skulle plasseres i det nye anleggets lagerdel i påvente av en nærmere beslutning om fremtidig deponering. Resten av tønnene skulle plasseres i det nye anleggets deponidel sammen med de avfallsbeholderne som i dag lagres i lagerbygg på IFEs område på Kjeller.

Som en forberedelse til arbeidet med å fjerne de nedgravde tønnene, er det foretatt nærmere undersøkelser av omfang og aktivitetsnivå av radionuklider i leire og vann i og rundt tønnefeltet og av tønnenes tilstand. På grunn av tønnenes variable tilstand vil de bli plassert 4 og 4 i nye stålkontainere og omstøpt med betong før de fraktes og plasseres i det nye lager/deponi i Himdalen. Undersøkelsene viser at det skal være mulig å håndtere tønnene mekanisk og på den måten redusere stråledosene til et minimum. Den totale individuelle stråledosen til personell er anslått til 2 - 3 mSv eller lavere. Beregninger viser at mengden forurenset leire som må samles og behandles som radioaktivt avfall kan være av størrelsesorden 100 m^3.

1. Etablering av deponiet

Institutt for Energiteknikk (IFE) i Norge ble etablert i 1948. Fram til i dag har Instituttet vært eier og operatør av fire forskningsreaktorer hvorav to fremdeles er i drift, Haldenreaktoren og JEEP II på Kjeller utenfor Oslo. Instituttet har også produksjon av radiofarmaka, driver et bestrålingsanlegg, undersøker eksperimentalbrensel fra Haldenreaktoren og driver Norges eneste anlegg for mottak og behandling av radioaktivt avfall. Et pilotanlegg for reprosessering av uran var i drift mellom 1961 og 1968. Dette anlegget er nå demontert.

På grunn av den nukleære aktiviteten i 1950- og 1960-årene ble det forutsett at IFEs lagre for lavt- og middels radioaktivt avfall ville være fylt opp i 1970. Det ble derfor bestemt at dette avfallet skulle deponeres i et overflatedeponi på Instituttets område på Kjeller.

Planer og sikkerhetsvurderinger for dette deponiet ble gitt i en søknad til Staten institutt for strålehygiene (SIS, nå Statens strålevern, NRPA) som ga tillatelse til å etablere deponiet etter spesifiserte krav.

2. Avfallsvolumer og radioaktivitetsnivåer

Totalt ble 1013 avfallsbeholdere og 19 andre gjenstander plassert i deponiet [1]. Avfallsbeholderne var standard 210 liters ståltønner. Den indre utforming og skjermingen i tønnene varierte for ulike typer avfall og aktivitetsnivåer. Avfallet bestod av laboratorieavfall (papir, plast, glass osv), organisk væske absorbert i vermikulite og ionebyttermasse hvor fuktigheten var ekstrahert. Flytende avfall ble overført til fast form ved å blande med sement og lagret i tønner med innvendig polyetylen belegg. Metallskrap ble omstøpt med sement. Metallskrap og flytende avfall med høy doserate (> 20 mSv/h) ble plassert i innertønner av bly.

Detaljert informasjon om avfallstønnene er vanskelig tilgjengelig på grunn av dårlig dokumentasjon om de enkelte tønnene. Den totale aktiviteten ved nedgravingstidspunktet pr 31. desember 1970 og pr 1. juli 1997 er gitt i Tabell I. "Andre nuklider" er ^{3}H, ^{14}C, ^{55}Fe og ^{204}Tl. Aktivitet pr 1. juli 1997 er oppgitt fordi dette er det første mulige tidspunkt for oppgraving av deponiet. Ved nedgraving i 1970 var den totale aktiviteten for nuklider med halveringstid > 1 år ca. 9000 GBq. I 1997 vil den totale aktiviteten i deponiet være sunket til ca 3000 GBq. Det største bidraget kommer fra ^{137}Cs og ^{90}Sr.

Deponiet inneholder i alt 80 GBq med ^{239}Pu (35 g) og 2,5 GBq (100 kg) med naturlig uran fra reprosesseringsanlegget. Av dette er ca 90 % plassert i tønner med innvendig polyetylen-belegg.

3. Utforming og sikkerhetsvurderinger for deponiet

Avfallstønnene ligger i to lag og er omsluttet av leire. Deponiet dekker et areal på 11,5 x 23 meter. Det øverste tønnelaget har en overdekning på 1,5 - 2 meter leire og jord. Figur 1 viser et bilde av deponiet.

Deponiet ble etablert i et gammelt bekkeleie og terrenget er svakt synkende i deponiets lengderetning. I enden av deponiet med lavest nivå er det plassert en dreneringskum 0,5 meter fra kanten av tønnene. I dreneringskummen samles alt regn- og grunnvann som renner gjennom deponiet. Det tas rutinemessig prøver av vann og slam i dreneringskummen i henhold til et monitoreringsprogram. Fra kummen føres vannet i rør til avfallsbehandlingsanlegget. IFE har tillatelse til å slippe ut flytende radioaktivt avfall til Nitelva som renner i utkanten av Kjeller-området. Statens strålevern har fastsatt grenser for utslipp slik at stråledosene til individer i området ikke skal overskride 1 μSv pr år. Vann fra dreneringskummen blir derfor analysert før det slippes ut i Nitelva.

Det ble opprinnelig anslått at ståltønnene ville forbli intakte i 10 år og at de deretter ville gå i oppløsning, inkludert betongskjerming og innertønne, over en periode på minimum 50 år. En forventet derfor ingen radioaktiv forurensning av miljøet utenfor leirebedet i de første 50 årene. Etter denne tidsperioden ble det antatt at det totale aktivitetsinnholdet i deponiet ville lekke ut over en periode på 10 år. Nye beregninger fra 1989 konkluderte med at levetiden for betongen i tønnene ville være i området 30 - 100 år under ugunstige forhold [2].

4. Monitoreringsprogram og lekkasje fra deponiet

I juli 1974 ble det utført et tracereksperiment for å kontrollere hvordan leirebarrieren og dreneringssystemet fungerte. I alt 80 m^3 vann ble tilført deponiet ved vanning. Gjennom 10 rør fra toppen av deponiet vannet ble det tilsatt 196 GBq ^3H i en liter vann pr rør [3]. Målinger utført over 3 måneder og viste at den totale ^3H-aktiviteten ble oppsamlet i dreneringskummen. Dette viste at leirebarrieren og dreneringssystmet fungerte tilfredsstillende.

Mengden vann som renner gjennom deponiet avhenger av nedbøren. Slamprøver fra drenskummen en gang pr år. Vannprøver tas som proporsjonalprøver av vann som renner fra dreneringskummen i rør til avfallsanlegget. Den midlere utlekking av ^{90}Sr, ^{137}Cs og ^{239}Pu pr år i vann og slam fra deponiet for fire perioder mellom 1970 og 1994. er gitt i Tabell II.

5. Bakgrunn for oppgraving av deponiet

I april 1994 bestemte det norske Storting at det skulle bygges et kombinert lager og deponi for lavt- og middels radioaktivt avfall (KLDRA) i Himdalen i Aurskog-Høland kommune. Området ligger 25 km fra IFEs område på Kjeller. Det ble videre bestemt at IFE skulle ha driftsansvaret for dette anlegget. Stortinget bestemte også at nedgravde tønner skulle graves opp og at de som inneholder plutonium skulle plasseres i anleggets lagerdel i påvente av vedtak om permanent deponi for dette avfallet. Andre nedgravde avfallstønner skal støpes inn i betongsarkofager i anleggets deponidel.

Konsekvensen av Stortingets beslutning er at deponiet må graves opp og at tønner som inneholder plutonium må identifiseres og separeres fra resten av tønnene. Antall tønner som inneholder plutonium er ca. 230.

6. Undersøkelser for planlegging av oppgraving av deponiet

Tidlig om morgenen den 7. september 1993 brøt Miljøstiftelsen Bellona seg inn på IFEs område og startet graving i deponiet med en gravemaskin. Etter 2 timer ble aktivistene fjernet av politiet. De hadde da gravet seg ned til øverste tønnelag og skadet en av tønnene. Før dette hullet ble fylt igjen, benyttet IFE anledningen til å hente opp 5 tønner for inspeksjon, tre fra øverste tønnelag og to fra nederste. Samtidig ble det tatt prøver av leire og vann mellom og under tønnene og rust fra overflaten av tønnene. Disse prøvene ble analysert av IFE og av NRPA med tanke på ^{137}Cs og ^{60}Co. Noen av prøvene ble også analysert for ^{90}Sr og plutonium.

I desember 1994 startet en systematisk undersøkelse for å fastslå kontamineringsnivået i og rundt deponiet og langs det gamle bekkeleiet. Hull ble gravet ned til 1,6 meters dybde og det ble tatt kjerneprøver herfra og ned til ca 4 meters dybde i 10 punkter rundt deponiet og langs bekkeleiet. Prøvene ble delt i seksjoner av 5 cm, tørket og analysert for ^{137}Cs og ^{60}Co. Prøver som inneholdt aktivitet av disse nuklidene ble også analysert for ^{90}Sr og plutonium.

For å skaffe mer informasjon om kontamineringsnivået inne i deponiet og tønnenes tilstand ble det gravet to hull, en i hver ende av deponiet. Fra hver av disse hullene ble

det tatt opp 5 tønner, tre fra det øverste tønnelaget og to fra det nederste. Det ble samtidig tatt prøver av vann og leire rundt og under tønnene. Siden noen av tønnene var gjennomrustet ble det tatt prøver av vann som rant ut av tønnene. Alle prøvene ble analysert for ^{137}Cs og ^{60}Co. Prøver som inneholdt aktivitet av disse nuklidene ble også analysert for ^{90}Sr og plutonium.

Under uttak av tønnene ble det utført målinger av doserater. Denne informasjonen sammen med tønnenes tilstand danner grunnlag for planlegging av utgraving av deponiet og bruk av hjelpemidler samt estimater over stråledoser til mannskaper.

6.1 Kontaminering rundt deponiet

Med to unntak ble det ikke funnet radioaktivitet i kjerneprøvene fra borehull rundt deponiet. I ett borehull nærmest drenskummen ble det funnet spor av ^{137}Cs fra 3,1 meters dybde ned til 4,2 meters dybde. Aktivitetsnivåene varierte fra 600 Bq/kg ned til 10 Bq/kg. Dette skyldtes antakelig lekkasje av forurenset vann og slam fra drenskummen i sjiktet mellom det gamle terrengnivået og ny masse langs det gamle bekkeleiet. Spor av ^{137}Cs ble også funnet i det siste borehullet langs det gamle bekkeleiet.

På bakgrunn av analyse av kjerneprøvene er det rimelig å konkludere med at leirbedet har fungert som en tilfredsstillende barriere for lekkasje fra deponiet.

6.2 Kontaminering av leire og vann inne i deponiet

Aktivitetsnivåene for ^{137}Cs i prøver av leire og rust tatt på og mellom tønnene i deponiet varierer fra 1,5 MBq/kg og nedover. Prøver av vann som ble tatt etter at tønner var fjernet inneholdt ^{137}Cs i området 1 - 50 Bq/l og ^{90}Sr i området 1 - 10 Bq/l. Etter filtrering av vann ble det målt opp til 3 MBq/kg for ^{137}Cs på filtrene. Profilprøver av leire og slam fra bunnen av deponiet under det laveste tønnelaget viste aktivitetsnivåer opp til 100 kBq/kg i de første 2 cm dybde og ned til 20 kBq/kg i 5 cm dybde.

Med unntak av en prøve som inneholdt 7 kBq/kg av ^{239}Pu, inneholdt prøvene aktivitetsnivåer fra 500 Bq/kg og nedover for denne nukliden. En leirprøve inneholdt 6 kBq/kg av ^{90}Sr. De andre prøvene hadde aktivitetsnivåer fra 550 Bq/kg og lavere for denne nukliden.

Generelt hadde leirprøvene som var tatt mellom og under det laveste tønnelaget vesentlig høyere aktivitetsnivåer enn prøver tatt fra det øverste tønnelaget. Profilprøver av leire fra overflaten av tønnene viste at radioaktiviteten var konsentrert i de første 2 cm med leire på tønneoverflatene.

6.3 Tilstanden til tønnene

Tønnene i det øverste tønnelaget er i en bemerkelsesverdig godt forfatning. De fleste av disse viste antydning til korrosjon bare på undersiden. Tønner i det laveste tønnelaget er vesentlig mer korroderte og for noen av dem er yttertønnen stedvis gjennomrustet. Til

tross for dette viste det seg mulig å heise opp tønner med kroker festet til stålfalsene i kantene av tønnene.

Totalt sett synes tønnene å være i en bedre forfatning enn det en kunne forvente, og dette vil forenkle arbeidet med oppgraving og håndtering.

7. Konklusjoner fra undersøkelsene

For å redusere stråledosene til personalet vil deponiet bli gravet opp seksjonsvis. Undersøkelsene viser at det er mulig å heise opp tønnene ved hjelp av kroker festet til tønnekantene. Leire som sitter på overflaten av tønnene vil bli skrapet vekk og behandlet som radioaktivt avfall, men det vil ikke bli foretatt noen ytterligere rengjøring av tønnene. På bakgrunn av tønnenes variable tilstand vil de bli plassert 4 og 4 i nye stålkontainere og omstøpt med betong før de fraktes til det nye lager/deponi i Himdalen.

Vann i deponiet vil bli pumpet opp og behandlet som radioaktivt avfall. Monitorering og oppsamling av forurenset leire og slam i deponiet synes å bli det største problemet ved oppgravingsprosessen. Den friklassingsgrensen som anbefales av Statens strålevern er aktivitetskonsentrasjonen fra radioaktivt nedfall fra Tsjernobyl-ulykken. For ^{137}Cs er dette ca 4 kBq/m^2 i området rundt Kjeller. Så langt har vi ikke funnet en effektiv metode for kartlegging av forurenset leire og slam i deponiet. Området rundt drenskummen og langs det gamle bekkeleiet må også graves ut og kontrolleres for forurensninger. Hvis vi antar at et 2 cm tykt leirelag rundt tønnene er forurenset, gir dette 60 m^3 leire som må tas hånd om som radioaktivt avfall. Totalt sett regner vi med at mengden forurenset leire og slam fra oppgravingen blir i størrelsesorden 100 m^3.

En vesentlig faktor er stråledoser til mannskaper som skal utføre utgraving og håndtering av tønnene. Målinger av doserate fra tønnene viser at den varierer kraftig, men kan være opp til 1 mSv/h i kontakt og 300 µSv/h i en meters avstand. Hvis vi antar at mekaniske hjelpemidler kan brukes for å heise opp og håndtere tønnene slik at nærkontakt til en viss grad kan unngås, kan de totale individuelle stråledosene til mannskapene anslås til 2 - 3 mSv eller lavere.

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Tabell I

Radioaktivitet i deponiet pr 31. desember 1970 og 1. juli 1997

Isotop	Halveringstid (år)	Radioaktivitet 31. desember 1970 (GBq)	Radioaktivitet 1. juli 1997 (GBq)
^{60}Co	5.3	1850	60
^{90}Sr	28	2570	1340
^{137}Cs	30	2520	1360
^{239}Pu	24 000	85	85
Uran	$4.5 \cdot 10^9$	2.6	2.6
Andre nuklider		2240	< 100
Totalt (GBq)		≈ 9270	≈ 2900

Tabell II

Utslipp av radioaktivitet med vann og slam fra deponiet via drenskummen

Isotoper	Midlere verdier 1970 - 1979 (kBq/år)	1980 - 1989 (kBq/år)	1990 - 1994 (kBq/år)	1970 - 1994 (kBq/år)
I vann:				
^{90}Sr	2330	980	1100	1540
^{239}Pu	30	20	33	27
^{137}Cs	ikke målt	4000	2700	3600
I slam:				
^{90}Sr	ikke målt	0.7	0.8	0.7
^{239}Pu	"	6,0	9.8	7.3
^{137}Cs	"	40	66	49

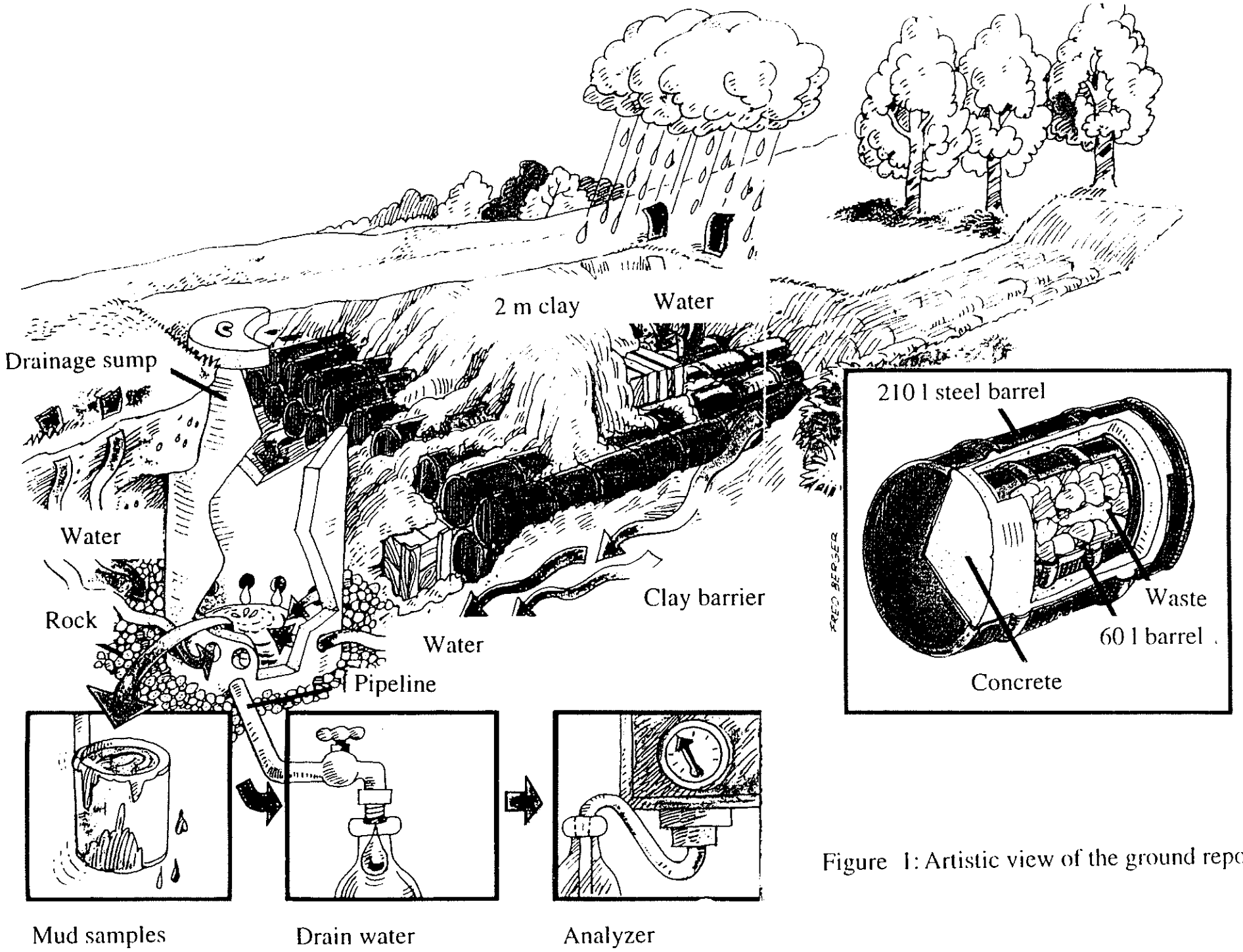

Figure 1: Artistic view of the ground repository

Det ellevte ordinære møte. Nordisk selskap for strålevern og Det syvende Nordiske Radioøkologiseminar, 26.-30. august 1996, Reykjavik

Kombinert lager og deponi for lavt- og middels radioaktivt avfall i Himdalen i Norge

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Oppsummering

Stortinget har bestemt at Norge skal bygge et kombinert lager og deponi for lavt- og middels radioaktivt avfall i Himdalen i Aurskog-Høland kommune. Anlegget vil bli bygget som fire separate fjellhaller med 50 meter fjelloverdekning. Tre av hallene vil bli benyttet for deponering av avfall og en til lagring av plutoniumsholdig avfall. I deponidelen vil avfallet bli støpt inn i betongsarkofager. Anlegget vil være i drift fram til år 2030 og er dimensjonert for i alt 10 000 tønner med radioaktivt avfall. Himdalen er et geologisk stabilt område og gradienten for grunnvannsstrøm vil være inn mot anlegget. Lekkasje av radioaktivitet fra fjellhallene vil derfor bare forekomme i anleggets dreneringssystemer eller ut gjennom adkomsttunnelen fordi hele anlegget har et fall mot tunnelåpningen på 1:50. Det vil bli konstruert to dreneringssystemer med dreneringsbrønner, et for vann som trenger inn i anlegget fra det omkringliggende berget og et for vann som eventuelt trenger ut av sarkofagene. Monitorering av dreneringsvann vil bli utført i hele driftsperioden og radionuklider i vann vill bli overvåket i en periode på 300 - 500 år framover

1. Innledning

Arbeidet med å etablere et sentralt deponi for lavt- og middels radioaktivt avfall i Norge startet i 1989. Etter flere utredninger om stedsvalg og konsekvenser vedtok det norske Storting i april 1994 at et kombinert lager og deponi for lavt og middels radioaktivt avfall (KLDRA) skulle bygges i Himdalen i Aurskog-Høland kommune i Akershus fylke.

Stortinget bestemte at avfallsbeholdere som inneholder plutonium utover en produksjon på 100 mg pr år skal lagres i anleggets lagerdel i påvente av en beslutning om endelig deponi av dette avfallet. Annet avfall skal støpes inn i sarkofager i anleggets deponidel.

Anlegget skal være i drift fram til 2030 og skal romme alt eksisterende radioaktivt avfall lagret og deponert opp til i dag, og det skal kunne ta imot alt avfall som produseres i Norge (med unntak av scale fra oljeindustrien) fram til år 2030. I år 2030 skal det tas stilling til den videre håndtering av avfallet i anleggets lagerdel, og deponiet skal lukkes. Det skal gjennomføres en overvåkning av utlekking fra deponiet i en periode på 300 - 500 år.

Den Internasjonale Atomenergikommisjonen IAEA har gjennomførte i 1995 en vurdering av anlegget gjennom sitt Waste Management Assessment and Technical Review Program (WATRP). Eksperter oppnevnt av IAEA, og som utgjorde WATRP-teamet, har avgitt sin innstilling i en egen rapport [1].

Det er bestemt at Statens bygge- og eiendomsdirektorat (Statsbygg) skal være byggherre og stå som eier av KLDRA-Himdalen mens IFE skal drive anlegget. I henhold til lover og forskrifter har Statsbygg søkt om konsesjon for bygging av anlegget

[2]. I sin konsesjonssøknad har Statsbygg beskrevet utforming og sikkerhet ved anlegget både på kort og lang sikt. Søknaden inneholder derfor en beskrivelse av geologi, hydrologi og andre forhold knyttet til stedsvalget og omfatter en sikkerhetsanalyse for unormale hendelser.

Som operatør av anlegget skal IFE søke om konsesjon for drift av anlegget. Denne søknaden er under arbeid.

Det forvente at bygging av anlegget vil starte i løpet av høsten 1996 og at anlegget vil stå ferdig for drift i løpet av 1997.

2. Avfallsvolumer og aktivitetsmengder

KLDRA-Himdalen er dimensjonert for å ta imot alt avfall som i dag er lagret og deponert på IFEs område på Kjeller og det avfallet som vil bli produsert i framtiden fram til år 2030. Dette omfatter alt driftsavfall som vil bli produsert ved IFEs anlegg fram til år 2030, annet radioaktivt avfall som produseres i Norge i denne perioden, og avfall fra nedlegging av IFEs nukleære installasjoner. Stortingets vedtak innebærer også at IFE må grave opp et overflatedeponi som ble etablert på Institutts område på Kjeller i 1970 og overføre avfallsbeholderne til det nye anlegget. Totalt gir dette følgende overslag over avfallsvolumer som skal lagres/deponeres. Mengden angis i tønneekvivalenter tilsvarende 210 liters ståltønner fordi det også benyttes andre typer avfallsbeholdere.

Deponert på IFE, Kjeller	:	1 000	tønneekvivalenter
Lagret i lagerbygg på IFE, Kjeller	:	2 000	---------"----------
Anslått driftsavfall 1995 - 2030	:	4 000	---------"----------
Anslått fra nedlegging av reaktorer og andre nukleære installasjoner	:	2 000	---------"----------
Sum	:	9 000	tønneekvivalenter

Det er anslått at den totale mengden lavt- og middels radioaktivt avfall som vil bli produsert i Norge opp til år 2030 tilsvarer 9 000 tønner a 210 liter. KLDRA-Himdalen er derfor dimensjonert til å ta imot 10 000 avfallstønner.

Siden det er stilt krav om at plutoniumsbærende avfall skal behandles spesielt er det gjort et separat overslag over mengden plutonium i radioaktivt avfall fram til år 2030. Dette er gitt i Tabell 1. Tabell 2 inneholder en oversikt over aktivitetsmengder for andre nuklider i avfallet produsert opp til 1992 og estimert for perioden 1992 - 2030. Verdiene er korrigert for decay fram til år 2030.

3. Geologi og utforming av KLDRA-Himdalen

Himdalen ligger ca 25 km sør for IFEs område på Kjeller. KLDRA-Himdalen skal bygges i fjell i en år med adkomstvei fra dalen. Fjellet består av prekambrium gneis og er den del av den gamle tektoniske strukturen i Skandinavia. Fjellet er tørt uten store sprekkdannelser. Den nærmeste svakhetssonen er i bunnen av Himdalen, men det har ikke vært vesentlig seismisk aktivitet i området de siste 100 millioner år. Belastninger fra jordskjelv i området er utredet som en del av konsesjonssøknaden for bygging [3].

Det konkluderes med at det er meget usannsynlig at jordskjelv vil forårsake vesentlig skade på anlegget [4].

Det skal bygges 4 fjellhaller hver med plass for 2 500 avfallstønner (Figur 1). De tre innerste hallene er deponidelen mens den ytterste hallen vil bli benyttet som lager. Deponidelen vil derfor romme 7 500 avfallstønner mens lagerdelen vil romme 2 500 tønner. Hallene har egne områder for manøvrering av transportkjøretøy. Adkomsttunnelen her en lengde på 138 meter. Både fjellhallene og adkomsttunnelen har et fall mot tunnelåpningen på 1:50. I adkomsttunnelen mellom tunnelåpning og fjellhallene vil det bli etablert lokaler for servicefunksjoner og besøksrom for omvisning (Figur 1).

I hver fjellhall vil det bli bygget 4 betongbunkere for avfallsbeholdere hvorav to og to henger sammen (Figur 2). På denne måten vil konstruksjonen bedre motstå jordskjelv enn en sammenhengene sarkofagstruktur. I deponidelen vil avfallsbeholderne bli omstøpt lagvis med 15 cm betongdekke på hvert lag. Etter at en bunker er fylt opp vil det bli montert et tak med vanntett membran som skal være i stand til å tåle steinras fra taket i hallen. Oppfylling av bunkerne og montering av tak vil bli en kontinuerlig operasjon Med unntak av drift i den første perioden hvor avfallsbeholdere som er lagret og deponert på IFEs område vil bli transportert inntil lagrene er tomme, vil avfallsbeholdere bli mellomlagret på IFE inntil det foreligger et tilstrekkelig antall for å fylle en bunker.

Alle avfallsbeholdere vil bli fraktet med lastebil fra IFE, Kjeller, til Himdalen. Beholderne vil bli lastet direkte fra bil til betongbunker med traverskran montert oppå bunkerne (Figur 2).

4. Hydrologi og dreneringssystem

Det nåværende grunnvannsnivået tilsvarer terrengnivået i området med unntak av toppen av åsen over anlegget. Etter utgraving av fjellhallene vil grunnvannsnivået synke til gulvnivå for hallene over en 15-års periode og stabilisere seg på dette nivå over en 55-års periode. Dette betyr at strømningsgradienten alltid vil være inn mot hallene og at utstrømning av vann fra hallene gjennom berget ikke er mulig. Dette umuliggjør transport av radionuklider ut av hallene gjennom berget og den eneste strømningsveien blir ut adkomsttunellen. Vannmengdene som vil strømme inn i hele anlegget er beregnet til mellom 0.14 og 1.4 m^3 pr time [5].

Det vil bli installert to dreneringssystemer. Et system skal samle opp og lede ut vann som renner inn i anlegget fra det omkringliggende berget. Det andre dreneringssystemet skal samle opp vann som av en eller annen årsak lekker ut fra betongsarkofagene og som kan inneholde radionuklider. På grunn av fallet mot tunnelåpningen vil begge systemer være selvdrenerende. Dersom dreneringssystemene blokkeres vil adkomsttunnelen fungere som dreneringskanal. To separate dreneringskummer vil bli etablert i serviceseksjonen, en for hvert dreneringssystem. Monitorering av dreneringsvann vil bli utført rutinemessig i hele driftsperioden fram til år 2030.

Radionuklideinnholdet i dreneringsvann og lekkasje fra deponiet vil bli overvåket i en periode på 300 - 500 år framover.

5. Radiologisk kartlegging

Et program for overvåking av radioaktivitet i omgivelsene rundt deponiet vil bli gjennomført i driftsperioden for å kunne påvise eventuell utlekkasje av radioaktivitet som skyldes avfallet. Men allerede nå finnes det radioaktive stoffer i naturen i dette området, både naturlige og slike som stammer fra prøvesprengninger i atmosfæren og fra Tsjernobylulykken. Det er derfor nødvendig med en forhåndskartlegging slik at referanseverdier kan etableres. Denne kartleggingen er allerede startet og omfatter representative prøver av grunnvann (i fjell og i drikkevannsbrønner), overflatevann, bergartsprøver (boreprofiler), sedimenter, myrjord og biota (fisk, melk, vegetasjon, viltkjøtt).

Slik deponiet konstrueres, er det sannsynlig at kun vann kan bli kontaminert. Men programmet omfatter også de andre prøveslagene for å få et mest mulig komplett sett av referanseverdier med tanke på fremtidige påstander om at radioaktiviteten i området skyldes deponiet.

Radionuklidene som inngår i kartleggingen er de som finnes både i det radioaktive avfallet og i omgivelsene. De viktigste er ^{3}H, ^{90}Sr, ^{137}Cs, ^{238}U og datterprodukter, Pu-isotoper og ^{241}Am. Både gammaspektrometri og radiokjemiske metoder vil bli benyttet.

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Table 1
Estimater over plutonium i avfallet i KLDRA-Himdalen

Kilde	Masse (g)	Aktivitet (GBq)	Kommentar
Nedgravd på IFE, Kjeller	35	84	Fra drift av URA [1)]
Lagret på IFE, Kjeller	10	24	Fra drift av URA [1)]
Lagret på IFE, Kjeller	1	4	Drift av Metallurgisk laboratorium opp til 1992
Drift av Metallurgisk	3	12	Anslått fra drift av Metallurgisk laboratorium mellom 1992 og 2020
Nedlegging av Metallurgisk laboratorium	41	168	Omkring år 2020
Nedlegging av Halden-reaktoren	10	41	Omkring år 2020
Fra andre kilder	9	21	Estimert fram til år 2030
TOTALT	110	354	

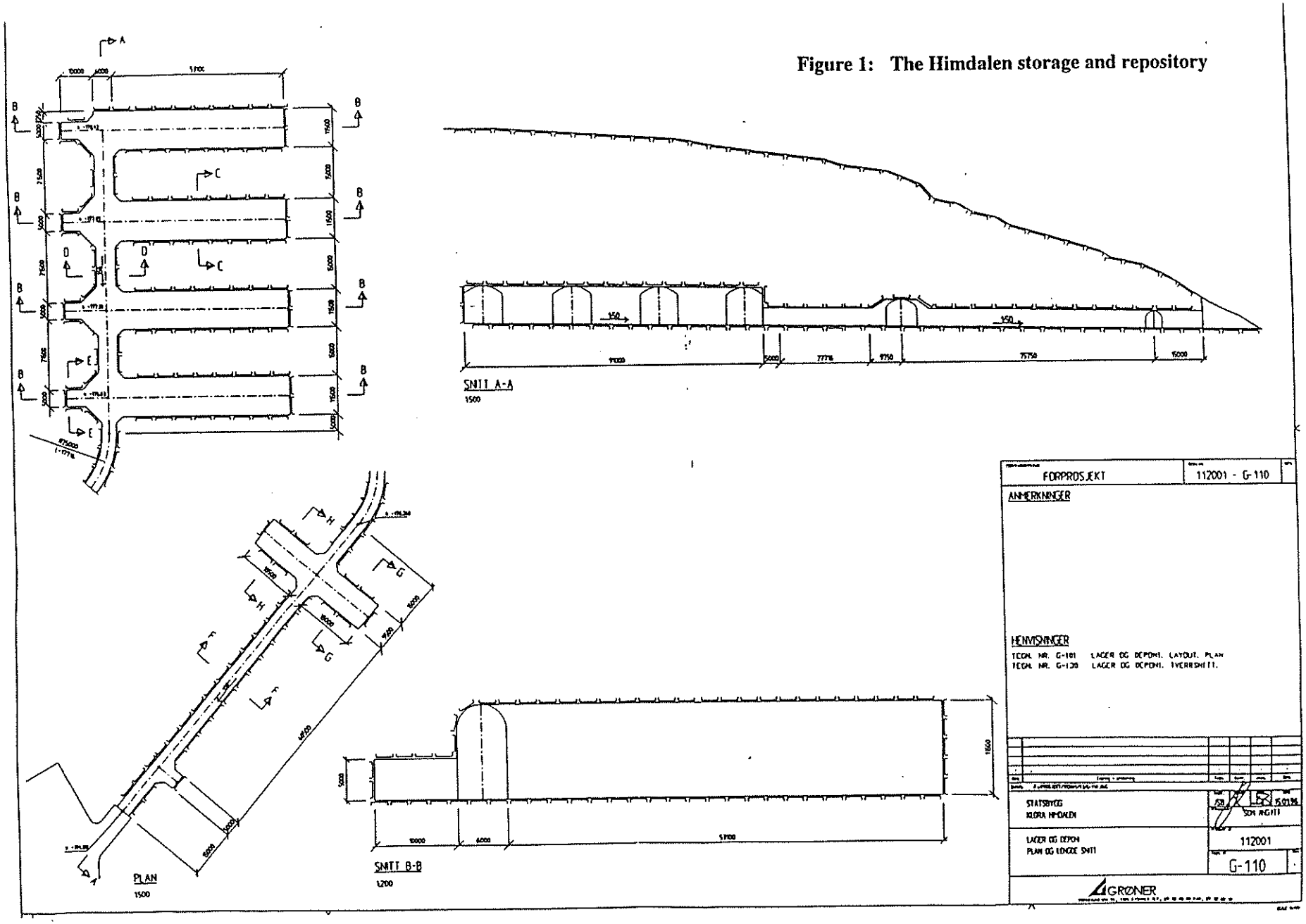
[1)] Uranium Reprocessing Pilot Plant

Table 2
Estimater over nuklidesammensetning og aktivitetsmengder i KLDRA-Himdalen

Nuklide	Halveringstid (år)	Lagret/deponert opp til 1992 [1] (GBq)	Estimert 1992 - 2030 [1] (GBq)	Totalt [1] (GBq)
^{60}Co	5,3	10	3 390	3 400
^{85}Kr	10,6	5	140	145
^{3}H	12,3	15 000	295 000	310 000
^{152}Eu	13	-	0.1	0.1
^{90}Sr	28	3 600	16 000	19 600
^{137}Cs	30	5 700	23 000	28 700
^{241}Am	458	11,5	5 330	5 340
^{226}Ra	1 622	11	20	30
^{14}C	5 760	40	108	148
^{41}Ca	103 000	-	50	50
^{36}Cl	310 000	-	0.4	0,4
$^{239,240}Pu$	24 000/6537	112	242	354
^{238}U	$4,5\ 10^{9}$	1,3	0,05	1,4
TOTALT	≈ 24 500		≈ 343 300	≈ 367 800

[1] Korrigert for decay pr år 2030

Figure 1: The Himdalen storage and repository



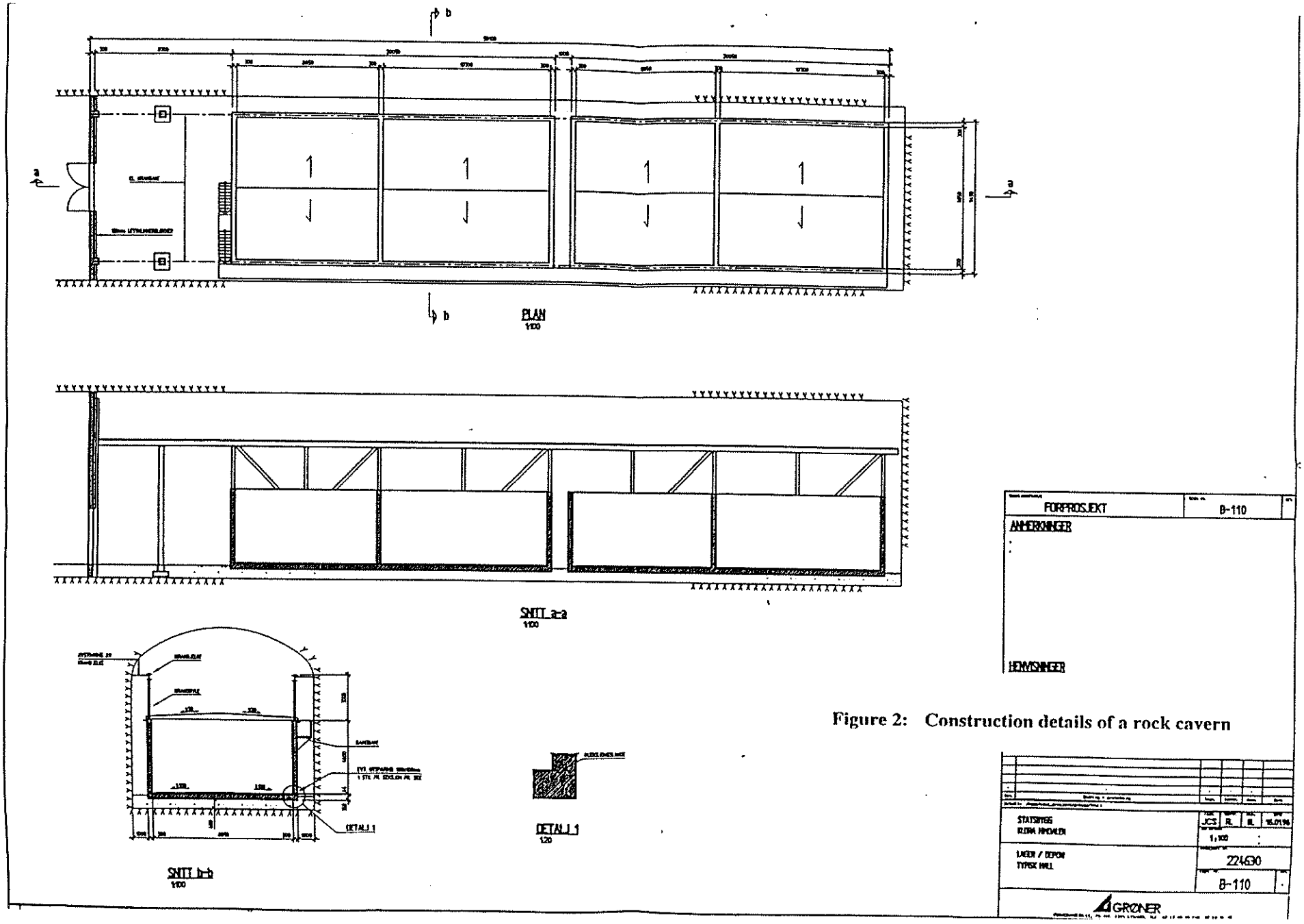


Figure 2: Construction details of a rock cavern

IAEA - WATRP REVIEW OF THE NORWEGIAN COMBINED STORAGE AND DISPOSAL FACILITY FOR LLW AND ILW

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ABSTRACT

The Norwegian Storting (Parliament) decided in April 1994 that a combined storage and disposal facility should be built in Himdalen. The Storting however recommended that a review of the storage and disposal facility should be made within the Waste management Assessment and Technical Review Programme (WATRP) of the International Atomic Energy Agency (IAEA), before a construction licence could be granted.

In December 1994 the Norwegian Radiation Protection Authority (NRPA) requested a WATRP review from the IAEA. The scope included review of the legal framework, long term safety and the site selection process. The review team consisted of five members with Mr. Zurkinden, Switzerland as the chairman. In September 1995 a review meeting was convened in Oslo where the team and IAEA representatives met with Norwegian experts and also visited the Himdalen site. The chairman presented the team's conclusions and recommendations at a press conference the last day of the meeting. Within the scope of the review the team is satisfied and believes that Himdalen with the technical concept is a suitable site.

NRPA is now in the process of handling the application for the construction licence, and in this work it is very valuable to have the review report as a "check list". The builder, Statsbygg has also used the recommendations of the team when making the final design and safety reports.

INTRODUCTION

The Norwegian Radiation Protection Authority (NRPA) is the regulatory body in Norway for both radiation protection and nuclear safety.

In 1989, a committee was appointed by the government to investigate possible solutions for final disposal of all the Norwegian LLW and ILW (1).

In 1992, an impact assessment was performed including three possible disposal sites, with a recommendation for one of them. This was an engineered rock cavity facility in Himdalen, 25 km from the Kjeller waste conditioning plant (2). During the Parliamentary committee deliberations it was proposed that it should be a combined storage and disposal facility, with storage of the plutonium bearing waste and disposal of the short lived waste.

In April 1994 the Storting (Parliament) decided to go on with investigations at the Himdalen site and that it should be a combined facility, it was also recommended that an IAEA - WATRP review should be performed before the granting of a construction licence. In December 1994 NRPA sent such a request to IAEA.

Norway does not have any nuclear power plants, although there are two research reactors.

The Institute for Energy Technology (IFE) operates these two reactors located in Halden, 10 km south-east of Oslo and at Kjeller, 25 km east of Oslo. IFE is an industrial foundation funded from the national budget and from commercial research programmes.

Main research areas at the Halden reactor are reactor safety, technological research and development. Fuel testing, the OECD/Halden project, and research on man-machine interactions are two important tasks. The waste is mainly ion exchange resins. Tested fuel elements are returned to their owners. Spent fuel from the reactors is stored at the sites.

The JEEP II reactor at Kjeller is used to produce pharmaceutical products and irradiation services for medicine, industry and research. Neutron beams from the reactor are used to study the fundamental physical characteristics of solids and liquids.

The Norwegian waste originates from activities at IFE and from industrial and medical applications.

Between 50 and 100 drums of waste are generated per year. About 80 per cent of the waste volume originates from the activities connected to the research reactor programme, while industry and medicine generate about 10 per cent each. Up to the year 2030 it is estimated that an equivalent of about 10 000 drums of low and intermediate level waste with a total activity of approximately 370 TBq including about 110 g of plutonium will have been generated (3).

DESCRIPTION OF THE COMBINED STORAGE AND DISPOSAL FACILITY

The combined storage and disposal facility will be built into a hillside in crystalline bedrock. The rock formation on the Himdalen site is Precambrian Mylonitised Gneiss.

The facility will have four caverns for the waste packages and one slightly inclining 150 m long access tunnel for vehicles and persons. A service and control room, with certain service functions for the personnel, will be built along the tunnel and there will be a visitors' room inside the facility. The rock caverns will be excavated with about 50 meters of rock covering. The geological covering is for protection against intruders, plane crashes etc. and it is not intended to be taken into account as a main barrier in the long term safety calculations.

In each cavern a solid sarcophagus incorporating the concrete floor and walls will be created. The roof of the sarcophagus will be shaped to shed infiltrating ground water and a waterproof membrane will be fixed to the concrete roof. Three caverns will be used for disposal with the drums stacked in four layers. One cavern is for storage.

Operation is planned to start in 1997, the operational period is expected to last until the year 2030, this would take care of all Norwegian low and intermediate level waste, including decommissioning of the research reactors and laboratories at IFE. Based on the experience during the operational period and safety reports for the closure, it will be decided whether to retrieve or dispose of the waste in the storage part by encasing it with concrete. During operation one does not intended to retrieve any of the waste that has been placed into the storage facility.

An anticipated requirement is that the caverns must be maintained in a drained condition after closure of the repository. The facility will therefore be closed by backfilling the disposal caverns in such a way as to permit functioning of the drainage for a very long time.It is intended to have an institutional control period of 300 -500 years with monitoring the discharges and restrictions on land use.

THE WATRP REVIEW

The Waste Management Assessment and Technical Review Programme (WATRP) has been developed by the International Atomic Energy Agency (IAEA) to serve Member States through technical assessments and peer reviews of their national policies, programmes or facilities related to the management of radioactive waste (4).

Upon the request of a Member State, or an organization within a Member State, the IAEA undertakes the responsibility of convening an international panel of experts to perform such a peer review in accordance with the terms of reference established by the requesting Member State or organization.

The mechanism used for this purpose typically includes a review of source material, an exchange of information with the applicant's experts in a review meeting and a compilation of the findings in a review report.

The advantage of such a peer review for the requesting Member State or organization consists of obtaining independent international expert opinions and advice. The experts perform their reviews in their personal capacities independent of their governments or the IAEA, which means that the final review results present the insight of the experts and not necessarily of their country or organization or the IAEA.

The conclusions and recommendations of the WATRP experts are based on IAEA standards, internationally accepted principles and proven practices taking into account the general consensus on the most viable options for solving particular tasks .

A prepatory meeting in Oslo with representatives from IAEA was held to discuss the scope of the review, the time schedule and possible experts (to avoid that IAEA would contact experts that had been involved in the process earlier). In December 1994 NRPA sent the request to IAEA for a WATRP review (5).

The scope of the requested Norwegian review was to evaluate:
- the approach to the selection of a site
- the technical concept ("mountain hall", combined storage/disposal)
- the long term safety of the facility

The main purpose was to review the safety and radiation protection aspects of the chosen solution as outlined: Waste management in Norway:

1. Legal framework
   - organisations involved/independency
   - licensing procedure
   - long term safety criteria

2. Concept and site
   - host rock
   - design of the facility
   - waste inventory
   - storage/repository
   - post operational phase
3. Safety assessment
   - input data
   - scenario
   - dose calculations

NRPA provided the expert team with an overview report of the siting procedure, the legal framework, the site and the concept (Feb. 95) in order to make them acquainted with the situation in Norway.

All relevant Norwegian experts currently working on the project and also those that had been involved earlier participated at the review meeting. Two days of the meeting were used for discussions, one day for an excursion to the Himdalen site and to the waste conditioning facility at IFE and two days for drafting the review report. On the third day the team presented their preliminary conclusions for discussion with the Norwegian experts and then started to write the report. The review meeting ended with a press conference and a press release.

The main findings, views, conclusions and recommendations of the team are summarised in the review report. The report was printed in a limited number and as confidential. IAEA submitted the report to the team members and NRPA. The report was translated and declassified and published as a two language document, the original received from the IAEA and the Norwegian version, by NRPA (5).

After the press conference five representatives from groups of concerned sitizens had a long discussion with the chairman Mr. Zurkinden and the IAEA scientific secretary Mr. Warnecke. The members of the groups claimed that the team had not been provided with a complete documentation. It was of course a big disappointment for them that the team approved the site and the technical concept chosen.

The WATRP press release was covered by TV (short announcement at the news and text-TV) and the main newspapers. They were not present at the press conference. The local press (small communities) has been following the process closely, especially the initiatives of the groups of concerned citizens.

WATRP TEAM RECOMMENDATIONS - FOLLOW UP

Within the scope of the review, the team was satisfied with the overall approach taken by the responsible Norwegian organization in the development of the storage and disposal facility for low and intermediate level waste. Based on the existing information the review team believed that the Himdalen site, in combination with the engineering concept (sarcophagus), can be suitable for the storage and disposal of the relatively small amounts of Norwegian low and intermediate level waste. The review team emphasised that it is only necessary to find a

suitable site, and in fact it is not necessarily possible to find the best site. The team encouraged the responsible Norwegian organisations:

- to select a storage and disposal concept among the various options and develop the detailed plans for this concept;
- to excavate the respective storage and disposal rooms and collect further detailed site specific data;
- to update the safety assessment accordingly; and
- to confirm the suitability of the site (5).

Statsbygg has used the recommendations from the team when making the license application and the safety reports. In the application they have listed all recommendations relevant for them and showed in which way they have been taken into account.

Some examples; "a design earthquake should be established for the storage and disposal facility and the potential impacts of such an earthquake on the integrity of the sarcophagus, the groundwater inflow conditions, rockfall and other relevant aspects and components of the disposal facility should be assessed". Earthquake design was done by NORSAR (6) and calculations by NGI (7). "The gas generation in the storage and disposal facility and the consequences for the facility should be evaluated". This has been calculated by AEA Technology (UK) in connection with the scenario and release calculations (8). AEA Technology is a subcontractor to Statsbygg.

The team has reviewed the legal framework and the requirements set by the authorities for the applications and the safety reports. In the review report the team has discussed these aspects and agrees with the NRPA criteria. This is a help for the NRPA when handling the license applications and safety reports. The review report can also be used as a "check list".

The IAEA-WATRP report is also an important support when answering questions raised by politicians and the media.

CONCLUSION

The WATRP review contributes to improving the confidence level of the planned waste management systems, and helps to ensure that the systems perform in a safe and reliable manner. The WATRP review also assists in improving public acceptance of the planned storage and disposal facility.

The review report is written in short, numbered paragraphs and the recommendations are highlighted. This ensures its easy use and easy reference.

The WATRP review has turned out to be a useful assessment of the work done during the siting activities, legal framework and the technical concept. The review report from IAEA, written in the personal capacity of five international expert, presents itself as an objective report that Norwegian authorities and others can use in assessing waste management practises in Norway now and in the future.

The NRPA can highly recommend this service provided by the IAEA. The review process is carried out in a professional way with highly qualified experts. To use experts from different countries and from different parts of the world adds an (extra) international weight to the

report. Norwegian authorities would consider the use of the service again whenever a waste management project is developed.

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Recommendations on Exposure to 50 Hz Magnetic Fields in Sweden

Enn Kivisäkk, Lars Erik Paulsson.

Abstract

The Swedish Radiation Protection Institute (SSI) has in 1993 issued recommendations on 50 Hz magnetic fields. The recommendations are based on a "prudent avoidance" approach. Several official authorities in Sweden are collaborating to further promote this issue. The recommendations state:

* *When installing new equipment or new power lines that may cause high magnetic field exposures, solutions giving lower exposures should be chosen provided these solutions do not imply large inconveniences or costs.*

* *In cases where the exposure levels exceed the normally occurring levels by more than tens of times steps can be taken to reduce the magnetic field, provided such reductions can be done at a reasonable cost.*

* *In situations where the exposure levels from existing installations do not exceed tens of times the normally occurring levels, costly rebuilding should be avoided. "*

SSI has used its policy statement when responding to specific questions about actual situations. The SSI policy has at time being not been used for generating regulations or mandatory restrictions. Later during 1993 an information brochure was issued for use by the general public. The brochure was a result of an interagency collaboration with the following participants: National Board of Housing, Building and Planning, National Electrical Safety Board, National Board of Health and Welfare, and Swedish Radiation Protection Institute. The SSI recommendations were essentially copied in that brochure.

For the occupational environment there is a long tradition of mandatory regulations for identified risk factors. In Sweden such factors are usually identified and the basis for the regulations developed by so-called criteria groups within the National Institute of Occupational Safety and Health (now National Institute of Working Life) and in which SSI takes an active part. The Criteria Group for Physical Risk Factors (chaired by Prof. B Knave) concluded in 1995 that there was no scientific basis for traditional regulations regarding low frequency magnetic fields. This conclusion was based upon a comprehensive report from a working group on an evaluation of all the epidemiological and experimental studies on cancer and low frequency fields.

Optical radiation from two industrial laser stations

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Abstract - Two powerful laser working stations have been studied. One CO_2-laser (10600 nm) at the Technical University in Luleå, Sweden and one quadruple-coupled Nd-YAG laser (1060 nm) at the Force Institute, Copenhagen, Denmark.

Several different types of processes were studied by the CO_2-laser, run at laser power from 1000 W to 4500 W. The Nd-YAG-laser was run at 2000 W in all cases.

Measurements have been done of reflected laser radiation and, spectrally, in the ultraviolet and visible regions.

The results were assessed versus exposure threshold limits given by A.C.G.I.H. Radiation levels are calculated and shown at the distance 0.5 m.

The most impressing finding was the very powerful optical source created by plasma in CO_2-welding as well as surface alloying. Especially when argon was involved as shielding gas or carrier gas large emittance of ultraviolet and short-wave visible radiation was recorded. The shortest times giving the threshold ultraviolet exposure is around 0.5 second. Also the time limits for retinal burn hazards are very short, in the order of milliseconds. In these processes reflected laser radiation are however not hazardous. Reflected CO_2-laser radiation, occurred rather in processes where the laser beam was not focused such as in surface treatments.

The Nd-YAG-laser did not create a plasma. Small glowing particles were thrown out from the material of moderate temperature emitting a spectrum close to black-body radiation. No significant ultraviolet radiation was emitted. However, in all cases reflected Nd-YAG radiation exceeding the exposure limits were found.

Introduction

This work has been done as a part of an Eureka-project (643) concerning the working environment in general around industrial lasers, used for example for welding, cutting or surface treatment. The Swedish part of this project is managed by the Technical University in Luleå. This paper deals with measurements and risk assessments of optical radiation that is emitted from different industrial processes.

Two powerful laser working stations have been studied. One CO_2-laser (10600 nm) at the Technical University in Luleå, Sweden and one quadruple-coupled Nd-YAG laser (1060 nm) at the Force Institute, Copenhagen, Denmark.

Instrumentation

Spectral measurements were performed with an Optronic 742, provided with a double monochromator, for ultraviolet and visible radiation, scanning every 5th nm between 230 and

780 nm. The spectral data were recorded by a small computer HP85 which also controlled the radiometer. Later the recorded files were transferred into the PC-world for further analysis.

Reflected CO_2 radiation was measured by a Laser Precision Rk 3440 pyroelectric radiometer which was connected to an Y-T recorder.

An UDT 40X Opto meter was used, in particular, for measurements of reflected Nd-YAG laser radiation (which the Optronic was not able to see). The UDT was filtered by a Kodak Wratten filter 87A. The response of the UDT was thus reduced within a narrow spectral band 960 - 1100 nm. Without filter the spectral range is 380 - 1100 nm.

Measurements procedures

Reflected CO_2 laser radiation

The pyroelectric radiometer has a very wide spectral response. It can see the CO_2 wavelength as well as ultraviolet, visible and near infrared radiation. In order to measure the laser radiation only, the visible and near infrared radiation was in some positions measured simultaneously by the unfiltered UDT. The Rk 3440 readings reduced by the readings from the latter instrument should be regarded as the proper values of the laser radiation. However it turned out that there was no need for such adjustments in surface treatment since the readings of visible and near infrared from the irradiated spot in most positions were several orders of magnitude less than the readings of the total radiation. In welding and cutting when plasma was produced the radiation was dominated by ultraviolet and visible (referred to as secondary emission) and thus measured spectrally.

The Rk 3440 radiometer was placed in several different positions around the work-piece directed towards the irradiated spot. A system of co-ordinates was invented the origin of which was the irradiated spot at the work-piece. As the readings varied in time the radiometer was connected to an Y-T recorder.

The object to be measured was a CO_2-laser station melting a hard coating on a work-piece (an iron tube). The tube was rotating and the laser scanned axially a length of about 0.1 m. Thus the coating was applied like a thread. In this coating process the laser beam is unfocused. Another set-up, something like cutting, was also measured. In all occasions the laser output power was set to 3 kW.

Spectral measurements of ultraviolet and visible radiation.

Secondary emission was measured in a number of welding or surface alloying procedures using some different settings of laser power. Measurements were made at every fifth nanometer in the wavelength range 230 - 780 nm using an Optronic 742 spectroradiometer. The radiometer was placed at distances where good readings were obtained without overflows.

All results were afterwards normalised to a distance of 0,5 meter using the inverse square law, an operation that was regarded legal since the plasma droplets were small with respect to the distances involved. The distance was chosen to allow comparison to data on conventional welding where 0.5 metres, or close to it, is often found in literature (1, 2, 3). Frequently the ambient light was measured, the results of which were spectrally subtracted from the secondary emission data prior to calculating the irradiance at 0,5 meter.

Reflected NdYAG radiation

The instrument available that could see radiation of wavelength 1060 nm was the UTD 40X radiometer. In order to make the rather broad-band instrument selective a Wratten 87 A filter

was attached to the detector. Thus the instrument became sensitive in a rather narrow band between 960 nm and 1100 nm. The UDT instrument was also provided with a 7 mm circular entrance aperture.

The previous spectral measurements showed that the secondary emission from the NdYAG laser procedures mainly is thermal, i.e. follows Planck's law. The expected contribution from that black body radiation could be extrapolated and subtracted from the readings on the total radiation.

Several positions versus the angle of the laser beam incidence were mapped.

Results and risk assessment

Reflected CO_2 radiation

The results show that a considerable fraction of the laser radiation is reflected when the beam is not focused. Since the beam never is aimed exactly perpendicular to the work-piece, in which case the reflected radiation will go back and destroy the optics, there is generally a not shielded direction out of the work-piece into the room. Irradiances exceeding the maximum permissible exposure (MPE) (4) were found in several positions in proximity to the reflection angle.

In those cases where the laser-beam was focused the main part of the laser energy was absorbed into the work-piece. Reflected laser radiation exceeding the exposure limits was never found.

One example of an Y-T recording concerning reflected CO_2 radiation is shown in the diagram 1.

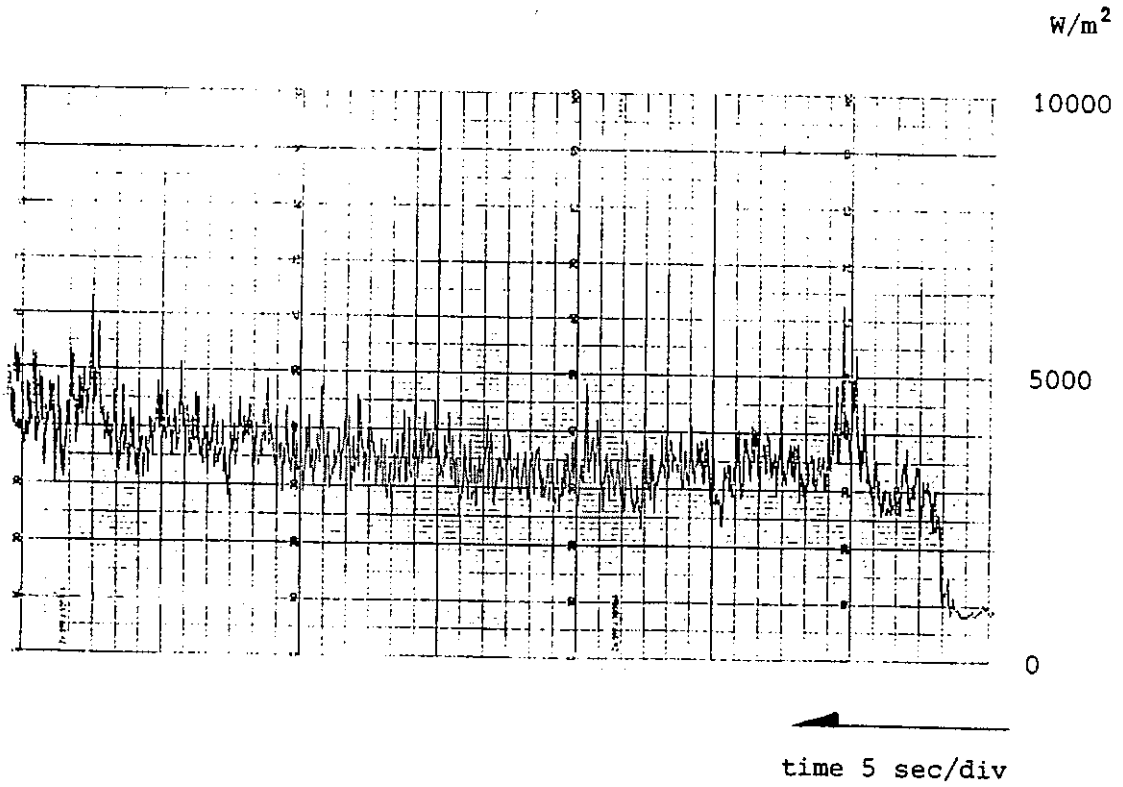

Diagram 1. Irradiance of reflected CO_2 radiation close to the reflection angle. The distance is 0.6 metres.

Defocused CO_2 laser processes, for example coating and surface treatment, give reflected laser radiation exceeding the MPE in particular in the geometric angle of reflection. The MPE-values for 10600 nm are rather generous due to the fact that the long wavelength radiation does not reach the retina and thus is not focused into the eye (5). The long term MPE (1000 W/m^2), for eyes as well as skin, implies that a sensation of heat is clearly present in the neighbourhood of the MPE.

The example above is one of the strongest reflexes found in this study. A safety distance for long term exposure would be around 1.5 metres.

Ultraviolet and visible radiation

Spectral measurements on the CO_2 laser processes where a focused beam was used showed the emission from the created plasma. The spectral results demonstrated powerful emission of ultraviolet radiation as well as visible radiation, in particular blue, coming from small plasma droplets. Examples on results are shown in diagrams 2 and 3.

The results were evaluated according to exposure limits given by A.C.G.I.H. (6) concerning exposure to ultraviolet radiation, blue light hazards and retinal burn hazards. It turned out that the NdYAG process did not create a plasma as in the case of the CO_2 laser. The secondary emission came from small glowing particles sparking out from the work-piece. No emission of importance of ultraviolet or retinal burn hazards was found. An example is shown in diagram 4.

A compilation of all spectral measurements with calculated exposure times is shown in table 2.

With regard to the ultraviolet radiation eye protectors, as a minimum, are strongly recommended, though a face shield is better. Even if the irradiance decreases as the inverse of the squared distance the biologic effect is accumulative in a working day and the recommended exposure limits are easily exceeded. For direct viewing of the plasma dark glasses are needed.

Reflected Nd-YAG radiation

A general observation was that the irradiance of reflected laser radiation peaked at the angle of reflection. Moving in the same plane the readings decreased from the angle of reflection. However there were irregularities not allowing a general description of the angle dependent irradiances. Some results are shown in table 1.

Process			Irradiance 0° from the angle of reflection (W/m^2)	Irradiance 45° from the angle of reflection (W/m^2)	Irradiance 90° from the angle of reflection (W/m^2)
Mild steel welding	He	focus 2 mm under surface	460	330	11
Stainless steel welding	He	focus at surface	110	65	9
Stainless steel welding	He	focus 2 mm under	220	160	7
Stainless steel welding	Ar	focus 2 mm under	370	180	15
Stainless steel welding	Ar	focus at surface	550	55	9

Table 1. Reflected Nd-YAG radiation for different processes and different angles

The MPE-values imply short exposure times. For instance, an irradiance of 200 W/m^2 allows an exposure time for 0.04 seconds. For long term exposure a safe level is 16 W/m^2. The corresponding safety distances will obviously be considerable in some directions. Laser safety goggles fit for 1060 nm are needed.

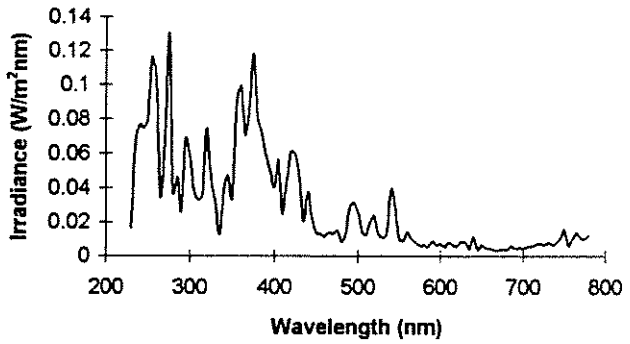

Diagram 2 Mild steel welding, CO_2 4500 W, shielding gas He

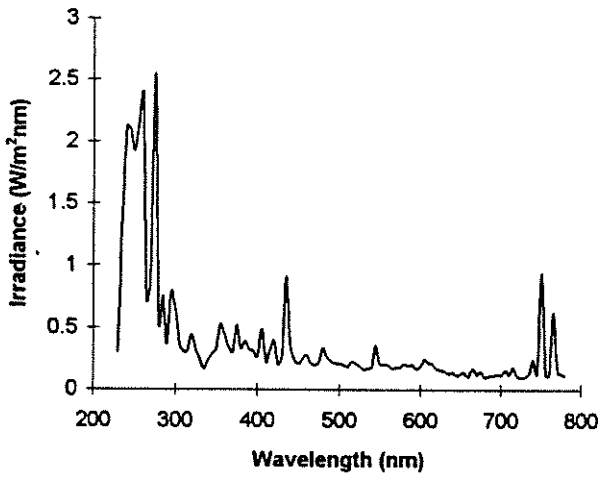

Diagram 3 Mild steel welding, CO_2 4500 W, shielding gas Ar
Note the different scale of the Y-axis. When argon is involved strong ultraviolet radiation is emitted.

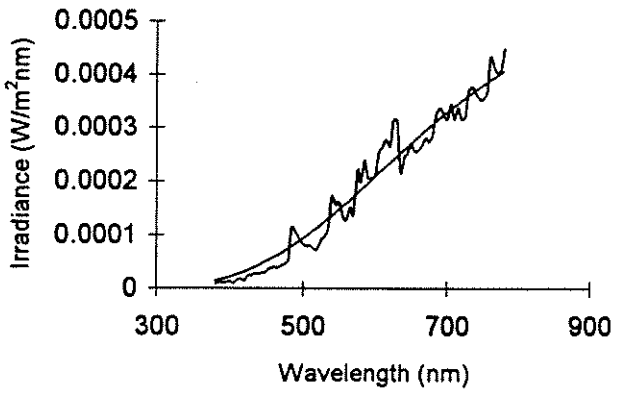

Diagram 4 Stainless steel welding, Nd-YAG 2000 W, Shielding gas He. Smooth line is the black body radiation curve at a temperature of 2800 K

	UV_{eff}	Area m^2	E_{BLUE}	E_{BURN}	t_{UV} sec	t_{BLUE} sec	t_{BURN} sec
MS CO_2 1300W He	0.079	2.00E-06	0.051	0.585	379.75	157	> 10
MS CO_2 4500W He	2.93	7.07E-06	1.92	21.4	10.24	14.7	4.85
MS CO_2 3800W He	1.54	7.07E-06	0.6	6.91	19.48	47.1	> 10
MS CO_2 4500W N_2	1.76	3.14E-06	2	22.7	17.05	6.28	1.91
MS CO_2 4500W Ar	55.6	2.00E-05	21.6	261	0.54	3.7	0.07
SA CO_2 3600W Ar Ar	75.3	7.07E-06	33.5	382	0.40	0.844	0.015
SA CO_2 3600W He Ar	13.6	7.07E-06	2.63	29.3	2.21	10.8	2.59
SA CO_2 3600W He Ar	47.5	8.00E-06	7.95	86.6	0.63	4.03	0.28
SA CO_2 3600W Ar Ar	71.2	6.30E-05	26.9	307	0.42	9.37	0.17
SA CO_2 3600W Ar Ar	41.7	6.30E-05	17.1	193	0.72	14.7	0.43
Surf. treat. CO_2 1050W	9.22E-04	1.02E-05	0.113	2.91	> 30 000	361	> 10
MS NdYAG 2000W Ar	1.01E-04	1.80E-06	2.20E-02	1.04	> 30 000	327	> 10
MS NdYAG 2000W He	1.11E-05	7.90E-07	2.10E-03	9.46E-02	> 30 000	1500	> 10
SS NdYAG 2000W He	6.10E-06	1.50E-04	2.50E-03	8.98E-02	> 30 000	> 30 000	> 10
SS NdYAG 2000W He	5.20E-05	1.50E-04	8.90E-03	2.68E-01	> 30 000	> 30 000	> 10
SS NdYAG 2000W Ar	0.000349	1.50E-04	0.048	1.27	> 30 000	12500	> 10
SS NdYAG 2000W Ar	6.67E-05	1.50E-04	0.011	0.38	> 30 000	> 30 000	> 10

Table 2 A compilation of the results of the spectral measurements. UV_{eff} is the biologically weighted irradiance (W/m^2) for ultraviolet. Area is the area of the plasma droplet. E_{BLUE} and E_{BURN} are the weighted irradiances for retinal burns and blue light hazards respectively. t_{UV}, T_{BLUE} and T_{BURN} are the maximum recommended exposure times. Exposure times longer than 30000 or 10 seconds respectively could be regarded as safe.

Abbreviations used: MS = mild steel welding SA = surface alloying SS = stainless steel welding He Ar (for instance) means helium as shielding gas and argon as carrier gas.

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Testing of the Sunscreens on the Swedish Market

Enn Kivisäkk, Ulf Wester, Olle Larkö

Abstract

Different kinds of skin cancers caused by exposure to solar ultraviolet radiation constitute a steadily growing health problem in Sweden. A nation-wide campaign was started at the end of the eighties in order to reduce the incidence of such health injuries. Recommendations for sun protection form an important element in the campaign.

This survey was carried out as one element of the above-mentioned campaign. The objective has been to procure information on the testing of sunscreen products marketed in Sweden, and to form a basis for regulations or guidelines in this field.

The questionnaire inquired about respective company's number of products, their Sun Protection Factor (SPF), how the SPF was determined, if UVA-protection was assessed, production quality assurance, storage durability, declaration of contents and about toxic, allergenic or carcinogenic qualities of the product or of it's ingredients.

Information on the testing of the sunscreens marketed in Sweden 1993 was investigated by means of a questionnaire, sent out to all companies marketing such products. Also questions concerning the quality control of the products were included. Information was returned by approximately half of the companies.

The answers show that most of the sunscreens are tested for one or more properties relevant for the effectiveness and safety of the product. Different methods and standards are however employed, which among other things means that the results of the testing are not comparable. A majority of the companies express a desire for standardized guidelines for the testing and quality control of the products.

The authors suggest some recommendations to the Swedish authorities concerned to take steps in order to improve the testing and control of the products.

A sensitivity analysis of a radiological assessment model for Arctic waters

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Abstract A box model has been used to calculate the environmental dispersion of radionuclides in Arctic waters and the subsequent exposure of man from a range of exposure pathways. A sensitivity analysis has been carried out to identify components of the model that are potentially important contributors to the predictive accuracy. The components investigated include features associated with water transport and mixing, particle scavenging, water-sediment interaction and biological uptake.

Introduction

Information on disposal of radioactive waste in the shallow waters of the Arctic Seas by the former Soviet Union became available to the international community from official Russian sources in 1993 in the White Book [1]. Since then several initiatives have been taken to assess the potential impact to man and the environment from this waste. Dumping has taken place in the Barents Sea west of Novaya Zemlya and in the Kara Sea east of Novaya Zemlya during the period 1960-1991.

The countries bordering the Arctic Seas are particularly concerned about the possible contamination of marine produce from these waters. Norway and the Russian Federation have entered into close co-operation to investigate the present levels of artificial radionuclides in the environment near the sites where solid reactor waste and spent nuclear fuel have been dumped in the Kara Sea. The International Atomic Energy Agency (IAEA) launched in 1993 the "International Arctic Seas Assessment Project (IASAP)" which is planned to last for four years. The programme is organised by IAEA in co-operation with Norway and Russia. Several projects supported by the European Union include supplementary investigations of the potential impact. The present paper describes a parameter sensitivity analysis of a radiological assessment model for the Arctic waters for the purpose of identifying important contributions to the predictive accuracy of the model.

Materials and methods

Model description

The marine dispersion model covers the Arctic Ocean and the North Atlantic Ocean. Box-model analysis used to simulate the movement of radioactive material between the different boxes into which the marine environment is subdivided. This type of modelling assumes instantaneous uniform mixing within each box with rates of transfer being proportional to the inventories of material in the source boxes.

Box models have been used in connection with studies of the dispersion of discharges of radioactivity from European civil nuclear installations [2]. The present model is a combination of two models: 1) an adjusted version [3] of a regional box model used for radiological assessments in north-west European coastal areas [4]; and 2) a larger box model covering the Arctic Ocean and the North Atlantic [5]. The larger box model is derived from a world ocean General Circulation Model [6] from which the results have been used for the design of a box structure in the Arctic Ocean and surrounding waters.

The box-model analysis uses first order differential equations to describe the transfer of contaminants between the boxes. The equations are of the form:

$$\frac{dA_i}{dt} = \sum_{j=1}^{n} k_{ji} A_j - \sum_{j=1}^{n} k_{ij} A_i - k_i A_i + Q_i$$

where $k_{ii}=0$ for all i, A_i and A_j are activities (Bq) at time t in boxes i and j, k_{ij} and k_{ji} are rates of transfer (y^{-1}) between boxes i and j, k_i is an effective rate of transfer of activity (y^{-1}) from box i taking into account loss of material from the compartment without transfer to another, for example radioactive decay, Q is a continuous source of input into box i (Bq y^{-1}) and n is the number of boxes in the system.

The rates of transfer between the aquatic boxes, k_{ij} are related to the volume exchanges, R_{ij}, according to:

$$R_{ij} = k_{ij} \cdot V_i,$$

where V_i is the volume of water represented by box i.

Figure 1 shows the regions used in the marine box model in the Arctic Ocean. Each of the water compartments contains associated suspended sediment, and the water compartments in contact with the seabed have underlying seabed sediment compartments. Details of compartment volumes, depths, suspended sediment loads, sedimentation rates and volume exchange rates may be found in [7]. The suspended sediment loads and sedimentation rates are default values from the regional model mentioned above [4].

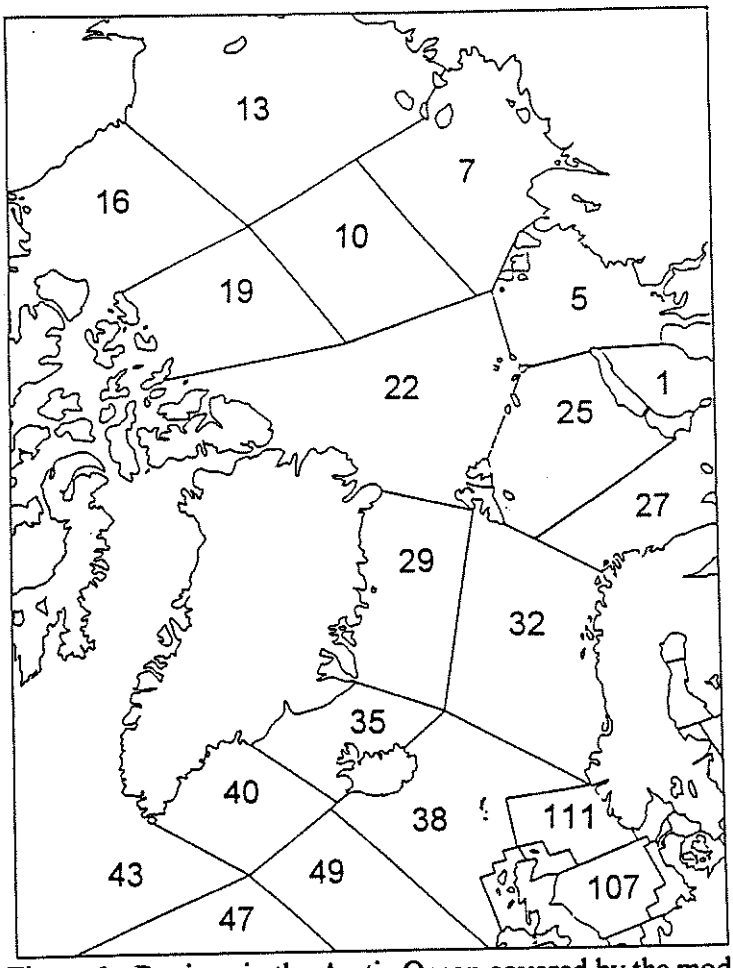

Figure 1. Regions in the Arctic Ocean covered by the model. The numbers refer to surface water boxes.

At any given time the activity in the water column is partitioned between the water phase and the suspended sediment material. The fraction of the activity (F_W) in the water column which is in solution is given by:

$$F_W = \frac{1}{1 + K_d SSL},$$

where K_d is the sediment-water distribution coefficient and SSL the suspended sediment load. Activity on suspended sediments is lost to the underlying boxes when particulates settle out. The fractional transfer from a water column (box i) to the sediments (box j) due to sedimentation is given by:

$$k_{ij} = \frac{K_d SR_i}{d_i (1 + K_d SSL_i}),$$

where d_i is the mean water depth of the water column and SR the mass sedimentation rate.

Transfer between the sediment box and the water column is parameterized by diffusivity through the pore water and diffusivity due to bioturbation which gives the following transfer across the boundary:

$$\frac{dA}{dt} = (B + \omega D) S \frac{C_S - C_W}{h_B},$$

where A is the radionuclide inventory in the water column, B the bioturbation coefficient, ω the sediment porosity, D the pore water diffusion coefficient, S is the surface area of the interface, C_S the radionuclide concentration in sediment pore water, C_W the radionuclide concentration in the water column, and h_B the thickness of the bioturbated layer. Furthermore, removal of activity from the top surface sediment to lower sediment layers is taken into account by assuming that the burial rate is equal to the flux of particles which settle from the overlying waters. Radioactive decay is included in all the boxes.

The contamination of fish, crustaceans and molluscs is calculated from the radionuclide concentrations in filtered seawater in the different water regions. For this purpose, concentration factors for biological material are used. Data for the world catch of seafood in the various regions have been compiled from the Marina Study [2] and FAO [8]. The following assumptions for the edible fractions of seafood catch are used: 50% for fish, 35% for crustacea and 15% for molluscs. Collective doses to the world population are based on the ingestion of seafood only. Critical groups have been assumed located various distances from the dump sites: on the Yamal peninsula at the Kara Sea, on the Kola peninsula at the Barents Sea and in northern Norway. The following assumptions have been made for their habits for the estimation of radiation doses: ingestion of fish (200 kg y^{-1}), crustaceans (50 kg y^{-1}) and molluscs (50 kg y^{-1}); exposure to external gamma radiation for 250 h y^{-1} in coastal areas; and inhalation for 2000 h y^{-1} of resuspended beach sediment and seaspray at concentrations of 0.25 μg m^{-3} and 10 g m^{-3}, respectively. Furthermore, a hypothetical critical group of military personnel staying at the bays with dumped nuclear waste has been included. The exposure pathways of this group, however, cover inhalation and external exposure from beach occupancy only.

Source terms

The source terms used for the calculations were obtained from the IASAP project. The best-estimate release scenario was selected comprising release rates of radionuclides from shallow bays on the east coast of Novaya Zemlya (Abrosimov Bay and Tsivolki Bay) and from the deeper waters of the Novaya Zemlya Trough. The radi-

onuclide specific releases considered are shown in Table 1, and Fig. 2 shows the release rate assumed for the Tsivolki Bay where the reactor compartment and spent nuclear fuel from the Lenin icebreaker was dumped in 1967. The graph shows that the best-estimate scenario assumes release of the activation products only (Co-60, Ni-59, Ni-63) until the year 2300 due to corrosion of the protective barriers. At that time contact is assumed between the seawater and the spent nuclear fuel, and a peaked release of fission products (Cs-137 and Sr-90) and transuranics (Pu-239, Pu-240, Am-241) is assumed followed by a steady release due to corrosion of the nuclear fuel.

Table 1. Integrated release of radionuclides (TBq) dumped in the Kara Sea, IASAP best-estimate scenario.

	Tsivolki Bay	Abrosimov Bay	Novaya Zemlya Trough
Pu-239	4.9	0.73	0.09
Pu-240	2.1	0.32	0.04
Am-241	0.41	0.12	0.01
Co-60	0.03	2.9	0.08
Ni-63	370	110	1.9
Ni-59	40	110	2.3
Cs-137	0.10	41	8.7
Sr-90	0.07	34	7.4

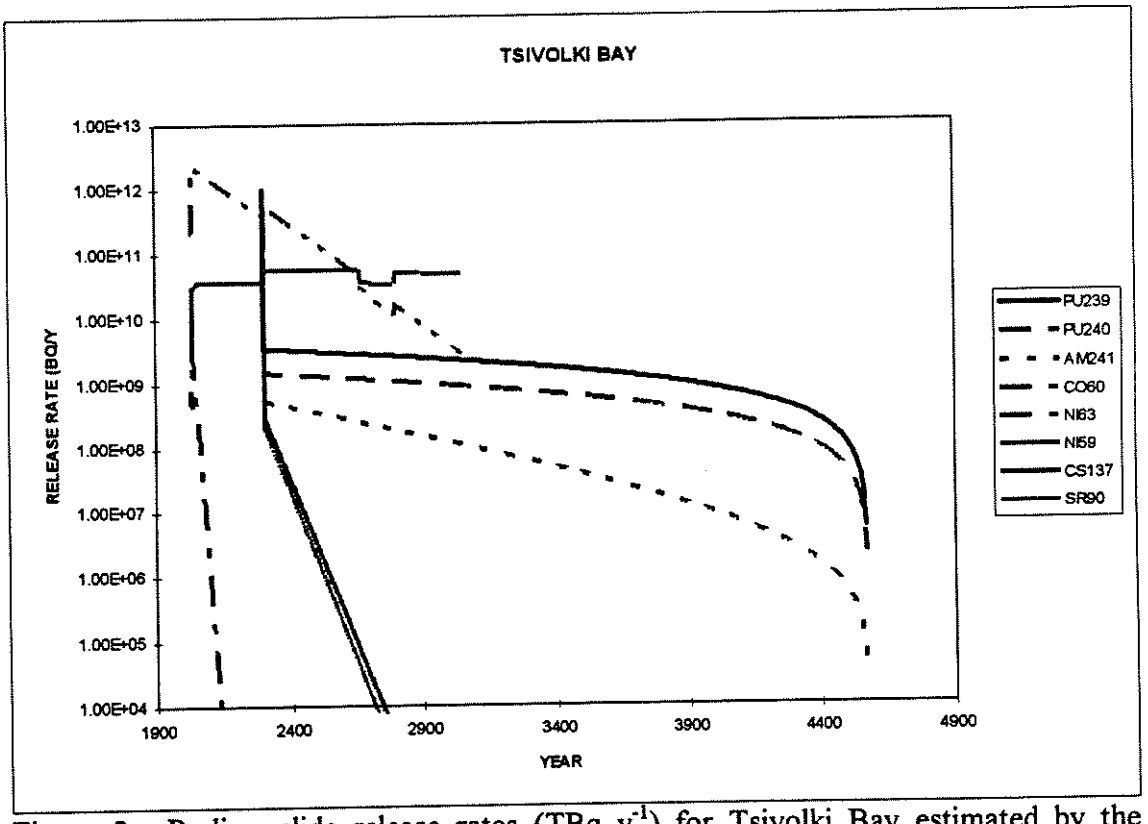

Figure 2. Radionuclide release rates (TBq y^{-1}) for Tsivolki Bay estimated by the IASAP Source Term Group.

Parameters investigated

The parameter sensitivity of the doses to individuals and populations has been investigated concerning three main processes: water movement and mixing, sediment-water interaction and biological transfer. For the hydrodynamical processes, four parameters were investigated: the advection and mixing between the Kara Sea and the Laptev Sea to the east, the advection and mixing between the Kara Sea and the Barents Sea to the

west, the vertical mixing between surface and deeper waters over the Novaya Zemlya Trough, and the rates of exchange of water between the bays containing the dumped waste and the open Kara Sea. The sediment-related parameters were the following: sedimentation rates, sediment distribution coefficients (K_d), suspended sediment loads, depth of the mixed surface sediment layer, and the mixing rates in the surface sediments. For the biological transfer processes, the three parameters representing biological concentration factors for fish, crustaceans and molluscs were included.

The parameter sensitivity analysis was carried out by assigning identical variabilities of 10% to all the above mentioned parameters and running the model repeatedly (about 500 times). Correlation coefficients were calculated between parameter values and dose values, and the square of the correlation coefficients (R^2) were interpreted as how much of the variation of the doses is explained by the linear relationship to the parameters. The parameter sensitivities are thus expressed in percent of the total variability.

Results

Collective doses

The results of the parameter sensitivity analysis for the collective doses are shown in Table 2 which summarises the integrated releases, the collective doses and the main parameter sensitivities (> 1%). The total collective dose is calculated to 0.4 manSv (truncated at 1000 y) with dominating contributions from plutonium isotopes of 57% and from Cs-137 of 37%. The parameter sensitivities are seen to vary across the radionuclides from Sr-90 which has a low K_d to the transuranics and activation products which have high K_d's. For the total collective dose the main parameter sensitivities are due to sedimentation processes (sedimentation rate, K_d, suspended sediment load) and biological transfer processes (concentration factor for fish). Furthermore, it is noted that the collective dose is very small.

Table 2. Integrated releases (TBq), collective doses (manSv) and main parameter sensitivities (%).

Nuclide	Cs-137	Sr-90	Pu-239	Pu-240	Co-60	Am-241	Ni-63	Ni-59	Total
Total release (TBq)	49	42	6	2.4	3	0.5	480	150	
Coll. dose (manSv)	0.14	0.02	0.16	0.06	7E-05	0.001	0.002	0.001	0.4
Coll. dose (%)	37	5	41	16	0.02	0.4	0.6	0.3	100
Sedimentation rate, (%)	19		66	66	54	61	53	53	45
Conc. factor, fish (%)	65	90			13		4	4	28
K_d (%)	11		13	13	25	7	24	24	12
Susp. sediment load, (%)			16	16	4	14	1	1	9
Conc. factor, molluscs (%)			9	9		5	1	1	5
Adv., Kara Sea, west (%)	4	1					4	4	2

Individual doses

The results of the parameter sensitivity analysis for the maximum annual doses to individuals in the critical groups are shown in the Tables 3 to 6 which summarise the dose rates from each nuclide and the main parameter sensitivities (> 1%). For all the critical groups, the time of occurrence of the maximum dose rate is related to the peaked release of radionuclides from Tsivolki Bay at year 2300 when the protective barriers of the spent nuclear fuel from the Lenin icebreaker are penetrated due to corrosion. The late occurrence of that event means that the shorter-lived fission products Cs-137 and Sr-90 have decayed and that the transuranics dominate the release. These radionu-

clides associate readily with sediments for what reason the main parameter sensitivities for long range transport are related to sedimentation processes (e.g. sedimentation rate, K_d, suspended sediment load). The advection and mixing of water from the Kara Sea to the Barents Sea also has some influence on the maximum dose rates to the critical groups in northern Norway and on the Kola peninsula. The maximum dose rates to these groups are very low (of the order of about a nanosievert per year).

Table 3. Maximum dose rates (Sv y^{-1}) in year 2300 to a critical group in northern Norway and main parameter sensitivities (%).

Nuclide	Cs-137	Sr-90	Pu-239	Pu-240	Co-60	Am-241	Ni-63	Ni-59	Total
Dose rate (Sv/y)	7E-13	7E-14	2E-10	8E-11	0E+00	2E-11	5E-14	5E-15	3E-10
Dose rate (%)	0.2	0.0	64	28	0.0	7	0.0	0.0	100
Sedimentation rate (%)									43
Adv., Kara Sea, west (%)									15
Kd (%)									14
Conc. factor,molluscs (%)									10
Bay flushing rate (%)									7
Susp. sediment load (%)									5

Table 4. Maximum dose rates (Sv y^{-1}) in year 2300 to a critical group on the Kola peninsula at the Barents Sea and main parameter sensitivities (%).

Nuclide	Cs-137	Sr-90	Pu-239	Pu-240	Co-60	Am-241	Ni-63	Ni-59	Total
Dose rate (Sv/y)	7E-12	6E-13	2E-09	7E-10	0E+00	1E-10	4E-13	5E-14	3E-09
Dose rate (%)	0.3	0.0	66	29	0.0	4	0.0	0.0	100
Sedimentation rate (%)									34
Kd (%)									21
Adv., Kara Sea, west (%)									20
Conc. factor, mollusc (%)									11
Bay flushing rate									9

For the critical group on the Yamal peninsula at the Kara Sea, the concentration factor for molluscs and the bay flushing rate have higher sensitivities than those of the sediment-related parameters. This is because the Kara Sea is the first recipient of the radioactivity from the Tsivolki Bay. The maximum dose rate to the critical group is very low, of the order of about a microsievert per year.

Table 5. Maximum dose rates (Sv y^{-1}) in year 2300 to a critical group on the Yamal peninsula at the Kara Sea and main parameter sensitivities (%).

Nuclide	Cs-137	Sr-90	Pu-239	Pu-240	Co-60	Am-241	Ni-63	Ni-59	Total
Dose rate (Sv/y)	2E-09	1E-10	7E-07	3E-07	0E+00	1E-07	2E-10	2E-11	1E-06
Dose rate (%)	0.1	0.0	63	28	0.0	9	0.0	0.0	100
Conc. factor, mollus (%)									32
Bay flushing rate (%)									31
Sedimentation rate (%)									20
Kd (%)									15
Conc. factor, crustac (%)									3

For the critical group located at the Tsivolki Bay only external exposure and inhalation pathways are considered. Inhalation dominates due to the transuranic nuclides. The most sensitive parameter is the sediment mixing depth which shows a strong negative correlation with the dose rates and a high sensitivity (56%). This is due to the association between sediments and transuranic elements, for what reason the transuranic

elements are removed by sedimentation from the water column to the sediment where the concentration (activity/volume) depends on the value of the sediment mixing depth. A small sediment mixing depth gives a small sediment volume and a high sediment concentration from which resuspension causes high doses due to inhalation. The maximum exposure of the critical group is estimated to an annual dose of the order of about ten millisieverts per year.

Table 6. Maximum dose rates (Sv y^{-1}) in year 2300 to a hypothetical critical group located at the Tsivolki Bay and main parameter sensitivities (%).

Nuclide	Cs-137	Sr-90	Pu-239	Pu-240	Co-60	Am-241	Ni-63	Ni-59	Total
Dose rate (Sv/y)	1E-07	4E-07	5E-03	2E-03	0E+00	6E-04	7E-09	6E-10	7E-03
Dose rate (%)	0.0	0.0	64	28	0.0	8	0.0	0.0	100
Sedim. mixing depth (%)									56
Kd (%)									18
Bay flushing rate (%)									10
Sedimentation rate (%)									9
Susp. sediment load (%)									3

Conclusions

A parameter sensitivity analysis has been carried out for a radiological assessment model used for prediction of doses to man from dumping of radioactive waste in the Kara Sea. The model covers the Arctic and Atlantic Oceans and calculates doses to the world population as well as to critical groups from a range of exposure pathways. Release rates of radionuclides from waste dumped in the Kara Sea have been used as source terms. These data were made available from the International Arctic Seas Assessment Project of the International Atomic Energy Agency. The sensitivity analysis has focused on three areas: hydrodynamical processes, water-sediment interaction processes, and biological transfer processes. The doses to man are generally dominated by contributions from long-lived transuranic radionuclides (plutonium and americium isotopes) which associate readily with sediments. For that reason the sediment-related processes and parameters generally show high sensitivities, particularly where doses are delivered some distance from the dump sites, e.g. in the Barents Sea. At closer distances, in the Kara Sea and in the Novaya Zemlya bays, biological transfer and hydrodynamical processes show high sensitivities.

This study illustrates that more detailed information on sediment-related processes, biological transfer processes and hydrodynamical processes in the Arctic waters will help reduce the predictive uncertainty of the dose assessment.

Acknowledgements

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Caesium-137 in the Icelandic Marine Environment 1989-1995

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Abstract - Iceland is located at the boundary where warm and cold currents meet. The island is nearly bathed with warm and high saline Atlantic water that is carried by a branch of the Gulf stream. North east of the island a boundary is formed with the Arctic water and even stronger boundary is formed north west of Iceland where the Atlantic water meets the ice cold Polar water coming from the north. Temperature and salinity data can help tracing the origin of radioactive isotopes measured in seawater samples. Radiocaesium in the Icelandic marine environment has been monitored at Geislavarnir ríkisins since autumn 1989. The ^{137}Cs concentration in the Icelandic marine environment is very low and ^{134}Cs has not been detected. The values for ^{137}Cs in seawater and seaweed are higher north and east of Iceland than south and west. This is due to a contributor from the reprocessing plants as well as old fallout from the bomb testing which is the main contributor to the values of ^{137}Cs in the Icelandic marine environment. Difference in ^{137}Cs levels in seawater of certain areas at different times is associated with fluctuation in temperature and salinity values due to displacement and mixing of water masses. It is therefore perhaps more realistic to correlate the ^{137}Cs activity values in Icelandic waters with different water masses instead of the usual geographical representation.

INTRODUCTION

Iceland is located on the Greenland-Scotland Ridge where boundaries are formed between the relatively warm and saline Atlantic waters and the cold Arctic water (figure 1). A branch of the Gulf Stream carries Atlantic water to the south coast of Iceland. This water flows clockwise along the south and west coasts. At the Iceland-Greenland Ridge the current splits into two parts: one swinging to the west and south-west and the other following the north shore to the east. This influx of Atlantic water to the north region appears as a tongue of relatively warm and saline water where the Atlantic component decreases due to mixing in the direction of the flow [5].

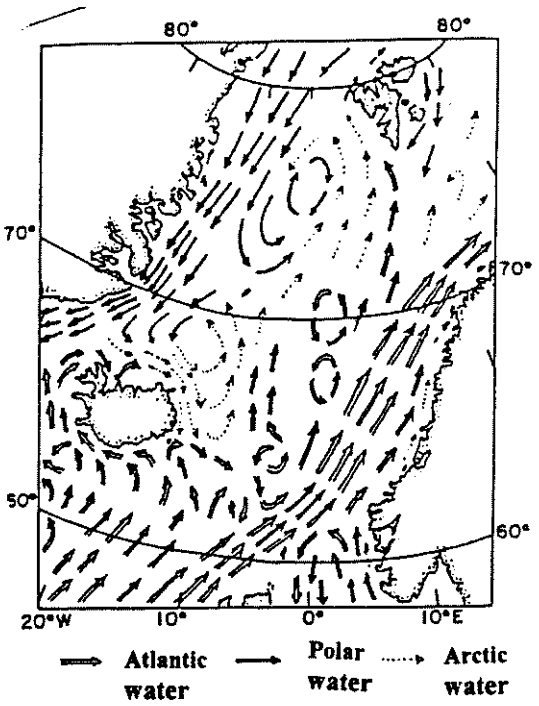

Figure 1 Ocean Currents around Iceland (Stefánsson 1990)

The East-Icelandic Current flows south-east along the Icelandic continental slope. Its origin is partly from mixed Atlantic water that flows

east along the north Icelandic shelf area. Another contribution to this current is Arctic water from the eastern part of the East Greenland Current.

Because of the conditions north and north-west of Iceland, where the relatively warm Atlantic water meets the cold and low salinity water of the East Greenland Current, the conditions are highly sensitive to meteorological changes. Variable influx of Atlantic water and variable admixture of polar water in the surface layers may lead to large changes in salinity and temperature of this area in space and time [5].

Temperature and salinity data can help to trace the origin of radionuclides measured in seawater samples. Furthermore, mixing of water masses can cause the activity to vary. Radiocaesium in the Icelandic marine environment has been monitored at Geislavarnir ríkisins since autumn 1989. This paper presents the main results of this study.

SAMPLING

Samples of surface water in the Icelandic zone have been collected and measured at Geislavarnir ríkisins since 1990, and deep water samples since September 1994. Since 1995 samples of marine sediment have been analysed. Seaweed samples have been collected at six locations since 1989. These were collected every third month from 1989-1992, but since 1993 samples are collected only in September. Samples of dab (*Limanda limanda*), common mussel (*Mytilus edulis*), capelin (*Mallotus villosus*), cod (*Gadus morrhus*), herring (*Clupea harengus*) and plaice (*Pleruonectes platessa*) have been taken in the economic zone since 1989. Samples from eiderduck (*Somateria mollissima*), polar bear (*Thalarctoc maritimus*) and harbour seal (*Phoca vitulina*) have also been collected and analysed. The locations of the sampling stations are shown in figure 2.

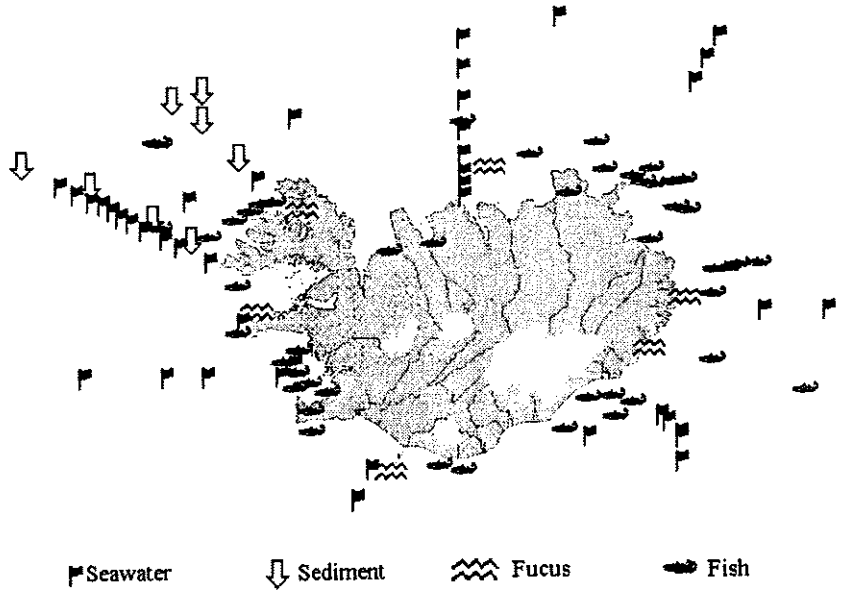

Figure 2 Sampling locations 1990-1995

RESULTS

The ^{137}Cs concentration in the Icelandic marine environment is very low and ^{134}Cs has not been detected in the samples taken in the economic zone. Typical levels of ^{137}Cs for samples monitored at Geislavarnir are [2]:

	south/west area	north/east area
Seawater:	2,5 Bq/m^3	2,5-10 Bq/m^3
Seaweed:	0,3 Bq/kg d.w.	0,5 Bq/kg d.w.

Sediment:	0,5-9 Bq/kg d.w.,
Fish:	0,1-0,4 Bq/kg f.w.,
Eiderduck:	<0,3 Bq/kg d.w.,
Polar bear and seal:	0,2 Bq/kg f.w..

There is a difference in the ^{137}Cs activity from sea water samples taken south and north of Iceland. The values are higher in samples from north of Iceland. The levels south of Iceland represent the radiocaesium fallout from nuclear testing in the early sixties. The levels north of Iceland are higher, mostly due to releases from the European reprocessing plants but relatively small contribution from the Chernobyl accident is also detectable[1]. Further more, there is great variation in activity levels at the same sampling stations north east of Iceland. But even greater variation in activity is noticeable at the region north west of Iceland [4].

Sediment samples are mainly collected as 30 cm cores. Each core is sliced into 2,5 cm sections and analysed. Some of the cores show thorough vertical mixing.

Analyses of variance for values of ^{137}Cs in seaweed samples, shows a statistically significant difference between values from the north and east coast on one hand and the south and west coast of Iceland on the other. The values are higher in samples taken at the north and east coast. Measurements of ^{99}Tc in seaweed samples from Vestmannaeyjar (an island south of Iceland) and Grimsey (an Island north of Iceland), done at the Risø Research Institute in Denmark, show also a significant difference[3].

Activity values in fish samples are very low. Statistically significant difference between sampling areas is not detected.

DISCUSSION

Seawater can be classified into different water masses, based on a certain temperature and salinity values. There are mainly five different types of water masses around Iceland. The warm and high saline Atlantic water, cold Polar water, Arctic surface water, Arctic bottom water and North Icelandic winter water. Furthermore mixing with fresh water from land causes low saline coastal water close to the shore.

Mixing between two or more water masses can also be detected as well as attenuation due to mixing with coastal water or ice melting. Higher salinity may occur by ice

formation and fluctuation in temperature due to summer heating or winter cooling are also common. This can be deduced from salinity and temperature data.

Great variability in activity from seawater samples is noticeable at the region north west of Iceland. This is because of the location near the boundary where the warm Atlantic water meets the Polar water and can be seen by measuring salinity and temperature. The fluctuation of the boundary is due to meteorological changes.

Variable influx of Atlantic water and/or variable admixture of polar water in the surface layers north of Iceland, may lead to large change in ^{137}Cs levels as well as the salinity and temperature.

Variance in ^{137}Cs activity values in certain areas of different times is accompanied with fluctuation in temperature and salinity values due to displacement and mixing of water masses. It is therefore perhaps more realistic to correlate the Cs-137 activity values of the Icelandic marine environment with different water masses instead of the usual geographical representation.

Further work with regard to this aspects to be carried out at Geislavarnir.

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Tracer Studies on Sediment-Water Interaction Kinetics of Sr-90 and Cs-137

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Sediments can act as both sinks and sources of radionuclides in the marine environment. The mobility of any radionuclide in a water-sediment system will be limited by both the strength of sorption to the sediment, the reversibility of the sorption mechanism and the kinetics of the sorption processes. Hence, the main aim of the laboratory studies reported here was to provide some information on the kinetics of radionuclide water-sediment interactions and influencing parameters.

Tracer studies (^{134}Cs and ^{85}Sr) have been used to investigate the kinetics of sorption and mobilization of radionuclides from marine sediments. Distribution coefficients (K_D ml g^{-1}) for ^{134}Cs in seawater-sediment systems varied with contact time, mixing conditions, sediment type, and salinity, and appeared to be correlated with cation exchange capacity. After 1 year contact time under static conditions, K_d for ^{134}Cs ranged from 740 to 1680 ml g^{-1}. Extraction studies showed that the majority of ^{134}Cs was strongly bound to sediment components, although a small fraction (1-10%) was easily displaced and remained in dynamic equilibrium with soluble species. Kinetic modelling using a simple box model indicated that the uptake of ionic Cs to sediments could be described by a two-component function. The relatively fast component probably reflects diffusion through the sediment-water microlayer followed by surface sorption to sediment (ion exchange). The slow component is thought to represent strong fixation within the mineral lattice.

Ionic ^{85}Sr was rather conservative in seawater, showing little transfer to sediments: K_d was 11 ± 2 ml g^{-1}. Furthermore, ^{85}Sr bound to sediments by ion exchange was easily displaced. Sequential extraction of Irish Sea and Stepovogo sediments indicated, however, that ^{90}Sr was rather less mobile than the ionic tracer. This might reflect the presence of different ^{90}Sr species in the discharge (i.e. non-ionic) or a different water-sediment interaction mechanism in the natural environment.

For both radionuclides, K_D in freshwater was a factor of 100 higher than in seawater, indicating that exchangeable ^{137}Cs and ^{90}Sr can be mobilized if freshwater sediments are transferred to the marine environment. However, the increasing ionic strength in estuaries can also enhance flocculation and precipitation of colloids.

Nuclear power and the environment in Finland - with special emphasis on the marine environment

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Abstract - Environmental impacts of two Finnish nuclear power plants are assessed on the basis of some 20 years of experience. The extent of their radiological impact has been insignificant. Local discharges of radionuclides are found in the terrestrial environment only occasionally, and in very small concentrations. Samples from the marine environment frequently contain traces of local discharges, and even at greater distances from power plants. The activity concentrations have been low, however. The most notable thermal effects of cooling water have been changes in winter ice-conditions which have made winterfishing more difficult in areas close to power plants. The most obvious biological effect has been significant increase of littoral vegetation in the vicinity of cooling water outlets.

Introduction

Finland already has some 20 years of experience in the use of nuclear power. Two 445 MWe PWR units have been operating in Loviisa (on the south coast) since 1977 and 1980, and two 710 MWe BWR units in Olkiluoto (on the west coast) since 1978 and 1980. This is therefore a good time to assess the environmental impact of the power plants on the basis of the knowledge that has been obtained up to now.

When construction of the existing units was started, there were plans to build several more units in Finland, but public opposition to nuclear power arose soon thereafter, delaying implementation of the plans. A decision to start building a fifth reactor was, however, in preparation just before the accident at the Chernobyl NPP in 1986. The accident had an influence on politicians, and the decision made in the Parliament in 1993 was against the new reactor unit.

The principal issues of contention in the debate over nuclear power have been: 1) fear of a nuclear power plant accident and its eventual consequences for man and the environment, 2) questions of nuclear waste, 3) environmental effects of the normal operation of nuclear power plants including the environmental impact of radioactive discharges and the thermal effects of heated cooling water on the bodies of water receiving the discharges. This paper reviews both aspects of the last-mentioned issue.

Material and methods

The authorities were well prepared from the very beginning to conduct surveys of the environmental impact of nuclear power plants in Finland. Base line studies were initiated at boths sites well before the start of construction work, at Loviisa in 1967, and at Olkiluoto in 1972. Permanent programmes for monitoring environmental radiation were launched in both areas one year before the power station became operational. The volume of the monitoring programmes has remained at the same level throughout the operational history of the power plants, though some details have been revised every five years.

The Finnish Centre for Radiation and Nuclear Safety (STUK) has carried out ecological investigations in the sea areas off Loviisa and Olkiluoto together with the radiation monitoring programmes. The aim has been to obtain the neccessary ecological background for the radiation monitoring programmes and radioecological studies carried out in the sea areas. In addition, the aim has been to document all the environmental impacts of the power plants (including thermal effects) by means of these investigations.

The environmental radiation monitoring programmes include altogether some 1000 annual sample measurements from Loviisa and Olkiluoto. The number of marine samples is 140. The monitoring objects in the terrestrial environment are external radiation (dose rate), air, deposition, soil, hair moss, forest berries, mushrooms, grazing grass, milk, lettuce, garden berries, fruits, grain, meat and drinking water of the power plants and the cities closest by.

The monitoring objects in the aquatic environment are sea water, sinking matter, bottom sediments, seaweed, benthic animals, and wild fish and fry from fish farms utilizing warm cooling water from the power plants. The sampling programme functions year-round, but it is weighted in late spring, summer and early autumn. The most important areas in the spreading of discharges in terrestrial and marine environments have been taken into account in placing the sampling stations. Most of the sampling stations are within a 10 km radius of the power plants.

The marine ecological investigations have focused on conventional hydrographic parameters such as water temperature, transparency, salinity, pH and oxygen concentration, total phosphorus and total nitrogen in water, and on a number of biological monitoring objects such as phytoplankton and its primary production, primary production capacity, littoral vegetation and zoobenthos.

Standard STUK methods have been used in sampling, pre-treatment and sample analysis. The methods have already been described in several STUK publications and Annual Reports.[1-5]

The sea areas off Loviisa and Olkiluoto

The Loviisa NPP is situated in the inner archipelago zone of the Gulf of Finland. The discharge area of the power plant, the bay of Hästholmsfjärden, is a semi-enclosed basin between the mainland and the outer archipelago where many islands, narrow and shallow sounds and underwater sills limit the exchange of water. This limited water exchange tends to retard the mixing and dilution of radioactive and thermal discharges in large masses of water, but on the other hand, it also limits the spreading of discharges to more distant areas. In addition, it tends to disrupt oxygen conditions in the deep water areas of the basins.

The low salinity of water is another significant characteristic of the sea area surrounding the Loviisa NPP. The average salinity of surface water during the growing period is 4-5‰ and in winter and early spring the salinity can be much lower. These circumstances are extreme ones for many marine organisms, but also for many freshwater organisms. Consequently, the biota in the area is very poor and sensitive to all environmental changes.

The sea area off Olkiluoto is more of a marine environment. The salinity of surface water during the growing period is 6 ‰ on average, and the biota is more of a marine nature, as well as being more abundant and more resistant to environmental changes. Despite the many skerries off Olkiluoto, the area is more open than that of Loviisa and the exchange of water is more effective there.

Discharges from power plants

H-3, C-14 and I-131 are the dominant radionuclides in the airborne discharges from both power plants. Ag-110m and Sb-124 have been the next most abundant nuclides in the airborne discharges from Loviisa and Cr-51, Co-60 and Mn-54 in those from Olkiluoto. In the period 1984-1995, the total discharge of tritium into the air was 14 TBq from Loviisa and 2.4 TBq from Olkiluoto.

Tritium is also the most abundant radionuclide in liquid discharges. In the period 1984-1995, the total discharge of tritium into the sea was 140 TBq from Loviisa and 21 TBq from Olkiluoto. The next most abundant radionuclides in the liquid discharges from Olkiluoto were Co-60 (49 GBq in 1984-1995), Cr-51, Mn-54 and Co-58. The next most abundant radionuclides after tritium in the liquid discharges from Loviisa

were Cs-137 (49 GBq in 1984-1995), Sb-124, Cs-134, Co-60 (19 GBq in 1984-1995) and Ag-110m. (All the numbers given here are non-decay-corrected total sums). It should be noted, however, that discharges of the last-mentioned nuclides, especially from Loviisa, have decreased substantially over the last few years due to a new waste management system.

When operating at full capacity, the thermal discharge from the Olkiluoto NPP amounts to about 60 m^3s^{-1} of cooling water, the temperature of which is 11.5°C higher than that of the intake water. The cooling water discharge at Loviisa is 50 m^3s^{-1} with operation at full capacity, and the temperature increase is 10°C. The radioactive discharges are mixed in with the outflowing cooling water, being thus diluted by large masses of water.

Results

The most abundant discharged nuclides are most frequently found in environmental samples. Observations of locally discharged nuclides are relatively rare in the terrestrial environment. On the contrary, in a marine environment locally discharged nuclides are regularly found in some sampling objects in areas close to cooling water outlets. Fallout from the Chernobyl accident made it difficult for a long time to distinguish locally discharged Cs-137 and Cs-134 in environmental samples.

Radioactive substances in the terrestrial environment

Locally discharged nuclides have been observed a few times each year in both areas in the continuously collected samples of ground level air and deposition. In air samples, the highest activity concentrations have been 100 μBq m^{-3}. In deposition, the H-3 concentrations have been less than 10 kBq m^{-3} and those of other nuclides less than 5 Bq m^{-2} (the different units depend on different sampling methods).[5]

In all other terrestrial samples, locally discharged nuclides are seldom observed and then in low concentrations. There have been occasional findings (less than once a year) of discharged nuclides in soil and hair moss in both areas; at Olkiluoto there have been some traces in lettuce as well. The activity concentrations have been less than 12 Bq kg^{-1} dry wt. Trace amounts of discharged nuclides have sometimes been found in the drinking water of both power plants, but in some cases at least, the obvious explanation is contaminated sampling.

Discharged nuclides of local origin have never been found in grazing grass, milk, meat, grain, mushrooms, forest berries, garden berries and fruits collected from the Loviisa and Olkiluoto areas.

Radioactive substances in the marine environment

In the case of the marine environment, locally discharged nuclides are found regularly (every sampling occasion) in seaweed, sinking matter, bottom sediments and benthic animals. The findings are sporadic in sea water and fish.

Some elevated concentrations of tritium in sea water caused by local discharges are found every year in both areas. At Loviisa, the highest concentration has been 35 kBq m^{-3}; at Olkiluoto, 25 kBq m^{-3}. Other discharged nuclides are observed less frequently in sea water and their concentrations have been 35 Bq m^{-3} at their highest.

Traces of locally discharged nuclides were detected only in a few wild fish samples caught in Loviisa during the monitoring period. At Olkiluoto, these findings have become somewhat more frequent in recent years. In all cases, the activity concentrations have been less than 1 Bq kg^{-1} fresh wt. Locally discharged nuclides have never been observed in fry from the Loviisa fish farm, but have been observed twice at Olkiluoto (less than 0.15 Bq kg^{-1} fresh wt.).

In the case of sinking matter and seaweed (macro algae), the acticity concentrations of discharged nuclides have been about 400 Bq kg^{-1} dry wt at their highest. Co-60 has been the most abundant discharged nuclide, especially in *Fucus vesiculosus*, which has proven to be an excellent indicator of cobalt. Besides Co-60, there was earlier abundant Ag-110m in sinking matter at Loviisa.

In the case of benthic animals (mussels and crustaceans), the highest activity concentrations of discharged nuclides have been less than 100 Bq kg^{-1} dry wt. The relict crustacean *Saduria entomon* has proven to be a good indicator of Ag-110m.[6]

In the case of bottom sediment samples, the activity concentrations of discharged nuclides have been 70 Bq kg^{-1} dry wt at their highest. The favourable sedimentation conditions in some sedimentation basins of both areas have been creating archives in which the history of discharges can be read.

The concentrations of discharged nuclides are generally highest in the vicinity of the cooling water outlets and decrease rapidly with increasing distance. In the surveys made with *Fucus vesiculosus* in the vicinities of the Loviisa and Olkiluoto NPPs and along the entire Finnish coast in 1980, 1987, 1991 and 1995, the farthest observations of Co-60 were made at a distance of 10 km from Loviisa, but as far as 140 km from Olkiluoto (0.19 Bq kg^{-1} dry wt at Närpiö, NE Bothnian Sea). This difference is based on more abundant Co-60 discharges from Olkiluoto and the openness of the Olkiluoto sea area, which promotes the spreading of discharges.

In a study carried out in Hästholmsfjärden at Loviisa, concentrations of radionuclides in different levels of the food web were widely investigated.[7] The highest concentrations of the Chernobyl derived Cs-137 (150-170 Bq kg^{-1} fresh wt) occurred in the flesh of some predatory fish (e.g. perch, burbot and pike) as expected. The concentrations were clearly lower in local water birds and in their inner organs and eggs as well as in seal samples (and their organs) obtained from the outer Gulf of Finland. A key finding was that locally discharged nuclides (Ag-110m, Co-60, Mn-54) were only found in the lower trophic levels of the ecosystem (phytoplankton, zooplankton, macro algae and benthic animals).

Impacts of cooling water discharges

Discharges of cooling water physically affect temperatures and currents in the receiving water bodies. In open water circumstances (spring, summer, autumn) elevated temperatures are found only near the cooling water outlets, because mixing caused by winds and evaporation into the atmosphere are effective at lowering temperatures. In Hästholmsfjärden at Loviisa, thermal discharges have increased the mean surface water temperature of the growing period by 2-4°C at a distance of 0.5 km from the outlet. Outside the Hästholmsfjärden bay, slight temperature increase can only be observed occasionally during the open water season.

In winter, thermal discharges may significantly affect the ice conditions in the neighbourhood of the power plants. During winters of normal ice-formation areas close to cooling water outlets are open throughout the winter and the size of the open area may change day by day (ranging between several hectares and several square kilometres) according to the weather conditions. If the surrounding area is ice-covered, heated water may drift some 20 kilometres under the ice, then come to the surface and melt the ice. Difficult ice conditions can thus hinder winter-fishing in quite extensive areas.

Strong currents outside the cooling water outlets produce erosion and less vegetation in the vicinity of the outlets, and accumulation of drifted material on the bottom in deeper basins farther from the outlets. At Olkiluoto this has caused changes in zoobenthos at some sampling stations.

General eutrophication has been a conspicuous event in the sea area surrounding the Loviisa NPP during the past 30 years. Primary production of phytoplankton has doubled or trebled, the share of blue green algae has increased especially in the discharge area in late summer, depletion of oxygen has become a repeated autumnal event in the deep basins of the area, bottom sediments have changed to more organic and unaerobic types, and the sparse zoobenthos has decreased further and even disappeared from some sampling sites. However, since the late 1960s, total phosphorus and total nitrogen concentrations of water have also doubled throughout the sea area, and throughout the eastern Gulf of Finland. This is certainly the main reason for the eutrophication trend and not the thermal discharges.

The clearest change in biota specifically caused by warm water has been the significant increase of littoral vegetation in the vicinity of the cooling water outlets at both Loviisa and Olkiluoto.[8,9] The abundance of the vascular plants *Myriophyllum spicatum*, *Potamogeton perfoliatus* and *Potamogeton pectinatus* has increased strongly and in some places they form together with their epiphytes, filamentous algae (*Cladophora glomerata*, *Ectocarpus siliculosus* and *Pilayella littoralis*), almost inpenetrable populations.

Discussion

Average radiation doses from the local discharges received by critical groups of people living in the vicinity of the Finnish nuclear power plants have been estimated to be between 0.1 and 3.7 $\mu Sv\ a^{-1}$ at Loviisa and between 0.1 and 1.5 $\mu Sv\ a^{-1}$ at Olkiluoto over the past years. At their highest, these values represent only several percent of the annual dose limits set for the critical groups (0.1 $mSv\ a^{-1}$). The average annual radiation dose received by Finns is estimated to be about 3.7 mSv, of which indoor radon makes up 2 mSv. Compared with these values, the significance of the additional dose caused by the radioactive discharges to the critical groups is marginal. This also obtains for general radiological effect of the normal operation of the nuclear power plants.

As concerns the thermal effect of heated cooling water on receiving bodies of water, changes in winter ice-conditions and temperatures have been the most notable physical deviations, and winter-fishing has consequently become more difficult in the areas close to the power plants. This has also been the most important consequence in an economic sense.

Significant changes have occurred in the ecological status of the sea area off Loviisa during the past 30 years. They have primarily been caused by the general eutrophication of the entire Gulf of Finland. Thermal discharges may have partly increased the eutrophication rate in a small area close to the cooling water outlet. The most important biological effect caused specifically by the thermal discharges in both study areas has been the strong increase of littoral vegetation in the vicinity of cooling water outlets. All other biological effects have been significantly weaker at Olkiluoto, where the nutrient concentrations in water are clearly lower than in Loviisa. As a whole, the ecological effects caused exclusively by thermal discharges have been relatively slight and have been restricted to relatively small areas close to the power plants.

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Lokalisering og inspeksjon av dumpet radioaktivt avfall i fjordene på østkysten av Novaja Semlja - Resultater av felles Norsk-Russisk samarbeid innen radioaktiv forurensning i de nordlige havområder.

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SAMMENDRAG

I 1992 ble det som følge av påstått dumping av radioaktivt materiale i Barents- og Karahavet nedsatt en norsk-russisk ekspertgruppe for undersøkelse av radioaktiv forurensning i de nordlige havområder.

Det er gjennomført tre, 1992 [1], 1993 [2,3] og 1994 [4,5], felles norsk-russiske ekspedisjoner med forskningsfartøyet «Viktor Buinitskiy» til dumpeområdene. I 1993 og 1994 var en av hovedmålsetningene å lokalisere og identifisere dumpet radioaktivt materiale i Tsivolki, Stepovogo og Abrosimov fjordene på østkysten av Novaja Semlja samt å samle inn prøvemateriale nær gjenstandene.

Ved hjelp av høyfrekvente sonarer og en mini-ubåt (ROV, Remote Operated Vehicle) (Fig.1) ble gjenstander lokalisert og inspisert. De fleste ettersøkte gjenstander ble funnet og inspisert med unntak av reaktor og brenselselementer fra isbryteren «Lenin» i Tsivolki fjorden.

INNLEDNING

De første offisielle informasjoner om de tidligere sovjetiske dumpingene av radioaktivt materiale i havet ble presentert i «Hviteboken» Nr.3 (1993) [6]. Rapporten gir opplysninger om dumping av både flytende og fast radioaktivt avfall i Barents- og Karahavet i perioden 1960-1990.

Fast lav og middels radioaktivt avfall dumpet i Karahavet består av forskjellig typer materiale fra den operasjonelle nordflåtens atomdrevne marinefartøy og sivile isbrytere. I hovedsak er lav og middels aktivt avfall dumpet i Karahavet forseglet i stålkontainere. Større objekter er blitt dumpet separat eller plassert innvendig i egnede skip og lektere.

I tre fjorder på østkysten av Novaya Zemlya og i et område i det åpne Karahavet skal det totalt være dumpet 6 ubåt-reaktorer med brukt brensel, deler av brukt brensel fra en av reaktorene på isbryteren «Lenin» og 5 objekter inneholdende 10 reaktorer uten brensel. De fleste av reaktorene med brukt brensel skal i.h.t. «Hviteboken» Nr.3 (1993) [6] være behandlet med et furfurol-basert materiale som en beskyttende barriere.

I.h.t. «Hviteboken» Nr.3 (1993) [6] er den maksimalt mulige aktivitet i de dumpede reaktorer med brukt brensel kalkulert til 85 PBq og den totale induserte aktivitet i reaktorene uten brensel mindre enn 3.7 PBq referert til tidspunkt for dumping.
Innenfor IASAP's rammeprogram (IAEA's International Artic Seas Assessement Project) er det av det russiske Kurchatov instituttet blitt foretatt nye beregninger som viser verdier (37 PBq) som er en faktor tre lavere enn de som presenteres i «Hviteboken» Nr.3 (1993) [6]. Her er det bl.a. tatt hensyn til at de fleste reakorene ble utsatt for en ulykke kort tid etter at de ble satt i drift.

LOKALISERING OG VISUELL INSPEKSJON AV DUMPEDE GJENSTANDER

Lokalisering.

Med utgangspunkt i «Hvitebokens» opplysninger om det dumpede materiale og de der oppgitte posisjoner ble gjenstandene forsøkt lokalisert med sonarsøk. Områdene ble scannet med bruk av en høyfrekvent (120/330 kHz) sidesøkende sonar av typen Simrad Mesotec 992 som ble slept etter «Viktor Buinitskiy». På grunn av skipets dårlige manuvreringsegenskaper viste denne metoden seg å være svært tidkrevende og vanskelig å gjennomføre. For bedre å kunne lokalisere de dumpede gjenstandene utrustet vi en 15 fots gummibåt (med påhengsmotor) med instrumenter og vekselstrømkilde slik at vi kunne søke detaljert i de aktuelle dumpeposisjoner. Med en hastighet på 2-3 knop ble en 360^0 søkende høyfrekvent (675 kHz) sonar (Simrad Mesotec 990) slept etter gummibåten på et dyp ca. 10 meter over havbunnen. Dette viste seg å være en effektiv, men tidkrevende, måten å lokalisere dumpede gjenstander på. De best definerte gjenstandene på sonarbildene ble merket ved at en forakret bøye ble droppet rett over de lokaliserte gjenstandene på havbunnen.
Mens dette lokaliseringsarbeidet pågikk ble det fra «Viktor Buinitskiy» gjennomført en systematisk innsamling av bunnsedimenter.

Visuell inspeksjon av dumpede objekter.

Etter at søkearbeidet med sonar var avsluttet kunne forskingsfartøyet bevege seg til de forskjellige markeringene og droppe anker for å foreta en mer omfattende inspeksjon av objektene ved bruk av en Remote Operated Vehicle (ROV) (Fig.1).

Den medbragte ROV var utstyrt med en høyfrekvent 360^0 sonar, video kamera, NaI-gammadetektor, BF_3-nøytrondetektor, sedimentprøvetaker og instrumenter for måling av korrosjonshastigheter i metaller (Corintech CP-probe).
Den påmonterte prøvetaker gjorde det mulig å ta prøver av havbunnen på eksakte posisjoner tett inntil objektene.

Det ble også benyttet en nedsenkbar NaI-detektor (80x400) for spektroskopisk screening av havbunnen nær objektene. En mini-CTD ble senket ned på alle stasjoner for å logge temperatur og salinitet i vannprofilene. Fig. 2 og 3 viser temperatur og saltholdighetsprofiler fra henholdsvis Tsivolki fjorden og Abrosimov fjorden.

Alle bilder fra sonarsøk og filming av undervannsoperasjoner ble lagret på videokassetter for senere studier.

I Tsivolky Fjord, Stepovogo Fjord og Abrosimov Fjord ble en rekke dumpede gjenstander lokalisert, både skip, lektere og beholdere. Graden av korrosjon og skader på objektene kan være vanskelig å fastslå p.g.a. tilgroing , men videobilder av enkelte beholdere viste stor grad av deformasjon.

Tsivolky Fjord

En rekke gjenstander av forskjellig størrelse ble lokalisert i Tsivolky Fjord. Et uventet høyt partikkelinnhold i vannet ga en svært dårlig sikt og gjorde søkearbeidet vanskelig. Allerede på 2 meters dyp var det så mørkt at det måtte benyttes lys på ROVen, men med lyskastere på ble sikten enda dårligere. Dette gjorde arbeidet svært tidkrevende og ga dårlig kvalitet på undervannsbildene.

Sonarscreening av områder i fjorden på posisjoner der reaktordeler fra isbryteren «Lenin» ifølge «Hviteboken» skal være dumpet viste gjenstander som av størrelse kan være deler fra denne. P.g.a. tidsnød var det ikke mulig å inspisere disse objekter nærmere med ROV. Det mest markante funnet i Tsivolky fjorden var et helt lasteskip.

Stepovogo Fjord

I den ytre delen av Stepovogo fjorden ble den dumpede ubåten med betegnelsen NS No. 601 inneholdende to reaktorer med brukt brensel lokalisert på den oppgitte posisjonen fra «Hviteboken». Det ble forretatt inspeksjon av denne med ROVen og det ble tatt relativt gode detaljbilder av skroget (Fig.4).

I den indre delen av fjorden ble det lokalisert et sort antall (flere hundre) dumpede firkantede beholdere, tydeligvis av forskjellig sørrelse og med varierende vekt. Enkelte av disse ser ut til å være i svært dårlig forfatning. Videobilder viser detaljer der enkelte er gjennomrustet og har store hull. Forhøyede konsentrasjoner av ^{137}Cs og ^{90}Sr kunne detekteres i sedimentprøver tatt lokalt ved beholderene.

To beholdere ble også funnet på stranden (Fig.5). Innholdet i disse (plastikk, laboratorietøy, gummihansker osv.) gir en indikasjon om innholdet i de åpenbart lette beholderene som ble observert på bunnen.

Abrosimov Fjord

I Abrosimov Fjord ble en rekke objekter lokalisert ved bruk av sonar. Fjorden er relativt grunn og samtlige objekter ligger på ca. 10-20 meters dyp. Flere av disse ble nærmere inspisert ved bruk av ROV. Tre lektere ble lokalisert og identifisert til å være av typen MBCH og MBHH. En av disse hadde to store sylindriske beholdere plassert på dekk.

Beholderene ble anslått til å ha en lengde på 8-10 meter med en diameter på 2-3 meter.

Tre objekter som antas å være dumpede reaktorer ble også lokalisert. Basert på gamma-målinger tett inntil objektene, kunne ^{137}Cs detekteres fra to av disse og ^{60}Co fra den tredje.

Det ble også lokalisert en rekke firkantede beholdere av samme type som ble funnet i Stepovogo fjorden. Flere av disse var tydelig angrepet av korrosjon.

KONKLUSJON

De benyttede metoder for lokalisering og inspeksjon av dumpede gjenstander i fjordene på østkysten av Novaya Zemlya viste seg utifra gitte forutsetninger å være svært effektive. Forskningsfartøyet hadde vanskeligheter med å manuvrere etter eksakte posisjoner i de trangeste partiene i fjordene. Sonarsøk fra spesialutstyrt gummibåt etter veldefinerte mønstere gjorde det svært effektivt å søke over større områder. Markering av lokaliserte objekter med forankrede bøyer gjode det enkelt å oppsøke disse for nærmere inspeksjon med ROV. Tatt i betraktning til dels svært dårlig sikt i vannmassene, ga videobildene allikevel et godt grunnlag for videre arbeid med å fastslå hvilke gjenstander som er lokalisert, hva de inneholder og hvilken forfatning de befinner seg i.

Forhøyede nivåer av kustige radionuklider i sedimenter innsamlet lokalt ved nærmest alle objektene indikerer at det er oppstått lekkasje til omgivelsene. De høyeste verdiene av ^{137}Cs, ^{60}Co, ^{90}Sr og $^{239,240}Pu$ ble observert i sedimentprøver tatt lokalt ved dumpede kontainere i Abrosimov Fjord og Stepovogo Fjord. Nivåene viste seg å være mindre ved de lokaliserte lektere og reaktorseksjonene.

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Figurtekst :

Fig.1. Bilde av benyttet mini-ubåt (ROV, Remote operated vehicle).

Fig.2. Temperatur og saltholdighetsprofiler fra Tsivolki fjord.

Fig.3. Temperatur og saltholdighetsprofiler fra Abrosimov fjord.

Fig.4. Detaljer fra skroget til dumpet ubåt i Stepovogo fjord.

Fig.5. Fotografi av en beholder funnet på stranden i Stepovogo fjord.

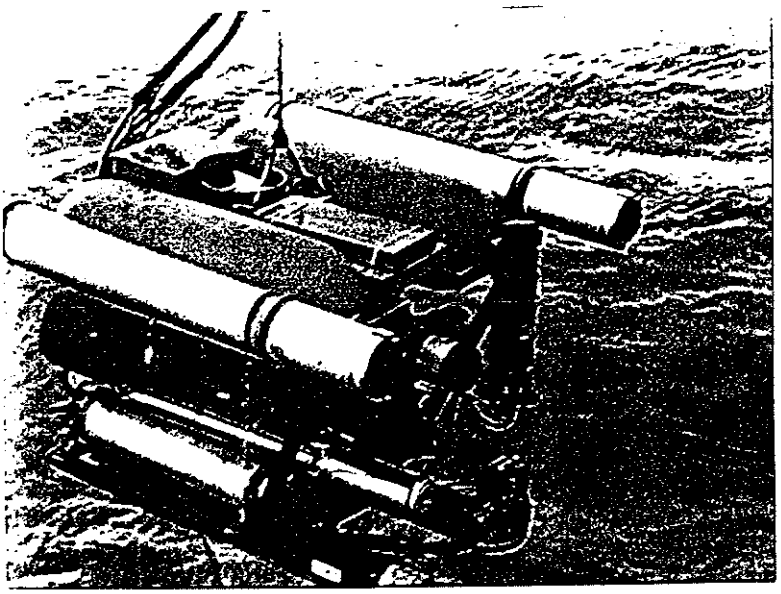


Fig. 1

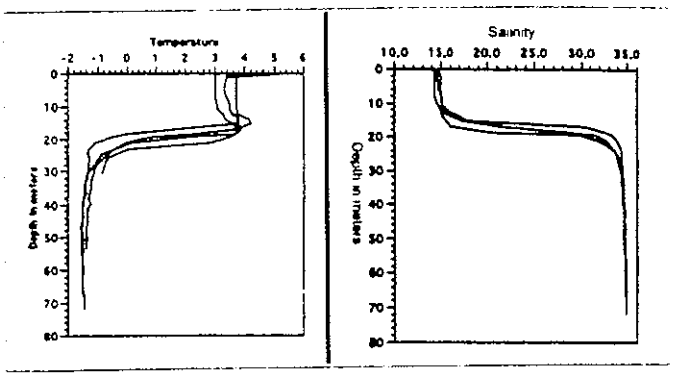
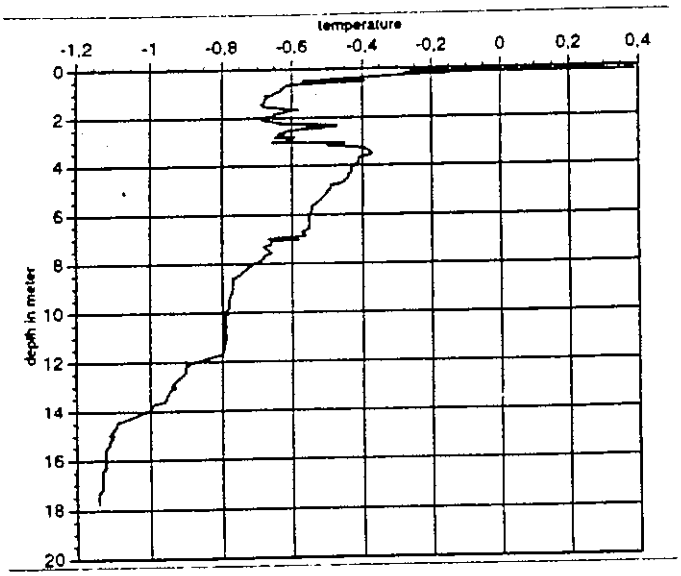


Fig. 2



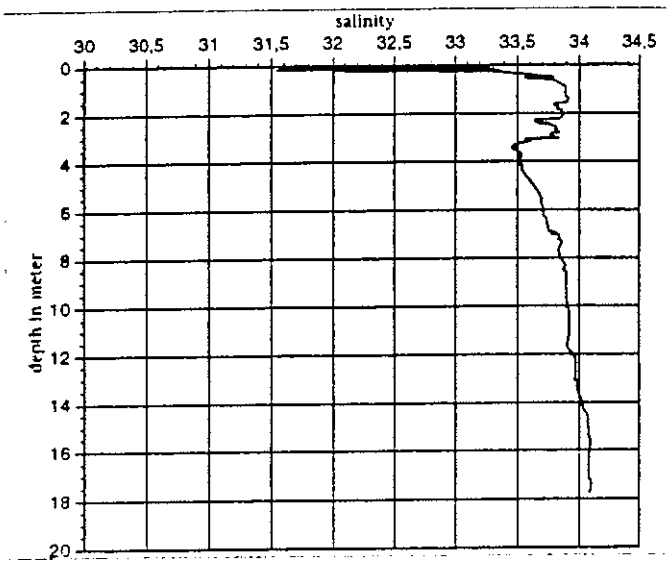


Fig. 3

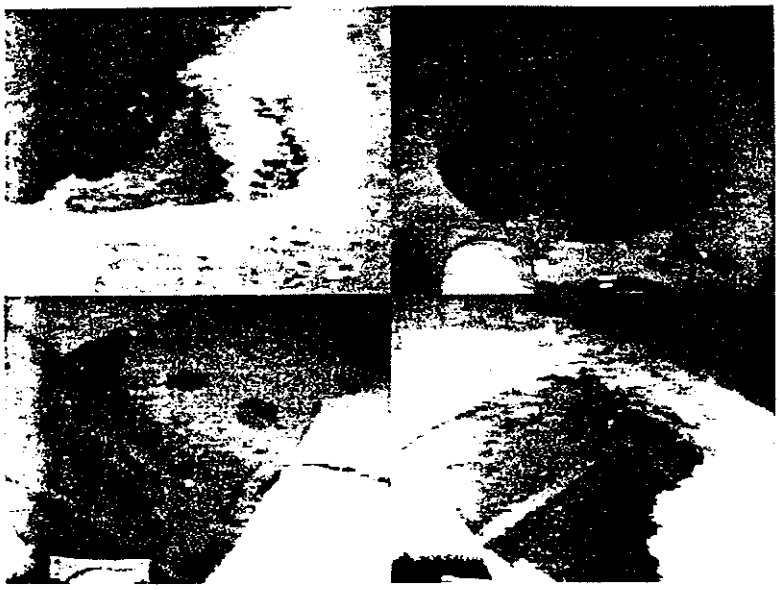


Fig. 4

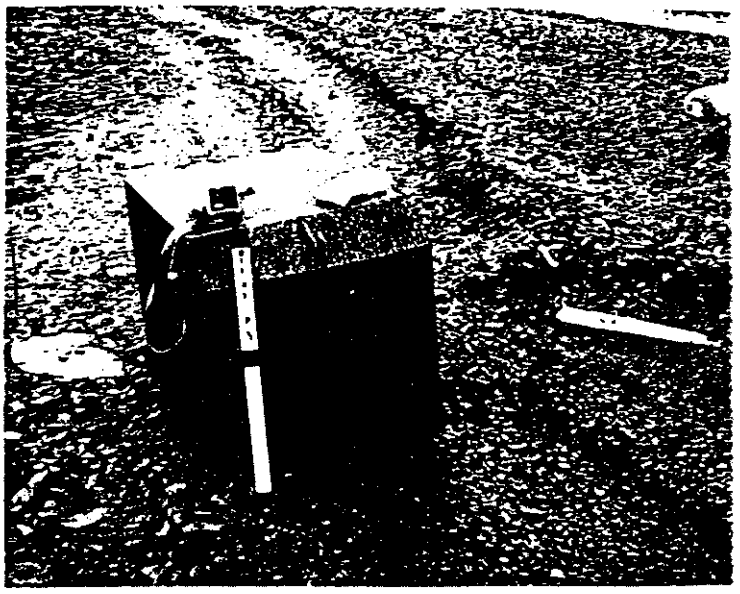


Fig. 5

A STUDY ON THE BEHAVIOUR OF CESIUM AND PLUTONIUM IN THE SEDIMENTATION PROCESS UNDER VARYING REDOX-CONDITIONS

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Abstract - The effect of varying redox-conditions on remobilization of cesium and plutonium from sediment to water has been investigated in the water deep of Hästholmsfjärden in the discharge area of the Loviisa NPP in 1995. The total depletion of oxygen occurred during two periods in the late summer and autumn. No significant increase in the Cs-137 and Pu-239,240 concentrations of the water samples was found. During the latter anoxic period the cesium concentration was, however, slightly higher than during oxic periods in early summer and later autumn. In sediments, the concentrations for different sampling times differed from each other, but no clear effect of anoxic conditions can be seen. The project continues by repeating the sampling and analysing procedures in 1996 with a more spesific sampling method. Theoretically, if about four percent of the total Cs-137 and two percent of Pu-239,240 adsorbed in surface sediment in this deep is released to water, it is possible to observe the subsequent concentration changes in near-bottom water.

Introduction

The sedimentary behaviour of radionuclides, especially cesium and plutonium, and the effect of anoxic conditions in near-bottom waters has been discussed largely in literature [1,2,3,4,5,6,7,8]. Varying redox-conditions may change occurence and concentrations of radionuclides in bottommost waters and surface sediments and affect remobilization of radionuclides from sediment to water.

Sediment inventories of the Baltic Sea, made by the HELCOM/MORS Expert Group after the Chernobyl accident, have shown that about 1.4 PBq of Cs-137 and 18 TBq of Pu-239,240 are bound in the Baltic Sea sediments. About 19% of the whole sea area, the deepest part of the Baltic Sea (70 000 km^2) has during the last years been continuously anoxic. It is important to know to what extent the depletion of oxygen affects the behaviour of radionuclides in near-bottom waters, i.e. sedimentation and remobilization from sediments. These processes have been investigated in an archipelago basin on the Finnish south coast.

The discharge area of the Loviisa NPP, Hästholmsfjärden, is a semi-closed water basin between the mainland and the archipelago, being in connection with the open Gulf of Finland only through narrow and shallow sounds (Fig.1). The mean depth of the deep of Hästholmsfjärden is about 15.5 meters and maximum depth 18.1 meters. The bottom area being beyond 15 metres depth is 175 000 m^2. The bottom sediment in the deep is very watery and organic mud.

Poor exchange of water together with thermal discharges causes disturbance in the oxygen balance of the basin. The low oxygen concentrations conjoin with strong remobilization of e.g. phosphorus and nitrogen from sediment to water. The phosphorus concentrations have been even 40 times higher in the hypolimnion than in the epilimnion of Hästholmsfjärden and high concentrations have been observed in surface water, too. In addition, slight indications of remobilization of certain radionuclides were found in 1983-1984, when preliminary experiments were carried out in the deep, though total depletion of oxygen did not occur [9].

Materials and methods

The oxygen, total phosphorus and total nitrogen concentrations as well as the temperature, salinity and pH of water were monitored in the deep of Hästholmsfjärden during June-October in 1995. The samples were taken with a Limnos water-sampler (volume 2 l) from 0.5 metres above the bottom.

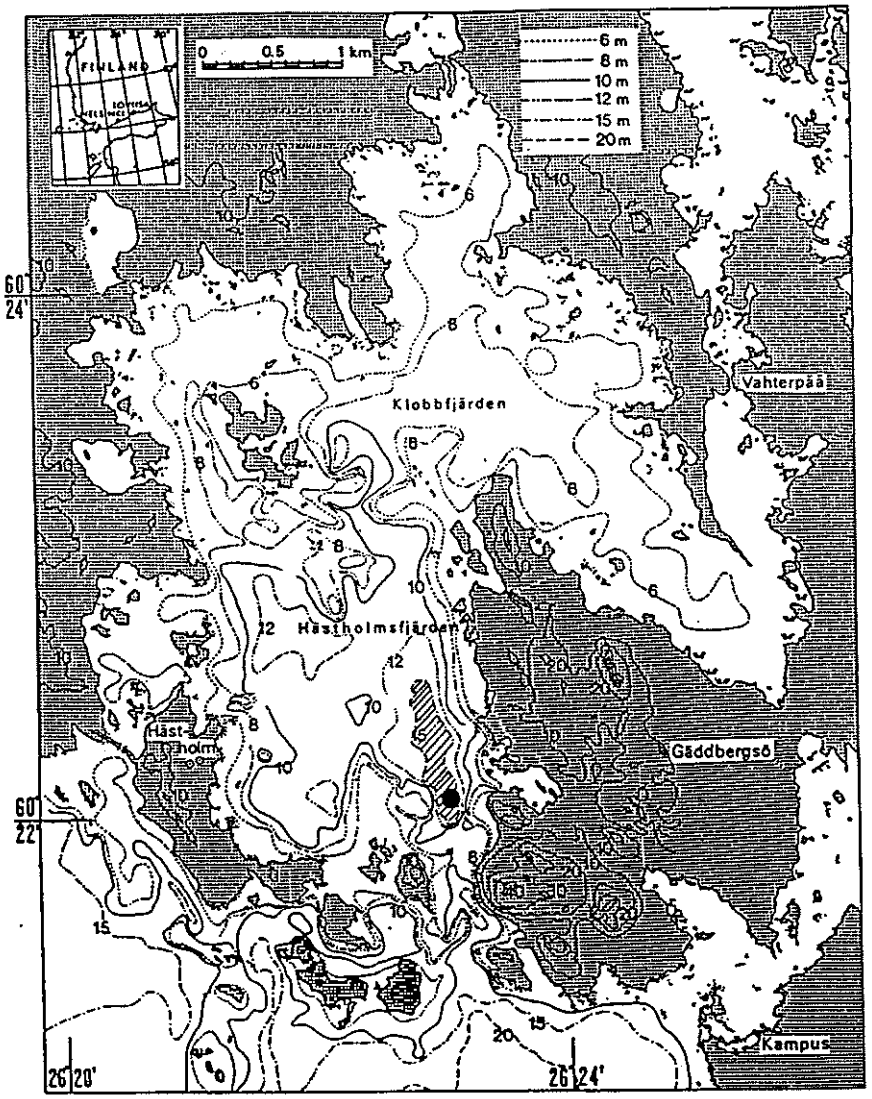

Figure 1. Hästholmsfjärden, the discharge area of the Loviisa NPP and the site of the sampling station.

Water samples for analysing the gamma-emitting nuclides and plutonium were taken five times with a submerged pump-hose system from 0.5 metres above the bottom. One part of water samples were taken directly as raw water, the other part was filtered. Filtering was performed through 0.45 µm membrane filters. The volumes were 30/100 l (raw/filtered) for gammanuclides and 100/200 l for plutonium analyses. During the sampling the boat was moored to a buoy placed at the site.

Sediment samples were taken with a Gemini twin corer. Four parallel tubes were taken and the cores were sliced in 1-cm-slices. The parallel slices were combined for analyses of gamma-emitting nuclides and plutonium.

Before measuring by gammaspectrometry the water samples were evaporated to more concentrated volumes (0.5 l). Sediment samples were freeze-dried and homogenized before analyses. Membrane filters were air dried and pressed into a geometry suitable for gamma measurement. Gamma spectra were measured with high-purity germanium detectors in low-background shields[10].

Water samples were analysed for plutonium by coprecipitating plutonium from water column with hydroxides. Plutonium was then separated by anion exchange and electrodeposited on a steinless steel disc. Sediment samples and membrane filters were wet ashed and acid leached with strong acids before anion exchange. Plutonium activities were measured with an alpha spectrometer and a multichannel analyzer. Pu-242 was used as a tracer [11].

Results and discussion

The total depletion of oxygen with increased nitrogen and phosphorus concentrations in the hypolimnion of the deep occurred during two periods in the late summer and autumn (August 16-24 and September 13-28 1995). During these periods the total phosphorus and total nitrogen concentrations of the near-bottom water increased at their highest to 440 mgP/l and 1355 mgN/l , as the corresponding values in surface water were 37 mgP/l and 660 mgN/l (Fig.2).

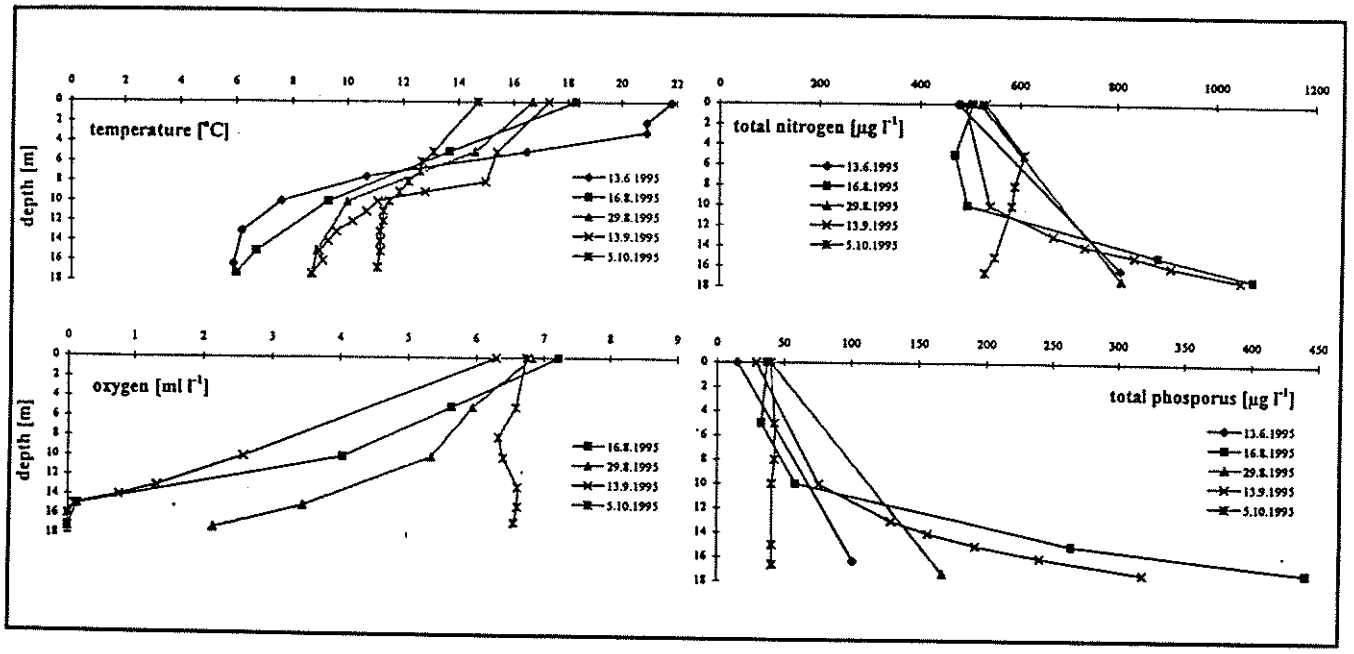

Figure 2. Variations of temperature, concentrations of oxygen, total nitrogen and total phosphorus in the water of the deep during June - October 1995.

Cs-137 and Cs-134 were the only artificial gamma-nuclides detected in the water samples. No significant increase was found in their concentrations (Fig. 3). More than 95% of Cs-137 was in soluble form. During the latter anoxic period the Cs-137 concentration was slightly higher than during the oxic period afterwards.

25-70% of plutonium was in the filtered water, but no systematic variation with oxygen depletion was observed in the concentrations of Pu-239,240 (Fig.3). The larger total amounts of plutonium in water phase in the first two samples were probably due to the greater amounts of particulate matter in these samples.

The measured water contents of the topmost sediment slice varied at differed sampling times between 93-96%. Values are typical for the sediment accumulation areas[12]. Only at the first sampling date about 0.5 cm oxidized layer at surface of the sediment was visible. At other sampling times, according to the redox measurements, the surface sediment was clearly reduced.

The results of Cs-137 from uppermost sediment slices and those of Pu-239,240 are presented in Table I. The contents of cesium and plutonium in surface sediments (Bqm^{-2}) have also been calculated in relation to the cumulative dry matter ($kg\ m^{-2}$) to avoid the effects, which depend on sediment compression, so that the results for different sampling times could be more comparable with each other (Fig.4). The results differed from each other, but no effect of the anoxic conditions can be seen. However, dry matter contents in the topmost sediment slice were higher during the anoxic period. This could be caused by differences in slicing or by the softest uppermost layer or part of it being remobilized under these conditions.

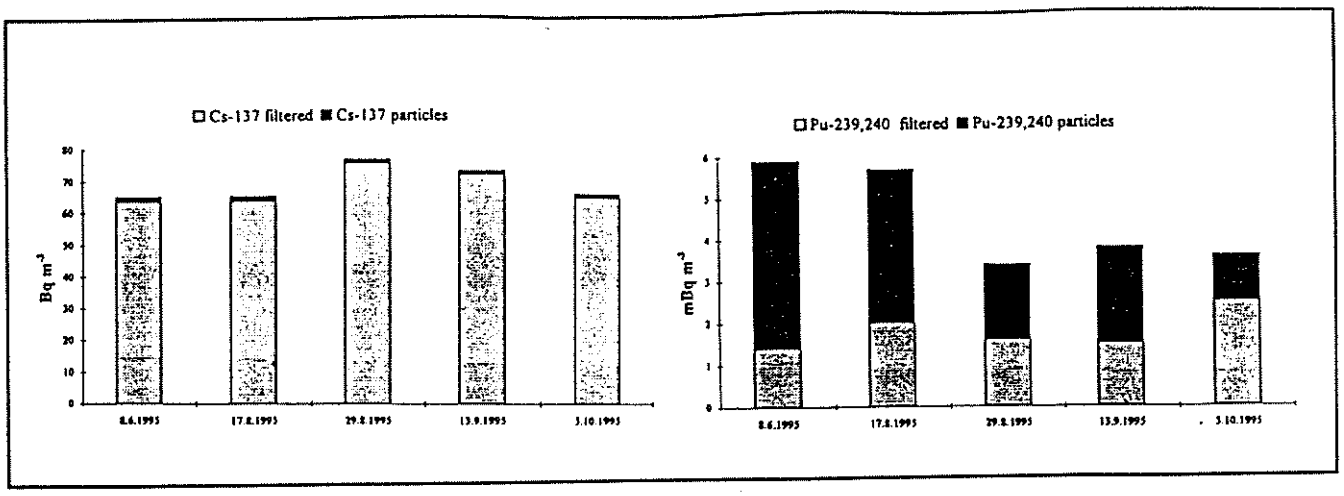

Figure 3. Concentrations of Cs-137 and Pu-239,240 in filtered water and particulate matter of near-bottom water.

Table I. Cumulative dry matter contents and concentrations of Cs-137 and Pu-239,240 in uppermost sediments (Bq kg $^{-1}$ d.w.) in 1995.

Date	Slice cm	Cs-137	Pu-239,240	Cum. dry matter kg m^{-2}
8.6.	0 - 1	850	1.08	0.527
	1 - 2	1280	1.45	1.467
18.8.	0 - 1	850	1.09	0.468
	1 - 2	950	1.15	1.268
29.8.	0 - 1	990	1.32	0.647
	1 - 2	1170	1.27	1.636
13.9.	0 - 1	1190	1.25	0.945
	1 - 2	1360	1.18	2.188
5.10.	0 - 1	870	1.15	0.547
	1 - 2	950	1.25	1.278

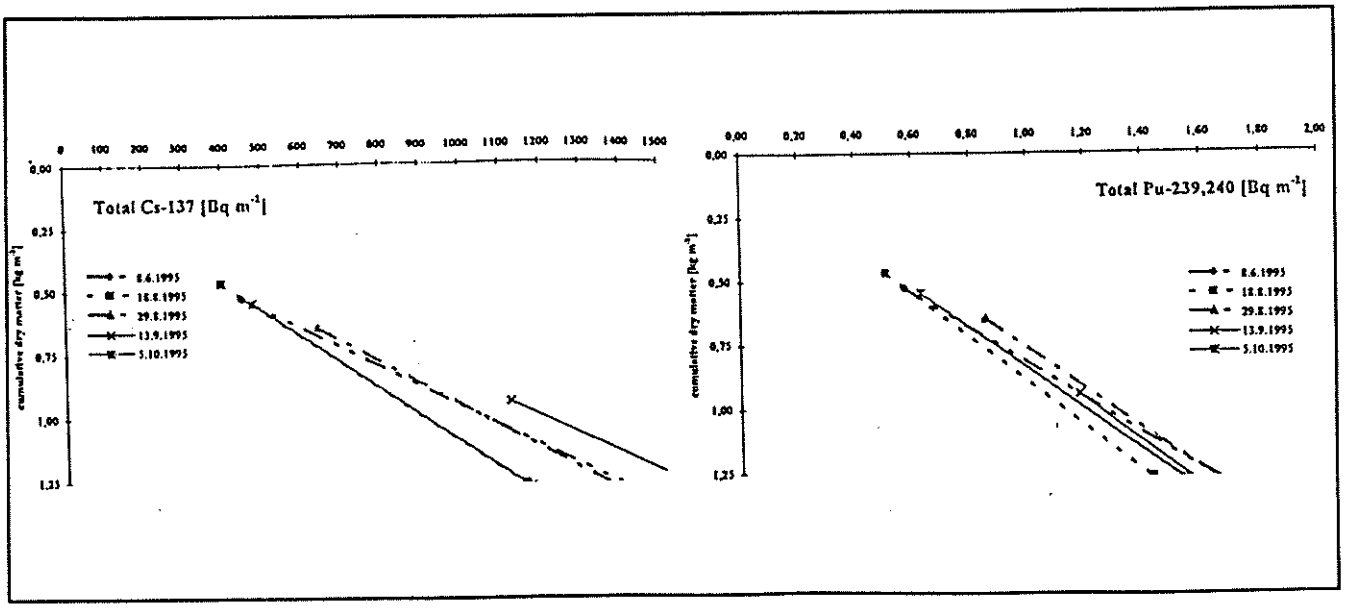

Figure 4. Total amounts of Cs-137 and Pu-239,240 for cumulative dry matter contents in surface sediments.

Concentration factors (C_f = concentration in sediment $Bqkg^{-1}$ / concentration in filtered bottom water Bql^{-1}) have also been calculated. Values of C_f for Cs-137 varied from 1.4 - 1.7 x 10^4, the average was 1.6 x 10^4 . Respectively, the factors for Pu,239,240 varied from 4.6 - 8.3 x 10^5 , the average being 6.9 x 10^5. Thus the values for different sampling times did not differ much from each other. If any release occurred the C_f should be smaller during and just after the anoxic periods. The concentration factors of plutonium were of same order of magnitude as the distribution factors (K_d) reported in other studies for marine sediments. For cesium they were about 10- 10^2 times higher, but most of the K_d-values presented in literature were calculated for pore waters[1,13,14,15].

As a whole, no clear remobilization of the Cs-137 or Pu-239,240 from sediment to water caused by the depletion of oxygen can be seen. The sampling was, however, performed with a relatively rough sampling method of near-bottom water. The project continues by repeating the sampling and analysing procedures in 1996 with a more specific sampling method by which an attempt will be made to determine the activity gradients of cesium and plutonium between the top of the sediment and the water. We also propose to evaluate more specifically the amount of particulate matter in water and resolve its contribution to the measured activities of the water samples.

In a study of lake sediments it has been calculated that 18-33% of cesium but much less of plutonium should be in mobile form in sediments [8]. Theoretically, if about four percent of the total Cs-137 adsorbed in surface sediment in this deep is released to water, it is possible to observe the subsequent concentration change in near-bottom water but not clearly in sediment, assuming that the released cesium stays in the bottommost one metre layer. Respectively, release of two percent of Pu-239,240 from sediment to water can be observed. A release of this order of magnitude should result in an increase in activities of 2% for Cs-137 and even 26% for Pu-239,240 in the whole water volume of the deep of Hästholmsfjärden.

This project is included to the NKS/EKO-1 Marine Radioecology Project.

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DETERMINATION OF PB-210 IN SEDIMENT SAMPLES BY GAMMA RAY SPECTROMETRY: APPLICATION OF AN EFFICIENCY TRANSFER METHOD

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Abstract - A semi-empirical method to obtain germanium detector efficiencies was applied to the analysis of Pb-210 by gamma ray spectrometry. The computer method was used to transfer a measured full energy peak efficiency of a standard geometry to various heights and densities of sediment samples. The obtained efficiencies were utilized in the determination of Pb-210 concentration of Baltic Sea sediments. The results show a good overall agreement to those obtained by conventional alpha spectrometry of Po-210.

Introduction

When analysing Pb-210 activity in environmental samples by gammaspectrometry one has to take into account the self-absorption of low energy gamma rays in the samples. Due to the wide variety in sample size, density and elemental composition, efficiency calibration by measuring standard activity materials would be impractical. Therefore several procedures have been developed to calculate the effect of photon attenuation in the sample. In this study a method to calculate germanium detector efficiencies for higher photon energies (60 keV - 1800 keV) was applied to low energy (46.5 keV) gamma rays of Pb-210.

Material and Methods

The full-energy peak efficiencies for 46.5 keV photons of three n-type hyperpure germanium detectors (Table I) were first determined measuring a known amount of Pb-210 in water solution (30 ml). Diameter and height of the reference geometry were 42 mm and 22 mm, respectively.

The computer method was then used to transfer the measured efficiencies to other sample heights, densities and material. The basis of the method is the assumption that the probability for *a detected* photon to be registered at its full energy peak depends on photon energy only and not on geometrical factors. Thus from the peak efficiency of a detector in one counting geometry one can derive efficiency for any geometry by integration over the source and the detector and taking attenuation in all absorbing layers into account.[1,2,3]

The results of the calculations are three sets of correction coefficients, one for each detector, covering the sample heights from 15 to 25 mm and densities from 0.5 to 1.4 g/cm^3. The mass attenuation coefficient of sample material was supposed to be equal to SiO_2.

The obtained efficiencies were utilized in the determination of Pb-210 profiles of four Baltic Sea sediments cores which were collected for the NKS/EKO-1 Marine Radioecology Project. The aim of this project is determination of sedimentation rates in the Baltic Sea by means of four different evaluation methods.

A routine method for dating the sediments is based on Po-210 measurements with alpha spectrometry, assuming that Po-210 is in equilibrium with its grandmother Pb-210. After decomposing the sediment sample in strong acids Po-210 was spontaneously deposited on a silver plate. For yield determination Po-209 was used as a tracer. Alpha-activity was measured with a PIPS-detector and a multichannel analyzer.

Results and discussion

The three set of results obtained by the two methods, Pb-210 and Po-210, are compared in Figs. 1. A good overall agreement is obvious. The error bars show the statistical uncertainty of the counting after subtraction of background counts. Summarising altogether 100 results yields average deviation of 10 %. Detailed investigation of the results shows no correlation between deviation and sample height nor between deviation and detector. All except six samples have density in the range 0.5-0.8 g/cm^3 and among them there is no correlation between differences in Po-210 results and density. Higher density samples are all from the same sampling station (Hanko Bay), which has unusual Pb-210-profile probably indicating varying composition. Thus increasing relative deviation with increasing densities (in the range 0.9-1.4 g/cm^3) could be caused by higher attenuation than that of SiO_2.

The comparison of the methods will continue when further EKO-1 material is available. These preliminary results, however, encourage to use gammaspectrometry instead of slower and more expensive Po-210-method, at least within the accuracy needed for sediment dating purposes.

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Table I. Detailed description of detector dimensions is essential for accurate efficiency calculation. All dimensions in mm.

DETECTOR	N1		L1		L8	
	Length	Diameter	Length	Diameter	Length	Diameter
Crystal	51.0	52.2	54.5	53.6	52.1	53.2
Core	46.3	8.0	47.1	10.2	35.1	9.0
Absorbing layers	top	side	top	side	top	side
Ge	0.0003	0.0003	0.0003	0.0003	0.0003	0.0003
Al	-	0.5	-	0.5	-	0.7
Mylar	0.013	-	0.013	-	-	-
Vacuum	7.2	8.4	4.2	7.2	7.0	9.3
Al	-	1.2	-	1.27	-	1.5
Be	0.5	-	0.75	-	0.5	-
Air	5.5	-	3.7	-	3.7	-
Polystyr.	1.0	-	3.15	-	0.25	-
Cu	0.11	-	-	-	-	-

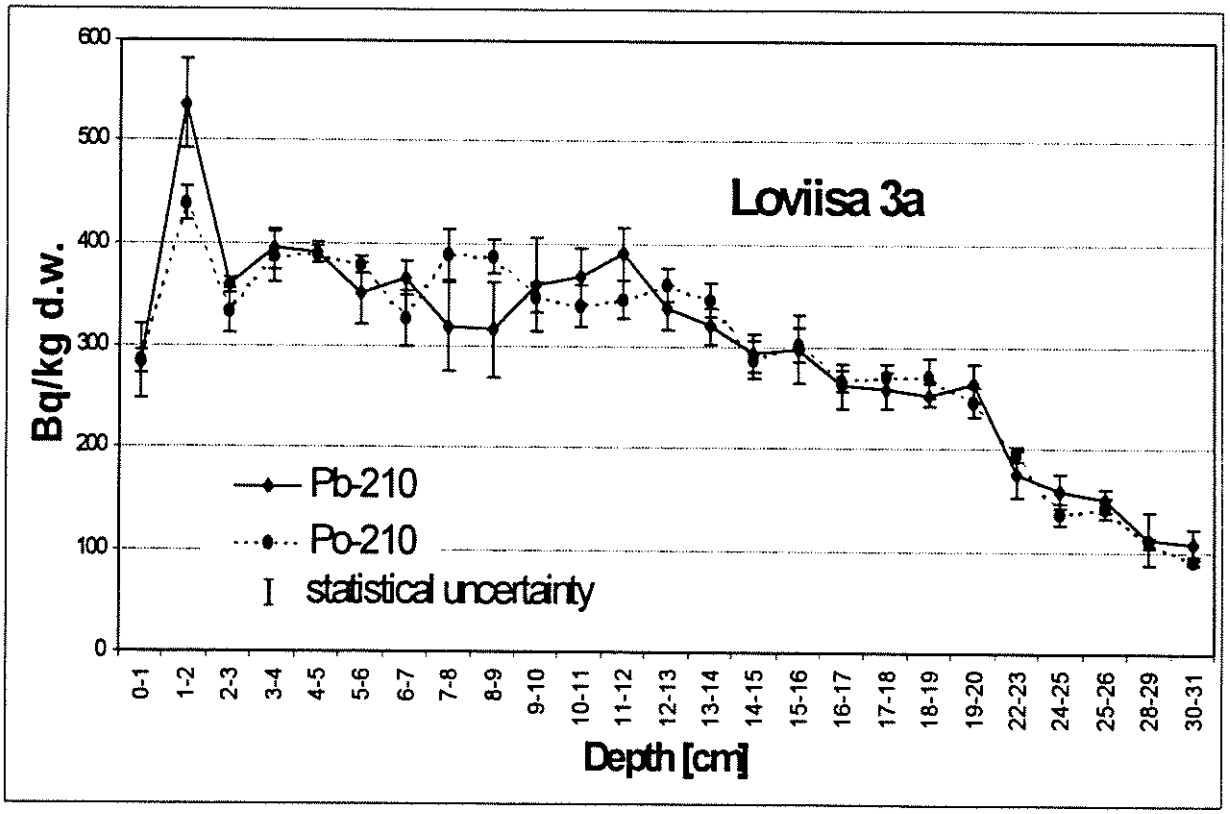
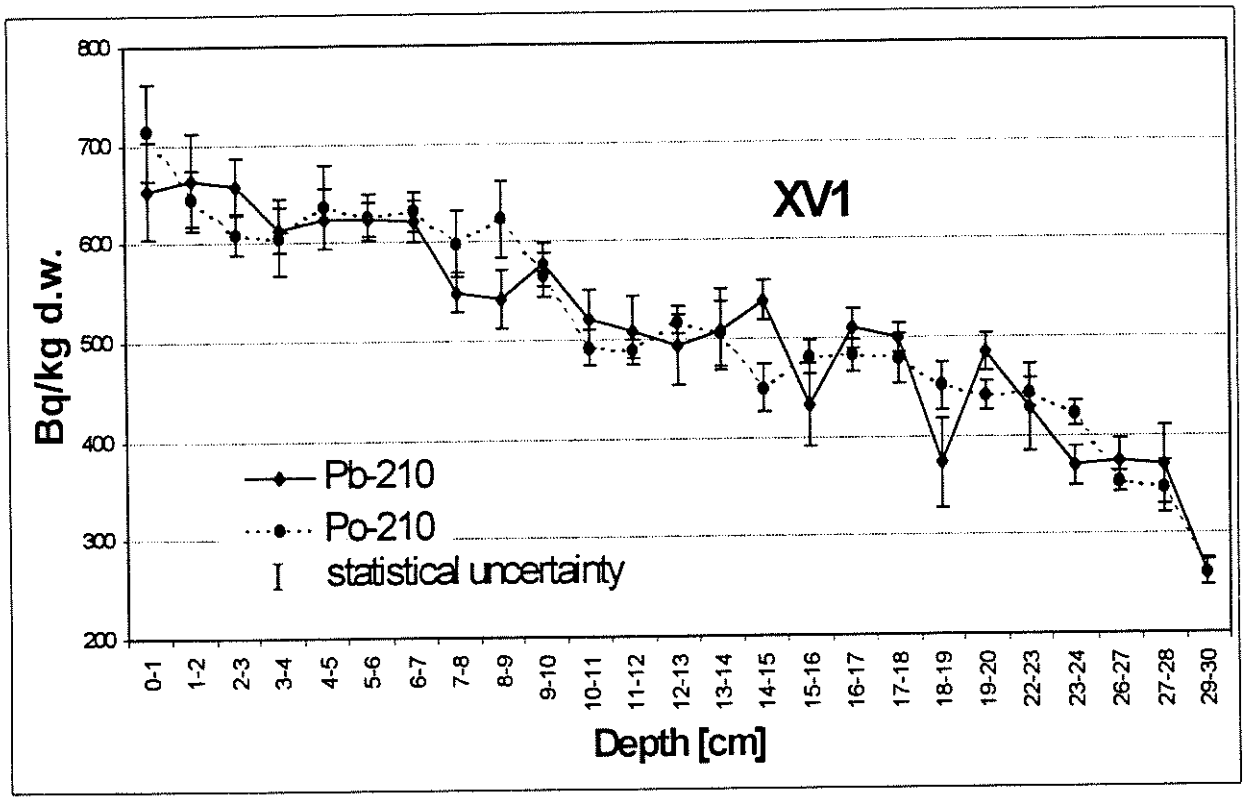


Fig. 1. Pb-210-activities obtained by gammaspectrometry compared to Po-210-results at three sampling stations. Error bars indicate statistical uncertainty after subtraction of background counts, no calibration error included.



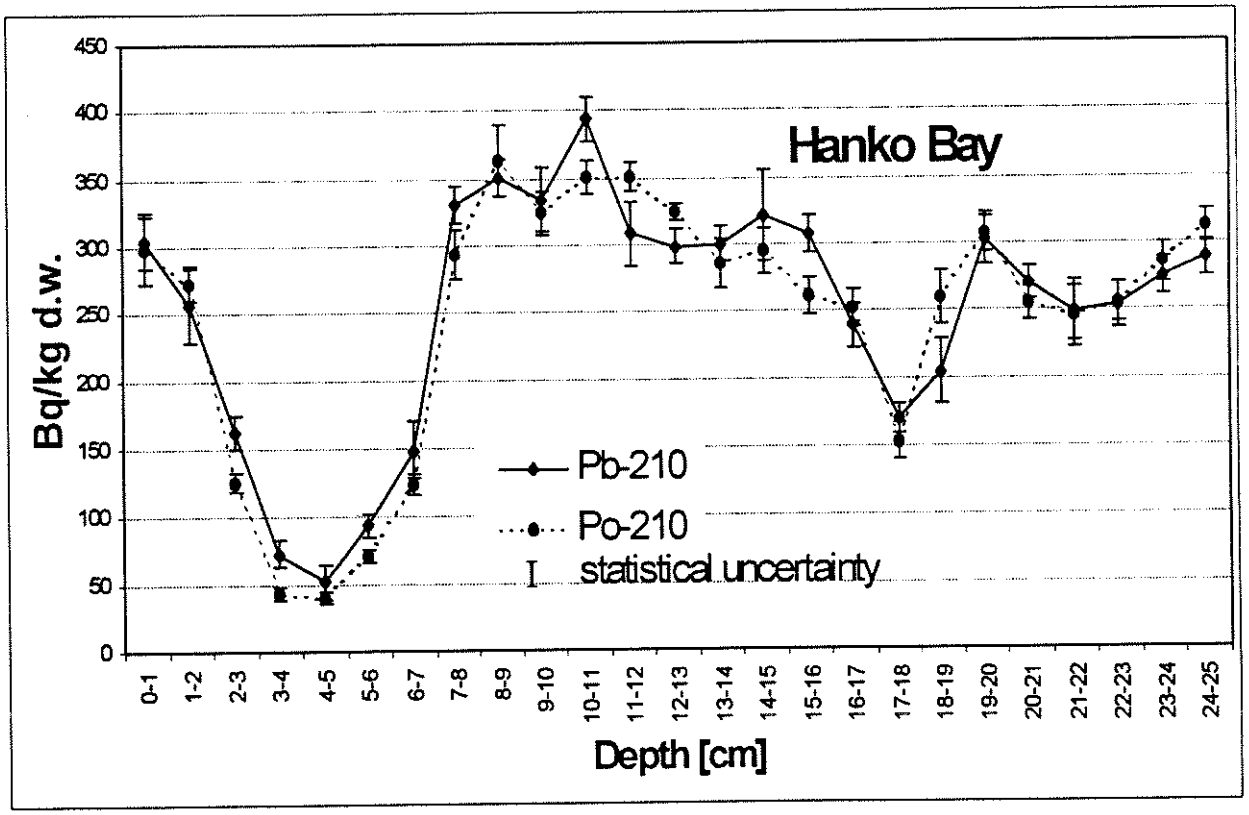


Fig 1. continued.

Datering ved hjelp av Po-210 og Pb-210 i sjø- og elvesedimenter

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Sammendrag

Sedimenter fra Karahavet (marint) og vannsystemene rundt PA Majak (limnisk) har vært datert ved hjelp av Po-210 eller Pb-210. I prøver med høy aktivitet av andre γ-emitterende nuklider (som Cs-137 eller Co-60) er Pb-210 vanskelig å bestemme ved hjelp av germanium-detektor. Disse prøvene har derfor vært analysert bare med hensyn på Po-210. De to metodene viser god overensstemmelse, og sedimentasjonshastigheter i ulike områder er bestemt. Sedimentasjonshastigheter er en viktig parameter når en skal vurdere konsekvensene av forurensing i elver og havområder.

Innledning

Nuklidene Pb-210 og Po-210 brukes ofte til å datere sedimenter. De er begge datternuklider av Rn-222 som produseres av Ra-226. Radon unnslipper fra jordens overflate i form av gass, og datterproduktene vaskes senere ut fra atmosfæren. Pb-210 er det datterproduktet som har lengst halveringstid - 22 år.

I akvatiske systemer adsorberes Pb-210 til partikler i vannfasen, og sedimenteres sammen med disse. Ved å måle Ra-226 og Pb-210 innholdet i sedimentene er det derved mulig å bestemme sedimentasjonshastigheten.

Metoder

Analyse av Pb-210

Pb-210 sender ut 4% γ-stråling ved energi 46,5 keV. Ved så lave verdier er det viktig å korrigere for selvabsorbsjon i prøven. Prøvene måles på Ge-detektor med beryllium-vindu (n-type). Målegeometrien er valgt for å minimalisere effekten av absorpsjon av γ-stråling i selve prøven og høyden på prøvene er bare 8 mm. Ved å måle prøven med og uten Pb-210 kilde kan en korrigere for den γ-strålingen som måtte bli absorbert i prøven.

Korreksjon for selvabsorbsjon er beregnet etter (Cutshall et al.) [1]. Forholdet mellom A, foton utsendelseshastigheten fra prøven, og O, dempet utsendelse fra prøven kan bestemmes ved målinger av T og I:

$$\frac{A}{O} = \frac{\ln(T/I)}{T/I - 1}$$

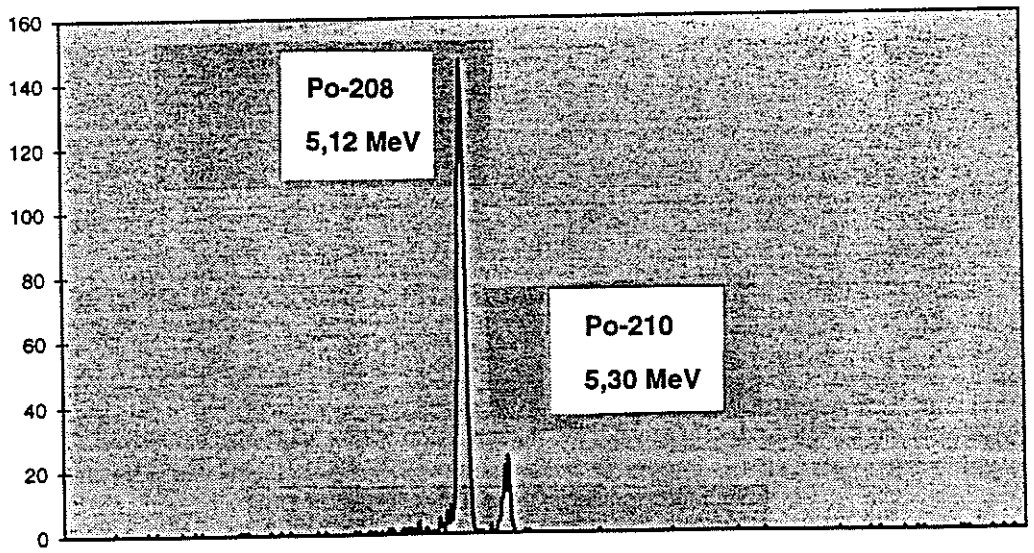
T = dempet γ-stråling og bestemt ved telling av prøven med ekstern Pb-210 kilde

I = udempet γ-stråling og bestemt ved telling av Pb-210 kilde uten prøve

I prøver med høy aktivitet av f.eks. Cs-137 eller Co-60 vil bestemmelse av Pb-210 aktivitet vanskeliggjøres på grunn av høy compton-bakgrunn, da Pb-210 aktiviteten ofte er lav [2].

Analyse av Po-210

En prøvemengde på 1-2 gram ble tilsatt spike (Po-209 eller Po-208) og behandlet med konsentrert HCl ved 140-150°C. Po-210 i løsningen ble deretter avsatt på en Ni-plate i løpet av 3-4 timer ved 65°C. Analyseprosedyren er tilpasset i forhold til prosedyre beskrevet av Häsänen [3]. Aktiviteten på platen ble bestemt ved hjelp av alfaspektrometri. Prøvemateriale som ikke løste seg ble tørket og veid for å bestemme utbytte ved oppslutning. Mengden prøvemateriale som løses varierte fra 20% til 60% avhengig av sedimenttype. Vi antar i analysen at alt Po-210 i prøven følger løsningen.

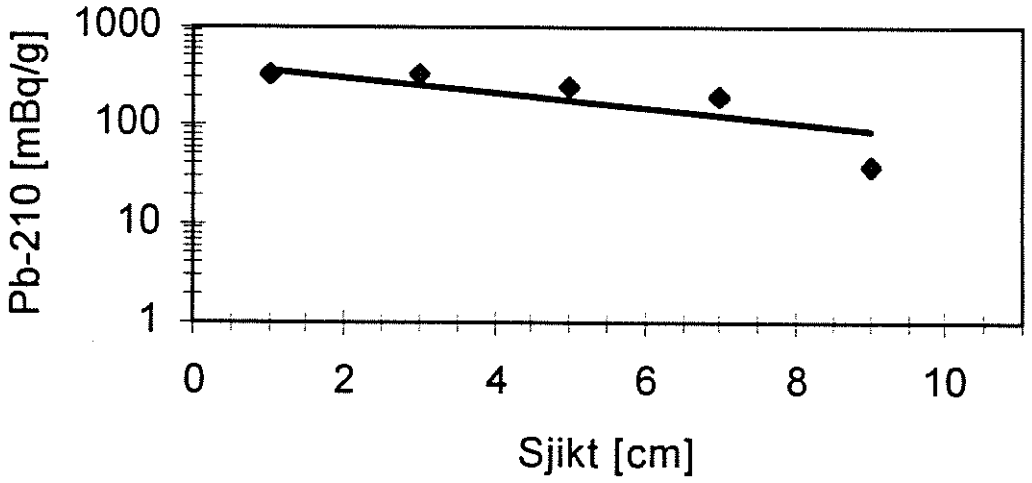


Figur 1. Eksempel på alfa-spekter med Po-208 og Po-210.

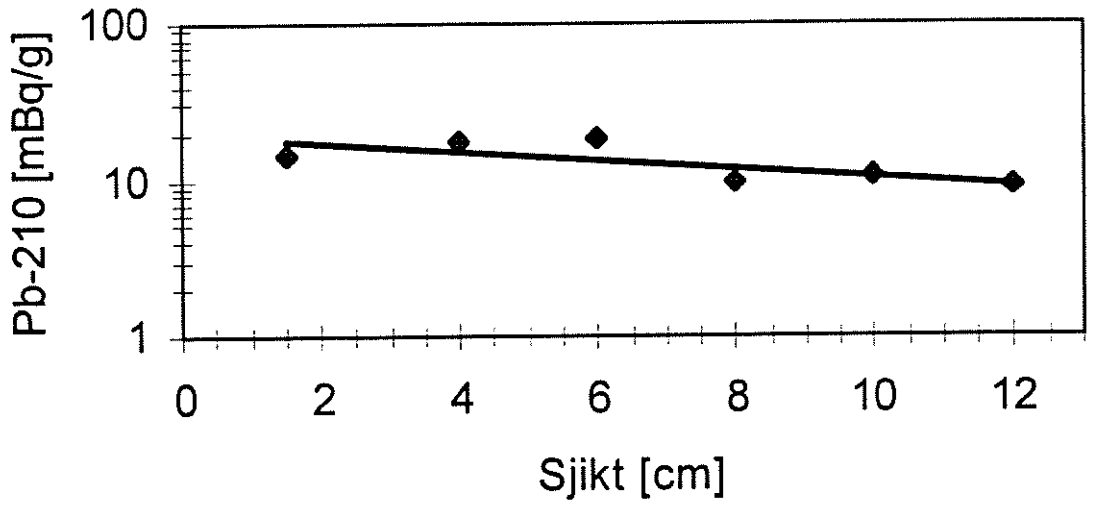
Resultater og diskusjon

Den største usikkerheten ved datering oppstår i forbindelse med selve prøvetakingen. Sammenpressing av sedimentet skjer i det prøven hentes opp, det er større friksjon langs kantene på prøvetakingsrøret, og en ny sammenpressing kan forekomme i forbindelse med sjikting av profilen [4]. I marine miljø er sedimentasjonshastigheten som regel lav, og 1 cm sediment kan ofte representere 5-10 år. Tynne sjikt vil imidlertid som regel introdusere nye feilkilder og dateringen blir derved mer usikker. Biologisk aktivitet i sedimentene (mark o.l.) vil også medføre at resultatene blir vanskelige å tolke. I dette prøvematerialet ble marine sedimenter sjiktet i 1 cm tykkelse, og ferskvannssedimenter i 2 cm tykkelse. Hele profilen ble brukt, men for sikrere bestemmelse hadde det vært ønskelig med lengre profiler.

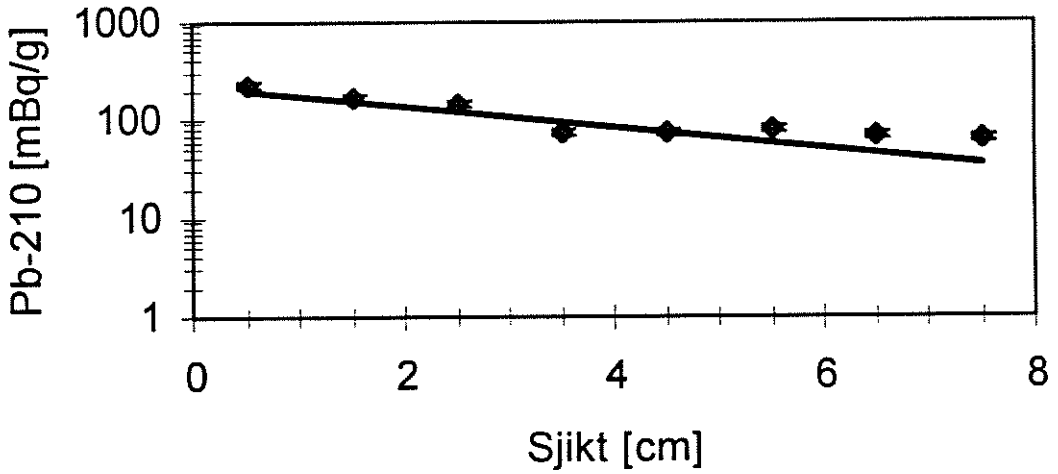
Ved kjemisk analyse av Po-210, som er en α-emitter, antar man likevekt mellom Pb-210 og Po-210. Dette er ikke nødvendigvis tilfelle i biologisk materiale. Videre antar man at all Po-210 lar seg løse, og at ikke Po-210 unnslipper fra løsningen under analyse (kokepunkt 254°C). Parallelle prøver viser imidlertid god overensstemmelse (5-7% avvik) og høyt kjemisk utbytte av spike tilsier at Po-210 holdes i løsningen.



Figur 2. Resultater fra analyse av sedimenter fra basseng nr. 10 ved PA Majak. Sedimentasjonshastighet er bestemt ved analyse av Po-210 til 1,2 mm/år.



Figur 3. Resultater fra analyse av sedimenter fra elva Techa ved PA Majak. Sedimentasjonshastighet er bestemt ved analyse av Po-210 til 5,2 mm/år. Resultatene indikerer en viss blanding i de øverste lagene.



Figur 4. Resultater fra analyse av sedimenter fra Stepovogo-bukta ved Novaja Semlja. Det er tatt to parallelle analyser i hvert sjikt. Sedimentasjonshastighet er bestemt ved analyse av Pb-210 til 1,4 mm/år.

Typiske sedimentasjonshastigheter i Barentshavet ligger på 0,1-0,5 mm/år. Noe høyere verdier er forventet i Karahavet, spesielt i buktene langs Novaja Semlja og i dyprenna.

Sedimentasjonshastigheten i indre del av Stepovogo Bukta er bestemt til 1,4 mm/år. IAEA-MEL har tidligere bestemt sedimentasjonshastigheten i Stepovogo til 1,3 mm/år.

Sedimentasjonshastighet i basseng 10 i Majak er rundt 1,2 mm/år, mens den er vesentlig høyere i elva Techa, 5,2 mm/år.

Konklusjon

Analyse av Pb-210 og analyse av Po-210 viser rimelig god overensstemmelse med hverandre. Po-210 analyser er vesentlig mer arbeidskrevende enn Pb-210 analyser, men er nødvendig i de tilfellene en har mye γ-aktivitet i prøvene.

Sedimentasjonshastighet i Stepovogo-bukta er bestemt med god overensstemmelse med tidligere undersøkelser utført av IAEA [5]. Det er ikke gjort tilsvarende undersøkelser av sedimentasjonshastighet i bassengene ved PA Majak.

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Det 11. Ordinære möte, Nordisk Selskap for Strålevern
Det 7. Nordiske Radioökologiseminar
Reykjavík, Island, 26-30 aug. 1996

Mätning av radioaktiva ämnen i betesgräs och mjölk i händelse av en kärnenergiolycka. Förslag till beredskapsorganisation i Sverige

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Abstract-Statens strålskyddsinstitut (SSI) skall i händelse av en kärnenergiolycka ge råd till övriga myndigheter angående rekommendationer i syfte att minska stråldosen till befolkningen. SSI måste därför snabbt få en korrekt bild av den aktuella strålningssituationen, särskilt inom lantbruks- och livsmedelssektorn och där speciellt inom näringskedjan betesgräs-ko-mjölk- människa. Förslag till en beredskapsorganisation för mätning av radioaktiva ämnen i betesgräs och mjölk har utarbetats. Förslaget bygger på utnyttjande och samordning av befintliga instanser, ansvariga för olika former av beredskap, provtagning eller mätning. Ett beredskapsregister upprättas över mjölkkogårdar omfattande ca. 5 % av totala antalet för landet, s.k *registergårdar*. Vid ett radioaktivt nedfall väljs ur registret *provgårdar* i aktuella nedfallsområden för provtagning och mätning av betesgräs samt *försöksgårdar* för betesförsök, där överföringsfaktorer för gräs-mjölk studeras.

Det följande utgör en sammanfattning av utredningsförslag av Andersson & Finck (1996).

Bakgrund och syfte

I Regeringens proposition 1991/92:41 om samhällets åtgärder mot allvarliga olyckor åläggs Statens jordbruksverk (SJV) och Statens livsmedelsverk (SLV) att formalisera sin planering för åtgärder i samband med utsläpp av radioaktiva ämnen samt att tillsammans med Statens strålskyddsinstitut (SSI) utarbeta planer och metodbeskrivningar för provtagning och mätning.

SSI skall i händelse av en kärnenergiolycka eller annan radiologisk hotsituation ge råd till övriga myndigheter angående föreskrifter och rekommendationer i syfte att minska stråldosen till befolkningen. SSI bör därför i ett sådant läge så snart som möjligt få en korrekt bild av den aktuella strålningssituationen, särskilt när det gäller lantbruks- och livsmedelssektorerna. Inom dessa är näringskedjan betesvegetation-ko-mjölk-människa av speciellt intresse, eftersom radioaktiva ämnen snabbt tas upp hos betande kor och

utsöndras i mjölken. Mjölk och mjölkprodukter är viktiga baslivsmedel (den årliga konsumtionen motsvarar ungefär 350 kg mjölk per person) och kan på så sätt ge ett väsentligt bidrag till internstråldosen.

Det kan nämnas att antalet mjölkkor totalt för landet uppgår till drygt 480000, fördelade på ca. 17000 besättningar. Mjölkproduktionen är koncentrerad främst till de södra delarna av landet. Storstadsregionerna är underskottsområden, när det gäller "mjölkbalans", dvs mjölkproduktionen (antal mjölkkor) relaterad till konsumtionen (befolkningstalet). Totalt invägd mängd mjölk vid landets mejerier var 1994 drygt 3,3 miljoner ton, varav 42 % användes till konsumtionsmjölkprodukter och 37 % till ostframställning.

Utförande

Som ett led i uppbyggnaden av en beredskapsorganisation har SSI i samråd med de övriga nämnda myndigheterna formulerat ett utrednings- och forskningsprojekt gällande provtagning och mätning av radioaktiva ämnen i betesvegetation (här benämnd betesgräs) och mjölk i händelse av en kärnenergiolycka. Utredningsarbetet har utförts av Inst. för jordbrukets biosystem och teknologi, Sveriges lantbruksuniversitet (SLU), Alnarp, i samarbete med SSI och i samråd med Inst. för radiofysik, Malmö, samt med företrädare för SJV, SLV, Statistiska centralbyrån (SCB), Svenska Mejeriernas Riksförening (SMR) och Jordbrukets beredskapsorganisation.

Utredningen identifierar strålningsmässiga problem och diskuterar möjliga åtgärder för att begränsa aktivitetskoncentrationen i mjölk.

Principen för den nya beredskapsorganisationen för provtagning och aktivitetsmätning, som dimensioneras för betesperioden, är att resurser och verksamheter vid befintliga organisationer skall utnyttjas och samordnas. Utredningen beskriver nuvarande mätberedskap och mätprogram och befintliga instanser som är ansvariga för olika former av beredskap eller för provtagningsverksamhet.

Utredningen föreslår bl.a. att:

* Ett beredskapsregister, baserat på ett urval av gårdar i Lantbrukets företagsregister (LBR), upprättas för varje län över jordbruksföretag med mjölkkobesättningar (gårdar). Beredskapsregistret omfattar tre kategorier beträffande antalet gårdar och provtagningsprogram: registergårdar, provgårdar och försöksgårdar. *Registergårdar* består av en förteckning över ca. 5 % av antalet mjölkkogårdar i varje kommun, dvs totalt ca. 900 st. *Provgårdar* väljs ur registret vid ett radioaktivt nedfall. Antalet sådana

gårdar bestäms av den aktuella nedfallsbilden, men kan vara 10-30 st per län. Vid provgårdarna tas betesprover för aktivitetsmätning. *Försöksgårdar* väljs vid ett radioaktivt nedfall ur registret för betesförsök med hela, alternativt delar av besättningen och med provtagning och mätning av betesgräs och mjölk. Antalet försöksgårdar bör vara 2 per kontraktslaboratorium (se nedan), vilket innebär totalt 18 st. Urvalskriterier för de olika kategorierna av gårdar lämnas.

* Lantbruksenheten eller motsvarande vid resp. länsstyrelse upprättar, administrerar och uppdaterar registret.

* Jordbrukets beredskapsorganisation (region-/områdessamordnare) anlitas som rådgivare vid registeruppläggning och vid urval av gårdar.

* Lantbrukarna vid registergårdarna informeras. Avtal upprättas med försöksgårdarna i en nedfallssituation, då också åtaganden från försöksgårdarna uppmärksammas och ekonomisk ersättning garanteras.

* SCB:s Objektiva skördeuppskattningar (provtagningsorganisationen) engageras för provtagning av betesgräs vid provgårdar, senast 4 dygn efter ett larm. Om SCB inte kan medverka, måste andra alternativ diskuteras.

* Mjölkbedömningen ställer mjölkprover från register- eller provgårdar till förfogande för aktivitetsmätning.

* SSI:s kontraktslaboratorier (f.n. 9 st, fördelade över landet) medverkar vid urval av prov- och försöksgårdar. Utför betesförsök på försöksgårdar (se ovan). Provtar betesgräs på provgårdar (tills SCB:s provtagarorganisation övertar funktionen) och utför aktivitetsmätningar. Medverkar vid urvalet av mjölkprover vid mjölkbedömningen. Beräknar aktuella samband mellan aktivitet i betesgräs och i mjölk. Utför ev. fältgammamätningar. Rapporterar resultaten till SSI. Medverkar vid utbildning av SCB:s provtagare. Övar provtagningsrutiner och uppgiftsinsamling, eventuellt på "fast" försöksgård.

* Kärnkraftverken eventuellt medverkar vid mätning av prover.

* SSI i samråd med SJV och SLV har det övergripande ansvaret för beredskapsorganisationen. Verkar för att länsstyrelserna upprättar beredskapsregistret. Förbereder utformning av avtal för försöksgårdar. Medverkar vid utarbetande av informationsmaterial till länsstyrelser samt till register-, prov- och försöksgårdar. Förtecknar ansvariga handläggare inom de medverkande organisationerna. Medverkar vid utbildning av SCB:s provtagare.

* SLU i samarbete med SSI utarbetar instruktioner för provtagning och uppgiftsinsamling vid försöksgårdar och vid provgårdar. Utformar i samråd med SCB instruktioner gällande provtagning vid provgårdar (att infogas i SCB:s instruktionspärm för provtagare). Medverkar vid utbildning av SCB:s och kontraktslaboratoriernas personal. Utarbetar informationsmaterial till samtliga tre gårdskategorier.

Åtagandena från myndigheter och organisationer anges i utredningen också som *grundförberedelser* resp. *agerande i akut skede.*

Med den här föreslagna beredskapsorganisationen för provtagning och aktivitetsmätning av betesvegetation och mjölk ges möjligheter att på ett effektivt sätt ta fram de beslutsunderlag som myndigheterna behöver. Tillförlitliga beslutsunderlag gör det möjligt att genom olika åtgärder begränsa internstråldosen till befolkningen. Korrekta beslutsunderlag innebär också ekonomiska besparingar när det gäller djurhållningen till gagn för såväl samhället som för den enskilde lantbrukaren.

Provtagningsrutiner gällande betesgräs kommer att beskrivas i en följande del. Principerna för provtagning bör diskuteras för eventuell samordning inom de nordiska länderna.

Referens

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Local reference measurements of gamma radiation in Sweden Experiences from seven years of measurements

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Abstract - In Sweden a system of portable dose rate meters is used for emergency preparedness in case of situations with fallout from nuclear accidents or nuclear weapons. Portable instruments were chosen rather than a stationary monitoring system because they provided mobility and better spatial resolution. In 1987, the health and safety administrations in the 286 local districts were equipped with instruments and an education programme was carried through. To maintain knowledge and competence, a quality programme was defined by the Swedish Radiation Protection Institute. The quality programme stated, among other things, that measurements should be done in 2 - 4 well defined reference points in each local district every seventh month. The purpose was to gain understanding of the seasonal variation in the radiation background and to maintain the knowledge of how measurements are performed and evaluated. In all, about 900 reference points were defined and labelled. Up to 1996, more than 10000 reference measurements of the dose rate have been performed, covering all months of the year. The effects of different thickness of the snow carpet can be clearly seen in the data and the decay of cesium-134 in the Chernobyl fallout in the northern part of Sweden can be identified. The experiences of the system and procedures are good. The large number of measurement points provides, at relatively low cost, a good spatial resolution. In case of a fallout situation there will be enough measurement data to construct maps of the dose rate for the whole of Sweden within 12 - 24 hours after an alarm. Since the background radiation and its variation is well known in each reference point, a dose rate increase in the order of 0,05 μSv/h above background can be identified. The most important experience is that the system gives the local districts the possibility to make reliable dose rate measurements without having to share resources provided by central authorities.

Introduction

In Sweden, before the Chernobyl accident, portable dose rate instruments were aimed at measuring the relatively high dose rates that could be feared from nuclear weapons fallout. These instruments were not sensitive enough to be operated in the low dose rate region. The only measuring system that could be used for large areas was an airborne spectrometer operated by the Swedish Geological CO and a limited number of survey instruments for prospecting of uranium. Using aerial spectrometry the first map of the Chernobyl fallout in Sweden was produced within two weeks. The distance between flight lines was 50 - 100 km, thus leaving large areas uncovered at this early stage. In addition there were 24 fixed monitoring stations which gave certain information of the time variation in the dose rates.

The lack of suitable instrumentation for rapid, large scale dose rate monitoring at dose rates near the background level was recognised by the Swedish Radiation Protection Institute (SSI). In 1987, SSI bought 300 dose rate instruments to be distributed to the 284 local districts and 24 regional county administrations. The intention was to create capabilities for each local district to measure dose rates in their own district. The instruments were specially designed for this purpose and could measure dose rates at the natural background level. A two day education programme was arranged to inform the health and safety authorities in the local districts and county administrations how to operate the instrument.

Measurements

A programme was defined by SSI to assure that high quality measurements could be performed (Wallin *et. al.* 1989). The quality program recommended, among other things, that specific reference point should be chosen where dose rate measurements should be performed. Between 2 - 4 reference points should be defined in each local district. They should be on open, grass covered ground and the position of each reference point should be exactly identified. In all, about 900 reference points were defined and labelled. Many local districts chose golf greens and football fields.

To maintain knowledge and competence to perform dose rate measurements in case of an emergency SSI recommended that dose rate measurements should be performed every seventh month in each reference point. This would provide local knowledge of the radiation background and an understanding of the seasonal variations. In case of a fallout situation it would be possible to draw conclusions of an additional dose rate above background. The first reference measurements started in September 1989. Until September 1996, more than 10000 reference measurements have been reported.

The dose rate instrument is based on a GM-tube and calibrated for ambient dose equivalent rate H*(10) at 662 keV. The uncertainty at other energies is in the order of ±30 %. Measurements are integrated during 10 minutes due to the low efficiency of the GM-tube. At the natural background level, this will give a standard deviation about ±20 %. The dose rate measurements are made 1 m above ground. A map of the reference points is shown in Fig 3.

Results

All measurements are reported to SSI and data are compiled into maps for each seven-month round. From the maps it is possible to visualise the seasonal variations and to identify the decay of the Chernobyl fallout.

In Fig 1 the average of all measurements during the summer period is mapped (June-91, August-92, May-94 and July-95). Average values for the winter period is shown in Fig 2 (January-92, March-93, December-94 and February-96). The shielding effect of the snow cover can be clearly seen in the northern part of Sweden. In this part the snow cover is usually several tens of centimetres and in the Northwest part more than one metre. Snow will cause a dose rate reduction of the combined background and Chernobyl fallout in the order of 0.05 - 0.10 μSv/h as can be seen in Fig 6.

If a fallout situation should occur the net contribution can be calculated by subtracting the known background. As an example such a subtraction was made to see if the decay of the Chernobyl fallout can be identified for the period 1991 to 1995. Fig 2 shows the difference in dose rate where the July-95 values have been subtracted from the June-91 values. In the northern part (the counties of Västerbotten and Västernorrland) a decrease in the order of 0.01 - 0.05 μSv/h can be seen for this time period. The decrease is mainly due to the decay of Cs-134. An example of the decrease is also shown for the local district of Gävle in Fig 5. This district received the heaviest part of the Chernobyl fallout in Sweden.

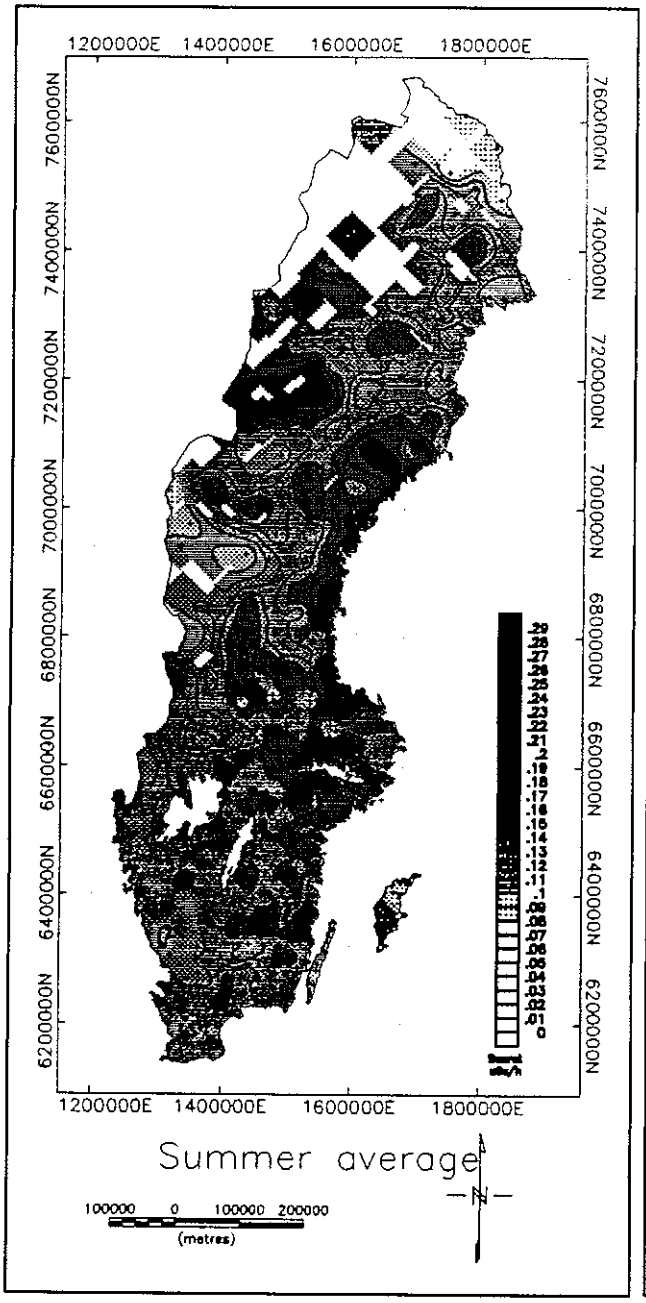

Fig 1. Average dose rates measured at the reference points during the summer period. (Jun-91, Aug-92, May-94, Jul-95).

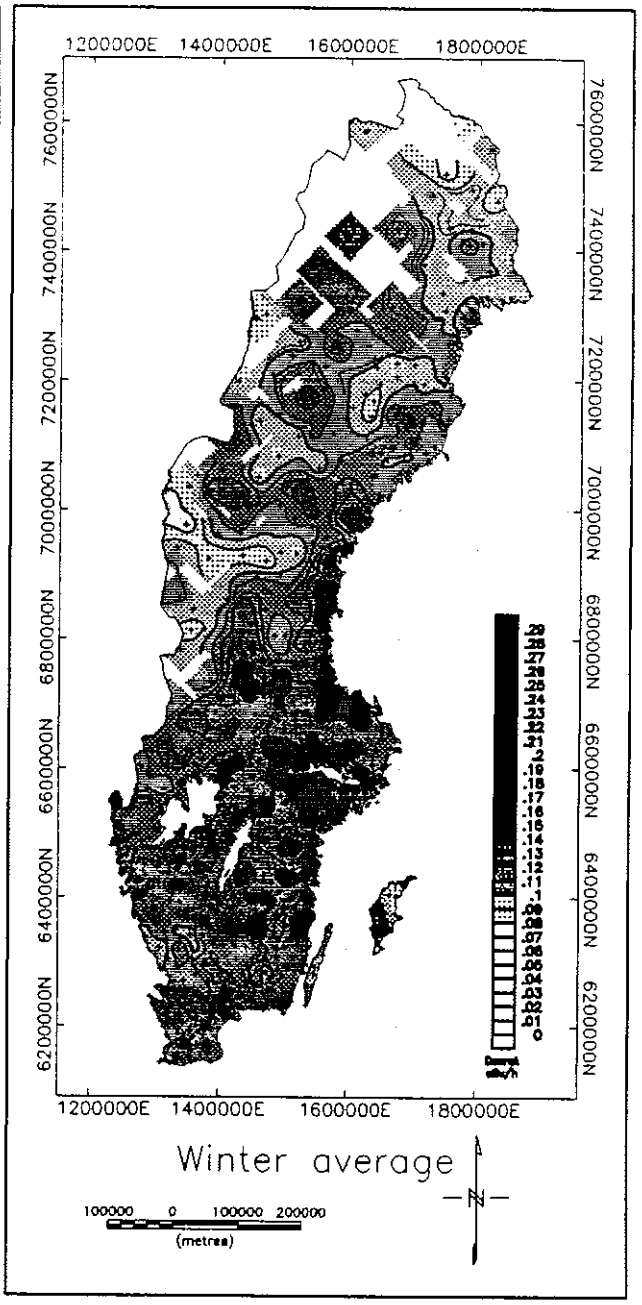

Fig 2. Average dose rates measured at the reference points during the winter period. (Jan-92, Mar-93, Dec-94 Feb-96)

Conclusions

From the about 900 reference points in Sweden where dose rate measurements have been performed with portable instruments during seven years, the seasonal variations can be clearly seen and the decay of the Chernobyl fallout identified for the most affected part of the country. The system provides local authorities with the capability of assessing dose rates in their district. During exercises it has been shown that measurements, reporting and mapping of the dose rate can be made in 6 - 12 hours for parts of Sweden and within 12 - 24 hours for the whole country. The system with portable equipment provides, at relatively low cost, better spatial resolution than an

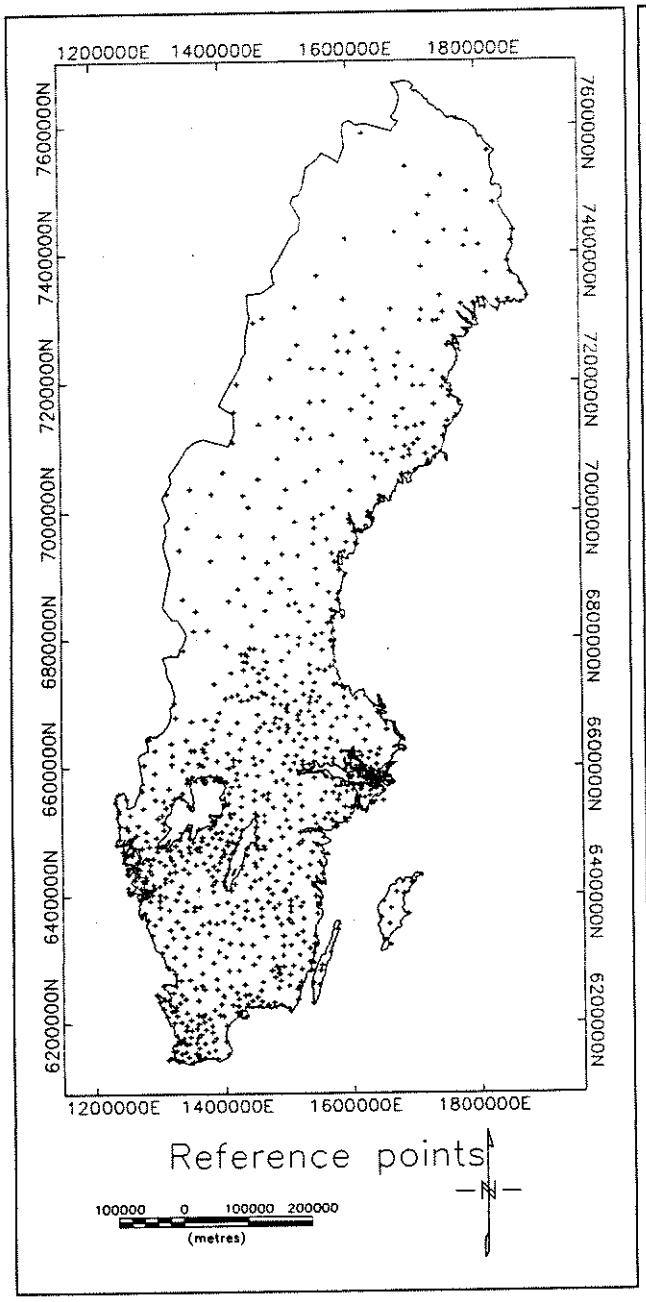

Fig 3. Reference points in Sweden for external dose rate measurements performed by the local districts.

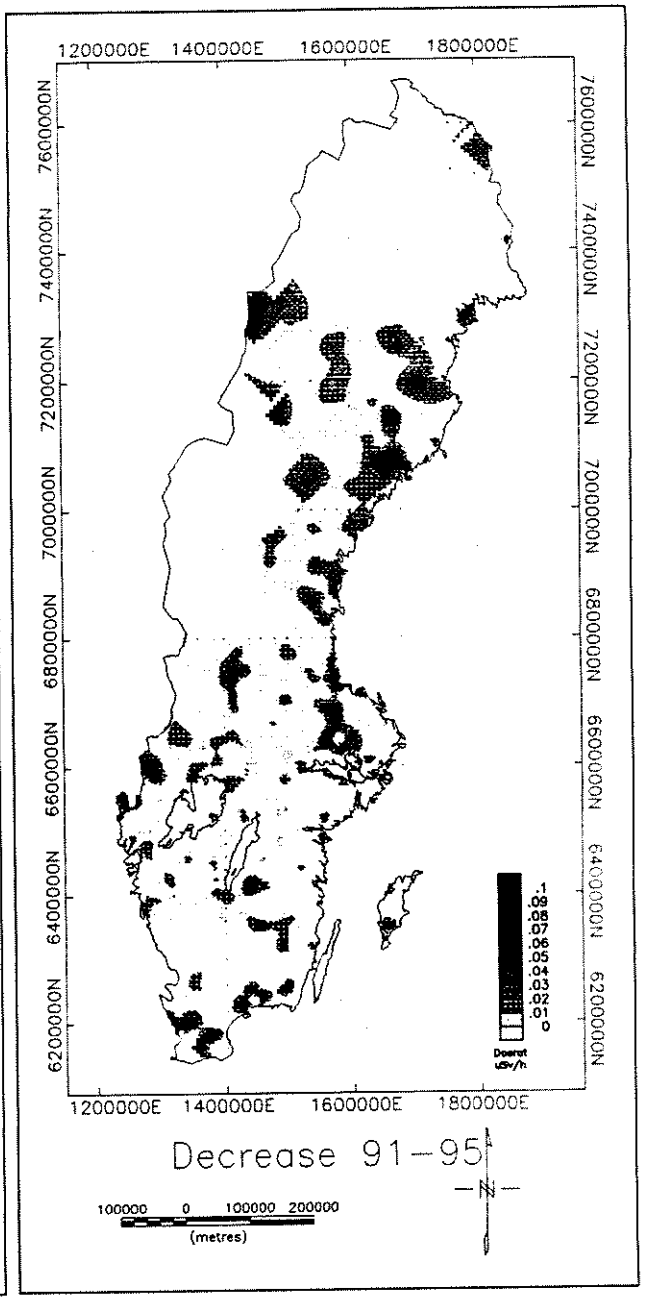

Fig 4. The reduction in external dose rate from June 1991 to July 1995 as calculated from reference measurements.

extensive stationary system would give. The disadvantage is that the portable system is slower than a stationary system and that it is less feasible for warning purpose. It should therefore be seen as a complement to a stationary system.

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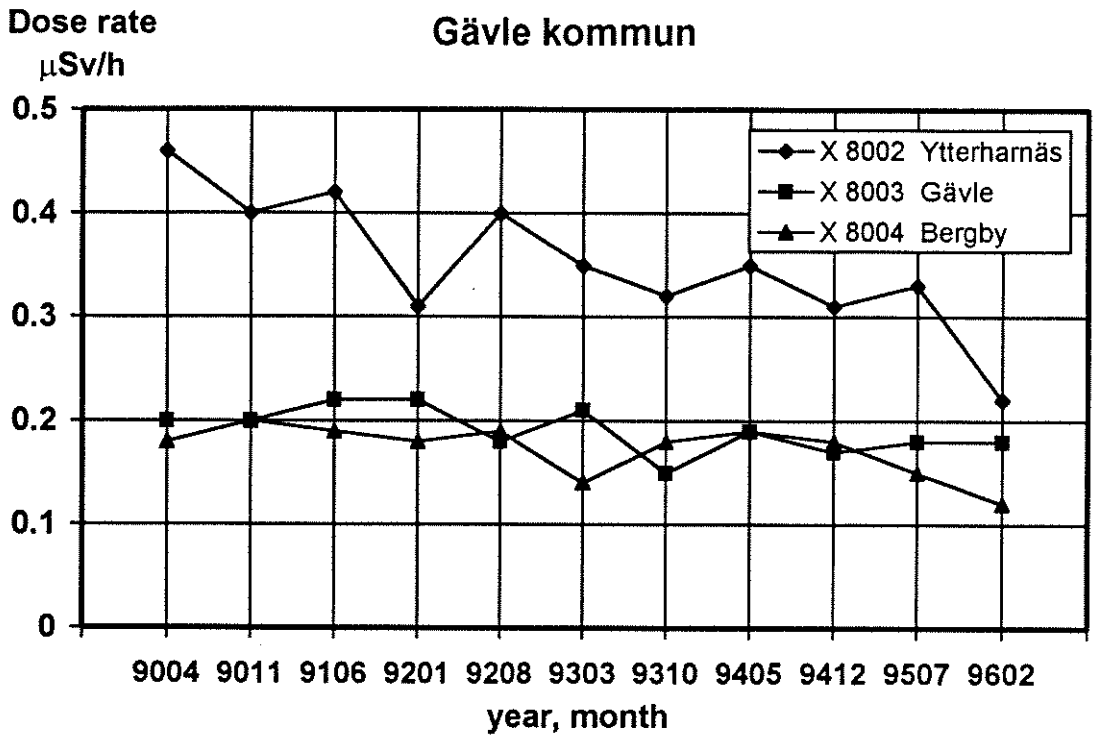

Fig 5. Dose rate measurements from the local district of Gävle at the east coast of middle Sweden. Part of the area was largely affected by the Chernobyl fallout (50 - 100 kBq/m^2 of Cs-137).. The reduction in dose rate for the reference point X-8002 is due mainly to the decay of Cs-134.

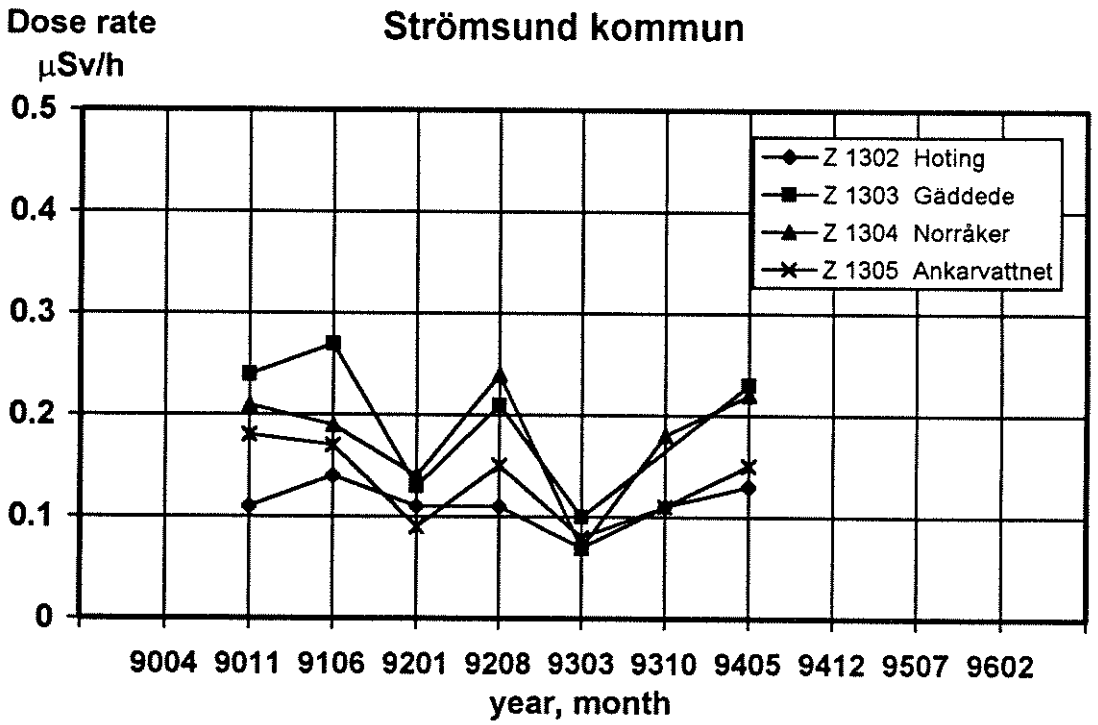

Fig 6. Dose rate measurements from the local district of Strömsund in the north-western part of Sweden. The area was affected by the Chernobyl fallout. The shielding effect of snow cover during the winter season is clearly seen.

Overvåking av utslipp av radioaktive stoffer fra IFE-Kjeller 1948-1994

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Sammendrag I 1948 ble Institutt for atomenergi, nå Institutt for energiteknikk opprettet med formål å drive forskning med sikte på anvendelse av atomkraft. Den første reaktoren (JEEP I) var i drift fra 1951 til 1967 og ble erstattet av den nåværende reaktoren JEEP II i 1969. Reaktoren brukes i dag utelukkende til forskning og isotop-produksjon. I tillegg til reaktoren har instituttet i dag anlegg for isotop-produksjon, undersøkelser av brukt brensel og avfallsbehandling. I hele driftsperioden har det vært små, kontrollerte utslipp av radioaktivitet til atmosfæren og via avløpsvann. Ved utslipp til luft finnes bare ^{41}Ar og ^{131}I i påvisbare mengder, mens følgende nuklider dominerer avløpsvannet: ^{3}H, ^{60}Co, ^{90}Sr, ^{129}I, ^{131}I og ^{137}Cs. Utslippene har i hele perioden ligget langt under gjeldende utslippstillatelse. For utslipp til luft utgjør ^{41}Ar og ^{131}I henholdsvis ca 13 % og 0,3 % av tillatt utslippsmengde. Utslipp av radioaktive stoffer i avløpsvann til den nærliggende Nitelva ligger i gjennomsnitt på ca. 10 % av tillatt utslippsmengde.

Utslippsveier

Det er to utslippsveier fra instituttets anlegg til omgivelsene. Den ene går gjennom ventilasjonsanleggene til atmosfæren, og den andre til Nitelva gjennom instituttets spesielle utslippsledning, NALFA-ledningen (Ny (1967) avfallsledning for lavaktivt flytende avfall).

Atmosfæren

Alt utslipp til atmosfæren er under kontinuerlig overvåking. Det er bare ^{41}Ar og ^{131}I som kan måles, andre nuklider forekommer ikke i påviselige mengder. ^{41}Ar utgjør største delen av utslippet og kommer fra bestrålt luft i reaktoren JEEP II. I reaktorens skorstein er det montert detektor som måler ^{41}Ar-utslippet. Utslippet er proporsjonalt med antall driftstimer for reaktoren. ^{131}I kommer dels fra isotop-laboratoriene og dels fra reaktoren hvor tellur-kontaminering av bestrålingskanaler og lekkasje fra produksjonsboksene av og til kan forekomme. Ved alle bygg eller anlegg hvor det håndteres radioaktive stoffer, filtreres ventilasjonsluften i absolutt-filtre og/eller kullfiltre. Etter filtreringen blir luften kontrollmålt ved at en representativ del tas ut i et "by pass" og suges gjennom et glassfiberfilter og/eller et impregnert kullfilter. Filtrene blir kontinuerlig målt for oppsamlet beta/gamma-aktivitet og resultatene automatisk registrert. Ved noen bygg og anlegg brukes spesielt utviklede målepatroner ilagt aktivt kull for oppsamling av I-nuklider (ref 2). Bygg hvor slike filterpatroner er i drift er Isotoplaboratoriene, Metallurgisk laboratorium og reaktoren JEEP II. Gjennomsnittsutslipp av ^{41}Ar 1984 til 1994 er 12,9 % av sin grenseverdi. I-utslipp til atmosfæren er meget begrenset og gjennomsnittsutslipp 1984 til 1994 er 0,33 % av sin grenseverdi. Utslippene av ^{41}Ar og ^{131}I til atmosfæren 1984-1994 er vist i tabell 1.

Elven som resipient for utslipp

I samsvar med instituttets utslippstillatelse blir lavaktivt avløpsvann sluppet fra avfalls-anlegget til Nitelva. Avfallsvannet kontrolleres først av instituttets Miljø og strålevern-avdeling, hvoretter godkjennelse for utslipp blir gitt til seksjon Radavfall. Samlet utslipp til Nitelva 1984 til1994 er gjennomsnittlig 9,6 % av grenseverdiene. Resultatene er gitt i tabell 3.

Historisk tilbakeblikk; IFEs utslippstillatelser

Institutt for atomenergi (IFA), nå Institutt for energiteknikk (IFE) ble opprettet i 1948 med formål å drive forskning med sikte på anvendelse av atomkraft. Behovet for utslipp av radioaktive stoffer til omgivelsene ble tidlig påtrengende for instituttet. Allerede i 1954 startet en omfattende prosess med respektive myndigheter. Hvordan holdningen til forurensing av

naturen var i datidens kontekst, kan illustreres i sitat fra IFAs brev til Norges vassdrags- og energiverk, hydrologisk avdeling:

"Jeg tillater meg hermed å spørre om De kan gi oss noen opplysninger om vannføring i Nitelva. Vi er nemlig interessert i å se hvilken fortynning vi oppnår dersom vi sender vårt radioaktive avfall ut i Nitelva."

Her er ingen reservasjoner om forurensing av elva eller tanker om eventuell tillatelse til utslipp, og det var dermed starten på en prosess hvor Statens radiologiske-fysiske laboratorium (i dag Statens strålevern) var en av hovedaktørene. Institusjonen godtok i realiteten utslipp av radioaktive stoffer til Nitelva i brev til IFA 1954:

" Slik som De fremstiller saken, ser vi ingen betenkeligheter i at De slipper nevnte mengder av radioaktivt materiale i Nitelven. Vi ber om å få beskjed når forholdene blir forandret",

Dette brakte Norges vassdrags- og energiverk inn i diskusjonen. Deres mening om IFAs utslipp ble klarlagt i brev til IFA 1956:

" Hovedstyret er underhånden gjort oppmerksom på at instituttet fra sin atomreaktor på Kjeller har sluppet radioaktive stoffer således at disse kommer ut i vassdrag. Man ber meddelt om det er så eller om hvordan slike stoffer ellers kan fjernes".

Andre statlige og kommunale institusjoner som Det Kongelige Departement for Industri og Håndverk, Helsedirektoratet, helserådet i Lillestrøm og helserådet i Skedsmo ville også ha saken belyst. Etter lang tid med diskusjoner og brevskrivning førte prosessen frem til IFAs første søknad om tillatelse til utslipp. Søknaden ble innvilget i 1963 av det Kongelige Departement for Industri og Håndverk, og instituttet fikk:

" Tillatelse til utslipp til Nitelva av inntil 2 curie flytende radioaktivt avfall pr 30 dager."

Det heftet omfattende betingelser om analyse av utslipp, oppbevaring av kontraprøve, minimum vannføring i elva og journalføring.

Någjeldende utslippstillatelse

Instituttets tillatelse av 1963 virket til 1984 da den ble fornyet. Den nye tillatelsen er formulert i en ny tankesammenheng. Basert på fastlagte grensesettende maksimalverdier for effektiv dose til utsatt befolkningsgruppe, skulle nå avledede grenseverdier for nuklider beregnes.

Den nye tillatelsen formaliserte også instituttes utslipp til atmosfæren. Någjeldende tillatelse fra 1992 er en kopi av tillatelsen av 1984, og den er knyttet til følgende betingelser:

"Utslippet skal for individer i den mest utsatte befolkningsgruppe rundt anleggene holdes under de mengder, som kan avledes av fastsatte grenseverdier for effektiv dose som følge av utslippene".

Fra utslipp til atmosfæren totalt	*100 µSv/år*
Fra utslipp til atmosfæren av I-nuklider	*10 µSv/år*
Fra utslipp til vann totalt	*1 µSv/år*
Kollektivdose (befolkningsdose)	*0,5 manSv/år*

Grenseverdiene ved lignende forskningsinstitusjoner i øvrige skandinaviske land har vært retningsgivende for grenseverdiene fastlagt ved IFE. Eksempler på utslippsgrenser:

Til atmosfæren:

^{41}Ar	280 TBq/år (grensesettende dose 100 µSv/år)
^{131}I	0,2 TBq/år (grensesettende dose 10 µSv/år)

Til Nitelva: Grensesettende dose 1µSv/år

^{3}H	4300 GBq/år
^{60}Co	30 GBq/år
^{131}I	3,4 GBq/år
^{137}Cs	4,3 GBq/år
^{134}Cs	4,3 GBq/år

Beregningene av utslippsgrenser er gjort innom rammer og forutsetninger som antas å ville gi eksponeringer langt utover de som i virkeligheten kan forekomme. Beregningene er basert på IAEAs og ICRPs modeller. Detaljert beskrivelse finnes i ref 1.

I utslipp til atmosfæren påvises som nevnt kontinuerlig bare to nuklider, og utslipp av betydning av andre nuklider har ikke forekommet. Ved utslipp av radioaktive stoffer til Nitelva er nuklidene derimot mer mangeartede, og fra tid til annen påvises nye. For raskt å kunne fastsette grenseverdien for nye nuklider, er det derfor utledet følgende formel :

$$Q_{maks} = 4{,}3\ 10^{-3}\ (ALI_{20})\ MBq/år$$

ALI_{20} (Annual Limit of Intake) er det årlige inntak av en nuklide som gir en effektiv dose på 20 mSv. Formelen baserer seg på den fastsatt maksimale strålebelastning 1 µSv/år til utsatt gruppe personer. Det forutsettes 1 l inntak av Nitelvvann pr dag. Utledning av formel er gitt nedenfor.

Effektiv dose som følge av et års inntak av L liter vann pr døgn:

$$H = 20 \frac{C_w\ L\ 365}{(ALI_{20})}\ mSv$$

C_w = gjennomsnittlig årskonsentrasjon av nukliden i Bq/l
(ALI_{20}) = inntak i Bq som gir 20 mSv

$C_w$for Nitelva = $3{,}2\ 10^{-11}$ Q, forutsatt fullstendig blanding og gjennomsnittlig vannføring lik 1 m^3/sek
Q = utslippsmengde pr år (Bq)

Settes inn for C_w i formelen for H blir:

$$H = 20 \frac{3{,}210^{-11}\ Q\ L\ 365}{(ALI_{20})}\ mSv$$

herav fås:

$$Q = \frac{H(ALI_{20})}{20\ 3{,}210^{-11}\ L\ 365}\ Bq$$

H= 1µSv og L= 1 l/døgn og omgjøres til MBq, blir: $Q_{maks} = 4{,}3\ 10^{-3}\ (ALI_{20})$ *MBq/år*

Dette gir en letthåndterlig formel for beregning av grenseverdier for de enkelte isotoper, idet ALI_{20} finnes i tabell i ICRP publikasjon nr 61(ref 3).

Betingelse ved samtidige utslipp av flere nuklider (samme år):

$$\text{Sum}_{(i)} \frac{Q(i)}{Q(i)maks} < 1$$

Utover dette skal instituttet alltid begrense sine utslipp mest mulig ut fra en balansert vurdering av kostnader forbundet med å redusere utslippene og den strålerisiko utslippet kan medføre.

Utslipp til atmosfæren

Tabell 1 Utslipp til atmosfæren i % av grenseverdi, 1984 til 1994.

	^{41}Ar utslipp TBq	% av grenseverdi	^{131}I utslipp TBq	% av grenseverdi
1984	39,2	14	$1{,}9.10^{-4}$	0,095
1985	36,3	13	$3{,}7.10^{-4}$	0,19
1986	35,7	12,8	$9{,}8.10^{-4}$	0,49
1987	33,3	11,9	16.10^{-4}	0,8
1988	38,6	13,8	$3{,}.910^{-4}$	0,2
1989	38	13,6	$3{,}5.10^{-4}$	0,18
1990	38,6	13,8	$3{,}4.10^{-4}$	0,17
1991	33,3	11,9	$4{,}8.10^{-4}$	0,24
1992	37,4	13,4	$8{,}4.10^{-4}$	0,42
1993	34,6	12,4	12.10^{-4}	0,6
1994	31,7	11,3	$4{,}3.10^{-4}$	0,22
Gjennomsnitt	36	12,9	$6{,}5.10^{-4}$	0,33

Kontrollprøver for utslipp til atmosfæren

Prøver som tas for kontroll av instituttets utslipp til atmosfæren er friskluft, nedbør, gressprøver, jordbruksprøver og melk. Resultater av ^{131}I-målinger av utendørs friskluft fra 1984 til 1994 er gitt i tabell 2.

For kontroll av utendørsluft er det satt opp to målestasjoner med spesielle målepatroner for oppsamling av I-nuklide, en i inntaksluften til reaktoren JEEP II og en på taket av Miljø- og strålevernbygget. Filtrene kontrolleres rutinemessig.

Tabell 2 Årsgjennomsnitt av ^{131}I i utendørsluft ved IFE 1984 til 1994. mBq/m^3.

År	Målestasjon LU 1 VE-bygg	Målestasjon LU 2 JEEP II
1984	10	67
1985	6	38
1986	540	625
1987	8	31
1988	7	37
1989	9	74
1990	12	120
1991	16	182
1992	24	295
1993	2	499
1994	7	350

De markert høyere verdier målt ved målestasjon LU 2 JEEP II, er forårsaket av målepunktets nære beliggenhet til utslippstedet. Høye 1986-verdier skyldes Tsjernobyl-ulykken.

Utslipp til Nitelva

I tabell 3 nedenfor er utslippsmengdene av de ulike nuklider pr år summert og beregnet i % av den totale mengde grenseverdiene for utslipp tilsier.

Tabell 3 Samlet utslipp til Nitelva 1984 til 1994.

År	Utsluppete nuklider	% av utslippsgrense
1984	^{131}I, ^{60}Co, ^{3}H, ^{35}S, ^{90}Sr, $^{239,240}Pu$	4
1985	^{131}I, ^{60}Co, ^{3}H, ^{35}S, ^{90}Sr, $^{239,240}Pu$	1,7
1986	^{131}I, ^{137}Cs, ^{60}Co, ^{3}H, ^{35}S, ^{90}Sr, $^{239,240}Pu$	17
1987	^{131}I, ^{137}Cs, ^{60}Co, ^{3}H, ^{35}S, ^{90}Sr, $^{239,240}Pu$	14
1988	^{131}I, ^{137}Cs, ^{60}Co, ^{3}H, ^{35}S, ^{90}Sr, $^{239,240}Pu$	6
1989	^{131}I, ^{137}Cs, ^{60}Co, ^{3}H, ^{35}S, ^{90}Sr, $^{239,240}Pu$	7,6
1990	^{131}I, ^{137}Cs, ^{134}Cs, ^{60}Co, ^{3}H, ^{35}S, ^{90}Sr, $^{239,240}Pu$	13
1991	^{131}I, ^{137}Cs, ^{134}Cs, ^{56}Co, ^{60}Co, ^{3}H, ^{35}S, ^{90}Sr, $^{239,240}Pu$	12
1992	^{131}I, ^{125}I, ^{137}Cs, ^{134}Cs, ^{60}Co, ^{153}Gd, ^{65}Zn, ^{35}S, ^{90}Sr, $^{239,240}Pu$, ^{3}H	15
1993	^{131}I, ^{125}I, ^{134}Cs, ^{137}Cs, ^{60}Co, ^{153}Gd, ^{59}Fe, ^{65}Zn, ^{35}S, ^{90}Sr, $^{239,240}Pu$, ^{3}H	3,7
1994	^{131}I, ^{125}I, ^{134}Cs, ^{137}Cs, ^{60}Co, ^{65}Zn, ^{35}S, $^{239,240}Pu$, ^{3}H	11,8
	Gjennomsnitt	**9,6**

Kontrollprøver for utslipp av lavaktivt avfallsvann til Nitelva

For kontroll av instituttets utslipp av radioaktive stoffer til Nitelva innsamles prøver av vann, sedimenter, fisk og vannplanter. Resultater av målingene vises i tabellene nedenfor:

Tabell 4 ^{137}Cs, ^{90}Sr, $^{239,240}Pu$ i vann 1984-1994, 87 enkeltprøver. Gjennomsnitt, høyeste og laveste verdier angitt i mBq/l.

	^{137}Cs*			^{90}Sr			$^{239,240}Pu$		
Prøvested	gj.snitt	høyeste	laveste	gj.snitt	høyeste	laveste	gj.snitt	høyeste	laveste
Kjellerholen	570	730	25	24	37	8,3	0,78	3,4	0,11
L.strøm bru	1300	2400	530	29	69	11	0,74	2,6	0,3
Ræl.bru	730	1700	150	21	34	3	0,84	2,2	0,12

*61 enkeltprøver<DL. Gj.snitt er av antall registrerte verdier >DL. DL=laveste målbare konsentrasjon.

Tabell 5 ^{137}Cs, ^{90}Sr og $^{239,240}Pu$ i sedimentprøver 1984-1994, 55 enkeltprøver. Gjennomsnitt, høyeste og laveste verdier angitt i Bq/kg glødet slam.

	^{137}Cs			^{90}Sr			$^{239,240}Pu$		
Prøvested	gj.snitt	høyeste	laveste	gj.snitt	høyeste	laveste	gj.snitt	høyeste	laveste
Kj.holen	110	170	22	2,1	3,6	1,1	0,5	0,48	0,29
L.strøm bru	110	180	52	2,3	3,6	1,3	11	31	0,6
Ræl.bru	88	150	9	1,8	3,2	1	8,4	20	2,6
Svellet	44	83	21	1,6	3,7	1	1,3	4,7	0,5
Øyeren	16	39	5	0,93	2,1	1,1	0,25	0,8	0,025

Tabell 6 *^{137}Cs, ^{90}Sr og $^{239,240}Pu$ i vannplanteprøver (potamogeton perfoliatus) 1984-1994, 44 enkeltprøver. Gjennomsnitt, høveste og laveste verdier angitt i Bq/kg våtvekt.*

	^{137}Cs*			^{90}Sr			$^{239,240}Pu$		
Prøvested	gj.snitt	høyeste	laveste	gj.snitt	høyeste	laveste	gj.snitt	høyeste	laveste
Svellet	1,1	2	0,52	2,4	5	1	0,043	0,11	0,018

* Av 44stk enkeltprøver er 15 <DL

Tabell 7 *^{137}Cs, ^{90}Sr og $^{239,240}Pu$ i fisk 1984-1994, 102 enkeltprøver. Gjennomsnitt, høyeste og laveste verdier angitt i Bq/kg våtvekt.*

	^{137}Cs			^{90}Sr			$^{239,240}Pu$		
Prøvsted	gj.snitt	høyeste	laveste	gj.snitt	høyste	laveste	gj.snitt	høyeste	laveste
Svellet/Øyern	9,5	22	0,6	5,2	7,2	3,8	0,024	0,079	0,010

Konsekvenser av utslippene

Ved beregning av utslippsgrensene er det gjort visse antagelser og forutsetninger om forhold som kan føre til maksimal strålebelastning til en eventuell utsatt befolkningsgruppe (ref 1). Dosebelastningen vil derfor være, hvis disse forhold er tilstede, direkte relaterbar til mengdene av utslipp. Mengdene av utslipp fra 1984 til 1994 er gitt i tabell 1 og 3.

I det etterfølgende er det gjort beregninger for hvilke maksimale doser til eventuell utsatt befolknings-gruppe IFEs utslipp ville medført dersom forutsetningene gitt i beregnings-grunnlaget for grensesettende dose hadde vært tilstede.

Utslipp til atmosfæren

Gjennomsnittsverdiene for ^{41}Ar og ^{131}I i % av grenseverdiene som er avledet av den grensesettende effektiv dose 100 µSv/år, er henholdsvis 12,9 og 0,33 % (tabell 1) for årene 1984 til 1994. Total dosebelastning til eventuell utsatt befolkningsgruppe vil være:

$$^{41}Ar:\ 100\ \mu Sv \times 0{,}129 \times 11 = 142\ \mu Sv$$

$$^{131}I:\ 10\ \mu Sv \times 0{,}033 \times 11 = 3{,}6\ \mu Sv$$

Utslipp til Nitelva

Gjennomsnittsutslippet i % av grenseverdiene som er avledet av den grensesettende dose 1 µSv/år, er 9,6 % (tabell 3) for årene 1984 til 1994. Total dosebelastning til eventuell utsatt befolkningsgruppe vil være:

$$1\ \mu Sv \times 0{,}096 \times 11 = 1{,}06\ \mu Sv$$

Konklusjon

På bakgrunn av dette vil doserelaterte konsekvenser umulig kunne registreres, og de kan dermed sies å være neglisjerbare.

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Decontamination in a Russian Settlement

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Abstract

Decontamination was carried out in and around three houses in Novo Bobovichi in Russia during the autumn of 1995. It was demonstrated that significant reductions in the dose rate both indoor (DRF=0.34) and outdoor (DRF=0.20) can be achieved when a careful cleaning is undertaken. The decontamination work carried out and the results obtained are described. The roofs of the houses were swept and cleaned by special roof cleaning equipment. The top soil around the houses was removed by hand while carefully monitoring the ground for residual contamination. By monitoring the decline in the dose rate during the different stages of the work the dose reducing effect of each action has been estimated.

A method for ^{90}Sr-determination in environmental samples shortly after a nuclear accident

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Abstract- After a recent nuclear accident determinations of ^{90}Sr in environmental samples is difficult due to high activities of interfering radionuclides. The possibility of an earlier developed method to conduct proper separations under such conditions has been investigated. The results show that, if the activities of the interfering nuclides are not extremely high, activities down to approximately 65 mBq ^{90}Sr/g sample ash can be detected in about 3 days after sampling.

Introduction

In case of a release of radioactivity from a nuclear accident, it is important to have fast and reliable methods for the analyses of contaminated environmental samples. ^{90}Sr is usually considered one of the most hazardous fission products due to its high fission yield and its long physical and biological half-life. In the analysis of ^{90}Sr, a radiochemical procedure is required as both ^{90}Sr and, in practice, its daughter ^{90}Y are pure β-emitters. The classical procedure [1] is laborious and time-consuming and therefore, a simple and fast method has been developed and applicated to fallout from the Chernobyl accident [2]. The main steps in this method are: i) Drying and ashing, ii) Leaching, iii) Solvent extraction of ^{90}Y, iiii) Cherenkov counting of ^{90}Y in the organic phase. In determinations of "old" fallouts, ^{90}Y has to be separated only from a few interfering radionuclides while, after a recent accident, there are a number of interferences. In order to plan the separation scheme, one must know their approximate activity ratios relative ^{90}Sr. It has been shown [3,4] that a considerable fraction of the ^{90}Sr in the fallout from the Chernobyl accident was incorporated in fuel particles, at least less than 60 km from the reactor. The characteristic sizes were 20-400 μm, and the estimated fuel burn-up values indicated that the particles originated from fuel of all ages, with a peak for very young fuel. If an environmental sample contains one of these particles, and if they are at least partly soluble, the activity ratios in the sample can not be estimated from the average age of the fuel. It is therefore interesting to calculate approximate activity ratios as a function of fuel irradiation time and propose a separation scheme, aiming at isolating yttrium. In an emergency case, it is also interesting to know the approximate time required to detect a certain activity of ^{90}Sr in an environmental sample. The aim of is work is to investigate the applicability of the earlier developed method to the case of a recent accident.

Calculations

Activity ratios

All fission products with sufficient decay energies to produce Cherenkov radiation in the organic phase (n=1.44) have been considered. The activity ratios were calculated considering ^{235}U-fission, (n,γ)-production/consumption and radioactive decay, at an

average neutron flux of 10^{13} n cm^{-2} s^{-1} (10^{14} for activation products). Fission yields, thermal neutron cross-sections and half-lives are taken from Ref. 5. Because fission yields for thermal neutrons are used, and because fission of ^{239}Pu is neglected, the activity ratios for nuclides with mass numbers of about 105-120 might be up to 10 times higher.

Separation (Solvent Extraction)

The extraction is made with 1 M HDEHP (di(2-ethyl-hexyl)-phosphoric acid) dissolved in Nysolvin 75A (an aliphatic kerosene). The distribution factor (D) is c_{org}/c_{aq} and the percent extracted (%E) is calculated according to: %E = 100*D*θ/(1+D*θ), where θ = phase volume ratio (v_{org}/v_{aq}).

Radioactivity measurements

The Cherenkov light-pulses in the organic phase are registred using a liquid scintillation counter. The decay curve is followed by counting each sample a number of short time-periods (usually 18 min each). The count rate of ^{90}Y at the time for $^{90}Y/^{90}Sr$-separation, and its uncertainty, are obtained by minimizing the sum of squares of the data points to the principal function:

$$a_0 + a_1 \cdot x_i = y_i \pm \sigma_i,$$

where a_0 = sample background count rate, a_1 = count rate of ^{90}Y at the time of $^{90}Y/^{90}Sr$-separation, $x_i = \exp(-\lambda(^{90}Y)*t_i)$ (where t_i = time since separation), y_i = measured count rate at t_i and σ_i = the uncertainty of y_i used (squared) in the data point weighing. The function can be used if no short-lived interferences are present. The expression for calculating a_1 (the inital count rate of ^{90}Y) is:

$$a_1 = \frac{1}{D}\left[\left(\sum_i \frac{1}{\sigma_i^2}\right)\cdot\left(\sum_i \frac{x_i y_i}{\sigma_i^2}\right) - \left(\sum_i \frac{x_i}{\sigma_i^2}\right)\cdot\left(\sum_i \frac{y_i}{\sigma_i^2}\right)\right]$$

where

$$D = \left(\sum_i \frac{1}{\sigma_i^2}\right)\cdot\left(\sum_i \frac{x_i^2}{\sigma_i^2}\right) - \left(\sum_i \frac{x_i}{\sigma_i^2}\right)^2$$

The uncertainty of a_1 (if the errors are normally distributed) is calculated according to:

$$\sigma_{a1} = \sqrt{\frac{1}{D}\sum_i \frac{1}{\sigma_i^2}}$$

If the time of each measurement is *e.g.* 18 min, and if one uses "counts per 18 min" as count rate, the expected uncertainty in each data point becomes the square root of the total number of counts in each data point, *i.e.* $\sigma_i = (y_i)^{1/2}$. (y_i is the total counts from

sample background and ^{90}Y during 18 min.) By substituting σ_i^2 with y_i in the expressions for a_1 and its uncertainty, an expression for the expected relative uncertainty (σ_{a1}/a_1) in the initial count rate of ^{90}Y (at the time of $^{90}Y/^{90}Sr$-separation) can be formed. The expected relative uncertainty is thus a function of sample background count rate, initial ^{90}Y count rate and the characteristics of the data points (number, distribution in time). If, for instance, the sample background and the initial count rate of ^{90}Y can be estimated, and the time between each data point is fixed (*e.g.* owing to the number of samples), it is now possible to calculate the expected relative uncertainty of the initial ^{90}Y count rate as a function of the number of the data points. Hence, the time required to reach below a specified uncertainty can be calculated.

Results and Discussion

Activity ratios

The activity ratios relative ^{90}Sr are very high for young fuel, as can be seen from fig.1. The shape of the curves are typical and reflect the half-lives of the fission products. No chemical aspects are considered, *e.g.* gases like Kr will not accumulate in the fuel.

If the fuel has been irradiated only for a couple of months, a separation of ^{90}Y is difficult. Out of an infinite number of possibilities, it was decided to use the activity ratios after 6 months of irradiation and 3 days of decay, for the separation scheme. A real case might be worse, but it is probably easier as the average fuel age is normally about 2 years.

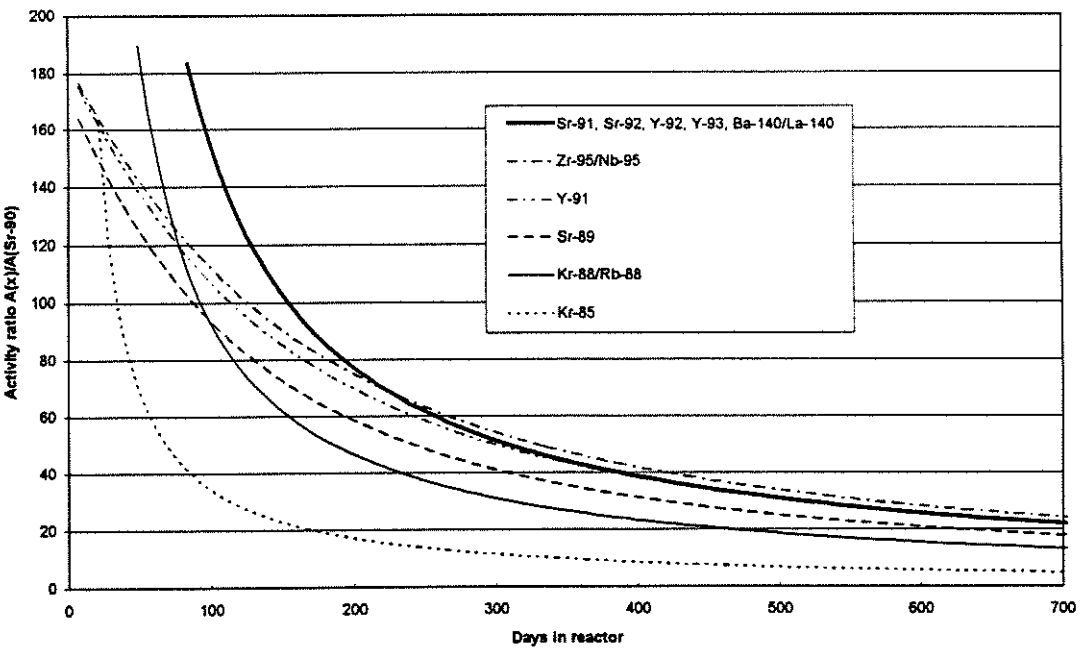

Fig.1 Activity ratios (rel. ^{90}Sr) of some fission products as a function of fuel irradiation time.

Separation

Table 1.
Relative activity ratios before and after the separation process.

Initial activity ratios (rel. Sr-90) after 6 months of fuel irradiation and 3 days of decay.
Separation process: 2 extr. at 0.3 M nitric acid, 24 h ingrowth, 1 extr. and 1 strip at 0.8 M nitric acid

Initial activity ratios			D-values (1 M HDEHP)			%E per extr.		Activity ratios in the stripped organic phase	
						θ = 1	θ = 0.25		
Fission product (x)	t½	A_0 (x)/ A_0 (Sr-90)	0.3 M HNO_3	0.8 M HNO_3	Ref.	0.3 M HNO_3	0.8 M HNO_3	A(x)/ A_0 (Sr-90)	A(x)/ A(Y-90)
Sr-90	28.5 a	1	0.001	0.0002	6	0.1	0.005	< 0.01	< 0.01
Y-90	64.1 h	1	800	162	6	99.9	97.6	0.22	1
As-77	38.8 h	0.04	?	?				low	low
Sr-89	50.5 d	60	0.001	0.0002	6	0.1	0.005	< 0.01	< 0.01
Sr-91	9.5 h	0.5	0.001	0.0002	6	0.1	0.005	< 0.01	< 0.01
Y-91	58.5 d	73	800	162	6	99.9	97.6	< 0.01	< 0.01
Y-93	10.1 h	0.6	800	162	6	99.9	97.6	< 0.01	< 0.01
Zr-95	64.0 d	78	230	700	7	99.6	99.4	< 0.01	< 0.01
Nb-95	35.0 d	77	2.4	1.1	6	71	22	0.30	1.4
Zr-97	16.8 h	4	230	700	7	99.6	99.4	< 0.01	< 0.01
Nb-97	74 m	4	2.4	1.1	6	71	22	< 0.01	< 0.01
Mo-99	66.0 h	40	200	160	7	99.5	97.6	< 0.01	< 0.01
Ru-103	39.4 d	40	0.009	0.002	6	0.9	0.05	< 0.01	< 0.01
Rh-105	35.5 h	5	?	0.001	7	?	0.025	< 0.01	< 0.01
Ru-/Rh-106	368 d	2	0.009	0.002	6	0.9	0.05	< 0.01	< 0.01
Pd-109	13.4 h	0.01	0.04	0.007	7	3.8	0.2	< 0.01	< 0.01
Ag-111	7.4 d	0.2	0.01	0.004	7	1.0	0.1	< 0.01	< 0.01
Cd-115	53.4 h	0.06	?	0.001	7	?	0.025	< 0.01	< 0.01
Sb-125	2.77 a	0.05	0.01	0.01	7	1.0	0.25	< 0.01	< 0.01
Sb-127	3.85 d	1	0.01	0.01	7	1.0	0.25	< 0.01	< 0.01
Te-127	9.4 h	1.2	0.01	0.01	7	1.0	0.25	< 0.01	< 0.01
Sb-128	9.0 h	0.02	0.01	0.01	7	1.0	0.25	< 0.01	< 0.01
I-131	8.0 d	30	?	?				?	?
I-133	20.8 h	9	?	?				?	?
I-135	6.6 h	0.05	?	?				?	?
Cs-137	30.2 a	1	0.01	0.01	7	1.0	0.25	< 0.01	< 0.01
Ba-140	12.75 d	75	0.01	0.01	7	1.0	0.25	< 0.01	< 0.01
La-140	40.3 h	75	0.55	0.048	6	35	1.2	< 0.01	0.03
Ce-141	32.5 d	76	2	0.11	6	67	2.7	< 0.01	0.03
Ce-143	33.0 h	19	2	0.11	6	67	2.7	< 0.01	< 0.01
Pr-143	13.6 d	80	2.7	0.14	7	73	3.4	< 0.01	0.03
Ce-/Pr-144	285 d	30	2	0.11	6	67	2.7	< 0.01	0.01
Pr-145	5.98 h	0.01	2.7	0.14	7	73	3.4	< 0.01	< 0.01
Nd-147	11d	27	2.5	0.19	6	71	4.5	< 0.01	0.02
Pm-149	53.1h	6	10	0.2	8	71	4.1	< 0.01	< 0.01
Pm-151	28 h	1	10	0.2	8	71	4.1	< 0.01	< 0.01
Sm-153	46.8 h	0.8	5.8	0.29	7	85	6.8	< 0.01	< 0.01
Eu-156	15.2 d	0.6	34	1.4	7	97	26	< 0.01	< 0.01
Cs-134*	2.06 a	0.9	0.01	0.01	7	1.0	0.25	< 0.01	< 0.01
Eu-154*	8.8a	0.05	34	1.4	7	97	26	< 0.01	< 0.01

*Activation products

The separation process chosen consists of three steps, i) Two extractions at 0.3 M nitric acid (θ=1), ii) Ingrowth of ^{90}Y (among others) for 24 hours, iii) One extraction

and one strip of the organic phase at 0.8 M nitric acid (θ=0.25). The organic phase, containing ^{90}Y corresponding to 22% of the initial ^{90}Sr activity, can thereafter be measured. The results are shown in table 1.

The separation of ^{90}Y is good, as seen from the last column. The activity of ^{95}Nb is quite high but it decays with a low energy and has consequently a low detection efficiency. Furthermore, its relatively long half-life makes it possible to include in the sample background count rate when calculating the activity of ^{90}Y.

It seems to be possible to purify ^{90}Y reasonably well shortly after a nuclear accident, however, there are a number of factors that can affect the actual activity ratios in the environmental sample. Firstly, to what degree the individual radionuclides are released from the reactor, and from fuel particles, depends on their chemistry. Secondly, activation products from the reactor material has not been considered. Thirdly, the separation might be worse than calculated due to incomplete phase separations. These factors are all hard to predict.

Time of analysis

The total measuring time required to reach below a certain error in the initial ^{90}Y count rate is shown in figure 2. The time between the data points is 3 hours, corresponding to *e.g.* 10 samples, 18 minutes measurement and full use of one detector. Thus, with a background of 25 cpm and an initial R(^{90}Y) of 2 cpm, the uncertainty is 100% after 48 hours and 50% after about 90 hours. The higher the ^{90}Y activity and the lower the background, the smaller the error. Experiments have shown that the errors are usually slightly higher than theoretical values due to interferences.

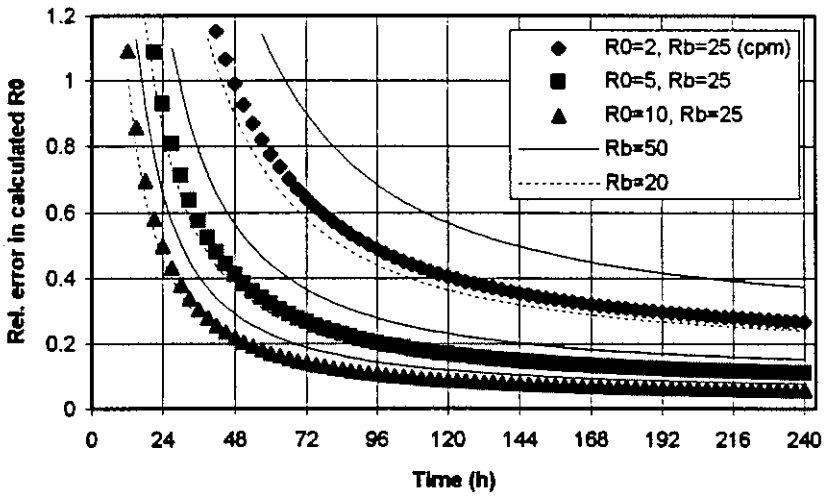

Fig 2. Expected relative error in the initial count rate of ^{90}Y (R_0) as a function of time at different R_0 and R_b (sample background count rate). The time between each data point is 3 h.

One might consider the detection limit to be reached when the count rate can be determined within an error of less than 100%. As an example, with a background of 25

cpm, an initial R(^{90}Y) of about 5 cpm can be detected after 24 hours, as seen in figure 2. With a detection efficiency of 0.65, this corresponds to an ^{90}Y activity of about 130 mBq. Using the separation scheme proposed above, the ingrowth of ^{90}Y is 22% of the ^{90}Sr activity. With a recovery of 90% and a sample size of 10 g sample ash, this corresponds to a detected ^{90}Sr activity of approximately 65 mBq/g ash (in each of 10 samples). The total time required to reach this limit, excluding drying and ashing, is about 50 hours (2h leaching, 24h ingrowth, 24h measurement). Drying and ashing might require another 24 hours, depending on the sample material. Irrespective of initial ^{90}Y and sample background count rates, an extended time of analysis lowers the errors in the determined activities and the detection limit.

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Mobile equipment for preparedness purposes.

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Abstract

Sweden is a fairly large country. Therefore, there is a relatively great need in our country of mobile equipment for measurements of fallout in case of nuclear accidents. For this purpose and on assignment from the Swedish Radiation Protection Institute (SSI), the Radiometry team from the Division of Ionizing Radiation and Fallout, FOA NBC-Defence, Umeå, is part of the National preparedness program for Northern Sweden. Other tasks of the Radiometry team are analyses of radioactive material, development of methods and equipment for preparedness purposes, and simulation of the distribution of deposited radioactive matter after nuclear accidents and nuclear weapons explosions.

Field gamma-ray spectrometry

Two HPGe-detectors, with relative efficiencies of 50% and 36%, respectively, are available for field gamma measurements. The detectors have been efficiency calibrated within the energy interval 240 - 1500 keV and for angle of incidence. The measurement data evaluation is based either on a slice distribution model with homogenous density and activity distribution in one ore more slices, or on a plane surface source distribution model.

Carborne measurements

We have recently mounted one of the two HPGe-detectors inside a four-wheel-drive vehicle for carborne spectrometry measurements. The vehicle also contains an air-sampling equipment, computers powered by 12 or 220 V, mobile phones and a fax. Other facilities are equipment for soil-sampling, dose-rate measurement and position estimation (GPS). To increase independence, the vehicle has both a heating system and a 220 V power-supply driven by gasoline.

Whole body counting

After the Chernobyl accident in 1986, a mobile whole body counter was constructed by FOA NBC-Defence and the Department of Radiophysics at Umeå University. It was built on an assignment from SSI. The whole body counter was placed in a standard ship-container and mounted on a truck trailer to facilitate transports. The whole body counter initially consisted of a lead-shielded geometry and a 5*5" NaI crystal with electronics. Recently, the NaI detector was substituted with the 50% HPGe detector. The detector is calibrated for photon energies between 500 - 1500 keV and for body-masses of 10 - 90 kg. For body-masses below 10 kg (babies), the detector has been calibrated for two specific nuclides (Cs-137 and K-40). The mobile whole body counter laboratory is also equipped with telephone, fax, electric heaters and air conditioning.

Simulation

A novel mathematical simulation method for activities and dose-rates in areas of fallout (MARS) has recently been developed. A microprocessor simulates the distribution of radioactivity based on a single mathematical expression, the geographical position and time. MARS can be used both in handheld and carborne applications. The simulator has been tested on a number of scenarios including the release of radioactive matter from nuclear reactors and the fallout after nuclear weapons explosions. The method is currently introduced in various simulation applications.

11th Meeting of the Nordic Radiation Protection
Society / 7th Nordic Radioecology Seminar
Reykjavík, August 26-30, 1996

THE METEOROLOGICAL INSTITUTE'S CONTRIBUTION TO THE NATIONAL RADIATION SURVEILLANCE PROGRAMME IN FINLAND

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Abstract - The Finnish Meteorological Institute (FMI) participates in the nation-wide radiation monitoring system of Finland with a monitoring network for atmospheric radioactivity and with dispersion forecasts. The main purpose of the monitoring network is the use of natural radionuclides as tracers in air quality and climate studies, but it operates also as a sensitive early detection network. The Institute is also, owing to its 24-hour duty service, the national contact point of the international radiation event notifications.

For operative use the FMI has two dispersion models. YDINO is a short-range dispersion model, which is used in case of emergencies at the Finnish nuclear power plants or at those nearest to the Finnish borders. It is a Gaussian surface trajectory dispersion model, which utilizes routine synoptic weather observations. TRADOS is an operational Gaussian long-range trajectory and dispersion model. Time-integrated air concentrations as well as dry and wet deposition forecasts for selected groups of radionuclides can be calculated. In the statistical mode also external dose rates and doses via several external and internal pathways can be estimated. The TRADOS model runs on the meteorological data from the Finnish version of the Nordic HIRLAM numerical weather prediction model or on the data from the European Centre for Medium-Range Weather Forecasts (ECMWF).

The FMI's radioactivity monitoring network consists of fourteen computer-controlled stations recording continuously aerosol β activity with a system where air is drawn alternately through one of two fixed filters wrapped around GM tubes. The discrimination between the short-lived ^{222}Rn progeny and artificial β emitters is based on different half-lives. Seven of the fourteen stations monitor also external radiation with scintillation detectors, rainfall intensity, and temperature of the air and five stations the electrical conductivity of the air.

Daily aerosol samples at four and daily precipitation samples at two stations are collected. These, and the weekly aerosol samples are measured with an automatic α/β analyser for long-lived β activity (mainly ^{210}Bi and possible artificial β emitters) five days after the end of sampling when the ^{220}Rn and the short-lived ^{222}Rn progenies have decayed. The daily aerosol and precipitation samples are measured also as soon as possible, usually 1-2 days after the sampling.

The daily aerosol and precipitation samples of two stations are examined with autoradiography to reveal any hot particles and the aerosol samples of one station are analysed with γ spectrometry. The three sounding stations operated by the FMI can also make radioactivity soundings of the upper atmosphere.

Introduction

The Finnish Meteorological Institute (FMI) is responsible for the dispersion forecasts during nuclear accidents and other emergencies requiring response actions. Because of its 24-hour duty service the Finnish Meteorological Institute is also, by an agreement, the national contact point of the international early notification system for nuclear events and other radiation

incidents. The Institute also maintains a monitoring network for atmospheric radioactivity. The main purpose of this monitoring network is the research on the use of natural radionuclides as tracers in air quality and climate studies but it operates also as a sensitive early detection network.

Dispersion models

The FMI's short-range dispersion model YDINO is used during emergencies at nuclear power plants in Finland (Loviisa and Olkiluoto) or at those closest to Finland (Forsmark, Ignalina, Kola and Leningrad). YDINO is a Gaussian surface trajectory dispersion model, which utilizes routine synoptic weather observations. The model's output files are transferred in real-time to the Finnish Centre for Radiation and Nuclear Safety, where they are utilized as radiological dosage model input.

An operational long-range dispersion model is available for other emergencies. This TRADOS model is a co-product of FMI and the Technical Research Centre of Finland. TRADOS is a Gaussian long-range trajectory and dispersion model (Fig. 1). Time integrated air concentration as well as dry and wet deposition forecasts for selected groups of radionuclides are computed in the operational emergency applications. In the statistical mode also external dose rate estimates and statistical evaluations of individual and population doses via several external and internal pathways can be computed. The model runs on the numerical meteorological data from the Finnish version of the Nordic HIRLAM (High Resolution Limited Area Model) numerical weather prediction model with a grid resolution of 55 km x 55 km. Similar data from the European Centre for Medium Range Weather Forecasts (ECMWF) can also be used (VALKAMA ET AL., 1995).

RADICAL is the decision makers' information system, which has been designed for managing radiation situations. The trajectory forecasts computed with TRADOS, wind and rain observations from synoptic hours and results of the FMI's radioactivity measurements are transmitted into the database of the RADICAL system.

SILJA is a modern high-quality dispersion model framework designed for multi-purpose use. It is a frame containing high-level tools for easy creation of dedicated dispersion models for different purposes. SILJA is developed in co-operation with the Technical Research Centre of Finland and an early version is now operational. It contains a mesoscale/long range random-walk particle model, dose assessment system and a three-dimensional trajectory model (SALONOJA, 1997; ILVONEN, 1997).

Continuous radioactivity measurements

The Finnish Meteorological Institute performs radioactivity measurements or sampling for laboratory analyses at 16 sites around Finland (Fig. 2). Aerosol beta activity is recorded continuously at fourteen stations: Mariehamn, Helsinki (two stations), Nurmijärvi, Virolahti, Vaasa, Tikkakoski, Joensuu, Kajaani, Oulanka, Rovaniemi, Sodankylä, Kilpisjärvi and Kevo. The stations in Helsinki, Tikkakoski, Oulanka, Sodankylä, Virolahti and Kilpisjärvi also monitor external radiation, rainfall intensity and temperature of the air. Electrical conductivity of the air is measured at Helsinki, Virolahti, Tikkakoski, Oulanka and Sodankylä. All the measured data can be interrogated remotely with a few minutes delay at the network's central station in Helsinki (PAATERO ET AL., 1994).

The aerosol beta activity is monitored with a system where air is drawn alternately through one of two fixed filters wrapped around GM tubes. The discrimination between the short-lived ^{222}Rn progeny and artificial beta emitters is based on different half-lives. The instrumentation detects an artificial beta activity in the aerosol of a few becquerels per cubic metre. The alarm limit for artificial particle-bound beta activity in the air is thus two or three orders of magnitude lower than with dose rate monitors. The filters are changed once a week and measured in the laboratory as weekly aerosol samples.

Two systems for measuring external radiation are currently in use. The first one is based on two NaI(Tl) scintillation detectors, one on top of the other separated by a 2.5 cm lead shield (Fig. 3). This counting geometry distinguishes the gamma radiation of suspended activity from the gamma radiation of deposited activity. In the case of precipitation an increase of 0.01 μSv/h in the external dose rate caused by artificial radioactivity can be distinguished from natural sources. Gross counts and single-channel analyser measurements are recorded. The other system uses only one NaI(Tl) scintillation detector but records the pulse height distribution with a multi-channel analyser in addition to gross counts.

The electrical conductivity of the air is measured with a Gerdien cylinder capacitor. Both positive and negative conductivities are recorded.

Laboratory measurements and radioactivity soundings

Daily aerosol samples (air filters, Munktel MGA, 3500 m^3/d) are collected in Helsinki, Nurmijärvi, Sodankylä and Ivalo. Daily precipitation samples from Helsinki and Ivalo (collected with 1 m^2 funnels) are evaporated to dryness and the residuals are transferred onto Whatman 42 paper filters for measurement.

The samples are measured in the laboratory with an automatic alpha/beta analyser.
The analyser incorporates five gas flow proportional counters: one alpha, one beta and one background detector above the sample, and one beta and one background detector below the sample. The counting efficiences for ^{210}Bi (beta) and ^{210}Po (alpha) are 75% and 42% respectively. The sample changer unit can handle 25 large-area (up to 240 mm diameter) air filters (MATTSSON ET AL., 1996).

All the daily aerosol and precipitation samples and weekly aerosol samples are measured for long-lived beta activity (mainly $^{210}Bi/^{210}Pb$ and possible artificial beta emitters) five days after the end of sampling when the ^{220}Rn progeny and short-lived ^{222}Rn progeny have decayed. Daily aerosol and precipitation samples are also measured as soon as possible, usually 1-2 days, after the sampling for ^{212}Pb and for surveillance purposes. In addition, the ^{210}Pb contents of the daily aerosol samples are assessed six months later by the alpha emissions of the in-grown ^{210}Po.

The daily aerosol and precipitation samples from Helsinki and Ivalo are also examined with autoradiography to reveal possible hot particles (highly radioactive agglomerates produced by condensation of radionuclides or by fragmentation of nuclear fuel). In addition, the aerosol samples from two stations are analysed with semiconductor gamma spectrometry.

Radioactivity soundings of the upper air can be performed at Jokioinen, Tikkakoski and Sodankylä. The radioactivity sonde consists of one beta-sensitive and one gamma-sensitive

GM tube. The sonde is attached to a normal weather sonde and is carried with a hydrogen-filled balloon to an altitude of 25-35 km. During the flight the pulse rate and the meteorological data are transmitted with radio to the ground station (Fig. 4).

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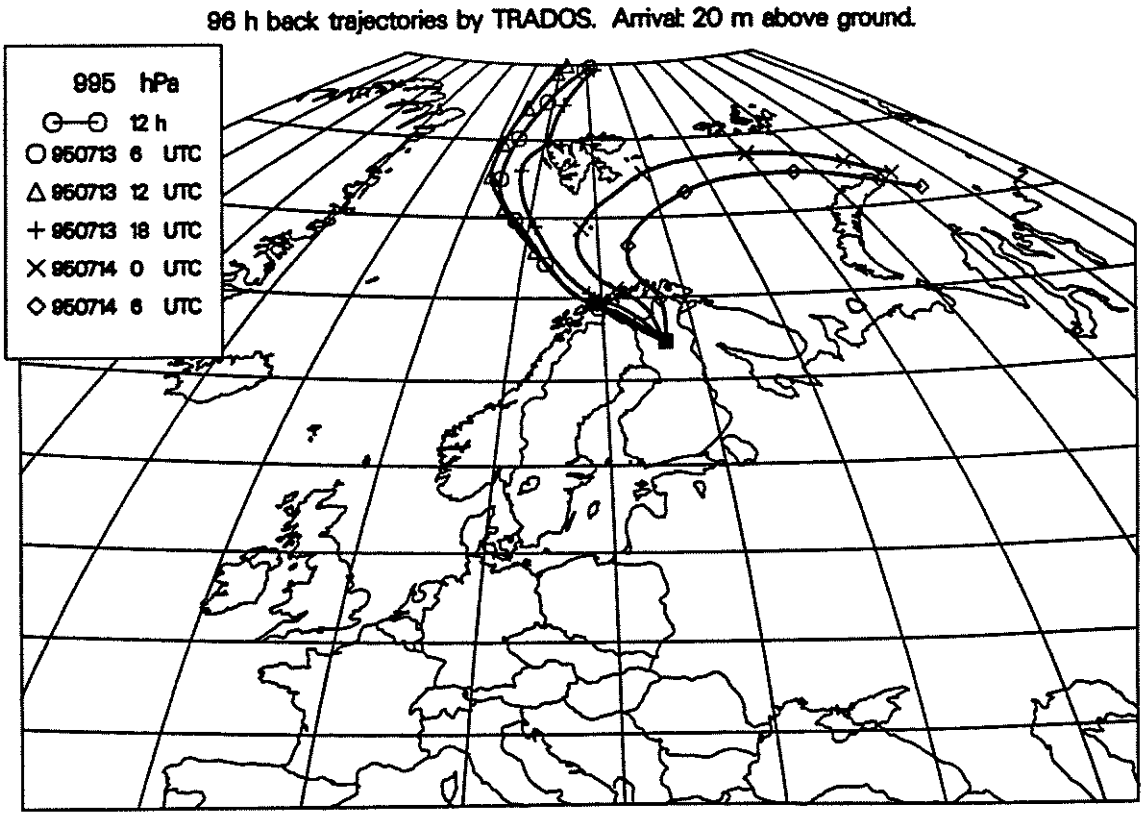


Figure 1. TRADOS trajectories computed 96 hours backwards in time, arrival at Sodankylä.

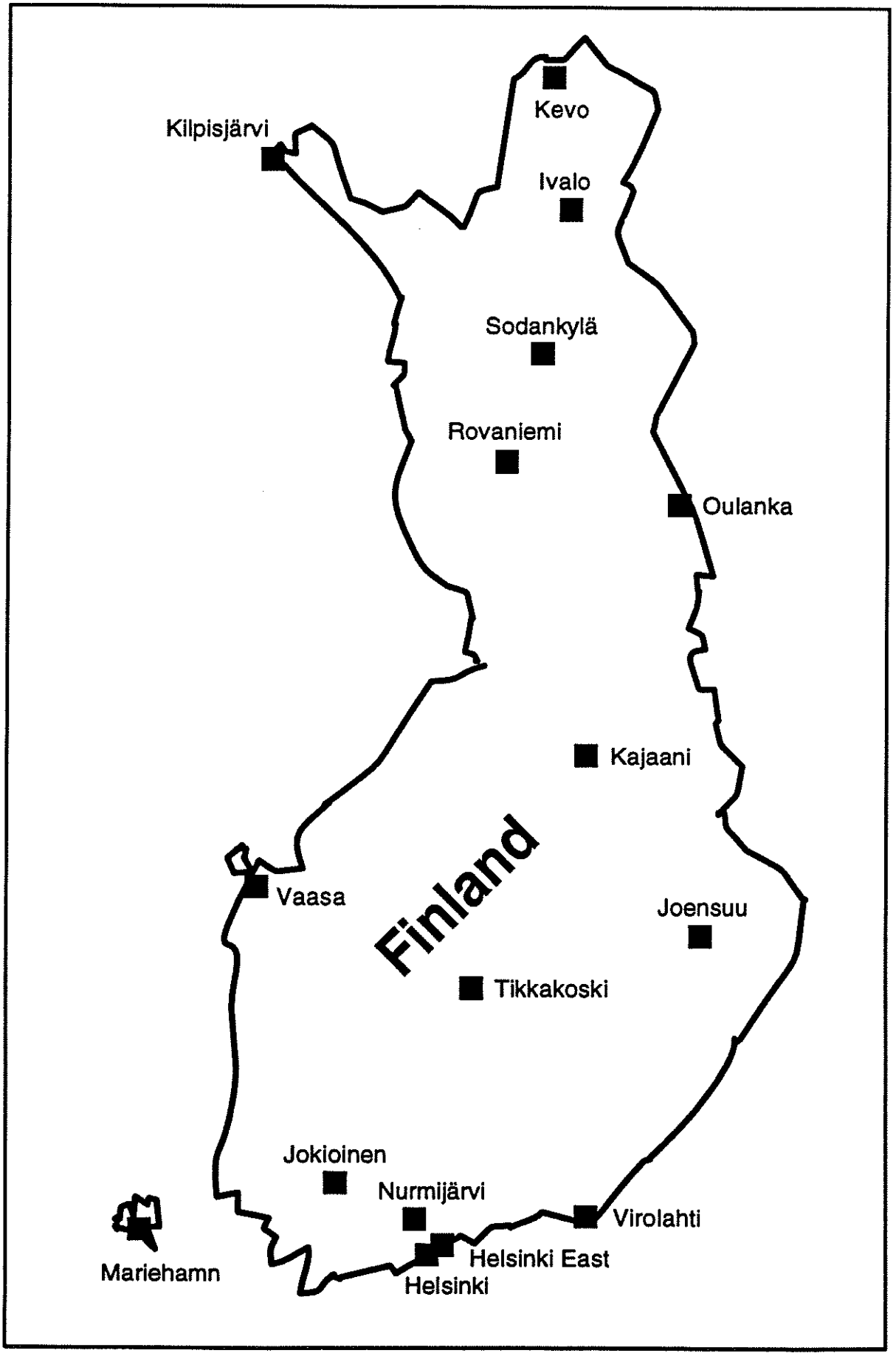


Figure 2. Radioactivity monitoring and sampling sites of the Finnish Meteorological Institute.

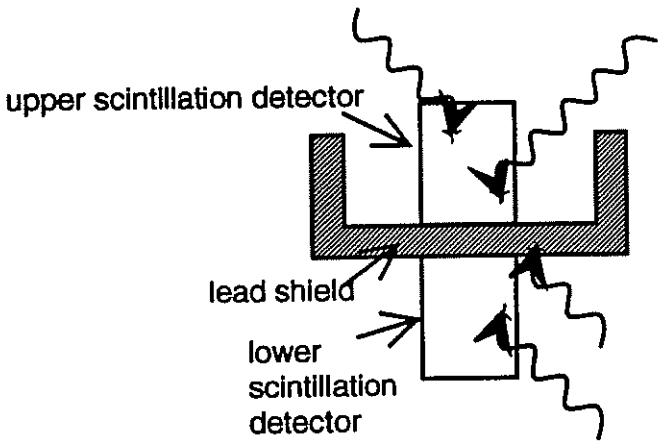


Figure 3. Detector arrangement for external radiation.

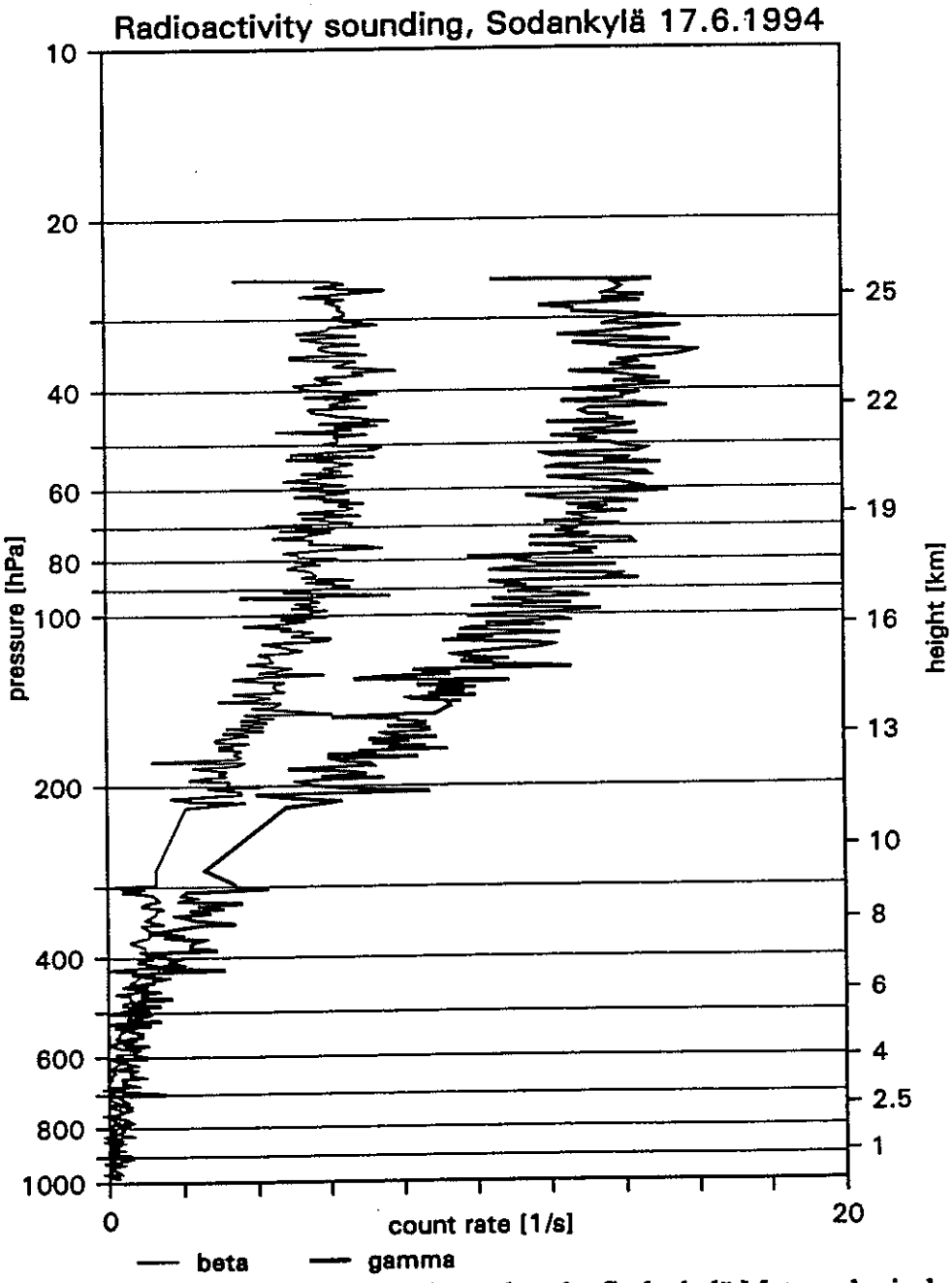


Figure 4. Radioactivity sounding performed at the Sodankylä Meteorological Observatory in June 1994.

Klaus Sjöblom (LOVIISA NUCLEAR POWER PLANT)

NORDISKA SÄLLSKAPET FÖR STRÅLSKYDD, MÖTE I REYKJAVIK 26-30.8.1996

EMERGENCY EXERCISE SCENARIO TOOLS

Abstract - Nuclear power plant emergency exercises require a realistically presented accident situation which includes various aspects: plant process, radioactivity, radiation, weather and people. Experiences from nuclear power plant emergency exercises show that preparing accident scenarios even for relatively short exercises is tedious. In the future modern computer technology and past experience could be used for making exercise planning more effective.

Emergency exercises

Emergency exercises are an essential prerequisite for appropriate preparedness. There is a large variety of exercises: from specialists' topical drills to large international co-operation exercises. Practising planned - or unplanned - countermeasures requires simulating an accident situation. This paper deals with tools for generating accident scenarios for utility, national or international level exercises. Some of these tools can be useful for internal training of an authority.

Accident Scenarios

An accident scenario describes the emergency situation from the point of view of the training organization. It can contain the following entities:

1. **Power plant**
   - a) Process
     - Initiating events
     - Defects in safety functions
     - Human failures
   - b) Component failures
     - Failure description
     - Repair times
   - c) Radiology
     - Core damage
     - Release path
     - Radioactivity on-site
     - Radiation on-site
   - d) Personnel
     - Injuries
     - Assembly and evacuation
     - Emergency organization

IVO GROUP

e) Rest of power plant
Electrical black-out, fire, flooding etc.

2. Environment
a) Transportation of radioactivity
Source term (nuclide types, quantities, time schedule)
Weather (wind direction and velocity, turbulence, precipitation)
Atmospheric dispersion
Ground deposition
b) Radiological consequences
Dose rates as a function of time and place
Doses (external, thyroid and effective whole body equivalent)
c) Radioactivity in foodstuff (water, milk, meat, fish etc.)

3. Society
a) Authorities (contacts, recommendations, decisions, protective actions)
b) Press (interviews and other information, also misinformation)
c) Population (protective actions, inquiries)
d) Other conditions (electric black out, extreme weather conditions, defects in infrastructure)

Full scale NPP control room simulation

Nuclear power plant process simulation has already traditions from several decades. In many countries it is even a regulatory requirement to have plant and unit specific simulators for training of all control room operators.

Loviisa Power Plant has had a "full scale" simulator since 1980. "Full scale" means that the simulator control room looks like the real one and most of the plant systems and redundancies have been included. The simulation grade has been improved and plant modifications have been added annually. Since 1985 the simulator has run in real time in utility level emergency exercises and also in larger rescue service exercises. During the past decade the following advances have been reached:

- The simulating of a two-phase flow in the PWR primary circuit has gradually become more reliable regarding to both the plant parameters and the simulator availability
- The simulator computer and the simulator's process computer have been connected to the plant process computer system
- An Emergency Operations Facility (EOF) for emergency management, which was established in 1990, has been equipped with a terminal connected to the computers mentioned above.
- Safety parameters can be automatically reported to command posts of STUK and IVO International in Helsinki
- The operators have been allowed to influence on the accident scenario.

- The simulator has increased the realism of an accident scenario compared with conservative licensing or PSA analysis. The operators' recovery actions and non-safety systems add a valuable credit to plant safety.

Yet there are many severe accident phenomena that the simulator is unable to simulate or the simulation results are less reliable, e.g. complicated core rewetting situations, containment hydrogen and pressure, fuel damage and radioactivity. So far the simulator has been oriented mainly for operator training, but it also provides other emergency management organizations with a new potential.

Other NPP process simulators

APROS

APROS (Advanced PROcess Simulation) provides a simulation environment for nuclear power plants or other processes. APROS applications include engineering and training simulators as well as comprehensive plant analyzers. APROS has been developed by VTT and IVO International, Finland. It can flexibly be adjusted to fit a specific plant. APROS models have been made e.g. for Loviisa 1/2, TVO 1/2, Forsmark 1, Ringhals 1 and Kola 1/2 & 3/4 plants.

As a tool for generating emergency exercises APROS has the following benefits: existing process models and experience from training use. There are plans for using it when estimating the releases in the Loviisa PSA study. On the other hand, APROS needs a work station, and additional radiological and other sever accident models require extra work.

MELCOR/MELSIM

MELCOR is a widely used code for severe accidents. It calculates two-phase situations in the reactor and the primary circuit, fuel damage, core melt, reactor pressure vessel penetration, core-concrete reactions, hydrogen production and behaviour, and containment functions. MELCOR is neither as accurate as RELAP nor as complete as a full-scale simulator, but it can analyze severe accidents beyond core uncover. Radioactive materials are divided into six groups: Xe, Cs, Ba, I, Te and others. The amount and concentration of them in fuel, primary system, containment/deposits, containment/airborne and environment are calculated.

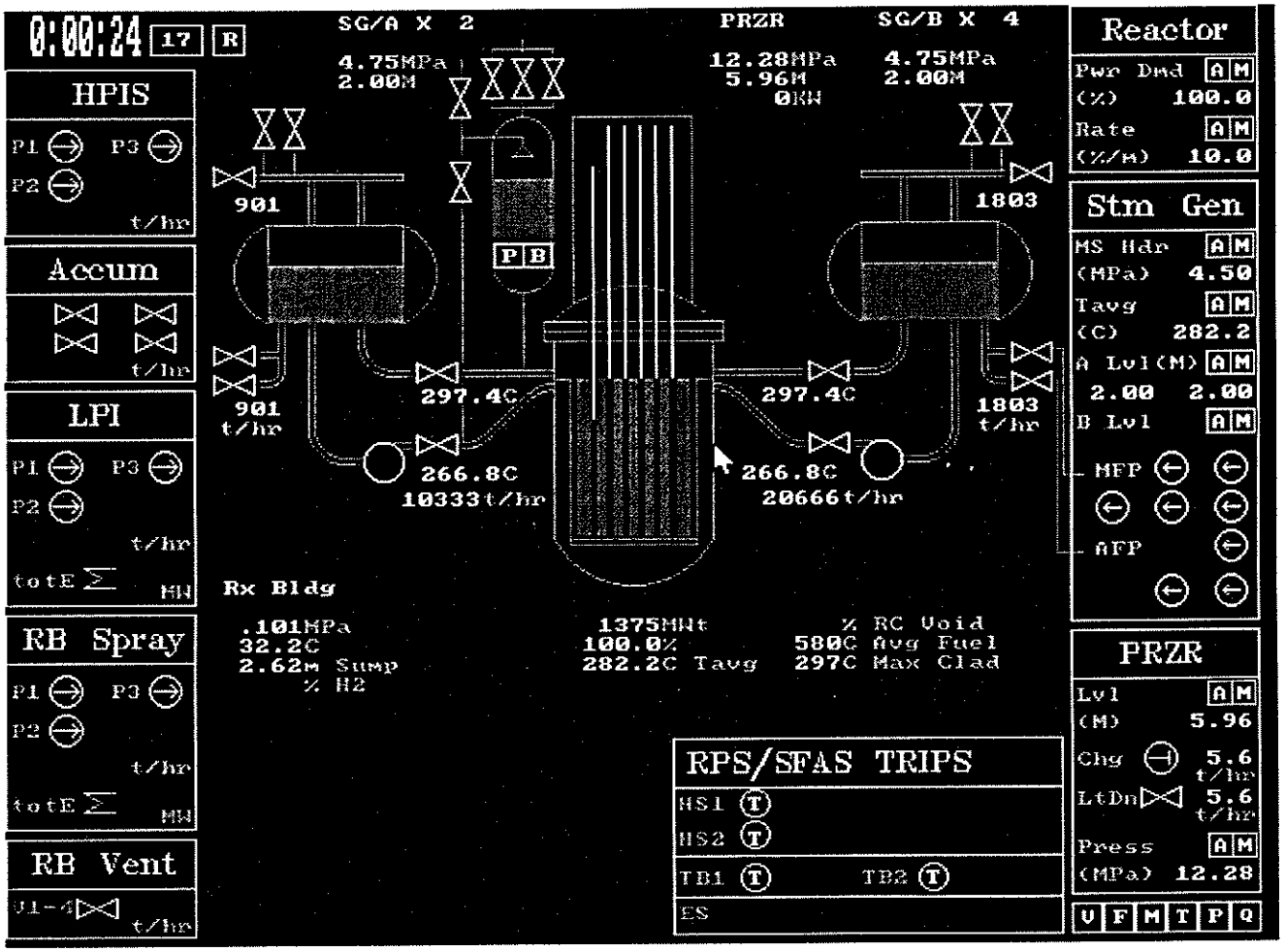
MELSIM is based on MELCOR but includes also a man-machine interface for interactive simulation sessions. MELSIM has been developed mainly for Technical Support (for operators) Center training rather than Emergency Operations Facility training.

MELCOR/MELSIM could benefit emergency preparedness by
- generating accident scenarios (in advance) for emergency exercises (like in Kola in 1995);
- providing an interactive plant simulator for operator training emergency exercises and
- generating releases of model accidents (quantity, quality & timing) to emergency plan.

PCTRAN

PCTRAN is a PC based NPP training simulator. It models severe accident phenomena inside a nuclear power plant. On a PC it can simulate much faster than real time (the user may use the simulation speed). PCTRAN can be used for self education. However, one shall note that PCTRAN is not a nuclear power plant analyzer.

The supplier is Micro-Soft Technology from the USA. It has a long list of clients (TVO and 16 others in 1995). A delivery can include a basic package, a plant specific model or severe accident and radiological models.



The Industrial Power Company Ltd (TVO) has used PCTRAN since 1995. It models the plant-specific features of the ABB Atom designed BWR, including its containment overpressure protection and filtered vent systems. Radiation leakage paths and dose rate distribution are presented graphically. Potential scenarios include core heat-up, hydrogen generation, core melt and vessel penetration.

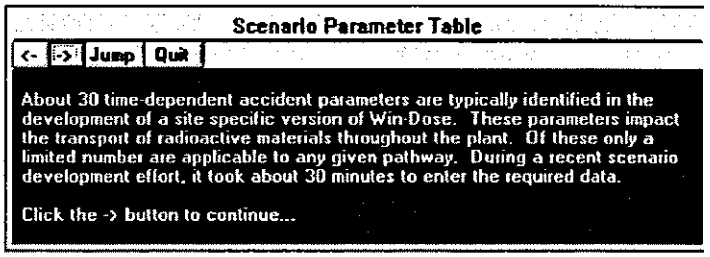
Plant radiation

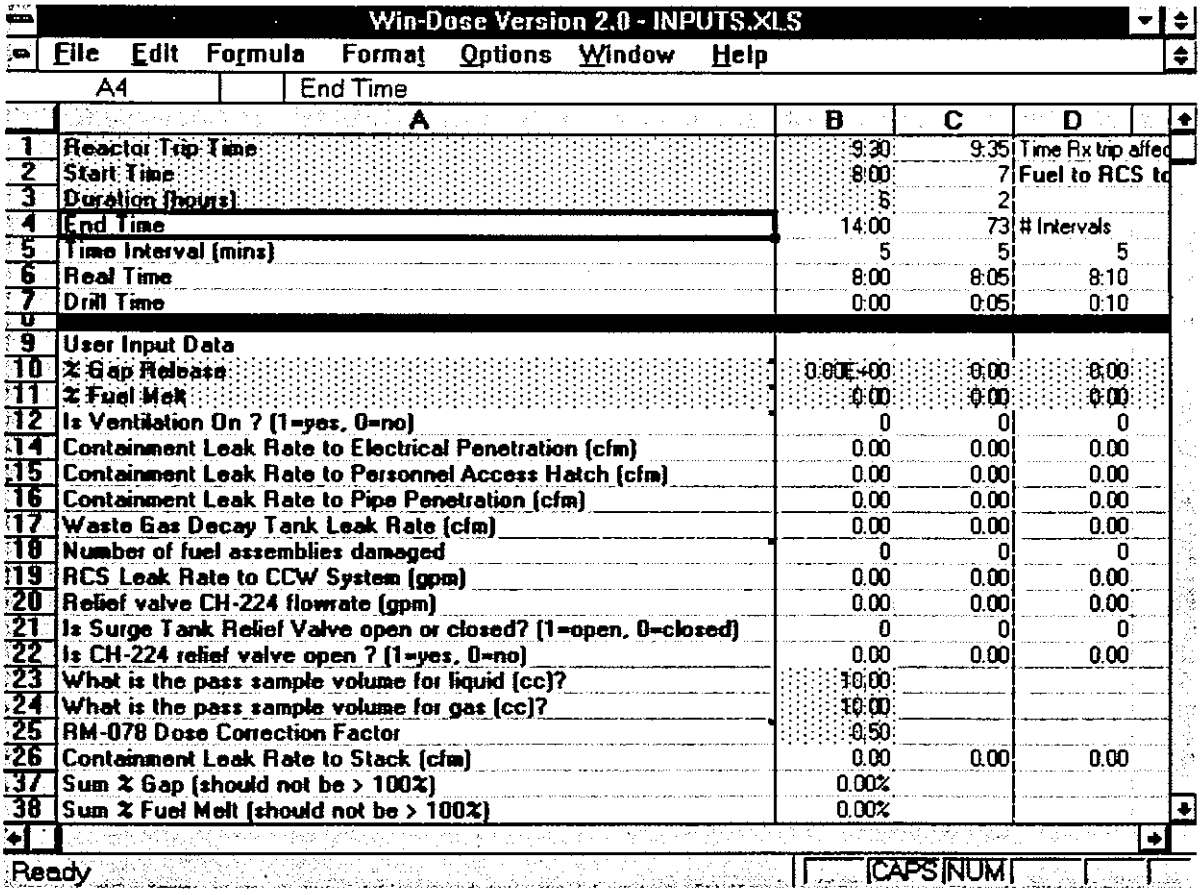
Win-Dose simulates radioactive materials and dose rates inside the plant. It uses specific data as accident types, source terms, radiation monitor information and compartment characteristics

As input it needs start time, duration, accident type, trip time, leak rates, leak pathway, fuel damage, plant status and operational data. Options are available to generate offsite radiological data, ingestion pathway data, accept downloaded data from a control room simulator and generate a file for upload to simulator.

The accident simulation of Win-dose includes: inplant contamination and dose rates, airborne radioactivity, post accident sample data and safety parameter display (SPDS) screens.

Win-dose has been developed by Earth Tech in the USA. Plant specific model development requires close work with plant knowledgeable staff and takes about 8 months.

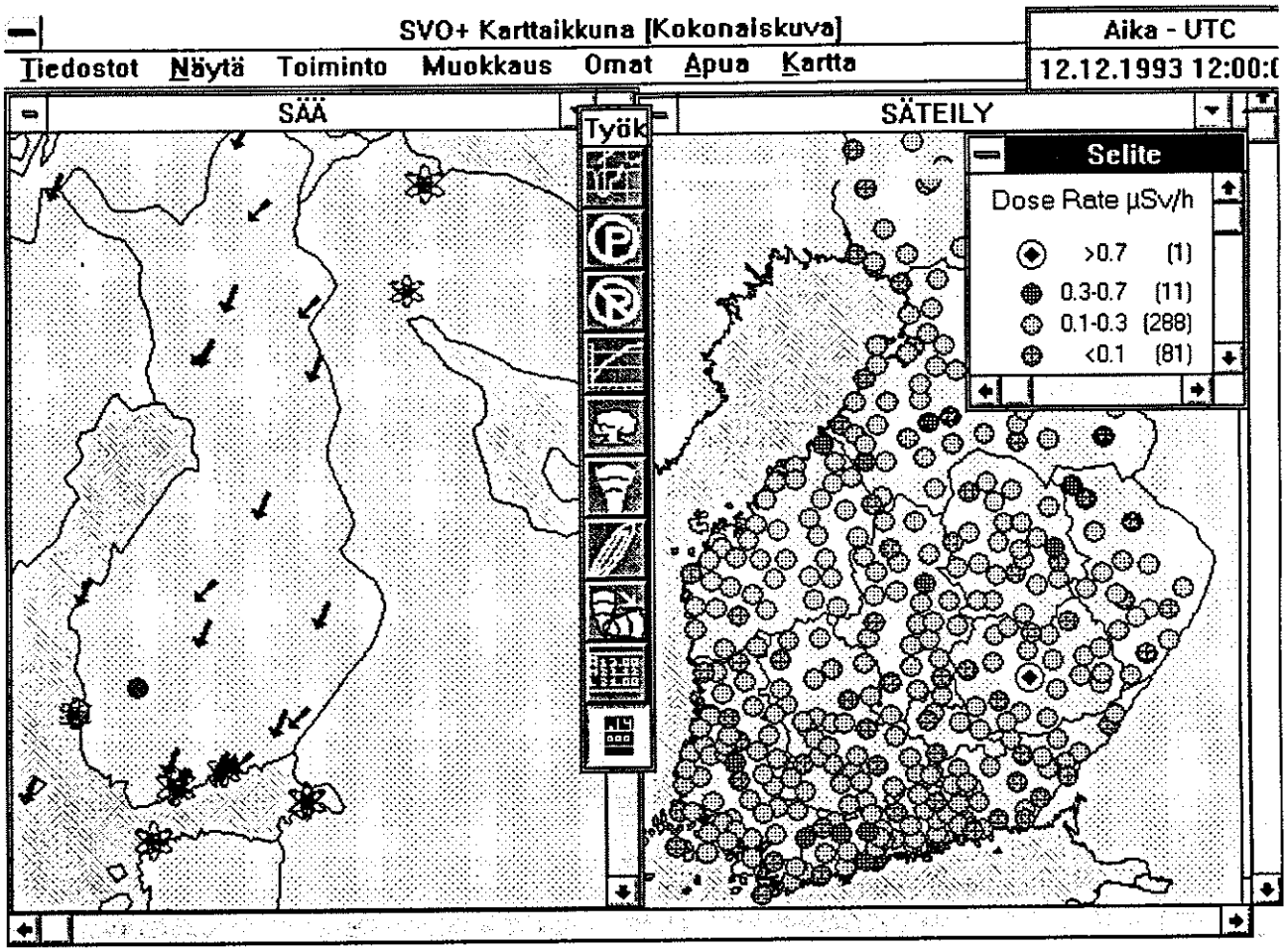




	A	B	C	D
1	Reactor Trip Time	9:30	9:35	Time Rx trip affec
2	Start Time	8:00	7	Fuel to RCS to
3	Duration (hours)	6	2	
4	End Time	14:00	73	# Intervals
5	Time Interval (mins)	5	5	5
6	Real Time	8:00	8:05	8:10
7	Drill Time	0:00	0:05	0:10
9	User Input Data			
10	% Gap Release	0.00E+00	0.00	0.00
11	% Fuel Melt	0.00	0.00	0.00
12	Is Ventilation On ? (1=yes, 0=no)	0	0	0
14	Containment Leak Rate to Electrical Penetration (cfm)	0.00	0.00	0.00
15	Containment Leak Rate to Personnel Access Hatch (cfm)	0.00	0.00	0.00
16	Containment Leak Rate to Pipe Penetration (cfm)	0.00	0.00	0.00
17	Waste Gas Decay Tank Leak Rate (cfm)	0.00	0.00	0.00
18	Number of fuel assemblies damaged	0	0	0
19	RCS Leak Rate to CCW System (gpm)	0.00	0.00	0.00
20	Relief valve CH-224 flowrate (gpm)	0.00	0.00	0.00
21	Is Surge Tank Relief Valve open or closed? (1=open, 0=closed)	0	0	0
22	Is CH-224 relief valve open ? (1=yes, 0=no)	0.00	0.00	0.00
23	What is the pass sample volume for liquid (cc)?	10.00		
24	What is the pass sample volume for gas (cc)?	10.00		
25	RM-078 Dose Correction Factor	0.50		
26	Containment Leak Rate to Stack (cfm)	0.00	0.00	0.00
37	Sum % Gap (should not be > 100%)	0.00%		
38	Sum % Fuel Melt (should not be > 100%)	0.00%		

Off-site radiation

The calculation of transportation of radioactivity and environmental radiation is relatively straightforward. There are plenty of verified models, from compact to big computer applications: RODOS, NORMEM, RIMPUFF, OIVA, AINO, LENA, ROSA etc. However, a realistic presentation could still be developed especially when real-weather is going to be used. One recent advance is a training version of SVO+. SVO+ is a Finnish real time radiation control system that receives and displays data also from other Nordic countries. SVO+ is managed by the Finnish Centre for Radiation and Nuclear Safety. The next figure shows as an example one SVO+ display.



There are also simulator systems for moving radiation monitoring teams. One such is KATASIMU-96. Together with KATAMAP-96, a radiation survey system, it can add the postulated radiation data to the real (normal background) radiation as a function of place and time. KATASIMU has been planned to be able to run in connection with the training version of SVO+. KATASIMU and KATAMAP are supplied by KATA ELECTRONICS in Finland.

Human behaviour

The human behaviour of on-site personnel, near-site population, press, authorities and public forms an essential part of the accidents. In this respect the humans are still irreplaceable.

Statens strålskyddsinstitut
Swedish Radiation Protection Institute

24 augusti 1996

EMERCON-ECURIE International Notification of Radiological Emergencies

Kjell Nyholm, John-Christer Lindhé, Statens strålskyddsinstitut, Stockholm

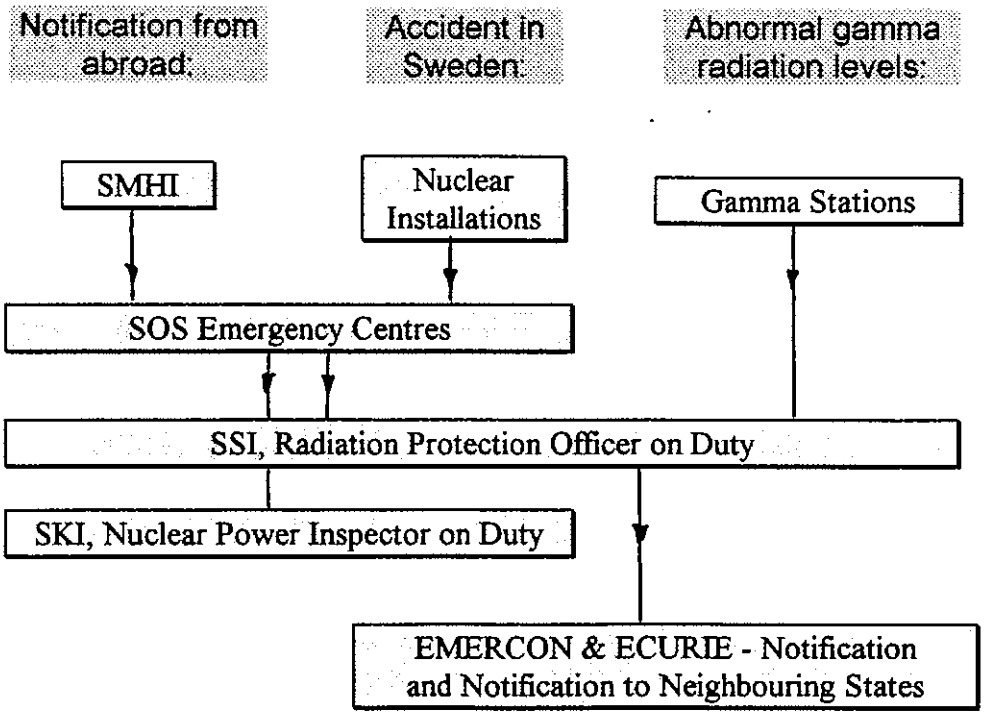
1. Swedish Notification Procedure for Nuclear Power Reactor Accidents and Other Radiological Emergencies

Introduction

The procedure is a bit different depending on whether the notification is coming to or from Sweden. In the first case all notification messages to Sweden are transmitted through SMHI, the Swedish Meteorological Services Centre in Norrköping with 24 h duty. From there the competent authorities especially SSI, the Swedish Radiation Protection Institute, and SKI, the Swedish Nuclear Power Inspectorate, are notified with the help of the SOS Emergency Centre in Stockholm where the updated telephone and pager numbers to authority personnel are kept.

In the case of a radiological emergency, such as a release or abnormal gamma radiation levels detected in Sweden, we are obliged by international and bilateral commitments to give prompt notification to other states that might be affected by a release. The notification will be made by the responsible authority SSI (in co-operation with SKI) directly to the International Atomic Energy Agency, IAEA and the Commission of the European Union and to the neighbouring states by fax and/or telex.

The notification network



2. Emercon and Ecurie Two International Brothers With Some Differences

Introduction

The Emercon and Ecurie notification systems are both offspring of the Chernobyl accident in 1986. Emercon is hosted by IAEA in Vienna and serves all countries connected to the Agency. The Ecurie system with centres in Luxembourg and Brussels is developed within the EU to meet the demands of the countries in the Union.

Differences Between the IAEA Convention and the EU Council Decision

The IAEA Convention (EMERCON)	*EU Council Decision (ECURIE)*
Range of accidents Notification is required for accidents in several specified facilities and activities whereas not listed facilities (i.e. military) are left for the accident state to decide on.	**Range of accidents** Notification is required for accidents in all types of nuclear installations within the border of a member state (thus including jointly operated international facilities).
Notification triggers When an accident within a member state has lead to or might lead to a trans-boundary release with radiological safety significant effects in another state. The accident state has the responsibility to make this judgement.	**Notification triggers** When emergency measures of a wide-spread nature are decided in the accident state necessitated by either an accidental release within the state or by the discovery of abnormal radiation levels.
Exchange of information 1. Only states that are or might be physically affected by the transboundary release have to be informed. The accident state must promptly notify the IAEA but can leave to IAEA to further notify other states that are or may be affected. 2. It is not required to give reports on measures taken or planned to inform the public. 3. It is only required that the accident state should report results from environmental monitoring to the other states that might be physically affected by the release. 4. It is not required to give reports on results of measurements of foodstuffs, feedingstuffs and drinking water. 5. Repeated reporting of results of measurements are required but does not have to include the measurements mentioned in point 4 above.	**Exchange of information** 1. All member states have to be informed and all information received by the Commission is redistributed to all states. The accident state must promptly notify the Commission and also directly notify other states that are or might be affected. 2. It is required to give reports on measures taken or planned to inform the public. 3. It is required that the accident state should report results from environmental monitoring to the Commission to be redistributed to all member states. 4. It is required to give reports on results of measurements of foodstuffs, feedingstuffs and drinking water. 5. All states have to give prompt reports on measures taken and recommendations given after having received a notification. Repeated reporting of results of measure-ments are required and should also include the subjects in point 4 above.

Other provisions	**Other provisions**
1. Testing of notification procedures are not required but are usually done annually. 2. English, French and Spanish are the only languages supported. 3. Procedures for settlement of disputes concerning the convention are specified but signatory states might have declared themselves not bound by the procedures.	1. Regular testing of notification procedures as well as consultations between the Commission and the member states on procedures are required. 2. All official languages will have logistic backup to enable communication. 3. The procedures and requests laid down by the Decision are legally binding for all EU-states. (Euratom Treaty).

Ref.: "Convention on Early Notification of a Nuclear Accident", Vienna 26 September 1986 and "Council Decision of 14 December 1986 on Community arrangements for the early exchange of information in the event of a radiological emergency"(87/600/Euratom)

The conclusion of the comparison is that if our procedures are designed to comply with the Ecurie requirements then all requirements for Emercon will automatically be satisfied.

3. The Emercon-Ecurie Notification PC-program for MS Windows

Introduction

The demands for quick notification in case of an accident are not easily met in a situation when everybody is busy collecting information and making analyses in order to understand what has happened and what might follow. The list of questions to be answered is long and some answers can be prepared in beforehand e.g. latitude and longitude of the nuclear installations in our country. The development of a PC-program to speed up the procedure was therefore of high interest.

Scoop of the Program

The program will present one question after the other and depending on the answers it will jump past irrelevant questions and will fill in evident data from a data base as well as propose likely or possible answers in some other cases.

After a complete input of the answers the program will present a file of answers in coded form ready to send as well as a parallel file with answers in a easy to check non-coded form. Answers can be edited right in the answer file or - if preferred - it will also be possible to go back to the relevant questions within the program and change the answers there.

Features of the Program

Not all questions has to be answered in the first notification. Therefore the program has a special choice {Meny1} for the first notification message. Only the relevant questions will show up in this alternative with jumps past the rest of the questions in order to speed up the procedure. (See figure 1 next page)

For reports to follow the first notification the program has an alternative {Meny2} where all the questions will show up and with the answers from the first notification available to be used or to be altered.

Whenever a time and date has to be given in an answer, the program will transform the local time from the PC system clock to UTC and will also take the sun-saving-time into consideration. (See figure 2)

Figure 1

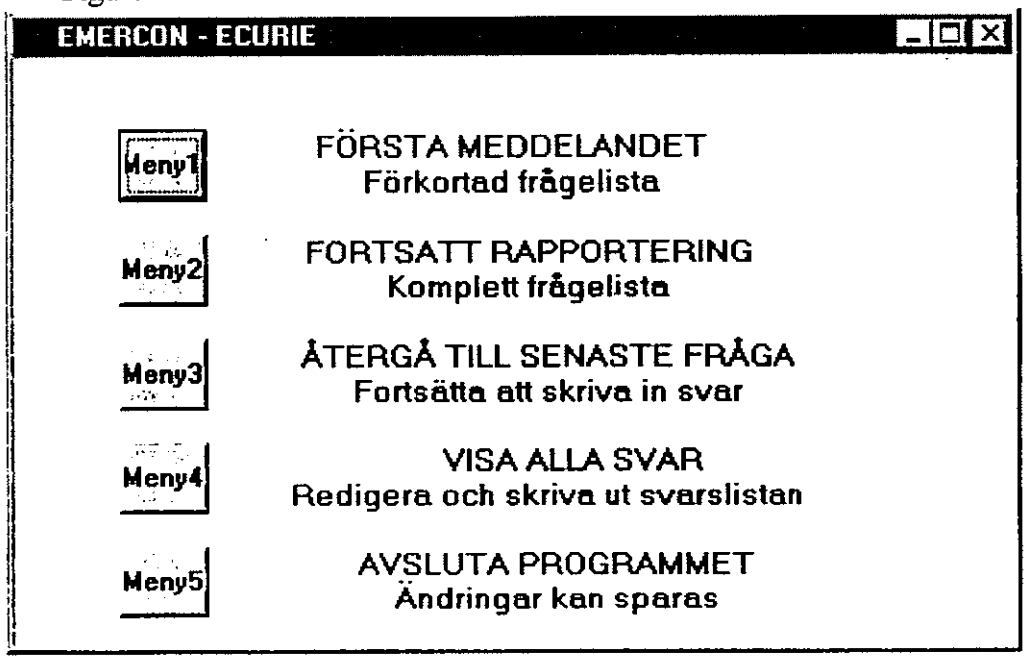
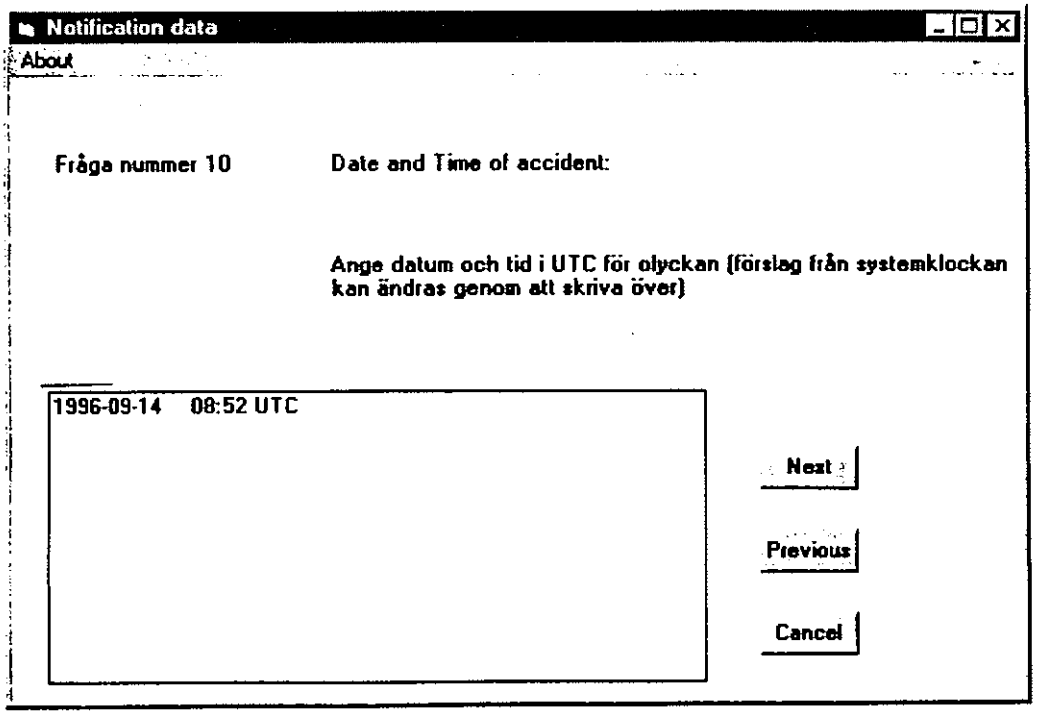


Figure 2



Radiation monitoring on the border and in the metal scrap recycling industry in Finland

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Abstract The aim of the control of passengers and goods for radiation at customs is to prevent the illegal or unintentional import and export of radioactive materials. The Finnish customs and the Finnish metal scrap recycling industry began radiation measurements soon after the Chernobyl accident with portable dose rate meters. The customs have now fixed installed measuring systems with large plastic scintillation detectors located at all crossing points on the eastern border of Finland. Fixed systems are also in some harbours and at the airport of Helsinki. The customs have also hand-held monitors with very high sensitivity to detect radioactive material. Today all trains, autotrucks and cars crossing the border from the Russian Federal Republic or coming by ships from Estonia can be monitored using modern technology. The metal industry has also fixed monitoring systems and hand-held radiation monitors to find radioactive material and to prevent it from getting into the melting process.

Monitoring equipments

The customs have today seven fixed portal monitors for trucks and cars, four monitors for trains and nine monitors for passengers. Fixed monitors are located at seven main crossing points on the FIN-RUS border, in Helsinki harbour and at Helsinki-Vantaa airport. The portal monitors for trains, trucks and cars are equipped with large (two 100 x 50 x 5 cm^3) plastic scintillation detectors to find sources emitting gamma radiation. The monitors have also some sensitivity for neutron radiation and for strong beta radiation via bremsstrahlung.

The fixed monitors for passengers and luggages are based on sodium iodide scintillation detectors or on GM counter tubes with big volume.

The customs have also about 70 hand-held survey or dose rate meters. Some portable survey meters are equipped with very sensitive plastic scintillation detectors or with 2" x 2" Na(Tl)I detectors to locate radioactive material in the case of an alarm from the fixed monitoring system. Dose rate meters are based mostly on GM tubes.

The Finnish steel industry has today six fixed monitors for trucks and six for trains to find radioactive material in scrap metal. The first fixed system was installed in summer 1994. The industry has also several tens hand-held instruments mostly based on GM tubes. In addition some survey meters, specifically designed to find radioactive material, are used. Some meters are also beta sensitive. The buyers of metal scrap often perform radiation monitoring in the country of origin before the scrap is transported to Finland.

Sources of radiation

Radioactive material can be found in the metal scrap. It may be true sealed sources, activated scrap or contaminated scrap. Some industrial raw materials contain naturally occurring radioactive materials in so high amount that they can be detected at the border control. Smugglers may also try to transport sealed sources or nuclear material. In addition the patients treated with radionuclides may cause an alarm in the fixed installed radiation monitors.

The Finnish experience

Because of the low sensitivity of the GM tube dose rate meters the Finnish customs had identified only few cases of radiative substances before 1994. Examples of radiation sources found include radium and uranium contaminated scrap and materials containing naturally occurring radioactive materials e.g. potassium carbonate or zirconium sand.

In the winter 1995 a Swedish company began the transport of several thousand tons of aeroplan scrap from Kazakhstan to Finland to be shipped further. The customs found among the scrap radioactive instruments. Because of great amount of radium luminous paintings the first transport was stopped in the harbour of Kotka, and the company was informed by the Finnish Centre for Radiation and Nuclear Safety (STUK) of radioactive material. The company promised to perform the decontamination of the scrap in Kazakhstan before the transport. When the rest of scrap arrived in Finland, almost all radioactive instruments were still left, and the transport was stopped again. Due to economical reasons the Swedish company disappeared in the autumn 1995. Today the radioactive components are separated from the scrap through very hard and expensive work and packed for return to Kazakhstan. The well-decontaminated aluminium scrap has been sold to a Finnish company at a reasonable low price. Some companies do not get back all money they have spent in handling or transportation of the scrap. This is an example of what may happen in the business with scrap containing radioactive material.

One smuggling case has also been identified. This involved the import of a sealed californium-source with very low activity hidden in a wooden pallet.

The metal industry has also performed intensive radiation measurements for scrap metal since the Chernobyl accident and particularly since the breakdown of the former Soviet Union. About 20 cases of contaminated scrap or sealed sources have been found. The main isotopes were ^{226}Ra, U (depleted) and ^{137}Cs. The activities of sealed ^{137}Cs sources were below 100 MBq. None of the findings has caused any significant risk for anybody.

Procedures

The main procedure, when a radioactive source or cargo containing radioactive material is found, is to halt the transport and return the radioactive material to the sender. The decision for the return is always made by the customs.

Additional measurements may also be carried out, mostly by STUK, to identify the source and its origin. The metal recycling industry does not receive radioactive scrap. Radioactive parts or the whole loads are returned to the suppliers based on their agreements. If the origin of the contaminated scrap is unknown, the radioactive components are stored by STUK.

Soaking foodstuff with lye as a counter measurement to reduce radio-caesium, potassium and polonium

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Abstract-We have been studying the concentrations of ^{137}Cs, ^{40}K and ^{210}Po in pike, reindeer and dried ling (stockfish) before and after the samples were soaked in lye (NaOH-solution). From these experiments we got information about the similarity between potassium and caesium behaviour in the cellar structure and the effect on different tissues when soaking with lye. Dose calculations were done before and after treatment and the dose reduction were estimated. Out of this we were evaluating if the old Scandinavian tradition of soaking fish in lye is a method to use when foodstuff is contaminated.

When soaking with lye, the lipid bilayer in the cell membrane is broken down by the hydroxide-solution to soap. This makes it possible for the intra cellular bonded ions to diffuse out to the extra cellular fluid and further out to the lye solution which the sample is placed in. It appearers that after four days the two nuclides (^{137}Cs and ^{40}K) reaches a state of equilibrium in the lye solution. The extraction velocity (percent of the total content of the isotope that extracts per day) were the same for Cs and K both in lye- and rinse-process. From this we can conclude that caesium is present intra cellular and in ion-form. The protein in the cell structure prove to have a slower brake down under the influence of hydroxide solution and form protainats. It is expected that polonium should follow the protein brake down, as polonium is associated to particles (colloids) in alkali solutions.

Reduction of the alkali nuclides ^{137}Cs and ^{40}K were between 80-100 %.^{210}Po reduction was between 0-70 % (sometimes even a higher concentration after soaking), and with an average reduction of 20 % . Radiation doses were reduced between 60-90%, where a big part of the dose came from ^{210}Po.

Introduction

There has been a need and a wish to find methods to reduce radionuclides which result in radiation dose to man. These efforts has ranged from blocking radionuclides, with analogue stable elements, to climb from one to another trophic level in the food chain, till preparing foodstuff in a certain way. Lars Håkansson and Tord Andersson[1] have investigated the effect of liming lake and wet land. Their aim was to reduce levels of radionuclides in fish by blocking Cs and exchange it with other alkali-elements which were put into the wetland environment. This is an exampel of a long term countermeasurement method. The effect of the treatment is seen after several months to years after the action. Petäjä and Rantavaara[2] described methods to reduce radio caesium concentration in meat and fish by soaking it with brine (NaCl-solution), whereby radio caesium exchanges with sodium. They studied several parameters as brine concentration, soaking time, sample size etc. In Norway Andersson Sørlie[3] used a dietary advice as a countermeasure to reduce dose to man. They had chosen a group of the population which has special food habits, such as farmers/hunters and Lapps. All these papers have studied radio caesium reductions. This paper also include investigation of ^{40}K and the α-emitting radionuclide ^{210}Po.

In this paper there has been an investigation if soaking with lye (NaOH-solution) is as effective as expected to reduce radio caesium. When the cell is alive there is a high intracellular concentration of potassium compared with extracellular (150 mM intra- and 4mM extra-cellular). When the transport of the cellfuel (glycose and ATP) is stopped, the active transport of Na- and K-ions through the membrane is stopped as well. This takes place when the cell dies and is a consequence of that the transport proteins have consumed the fuel and stopped working. If the isotopes were in the extra cellular fluid, they could be extracted in water. When the cell dies, ions inside the cell cannot pass the cell membrane unless there is a breakdown

of this membrane. The main effect of soaking with lye is that the lipid layer (a part of the cellmembrane) is dissolved by the hydroxide solution, as soap, whereby the alkali ions can move freely and the difference in concentration between intra- and extra cellular fluides comes to an equilibrium. The protein breakdown is a slower process and has probably not that much with Cs and K "extraction" to do. All of these elements (Na, K and Cs) are from the first group in the periodic table, and have therefore closely-related chemical properties.
There is no similar theory for polonium, which holds as a scientific argument. Polonium has no biological function in the body, but in some papers[4,5] they have observed some accumulation in fish liver. We know that polonium is strongly associated with particles in alkali environment, and there is a possibility that it may follow the protein breakdown. NaOH-solution is making protainates of proteins, and when the cell is dead some enzymes inside the lysosomes are set free into the cytoplasm. These enzymes are also breaking down the protein in the cell membrane. The effect of alkali treatment is a softening of the tissues due to the dissolution of proteins. After treatment only about half of the protein remains in the sample, and all the fat has been dissolved into the lye-solution. From this it is expected that Po would stay in the tissue, unless there are some particles (colloids) in the solution which they can be associated with.

Why soaking with lye?
Soaking stockfish in lye is an old Scandinavian tradition. In the Middle Ages there was even an occupation to make this preparation of the fish called "fiskblöteskor". The Swedish national skald, Carl Michael Bellman (1740-1795), mentioned this in one of his poetry[6]. Even earlier, in the beginning of 16 th century, the Bishop of Linköpings diocese mention lutefisk in his description of his Christmas smorgasbord. Nowadays Scandinavian people are eating Lutefisk for Christmas and Easter. Lutefisk was allowed to eat lutefisk for the fast. This because this food has a very low content of fat. Almost all of the Lutefisk which is eaten today is produced in special Lutefisk factories[7].

Material

-Ling material (stockfish), *Molva molva*
The ling was bought in "GLADA FISKEN" (a store in Malmö market hall) one year after it was caught in Skagerrak. The fish was in a form of a stockfish, which is a dried cleaned (head and gutter removed) ling. After the fish was caught, it was put in a drying room for three months and then delivered to the stores. In "GLADA FISKEN" they stored the stockfish hanging from the ceiling. The weight was 0.761 kilograms.

-Ling material from Skärhamns frys AB
This is a Lutefisk factory that produces about one third of the Swedish Lutefisk consumption. Samples from stockfish (with and without skin) was taken from here.These fishes were caught at the same time and same place in the North Sea. After cleaning they were dried in a drying house. 25 litre rinse water sample were taken from one of the producing tubes. The tube contained about 600 litres and 400 kilograms of Lutefisk. One sample, 25 litre, from the lye solution was taken from a drain well, containing several cubic metre of waste solution. About two kilograms of lutefisk was taken. This sample contained pieces from about fifty fishes.

-Pike material, *Esox lucius*
The pike material included three fishes from Ryssby lake 20 kilometres east of Ljungby,

Sweden. They were caught in the middle of May, 1995. Their weight was between 1.2 and 1.6 kg. Before soaking they were cleaned (gutted and heads removed), but not dried as a stockfish. By personal communication with Karl Simonsson, (who is "soaking with lye expert" at ABBA foodindustry in Kungshamn, Sweden) I was informed that the results of soaking a fresh or dry fish were almost the same. Two days after they were caught, the soaking process was started.

-Reindeer fillet, *Rangifer tarandus*
Apart from fish we also used reindeer meat in our experiments. The main reason for choosing reindeer was because of the high levels of fall-out radio caesium from Chernobyl in the meat. This makes it also easier to measure the radio cesium in the different steps performed. The reindeer-fillet was bought in Östersund market hall and transported down to Lund in an ice-cooled styrox-box. After defrosting the meat (two fillets with 0.35 kilograms total wight) the soaking process was started at once.

-Reindeer steak and liver
The steak and liver were bought at "Vilt-Specialisten", a butcher store in Malmö. The weight was 1.0 kg for the steak and 0.7 kg for the liver. These samples were from autumn slaughter (3 month before preparation). After defrosting the soaking process was started.

Methods

-Soaking process
For the preparation of the soaked ling with lye, the recepie from "Prinsessornas Kokbok"[8] was followed. The soaking process described in "Prinsessornas Kokbok" has four subprocesses. The first one is swelling the fish in fresh water. Instead of tap water, demineralizing water was used, produced of Millipore's model Milli-RO 6 PLUS. This was used to avoid contamination and undesired soaking processes, like diffusion between the minerals in the water and radionuclides in the sample. This subprocess was going on for one week. The water was changed four times in this period of time. Subprocess two is the soaking process. For every kilogram meat sample (dry wight): 0.1 kilogram NaOH, 0.5 litre $Ca(OH)_2$ and demineralized water was added. The chemicals were from Kebo Lab, Sweden. The chemicals were diluted with demineralized water so that sample/solution weight ratio was 1:5. K_2CO_3-solution was produced as described in "Prinsessornas kokbok"by boiling birch ash.(This was the old method to produce hydroxide solutions.) This process proceeded for one week. The third subprocess was rinsing in demineralized water during one week. The water was changed every day, sample/solution weight ratio was 1:5. These three subprocesses were done at the inner yard of Dep. of Radiation Physics (Lund) at a temperature between 10 - 15°C. Finally the samples were boiled in demineralized water. The recipe says that the water should contain salt (NaCl). In this experiment we omitted the salt because it is an effective decontaminator[4], and here we were only interested in studying the effect of soaking with lye.
For pike and reindeer the soaking process was a little bit different since we had fresh products. The swelling subprocess was therefore omitted as well as the boiling process for the reindeer.

-Gamma detection and activity-concentration determination for ^{137}Cs and ^{40}K
From every subprocess samples for gamma spectroscopy were taken. For Raw-material and Final-material all the drying was done in an oven at 80°C for about 70 hours.

Raw material

Ling:

Half of the dried fish was milled with a Arthur H. Thomas Co. mill. Cutting was done along the spine and assumed the same activity-concentration in both halves. The remaining half was cut in three pieces to fit in a 5 litre glass bowl.

Pike:

The three pikes were also cut along the spine and the halves were dried and milled. Before drying scale and bone were removed. These fishes were in the same size and caught in the same lake at the same time, accordingly to this we assume similar activity-concentration in the fishes.

Reindeer(fillet, liver and steak):

We took randomly small pieces of the meat, and it was assumed that the activity-concentration was the same all over. For liver and fillet we determined the activity-concentration on wet-weight and the sample was dried afterwards. This was done because we had limiting material. The wet-/dry- weight ratio was for that reason determined afterwards.

Waste solutions from sub-process 1-4

When the waste solution volume was small (<5 litre) it was evaporated to a volume smaller than 180 millilitre, otherwise caesium was preconcentrated with copperferrocyanide precipitation. ^{134}Cs was used as radiochemical yield determinant. The precipitations give only information about Cs- concentrations, so in this sub-process we did not get any information about ^{40}K , ^{210}Po. Activity concentration for ^{137}Cs was calculated as follows:

$$\frac{A_{137}}{A_{134}} = \frac{A_{read,137}}{A_{read,134}} \Leftrightarrow A_{137} = \frac{A_{read,137}}{A_{read,134}} \cdot A_{134} \quad ; \quad A_{134} = m_{yield} \cdot A_{K,\,yield}$$

$$A_{K,\,137} = \frac{A_{137}}{m_{sample}} \Rightarrow A_{K,\,137} = \frac{\left(\frac{A_{read,137}}{A_{read,134}}\right) \cdot m_{yield} \cdot A_{K,\,yield}}{m_{sample}}$$

there : A_{137} is total activity of ^{137}Cs in the sample, [Bq]
A_{134} is total activity of ^{134}Cs in the sample, [Bq]
$A_{K,\,137}$ is activity concentration of ^{137}Cs in the sample, [Bq/kg]
$A_{read,\,137}$ is activity concentration read from peak search program for ^{137}Cs [Bq/kg]
$A_{read,\,134}$ is activity concentration read from peak search program for ^{134}Cs [Bq/kg]
m_{yieild} is the mass of the radiochemical yield determinant put in the sample, [g]
$A_{K,\,yield}$ is activity concentration of ^{134}Cs in the yield solution [Bq/g]
m_{sample} is the dry mass of the sample [kg]

For the two processes with reindeer steak, build up of ^{137}Cs and ^{40}K was studied in a 1.5 litre calibrated tin on a HPGe-detector.

Final material

Before gamma spectroscopy was done, the samples were dried and milled. For the pike and ling the scale and bone were removed before drying.

Measurement

The milled samples were weighed and put in cans for γ-spectroscopy. There were three sizes of the cans, 35 , 60 and 180 ml. All measurement errors were less than five per cent, which require a measurement time between 2-5 days. The detectors were either Ge(Li)- or HPGe (highly purified germanium crystal)- detectors. The relative efficiencies varied between 9 - 15 % and FWHM between 1.8-2.0 keV. A ND6600 multichannel analyzer from Nuclear Data INC. was analysing the pulses from the detectors. Spectra were analysed by a computer program ran on a micro vax computer. This program was produced at the department and gives activity-concentration in Bq kg^{-1} and calculates the error of the activity concentration.

-Alpha spectrometry and determination of ^{210}Po

Alpha spectroscopy was only done on raw material and final material. From the gamma spectroscopy, samples of about 20 g were taken (for reindeer fillet only 6 g remained after soaking process and drying). These were wet ashed with nitric and perchloric acid. Polonium was spontaneous deposited onto nickel discs, ^{209}Po was used as yield determinant.

The discs were measured with surface barrier detectors from ORTEC. The detectors had a 300 mm^2 surface area. Pulses from the detector were analysed by a ND 6600 multichannel analyzer. Analysing time was determined from the total number of counts, in order to keep the error less than one standard deviation. From ND 6600 you get the total number of counts in the peak, from which the background must be subtracted to get the net counts. The background is updated every month for respective detector. Activity concentrations can be calculated as follows:

$$\frac{A_{210}}{A_{209}} = \frac{Count_{netto,210}}{Count_{netto,209}} \Leftrightarrow A_{210} = \frac{Count_{netto,210}}{Count_{netto,209}} \cdot A_{209} \quad ; \quad A_{209} = m_{yield} \cdot A_{K,\,yield}$$

$$A_{K,\,210} = \frac{A_{210}}{m_{sample}} \Rightarrow A_{K,\,210} = \frac{\left(\frac{Count_{netto,210}}{Count_{netto,209}}\right) \cdot m_{yield} \cdot A_{K,\,yield}}{m_{sample}}$$

there : A_{210} is total activity of ^{210}Po in the sample, [Bq]
A_{209} is total activity of ^{209}Po in the sample, [Bq]
$A_{K,\,210}$ is activity concentration of ^{210}Po in the sample, [Bq/kg]
$Count_{nett,\,210}$ is nett counts from ^{210}Po peak
$Count_{nett,\,209}$ is nett counts from ^{209}Po peak
m_{yieild} is the mass of the radiochemical yield determinant put in the sample, [g]
$A_{K,\,yield}$ is activity concentration of ^{209}Po in the yield solution [Bq/g]
m_{sample} is the mass of the sample [kg]

-Stable element analysis

Two samples from the Reindeer liver (before and after soaking) were sent to Svensk Grundämnesanalys AB for analysis of stable nuclides. This was done by analyse method M3 plus Hg-determination[9].

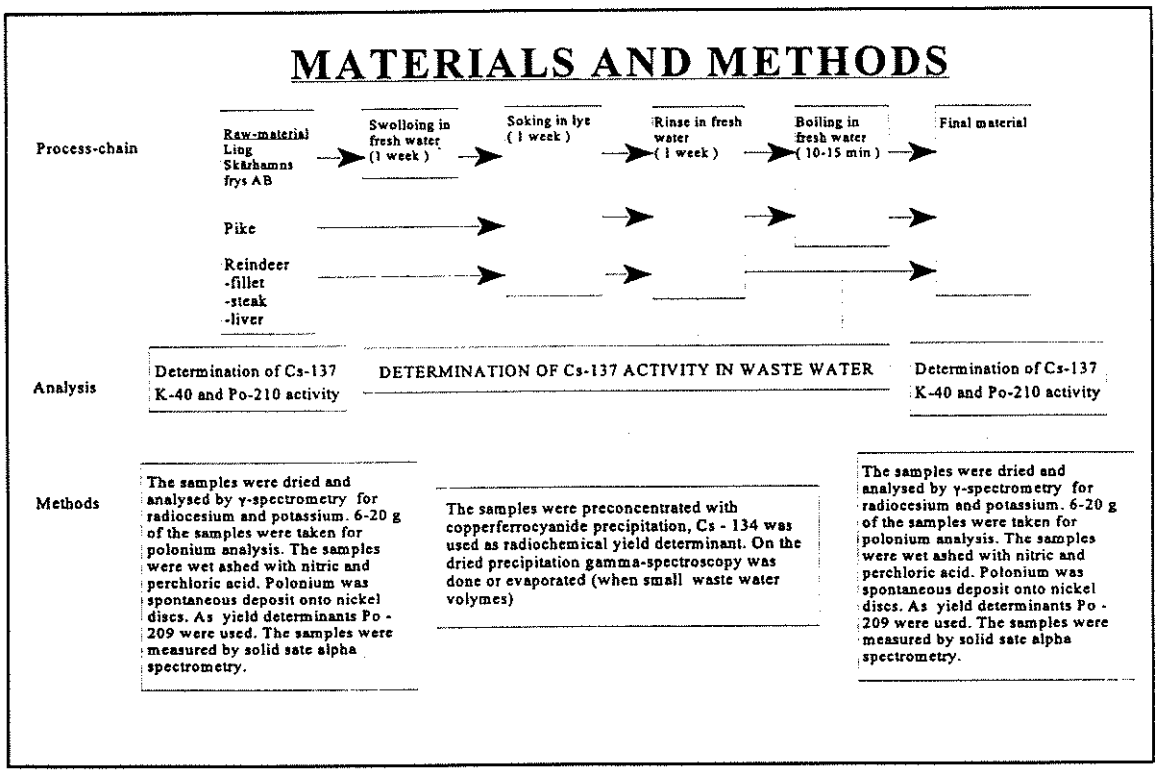

Figure 1. Flow scheme over material and methods

-Dose comparison before and after soaking

To calculate the doses, ICRP 30 was used. In this publication values for ALI_{oral} (Annual limits on Intake, oral), listed below, were used to get a dose factor $k_{dos,i}$, where i is the corresponding nuclide.

$$k_{dos,i} = \frac{50}{ALI_{oral,i}} \quad [mSv/Bq]$$

$$ALI_{oral,^{137}Cs} = 4 \cdot 10^6 \ Bq$$
$$ALI_{oral,^{40}K} = 1 \cdot 10^7 \ Bq$$
$$ALI_{oral,^{210}Po} = 1 \cdot 10^5 \ Bq$$

This factor was multiplied with the activity concentration, to get the dose per kg of oral intake. This was done both before and after processing:

$$D_{before} = \sum_i k_{dos,\,i} \cdot A_{conc\ before,\,i} \quad [Sv/kg]$$

$$D_{after} = \sum_i k_{dos,\,i} \cdot A_{conc\ after,\,i} \quad [Sv/kg]$$

The Dose Reduction Level is then calculated as:

$$Dose\ Reduction\ Level\ (DRL) = \left(1 - \left(\frac{D_{after}}{D_{before}}\right)\right) \cdot 100\ \%$$

Results

Ling

a, soaking effect

Ling	^{137}Cs [Bq/kg]	^{40}K [Bq/kg]	^{210}Po [mBq/kg]
Activity before treatment	2.42±0.14	275±26	238±15
Activity after treatment	0.04±0.16	-5±4	2030±45
Reduction level	98%	100%	increase

b, dose calculation

D_{before} = 1.52 ·10^{-3} mSv/kg
D_{after} = 1.02 ·10^{-3} mSv/kg
Dose Reduction Level =33 %

Ling from Skärhamns Frys AB

a, soaking effect

Ling	^{137}Cs [Bq/kg]	^{40}K [Bq/kg]	^{210}Po [mBq/kg]
Activity before treatment (with scale)	1.51±0.49	477±22	2451±50
Activity before treatment (fillet)	2.96±0.55	426±19	316±18
Activity after treatment	1.8±1.3	7.1±2.6	2101±46
Reduction $level_{scale}$	increase	99%	14 %
Reduction $level_{fillet}$	40%	98%	increase

b, activity in rinse process

1.58 mBq/litre (from a drain-tank which maby have been diluted by water)

c, dose calculation

D_{before} = 3.63 $\cdot 10^{-3}$ mSv/kg (with scale) and 2.32 $\cdot 10^{-3}$ mSv/kg (fillet)
D_{after} = 1.11 $\cdot 10^{-3}$mSv/kg
Dose Reduction Level = 69% (with scale) and 52% (fillet)

Pike

a, soaking effect

Pike	^{137}Cs [Bq/kg]	^{40}K [Bq/kg]	^{210}Po [mBq/kg]
Activity before treatment	95.8±3.5	524±20	793±28
Activity after treatment	19.3±7.9	46.8±7.9	465±22
Reduction level	80%	91%	41%

b, distribution of the soaked activity (only for ^{137}Cs)

	content	percent of the total soaked activity
lye solution:	15.5±3.9 Bq	47%
rinse solution:	16.4±4.0 Bq	52%
boiling water:	0.3±0.5Bq	1%

c, dose calculation

D_{before} =4.21 $\cdot$ 10^{-3} mSv/kg
D_{after} = 7.08 $\cdot 10^{-4}$ mSv/kg
Dose Reduction Level =83 %

Reindeer fillet

a, soaking effect

Reindeer fillet	^{137}Cs [Bq/kg]	^{40}K [Bq/kg]	^{210}Po [mBq/kg]
Activity before treatment	1825±72	386±26	8339±91
Activity after treatment	349±18	1150±34	2495±50
Reduction level	81%	increase	70%

b, distribution of the soaked activity (only for ^{137}Cs)

	content	percent of the total soaked activity
lye solution:	92±9 Bq	87%
rinse solution:	13±4 Bq	13%

c, dose calculation

D_{before} $=2.89 \cdot 10^{-2}$ mSv/kg
D_{after} $=1.14 \cdot 10^{-2}$ mSv/kg
Dose Reduction Level $=61\%$

Reindeer steak (soaked with NaOH)

a, soaking effect

Reindeer steak	^{137}Cs [Bq/kg]	^{40}K [Bq/kg]	^{210}Po [mBq/kg]
Activity before treatment	1017±32	616±25	11140±105
Activity after treatment	19±4	not measured	10286±101
Reduction level	98%	-	7.6%

b, build up activity in the lye solution

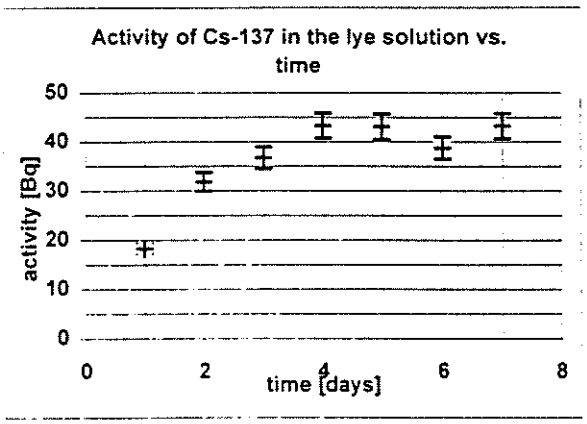

Diagram 1

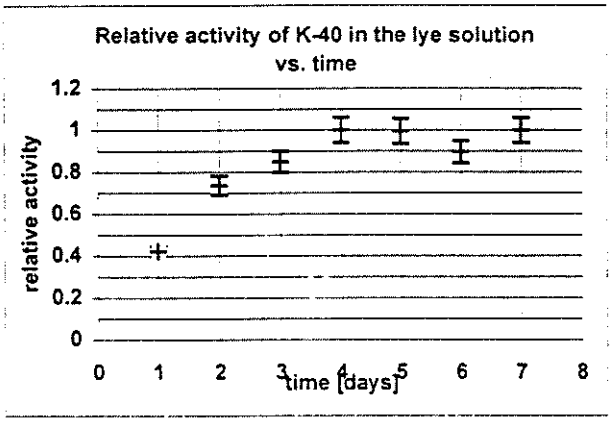

Diagram 2

c, activity in the rinse water

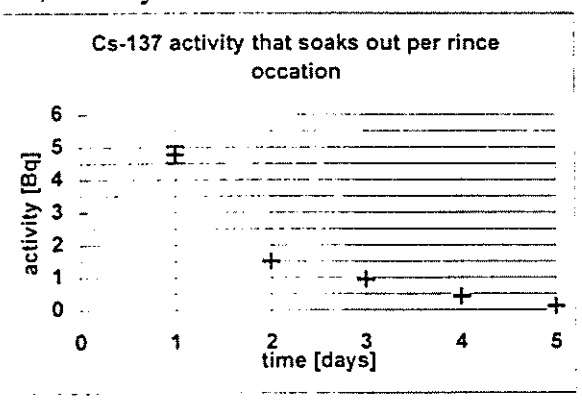

Diagram 3

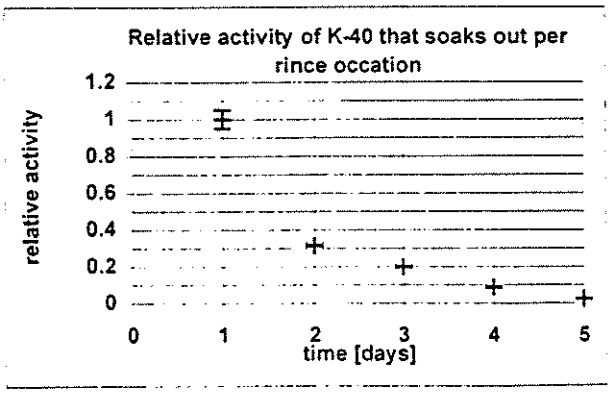

Diagram 4

d, dose calculation

D_{before} $=2.14 \cdot 10^{-2}$ mSv/kg
D_{after} $=5.38 \cdot 10^{-3}$ mSv/kg (exclude ^{40}K)
Dose Reduction Level $=75$ %

Reindeer steak (soaked with potash)

a, soaking effect

Reindeer steak	^{137}Cs [Bq/kg]	^{40}K [Bq/kg]	^{210}Po [mBq/kg]
Activity before treatment	1017±32	616±25	11140±105
Activity after treatment	196±14	477±22	8148±90
Reduction level	81%	23%	27%

b, dose calculation

$D_{before} = 2.14 \cdot 10^{-2}$ mSv/kg
$D_{after} = 8.91 \cdot 10^{-3}$ mSv/kg
Dose Reduction Level =58 %

Reindeer liver

a, soaking effect

Reindeer liver	^{137}Cs [Bq/kg]	^{40}K [Bq/kg]	^{210}Po [mBq/kg]
Activity before treatment	75.2±8.6	1763±42	8615±94
Activity after treatment	13.27±0.66	388±20	10330±102
Reduction level	82%	78%	increase

b, dose calculation

$D_{before} = 1.41 \cdot 10^{-2}$ mSv/kg
$D_{after} = 7.10 \cdot 10^{-3}$ mSv/kg
Dose Reduction Level =50 %

c, stable element analysis

Element	contents before soaking $[mg/kg]_{dw}$	contents after soaking $[mg/kg]_{dw}$
Al	43 ±15	139 ±9.4
As	<17	<33
Ba	<2.8	<5.5
Ca	190.43 ±0.35	53012 ±150
Cd	<2.8	<5.5
Co	<56	<109
Cr	<2.8	<5.5

Cu	96.10 ±0.70	153.1 ±1.7
Fe	290.4 ±1.4	427.9 ±3.5
Hg	59.1 ±3.2	100.5 ±3.0
K	6688 ±21	596 ±35
Mg	637.1 ±3.5	756 ±12
Mn	13.34 ±0.46	34.51 ±47
Mo	7.60 ±0.70	12.2 ±1.1
Na	2326.7 ±7.0	1117.0 ±5.9
Ni	<11	<22
P	12214 ±49	26420 ±170
Pb	<28	<55
Si	15.4 ±1.9	216.3 ±4.1
Ti	<2.8	7.45 ±0.76
V	<2.8	<5.5
Zn	117.22 ±0.70	252.4 ±1.2

Results from other countermeasurments

Some Remedial Measures / Countermeasures

Author	Method(s)	Results
A.Andersson-Sørlie, P.Strand and E.Bøe[5]	Dietary advice to special groups of the population to reduce Cs intake	40-80% did take precautions. Farmers /hunters reduce the intake with 25%. Some Lapps up to 90%
L.Håkanson and T.Andersson[3]	Liming lake and wetland to reduce Cs contents in fish	After 3 years there was no significant reduction of radio cesium in lake fish.
E.Petäjä,A.Rantavaara, O.Paakkola and E.Puolanne[4]	Soaking meat and fish in brine and water	Best result was a reduction of 90% of coarsely ground meat, with was soaked 17 h in running water.200g meat piece, 5% brine for 2+2 days reduce to about 80%

Discussion

-Soaking foodstuff with lye as a counter measurement

As can be seen from the results, this is an effective method to reduce radio nuclides as ^{40}K and ^{137}Cs. This was expected as these alkali-elements are present as ions and when the cell membrane is broken down the ions may easily diffuse out in to the lye solution. The extraction proceeds until a concentration equilibrium between the meat and the solution has occurred. The fast build up of ^{137}Cs and ^{40}K activity in the lye solution can been seen in diagram 1 and 2. After four days an equilibrium situation is obtained. This equilibrium occurs either because the lye solution's "dissolution

constant", (maximum of what the solution can carry of the nuclide), have been reached or because concentration-equilibrium between meat and solution have been reached. From diagram 3 and 4 we can understand that in this case it is because of a concentration equilibrium. Otherwise, we would have a more constant soaking effect (straight line) instead of an exponential decreasing. From this we would expect a better result if the meat/solution mass ratio were lower. From a countermeasurment point of view it would be sufficient with a short time (one day) in the lye solution and thin pieces of meat, so that the cell membrane is broken down. After that the meat pieces are rinsed in fresh water for some time. This could reduce att least the alkali radio nuclides sufficiently.
The results of the extraction for Cs and K is between 80-100 % reduction and for Po 0-70 %

-Differences in soaking different tissue
For the alkali metals (^{137}Cs and ^{40}K) the results are similar between different tissues and for the heavier elements no conclusion has been taken. The biggest change in shape and form was for dried ling. From 1 kg dryed ling we got about 7 kg lutefisk. 7 kg of lutefisk have a dry weight of 0.45 kg. On the other hand the Reindeer liver almost kept it's shape under the soaking process. Here the main effect was a big extraction of blood from the liver and a build up of a terrible smell from the sample. The steak and fillet changed colour but smell were unchanged.

-Differences in soaking with Sodiumhydroxide or Potash
There is a better result when soaking with NaOH than Potash for ^{137}Cs. This can be an effect of that the hydroxide-solution has a higher concentration of OH-ions for NaOH than the Potash. In the Potash there was also some small content of Cs from fallout.

-Similarity between Potassium and Caesium
If the ratio ^{137}Cs-/^{40}K- extraction (relative extraction) is plotted against time, we get straight lines, (both for the build up in the lye solution and for the rinse water) see diagram 5 and 6 below, with a constant value of 1.This means that the isotopes have same extraction velocity. Of this we can understand that Cs and K are acting almost in the same way in the tissue. As have been said before potassium is inside the cell membrane and in ionic form. We can now expect the same for Cs. If Cs had been outside the membrane the ratio curve would not been a constant as the extracellular fluid has a faster extraction.

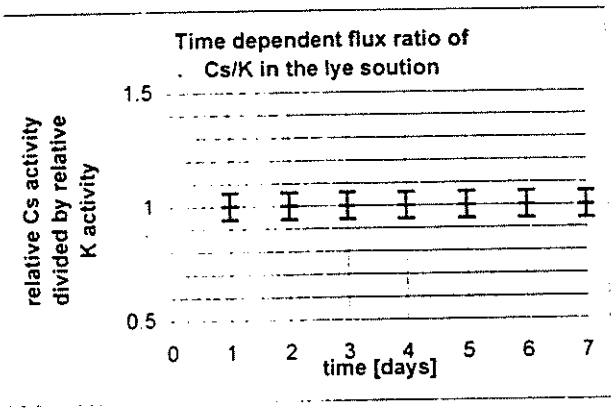

Diagram 5

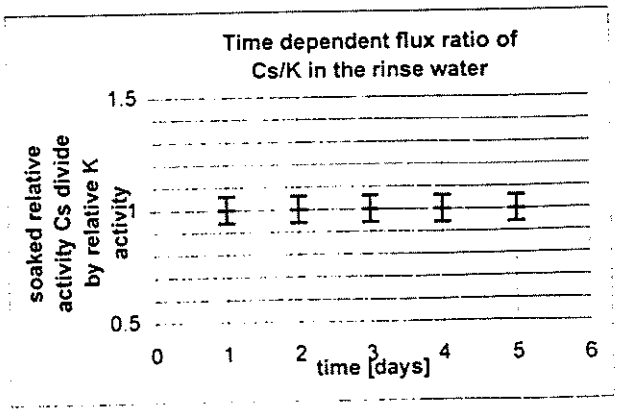

Diagram 6

-Alfa-emitting nuclide extraction when soaking

There is a big difference in reduction of ^{210}Po when soaking, (from no reduction at all, to seventy per cent). The differences are of two types. First there is a difference between dried and fresh material. Secondly, there is a difference caused from a bad result from the wet aching process. With bad means that the decomposing of the sample was not total. If the ^{210}Po is attached "inside" a protein which is not totally broken down, there could be an error in the calculated content, as this is not effecting the yield determinator, ^{209}Po. The result of this is a lower concentration of ^{210}Po.

From the results it is also seen that most of the polonium is sitting on the scale of the fish. The Lutefisk from Skärhamns Frys AB was a mixture of Lutefisk with and without scale, this can explain why the reduction was not total.

-Stable element analysis of the liver

The results show that almost all elements have higher concentrations after soaking. This can be an effect of that the liver contain much blood at the first measurement, which was rapid diluted into the lye solution. The blood is expected to have a low concentration of the elements measured. If it was possible to bring out the blood before the first measurement, the concentration of the elements would have a higher value and the increase in concentration when soaking would be lower. Another effect is that the chemicals used may have some of the elements in their content. These can diffuse in to the liver and combine with molecules. The two alkali elements, K and Na, are the only ones which have lower concentration after soaking. This may be an effect of that they are presented as ions in the cellar structure and then easy soaked out with diffusion. K has a higher reduction than Na, as an effect of that the lye was NaOH. When the liver is exposed to the lye it gets a very high concentration of sodium. When rinsed in water, the same percentage share of alkali elements are expected to diffuse out, whereby the reduction of Na is smaller, which agrees with the results.

Dosereduction

The low content of activity in the samples are of no danger, from a radiological point of view. For example it is necessary to eat 240 kg Pike, 660 kg Ling or 45 kg Reindeer to get 1 mSv, (raw sample), which is almost equal to the background radiation. The dose reduction, when soaking the foodstuff is 50 - 100 %. As seen from the results Polonium is responsible for a big part of the dose. If we only look at ^{137}Cs and ^{40}K the dose reduction would be 80 -100%.

Conclusion

This method is very good to reduce the alkali metals, as the cell membrane is broken down and the ions can move freely out in to the lye solution. It is shown that Caesium and Potassium have closely chemical similarities in the cell function or at least are located inside the cell. The results for Polonium shows that it is not in ion-form, and probably attached on proteins. As the whole soaking process is quite time consuming, it is not sutiable as an "every day" countermeasure of foodstuff, even thought it is, with some modification of the recipe (discussed above) it could be an excellent method to reduce dose to man. From this experiment can be said that it is safe, at least in a radiological point of view, to eat soaked foodstuff whit lye.

It could also be usable in practice as a method to make high-protein feedstuff out of large qantities contaminated meat.

Acknowledgments
I wish to acknowledge Skärhamns Frys AB for the kindness to let us take samples from the soaking process and for interesting information about Lutefisk manufacturing. I would also like to thank the staff in radioecology group at the Department of Radiation Physics.

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RaMona, Radiation Monitoring and Alarm System for the IFE-Kjeller Site

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Sammendrag Et system for kontinuerlig overvåkning av strålenivå og luftaktivitet i- og omkring nukleære installasjoner på IFE-Kjeller beskrives. Systemet, som for tiden er under oppgradering, baseres på computer-styrt utlesning og analyse av måledata. Formålet med systemet er å holde stråledosen til personalet så lav som mulig og å kontrollere utslipp til omgivelsene. Måledata fra overvåkningssystemet skal dessuten brukes som grunnlag for doseberegninger og spredningsmodellberegninger.

Introduksjon

På IFE's område på Kjeller er det samlet flere anlegg som behandler radioaktivt materiale. Det er derfor nødvendig å gjennomføre kontinuerlig overvåkning av strålenivå og luftaktivitet i- og omkring disse installasjonene.

Virksomheter hvor det antas å kunne oppstå situasjoner av radiologisk betydning er:

Isopharma AS (tidligere Isotoplaboratoriet-IFE) som produserer og analyserer radio-farmaka. Aktuelle radionuklider er først og fremst ^{131}I, men ^{125}I og ^{99m}Tc inngår også i produktutvalget. Det utføres kontinuerlig overvåkning av samtlige laboratorier som behandler radioaktivt materiale og av avgass fra bygningen. Totalt antall overvåknings-punkter er 30.

Forskningsreaktoren *JEEP II* benyttes til produksjon av ^{131}I, doping av Si og nøytron-spredningsfysikk. Anlegget overvåkes med hensyn på luftaktivitet, ekstern γ- og nøytron-stråling og utslipp til omgivelsene via ventilasjonsanlegget, totalt 24 punkter. I utslippssammenheng er nuklidene ^{131}I og ^{41}Ar mest interessante.

På *Metallurgisk lab.II* utføres re-instrumentering av brukt brensel i samarbeid med OECD Halden prosjektet. Det utføres kontinuerlig overvåkning av luftaktivitet, avgass og ekstern γ-stråling i bygningen, totalt finnes 11 monitoreringspunkter.

Avfallsanlegget utfører behandling og lagring av radioaktivt avfall. Luftaktivitet, avgass og ekstern stråling monitoreres kontinuerlig. Totalt er det etablert 29 punkter fordelt på avfallsbehandling og lagerbygg.

Det planlagte kombinerte lager og deponi for lav- og middelaktivt avfall i *Himdalen* vil også kreve kontinuerlig overvåkning av strålenivå. Anlegget vil være ubetjent, monitoreringen vil skje fra Kjeller. Detaljer i måleprogrammet er ikke fastsatt.

Hovedformålet med kontinuerlig overvåkning av romluft og strålenivå er holde stråledosen til personalet så lav som mulig. Ved hendelser som fører til en akutt forhøyning av strålenivå skal lokalene evakueres umiddelbart. Overvåkingssystemet aktiverer en alarm så hurtig som mulig når strålingsnivået overskrider en viss grenseverdi.

Under normale driftsforhold vil et kontinuerlig overvåkningssystem kunne gi informasjon om hvordan strålenivået kan relateres til ulike arbeidsoperasjoner. Dette vil være av stor betydning ved vurdering av dosereduserende tiltak.

Overvåkning av utslipp fra bygningen utføres for å dokumentere at utslippene ligger innenfor de grenseverdier som er fastsatt av myndighetene. For IFE er grenseverdiene:

- 100 µSv/år effektiv dose fra utslipp til atmosfæren til enkeltindivid i mest utsatt befolkningsgruppe
- 10 µSv/år fra jodisotoper til enkeltindivid i mest utsatt befolkningsgruppe
- 0,5 mannSv/år kollektivdose

Monitorering av utslipp vil også gi grunnlagsdata for sprednings- og deponeringsmodeller.

Monitoreringssystemet RaMona

En prinsippskisse for monitoreringssystemet er vist i *Fig.1*. Lokalt i den enkelte bygning styres utlesningen av detektorer av en PC. Samtlige detektorer avleses samtidig, én gang per minutt, dette kan imidlertid varieres etter behov.

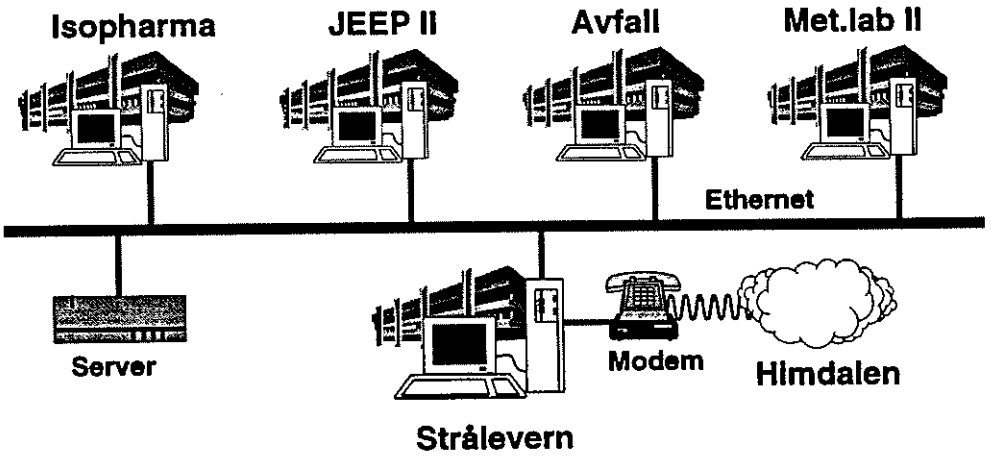

Fig.1 Skisse over kontinuerlig overvåking av strålenivå på IFE

Måledata presenteres på lokal PC slik at personalet har mulighet til å følge variasjon i strålenivå. Aktivering av alarmsystem for bygning og utsendelse av melding til alarmsentral styres av lokal PC.

Lokal PC er koplet til instituttets datanett. Det er dermed mulig å utveksle måledata mellom lokale enheter og sentral PC plassert ved avdeling Miljø- og strålevern. Utvekslingen skjer ved at alle måledata lagres på felles filtjener. De lokale PC'ene fungerer som selvstendige enheter og kopi av alle måledata lagres lokalt. Bortfall av filtjener vil dermed ikke ha noen betydning for alarmsystemets funksjon.

Den sentrale PC brukes av strålevern for å holde oversikt over strålenivå i alle anlegg. Her vil det også være tilgang til statistikk over variasjon i strålenivå over tid. Dette vil også tjene som dokumentasjon av utslipp fra virksomhetene.

Det planlagte kombinerte lager og deponi for lav- og middelaktivt avfall i Himdalen foreslås knyttet til system med en modem forbindelse. Måledata kan dermed kunne overføres jevnlig.

Utstyret for måling av aktivitet i luft er vist i *Fig.2*. Det er etablert sugepunkter i samtlige laboratorier hvor det arbeides med åpne radioaktive kilder. Sugepunktene er plassert ved ventiler for utluft fra rommet og i nærheten av arbeidsplass. Studier av luftstrøm viser at plassering i utluftkanal gir de mest representative resultater, det viser seg likevel at aktivitet i prøveluft kan variere fra 1 til 1/10 av aktivitet i den luft som faktisk pustes inn [1].

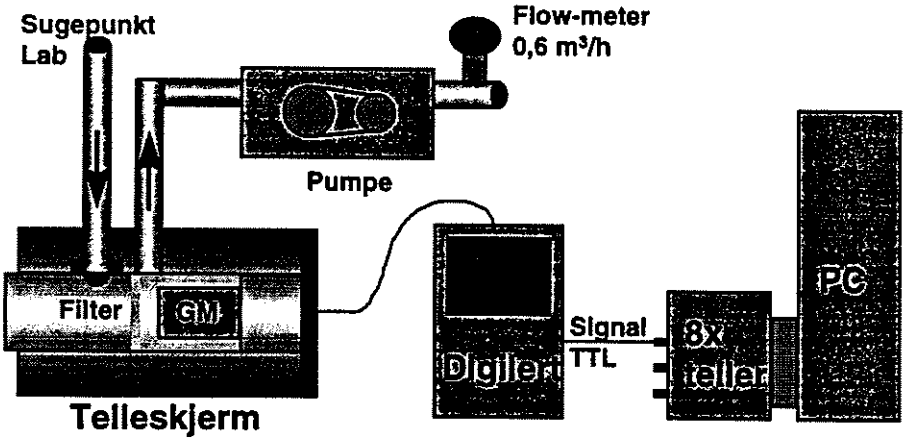

Fig.2 Detektorenhet for måling av aktivitet i romluft

Prøveluften suges gjennom et fileter som er plasser i en telleskjerm. Filteret som benyttes et papirmembranfilter som er satt inn med aktivt kull i de tilfeller det dreier seg om jod-isotoper. Filteret er plassert i umiddelbar nærhet av et GM rør koplet til et lavkost

måleinstrument. Signalene fra instrumentet er koplet til PC ved hjelp av et kommersielt tilgjengelig digitalt tellerkort [2], hvert kort betjener 8 detektorer.

Programvaren er utviklet i LabVIEW [3], et grafisk dataflyt språk spesielt beregnet på instrumentering. Det er lagt vekt på modularitet og mulighet for gjenbruk av programelement, systemet kan dermed lett tilpasses andre målesituasjoner og sensorer.

Alarmgrense

Alarmgrensen kan settes individuelt for det enkelte laboratorium, det er dermed mulig å ta hensyn til ulike ventilasjonsforhold og nuklidesammensetning. Generelt er alarmgrensen satt slik at en signifikant forhøyning av strålenivået gir alarm..

Operasjoner som forventes å oppstå med en viss hyppighet skal ikke gi større bidrag til total effektiv dose enn 500 μSv, dette gir et øvre tak for alarmgrensen. Hvis vi tar ^{131}I i gassfase som eksempel er effektiv dose koeffisient $2{,}0 \cdot 10^{-8}$ Sv/Bq [4]. Dette tilsvarer en alarmgrense på 2800 CPM under typiske forhold.

Doseberegning

Måledata fra overvåkningssystemet vil tjene som et supplement til beregning av dose til personalet. Resultatet vil sammenholdes med TLD persondosimeter og rutinemessig interndosimetri (thyroidea måling, helkroppsmåling og urinanalyse).

Bidrag til effektiv dose vil gjøres direkte tilgjengelig ved at effektiv dose koeffisient for de mest aktuelle isotoper legges inn i systemet. Beregning av dose til enkeltorganer vil bli utført med LUDEP [5] som er basert på ny modell for respiratorisk trakt [6].

Spredningsmodell

Hvis det skulle oppstå uhell som gir opphav til utslipp av radioaktivt materiale til atmosfæren vil det være nødvendig å ha tilgang til en modellberegning av utbredelsen av utslippet. Måledata fra monitoreringssystemet vil sammen med enkle meteorologiske observasjoner gi et grunnlag for slike beregninger. Observasjon av vindstyrke og retning vil gjøres kontinuerlig ved hjelp av en vindmåler plassert på et representativt sted på området.

Modellen som vil bli benyttet er en enkel Gaussisk spredningsmodell med korreksjon for effekter av bygningsmasse. Slike korreksjoner er nødvendige siden alle utslippspunkt ligger under 2,5 ganger bygningshøyde) [7,8]. Utbredelsen av radioaktivitet skal presenteres som aktivitetskonsentrasjonskonturer på et områdekart.

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Probabilistic assessment of doses and depositions after a hypothetical accident at the Barsebäck nuclear power plant, using two years' hourly meteorological data.

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Abstract

In 1995 the Swedish Radiation Protection Institute (SSI) was asked by the Governmental Energy Commission to investigate doses and ground deposition following hypothetical accidents at all four Swedish nuclear power plants. For this investigation the Swedish Nuclear Power Inspectorate published figures on source terms for "typical" accidents with a melted core, covering the range from a fully functioning consequence mitigating system to no mitigation at all.

This study uses these source terms combined with hourly meteorological data from Sturup for a full 2-year period in a probabilistic study (PSA 3) of external doses and ground deposition after a hypothetical accident at Barsebäck. This is the first time a true, although limited, level 3 PSA has been performed for a Swedish power plant.

Background

In 1995 the Swedish Radiation Protection Institute (SSI) was asked by the Governmental Energy Commission to investigate doses and ground deposition following hypothetical accidents at the four Swedish nuclear power plants. For this study the Swedish Nuclear Power Inspectorate published source terms for "typical" accidents with a melted core, covering the range from a fully functioning consequence mitigating system to no mitigation at all. Using these source terms, calculations for the four power plant locations were performed. The results were published as part of the Report of the Energy Commission [1] and as an SSI report [2]. A comprehensive summary of earlier Swedish studies of consequences of nuclear accidents is given in [1].

Meteorological time series for the Swedish power plant locations were not available to the SSI in 1995. Semi-probabilistic calculations were performed based on data extracted from time integrated meteorological values. For the present study SSI has received hourly data from the Swedish Meteorological and Hydrological Institute, covering two years at the Sturup Airport. This made it possible to make probabilistic calculations for the Barsebäck power plant. The calculations in 1995 were made with short notice and a tight time schedule. Due to this, a slightly incorrect population data base had to be used. This data base has now been updated.

The primary objective of the present study was to provide probabilistic consequence calculations that may be used when a final decision on the Swedish nuclear power programme will be taken. A secondary objective was to find out how the use of better meteorological data and a true probabilistic treatment influence the results of the calculations.

Source terms used

Three different source terms, described in [1], called "realistic", "nominal" and "rest risk", have been used. The "realistic" source term corresponds to a release when all mitigating systems function well. The "nominal" source term corresponds to a situation when these systems work enough to fulfil the design criteria but no more. The "rest risk" source term,

finally, is envisaged to give the maximum release deemed possible in the Barsebäck reactor. This is thus a highly improbable situation, with no mitigation. The first two source terms contain a first fast release of noble gases, and a delayed and prolonged release of the depositing nuclides. In the "rest risk" case all activity is released in three hours. The released fractions for the three source terms are shown in the table below.

	Fraction of core inventory that is released for the three source terms		
	realistic	nominal	rest risk
Noble gases	1.0	1.0	1.0
Cs, I, Rb, Sb, Te	$2 \cdot 10^{-6}$	$1 \cdot 10^{-3}$	$1 \cdot 10^{-1}$
Sr, Ru, Ba, Co, etc.	$4 \cdot 10^{-7}$	$2 \cdot 10^{-4}$	$1 \cdot 10^{-2}$
Zr, La, La, Pu etc.	$4 \cdot 10^{-8}$	$2 \cdot 10^{-5}$	$1 \cdot 10^{-3}$

No probabilities for the source terms above have been published.

Computer code, parameters

The calculations have been performed using a probabilistic version of the code LENA 3, called LENAP3. Earlier versions of this code are described in [3]. Version 3 differs mainly in the graphic interface and in some minor corrections such as an updated cloud dose correction procedure. A major difference exists, however, between LENAP3 on one side and earlier versions of the probabilistic code and the code used for the calculations for the Energy Commission on the other in the treatment of releases with prolonged duration. In LENAP3 the consequences of a release during several hours is calculated as the sum of a number of consecutive hourly releases. For each of these hourly releases, the initial wind direction and atmospheric stability are assumed constant along all the path. The other meteorological parameters (mixing layer height, wind speed, precipitation intensity) are, however, given new values every hour, as read from the data file.

For the calculations presented here the parameters of [1] were used. Thus for rural populations a common shielding factor of 0.3 was used for inhalation, cloud and ground doses. For urban populations, the cloud and ground shielding factors were lowered according to the size of the population centres. The lowest value (0.1) was used for inhabitants of cities with more than 100.000 inhabitants.

The following deposition parameters were used:

Dry deposition	cm/sec
iodine, non depositing	0,01
iodine, other	1,0
all other nuclides	0,1

Wet deposition	fraction / sec
iodine, non depositing	$2 \cdot 10^{-6}$
all other nuclides incl. iodine	$1 \cdot 10^{-4}$

The dose conversion factors are mainly from [4].

Here one may note that the deposition parameters for non depositing forms of iodine, as well as the amount produced and the dose factors, are extremely poorly known. The doses esti-

mated from non depositing iodine could therefor be in error by as much as a factor of 100. On the other hand, there are reasons to believe that only very minute amounts of non depositing iodine will be produced in an accident in a Swedish reactor.

The meteorological data set

The basis for the probabilistic treatment is a set of meteorological data from the meteorological station at Sturup Airport, complemented by radio sond data from Kastrup Airport. The time series that covers every hour for the period April 1989 to March 1991, contains wind direction, wind speed, atmospheric mixing layer stability classified as Pasquill B to F, mixing layer height, 12 or 24 hours integrated precipitation and some parameters that make it possible to calculate the precipitation for every hour. The time period can be regarded a grab sample.

End points

The collective effective dose equivalent is calculated for 24 hours and for 50 years, out to 200 and 2000 km. From these results it is possible to estimate probabilistic consequences. The results presented does not allow a calculation of acute injuries or deaths. No proper agricultural data base was available to the author, and therefor no intake doses have been calculated. (The EU-sponsored code COSYMA does contain such a data base. Large parts of Scandinavia and East Europe are, however, missing.)

Reasonable countermeasures do only to a limited extent affect the external collective doses. In a simple modelling of possible effects of evacuation and relocation, it has been assumed that individual doses above 200 mSv could be avoided.

Results

Below are shown CCDF curves (Complementary Cumulative Density Functions) for several calculations. It is usual to show CCDF curves with logarithmic x- and y-axes, and this practice is also followed here. Given that an accident occurs, and given the source term, these curves show the conditional probability that the consequences will be larger than the value on the abscissa. The only probabilistic element here is the weather.

From the graphs below it is found that the present calculations result in doses which are 2 - 2.5 times higher than those of [1]. If one corrects for the somewhat larger population in the present data base it remains anyway a factor of about two. The basic explanation to this is the difference between the hourly weather data used in the present work (below called "measured weather)" and the data used in the earlier study ("statistical data"). The following differences are can be identified:

i/ As an average, the measured mixing layers were about 30% lower that the statistical ones for all stability classes. Also, measured mixing layer heights only slightly above the release height were found in a number of cases, whereas in the statistical data such situations were not accounted for.

ii/ There is a difference in wind direction probabilities between the two sets of data: In the measured time series wind from west is more frequent than what was assumed from statistical data. In fact, the probability of winds in the sector covering Copenhagen was directly proportional to the relative sector width. The assumption that "it seldom blows from Barsebäck towards Copenhagen" was certainly not correct during the two years studied in this

report. On the other hand did it only on rare occasions blow in such directions that the nearby deposition mainly occurred on water, which also contributes to higher collective doses.

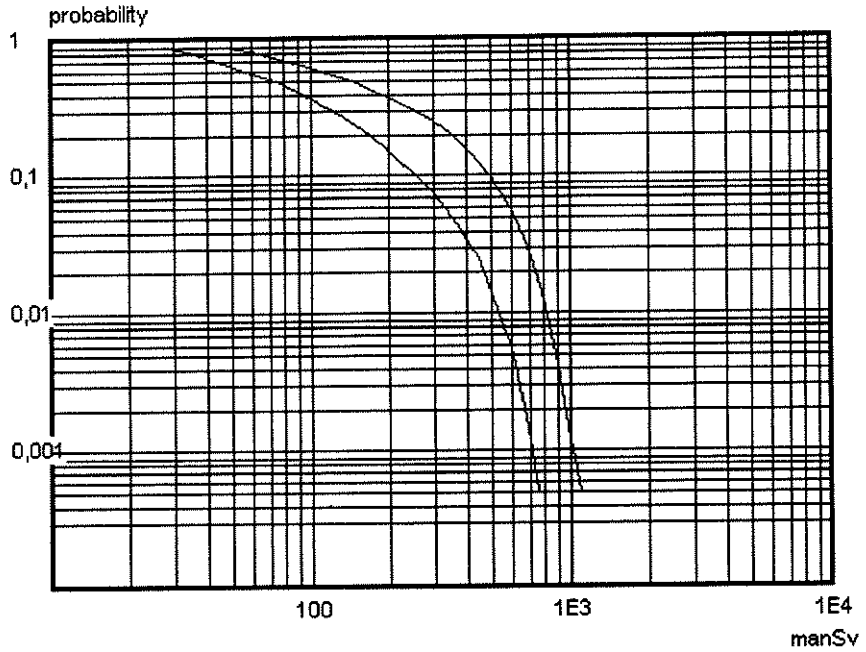

Figure 1. CCDF for a 'realistic' source term from the present study. Upper curve shows collective doses for 50 years out to 2000 km, lower curve out to 200 km, (intake not included). From this figure one finds, for example, that given an accident with the stipulated source term, there is according to the present study a 10% probability that the collective dose over 50 years out to 2000 km will be larger than 500 manSv.

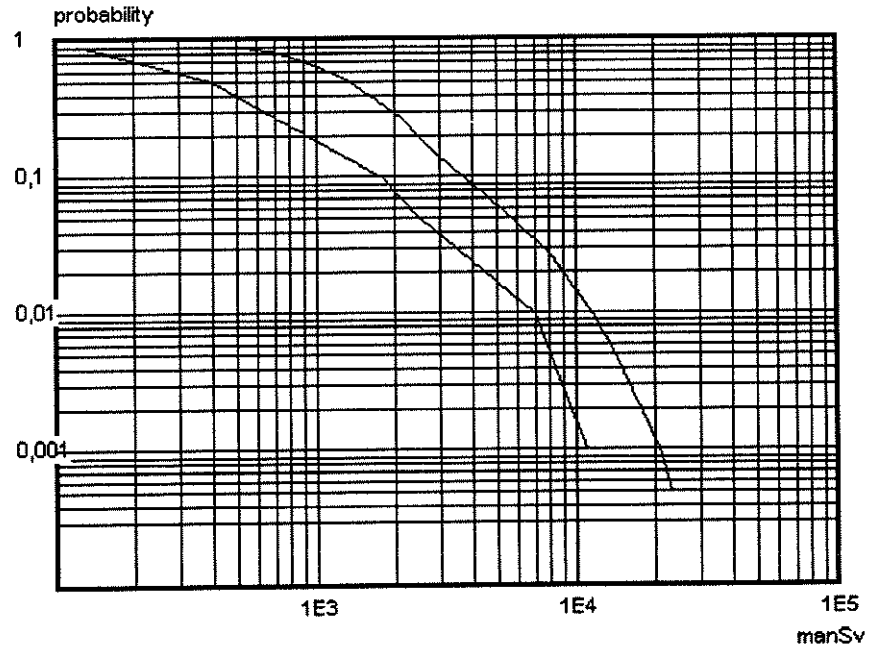

Figure 2. CCDF for a 'nominal' source term. Collective doses out to 2000 km for 50 years, intake doses not included. Upper curve shows results from the present study, lower curve results from [1].

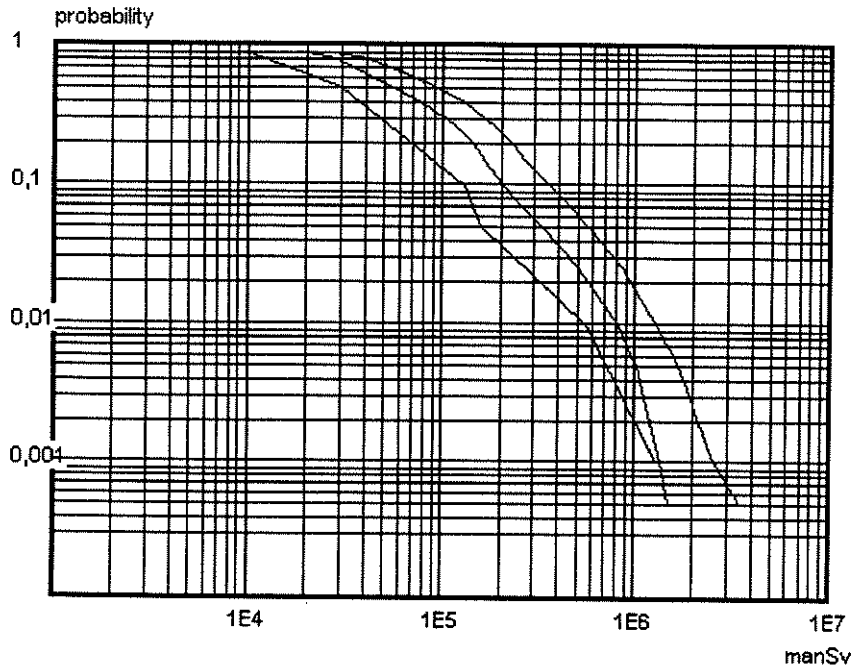

Figure 3. CCDF for a 'rest risk' source term. Collective doses out to 2000 km for 50 years, no intake doses. Uppermost curve shows results from the present study, lowest curve results from [1]. The curve in the middle shows the reduction when all individual doses above 200 mSv are excluded - a simplified evacuation.

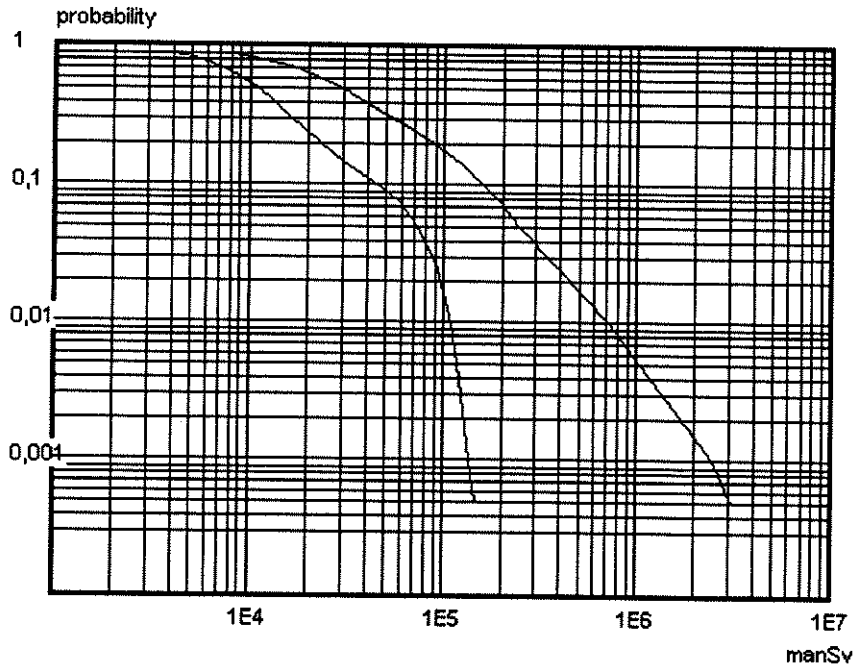

Figure 4. CCDF for a 'rest risk' source term. Collective doses out to 200 km for 50 years, no intake doses. The lower curve shows the reduction when individual doses above 200 mSv are excluded. Here the countermeasures result in a drastic decrease of the collective dose, since the high individual doses at close distances contribute more when doses outside 200 km are excluded.

iii/The measured total precipitation for two years was 1880 mm, which is 15 % more than was assumed in the earlier calculations. The probabilistic effect of this is not evident. However, it does at least increase the collective doses in the low probability end.

Moreover, the measured data set contained some periods of heavy rainfall, with winds blowing especially in the direction of Malmö. This resulted in an increase of the 0.1 % values.

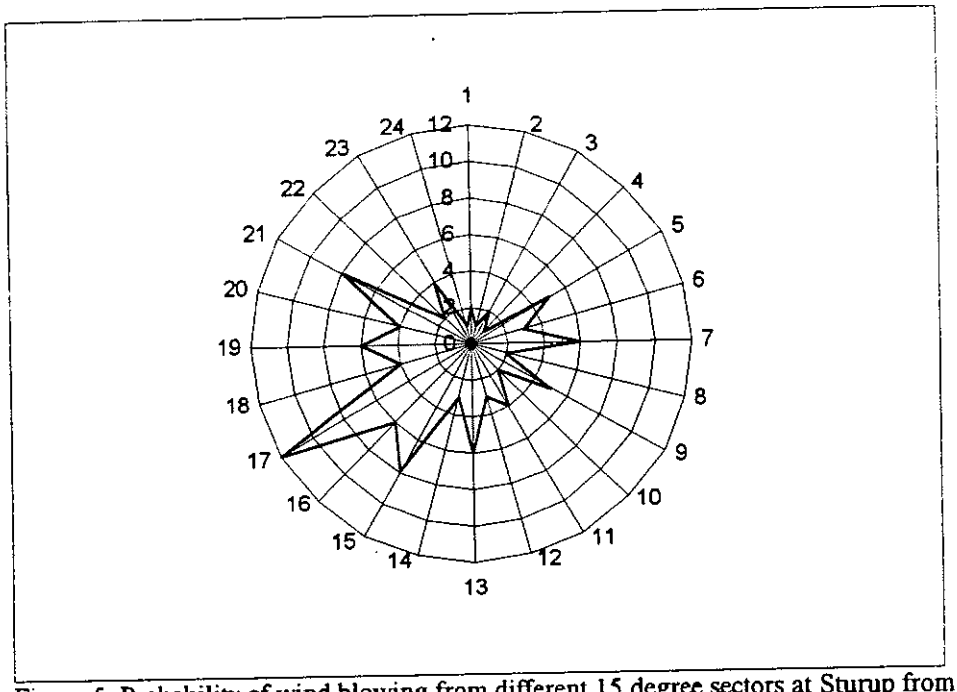

Figure 5. Probability of wind blowing from different 15 degree sectors at Sturup from April 1989 to March 1991.

Extreme values and low probabilities

In this study about 17500 sequences were used for every scenario. This made it possible to calculate a 0.1 percent value for the CCDF-curves, which is less than 20 sequences. Sequences that occur less than 20 times during 2 years are rare, but exist. But what are their "true probabilities"? From the data file one finds that out of the 20 worst situations, almost one third occurred during one day, about 1500 hours from the start. Had the calculations been limited to the first 1500 hours the results would have suggested that the 1% values were those that now are considered the 0.1%. Had, on the other hand, the data file been extended to cover ten years, it might have turned out that the probability of these extreme values was a factor of five lower. But then one gets in trouble anyway, since such a low probability calculated over a ten year period is not valid today. The evident conclusion is that probabilities of the order of 0.1% and below must be regarded with great scepticism.

A closer study of the meteorological data file reveals that on the day mentioned above there was 15 mm precipitation in three hours, 30 mm in twelve hours. All the time it blew 2 metres per second in the direction of Malmö, except for a few hours when it blew against Copenhagen. Due to these circumstances, a very large fraction of the activity was deposited on nearby, densely populated areas. In fact, the situation during these hours may very well be described as the "worst possible".

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Nordic Society for Radiation Protection
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26 - 30 August, 1996

International Intercomparison of Criticality Accident Dosimetry Systems at the Silene Reactor

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Abstract - The Silene Reactor at the Valduc Centre near Dijon in France can be operated to give a short intense neutron pulse. This is very useful for testing criticality accident dosemeters. In 1993 IAEA and EU jointly arranged an intercomparison of criticality accident dosimetry systems at the Silene Reactor. 18 laboratories from 14 countries with about as many different dosimetry systems participated. Each laboratory had the possibility of placing dosemeters at the front of a phantom and in the "air". Liquid from the phantom could be obtained after the irradiation for "biological" dosimetry. The Silene Reactor provided two sets of neutron pulses, one from a bare reactor and one with the reactor shielded with 10 cm of lead. After each neutron pulse the participating laboratories had two days to produce a first estimate of the neutron kerma. The agreement between the results obtained by the different laboratories was quite good.

Introduction

A criticality is characterised by an uncontrolled chain reaction of at least 10^{15} fissions. The radiation from a criticality is a prompt pulse of intense neutron and γ-radiation, followed by capture γ-radiation. It is not very easy to simulate radiation from a criticality, but a few reactors can produce a criticality. One is the Silene Reactor at the Valduc Centre of the Institut de Protection et de Sûrité Nucléaire in France. Only at these special reactors is it possible to test and to get practical training in reading out a criticality accident dosimetry system. Previous intercomparisons of criticality accident dosimetry systems have taken place at Valduc (1970), at Oak Ridge (1971), at Vinca (1973) and at Harwell (1975).
The experiment at Valduc was sponsored by the Commission of the European Communities (CEC) in collaboration with the International Atomic Energy Agency (IAEA) and the US Department of Energy (DoE).

The Silene Reactor

The Silene reactor has a cylindrical core of 36 cm diameter. The fuel is a uranylnitrate solution with 93 % enriched uranium, and 71 g of uranium per litre. A control rod can be moved in and out of the core vertically along the central axis. To simulate the accidental criticality excursion for the experiment, the reactor was operated in a mode called "Free Evolution" for about 2-3 minutes, giving a neutron pulse corresponding to $\sim 2 \cdot 10^{17}$ fissions. To accommodate all dosemeters from the 18 laboratories the reactor was operated to give a total of four pulses, two with a 10 cm lead shield surrounding the reactor and two with a bare reactor.

The Experimental Set-up

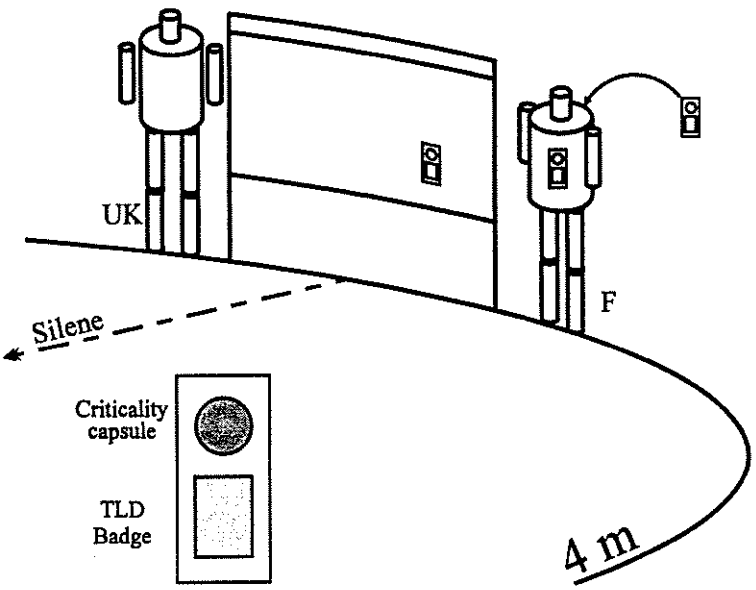

Figure 1. Sketch of the experimental set-up.

Two phantoms and a plastic sheet were placed 4 metres from the centre of the reactor. Personal dosemeters were placed at the front and back of the phantoms, free air dosemeters were placed at the plastic sheet. The Risø-dosemeter as shown in figure 1 consisted of a Harwell Criticality Capsule and a TLD badge.

Dosimetry Systems

The Harwell Criticality Capsule as used by Risø is shown in expanded view in figure 2.

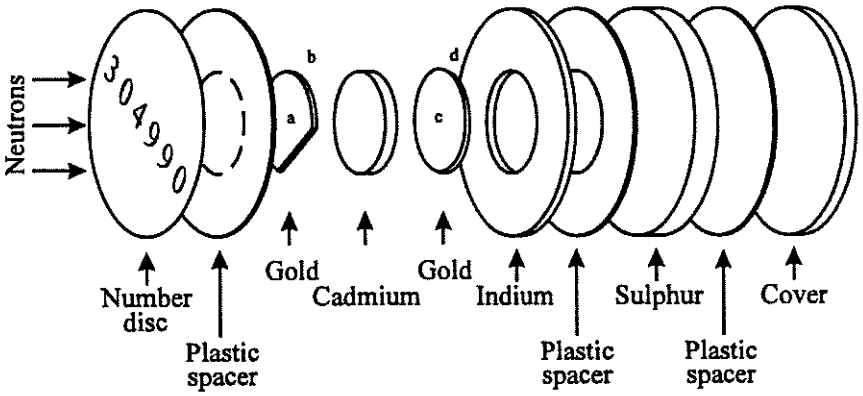

Figure 2. Sketch of a Harwell Criticality Capsule

The Harwell Criticality Capsule is an activation dosemeter for determining the neutron kerma and neutron fluence at different energy intervals. At the experiment at Valduc there were many different versions of activation dosemeters developed at different laboratories.

For γ-detection most laboratories used different thermoluminescent dosemeters (TLD) systems. A number of laboratories had experimental systems. The detector that deviated most from the others was a Superheated Drop Detector, it was a passive, integrating and instant read-out neutron detector. Also blood samples were irradiated for chromosome aberrations analysis.

Results

After each irradiation the participating laboratories had two days to produce an estimate of the neutron kerma and γ-dose and as many other parameters as possible. To refine the estimated results the laboratories had two months. Therefore, some of the measurements of activation could be repeated in the home laboratory.
The following parameters for the radiation could be determined from many of the dosimetry systems: neutron fluence in 6 energy intervals, neutron kerma, γ-dose, ^{24}Na-activity in the phantom, neutron dose of the recoil neutrons from the phantom and γ-dose from the $^{1}H(n,\gamma)^{2}H$ reaction in the phantom.
Two examples of final results are shown in the figures 3 and 4.

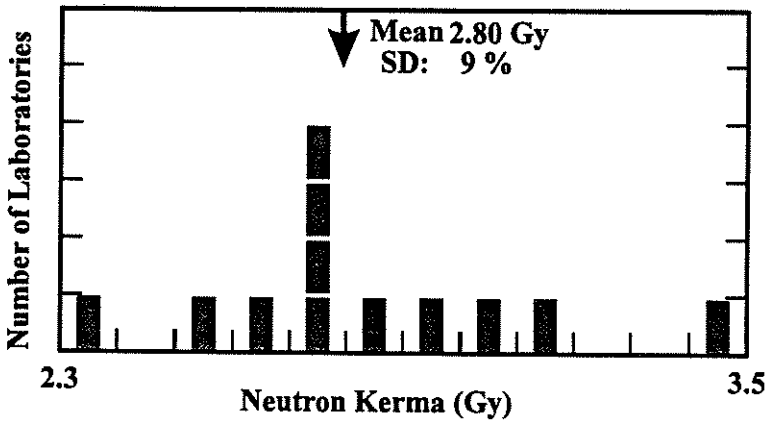

Figure 3. Final results of the neutron kerma in free air from a criticality in a bare reactor. The reference neutron kerma was 2.55 Gy. The 48h estimates gave a mean of 3.2 Gy with a standard deviation of 23 %.

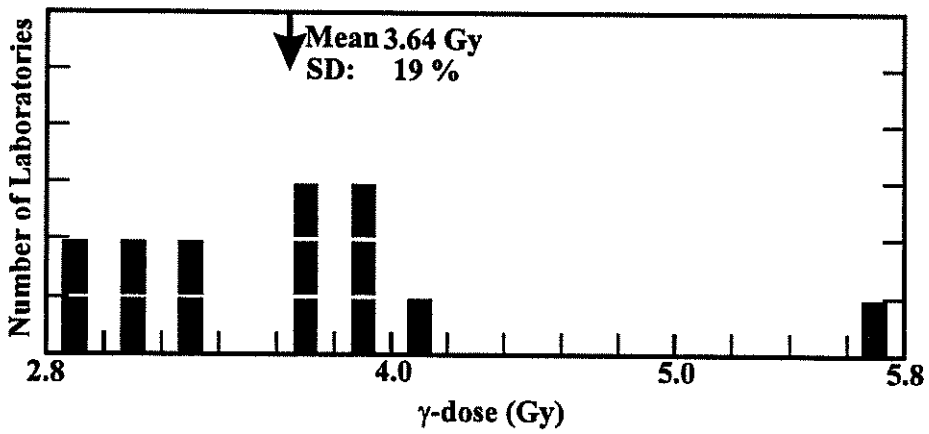

Figure 4. Final results of the γ-dose in free air from a criticality in a bare reactor. The reference γ-dose was 3.97 Gy. The 48h estimates gave a mean of 3.71 Gy with a standard deviation of 11 %.

Conclusions

Most laboratories managed to give reasonable estimates of the neutron kerma and the γ-dose within the first two days even though they had to use unfamiliar equipment. The standard deviations for the first estimates were about 10 - 20 %. The means of the final dose estimates delivered two month later had a standard deviation of up to 17 % for the neutron doses and up to19 % for the γ-doses. The difference between the mean value and the reference value varied from 1 to 17 % for the neutron kerma and from 9 to 24 % for the γ-dose.
Since 18 years had past from the previous intercomparison, an important result at Valduc was that a "new" generation of dosimetrists had the opportunity to test their criticality accident dosimetry system.

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International Exchange of Information in the Event of Nuclear Accidents.

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Abstract

It is today recognised that nuclear accidents may have considerable cross border consequences. It is therefore important also for neighbouring countries to have as much information as possible in order to assess consequences for their own countries.

Computerised decision support systems are now being developed for implementation in many countries. Their capability of collecting and processing information has improved considerably and will add a new dimension to the term decision support.

Rapid exchange of information between decision support systems therefore becomes important for the emergency preparedness organisation in order to establish the best possible background for decisions and information to the public in the event of a nuclear accident.

This exchange of information requires international consensus on strategies and methods and tools. This has not yet been achieved. The views from the Norwegian side on the principles and methods for this co-operation will be presented.

Radiation Risk Information to the Public: Principles or Common Sense?

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"Miscommunication of risks or lack of communication are costly: the public distrusts regulatory agencies and does not readily accept their decisions. When the public contests or, at a minimum, does not respect regulatory decisions, administrative agencies operate inefficiently."
(Stenzel 1987)

Abstract

The risk information strategies of radiation protection authorities and experts failed, to a large extent, after the Chernobyl accident, giving rise to distrust and "deafness" among the public side. In some countries, this distrust can be explained by purely historical and political reasons, but information failure also occurred in non-corrupt and democratic countries like Sweden. Radiation risk messages were not understood by the public, as the basic laws of risk communication were not obeyed by radiation authorities and radiation safety professionals. Remedial advice on trivial exposures makes no sense to the public, and is a misinformation that is costly in terms of distrust and disobedience. These negative effects must be included in any radiation exposure optimisation process worth the name. In addition, intervening against trivial or very low exposures will nourish hypersensitivity towards ionising radiation and in the long run support a public demand for costly remedial actions against trivial radiation threats. It is stressed that the primary goal of food regulation, i.e. a limited yearly dose or intake of activity, must be well-known to the media and the public before secondary Bq/kg limits are enforced. Overprotective food regulations cause confusion and discredit the regulating authorities. A draft of rules for radiation risk communication is presented, emphasizing the need for qualitative risk comparisons at the expense of probabilities, absolute numbers and technical terms and underlining that the radiation risk to the individual and his family always must be put first. It is concluded that the optimisation principle of radiation protection is consistent with common sense, but only if the optimisation is widened to include negative side-effects of low-cost remedial action and advice. Finding the right balance in radiation risk information is a great challenge for the future, both overprotection and radiation risk ignorance create distrust among the public towards radiation experts and authorities.

Introduction

The Swedish TV news, "Rapport" on Monday April 28 at 7.30 PM opened with: (my translation)

"*Big alarm this morning about high radioactivity around the Forsmark Nuclear Power Plant and about higher values measured in the whole of Mid-Sweden. Where is the leak? No Swedish plant seems to be guilty. Winds from the east have carried the fall-out from the Baltic States and the Soviet Union and we have just been informed that an accident has occurred at a nuclear power plant in the Ukraine.*" (Findahl et al. 1987)

The evening news is seen by millions of citizens, and they were struck by surprise; a large nuclear accident far away had affected Sweden. Unbelievable! Suddenly confronted with this unknown threat people's need for information was insatiable. Most people associate radioactivity with fearful things and consequently the prevailing and most important of questions from the individual citizen must have been "IS IT DANGEROUS?", and if it is, what precautions do I need to take for myself and my family?

Who could answer? We could. We, radiation safety experts. And we gave almost immediately the answer, "It is not dangerous". We had all the attention from the media we wanted, we were

on the air live in all the news programmes and we repeated it over and over again: "It is not dangerous", but this message did not get through. Instead we lost the people's ear and their confidence. Why?

The simple answer is that the message was not understood as we undermined the understanding by expressions like "unusually high local radioactivity", "pretty serious", "do not consume still water", "rinse leafy vegetables", expressions that in the ear of the public was in conflict with our assurance of a safe situation. We gave people millisieverts and becquerels when they needed risk information they could understand in the framework of everyday life.

We, as radiation protection experts, must take the major responsibility for the Chernobyl information failure. We can not blame the news media to any large extent. We, the experts, had ample time to present live and undistorted information to the media. The most severe consequence of this inadequate risk communication was not the anxiety caused among people, most people stayed calm, but instead the widespread loss of confidence and good will that radiation experts and authorities experienced.

The objectives of this contribution are several. The first is to illustrate how the optimisation principle of radiation should be applied to risk information strategies. The débâcle of Chernobyl information in Sweden will be used as a warning example of what not to do. Secondly, I will illustrate how misinformation can easily fuel unfounded anxiety. My third ambition is to summarize useful rules for radiation risk communication to the public, rules that must be implemented in order to regain trusting communication between authorities and experts, on one hand, and the media and public on the other.

General risk communication rules

It is easy to see why information strategies in the Scandinavian countries after the Chernobyl disaster were so unsuccessful. The basic "laws" of risk communication were, to a large extent, violated. Information messages concerning risks must be:

- Understandable and risk informative
- Related to the individual and
- Consistent.

The same goes for any imposed or proposed remedial measure, which in addition must be:

- Practical and
- Significantly risk reductive.

Regarding risk comprehension, it must be stressed that most audiences are poorly receptive to risk quantification. To phrase it simply, people listen more to words than figures, particularly in stressful situations.

The optimisation principle

In order to understand the Chernobyl information failure it is instructive to describe the optimisation principle of radiation protection, as this principle permeates the regulatory work of radiation protection authorities during controllable situations. The optimisation principle says

that the absorbed dose to people should be kept as low as readily achievable (ALARA) and emerges from the belief that the LNT (Linear No Threshold) hazard model is valid. The ALARA principle implies that also small, even trivial, doses should be avoided or decreased in cases where this can readily be done, for instance if the avoidance is associated with small enough costs or no cost at all. The outcome of the optimisation principle is schematically illustrated in Table 1, assuming that the dose-reducing measure in a low-dose scenario is either costly or free of costs.

It must stressed that the decision "Yes" to remedial procedures in Table 1 is not a very wise decision if "Yes" is associated with negative side-effects, immediate or in the future. It goes without saying that an optimisation without attention to all the positive and negative consequences, monetary or not, of a remedial proposal, is not a proper optimisation.

CONTROLLABLE PRACTICES Low-dose range		
	Action	Execute
•	Costly	No
•	Free	Yes

Table 1. Schematic decision making for remedial actions. If the action involves very small costs per eliminated mansievert (or is free of costs), the action should be taken.

Unfortunately, the optimisation scheme in Table 1 was also applied to the Chernobyl risk management by many national radiation authorities. Apparently, the fact was overlooked that remedial suggestions to the public regarding very small or trivial doses, were not in accordance with people's demand for efficient countermeasures, and not in accordance with the normal behaviour of agencies, viz. not to inform on or intervene in scenarios involving only insignificant risks. So, blacklisting chives, overprotective foodstuff regulations, and other interventions of authorities in the trivial dose domain, harmless as it may seem at first sight, lead to confidence losses and thus impairment of the future work of the regulating authority. Furthermore, remedial advice aimed at reducing of trivial absorbed doses, will nourish radiation phobias and support hypersensitivity to radiation in the media, among the public, and in other authorities with no radiation expertise of their own. In the long run, such currents of unfounded radiation fear, will support the public opinion that advocates the decision scheme of Table 2, a grand waste of money and a nightmare for all regulating authorities.

Action levels for foodstuffs

The way in which action levels for foodstuff were perceived by Swedish citizens is a good illustration of how important it is that authorities only present the simplest and most risk-relevant numbers to the public. As it was, the action level in Bq/kg was valued by the public as being highly risk relevant, which of course worked against the public's understanding of the risks from Chernobyl contamination, and was a hotbed for distrust in the

CONTROLLABLE PRACTICES Low-dose range		
	Action	Execute
•	Costly	Yes
•	Free	Yes

Table 2. Required decision making from a public that is unduly "sensitive" towards ionising radiation.

authorities issuing these foodstuff rules. An action level expressed in Bq/kg is of secondary nature, and should not be enforced without certifying that the public is aware of or understands the underlying primary goal. Furthermore, the action level of 300 Bq/kg of Cs-137 enforced in Sweden, was applied to all food irrespective of the amount consumed, which led to a ridiculous overprotection for herbs such as chives, parsley, and so on.

Foodstuff action levels expressed in Bq/kg are a nuisance from an information point of view. Attention should be placed where it belongs, i.e. on the primary action level expressed in a dose ambition and the corresponding yearly intake of activity. The recommended maximal yearly intake of activity combined with dietary guidance gives the hazard from the intake of radionuclides the correct perspective and the right impression of accuracy. Only after a yearly intake level has been established and is known to the public, should Bq/kg values be enforced, and then only for the basic diet. After the Chernobyl accident, the radiation protection authorities in Finland tried to follow this type of exemplary foodstuff policy, but they were criticized by the public and in the media, as their policy was not consistent with the overprotective Bq/kg values enforced by the other Scandinavian countries.

An example of misinformation

The region authority (Länsstyrelsen) for Malmöhus phoned me on Tuesday afternoon of April 29, and asked me to check the trucks on the incoming ferry from Poland arriving at Ystad at 11.00 PM. I agreed, went to Ystad, about 45 minutes' drive away, equipped with exposure meters and equipment for taking specimens. The only significantly contaminated vehicle I found was a Danish lorry. During Monday April 28 the driver had passed 400-500 km north of Kiev, according to his own estimate. Newspaper journalists followed my measurements closely and I explained that the lorry had contaminated wheel housings and the maximum contamination found, 50 times the natural background, was close to the airfilter. I emphasized to the journalists that neither the driver nor his clothes bore any sign of contamination and that the activity found on the lorry did not constitute any future risk to the driver. The press was also informed that the vehicle was free to return home to Denmark without further delay and that the driver, as an extra safety measure, was recommended to wash the truck and his clothes, and to take a shower when he arrived home. Länsstyrelsen in Malmö felt it necessary to report this case of vehicle contamination at the Swedish border to the Environmental Board in Denmark (Danska Miljöstyrelsen), and did so at once.

Now, let us examine the case described from an information point of view, not because it is an important intervention by the authorities and experts, but because it is a typical low-dose Chernobyl situation where authorities, experts and media interact.

The mere action of the authorities requesting controls at the border will support the view that this is a dangerous situation, but at the same time, an intervening operation reveals that the authority is alert and cares about its population. Thus, we have from a psychological point of view a two-edged sword in our hand.

<u>For a contamination check at the border</u>:
At the time, the Swedish authorities did not know how well goods and people were controlled in the Chernobyl area or at the Soviet-Poland border.
A check point at the Swedish border reassured the anxious Swedish public that at least <u>their</u> authorities had things under observation and control.

Against a contamination check at the border:
In the case at hand, the probability of finding severely contaminated persons or vehicles at the Swedish boarder were microscopically small.
The operation as such indicates a health threat to people that is not there.

Given the complete background, it can be said that, under the existing circumstances, setting up a checkpoint at the border was reasonable, and perhaps also recommendable, from an ethical point of view, but less motivated from a radiation protection perspective.

Well, accepting the *raison d'être* of the control of passengers and goods at Ystad, I will now look into the risk information, as presented to the public. How was the contamination at the border described to the press? Understandable? Was the information required to solve the burning question "is it dangerous or not" given? I am afraid that the answer is "No" to both questions, and I as an expert and sender of the message must take full responsibility for this information failure. My main error "informationwise" was that I presented my message "not dangerous" together with the quantification "50 times the natural background". Thus, first a statement was made assuring safety followed by a quantification that sounded dangerous (but actually says not a jot about risks). No wonder the journalist/reader became confused.

The leading morning paper in Sweden, Dagens Nyheter, reported the next day on page 7 "Radioactive lorry from Soviet; 50 times higher radiation" and the figure caption of the five-column article says that "The Danish long-distance lorry with its high radioactivity levels had not been closer to Kiev than 400 kilometres". This article must have given the reader a distorted and exaggerated view of radiation risks and no doubt, the information emanated from me as a radiation expert.

How should I have acted? Well, it is easy to make out that the message "not dangerous" must be followed by formulations that support that message, not contradict it. The exposure rate on the surface of the airfilter was presumably 5 μSv/h, so instead of "50 times natural background" I could have said "the radiation intensity at the very surface of the airfilter is below the intensity allowed at workplaces for work 8 hours a day" (at least is was at that time; 7.5 μSv/h has now been lowered to 2 μSv/h) or "one has to sit on the filter for more than 100 hours in order to receive a dose comparable to an X-ray investigation", or any similar comparison understandable to the reader and supportive to the assurance "not dangerous". But on the other hand, if I had presented my results like this, it is likely that the newspaper article would have been smaller or not printed at all. There is obviously a need for a consensus on a strategy for presenting radiation risks to the media and to the public. In the next section I have drafted rules on how to present information on radiation risks. The rules express my personal views and I very much welcome comments and criticism from the reader.

Rules for radiation risk communication

*Concentrate on risks to individuals, not to society.
*Ionizing radiation is alarming to most people, consequently your role is normally reassuring.
*Aim at simple and informative messages that can be understood by the receiver.
*Reduce numbers and quantities to a minimum.
*Remember that comparisons with familiar risk situations are much more informative than numbers.

*Avoid comparing your measured results with the natural background; compare them instead with accepted levels at workplaces or during medical examinations.
*Be careful with wording and how you frame your message, merely the words "cancer" and "radioactivity" may cause anxiety.
*Listen to the receiver and build on his/her common sense, not on intricate radiation protection principles.
*Do not try to educate the receivers during the acute phase of an accident. You must base your message mainly on what the audience already knows and is familiar with.
*Be aware that the receiver expects effective countermeasures.
*If you suggest purely psychological measures with no radiologically protective effect, you should be open about the true nature of these measures.
*Be consistent: if you have said the situation is not dangerous, you must act accordingly.

Conclusions

With the knowledge of how the Chernobyl information was received in low-dose countries like Sweden, it is clear that overprotection is costly in the long run and that the outcome of an optimisation (ALARA) process makes no sense unless all negative side-effects of low-cost remedial actions and advice are included in the optimisation. The policy dictating how action levels for food are presented must be changed, emphasizing the primary goal of dose reduction and yearly intake of activity. The secondary action level in the form of Bq/kg values should be launched only if the primary ambition is understood by the media and the public. The balance between allowing extra small doses and optimising doses downwards is delicate. Finding the right balance in radiation risk information present a great challenge, as both overprotection and radiation risk ignorance will give rise to public distrust. In order to regain the public's confidence, radiation safety authorities and organisations must come to a consensus on how to present information on radiation health hazards, especially in the low-dose range. Professionals in the field of radiation protection must learn that efficient radiation risk communication with the media and the public involves very little probabilities, absolute numbers and becquerels, but a lot of references and comparisons to familiar and understandable risk situations.

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Järnvägsvagn kontaminerad med Cs-137

Tord Larsson, Statens strålskyddsinstitut, Sverige, 1996-05-30

Inledning
Efter Sovjetunionens fall i början av 90-talet och Tjernobylkatastrofen för 10 år seden, har svensk stålindustri i allt större omfatting utrustat sig med automatiska detekteringssystem för joniserande strålning från gammastrålande radioaktiva ämnen. Systemen kallas allmänt gammaporter och placeras vid järnverkens infarter för järnvägs-och vägtransporter. Avsikten är att kontrollera återvinninngsbart skrot.

Gammaportarna har under senare tid givit larm vid upprepade tillfällen, men de flesta händelserna har kunnat härledas till enskilda metallstycken i lasten. Oftast har det varit naturligt förekommande radioaktiva ämnen som givit upphov till larmet. Som exempel kan nämnas beläggningar av radium-226 och uran-238.

Larm i gammaport
I augusti förra året larmade en gammaport vid Fundia Steel AB i Smedjebacken när en järnvägstransport med fragmenterat skrot passerade. Förnyad kontroll utfördes med handinstrument varvid kunde konstateras an en enstaka järnvägsvagn med last (ca 60 ton) strålade. Ytdosraten visade sig vara ca 30-50 µSv/h.

Vagnen återsändes til leverantören för undersökning. Lasten togs ur skopa för skopa och allt skrot kontrollerades med känsliga handinstrument. De sista skrotresterna skrapades samman med skyffel så att vagnen blev helt ren. I skrotet fanns ingen påvisbar förhöjd gammastrålning.

Det stod nu klart att det var vagnen själv som var orsaken till strålningen.

Järnvägsvagnen är inhyrd av Statens Järnväger från Tyskland sedan april 1994 tillsammans med ytterligare ca 150 vagnar. Underredet (boogie) tillverkades i Tyskland 1984.

Oregelbundet utspridd aktivitet
Vagnen togs omhand av Statens järnvägar och ställdes upp på ett avgränsat område i avvaktan på en kontroll av personal från Strålskyddsinstitutet. Kontrollen visade att vagnens ena boogie var kontaminerad med Cs-137. Aktiviteten var oregelbundet utspridd och återfanns på de flesta ställen på ramkonstruktionen. Prover togs från metallen i boogien och från ytan. Proverna visade att aktiviteten fanns i rost och smuts på ytan. Mängden radioaktivt ämne Cs-137 i beläggningen var ca 20 kBq/g. Ytdosraten på motsvarande ställe uppmättes till ca 100 µSv/h.

Tjernobyl uteslutet
Det fanns inga spår av radioaktivitet på vagnkorgen eller den andra boogie. Vidare kan konstateras att det inte fanns några spår av Cs-134 vars halveringstid är två år, vilket skulle ha varit fallet om boogien blivit utsatt för nedfall efter Tjernobylkatastrofen. Det har ännu inte gått att klarlägga på vilket sätt eller när kontaminationen har skett. Strålskyddsinstitutet har begärt att järnvägsvagnen skall undersökas ytterligare hos den tyske ägaren i samarbete med den tyska strålskyddsmyndigheten.

Radiological consequences in Denmark following a severe reactor accident at the Ignalina nuclear power plant.

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Abstract

The radiological consequences in Denmark are assessed, following a hypothetical severe reactor accident at the Lithuanian nuclear power plant at Ignalina. The release of radionuclides and subsequent spreading in the atmosphere is initially assumed to be similar in magnitude to that of the Chernobyl accident in April 1986. The atmospheric transport and deposition of radionuclides from the Ignalina power plant is modelled as a stochastic process, and a probability distribution is estimated for the deposition on Danish territory, based on the deposition pattern of radiocaesium after the Chernobyl accident. At the 1% probability level of the atmospheric transport and deposition, the collective radiation dose to the Danish population amounts to 100,000 manSv, assuming the accident to happen in the summer months shortly before harvest and with no intervention measures implemented. The most important pathways for radiation dose to the population will be external radiation from activity deposited on the ground and ingestion of contaminated food.

Introduction

In estimating the radiological risks to a population living nearby a nuclear installation, meteorological time series may be combined with standard atmospheric dispersion models in a probabilistic risk assessment (PSA 3). An example of such a level 3 PSA study of the Swedish Barsebäck reactor is shown in these proceedings [1]. A similar probabilistic risk assessment will invariably fail at very large distances from a nuclear reactor. This is partly due to insufficient amounts of reliable meteorological data and partly because of the inadequacy of standard atmospheric dispersion models in modelling atmospheric transport over very large distances.

On the other hand, atmospheric transport and deposition of radionuclides over large distances may essentially be viewed as a stochastic process in which the radionuclides are distributed in a random pattern. A realization of this stochastic process was the spreading of radionuclides from Chernobyl during the accident in April 1986, which took place under turbulent meteorological conditions.

The modelling of the radiological consequences in Denmark following a release of radionuclides from a distant nuclear reactor will be based on the stochastic process of long range atmospheric transport. In lieu of accurate, trajectory-based atmospheric dispersion model calculations, the distribution of radionuclides after the Chernobyl accident are used in assessing the risks to the Danish population after a hypothetical, severe reactor accident at the Lithuanian nuclear power plant at Ignalina [2].

The reactor accident is assumed to cause a release of radionuclides to the atmosphere which is of similar magnitude as the release resulting from the Chernobyl accident. Based on the spreading of radionuclides from Chernobyl, the probability distribution of radionuclide concentrations in Denmark is estimated, and the "worst credible" radiological consequences in Denmark are outlined.

Methodology

The stochastic modelling of the atmospheric transport and deposition of radionuclides contains two parts. First, average nuclide densities are estimated as the expectation value of a random walk process in the two-dimensional horizontal plane. Next, local fluctuations around the average densities are derived from the Chernobyl fallout data.

In the random walk model, the release of a radionuclide from the reactor gives the average deposition density [2]

$$c(r) = \frac{Q}{2\pi R^2} K_0\left(\frac{r}{R}\right), \tag{1}$$

and the average time-integrated concentration in air close to the ground surface,

$$I(r) = \frac{1}{v_{eff}} \frac{Q}{2\pi R^2} K_0\left(\frac{r}{R}\right) = \frac{1}{v_{eff}} c(r). \tag{2}$$

Here Q is the total release of the radionuclide, K_0 is the zero order modified Bessel function, the parameter R contains information about the atmospheric diffusion process as well as the decay and deposition, while v_{eff} is the effective deposition velocity of the nuclide. Both the density of the radionuclide in air and on the ground decreases monotonically with the distance r from the reactor.

The fluctuations in the radionuclide densities are obtained by comparing the measured concentrations to the expectation values (1) and (2). The parameters R and v_{eff} are fitted to the Chernobyl data. Assuming the fluctuations in the radionuclide densities to have the same r-dependence as the average values, measured densities at different distances from the reactor may be compared in order to obtain the probability distributions of the radionuclide densities.

Chernobyl data

In Figs. 1-2, the average deposition density and time-integrated concentration in air of ^{137}Cs, Eqs. (1-2), are compared to the densities measured after the Chernobyl accident [3]. From the figures, values for the diffusion length and the effective deposition velocity are estimated to be $R \approx 1000$ km and $v_{eff} \approx 1$ cm/s, respectively. The effective deposition velocity of 1 cm/s should be compared to measured dry deposition velocities for ^{137}Cs of the order of 0.1 cm/s [4], indicating that only approx. 10 % of the total deposition of caesium is due to dry deposition while the remaining 90 % stems from wet deposition.

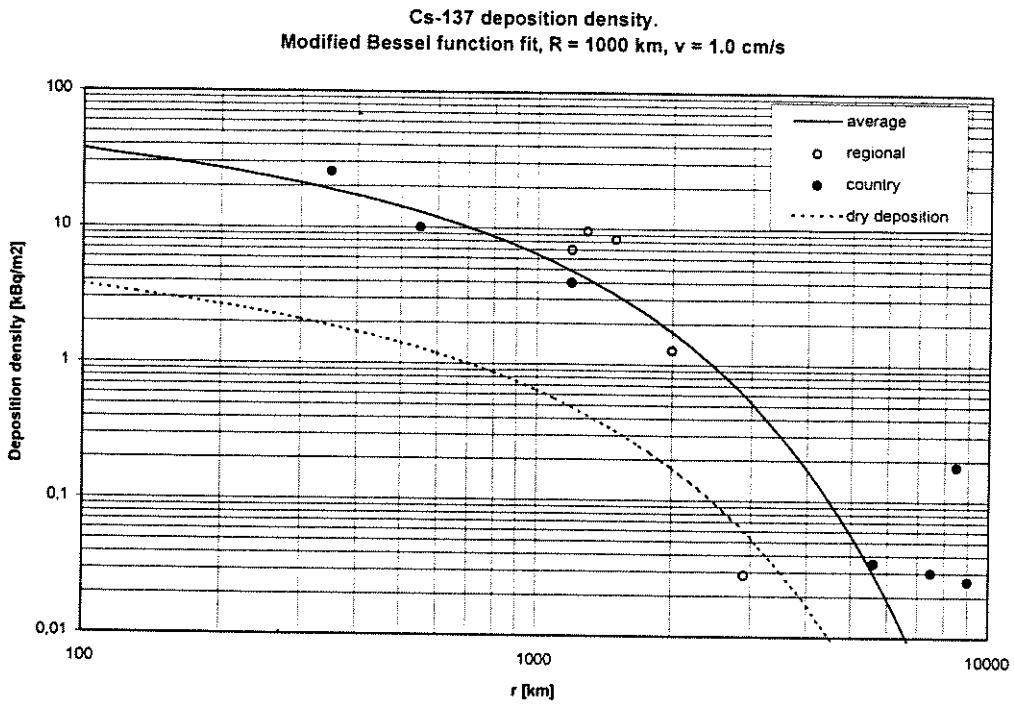

Figure 1. Deposition density of ^{137}Cs versus distance from Chernobyl.

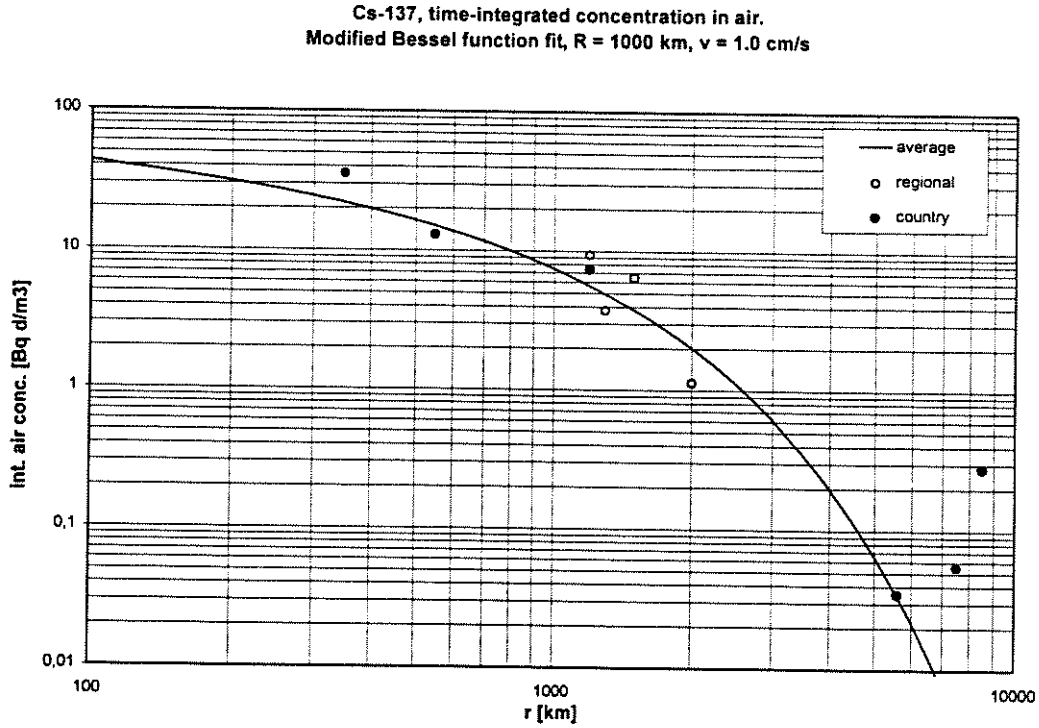

Figure 2. Time-integrated concentration in air of ^{137}Cs versus distance from Chernobyl.

In Figs. 3-4, the cumulative probability distributions are shown for the measured densities c and I. The densities represent average measurements for each country, or in some cases for a small part of a country. All densities have been scaled to the values they would have at the common distance of 1200 km from the reactor, using the functional form of the average densities, Eqs. (1-2).

From the figures, the size of the fluctuations can be estimated. Larger fluctuations are observed for deposited material (Fig. 3) than for airborne material (Fig. 4). At the 1 % quantile level the deposition density is approx. 10 times the average deposition density, $c(r)$ = 4.6 kBq/m^2, while the time-integrated concentration in air at the 1 % quantile level is 3-4 times the average value $I(r)$ = 5.3 Bq d/m^3.

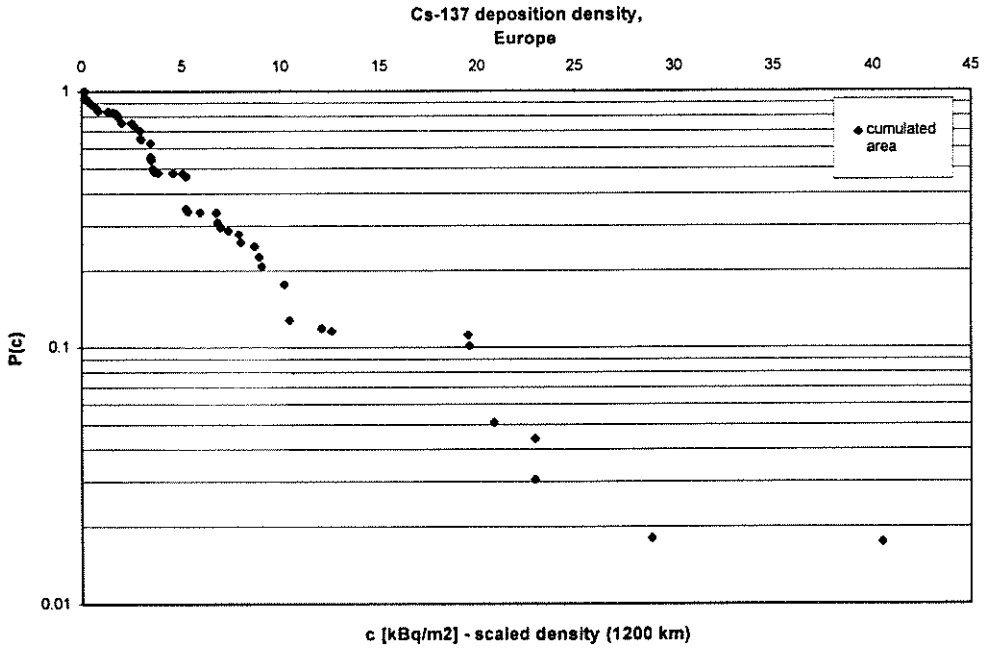

Figure 3. Cumulative probability distribution of deposited ^{137}Cs after Chernobyl.

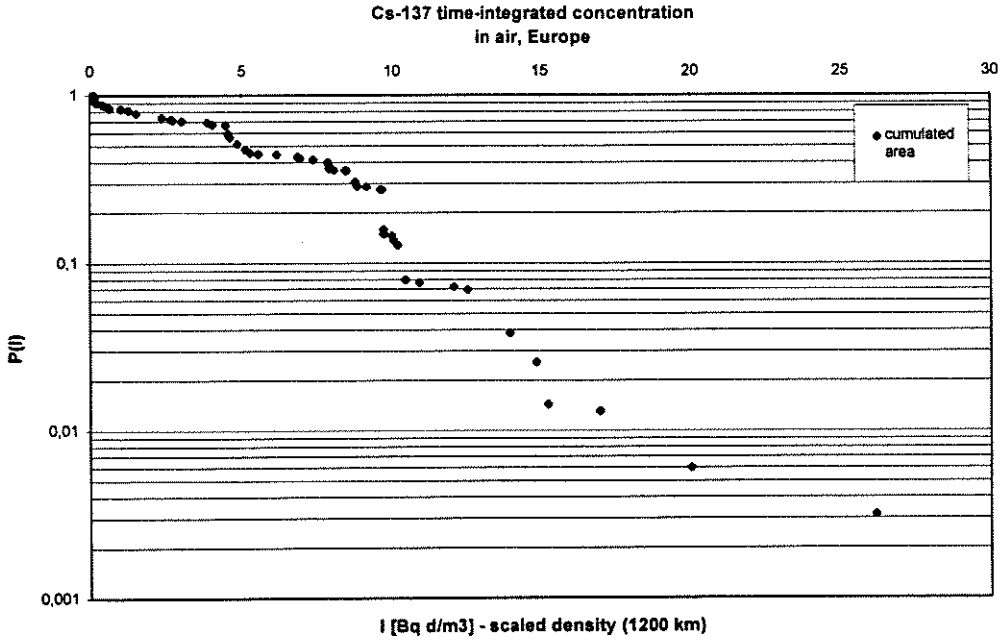

Figure 4. Cumulative probability distribution of airborne ^{137}Cs after Chernobyl.

Results

Each of the two units of the Ignalina power plant operates at 1200 MWe, compared to the 1000 MW electrical power of the Chernobyl unit 4 reactor. Recent safety measures introduced at Ignalina may largely have eliminated the possibility of having an accident of the same severity as the Chernobyl accident. Nonetheless, in a "worst credible" scenario, we assume an accident at one of the Ignalina units to have similar release fractions as those of the Chernobyl accident, implying that the order of 110 PBq ^{137}Cs would be released from the reactor, a slightly larger amount than released from Chernobyl.

Using the probability distributions for the atmospheric transport and deposition derived above, we obtain in Denmark, at a distance of approx. 1000 km from the reactor, the concentrations of ^{137}Cs shown in table 1.

^{137}Cs in Denmark	Concentration on the ground [kBq/m^2]	Concentration in air [Bq d/m^3]
Average	7	9
1 % quantile	70	40

Table 1. Average and 1 % quantile levels of ^{137}Cs in Denmark assuming that a nuclear accident at Ignalina, similar to the Chernobyl accident, has taken place.

From the radionuclide concentrations, the effective doses from external radiation and committed, effective doses from internal radiation are evaluated. The effective doses are estimated assuming normal shielding and filtration factors [3], but not allowing for specific protective actions such as sheltering recommendations. Other radionuclides than ^{137}Cs are included to the extent that they were widely measured after Chernobyl and are assumed to give the same relative contribution to the effective dose as they did in Europe after the Chernobyl accident.

Dose committed from the ingestion of foodstuffs is evaluated using transfer factors for iodine [3] and caesium [5], estimated on the basis of Chernobyl and weapon test fallout data, respectively. The transfer factors vary by at least one order of magnitude, depending upon the time of the year when the radionuclides are being transferred to the foodstuffs, with the largest transfer factors being associated with deposition of radionuclides during the summer months, shortly before harvest. In the calculations, the high 90 % quantiles of the transfer factor data sets have been used [2].

The collective, effective dose to the Danish population of approx. five million people is shown in table 2 below. The doses are estimated, based on the 1 % quantile level for the atmospheric transport and deposition. Using the 1 % quantile level amounts to assume rather unfavourable meteorological conditions prevailing at the time of the accident. That is, one percent probability level is likely to single out the most heavily contaminated country of all countries (of a similar size as Denmark) at a distance of approx. 1000 km from Ignalina.

Pathway	Collective dose [manSv]
External radiation from cloud	6
Inhalation	300
External radiation from ground	28000
Ingestion of foodstuff	65000
Total	94000

Table 2. Collective, 50 years effective dose in Denmark, based on the 1 % quantile level of the atmospheric transport and deposition model.

The dose estimate assumes that no protective intervention measures are implemented. Most of the dose is seen to stem from external radiation from deposited activity and from the ingestion of contaminated foodstuffs. The high collective dose obtained from the ingestion of foodstuff is primarily due to ^{131}I through the intake of milk products, and this part of the dose (approx. 30,000 manSv) may readily be avoided by enforcing restrictions on the use of milk products in contaminated areas.

In summary, the "worst credible" consequences in Denmark of a severe accident at the Ignalina nuclear power plant are outlined, the assumptions being 1) a release of radionuclides of the same magnitude as from the Chernobyl accident, 2) applying the highest concentration of radionuclides in Denmark consistent with a 1 % probability for the atmospheric transport, and, 3) that the accident takes place during summer without any protective intervention measure introduced. Clearly, modifying any of these assumptions will result in a lowering of the dose estimate.

Many constructive discussions with A. Damkjær and S.P. Nielsen are gratefully acknowledged.

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ON ETHICAL ISSUES IN RADIATION PROTECTION

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Abstract
From an ethical viewpoint the author surveys existing international radiation protection recommendations and standards. After outlining previous work on the ethics of radiation protection, professional ethics, and the ethics of human radiation experiments, the author discusses ethical thinking on seven key issues related to radiation protection and ethics.

INTRODUCTION

The paper is mainly based on an article by Professor K Shrader-Frechette, Dep of Philosophy, University of South Florida, Tampa and the author submitted in July 1996 to Health Physics - the Radiation Protection Journal (Ref.1).

The United Nations and its daughter organizations such as the International Atomic Energy Agency (IAEA) have devoted many efforts to protecting humans and the environment from ionizing radiation. The Universal Declaration of Human Rights, adopted in 1948 (Ref.2), stated in article 3: "Everyone has the right to life, liberty and the security of person."

In establishing ethical principles for radiation protection, the International Radiological Protection Commission (ICRP) has been prominent ever since the commission began in 1928 (Ref.3).

INTERNATIONAL RADIATION PROTECTION STANDARDS

ICRP publication no. 60

The latest ICRP recommendations were issued as publication no. 60 (Ref.3). This document elaborated a conceptual framework for radiation protection based on ethics, experimental work, and risk assessment. An important presumption of the ICRP conceptual framework is that even small doses of radiation may produce some deleterious effects. The three main principles of the ICRP for proposed or continuing radiation protection practices are:

a) the justification principle,
b) the optimization principle, and
c) the dose-limitation principle.

The justification principle prohibits practices involving additional radiation exposures unless they produce sufficient societal benefits. The optimization principle requires managers to keep radiation exposures as low as reasonably achievable (ALARA), taking into account economic and social factors. The dose-limitation principle limits exposure of individuals to radiation. Because medical exposures are intended to be as low as possible and to provide a direct benefit to the patient, ICRP recommends in publication 60 that no one apply dose limits to medical exposures.

The system of radiological protection that ICRP recommends for intervention is based

on two additional principles:

d) The proposed intervention should do more good than harm, and

e) One should optimize the form, scale, and duration of intervention.

Although the ICRP does not employ the term *precautionary principle*, it does use the concept, at least implicitly. In fact, the whole philosophy of protection against stochastic effects is not based on proved harm from radiation, because cancer and hereditary damage from radiation has never been demonstrated conclusively, either in humans or animals at low doses. Rather, the ICRP and other international agencies believe that adopting the linear, no-threshold hypothesis is reasonable, if one wishes to take ethical precautions in a situation of uncertainty.

As part of the process of justifying radiation exposures, the ICRP also implicitly uses the *substitution principle*. For example, in medical diagnosis of offspring of pregnant women, the ICRP recommends use of ultrasound rather than X-rays.

IAEA fundamentals for radiation

The IAEA adopted in 1995 safety fundamentals relating to radiation protection and the safety of radiation sources (Ref.4). These fundamentals are mainly ethical, and they are partly based on the protection principles given in ICRP publication 60. According to IAEA the primary aim of radiation protection is to provide appropriate standards of protection and safety for humans without either unduly limiting the benefits of practices giving rise to radiation exposure or incurring disproportionate costs in the case of intervention. This aim is expressed by two specific protection and safety objectives:

Protection Objective: to prevent the occurrence of deterministic effects in individuals by keeping the doses below the relevant threshold and to ensure that all reasonable steps are taken to reduce the occurrence of stochastic effects in the population at present and in the future.

Safety Objective: to protect individuals, society and the environment from harm by establishing and maintaining effective defenses against radiological hazards from sources.

These objectives are achieved by the application of the following eleven principles:

* A practice shall be justified on the grounds that it produces sufficient benefit to the exposed individual(s) and to society to offset the radiation detriment it may cause.
* For justified practices, other than those involving medical exposures, restrictions on the dose that individuals may incur (dose limits) are required in order to ensure that no person be subject to an unacceptable risk attributable to radiation.
* For any source, except for therapeutic medical exposure, the doses, the number of people exposed, and the likelihood of incurring exposures should all be kept as low as reasonably achievable.
* All reasonably practicable measures shall be taken to enhance operational safety and to prevent radiation accidents and mitigate their consequences, should they occur.
* Any proposed intervention shall do more good than harm.

* The form, scale, and duration of any intervention shall be optimized so that the net benefit is maximized.
* Sources shall be located, taking into account those factors which affect exposure, or potential exposure, of individuals and populations.
* The design and construction of nuclear facilities shall ensure that a source is suited for reliable, stable and easily manageable operation that ensures protection and safety. For this purpose, consideration must be given to defense in depth, human factors, system testing, and feedback of operational experience.
* The operation and use of sources shall be based upon procedures and conditions of operation which promote safety and security of the source, optimization of radiation protection, and which reflect the lessons learned from operational experience.
* The government shall establish a legal framework for regulation of practices and interventions, with clear allocation of responsibilities, including those of a Regulatory Authority.
* Parties responsible under the legal framework shall, as appropriate, provide for radiation protection and safety, verify its effectiveness, and prepare adequate emergency plans.

ILO Convention 115

ILO Convention no. 115, concerning the protection of workers against ionizing radiation (Ref.5), also contains some ethical principles. This convention came into force in 1962. Article 5 of the Convention states the main protective objective: "Every effort shall be made to restrict the exposure of workers to ionizing radiation to the lowest practicable level, and any unnecessary exposure shall be avoided by all parties concerned."

SOME WORKS ON ETHICAL ASPECTS OF RADIATION PROTECTION

Since the latest ICRP recommendations in publication 60, Silini (Ref.6), Wallace (Ref.7), Maushart (Ref.8), Prêtre (Ref.9), and Shrader-Frechette (Ref.10) have assessed some of the ethical issues related to radiation protection.

Silini asks whether the present system of radiation protection is founded on sound ethical principles. His answer is "yes," but he proposes clarifications in the meaning of the justification principle; inclusion of environmental, as well as human, protection in the standards; and worker protection equal to that of the public.

Wallace concludes that the policy-formulation process for the use of ionizing radiation in Australia is neither unethical nor completely satisfactory. He proposes ways to strengthen ICRP recommendations and provide for more public control of radiation protection.

Maushart emphasizes the importance of global acceptance and use of the present radiation protection principles. He points out the lack of uniform standards across the planet.

Prêtre connects the three ICRP protection principles with four key ethical concepts: responsibility and respect for life, justice, and common good. He shows that, in general, the ICRP principles of radiation protection reflect all four concepts.

Shrader-Frechette argues against extremist positions in radiation protection, and she criticizes both environmentalists who unrealistically favor zero risk, as well as industrialists who are too lax about radiation risk. She argues for ethical improvements in dealing with radiation risks that are uncertain, inequitably distributed, imposed involuntarily, or not compensable in full.

PROFESSIONAL CODES OF ETHICS

The American Health Physics Society has issued a code of professional ethics (Ref.11). Its principles are intended as guidelines by which members may determine the propriety of their conduct in relationships with employers, co-workers, clients, governmental agencies, other professionals, and the public. Seven of the nine of the Society's principles concern factors that could affect radiation protection:

* Members shall strive to improve their professional knowledge and skill.
* Each member shall be a judge of his/her competence and will not undertake any assignment beyond his/her abilities.
* Members shall never compromise public welfare and safety in favor of an employer's interest.
* No employment or consultation shall be undertaken which is contrary to law or the public welfare.
* Members will gladly accept every opportunity to increase public understanding of radiation protection and the objectives of the Society.
* Professional statements made by members shall have sound scientific basis. Sensational and unwarranted statements of others concerning radiation and radiation protection shall be corrected, when practical.
* Members shall protect the sources of confidential communications, provided that such protection is not in itself unethical or illegal.

It may now be timely for the Nordic Society for Radiation Protection to start the discussion of establishing a joint Nordic code of professional ethics.

ETHICS AND HUMAN RADIATION EXPERIMENTS

Reports on human radiation experiments have highlighted the ethical issues in this field of research. In the International Basic Safety Standards for Protection the IAEA (Ref.12) has argued that the exposure of humans for medical research is not justified unless it is:

a) In accordance with the provisions of the Helsinki Declaration and follows the guidelines for its application prepared by the Council for International Organizations of Medical Sciences and the World Health Organization, and

b) Subject to advice of an Ethical Review Committee, or any other institutional body assigned similar functions by national authorities, and subject to applicable national and local regulations.

Mossman (Ref.13) has reviewed US radiation experiments, especially during the Cold War. He raises a number of questions: Is it appropriate to scrutinize past scientific investigations in the light of current standards, when such studies were conducted in a different political climate? Is the process of informed consent adequate in addressing the needs of the subject? Are current U S federal regulations and policies adequate to protect human subjects in research settings?

Mossman recommends that strategies to protect informed consent in the face of ethnic, cultural, and language barriers be incorporated into the regulatory framework. He also says that risk comparisons may be of value in putting the radiation risk into perspective, and he recommends establishing a federal commission to address national radiation-protection issues.

KEY ETHICAL ISSUES

Among the many ethical issues concerning radiation protection, seven are considered by professor K Shrader-Frechette and the author as crucial.

Equity versus efficiency: the justification principle

The justification principle, a cornerstone of ICRP and IAEA ethical philosophy, expresses an important ethical principle: that one must always justify any additional radiation exposures in terms of the benefits the exposures bring. In other words, only good reasons justify additional exposures. The justification principle also allows one to exchange societal benefits for individual radiation exposures. As such, it sanctions a utilitarian ethics, an ethics whose fundamental ethical principle is to maximize benefits to the majority of people.

Because the justification principle allows benefits to society to offset radiation detriments to individuals, it is somewhat controversial. Proponents of egalitarian ethical theories, whose fundamental ethical principle is equal treatment or equal protection of all people, would argue that utilitarian benefits to a group can almost never, if ever, offset detriments to individuals because they are fundamentally unfair. They would say that, unless the exposure gainers somehow compensate the losers, the radiation exposure is unethical because it is discriminatory.

Health versus economics: the optimization principle

The optimization principle requires one to optimize radiation protection, to keep exposures as low as reasonable achievable (ALARA), taking into account social and economic factors. The principle of optimization acts to correct unreasonable protection costs and ensures that efforts to achieve a rational reduction of exposures are profitable.

Many egalitarians believe that money-for-safety tradeoffs are often unacceptable. Some harms are so serious that no amount of money could possibly compensate the victims or their survivors.

Individual rights versus societal benefits: the dose-limitation principle

Just as egalitarians disagree with utilitarians over whether to give priority to equity over

efficiency, and over whether to give priority to health over economics, similar battles occur over whether to give priority, in radiation protection, to individual rights or to societal benefits in deciding whether to limit doses or exposures.

Egalitarians tend to be proponents of individual rights to protection against radiation exposure. Egalitarian opponents of uniform standards argue that because the public is more averse to some threats, like cancer, different standards (including zero risk) are necessary for different risks. They also argue that because everyone has rights to protection against any unnecessary danger, all societal and workplace risks should be kept as low as possible, perhaps below the *de minimis* level. They say that uniform standards frequently are too lenient in protecting only "average" persons rather than especially sensitive individuals such as children.

Due process versus necessary sacrifice: liability principles

One important issue is the ethical justification for reparation or compensation for radiation risks. Another issue is whether those exposed to radiation deserve full and strict liability for the risks they bear. Stochastic effects are believed to occur, even if with low frequency, even at the lowest doses of radiation. Because of this ICRP assumption, citizens exposed to additional radiation, often without their consent, can argue that it is a risk for which they should receive full liability protection. Their argument is that, because some individuals likely will contract cancer or die as a consequence of any additional radiation exposure, they deserve liability protection, particularly if they receive no benefits from the exposure or do not consent to it.

Uniform versus double standards: public and worker risks

The ICRP principles also raise several ethical issues because they sanction higher dose limits for workers than for members of the public. ICRP recommendations for dose limits on occupational exposure are 20 mSv per year, averaged over defined periods of 5 years, with the restriction that dose in a single year not exceed 50 mSv. The effective dose limits for members of the public, on the other hand, are for an average of 1 mSv per year, averaged over 5 years, with the restriction that dose in a single year not exceed 5 mSv (Ref.3). The ICRP argument given in paragraph S24 in publication 60 is to establish for occupational exposures a level of dose above which the consequences for the individual would be regarded as unacceptable.

Utilitarian ethical theorists would be likely to sanction such a double standard for workers and the public on grounds of efficiency, whereas egalitarians would be likely to oppose it, on the grounds that workers deserve protection equal to that accorded members of the public. The utilitarian argument in favor of the double standard is that, if all things being equal, radiation workers receive higher wages for risky work. This "compensating wage differential" justifies their bearing greater risks than members of the public. According to the theory behind the differential, the riskier the occupation, the higher the wage required to compensate the worker for bearing the risk, all other things being equal. Because in reality there is often no real compensating wage differential it is quite difficult, ethically speaking, to justify the double standards of radiation protection for workers.

Stakeholder consent versus management decisions: controlling exposures
Just as much ethical debate focuses on worker consent to higher radiation risks, so also there is controversy over whether members of the public need to give consent to radiation exposures and risks. Should experts and regulators define and evaluate radiation risk because they have most of the supply of wisdom about radiation protection? One possible way to promote greater consent in the area of radiation protection would be to include representatives of stakeholders on radiation-protection committees.

Radiation protection of nature
The ICRP 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.

Considering the shift of ethical values that we have observed in the world (Ref.14), we may now be at a point in history when it is timely and when there also exist very good scientific reasons to set up a policy for the protection of the environment (including animals) against the harmful effects of radiation.

One way to go ahead is by extending the ICRP protection principles to the environment. The dose limits for different species may, however, be very different from the ones given by ICRP for humans due to the different sensitivities for radiation.

CONCLUSIONS
From the review based on the work of professor Kristin Shrader-Frechette and the author (Ref.1) the ethical issues in radiation protection are neither simple nor one-sided. The insights of ethical theories provide a number of ways in which current recommendations and standards for radiation protection could improve. They could be more equitable, more protective of human health and human rights, including due process and consent, more cautious in recommending worker exposure standards that are more lenient than public standards, and more protective of environmental welfare.

Lauriston Taylor, one of the fathers of radiation protection, once said: "Radiation protection is not only a matter for science. It is a problem of philosophy, morality, and the outmost wisdom" (Ref.15). I fully agree to his statement.

My feeling is that we in future research programs should give priority to study the ethical issues in radiation protection.

Finally it may now be timely for the Nordic Society for Radiation Protection to establish a code of professional ethics.

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'Potential Exposure' in Practice

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Abstract - Earlier, the philosophy of radiation protection was strongly targeted on exposures which were certain to occur. However, there are also potential exposures. These are not normally expected to occur, but can be anticipated, and the probability of their occurrence can in principle be estimated. This concept was first developed in connection with considerations about final deposition of long-lived radioactive waste, and further elaborated for major accidents such as nuclear disasters. The present paper discusses a third situation: limited accidents affecting one or a few persons only. In contrast to the other two situations, a risk constraint for such smaller accidents need only consider health detriment to the immediately affected victims. ICRP is at present contemplating recommendations for such limited potential exposures. Pending such recommendations, it seems clear that protection against potential exposures must start with a structured analysis of scenarios. Given probabilities derived from such analyses, and nominal detriment probability coefficients, the expected detriment from potential exposures can be computed. This detriment can be compared to a risk constraint, which need not be particularly complicated but should include consideration of deterministic damage.

Introduction

ICRP 60 (1) specifies two broad categories of situations affecting radiation exposure of humans: practices and interventions. Exposures which might result from a practice are further divided into normal exposure and potential exposure. Normal exposure is expected to occur with near certainty, from operations conducted as planned or after unintended, high probability but low consequence events.

In contrast, potential exposure is not certain to occur. It results from unplanned events such as equipment failure or departure from planned operating procedures. Such events cannot be predicted in detail, but they can be theoretically anticipated and a probability of their occurrence can be assigned.

The dose limits recommended by ICRP (1) for exposures under normal working conditions in practices using radiation do not apply directly to potential exposures. They must be supplemented by risk constraints which take account of both the probability of incurring a dose and the probability of various types of harm associated with that dose. The theoretical basis for such a tool was developed in ICRP 64 (2). Current work within ICRP is aimed at demonstrating how that tool could be applied in practice in 'limited accidents'.

The expression 'limited accidents' is used here to indicate events where the number of individuals affected is small. Primarily, potential exposures would affect persons who are also subject to the normal exposures in the practice. Detriment would be limited to health effects to the persons actually exposed. The processes leading to potential exposure would be relatively simple, and may be the predominant threat associated with the practice. Potential unsafe entry into an irradiation room at a sterilising installation may be mentioned as an example of this situation.

In contrast, large disasters such as major nuclear accidents invoke detriment beyond health effects to exposed persons (say, agricultural restrictions, food consumption controls, etc). Final disposal of solid waste adds a further dimension of potential exposures in the far future. The present paper does not deal with these more specialised problems.

Criteria of Acceptability

ICRP (1, 2) has recommended that health risks due to potential exposures be limited and constrained to the same order of magnitude as that implied by dose limits and constraints for normal exposures. It may be assumed that deterministic deaths are valued in the same manner as stochastic deaths attributable to radiation.

Health risks due to potential exposure involve two different probability distributions. First, the unsafe situation causing the exposure occurs with a specific probability, say, P. Given the exposure which occurs with probability P, the conditional probability of stochastic and/or deterministic health detriment is a function of the dose, say, f(E). The unconditional probability of health detriment is therefore the product of these two probabilities, i.e. $P \cdot f(E)$. It is this quantity which should, according to ICRP (2), be kept at or lower than the magnitude of a reference risk, say, **R**. For the limited accidents discussed here, **R** can be regarded as a source-related individual risk constraint.

The magnitude of the risk constraint will not necessarily be set once and for all in ICRP recommendations and national legislation. It could be case specific and may be adjusted for various reasons. However, the fact that it is intended to keep health detriment due to potential exposure constrained to the same order of magnitude as that implied by dose limits and constraints provides some guidance. Current discussions within ICRP are believed to aim at a generic risk constraint which could be used as a starting point when more case specific constraints are chosen. For instance, in optimised operations maximum normal annual doses are often of the order of 5 mSv. Using the ICRP cancer death risk coefficient for occupational exposure of $4 \cdot 10^{-2}$ Sv^{-1}, the generic reference risk **R** would be $2 \cdot 10^{-4}$ a^{-1}.

Using such a reference risk, it is possible to work backwards and assess whether the probability P of unsafe events is acceptably low, and in consequence whether the engineered safety of the installation in question and its operational procedures are acceptable. For instance, if the event under study is unsafe entry into a radiation room, where dose rates are high enough that deterministic death will follow an entry (i.e. $f(E) = 1$), the probability P should be kept less than **R**. If upon analysis P is found to exceed **R**, further steps need to be taken to improve safety in the practice concerned.

Identification, Structuring and Quantification of Scenarios

An analysis aimed at estimating the probability P of unsafe events occurring, needs to identify all types of scenarios which could be significant contributors to potential exposure. A structured approach to this listing of scenarios is advisable. Several techniques to formalise the analysis are known and publicised, such as Table Top Exercises, Hazard and Operability Studies, Failure Modes and Effects Analysis, etc (3).

For each such scenario, it is assumed that demands will be placed on a protection system to function at a certain rate. Analysis aims at determining whether the system fails following these demands. Demands will be both 'normal' actions which routinely challenge the system and 'random' failure events (including human and equipment failures, intrinsic or caused by external events).

Once all relevant scenarios are believed to be listed, their logical structure must be determined. This type of operation is well known in engineering but may be less common knowledge among radiation protection specialists. Usually, such structuring is presented in event trees or fault trees.

Event trees start with an initiating demand on a given system, and moves progressively through successive responses of the system, describing the outcome in terms of success or failure of individual steps and devices. The complete event tree allows assessment of the overall probability of failure of the system due to a given initiating event, and combined assessment of all applicable event trees allows evaluation of the 'safety' of the system.

Fault trees begin at the other end, with a specified unwanted outcome (say, the safety devices fail to prevent unsafe entry into a radiation room), and works backward to analyse possible ways in which this outcome could have occurred. In more complex situations, it is not uncommon to use both types of tree during different stages of the analysis in order to ensure full coverage of all possible sequences of events.

Conceptually, the mathematical analysis of such event and fault trees is simple, being based on the elementary probability theory. However, the practical application of this theory is often quite difficult because of the many alternative outcomes under study, and in all but the simplest cases computer processing is recommendable for correct computations. Many computer programmes for such purposes are generally available.

Application of Acceptability Criteria: Some Complications

The quantification of scenarios just described involves probability estimations or assignments for numerous types of events. Care must be exercised to ensure consistency and coherence between values derived from existing experience data and values based on expert opinion or extrapolation.

The reliability of components in a safety system is usually regarded as an intrinsic characteristic. Often, this failure rate is regarded as constant over a long period of time, after a relatively short period of so called 'infant mortality' and a final increase in failure rates after long service (the so called 'bath tub curve'). High rates of demand might complicate the situation by causing wear such that the probability of failure on demand increases with time. Preventive maintenance may counteract such tendencies.

Sometimes, the question of modification of proven designs arises. If such modifications are contemplated 'in the field', for an existing system, there is considerable risk of potential exposure because of a high probability of human error. Except for pressing circumstances where immediate action is necessary to prevent degradation of the situation, no modification should be performed until a thorough safety assessment has been carried out and the proposed modification and the assessment are clearly documented.

Sometimes, measures taken to reduce a particular type of exposure may be associated with a trade-off in the shape of a concomitant increase of other exposure. For instance, numerous inspections are performed regularly in order to detect defects before they cause component failure. An increased inspection frequency in, say, an accelerator installation may improve the rate of early detection of defects and therefore decrease the probability of potential exposure.

However, due to the radiation fields normally present in such an installation, it would also entail increased occupational exposure. Both types of exposure must therefore be taken into account in an optimisation of such a situation.

In medical radiation usage, an added complication is that not only excessive doses but also too small doses to the patient may be unsafe. This often precludes comparisons of estimated detriment with the risk criterion **R**. However, structured analysis as discussed here is still of interest in order to identify major contributors of risk. On a relative basis, measures that might be taken to reduce risk to the patient can then be contemplated.

Human errors are often the most significant contributors to the probability of potential exposure. Unfortunately, they are also among the most difficult to quantify. They are strongly dependent on the situation and the so called performance shaping factors (layout of the workplace, amount of noise and distraction, level of stress etc). The probability of repeated human error may also well increase with time, if on a first occasion a given error did not lead to untoward consequences.

The above considerations concern correct computation of the probability P of an unsafe event. The probability f(E) of detriment given a certain dose may also involve some complications. For instance, potential exposure situations may well entail doses high enough to cause acute death due to deterministic effects. This introduces the possibility of a greater loss of lifetime than that associated with stochastically caused death, which occurs later in life. However, no additional weighting for early death seems warranted, considering general uncertainty in the probability calculations.

Furthermore, in some practices and operational environments localised deterministic effects will dominate the threat associated with potential exposure. For normal exposures, ICRP has expressly chosen not to take such detriment into account since it is most unlikely to result from exposures within present dose limits. However, for potential exposure it will be inevitable to assess such detriment. At the same time, while certainly worth assessing, loss of a finger (say) can obviously not be equated to death. Thus some sort of weighting for severity is required. It is believed that a generic weighting factor of 0.25 is being discussed within ICRP, based on various insurance and compensation schemes and considering the need for a reasonably simple generic application of weighting.

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European Basic Safety Standards Directive 1996

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Abstract - The Euratom Treaty from 1957 prescribes that uniform Basic Safety Standards shall be laid down within the European Community to protect the health of workers and of the general public against the dangers arising from ionizing radiation. The first European Directive was issued in 1959, and a new revised Directive was adopted by the Council of the European Union in May this year to replace the previous directive from 1980/1984.

The objective of the revision of the Basic Directive has primarily been to assure that European national legislations on radiation protection are in line with the new recommendation from the International Commission on Radiological Protection from 1991 (ICRP No 60). The new Directive therefore introduces the distinction between practices and intervention and the concepts of dose constraint and potential exposure. The dose limits are with minor editorial changes identical to the recommendations in ICRP No 60.

The Directive retains the system of reporting or prior authorization of practices although with some changes in the grouping of types and classes of practices into the two schemes. New radionuclides specific exemption levels for reporting, expressed as quantity or concentration, are given based on well established dose criterias and generic scenarios.

The Directive do not apply to radon in dwellings and to uncontrollable exposures to natural radiation sources, e.g. K-40 in the body, cosmic radiation at ground level. However, work activities within which the presence of natural radiation sources can lead to a significant increase in the exposure of workers or members of the public are covered by the Directive. The Directive introduces a new general obligation on the Member States to be aware of the possibility of such exposures and to react if necessary. The circumstances and possibility to apply protection measures regarding exposure to natural radiation sources can vary within and between Member States, and the Directive therefore allows a considerably flexibility in implementing this new area of concern.

Subsidiarity and flexibility has been key words during the negotiations of the Directive in the Council and being a minimum directive the new Directive offers a high degree of freedom for the Member States for the implementation of the Directive into the national regulations. In order to help the Member States during this process the Commission with the assistance of the Article 31 Expert Group is preparing a general Communication on the implementation of the Directive and advice and recommendation on more specific issues, e.g. natural radiation sources. The Members States shall comply with the Directive before 13 May 2000.

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Radioecology, development and trends

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Introduction

Radioecology comprises the relationships between radionuclides or radiation and the environment. Hence this discipline includes the movement of radionuclides within ecosystems and their accumulation within specific environmental components such as air, water, soil and biota.

Radioecological studies have been closely connected to unintentional releases of radionuclides to the environment. Such releases have in particular been bound up with the military exploitation of nuclear energy. In the following an overview of the more spectacular radioecological events will be given; it appears that also civilian use of nuclear energy has resulted in environmental contamination with radionuclides. The last part of the paper is devoted to the future of radioecology, where the coming needs and goals for radioecological research is identified.

Major radioecological events during six decades

The forties

The nuclear age started in the early forties during the second world war, when the first nuclear reactor went critical on December 2, 1942 in Chicago. The information on environmental contamination from the nuclear activities in these early days are sparse. It is however evident that the plutonium producing reactors, at Hanford at the Columbia river in the State of Washington released substantial amounts of radionuclides both to the atmosphere and the river during their first years of operation (UNSCEAR, 1993). From 1944 to 1946 about 17.6 PBq ^{131}I was thus released to the atmosphere. The releases from 1947 to 1949 decreased to about 1 PBq ^{131}I . Information on the very early releases to the Columbia river are missing, but we know that in the mid sixties the annual releases of ^{32}P, ^{46}Sc, ^{54}Mn, ^{60}Co, ^{65}Zn, ^{137}Cs and $^{152, 154, 155}Eu$ amounted to about 10 PBq (Robertson *et al*, 1972).

In the late forties the first plutonium production reactors went critical in the Former Soviet Union (FSU). They were located in the Southern Urals between Chelyabinsk and Yekaterinburg (at that time Sverdlovsk). From 1949 to 1951 100 PBq of mixed fission products were discharged to the

nearby Techa river, which belongs to the Ob river system. The amounts of ^{90}Sr and ^{137}Cs in these releases were 11.6 and 12.2 PBq respectively. The major part of the activity was retained in a cascade of reservoirs and by-pass canals. It is, however, estimated that in the order of 1 PBq ^{90}Sr and ^{137}Cs may have reached the Arctic Ocean from these discharges (Trapeznikov *et al,* 1993).

Both in the United States and the FSU, studies of the environmental behaviour of radionuclides were initiated in the late forties and early fifties. In Russia the geneticist Nikolaj V. Timofeeff-Ressovsky organised a radiation biology laboratory in the southern Urals, where he studied the distribution, accumulation and migration of radionuclides in experimental and natural biological systems (Paul and Krimbas, 1992). In the USA, Howard T. Odum presented his dissertation on the bio-geo-chemistry of strontium (Odum, 1950) which became important for the later studies of ^{90}Sr in the environment.

The fifties

The testing of nuclear weapons including the military use at Hiroshima and Nagasaki began in the midforties. The environmental contamination from the early atmospheric tests, when the weapons were in the 20 - 100 kilotons range, remained below the tropopause and was thus not dispersed globally, but deposited around the latitude band of the test site. In 1952 the United States and one year later, the FSU tested the first thermonuclear devices. The fallout from these and the subsequent atmospheric test series in the fifties and sixties occurred world wide because these very powerful weapons (in the megatons TNT range) injected the radioactive debris in the stratosphere. The production of environmental contamination from these tests was 760 PBq ^{90}Sr and 1140 PBq ^{137}Cs. The global fallout initiated the world wide studies of environmental radioactivity. The results of these early studies which concentrated on ^{90}Sr were presented in the first UNSCEAR (1958) report. In 1957 two serious nuclear accidents happened, which both got a significant impact on radioecology. On the 29th Sept. 1957 at Kyshtym, at the nuclear weapons plutonium production site in the South Urals (MAYAK), a chemical explosion occurred in a tank containing highlevel radioactive waste. The explosion dispersed about 40 PBq of fission products over an area of 300x50 km^2. Most of the activity consisted ^{144}Ce and ^{95}Zr, but 2 PBq ^{90}Sr and 13 TBq ^{137}Cs were also spread by the accident. This contamination, which was kept secret by the FSU until 1989 (Nikipelov *et al,* 1989) has given much valuable information on the radioecology of ^{90}Sr.

Less than two weeks after the Kyshtym accident a nuclear reactor accident occurred on the 10th October, 1957 at Windscale in Cumbria, UK. A military, air-cooled graphite-moderated, natural-uranium reactor used for plutonium production caught fire during the liberation of Wigner energy. The emissions lasted for 18 hours. About 1 PBq ^{131}I and minor amounts of other fission products were released (Jackson and Jones, 1990). Milk was contaminated with radio-iodine and remedial measures were introduced in the UK. The Windscale accident initiated a variety of radioecological studies, in particular of the behaviour of ^{131}I in the grass-cow-milk foodchain (Garner *et al*, 1960).

The sixties

The nuclear weapons testing in the atmosphere peaked during 1961-62 just before the moratorium for atmospheric tests was signed by USA. FSU and UK. The deposition of global fallout had its maximum in 1963, when the annual deposition rates in the northern hemisphere were about 0.5-1 kBq ^{90}Sr m^{-2} and 1-1.5 kBq ^{137}Cs m^{-2}. On a world-wide scale global fallout is still the most important source of man made radioactivity in our environment.

In 1966 and 1968 two accidents with US nuclear weapons happened. The first one in Palomares, Spain and the second in Thule, Greenland (Wrenn, 1974). At Palomares the terrestrial environment was contaminated by plutonium. At Thule the contamination was limited to the marine environment. The remaining contamination of the seafloor at Thule is about 1 TBq $^{239,240}Pu$ (Aarkrog *et al*, 1984), and the transfer of plutonium to biota is essentially limited to the benthic community.

The seventies

In the seventies the radioecological studies were characterized by a change from the terrestrial towards the marine environment and ^{137}Cs became the radionuclide in focus. Around 1975 the BNFL' nuclear reprocessing plant Sellafield in Cumbria, UK had increased its waterborne releases of ^{137}Cs to 4-5 PBq per year. Eventually the total discharges amounted to about 40 PBq ^{137}Cs before they were essentially stopped by the late eighties. These releases were studies intensively in the NE Atlantic during the following years. The discharges were useful as marine tracers. Beside of ^{137}Cs other radionuclides such as ^{134}Cs, ^{99}Tc and $^{239,240}Pu$ were followed in the marine environment (Kautsky, 1988 and Aarkrog *et al*, 1987). Dilution factor for transport of pollutants from western Europe to Arctic waters and transit times were determined. Furthermore the uptake of these

radionuclides were determined for various bio-indicators e.g. blue mussels and brown algae (NKA, 1985).

The eighties

In April 1986 the Chernobyl accident in Ukraine happened. This turned again the radioecology back towards the terrestrial environment. About 100 PBq ^{137}Cs was released and large parts of Europe became contaminated. The most serious consequences occurred in the FSU, where in particular the contamination with ^{131}I have had measurable health consequences in the form of juvenile thyroid cancers (EC, IAEA, WHO, 1996). From a radioecological point of view the Chernobyl accident draw special attention to the vulnerability of seminatural and natural ecosystems such as those known e.g. in the Nordic countries (Dahlgaard, 1994).

The nineties

In the recent years the radioecological interest has focused on dose-reconstruction and site restoration in the FSU. When the "cold war" ended by the eighties a fruitful co-operation between radioecologists from the FSU and the western world was initiated not least as a result of the efforts by the International Union of Radioecologists (IUR). By support from the Commission of the European Union a number of sources to environmental radioactive contamination in the FSU was identified (Aarkrog *et al*, 1994). In particular the sites for nuclear weapons plutonium production at Cheliabinsk (MAYAK), Tomsk and Krasnoyarsk were essential. But the widespread dumping activities in the Arctic Ocean at Novaya Zemlya carried out by the Navy of the FSU also attracted attention (Joint Russian-Norwegian Expert Group, 1994). International studies such as the Arctic Monitoring and Assessment Programme (AMAP) and the IAEA's International Arctic Seas Assessment Project (IASAP) have recently been dealing with these problems.

Recent trends in radioecology

New environments

Most of the knowledge on radioecology has been achieved from temperate environments. In the above mentioned AMAP study interest on Arctic radioecology is emphasised. There is however a need for more knowledge on radioecology of tropical and subtropical ecosystems. The recently launched IAEA study of the French Mururoa and Fangataufa nuclear tests sites may improve our

knowledge on in particular tropical marine ecosystems. There is however still a lot to do for radioecology e.g. in terrestrial continental tropical ecosystems.

Multi-pathway studies

There may within radioecology be a tendency to focus on one environment only, typically either the terrestrial or the marine. Radioecologists are often specialised so they deal with only one of these ecosystems. It may be important in the future radioecological research to consider the whole global ecosystem as an entirety. Throughout the years pathways between atmospheric - terrestrial - marine ecosystems have been identified e.g. seaspray as a way to transfer transuranic elements from sea to land. In connection with radioecological studies of the Arctic seas the possible transport of contaminated coastal sediments by seaice has been discussed. The radioecology of urban systems may be another field of new research and in this context technological pathways may be mentioned for instance the ability of traffic to resuspend deposited radionuclides. Another technological pathway could be transfere of radionuclides through the forest industry (paper, furniture, building materials) or transfer of radionuclides from the sea to the terrestrial ecosystem through the manufacturing for instance seaweed to alginates or use of industrial fish for folder in husbandry.

Less well studied radionuclides, new methods

Radioecology has primarily dealt with those radionuclides which give the highest doses i.e. ^{137}Cs, ^{90}Sr, $^{239,240}Pu$ and ^{131}I. The radioecological knowledge on other radionuclides is less comprehensive. In connection with disposal and longterm storage of radioactive waste very long-lived radionuclides such as ^{99}Tc, ^{129}I and ^{237}Np are sometimes mentioned. The knowledge on especially ^{237}Np may need to be improved. The recent development of mass spectrometric methods, such as High Resolution ICPMS, may be a useful tool in such studies. Determination of transuranic isotope ratios by ICPMS for source identification is another promising research field within radioecology. It may e.g. be possible to distinguish between power reactor debris and weapon grade plutonium.

Experimental radioecology

Within the field of experimental radioecology there is a need for a better understanding of the mechanisms which are responsible for the ability of certain brown algae e.g. fucus sp. to concentrate certain radionuclides from seawater with several orders of magnitude e.g. Tc. It is also worthwhile to

continue studies of radionuclide uptake by different sorts of for instance barley and perhaps by genmanipulation in the future be able to influence the radionuclide uptake by vegetation.

Modelling

A new generation of environmental radioecological models should be developed. Such models might be based on neural networks, where the models "learned" from experimental data or field observations and in this way by a "feed back" procedure were currently improved.

Application

Studies of environmental radioactivity may be used for dose-reconstruction. This is for instance done in the FSU, when measurements of sediment cores are used for the evaluation of contamination levels in river systems in previous periods, when measurements were missing or incomplete.

Radioecoloical data may together with the Geographical Information System (GIS) be applied for identification of areas, which are especially sensitive to radioactive contamination. Such information are useful for preparedness planning and may also be applied in connection with siting of nuclear installations.

Information and education

It is a deplorable fact, that although radioecological studies have been carried out now for more than fifty years and although the knowledge on environmental radioactive contamination is superior to that of most other environmental pollutants, the knowledge and information on radioactivity in the general population including politicians, news media and many scientists have not improved significantly over the last many years - on the contrary.

Hence it is a continuos obligation for the radioecological community to inform on the behaviour and impact of radionuclides on man and environment. This information should be dissipated at all levels: to scientists, politicians, administrators, schoolchildren, journalists.

In case of a future major nuclear accident much unnecessary human suffering could be avoided, if the present poor information level of the public was improved significantly. This was a lesson learned from the Chernobyl accident ten years ago.

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Radiation protection.
Development and trends.

Jan Olof Snihs 1996-08-30
Nordic Radiation Protection Society Meeting
Reykjavik, Iceland, 22 - 29 August 1996

1. Basis for judgement of risks.

- radiobiology
  - epidemiological evaluations - lung, breast, thyroid cancer and leukemia
  - hereditary effects - multifactorial effects
  - combined effects of radiation and other agents - review analytical models
  - low-level radiation - ongoing discussions
  - effects of radiation on the environment - basis for judgement of environmental effects

- risk comparisons
  - continued interest

- new areas of concern
  - future generation - criteria, assessments, time of interest
  - environment - analysis of possible problems
  - potential exposures - rad. prot. and safety integrated
  - uncertainties - barriers, reduncies, safety indicators, reference biopheres etc leading sometimes to pragmatic approach like 10.000 years for quantative assessments followed by more qualitative judgements with 1 million years as a threshold for any interest/responsibility of future

2. Sources

- Nuclear power
  - releases - increased attention
  - occupational exposures - continued attention, more international integration by data banks, dose passport etc.
  - emergency preparedness - exercises, harmonization
  - waste disposal - information, safety analyses, criteria
  - decommissioning - planning

- Other industrial sources
  - level indicators - constant numbers and strengths. About 50% of all
  - density indicators - minor increase. About 20% of all
  - thickness gauges - constant numbers. Mainly Kr-85. About 10% of all
  - thickness gauges X-rays - constant
  - humidity - new uses
  - mobile equipment - roentgenfluoroscens analysis, increased number

For all these very few accidents, occasionally incidents.
Mainly inherently safe, no need of special competence except for mobile equipments

- gammaradiography - continous decrease of numbers
- roentgenradiography - continous decrease of numbers
- other uses - few

- Medical sources
  - follow the development of techniques and equipments to foresee possible problems
  - emphasis on the need of improved competence in radiation protection for radiologists
  - reference levels and need of dose assessments
  - systematic reviews of medical practices and improvement of the technique from radiation protection point of view

- Sources for research in universities etc.
  - emphasis on the need of improved competence in radiation protection for students and scientists

- Schools
  - no significant changes

- Consumer products
  - constant or increased use. Possible waste problems if the routines of handling and disposal of ordinary waste are changed e.g. to be selective

- Natural sources
  - radon in houses - continued efforts, the hard way
  - thoron in houses - some efforts
  - radon in water - continued efforts
  - radium etc. in water - continued efforts
  - radiation from building materials - continued interest
  - radiation in aircraft operation - measurements, guidance
  - other work activities of concern e.g.
    - residues

- Accidents and contaminations
  - continued follow up of Chernobyl accident consequences for the purpose of public information, finding unexpectations and of research

- Non-ionizing sources
  - solar radiation - continued attention

- other UV - continued attention
- ELF - continued interest

3. Regulations and supervision

- International recommendations
  - publ. 60 ICRP - to be considered
  - other ICRP publications to be expected

- EU regulations
  - BSS - to be implemented
  - "Patient" Directive - to be completed and implemented
  - "External Workers" Directive - to be implemented
  - others

"News"
  - reference levels for medical exposure
  - identification of work activities of concern, nat. rad.
  - doses to population; estimations, ALARA documentation
  - dose limits
  - exemption levels
  - clearance levels
  - potential exposure

- Supervision
  - "sustained"
  - requirements on competence, training
  - spot control

- Dosimetry
  - from film to TLD more common

4. Political, societal and economical factors

- Political influence
  - more political influence by instructions
  - more dialogues with politicians

- Public opinion
  - more information related to waste disposal, Rn in houses, various other environmental radiations
  - information on emergency preparedness for specific groups
  - strengthen confidence and reliability by competence, integrity, unbias, legal security

- Financial constraints
    - continued limited resources, the decrease levels off
    - increased requirements on prioritization, efficiency and quality

5. International cooperation

- EU - by necessity for memberstates
- ICRP - several recommendations to be considered
- IAEA - significance of TC programme increases, conventions
- NEA - focuses to more specific issues: risks, societal factors, consequences of scientific and technical develpoments etc.
- UNSCEAR - UNSCEAR 2000
- Eastern Countries - continued cooperation and support
- Nordic Cooperation - continues

6. Accidents

- emergency preparedness
    - criteria for interventions
    - exercises, national and international

- harmonization efforts regarding intervention levels, mutual understanding of national differences

- early warning system - expanded

- environmental measurements, communication systems, exchange of information
    - continued and expanded cooperation

Note of precaution

The above is based on "qualified observation and guess" of development and trends in mainly northern West Europe countries, particularly the Nordic countries, to day and 5 - 10 years in future.

EMERGENCY PREPAREDNESS, DEVELOPMENT AND TRENDS

Annuli Salo

Looking back regarding emergency preparedness important development can be seen in the past ten years.

Immediately after the Chernobyl accident two international conventions {International Atomic Energy Agency,IAEA) were established, namely that vi' "Early warning" (Convention on Early Notification of a Nuclear Accident) and that of "Assistance" (Convention an Assistance in Case of a Nuclear Accident or Radiological Emergency). IAEA has also established a reporting mechanism EMERCON, for assisting in the implementation of the conventions. Further more many coutntries, including Nordic Countries established bilateral agreements on early notification and exchange of information with neighbouring countries. Also European Union (EU) has its early warning arrangements and reporting system ECURIE

Much confusion was experienced after the Chernobyl accident regarding implementation of protective actions and application of the existing international guidance, which was translated to the national plans only in a few countries. The process of clarifying and further developing international guidance, mainly in International Commission on Rardiological Protection (ICRP), IAEA and EU, lasted for several years and the strong requirements to harmonize emergency response in different countries were not met quite as expected. However, the above organizations now have the same main principles of justifying the intervention actions and optimizing the form and scale of the protective measures.

These organizations have also developed the methodology how to implement the principles, thus contributing to similar responses by different countries in accident situations. Some important aspects in this development work are:
- making clear that intervention levels should be derived from the justification and optimization principles, not from dose limits,
- international intervention levels for pecific protective actions have been developed but they can only be generic in nature. In some situations specific intervention levels may be necessary , and such specific intervention levels need to be developed specifically for protective actions in a specific accident and site. At the implementation phase specific intervention levels may be desired to check the applicability of the generic ones,
- the quantity to be used for intervention levels is avertable dose except for deterministic effects for which it is projected dose.
- since intervention levels expressed in avertable dose are not practicable, action levels to trigger protective actions need to be derived from intervention levels or directly by optimization. Again international action levels can only be generic and they are even more dependent on accident and site conditions, if the same avertable doses are expected or aimed at in different accident and site conditions, 'They arc expressed as activity concentrations or dose tales, in other words as measurable quantities. To harmonize actions in different countries such generic action levels ate desirable. For instance for food in international trade such values have been adopted by Codex Alimentarius Commission, although their nature is rather of interdiction W an action levels. Strictly speaking such international "action levels" or "interdiction levels" can hardly be established on purely radiological grounds as radiological justification and optimization would result in different numerical values in different. situations. A political decision is normally needed and other factors also taken into account, if same numerical values for action levels are wanted for different situations.

To implement urgent actions emergency response plans are necessary and they should include specific action levels or other criteria. to trigger protective actions. Also long term protective actions equire planning in advance, but the documentation of such planning should rather include broad basic material to assist in decision making when the actual situation is known, than rigid plans.

It is getting recognized that proper information to the public is decisive for successful implementation of most protective actions. How to get information through to the public is far from being solved.

Without exercising the implementation of plans and various functions including decision making, one can hardly expect succeeding. In addition to national exorcises regional and international exercises are carried out.

As a summary regarding the past it seems to me that international guidance can not be expected to give more "recipes" to national planning before national studies and planning has been carried out to the level of present guidance. Regarding efforts to harmonize internationally the responses m different countries one has to know the implications in ones own country before making decisions on e.g. joint action levels.

Looking now forward from the Nordic perspective it seems to me that a number of useful tasks can be carried out simultaneously or co-ordinated at national and Nordic levels to assist national efforts and test possibilities for broader joint approaches in emergency response. Such issues which 1 consider important are:
-jointly develop methodologies for measurements, assessments, reporting,
- carry out studies on central protection issues in national groups involving all parties acting on the subject in an emergency situation in order to create networks and involve the best expertise .Plan and co-ordinate the work on the Nordic level,
- when ever ready, it is important that the authorities make formal decisions ,positive or negative, concerning joint strategies, otherwise motivation to carry out such studies suffers.

In the Nordic co-operation it is important to handle emergency response as an entity from threat situation until decision and implementation of protective actions.

The threat situations are differernt for the different Nordic countries. Therefore the threats should be carefully analyzed for each country and suitable scenarios for reference accidents developed. The joint Nordic features can he identified and used to find joint approaches and the differences in situations and in necessary responses can be explained to decision makers and to the public.

Work is underway on a joint intervention policy, hopefully with generic intervention levels and explaining their limitations. Flexibility is necessary in such a policy. A formal decision on such a policy is important if joint approaches are wished in other aspects of emergency preparedness.

For urgent protective actions after a. reactor accident, in particular close to the accident site, criteria based on plant conditions a e vital. Even fi not all Nordic countries have nuclear power plants more work could be done to understand and further develop such criteria.

Studies aiming at derivation of action levels for urgent and longer term protective actions are necessary to improve emergency preparedness further away from the site and later after an accident. Developing action levels involves work on accident and site specific parameters, modelling of environmental transport of radionuclides. Case studies for different types of reference accidents and sites as well as uncertainty analyses bring realism to planning.

One can not only have in mind pure radiation safety and scientific-technical a aspects in

mastering an emergency situation. Other relevant aspects such as economical, societal and psychological need to be considered. CO-operation and networks need to be extended to cover such expertise as well.

Consequently also the basis for overall decision making should cover :

- direct and indirect economic consequences of protective actions; economic models and data bases,
- social impact of protective actions; demographic data. bases quantification of psychological factors
- feasibility of protective actions; man power, technical, legal and societal

There are many levels m administration like experts, expert authorities, who often make recommendations and then political decision makers, who have tile final word. This should be taken into account while developing decision models. Particularly important is to develop:

- "decision model" to be used by authorities for longer term protective actions in advance planning, and adjustable to decsion making after an accident
- "presentation model" describing how complicated technical material should be presented to decision makers .for multifactor decisions.

Continuous competence building is a necessary prerequisite for emergency preparedness. The Chernobyl accident demonstrated well in many countries how this aspect had been neglected. There will always be new people involved and techniques will need to be modernized. The best ways to take care of competence building is through studies on and updating of measurement techniques, dose assessments, modelling, communication, iniormation e.t.c..

Carrying out exercises is an efficient way to test and improve emergency preparedness. Proper system fox' feed-back from exercises to emergency plans and planning should be developed. W Exercises are expensive and the lessons learned during them should be utilized much more effciently than so far.

Emergency preparedness is never complete and the next accident will certainly differ from the previous ones therefore high standard of knowledge and flexible plans are vital for coping, if an accident happens.