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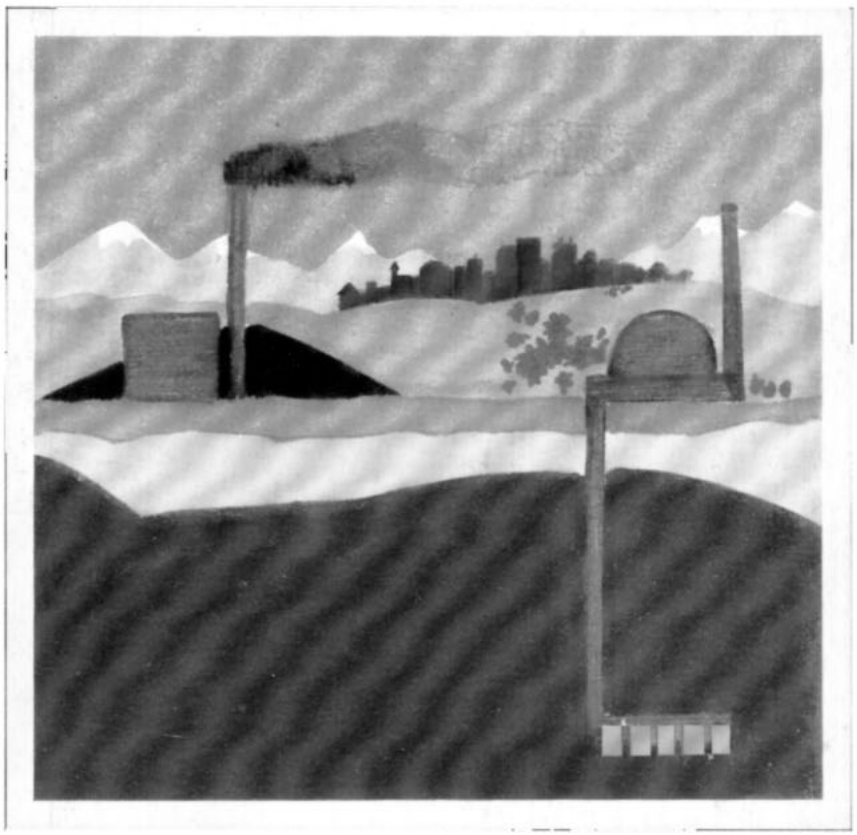

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NATURAL RADIATION, NUCLEAR WASTES and CHEMICAL POLLUTANTS

Final Report of the NKA Project RAS 430

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ABSTRACT

Doses from natural radiation to the population in the Nordic Countries are summarized and man made modifications of the natural radiation environment are discussed. An account is given of the radiological consequences of energy conservation by reduced ventilation. Risks from possible future releases of radioactivity from final repositories of spent nuclear fuel are compared to the risks from present natural radioactivity in the environment. The possibilities for comparison between chemical and radiological risks are discussed.

Key words Chemical effluents, comparative evaluations, energy conservation, fission product release, fossil fuels, natural radioactivity, nonradioactive waste disposal, radiation doses, radioactive waste disposal, radon, risk assessment, Scandinavia

PREFACE

This report is part of the safety program 1985-89 sponsored by NKA, the Nordic Liaison Committee for Atomic Energy. The work has been financed in part by The Nordic Council of Ministers, in part by the participating organizations: The Norwegian National Institute of Radiation Hygiene (SIS), The Finnish Centre for Radiation and Nuclear Safety (STUK) and The Swedish National Institute of Radiation Protection (SSI). Terje Christensen, SIS, has acted as the project leader, Raimo Mustonen, STUK, was project leader in the preliminary project. The other participants in the reference group have been Hans Ehdwall, SSI, Erling Stranden, SIS, (presently RADFORSK A/S) and Arne Sörensen, Risö National Laboratory, Denmark, later replaced by Heinz Hansen, Risö National Laboratory. Each participant had the responsibility for separate working reports. The working reports are published as SIS-report 1990:1 and can be obtained from SIS, Postboks 55, N-1345 Østerås, Norway.

Several consultants have also contributed to the present work: Christel Benestad, Center for Industrial research (SI), Oslo, Jan Fuglesvedt, Council for Environmental Studies, University of Oslo, Vesa Suolanen and Timo Vieno, Technical Research Centre of Finland, and Gustaf Åkerblom, Swedish Geological Consultants.

Summary

In this project the dose to the Nordic population from different sources of natural radiation is investigated, and also how man-made alterations influence the dose picture.

Doses from natural radiation vary within a wide range in the Nordic countries. Thus, there is a factor of 4 between the average doses to the population of Iceland and to that of Sweden. For individuals a factor of 1000 between the highest and lowest doses can be found. An important factor is the large variation in the content of natural radionuclides in different geological materials. The largest single contributor, however, is radon in dwellings.

Our natural background radiation is modified due to human activities such as industrial production, energy production, use of fertilizers, tightening of dwellings, etc. Some of these human activities also bring about other inconvenient effects such as chemical hazards. Even though it is difficult to compare radiological and chemical risks, models for such comparisons can be used if some limitations are recognized.

Production of electric power from peat and coal causes an increase of radiation doses, not only through emissions but also from waste disposal. Disposal of spent nuclear fuel brings about a potential for increased radiation doses in the future. Model calculations show that releases from a repository with spent nuclear fuel may occur after millions of years, and that they at that time may cause minor pertubations in the activity content of various environments. Dose estimates may be meaningless on such a time scale, but for the purpouse of comparison it can be calculated that possible releases may give rise to small doses compared to the natural radiation doses of today. In fact, the future population doses will be much more dependent on local geological conditions then these calculated releases.

To-day, the use of fertilizers causes additional radiation doses, since they increase the natural radionuclides in the soil. For the matter of comparison, to-day's increase due to fertilizers is larger than what can be calculated for future effects due to leakage from a repository with spent nuclear fuel.

Waste from coal- and peat fired power plants has very little effect on our radiation environment, but the activity is still in the same order of magnitude as what can be expected from a repository of spent nuclear fuel. With respect to normal operation, occupational radiation doses as well as emission of radioactivity occur constantly from power plants, be it nuclear or coal/peat fired ones. In this aspect, it is interesting to note that emissions from coal and peat powered stations in the Nordic Countries cause doses more than 100 times higher than doses caused by emissions from the Nordic nuclear power stations during normal operation. It should also be noted that some workers in underground hydro-electric power stations may receive large radiation doses from radon progeny.

Risk potentials and person equivalents have been used to compare the detriment from spent nuclear fuel to that of chemical waste produced from fossil fuels. These evaluations do not take into account the probability for a release to the environment but should only be regarded as a measure of the need to take care of the two types of waste in a safe manner. Only potential direct health risks are taken into account in the calculations, while possible effects on nature that could be caused by the chemical pollutants are not considered. Waste from coal-powered plants clearly has the highest potential risk among the fossil fuels, and depending on what set of limits we use, the chemical risk potential of such waste could be in the same order of magnitude as the radiological risk potential from nuclear waste. This suggests that the problem of waste disposal is a problem not only related to nuclear power. This is even more true when the possible detrimental chemical environmental effects of fossil fuel are taken into account.

An alternative to energy production is energy conservation. If we conserve energy by reducing ventilation in dwellings this causes increased doses from inhaled radon daughters. It is estimated that the doses from indoor radon have nearly doubled in the Nordic countries during the last 20 years. This is due to reduced ventilation. The corresponding increase in collective dose is moore than 100 times as high as the total collective dose from all energy production in the Nordic countries.

The radon concentration could be lowered by an increase in air exchange rates, but by this method, the resulting reduction in radiation dose will be very costly. From a radiation protection point of view, other mitigation methods should therefore be preferred. However, insufficient ventilation also causes other types of negative effects, and if all effects are taken into account, increased ventilation may after all be cost effective, especially if heat exchange systems are used. Further energy conservation by reducing ventilation rates should definitly be avoided in the future both due to the radiological consequences and in view of the indoor air quality in general.

Sammanfattning

I detta projekt har stråldosen från olika naturliga strålkällor till nordens befolkning undersökts och även hur mänskliga förändringar påverkar dosbilden.

Doser från naturlig strålning varierar inom vida gränser i de nordiska länderna. Så till exempel är den genomsnittliga befolkningsdosen 4 gånger högre i Sverige än på Island. För enskilda individer kan det skilja så mycket som 1000 gånger mellan det lägsta och det högsta värdet. En viktig faktor i det sammanhanget är den stora variationen avseende förekomsten av naturligt radioaktiva ämnen i olika geologiska mineral och bergarter. För det största enskilda bidraget svarar dock radon och dess dotterprodukter i bostäder.

Vår naturliga strålningsbakgrund påverkas av olika mänskliga verksamheter, som industriell verksamhet, energiproduktion, användandet av gödningsämnen, energisparande genom tätning i bostäder mm. Några av dessa verksamheter medför även andra olägenheter, tex kemiska föroreningar. Trots att det kan vara svårt att adekvat jämföra radiologiska och kemiska risker, kan inom vissa gränser modeller för sådana jämförelser användas.

Elproduktion baserad på kol och torv medför ökade stråldoser både genom luftutsläpp och läckage från avfallsupplag. Deponering av använt kärnbränsle medför en viss risk för ökande stråldoser i framtiden. Modellberäkningar visar att utsläpp från ett kärnavfallslager sker först efter miljontals år och att de då endast förosakar mindre störningar, vad avser aktivitetsinnehållet i olika miljöer. Att i en sådan tidsskala utföra dosberäkningar är inte meningsfullt, men beräkningarna visar klart att utsläppen till omgivningen endast kommer att medföra stråldoser som är små i förhållande till vad den naturliga strålningen förorsakar. Geologiska förhållanden kommer att i högre grad påverka befolkningsdoserna än sådana utsläpp.

Användandet av konstgödning i skog och mark bidrar till förhöjda stråldoser beroende på dess innehåll av radioaktiva ämnen. Som jämförelse kan nämnas att dagens ökande stråldoser beroende på användandet av konstgödsel, är större än vad som kan uppskattas från beräkningar vad gäller läckage från kärnavfallslager i framtiden.

Avfallet från kol- och torveldade kraftanläggningar har liten inverkan på vår strålningsmiljö, men mängden radioaktiva ämnen är ändock i samma storleksordning som vad som uppskattats från ett kärnavfallslager. Vid normaldrift förekommer yrkesbestrålning såväl som utsläpp av radioaktiva ämnen både när det gäller kärnkraftanläggningar och kol- eller torvanläggningar. Det kan i det här sammanhanget vara värt att notera att utsläppen från kol- och torveldade kraftverk i de nordiska länderna förorsakar mer än hundra gånger så höga stråldoser som de nordiska kärnkraftverken. Det kan även noteras att arbetstagare i vissa underjordiska vattenkraftverk får höga stråldoser på grund av förekomsten av radon och dess dotterprodukter i kraftverken.

Riskpotentialer och personekvivalenter är begrepp som använts för att kunna jämföra den totala påverkan av använt kärnbränsle, med avfall från fossileldning. De beräkningar som gjorts beaktar dock inte sannolikheten för framtida utsläpp till omgivningen, utan utgör mer ett mått på behovet av att ta hand om olika typer av avfall på ett godtagbart sätt. Vi har enbart studerat potentiella hälsorisker vid beräkningarna. Andra tänkbara effekter på naturen förorsakade av kemiska föroreningar har ej beaktats. När det gäller fossila bränslen svarar koleldade anläggningar för de klart största olägenheterna, och beroende på vilka avgränsningar som görs så är den kemiska riskpotentialen för koleldade kraftverk av samma storleksordning som den radiologiska riskpotentialen för kärnkraft. Detta visar att problemet med avfallsdeponering inte enbart är relevant för kärnkraft. Om den potentiella kemiska påverkan på miljön räknas med, framstår det senare än mer tydligt.

Ett alternativ till energiproduktion är energisparande. Om vi spar energi genom att minska ventilationen i våra bostäder kan radonhalterna och därmed stråldoserna öka. Man har uppskattat att stråldoserna från radon i våra bostäder nästan har fördubblats de senaste 20 åren och detta beror nästan helt på minskad ventilation. Kollektivdosen på grund av detta har ökat till att vara nästan 100 gånger högre än stråldosen från vår samlade energiproduktion i Norden. Radonhalterna inomhus kan minskas genom ökad ventilation, men genom sådana åtgärder blir kostnaden för motsvarande dosbesparing

hög. Från strålskyddssunpunkt är därför andra åtgärder motiverade. Men otillräcklig ventilation har också andra nackdelar och om man tar hänsyn till alla negativa effekter kan ökad inomhusventilation ändock vara kostnadseffektiv. Detta gäller speciellt om system med till exempel värmeväxlare används. Det måste bestämt avrådas från att i framtiden använda sig av ventilationsminskning inomhus som metod för energisparande. Detta gäller både på grund av de radiologiska konsekvenserna och på andra försämringar av luftkvaliten i övrigt.

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1. Introduction

In addition to releasing artificial radionuclides into the environment, we have also to an increasing degree modified the radiation we receive from natural sources. When natural radiation doses are increased due to some kind of human activity, we often call it "technologically enhanced natural radiation".

The boarder line between "true" and "technologically enhanced" natural radiation is often vague. The radiation doses we receive from radon progeny are e.g. dependent on how we build our houses, the air exchange rates and our living habits, in addition to geological and climatic conditions. Some of the doses from radon progeny are therefore caused by different technical and human activities and may thus be regarded as technologically enhanced. On the other hand, it should be regarded as natural to live in a dwelling, and a certain level of radon must thus be regarded as natural. In addition, we also have a certain, low radon level outdoors.

In some cases a release of natural radioactive nuclides to the environment and/or an increase in radiation doses from natural radioactivity to humans may occur due to different types of industrial processes. Energy production by burning coal, peat or natural gas will e.g. result in natural radioactive effluents to the environment. The use of phosphate fertilizers may also add natural radioactivity to the soil, resulting in increased doses from external radiation and from the intake of food grown on these soils. In some cases by-products from different industrial processes are used as building materials and sometimes these building materials have enhanced concentrations of natural radionuclides, compared to traditional building materials.

Different parts of the nuclear energy cycle will release some radioactive material to the human environment. The ionizing radiation that people are exposed to in this way, is in no aspect different from the radiation we receive from any natural radioactive nuclides in the environment, and a living cell can in no way "know" whether the radiation is caused by a natural or an artificial source.

Since there is no difference in health risk from a certain radiation dose whether it originates from natural or artificial sources, it is possible to compare the radiological consequences from different parts in the nuclear fuel cycle to radiation risks caused by natural sources. **It also seems correct to use the same basic radiation protection philosophy both for nuclear power and for technologically enhanced radiation.** This is perhaps especially valid when comparing different forms of energy production as both the benefits and risks of the different energy forms are comparable.

Industrial processes, such as energy production, may also result in chemical, non-radiological pollutants. Such pollutants may have detrimental effects on both man and his environment. It is however difficult to compare chemical and radiological pollutants as their effects may be completely different. If such comparisons are to be performed, it is necessary to define what is meant by "health risks", and also to limit the discussion to clearly defined areas where the comparison is appropiate. Even though such comparisons never will be quite satisfactory, they should at least be discussed, especially when different forms of energy production are evaluated. However, it is of crucial importance that the bases and limitations of each comparison are clearly defined and discussed.

In a comparison of this kind, the most complete procedure would be to go through the whole nuclear fuel cycle and compare its radiological consequences to the consequences of other types of activity. In this respect, the present project is limited to evaluate the consequences of an ultimate nuclear waste disposal to the consequenses of natural radiation. Disposed waste will remain for millions of years with progressively reduced radioactivity. There is no way to predict what the world will be like in such a time perspective. Thus there is a need to define indirect indexes for the radiological health impact. This has been considered and is discussed further in the report.

An alternative to energy **production** is energy **economization.** Energy conservation could in itself result in different negative social, economic, environmental and physiological consequences. However, it would be beyond the objective of this report to discuss these aspects in detail. The discussion is therefore limited to the **radiological** consequences of energy conservation by a reduction of air exchange rates in dwellings.

2. Natural radiation in the Nordic Countries

2.1 Radiation doses

Man has always been exposed to natural radiation. We receive radiation doses from cosmic radiation, internally deposited radionuclides, inhaled radon- and thoron daughters and gamma radiation from radionuclides in the ground and building materials. Collective doses have been estimated for the Nordic population. In sub-report 1 (1), the individual and collective doses from the following sources are discussed: external cosmic radiation, internal radiation from cosmogenic radionuclides, terrestrial external radiation indoors and outdoors, radon- and thoron-exposure indoors and outdoors and internal radiation from long-lived natural radionuclides.

2.1.1 Cosmic radiation

The cosmic radiation varies by the geomagnetic latitude, the altitude and the shielding properties of the buildings. The variations within the Nordic Countries are small, and the mean effective dose equivalent in the Nordic Countries is estimated to 0.3 millisievert per year (mSv a^{-1}) (1). This equals an annual collective dose equivalent for the entire Nordic area to 6800 mansievert (manSv).

2.1.2 Cosmogenic radionuclides

Cosmic radiation produces radioactive nuclides, the so-called cosmogenic radionuclides, through nuclear reactions in the atmosphere. The main cosmogenic radionuclides are ^{3}H, ^{7}Be, ^{14}C and ^{22}Na. Both the external and internal dose contributions are very small compared to the doses from other sources of natural radiation. The variations in individual doses are small, and there are no significant geographical variations. Mean effective dose equivalent is estimated to be 0.015 mSv a^{-1} corresponding to a collective effective dose equivalent of 330 manSv a^{-1} (1).

2.1.3 Terrestrial gamma radiation

Terrestrial gamma radiation originates from natural radionuclides in soil, bedrock and in building materials. From a radiological point of view, ^{40}K and the members of the ^{238}U and ^{232}Th-series are the most important radionuclides.

The outdoor gamma radiation is dependent on the contents of these nuclides in soil and bedrock. Depending on local geology, there are relatively large regional variations in outdoor gamma dose rates. In Table 2.1, the mean dose rates from terrestrial gamma radiation outdoors in the Nordic Countries are given.

Table 2.1. *Population averaged outdoor dose rates from terrestrial gamma radiation in the Nordic Countries in nanogray per hour. Cosmic radiation not included (1).*

Country	Absorbed dose rate in air ($nGy\ h^{-1}$)
Denmark	37
Finland	71
Iceland	28
Norway	73
Sweden	80

As seen from the table, there is a clear difference between Iceland and Denmark on one hand, and Finland, Norway and Sweden on the other. This is due to differences in geology.

Even within the individual country, there are variations in external gamma radiation depending on local geology, and in some "hot spots" there are strong local anomalies.

External gamma radiation in buildings depends mainly on the radionuclide contents of the building materials. In Table 2.2, measured values for different types of houses are summarized.

The dose rates are significantly higher in houses with "stone based" building materials than in wooden dwellings. An interesting special case is a light weight concrete based on alum shale earlier used in Sweden and Denmark. The radium content of this concrete is very enhanced (620-1300 $Bq\ kg^{-1}$) compared to "normal materials" (1).

Taking into consideration the relative use of different types of materials in the Nordic Countries, the following population averaged dose rates due to indoor gamma radiation have been estimated:
Denmark 63 $nGy\ h^{-1}$, Finland 80 $nGy\ h^{-1}$, Iceland 23 $nGy\ h^{-1}$, Norway 79 $nGy\ h^{-1}$ and Sweden 110 $nGy\ h^{-1}$.

Table 2.2. *External gamma radiation in dwellings in the Nordic Countries. Cosmic radiation not included. (1)*

Material	Country	Dose rate in air (nGy h^{-1}) Min	Mean	Max
Clay brick	Denmark	58	65	90
	Finland	82	120	150
	Norway	87	120	150
	Sweden	30	85	280
Concrete	Denmark	39	45	75
	Finland	73	120	160
	Iceland	14	23	32
	Norway	64	110	150
	Sweden	31	120	280
Wood	Denmark	30	36	59
	Finland	32	44	75
	Norway	47	71	110
	Sweden	15	57	210
Shale-based light weight concrete	Denmark	110	180	260
	Sweden	28*	180*	830

* Also contains sand-based light weight concrete.

By assuming that 80% of the time is spent indoors and 20% outdoors, the annual effective dose equivalents in the Nordic Countries is estimated in Table 2.3.

Table 2.3. *Effective dose equivalents for external gamma radiation in the Nordic Countries.(1)*

Country	Individual (mSv a^{-1})	Collective (man Sv a^{-1})
Denmark	0.36	1800
Finland	0.49	2300
Iceland	0.15	300
Norway	0.48	1900
Sweden	0.64	5300

Within each individual country, the doses will vary considerably, mainly due to differences in building materials. In Sweden, it has been estimated that about 5% of the population receives more than 1 mSv a^{-1} from external gamma radiation (1).

The values in Table 2.3 correspond to a collective effective dose equivalent of 11600 manSv per year for the Nordic population.

2.1.4 Radon and thoron

^{222}Rn (radon) is produced by the decay of ^{226}Ra. The main sources of radon to indoor air is soil and bedrock, building materials and radon-rich household water.

Normally, the outdoor radon concentration is very low compared to the indoor levels.

In Table 2.4, the mean radon concentrations in dwellings in the Nordic Countries are given.

Generally, the radon concentrations in Finland, Sweden and Norway are very similar, both in mean values and distributions, while the values are significantly lower in Denmark. In Iceland, the values are even much lower than in Denmark.

Table 2.4. *Radon in dwellings. (1)*

Country	Mean radon concentration ($Bq\ m^{-3}$)	Percentage of dwellings above given values ($Bq\ m^{-3}$)		
		>200	>400	>800
Denmark	50	2	0.1	0
Finland	100	10	3	1
Iceland	10	0	0	0
Norway	100	10	3	1
Sweden	100	10	3	1

Within the different countries, large regional variations have been found, and even large local variations are common. The highest values have been found in areas with very permeable underground (eskers), in areas with alum shale and in some granite areas.

Certain areas within the Nordic Countries must generally be regarded as "high radon risk areas". Some such areas are marked on the map in fig 2.1. It is important to stress that there are large variations within these areas, and that other areas also may be high risk areas. The map is only intended to illustrate the magnitude of the problem.

Effective dose equivalents from radon in dwellings in the Nordic Countries are estimated in Table 2.5.

Fig 2.1. Some areas in the Nordic Countries where the probability of enhanced indoor radon levels is much higher than in most other areas. The map has been drawn based partly on data from indoor radon measurements and partly on geological data. No such areas are found in Iceland.

Individual doses vary over a very wide range, and radon concentrations giving effective dose equivalents of more than 500 mSv a^{-1} have been reported in some cases.

Table 2.5. *Effective dose equivalent from indoor radon.(1)*

Country	Individual (mSv a^{-1})	Collective (man Sv a^{-1})
Denmark	1.5	8000
Finland	3.0	14000
Iceland	0.3	1000
Norway	3.0	12000
Sweden	3.0	25000

Outdoor radon concentrations are low, and the outdoor occupancy is short. The effective dose equivalent from outdoor radon is estimated to: 0.035 mSv a^{-1} for Denmark, Sweden, Norway and Finland and 0.004 mSv a^{-1} for Iceland.

The total collective effective dose equivalent from radon in the Nordic Countries is estimated to 60 000 manSv a^{-1}.

^{220}Rn (thoron) is produced by the decay of ^{224}Ra. Normally, thoron is a small problem compared to radon, and the collective effective dose equivalent from inhaled thoron daughters is estimated to 4 800 manSv a^{-1}.

2.1.5 Long-lived radionuclides

Long-lived radionuclides enter our bodies through food, water and air. It is estimated that the effective dose equivalent from an intake of long-lived radionuclides is 0.36 mSv a^{-1} in the Nordic Countries except Iceland. Due to different geochemical factors, the doses are believed to be somewhat lower in Iceland (around 0.3 mSv a^{-1}). The dietary habits are important for the individual doses, especially from ^{210}Po and ^{210}Pb. People having reindeer meat as the main diet could receive more than 1 mSv a^{-1} from ingested ^{210}Po and ^{210}Pb. Also people that regularly drink ground water in certain areas receive relatively high doses from long-lived natural radionuclides. The collective effective dose equivalent from an intake of long-lived radionuclides in the Nordic Countries is estimated to be about 8100 manSv a^{-1}.

2.1.6. Total doses from natural sources in the Nordic Countries

In Fig 2.2, the total individual effective dose equivalent from natural radiation in the Nordic Countries is shown. This illustrates the variations between the different countries.

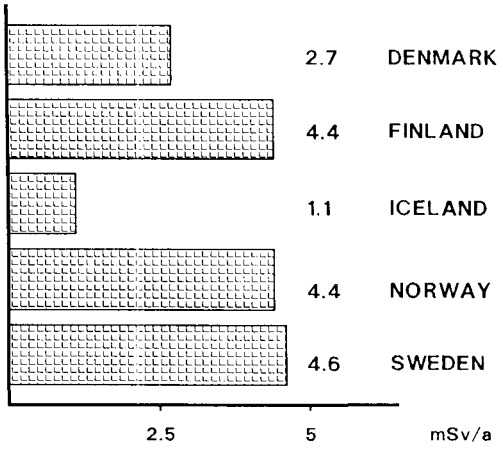


Fig 2.2. Radiation received by individuals in the Nordic Countries: Mean effective dose equivalent from natural radiation.

The contribution of the different modes of expopsure (collective effective dose equivalents) are shown in Fig 2.3. From this figure it is evident that radon exposure is the dominating mode of exposure in the Nordic Countries. Furthermore, the individual doses from radon may in some cases be very high.

2.2 Radiological environments

Many measurements of natural radionuclides have been performed in different environments within the Nordic Countries. Such investigations have often been directed to find anomalies, and the data are therefore not necessarily generally valid. The data that are presented in this report are therefore only to be regarded as examples of more or less typical values for the different environments.

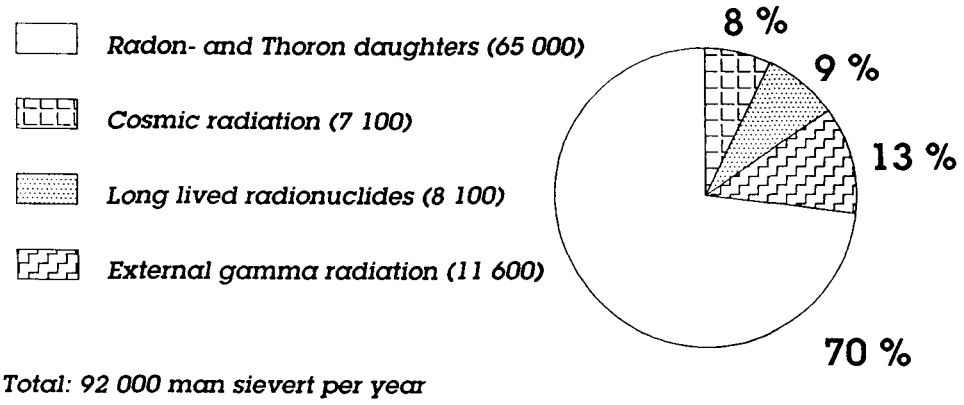


Fig 2.3. Collective effective dose equivalents for the different modes of exposure to natural radiation for the total nordic population.

In the Nordic Countries there are different geological areas. This will also be reflected in the radioactivity concentrations in soil and bedrock. The same is true for ground water, lake water and sediments. Fig 2.4 shows a map of the geology of the Nordic Countries. From a radiological point of view, Denmark and Iceland differ from the other countries. This is also reflected in the radiation doses from natural sources in the different countries.

The following radiological environments have been evaluated:

- Bedrock
- Soil
- Well water
- Lake water and sediments
- Baltic Sea water and sediments.

In Table 2.6, typical values of radioactivity in different soils are shown. Values for different types of rock are shown in Table 2.7 and activity concentrations in well water and in lakes and the Baltic Sea are given in Table 2.8.

Table 2.6. *Natural activity in soils (3)*

Soil type	Activity concentration (Bq kg $^{-1}$)		
	^{226}Ra	^{232}Th	^{40}K
Sand and silt	5-25	4-30	600-1200
Clay	20-120	25-80	600-1300
Morene	20-80	20-80	900-1300
Soils containing alum shale	100-1000	20-80	600-1000

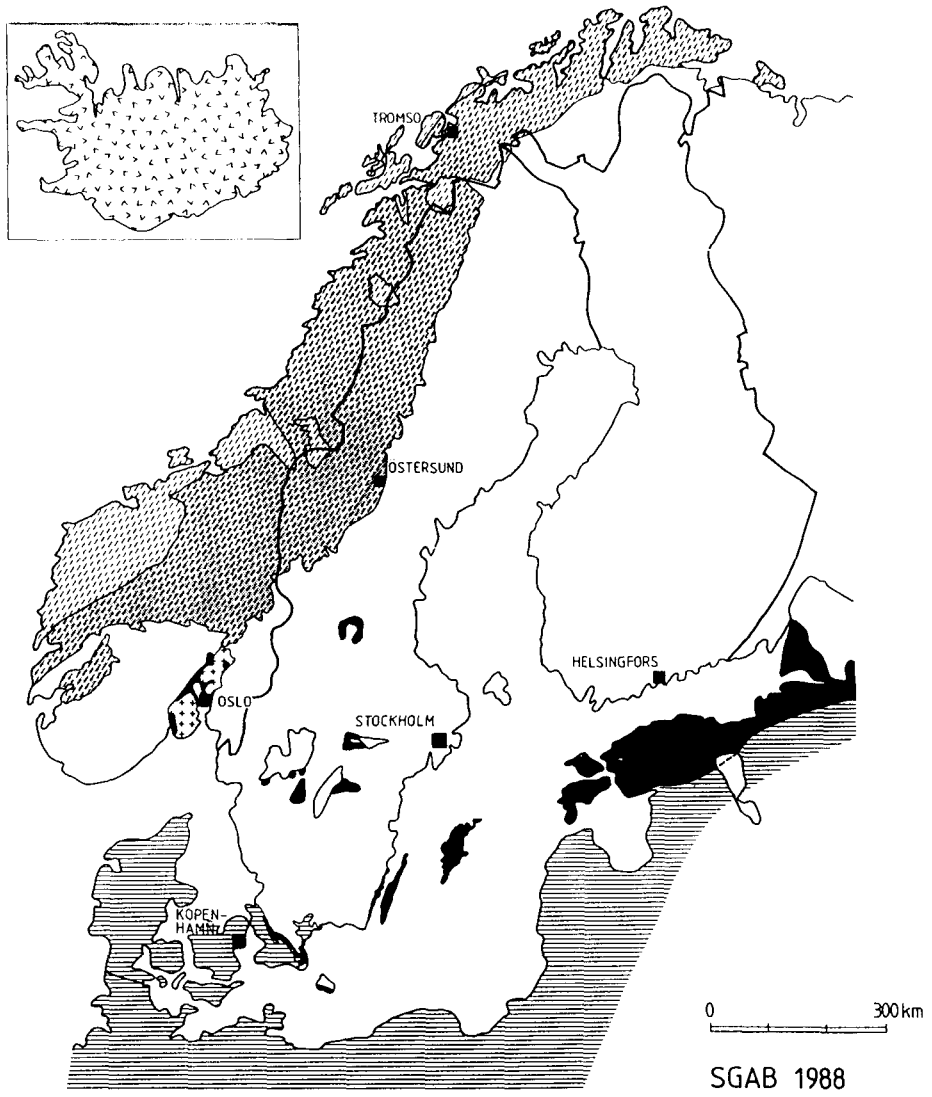
Table 2.7. *Radioactivity concentrations of different types of rock in the Nordic Countries (3).*

Rock type	Activity concentration (Bq kg^{-1})		
	^{226}Ra	^{232}Th	^{40}K
Normal granite	20-120	20-80	600-1800
Thorium and Uranium rich granite	100-500	40-350	1200-1800
Gneis	20-120	20-80	600-1800
Diorite	1-20	4-40	300-1000
Sandstone	5-60	4-40	300-1500
Limestone	5-20	1-10	30-150
Shale	10-120	8-60	600-1800
Alum shale[1]	120-600	8-40	1000-1800
Alum shale[2]	600-4500	8-40	1000-1800

1) Middle Cambrian

2) Upper Cambrian or Lower Ordovician

As seen from Table 2.8, a very large range of values is found in the different environments, and it is difficult to define "typical values". To be able to make some comparisons of the consequences of releases of nuclear waste to these environments, it is necessary to choose some specific values for the calculations. In Table 2.9, the values we have chosen from the data in Tables 2.6 - 2.8 are listed. It is however important to bear in mind that these data may not be representative and that there are very large variations within the same type of environment.



Sedimentary rocks(350-2 Ma).

Sedementary and volcanic rocks and granites(350-245 Ma).

Caledonian mountain range (800-400 Ma).

Caledonian mountain range(> 1000 Ma).

Sedementary rocks - sandstone, shale, alum shale, limestone. (800-400 Ma).

Sedementary and volcanic rocks, granites and gneisses etc. (2800-800 Ma).

Volcanic rocks on Iceland. Mainly basaltic rocks.(20 Ma - present day).

Fig. 2.4. The bedrock of the Nordic Countries.(2) (Ma = million years)

Sediments are formed by deposition of sand and clay particles combined with organic material. The amount of organic material varies as in shales, and the potassium content is dependent on the water salinity. In some cases, the sediments are enriched in uranium and radium (especially associated with precipitation of iron), but in principle, there is no difference between sediments and different soils and rocks. Data on activity in sediments are scanty, and we have, for the sake of the comparisons, chosen the same values for sediments as for soils.

Table 2.8. *Activity concentrations in different aquatic environments (3)*

Environments	Activety concentration Bq l^{-1}
	^{226}Ra
Well water	0.001 - 10
Lake and Baltic sea water	0.0005-0.007

Table 2.9. *Activity concentrations chosen for comparison to nuclear waste.*

Environment	Activity concentration Bq kg^{-1}		
	^{226}Ra	^{232}Th	^{40}K
Soil and sediments	50	50	800
Well water	0.1	-	-
Lake and Baltic Sea water	0.003	-	-

3. Industrial modification of the natural radiation doses

3.1 Mining

The main radiological impact of non-uranium mining is the exposure of miners to elevated levels of radon in underground air. In Table 3.1, the effective dose equivalents to miners in the Nordic Countries are summarized. There are no mines in Denmark and Iceland.

Table 3.1. Effective dose equivalents in millisieverts per year for individual doses and collective doses in mansieverts per year to miners in the Nordic Countries (4)

Country	No of miners	Individual effective dose equivalent mSv a^{-1}	Collective effective dose equivalent manSv a^{-1}
Norway	1130	3.4	4
Sweden	3320	9	30
Finland	1090	2.7	3

Close to some mines, the waste from the mining could give rise to a radiological pollution if the mined minerals are associated with high radioactivity concentrations or if there is an enrichment of radioactivity in the tailings. There are no systematic investigations on such problems in the Nordic Countries, but in one Norwegian area (Fen, near Ulefoss in Telemark) an investigation on the radiological impact of earlier mining activities was performed. This area is especially known for the enhanced thorium concentrations in some types of rock. In Table 3.2, the activity concentrations of some of the wastes from mining in Fen, Norway are shown.

Table 3.2. Activity concentrations of waste rock from mining in Fen, Norway (4)

Type of rock	Activity concentration (Bq kg^{-1}) ^{232}Th	^{226}Ra
Red rock	4 000 (170-12 000)	160 (44-550)
Rauhaugite	560 (160-2 000)	130 (20-290)
Soevite	310 (10-780)	310 (10-1 400)
Tailings	47 000	46 000

It should also be mentioned that there are large piles of waste from alum production in certain areas of Sweden. Alum was produced by burning alum shale, and the wastes have been left and are now scattered over relatively large areas. Building houses on such ground has in some cases caused large indoor radon concentrations.

No evaluation of the radiological consequences of a possible future uranium mining has been performed.

3.2 Energy production

The sub-report 4 (5) deals with natural radiation associated with energy production in the Nordic Countries. The radioactive emissions from coal and peat powered stations are discussed together with the use of by-products from these industries. Geothermal energy, oil and gas, and the occupational exposure in underground hydro power stations are also discussed.

3.2.1 Emissions from coal and peat powered plants.

Radioactive emissions from coal and peat powered plants can cause radiation doses in four different ways:

- Internal doses through inhalation of radionuclides

- Internal doses through contaminated food

- External radiation from radionuclides in air

- External radiation from radionuclides deposited on the ground

In Table 3.3 calculated maximum values are given for the radionuclide concentration in outdoor air caused by average emissions from 1000 megawatt thermal (MW_t) coal and peat powered plants. The effective dose equivalent through inhalation is also given. The inhalation dose is estimated to be 80% of the total dose from radioactive emissions.

By using the data in Table 3.3 together with the effective heat value of coal and peat, the annual effective dose equivalent can be roughly estimated. The results of such estimates are: 8, 4, 2, 0.3 microsievert per year ($\mu Sv\ a^{-1}$) for Denmark, Finland, Sweden and Norway respectively. Peat and coal are not used in significant quantities in Iceland. The collective effective dose equivalent for the total Nordic population is thus about 80 $manSv\ a^{-1}$.

Table 3.3. Max concentrations and individual effective dose equivalents caused by emissions from 1000 MW_t coal and peat power plants.(5)

Nuclide	Coal		Peat	
	nBq m^{-3}	μSv a^{-1}	nBq m^{-3}	μSv a^{-1}
U-238	370		380	
Th-230	310		380	
Ra-226	290	0.36	380	0.40
Pb-210	1400		8400	
Po-210	1400	0.07	8400	0.42
Th-232	220			
Ra-228			230	
Th-228	230	0.51	250	0.55
U-235			14	0.002
TOTAL		1.0		1.4

3.2.2. Natural radioactivity in waste from coal and peat burning.

When burning coal and peat, the unburnable fraction, the ash, will be divided into bottom ash and fly ash. The bottom ash is often in the form of larger pieces of sparingly soluble silicates. The fly ash follows the smoke gas and is removed from this by electro filters. The ash content varies according to the type of coal or peat burnt, and may be in the range 6-15% for coal and 3-8% for peat. As average figures, 12% can be assumed for coal and 5% for peat. The division between bottom ash and fly ash depends mainly on the process of burning. In Finnish power plants it is estimated that there is on average 22% bottom ash and 78% fly ash in coal plants, and 28% bottom ash and 72% fly ash in peat plants.

In Table 3.4 mean radionuclide concentrations in ash from Finnish power plants are shown.

Table 3.4 *Mean radioactivity in ash from Finnish power plants (5)*

	^{238}U	^{226}Ra	^{210}Pb	^{228}Ra	^{228}Th	^{40}K
	Bq kg^{-1} (dry weight)					
Coal:						
Bottom ash	80	100	50	60	60	500
Fly ash	150	160	160	80	80	700
Peat:						
Bottom ash	30	30	85	15	15	350
Fly ash	160	120	1000	50	50	390

The concentrations in individual samples may deviate from the mean value by a factor of 10.

In Table 3.5, the estimated consumption of peat and coal in the Nordic Countries is shown together with estimated values for the produced amount of ash.

Table 3.5 *Coal and peat used in the Nordic Countries and the resulting amount of ash (5)*

	Denmark	Finland	Iceland	Norway	Sweden
			kg a^{-1}		
Coal	8.5×10^9	4.1×10^9	-	3.9×10^8	2.4×10^9
bottom ash	2.0×10^8	1.0×10^8	-	9.4×10^6	5.8×10^7
fly ash	7.2×10^8	3.5×10^8	-	3.3×10^7	2.0×10^8
Peat	-	3.7×10^9	-	-	3.7×10^8
bottom ash	-	2.6×10^7	-	-	2.6×10^6
fly ash	-	6.7×10^7	-	-	6.7×10^6

Some of the coal fly ash is used in concrete. In the NKA project REK-4 (13) it was concluded that fly ash in concrete does not result in a significant increase in radon exhalation from the concrete, but the fly ash will cause a slight increase in external gamma radiation in concrete buildings. The fly ash is estimated to result in an increase in gamma radiation of about 0.1 mSv a^{-1} in blocks of flats having fly ash concrete as the main building material.

It is difficult to assess collective doses from the use of fly ash in concrete. It may, however, give the most significant contribution to the doses from the use of peat and coal.

3.2.3 Oil and gas.

In oil and gas production in the North Sea, salts from over-saturated brines are often precipitated after expansion, and scale is formed on the inner walls of tubings etc. The scale has enhanced concentrations of radium. The equipment must from time to time be cleaned and the scale removed. This creates a certain waste problem and may also result in radiation doses to the workers handling the equipment. Investigations into this problem suggest that the problem today is small, and that both individual and collective doses are low.

By using natural gas in the households, some radon will be released to the indoor air. The resulting doses are very low.

3.2.4 Geothermal energy

In Iceland, geothermal energy is used for heating of dwellings. Hot spring water is used directly as hot water in the households. About 80% of the dwellings in Iceland are heated directly by such water.

Use of this water may cause a radon release to the indoor air. Much of the radon is released from the water between the source and the consumer and the radon concentration in the household water is low (<0.4 $kBqm^{-3}$). This suggests that radon in the household water does not cause significant radon concentrations in the indoor air. Future technological developments are believed to reduce the problem further.

3.2.5 Hydro electric power plants.

In underground hydro electric power stations very high radon concentrations have been measured in several cases. This causes radiation doses to the workers, and in some Norwegian hydro electric power stations, individual effective dose equivalents of more than 50 $mSv\ a^{-1}$ have been estimated in some cases.

The collective effective dose equivalent from radon exposure in Norwegian hydro electric power stations is estimated to be 1.5 $manSv\ a^{-1}$.

3.2.6 Collective doses from non-nuclear energy production in the Nordic Countries.

In Fig. 3.1 collective effective dose equivalents from non-nuclear energy production in the Nordic Countries are given. The figure clearly shows that coal and peat powered plants give the dominant contributions.

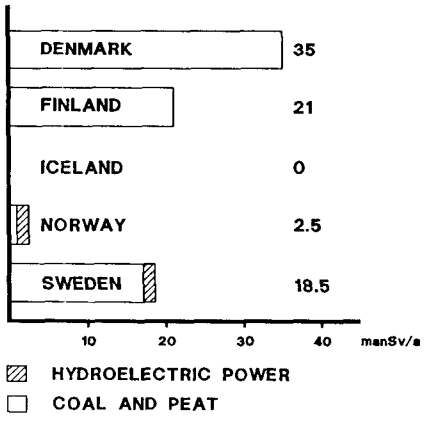


Fig.3.1 Collective effective dose equivalents from non-nuclear energy production in the Nordic Countries.

3.3 Production and use of fertilizers

Intensive use of agricultural soil, as it has been practiced in the Nordic Countries during the last decades, is based on a steady supply of fertilizer. Phospate fertilizers are made from phosphate rock which in some cases have elevated levels of nuclides in uranium series. Through the production the fertilizer get enhanced concentrations of ^{238}U and ^{226}Ra. Potassium fertilizers are rich in potassium, and thus in the radioactive isotope ^{40}K. The use of such fertilizers will thus cause an increase in natural radionuclides in the soil. During production of fertilizer, the radionuclides originally bound to the raw materials will be released. When the fertilizer is spread over the agricultural soil, it causes an increase of the external radiation doses to the population. The food that is produced on the soil, will take up radionuclides from the fertilizer. This will cause increased internal doses through ingestion of the food.

Waste products are produced during the process, and the radionuclide contents of these waste products will also represent a potential source of exposure to the public.

Radiation doses from the production, use and waste products of fertilizers are estimated in the sub-report 5. (6)

Table 3.6 shows the consumption of phosphate and potassium fertilizers in the Nordic Countries.

Table 3.6 *Consumption of phosphate and potassium fertilizer in the Nordic Countries. (6)*

	Total (10^6 kg a^{-1})		Per ha (kg ha^{-1} a^{-1})	
	Phosphate	Potassium	Phosphate	Potassium
Denmark	49	124	19	47
Finland	70	128	29	52
Norway	25	67	29	8
Sweden	44	84	15	29

The external radiation levels is estimated to increase by 0.5 nGy h^{-1} after 75 years of supply of fertilizers. This is an increase in the external dose rate of about 1% compared to the dose rate above unmodified soil. One year of supply is estimated to give an increase of about 0.02 nGy h^{-1}.

The supply of potassium will not have any effect on the internal doses because the potassium contents of our body is under homeostatic control. The increase of internal doses are therefore mainly caused by ^{226}Ra and its daughter products. The increase in internal doses due to fertilizers is estimated to be about 4% of the dose from ^{238}U and its daughter products originating from other sources than the fertilizer. This corresponds to an effective dose equivalent of a few $\mu Sv\ a^{-1}$. The collective effective dose equivalent from internal radiation caused by the use of fertilizer is thus about 50 $manSv\ a^{-1}$ for the Nordic population.

The external doses depend on the time spent on agricultural soil and the increase in dose rate above the soil. If we assume that the populations spend on average 1% of their time on agricultural soil, the resulting collective effective dose equivalent to the Nordic population is lower than 1 $manSv\ a^{-1}$.

From radiological point of view, gypsum is the most important by-product of fertilizer production from phosphoric acid. This is because the main radium content of the phosphate will end in this so-called by-product gypsum.

The utilization of the by-product gypsum as a building material will cause an increase in the radiation doses both through external gamma radiation and through radon released from the gypsum.

It is estimated that the use of 4 tons of the by-product gypsum as a substitute for natural gypsum in a dwelling will cause an increase of about 70 $nGy\ h^{-1}$ in external gamma radiation and that the radon concentration will increase by about 10 $Bq\ m^{-3}$ (6). These values represent a maximum use of the by-product gypsum, and such a use will cause an increase in effective dose equivalent of about 0.6 $mSv\ a^{-1}$, half of which is caused by radon daughters and half by external gamma radiation. Such gypsum is used in modest quantities in the Nordic Countries as a whole. The total use is probably less than 1% of the maximum use, meaning that the total collective effective dose equivalent is certainly lower than 100 manSv per year.

In Denmark and Sweden the gypsum has been released to the sea for many years. As an example, the calculated equilibrium inventory of ^{210}Po in Danish waters is about 1.8×10^{9} Bq from the Superfos Company in Denmark. The "background inventory" in Danish waters is about 1.5×10^{12} Bq. Due to a high level of enrichment in fish meat from sea waters, this nuclide is the most significant of the releases nuclides. The corresponding collective effective dose equivalent is estimated to be 0.14 $manSv\ a^{-1}$ (6).

In Table 3.7 the collective effective dose equivalents for different parts of the fertilizer industry are summarized.

Table 3.7 *Collective effective dose equivalent from fertilizers in the Nordic Countries. (6)*

Mode of exposure	Collective effective dose equivalent (manSv a^{-1})
Use of fertilizer	
External	0.5
Internal	50
Release to sea	
Internal	0.1
By-product gypsum as building material	<100*

* In some cases the individual doses are significant (0.6 mSv a^{-1}).

3.4 Total non-nuclear industrial modifications of the natural radiation doses.

In Table 3.8 collective effective dose equivalents for the different industrial modifications of the natural radiation environment in the Nordic Countries are summarized.

Table 3.8 *Industrial modifications of the natural radiation doses.*

Industry	Collective effective dose equivalent in the Nordic Countries (manSv a^{-1})
Non-nuclear energy	
Peat and coal	80
Hydro-electric	3
Mining	37
Fertilizer	
Use	50
Release to sea	0.1
By-product gypsum	<100

4. Doses from nuclear power

4.1 Normal operation

The emission from nuclear power plants in Finland is estimated to give 0.1-0.2 manSv a^{-1} in Finland in 1986 (12). This indicates that the total collective effective dose equivalent from nuclear power plant emissions in the Nordic Countries is at least lower than 1 manSv a^{-1}. The collective effective dose equivalent from emissions is thus about 100 times higher for coal and peat than for nuclear power (chapter 3).

The doses to workers will dominate the collective effective dose equivalent from nuclear power, but even taking this into consideration, the total collective dose equivalent from nuclear power plants in the Nordic Countries is only a few tens of manSv a^{-1}, which for instance is comparable to the collective effective dose equivalent for mining (4).

4.2 The Chernobyl accident

After the Chernobyl accident large resources were used in the Nordic Countries to measure the consequences of the accident and to apply counter measures. Compared to the radiation doses from natural radiation the consequences of the Chernobyl accident were however small. This is illustrated in Fig 4.1.

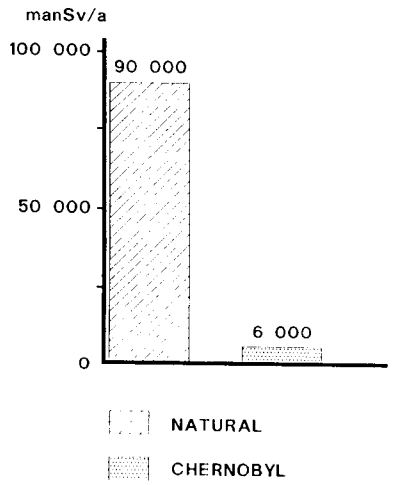


Fig 4.1. Collective effective dose equivalent from the Chernobyl accident in the Nordic Countries the first year after the accident compared to natural radiation. (Estimates from data given by the radiation protection authorities.)

In the figure we used values estimated for Chernobyl fallout by the radiation protection authorities in the Nordic Countries. The following values for the individual doses in the first year after the accident were used:

Norway, Sweden and Finland: 0.3 mSv a^{-1};
Denmark and Iceland: 0.1 mSv a^{-1}.

It is clearly seen from the figure that the Chernobyl accident only caused a small change in the total radiation doses to the Nordic population.

5. Radionuclides in the biosphere from a final disposal of spent nuclear fuel.

In this chapter, results of calculations of the concentration of radionuclides in the biosphere caused by a leakage from a final disposal of spent nuclear fuel in the Nordic Countries are presented. Details of the model and results are given in sub-report 8 (7).

It is planned that spent nuclear fuel will be disposed of at a depth of 500 m in a dense bedrock. The repository is made up of parallel tunnels separated by 25 m. The fuel elements are encapsuled in copper and stored in holes drilled in the tunnel floor. The gaps between the copper-enclosure and the rock are filled with bentonite clay. The safety is based on the fact that the radioactive material is surrounded by several barriers. This is illustrated in Fig 5.1.

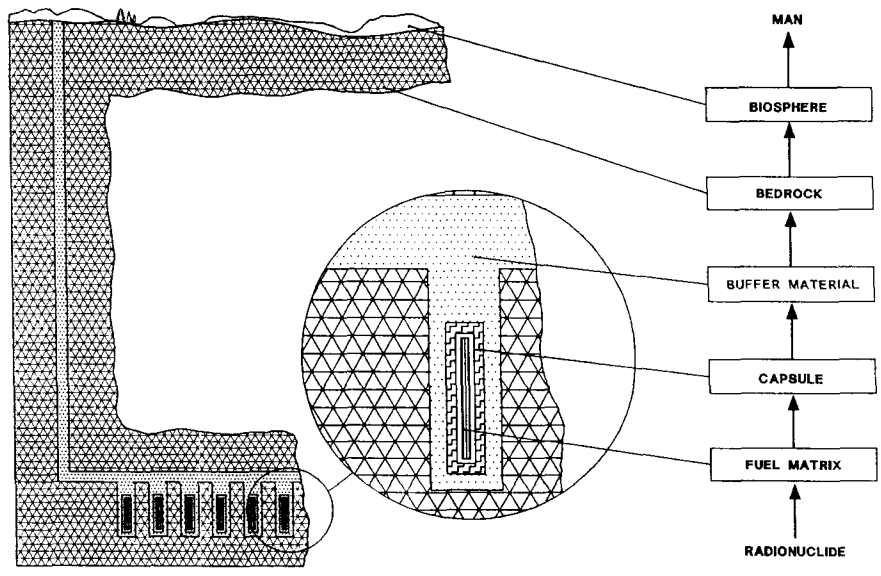


Fig 5.1. Principle for final disposal of spent nuclear fuel elements.

In a safety analysis of the final disposal system, the probabilities of different possible events in each barrier are systematically investigated. Then the consequences of the different cases are evaluated by mathematical models.

Several scenarios are analyzed. In the **basic case** it is assumed that the Cu-capsules lose their tightness after several thousands of years whereafter dissolution of the fuel in the ground water starts. Finally, the fuel material is transported over a long time period via cracks in the bedrock to the biosphere. The parameters used in the **basic case** are chosen in such a way that the release to the biosphere with a large probability is larger in the calculations than in a real case.

The analysis has also been performed in **special cases** and in **extreme cases.** Special cases are defined as cases where the properties of one of the barriers are weakened much more than expected, and extreme cases are when the properties and functions of several barriers are weakened or destroyed at the same time.

Results of radionuclide concentrations are presented for the basic case and for the special case where the ground water in the bedrock is assumed to be oxidizing. This special case provokes the largest releases in the analysis, except for the extreme case where a bedrock displacement is assumed to pass right through the disposal site. The probability of such an event is so low that it is not dealt with in this report

A compartment model is used for the biosphere. This is illustrated in figure 5.2.

The compartment model is based on a system of first degree linear differential equations with time dependent transfer coefficients together with well defined physical volumes. The recipients studied are a well and a small lake. Nuclides are spread from the lake to the Baltic Sea. The radionuclides may also be transferred to agricultural soil by watering or be deposited on bottom sediments.

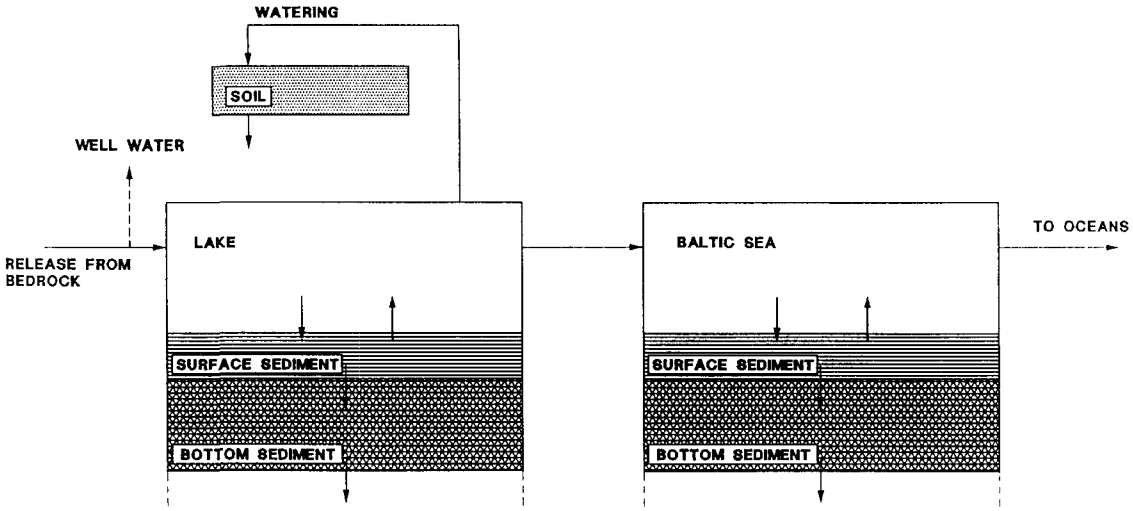


Fig. 5.2. Principle for the compartment model.

The safety analyses are made for the case of a disposal of 1270 tons of fuel (Finland). The results are also scaled up to the 6000 tons of fuel that are expected for Sweden.

The main results of the analysis can be summarized as follows:

- Both in the basic and the special case, the leakage will take place after a very long time span (1 million years or more).

- In the basic case, ^{226}Ra is the critical nuclide, while ^{237}Np is most important in the special case. In the special case , ^{233}U, ^{234}U, ^{236}U and ^{238}U also play an important role in addition to ^{226}Ra and ^{237}Np.

In the figure 5.3, examples of the time development of the radioactivity concentrations in the different environments is given.

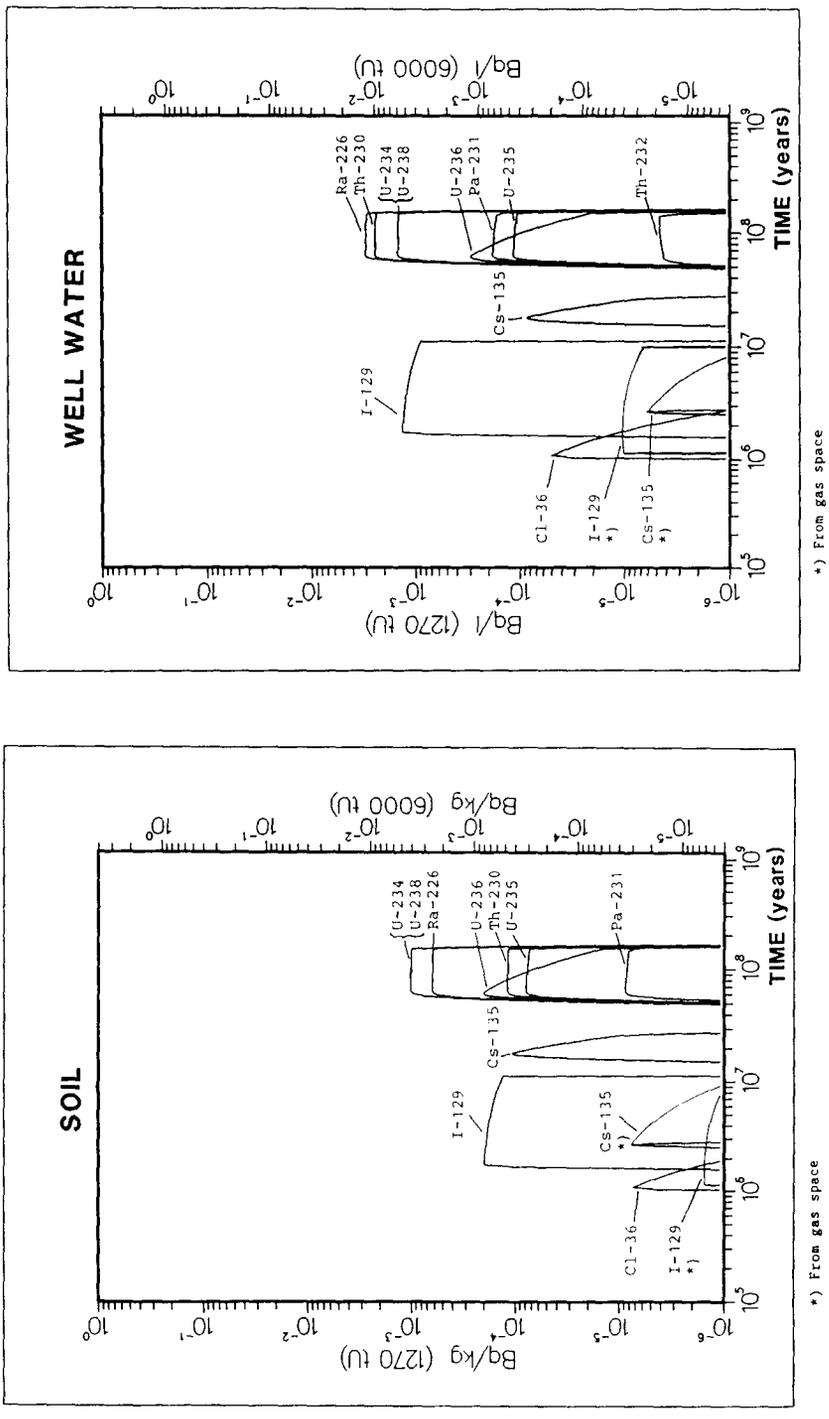


Fig 5.3 Calculated activity concentrations in soil and well water.

6. Chemical pollutants

Different industrial processes, lead to the release of various chemical, non-radioactive substances. Some of these substances may have adverse effects on health and environment, in the same way as radioactive substances.

Todays knowledge is insufficient to make a complete risk evaluation of non-radioactive substances in our environment. We have choosen to study the potential chemical risk from burning fossil fuels. This potential risk is then compared to the radiological risk of a final disposal of spent nuclear fuel.

The risk of energy production is in the broadest sense related to many aspects, e.g. production of the fuel, transport, occupational risks etc. In addition to radiation, carcinogenic substances and other directly related health risks from the wastes, there may be other negative effects on the environment. These may at a later stage influence man's health. Energy production can also cause effects of a socioeconomic, cultural or political nature. **In this report we have, however, limited the discussion to the direct health risk associated with the the waste substances.**

In the sub-reports 6 and 7 (8,9), the chemical pollutants are discussed in detail. The aim of this sub-project can be summarized as follows:

1 To outline a method that can be used to compare potential risks of activities leading to chemical and/or radiological pollutants.

2 To apply the method on the waste from traditional power plants and from nuclear power plants.

3 To discuss some possibilities and limitations of the method.

So called risk potentials are defined as the volume of water or air into which a given type of waste substance must be diluted in order to reach the accepted concentration limit in water or air for that particular substance. Mathematically this may be expressed as:

Risk potential = (Total amount of waste)/(concentration limit).

A significant difficulty of this definition is that concentration limits may be defined in different ways for different subtances, both regarding the accepted level of risk and the kind of effects that are taken into account. It is very important to be aware of this when such comparisons are performed.

From the risk potential, it is possible to define an index called the "person equivalent index". This quantity is obtained by dividing the risk potential of a given substance by the mean individual consumption of air or water (depending on the transport route of the

pollutant that is beeing considered). Mathematically, the person equivalent index is given as:

Person equivalent index = (Risk potential)/(Annual consumption).

The person equivalent index expresses the maximum number of people that could be exposed to the annual limit of intake. By using the person equivalent index, the potential risk may be compared regardless of whether the waste potentially leaks out to air or water.

In the figures 6.1 and 6.2 partial and total person equivalent indexes per produced energy unit are presented for different fuels. These figures clearly show that coal burning is associated with the highest potential chemical risk compared to the other types of power plants. In chapter 7 in the present report we will discuss the chemical pollutants from fossile fuels together with the radiological risk of used nuclear fuel.

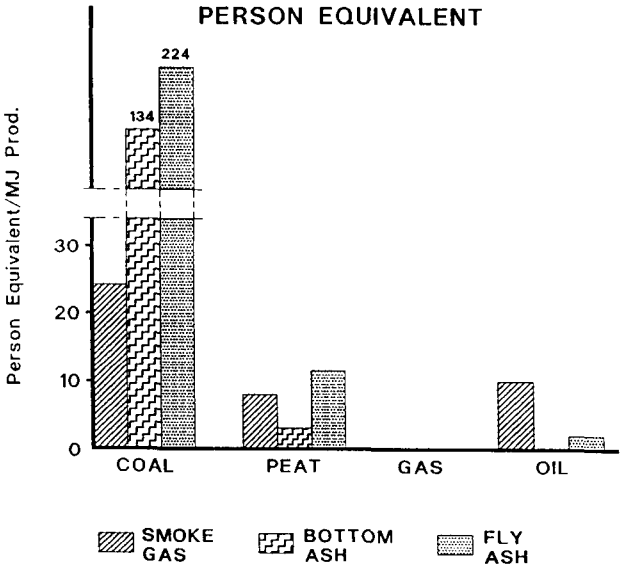


Fig 6.1. Person equivalent index per produced energy unit for smoke gas, bottom ash and fly ash from coal-, peat-, gas-, and oil-fired power plants.

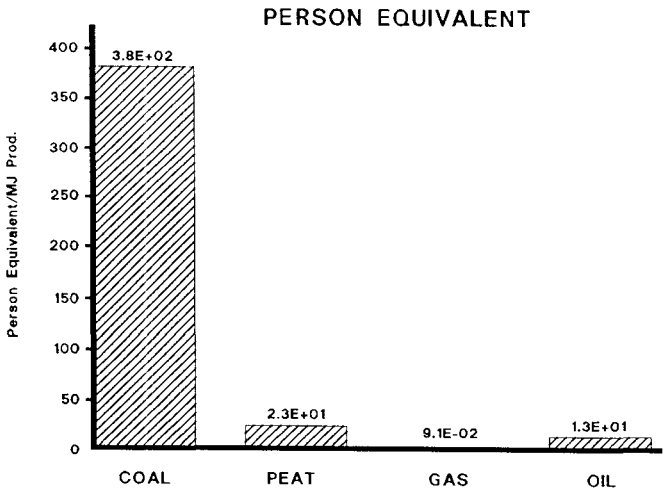


Fig 6.2. Total person equivalent index per produced energy unit for coal-, peat-, gas- and oil-fired power plants.

7. Discussion

7.1 The consequences of a final disposal of nuclear fuel compared to the consequences of natural radiation.

To be able to compare the radiological consequences of a final disposal of nuclear fuel to the consequences of natural radiation without estimating possible future doses, it is necessary to define a common index of risk for all the different nuclides involved. In our discussion we have chosen to divide the concentration of a nuclide by the ALI (Annual Limit of Intake) value for that nuclide. The sum of these values, for natural and nuclear fuel nuclides respectively, then represents an index for the "radiological danger" of the two groups of radionuclides.

Danger index = (activity concentration)/ALI

Before such comparisons are performed, it is necessary to point out the limitations and pitfalls of such a comparison:

-This type of comparison does not take into account whether the radionuclides are taken up in the food chain or not.
-The time scale of the model calculations is very long, and there is no way to predict what the world looks like in millions of years.

We have therefore chosen to use the **maximum values** in our evaluations, even though the model calculations show that this will occur after such a long time that it is impossible to tell how life on Earth may be then.

In table 7.1. the "danger index" is given for the maximum calculated activities in the environment discussed in chapter 5. The indexes for the natural nuclides are calculated by assuming radioactive equilibrium in the radioactive series. This is not necessarily the case, especially not in the aquatic environments, but such a calculation will at least give a fairly good idea of the magnitude of the index.

In the calculations, based on the present nuclear program, it is assumed that 6000 tons of uranium fuel will be disposed of in Sweden.The consequences for 1270 tons (Finland) may be calculated by scaling the results down. A leakage in one country would not effect the environment in the other country. If, however, a leakage should take place in Finland and Sweden at the same time, the activity released to the Baltic Sea would be the sum of the contribution from each country.

In table 7.1. only the mean values for the natural nuclides have been used. The activity concentrations vary within a wide range, and values 10 times higher than those listed in the table could be found in several cases. Furthermore, the largest doses from natural radiation are caused by radon in dwellings (chapter 2). This is not taken into consideration in the indexes. In the natural danger index for well water, radon is not included, since radon in water only represents a problem as a source to radon in indoor air.

We of course do not know anything about how life on earth will be millions of years from now. If we still, as a hypothetical case, assume that there are people on earth and that they consume 1 l of well water per day, they will receive an annual effective dose equivalent of about 0.5 mSv per year from the nuclear waste nuclides in well water in the special case. This is an increase by 10-15% of the mean total effective dose equivalent from natural radiation to the Nordic population today. In the basic case, the extra dose would be a factor of 1000 lower than this.

Table 7.1 *"Danger index" for the maximum values of release from nuclear fuel disposal and for the natural radiological environments.*

Environment	Danger index (kg^{-1})		
	Natural	Nuclear waste	
		Basic case	Special case*
Agricultural soil	$2x10^{-2}$	$7x10^{-8}$	$5x10^{-6}$
Well water	$2x10^{-5}$	$3x10^{-8}$	$3x10^{-5}$
Lake water	$4x10^{-8}$	$1x10^{-9}$	$4x10^{-8}$
Lake sediment	$2x10^{-2}$	$9x10^{-6}$	$1x10^{-3}$
Baltic Sea water	$4x10^{-8}$	$2x10^{-14}$	$3x10^{-11}$
Baltic Sea sediment	$2x10^{-2}$	$2x10^{-10}$	$3x10^{-8}$

* Oxidizing ground water.

7.2 Radiological waste from coal and peat powered plants and from fertilizers compared to the final disposal of nuclear fuel.

In chapter 3 the radiological consequences of different types of electric power production and of the production and use of fertilizers were discussed. It is very difficult to compare these consequences to the consequences of a final disposal of nuclear waste. Among other things, the time scale must be considered. The model calculations for nuclear waste have been projected a very long time into the future, while we receive the doses from both use, production and disposal of coal, peat and fertilizers here and now. Such evaluations will therefore be very qualitative.

7.2.1 Peat and coal powered plants

The doses from peat and coal powered plants are dominated by emission during production rather than the waste disposal. From table 3.3. (chapter 3) we can see that the mean activity concentrations in ash are somewhat higher than what we find in soil, but levels are within the range that is usually found in natural geological materials. This suggests that spreading the ash in agricultural soil or other environments will result only in a minor change in the activity concentration of the soil. If however, houses are built directly on ash piles the indoor radon concentrations may be enhanced. Use of fly ash in building materials could in some cases cause increased indoor radiation levels.

The radiological risk potentials of coal and peat powered plants are low compared to the chemical risk potentials of the waste.

The concentrations in agricultural soil of radionuclides caused by waste from coal and peat as it is handled today are very low compared to the natural radio nuclide concentration. They are, however, in the same order of magnitude as the calculated values for nuclear waste. The calculated concequences to ground water are larger for nuclear waste than for waste from coal and peat.

The emission of radionuclides from coal and peat powered plants is estimated to give an annual collective effective dose equivalent of 70-80 manSv in the Nordic Countries as a whole.

As seen from chapter 4, the radioactive emissions from Nordic nuclear power plants cause collective effective dose equivalent of less than 1 manSv a^{-1} for the Nordic Countries. The collective effective dose equivalent for emissions is thus about 100 times higher for coal and peat than for nuclear power. The doses to workers dominate the collective dose equivalent from nuclear power, and the total collective effective dose equivalent from nuclear power in the Nordic Countries is estimated to a few tens of mansievert per year when occupational doses are included.

7.2.2 Fertilizers

The largest radiological impact of fertilizers is the use on agricultural soil, and 80 years of use will result in an increase of 1 Bq kg^{-1} of radium in the soil. This is about 4% of the mean radium concentration. Thus the use of fertilizer represents a higher "danger index" for agricultural soil than that given by calculations for a final disposal of nuclear waste. The contribution by fertilizers to the activity in ground water and lake water will however probably be very small.

The most important by-product of the fertilizer industry is the by-product gypsum. In sub-report 5 (6) it is estimeted that the equilibrium inventory of ^{210}Po in Danish waters, due to release from fertilizer production in Denmark, will be about 1% of the natural ^{210}Po inventory. This will cause very low doses to the population. Use of the by-product gypsum as building materials will both cause increased gamma radiation and radon concentration indoors. The collective doses are of the same order of magnitude as the doses from the use of fertilizer on soil.

The radiological consequences of the everyday use of fertilizers and wastes from their production are today much larger than the consequences of any corresponding industrial effluents released to the environment. The contribution of fertilizes to the radioactivity in soil is significantly larger than the calculated maximum levels from a release of activity after a nuclear waste disposal site.

7.3 Radiological consequences of energy conservation

A significant fraction of the energy we produce is used for heating our dwellings. In Finland about 17% of the energy was used for heating dwellings in 1984 (industrial an office buildings not included). Due to the so-called "energy crises", efforts have been taken to conserve energy by reducing the cost of heating. This may have negative radiological consequences if the energy conservation is obtained by reducing the air exchange rates in the dwellings. If the air exchange rates are reduced without any other measures, indoor radon concentrations will increase. This will cause increased radiation doses to the inhabitants. It should, however, be pointed out that energy conservation in itself does not necessarily result in increased radiation doses; this is only the case when air exchange rates are reduced.

In the discussion, the following assumptions have been made:

1 The saved heating costs are proportional to the reduction in air exchange rates.

2 The indoor radon concentration is inversely proportional to the air exchange rate.

The trends in energy consumption for heating may be found from available statistics. The development in energy consuption for heating in Finland between 1970 and 1985 is shown in Fig 7.1, and the increase in total number of dwellings in the same period is shown in Fig. 7.2. By using Fig 7.1. and Fig 7.2., it is possible to present the development in energy consumption for heating per dwelling. This is shown in Fig 7.3.

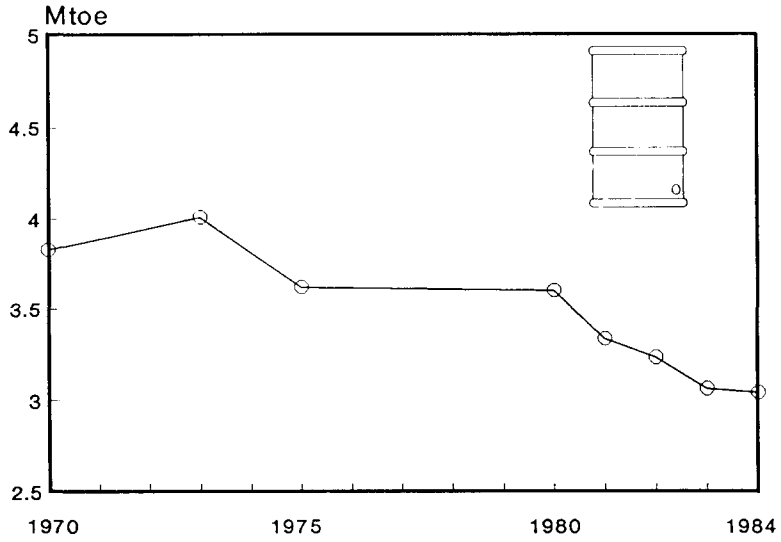


Fig 7.1. The energy used for heating of Finnish dwellings in the period 1970-1984. (10). (M_{toe} = millions of tons oil equivalent)

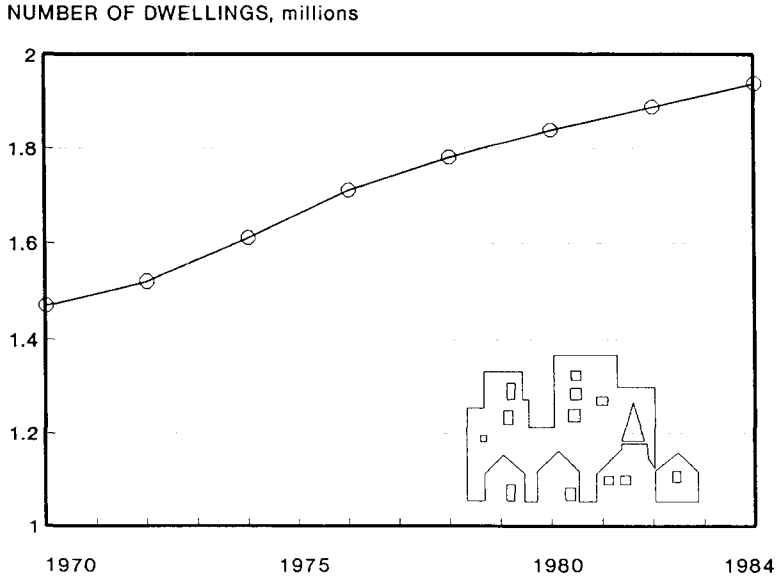


Fig 7.2. Number of Finnish dwellings (11).

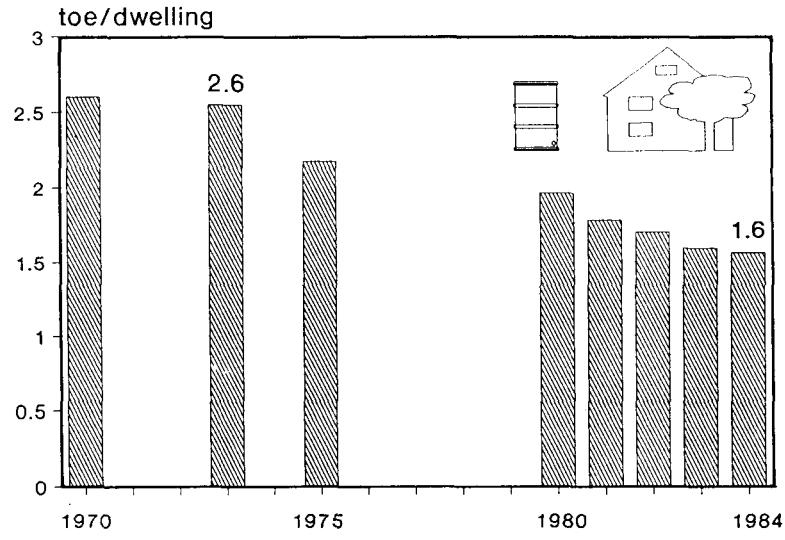


Fig 7.3. The development of energy consumption for heating ***per dwelling*** *in Finland.*

From Fig 7.3. it is seen that the mean heat consumption for a Finnish dwelling is reduced from about 2.6 toe (29 MWh) in 1973 to 1.6 toe (18 MWh) in 1984. This corresponds to a reduction of about 39%. By using the assumptions mentioned earlier, it can be estimated that the mean air exchange rate in Finnish dwellings has decreased by about 39% in the same period. From the assumption that the radon concentration is inversely proportional to the airexchange rate, we may derive that the radon concentration in Finland was 39% lower in 1973 than in 1984. The data on radon concentrations in dwellings in the early seventies are very limited, but some Swedish results suggest that there has been a significant increase in the last 10-15 years. The Swedish data indicate that the increase might be even more dramatic than what we find from our assumptions. Use of data on heating costs seems therefore to be a feasible way to estimate the radiological consequences caused by the energy conservation after the energy crises.

From the data on radon in Nordic dwellings in the eighties, the increase in collective doses since 1973 may be estimated from the following assumptions:

1 The mean indoor radon concentration in 1973 was 61% of the 1984-value.

2 The annual heating energy per dwelling was 11 MWh lower in 1984 than in 1973.

3 There are 2.7 inhabitants per dwelling.

These are of course rather crude estimates, but considering the above mentioned Swedish radon data, they should at least give a qualitative picture of the radiological costs of energy conservation. The results of the estimates are given in Table 7.2.

From the table we find that the annual collective effective dose equivalent in the Nordic Countries has increased by 23 000 manSv a^{-1} due to energy conservation between 1973 and 1984. This corresponds to an increase of about 2.3 manSv per MWa. For the individual country, the conditions may be different, because the increase in dose is largest where the absolute dose is largest. The heating costs are probably also higher in these countries. The results in Table 7.2 must therefore be interpreted with care. It should however be noted that the increase in collective doses is apparently more than 200 times higher than the total collective doses in the Nordic countries from coal and peat power stations.(5)

Table 7.2 *Estimated radiological consequences of energy conservation in the Nordic Countries in the period 1973-1984.*

Country	Difference in heating energy equivalent E(1973)-E(1984)	Difference in collective dose S(1984)-S(1973)
Denmark	2300	3 000
Finland	2200	5 700
Iceland	120	27
Norway	1900	4 800
Sweden	3800	9 800
TOTAL	10 320	23 000

E = Annual heating energy MWa

S = Annual collective effective dose equivalent from radon daughters manSv

Let us now assume that we for radiation protection purposes again increase the air exchange rates (and the energy consumption) in Nordic dwellings. In this way it is possible to estimate the costs in increased energy per manSv saved. The cost per produced energy varies between the countries, but for the sake of the example we have chosen a value of NOK 0.20 per kWh. This corresponds to NOK 1.7×10^6 per MWa. If the Nordic Countries increase their annual heating energy by 10 000 MWa, the annual collective effective dose equivalent will decrease by about 20 000 manSv. This corresponds to a cost in the order of 1 million NOK per man sievert.

A cost of about NOK 150 000 per manSv is often used in optimization of radiation protection. Compared to this, dose reduction by increasing the natural ventilation, will be very expensive in heating costs. There are, however, today methods for radon reduction that are very effective, and for purely radiation protection, such methods should be used rather than a general increase in ventilation.There are however other negative effects of low air exchange rates, and if such effects also are taken into consideration, increased general ventilation is probably recommendable, especially if heat exchangers are used.

It should be pointed out that the monetary value per manSv is to be used when the costs of a **dose reduction** are evaluated. A high monetary value per manSv should not be used as an argument for saving money by increasing the doses. The high value per manSv calculated above, should thus in no way be used to legitimize the dose increase caused by energy economisation during the last decades. This is especially true when also other negative health effects from a poor indoor climate are taken into account.

In this chapter estimates of the radiological consequences of energy conservation have been given. In our example, we have only taken into account the radiological effects, but there are also other factors that should be taken into consideration. Such factors can be economic, social and political as well as factors causing negative health effects. On one hand, a reduction in air exchange may cause increased "well being" and health if the dwelling originally is very drafty. On the other hand, if the dwelling is well insulated and has a low air exchange, a further decrease of ventilation may cause increases in allergenes etc. There are also cases where very low air exchange rates cause damage to the dwelling itself. The energy consumption has also a global political side that can be difficult to assess in money or health.

Many effective radon counter-measures have been developed during the last years. Many of these measures are energy efficient and have relatively low installation costs. From a radiation protection point of view, it is cost effective to perform such countermeasures in Nordic dwellings even when the energy costs are taken into consideration.

7.4 Chemical pollution from fossil fuel energy production.

In chapter 6, methods for calculation of risk potentials and person equivalents for different types of pollution are discussed.

In order to be able to perform relevant model calculations, sufficient data must be available. In our model calculations, the data is relatively incomplete, and the results must therefore be regarded as illustrating examples rather than a real description of the true situation. It is also important to note that the calculated person equivalents are purely theoretical quantities giving the number of persons exposed to the limit of intake if the emission is evenly distributed among these persons. The person equivalents do not imply that such distribution in fact will or could occur.

The calculations are conservative, because we have calculated the total relative risk potential for each power plant as the sum of the risk potentials for the different individual pollutants. Any individual additional substance will thus represent an increase in the total risk potential. If significant substances have been left out, large errors may occur. Furthermore, one cannot exclude the possibility that the sum of a number of individual substances does not make up the total, in fact they may cause much more serious effects when acting together. Such errors will always represent an underestimate of the total risk potential. It is also possible that some unidentified detrimental substances may have been left out in the analysis. This is especially valid for organic micro pollutants where only a few out of about hundred are known. Benzo-A-pyren (BaP) represents these substances in the present analysis because it is carcinogenic and also the most investigated polycyclic aromatic compound (PAH) and thus the substance for which most data is available. The PAH-substances are considered to be among the most detrimental substances in fuel burning processes.

Metals dominate among the inactive pollutants in the effluents and it is characteristic that in some cases some of these give the largest contribution to the risk potential. Chromium contributes for instance with 76% of the total risk potential in bottom ash from coal fired power plants, while nickel plays a similar role in both the smoke gas and fly ash from oil fired power plants. In such cases the neglection of some minor components result in relatively small errors, even if the components have high toxicity.

In chapter 6, a model for the assessment of potential health risks for different types of energy production is presented. The model can be a useful tool for such evaluations, but all the limitations of the model must be stressed. If the correlation between the pollution at the site of emission and the corresponding health risk were known, sufficient comparisons between risk potentials from different types of energy production, and thereby a ranking of the different energy forms based on potential risk, could be performed. As the situation is, we can only evaluate a limited part of the total complex. An important limiting factor is for instance that the different pollutants have different effects on man. The risk potential only gives risk relative to accepted limits. These limits have been based on different types of evaluations. Where a low value is readily reached there is a tendency to set a low limit, even if actual evaluation of the health risk would have implied a higher limit. On the other hand, a high limit may be set because it is difficult to reach low values. The detriments of different types of health risk such as cancer and genetic effects are not always easily compared. The different types of power production may cause different effects on nature, which might in later stages cause health effects on man. In our evaluations only **direct health effects** have been considered.

If "generally accepted" limits are used to rank the different types of energy production, the results in Table 7.3 are obtained.

Table 7.3 *Calculated person equivalents per MJ produced energy for different fuels. This index gives the number of persons, that theoretically might be exposed to a concentration equal to the appropriate limit if the total amount of waste is spread evenly among them.*

Type of energy	Main type of risk	Person equivalents per MJ produced
Coal	Chemical	380
Nuclear	Radiological	80
Peat	Chemical	23
Oil	Chemical	13
Gas	Chemical	0.1

This information could in theory be used in order to define the required safety measures for differnt types of waste.

The chosen limit for radiation induced cancer (1 mSv a^{-1}) corresponds to a risk of 10^{-5} - 10^{-4} per personyear. For the most dominating chemical carcinogens in this study, Cr and Ni, limits based on a risk 100 to 1000 times lower than this are used. If we use the same level of accepted risk for radiological and non-radiological effects, the number of person equivalents for nuclear power must be multiplied by about 500. The person equivalent index for nuclear fuel must be increased compared to the index for fossil fuels if the same level of risk should be applied for chemical and radiological pollutants.

These calculations clearly show that the models are very sensitive to how the limits are defined. If we use the presently accepted limits, nuclear power is ranked after the chemical risk of coal. If we, however, use limits accepting the same level of risk, nuclear power will represent the highest risk potential.

The discussions regarding risk potentials and person equivalents may be summarized by:

- The models are useful tools for comparing the risk potentials of different types of energy production, as long as comparable effects are evaluated.

- The results are very dependent on the choice of limits.

- The models deals only with direct health effects, and do not consider potential environmental effects that in later stages might cause serious detriments to mankind.

- The results should only be used in qualitative rankings of the potential risks of different energy forms.

- Coal clearly represents the highest potential risk among the traditional energy forms.

- Nuclear power has, based on these models, a potential risk that might be higher or lower than the potential risk of coal, depending on what set of limits are used.

- The results, given as risk potential or person equivalents, do not take into consideration whether the waste is safely disposed of or released to the environment. They are only indexes for the potential risk of the waste. If, through our method, a higher potential risk is found, it is an argument for a safe disposal of the waste. It is important to stress that the model does not take into consideration the probability for a release of the waste to the environment, but only evaluates the consequences of a total release to the human environment.

8. CONCLUSIONS

From the available data and models used, the conclusions of the report may be summarized as follows:

1 The radiation doses from natural radiation and the concentrations of natural radionuclides in different environments show large variations. The highest doses are received from inhaled radon daughters in dwellings. Possible releases from a final nuclear waste disposal site will first occur after millions of years, and it is meaningless to asses actual doses to people in such a time perspective. Calculations show, however, that the radiological concequences of such a release to the environment will at any rate be much smaller than the present variations in natural radiation caused by local geological differences.

2 Phosphate fertilizers cause increased radiation doses from natural radionuclides both during production and use. The use of such fertilizers increases the radioactivity in agricultural soil more than would be caused by a release from a final repository of spent nuclear fuel. The use of by-product gypsum as building material is also a route of exposure to enhanced natural radiation.

3 Coal and peat powered plants cause increased radiation doses both through emission and waste disposal. Thus nuclear power is in no way radiologically unique, compared to other types of energy production, neither regarding emission of radioactivity to the environment nor regarding occupational radiation doses during normal operation. An interesting fact is that workers in underground hydro-electric power stations receive large doses from the inhalation of radon daughters. The occupational exposures there are generally higher than doses received by workers at nuclear power stations.

4 The use of risk potentials and person equivalents is a useful concept to compare *potential* risks from different types of waste. **This procedure is limited by the fact that the risk potentials do not take into account the probability of release of waste into the environment, but only considers the potential consequences of a total release.** We have only considered possible *health effects on man* and not dealt with negative effects on nature caused by chemical pollutants. **The use of our model shows that the waste disposal is a problem not only related to nuclear power.**The chemical risk potential of coal power is the highest among the fossil fuels and may be in the same order of magnitude as the radiological risk potential of nuclear power.

5 One way of energy conservation is to reduce the air exchange rates in dwellings. This has caused increased indoor radon concentrations with resulting increases in collective doses, especially following the energy crises in the early seventies. The annual collective dose equivalent caused by energy conservation has been estimated to be more than two orders of magnitude higher than the total doses from the annual energy production in the Nordic Countries. Energy conservation by reducing air exchange rates in dwellings should therefore be avoided, and measures should be taken to reduce radon concentrations in the future. This could be achieved by relatively simple, inexpensive and energy effective methods.

9. References

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Appendix

Some definitions

A.1. Radioactivity

Some atoms are unstable, they undergo spontaneous transformation into more stable product atoms. Substances consisting of such unstable atoms are said to be **radioactive** and the transformation process is known as **radioactive decay.** Some radioactive substances are naturally occurring while others are man-made. The **half-life** of a radioactive species is the time required for one half of the nuclei in a sample to decay.

In a radioactive species the number of nuclear transformations per unit time is called the **activity.** The activity is proportional to the number of unstable atoms, and is thus a measure for the amount of radioactive substance. The unit for activity is **becquerel (Bq).** In one Bq there is one disintegration per sec. It is often practical to measure activity per unit of mass (Bq kg^{-1}) or per unit volume (Bq m^{-3}). This is called **activity concentration.**

A.2. Ionizing Radiation

During radioactive decay, radiation is emitted from the atomic nuclei. There are three types of radiation: **alpha** (α), **beta** (β), and **gamma** (γ). **Alpha radiation** are particles consisting of two protons and two neutrons. The alpha particles are absorbed very easily (a thin sheet of paper will stop alpha particles completely). **Beta radiation** consists of high speed electrons which originate in the nucleus. Beta radiation is not as easily stopped as alpha particles, but about 1 cm of water will stop the particles completely. **Gamma radiation** belongs to a class known as electromagnetic radiation in the same way for instance as visible light. The energy of gamma radiation is however very high (short wave length) compared to the energy of light. It is very difficult to "stop" gamma radiation, but the intensity is gradually reduced by increasing thickness of the shielding material. When high-activity gamma sources are used in industry or medicine, thick shielding walls of lead or concrete are applied to get safe working conditions.

Alpha, beta and gamma radiation belong to the group of ionizing radiation. Other types of ionizing radiation are X-rays, neutron radiation and cosmic particle radiation.

Natural radiation is radiation caused by natural sources. The most important types of natural radiation from a radiation protection point of view are **external gamma radiation** from natural radioactive substances in bedrock, soil and building materials, **cosmic radiation** consisting of particles from outer space, **internal radiation** from natural radioactive material accumulated in the body and finally, radiation from inhaled daughter products of the radon gas isotopes ^{222}Rn (radon) and ^{220}Rn (thoron).

A.3. Radiation dose

Some radiation energy will be absorbed by irradiated tissues of the body. The amount of energy absorbed per unit mass is called **absorbed dose** or simply **dose.** The unit for dose is gray (Gy). A dose of 1 Gy is equivalent to an absorbed energy of 1 J/kg.

Different types of radiation do not necessarily cause the same biological effect per unit dose. To be able to compare biological effects from different radiation doses, the quantity **dose equivalent** has been introduced. The dose equivalent is obtained by multiplying the dose by a "quality factor" reflecting the biological efficiency of the radiation type. The quality factor is unity for gamma and beta radiation while it is 20 for alpha radiation. Dose equivalent is measured in the unit **sievert (Sv).**

The radiation sensitivity is different for the different organs of the body. The quantity **effective dose equivalent** is introduced in order to compare effects of different modes of exposure. This quantity is obtained by multiplying the dose equivalent to the individual organs by "weighting factors" reflecting the radiation sensitivity of the individual organs. The sum of these products is the effective dose equivalent given in Sv. This quantity may be regarded as a kind of "weighted whole-body dose equivalent", and by using this concept, it is possible to compare the risks of different modes of exposure.

In radiation protection the risk of cancer and genetic effects is considered to increase lineary with increasing effective dose equivalent. The total number of adverse effects in a population is thus proportional to the product of mean effective dose equivalent and population size. This product is called the **collective effective dose equivalent.** The unit is **mansievert (manSv).**

A.4. Potential risk

In activities involving production of pollutants it is important to have an index to measure the risk of these pollutants if they are totally released to the environment. The **risk potential** is found by calculation of how large volumes of water or air the amount of pollution must be diluted in to reach a concentration limit for air or water. Mathematically this may be expressed as:

Risk potential = (Total amount)/(concentration limit)

Thus the risk potential do **not** take into consideration how much pollution that **is** or even **could** be released.

By dividing the risk potential by annual individual consumption of air or water, the **person equivalent index** is obtained. This index gives the number of persons, that theoretically might be exposed to a concentration equal to the appropriate limit if the total amount of waste is spread evenly among them.

We have also in this report introduced a quantity obtained by dividing the concentration of a given radionuclide by the **annual limit of intake (ALI)** for the nuclide. In this way, the **potential danger** of different concentrations of different nuclides in the biosphere can be compared. We have chosen to call this quantity for the **danger index.** The **danger index does not** take into consideration the probability for the radionuclides to get into the food chain, but is only a measure of the number of ALI values per volume or mass unit.