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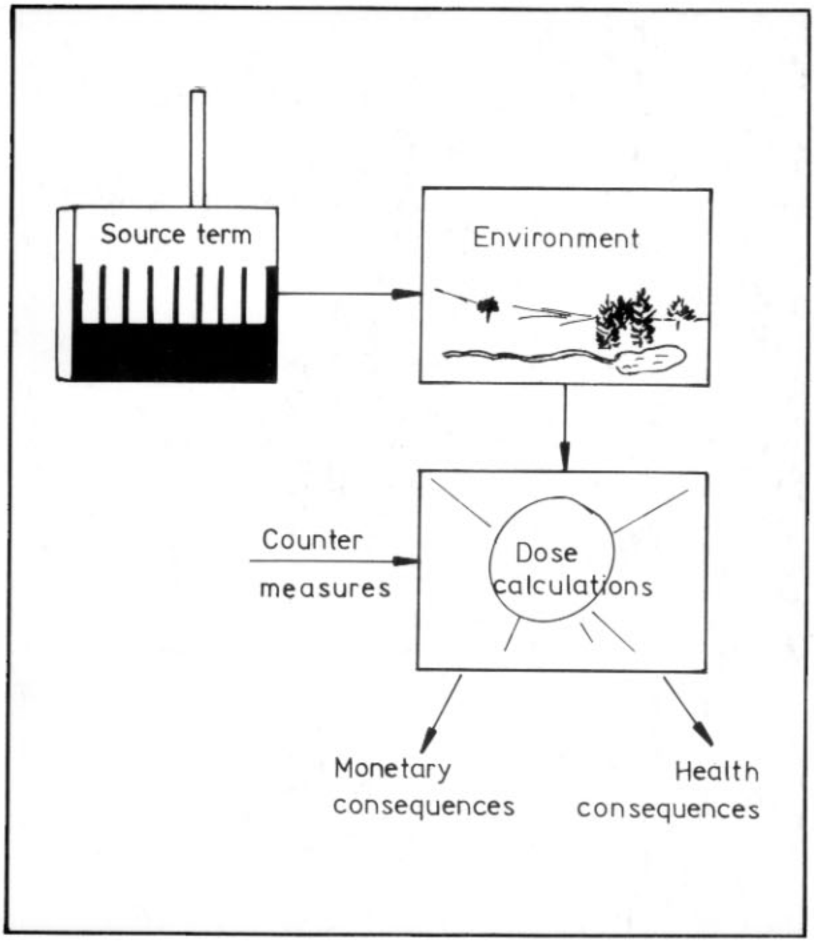
Towards More Realistic Assessment of Reactor Accident Consequences

A Nordic Project



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TOWARDS MORE REALISTIC ASSESSMENT OF REACTOR ACCIDENT CONSEQUENCES

A Nordic Project

Final Report of the NKA/REK-1 Project

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ABSTRACT

The project described in this report consists of 12 subprojects, all concerned with some aspect of accident consequence assessment. The purpose has been to improve the data base used in such assessments, and also to improve the assessment models. Both experimental and theoretical tasks have been carried out. The work has been partially financed by the Nordic Council of Ministers via the Nordic Liaison Committee for Atomic Energy.

KEY WORDS: Accidents; Agriculture; Computer codes; Decontamination; Deposition; Dose limits; Drinking water; Environmental exposure pathway; Fallout; Fish products; Irrigation; Lakes; Meat; Milk; Rivers; Runoff; Sediments; Shielding; Soil; Surfaces; Urban areas; Seasonal variations

Summary

A long chain of calculations has to be carried out, when the impact of a reactor accident shall be assessed. The last link in this chain is an analysis of the consequences of the release of radioactive materials on the environment. As input to this analysis is required the magnitude of the release that may result from certain accident sequences, as well as the probability that these sequences will occur. The results form part of the basis material upon which the safety aspects of a nuclear power plant are assessed.

A number of different phenomena are taken into consideration in such an analysis, like the chemical properties of the materials in the release, meteorological conditions, agrigultural practices in the area in question etc. The exact impacts are not always known, but in one way or another all these aspects must be taken into account in the analysis, and in this field it has become a tradition, whenever knowledge is incomplete, to treat the phenomenon in a "conservative" manner. This means that models and parameter values are chosen so that the resulting risk is intentionally overestimated.

With this overestimation at numerous points in an analysis, the end result (the calculated risk) may become significantly higher than what is realistic. The main purpose of the project described in the present report has been to bring more realism into such analyses, by increasing the information basis in some chosen subject areas, where local conditions are concerned, or where Nordic conditions are special (winter), and where there are special resources available in the Nordic countries for increasing the information base.

A release of radioactive materials may occur to the atomosphere or to water. The releases to the atmosphere are usually considered as being the most serious, where large reactor accidents are concerned. The rate at which the radioactive materials are diluted depends upon the meteorological conditions. Rapid dilution will reduce the consequences. The atmospheric dispersion models normally used in risk analyses are simplified (and cheap to run on the computer), and will sometimes no be sufficiently accurate. We have made a modification to an internationally well-known risk analysis program, so that it is now possible to calculate atmospheric dispersion under certain situations using more sophisticated methods.

When a plume containing radioactive materials passes, the population can be exposed directly by the radiation from the plume, and by inhalation of the radioactive materials. The exposure is reduced if a person stays in-

doors during plume passage. The building materials will reduce the direct radiation from the plume (shielding factor). The inhalation dose will also be reduced, as a portion of the radioactive materials entering the house does not reach the indoor air, but is caught in the walls (filtering factor). Both shielding factors and filtering factors for some typical Nordic houses have been determined.

As part of the project an investigation of dry deposition in urban areas has been carried out, as well as an investigation of run-off from roofs. It was found that dry deposition on surfaces typical of urban areas is much lower than what is now assumed in the analyses. It is also usual to assume that rain water does not remove previously deposited material. The investigation showed the run-off with rain was rather efficient: 40–75 %, depending upon the type of roof material. Accordingly the radiation dose in urban areas will be somewhat lower than estimated up till now.

One of the tasks in the project consisted of systematically collecting information on the freshwater exposure pathways in the Nordic countries. Much of this information was originally collected in connection with measurements on fall-out from atmospheric nuclear test explosions, and much of this information is relevant to large reactor accidents, because the dominating nuclides in the fall-out (Sr-90 and Cs-137) are also among the most important in connection with large reactor accidents.

Exposure via agricultural products can take place when the radioactive materials are deposited upon the plants, and brought with these to the consumer, or to domestic animals and subsequently to milk and meat, or via uptake of the radioactive materials to the plants from the soil. Such an uptake may take place in many growth seasons after the accident.

Our knowledge of deposition of radioactive materials on plants, and uptake from the soil, also comes from investigations of weapons fall-out. One part of the project consisted of collecting fall-out data from the Nordic countries in a common data bank, which can be directly utilized when it is desired to assess exposure via agricultural products.

The fall-out measurements have given the basis for the calculations, but much additional information is needed when resulting doses to a population shall be calculated. It is for instance necessary to know for each type of domestic animal how much of the intake is transferred to milk or meat. We have previously mentioned that we have incorporated a more advanced atmospheric dispersion model in a risk analysis program. In the same program we have also incorporated a more advanced nutrition exposure model, developed in Finland.

If an accident happens, there are several possibilities of reducing the health consequences, e.g. by evacuation or sheltering. Possible long-term actions are relocation from certain areas, decontamination, and restrictions on the use of agricultural products or drinking water from certain areas. The mitigating actions may be expensive, and guide-lines are needed to help make the best decisions. Such guidelines are often given in the form of so-called dose criteria. The project surveyed the manner in which dose criteria and related expressions are used in a number of countries today. This material is meant to help Nordic authorities in their continued work.

When a risk analysis shall be carried out, one is often forced to using parameter values "from literature". Often this means that some parameter values from the American Reactor Safety Study (WASH-1400) from 1975 are used. But these values are valid for American conditions, and can not always be transferred to Nordic conditions. Relevant Nordic values are being determined, however, and the present project has also contributed to this work. But there are also conditions that are specifically Nordic, winter conditions. Some studies of winter conditions have been carried out as part of the project. We have examined run-off of radioactive materials deposited upon snow, both for roofs and a field. And an experiment has been carried out, to examine decontamination efficiency, using ordinary snow removal equipment, when it is desired to remove materials deposited upon a snow surface. We found that run-off from the field was somewhat higher than under summer conditions, though not much. The retention on roofs was about half of that under summer conditions. The decontamination efficiency was unexpectedly high: more than 99% was removed. Accordingly the consequences of a radioactive contamination may be significantly less during winter conditions.

The project has contributed significantly in strengthening Nordic cooperation in the accident consequence analysis field. The results obtained have given a better base for performance of such analyses. The project has also been closely coordinated with the large present international efforts.

Sammendrag (Abstract in Norwegian)

Analyse av konsekvenser i forbindelse med større reaktoruhell bygger på et omfattende beregningsgrunnlag. Det siste ledd i slike beregninger er en analyse av konsekvensene for omgivelsene av et utslipp av radioaktive materialer. I denne analysen inngår dels beregnete utslipp som følge av visse ulykkesforløp, dels estimering av sannsynligheten knyttet til forskjellige ulykkesforløp. Resultatene vil være del av vurderingsgrunnlaget når det gjelder et kjernekraftanleggs sikkerhetsaspekter.

En slik analyse tar hensyn til en rekke forhold som bl. a. kjemiske egenskaper til stoffene i utslippet, meteorologiske forhold, jordbruksrutiner i angjeldende områder etc. De eksakte innvirkninger er ikke alltid kjent, men på en eller annen måte må imidlertid alle disse forhold tas hensyn til i analysen, og det har blitt en tradisjon innen fagfeltet at hvert enkelt trinn der kunnskapene er ufullstendige, blir behandlet "konservativt". Det vil si at modeller og tallverdier velges slik at den resulterende risiko bevisst overvurderes.

Med en slik overvurdering på mange trinn i analysen kan sluttresultatet (den beregnede risiko) bli vesentlig høyere enn det som er realistisk. *Hovedhensikten med det prosjektet som beskrives i denne rapporten er å bringe mer realisme inn i slike analyser ved å øke innsikten på noen valgte områder*, der lokale forhold spiller inn, eller der forholdene i Norden er *spesielle (om vinteren)*, eller der det i Norden finnes spesielle forutsetninger for å gjøre en viktig innsats.

Et utslipp av radioaktive stoffer kan skje til luft eller vann. De alvorligste utslippene i forbindelse med de store reaktorulykker regnes å skje til luft. Hvor fort fortynning skjer avhenger av de meteorologiske forhold. Rask fortynning tjener til å redusere konsekvensene. De modeller som vanligvis brukes i risikoanalyser til å beregne spredning, er enkle (og billige å kjøre på datamaskin) og vil noen ganger ikke gi tilstrekkelig nøyaktighet. Vi har gjort en utbygging av et kjent og internasjonalt hyppig brukt risikoanalyseprogram, slik at dette nå også kan beregne visse situasjoner ved hjelp av mer avanserte metoder for håndtering av meteorologisk spredning.

Når en sky med radioaktive stoffer passerer, kan befolkningen eksponeres ved direkte stråling fra skyen og ved at de radioaktive stoffene inhaleres. Eksponeringen blir mindre om en person oppholder seg innendørs under skypassasjen. Således svekker bygningsmaterialene den direkte strålingen fra skyen (*skjermingsfaktor*). Dessuten reduseres inhaleringen, fordi en del av de radioaktive stoffene som trenger inn i huset ikke når inneluften, men

blir sittende i veggene (*filterfaktor*). Både skjermingsfaktorer og filterfaktorer for noen typiske nordiske hustyper er blitt bestemt.

Prosjektet har også omfattet en undersøkelse av hvordan radioaktive stoffer avsettes på overflater (tørrdeponering) i tettbygde områder, samt en undersøkelse av avrenning fra tak. Det viser seg at tørrdeponering på flater typiske for tettbygde områder, er vesentlig lavere enn det som hittil vanligvis er antatt i analysene. Det har også vært vanlig å anta at regnvannet ikke skyller bort tidligere deponert materiale. *Undersøkelsen har vist at regnets bortvasking av slike stoffer er temmelig effektiv: 40–75%,* avhengig av takmaterialet. Slike forhold og prosesser vil bidra til å redusere stråledosen i tettbebygde områder.

En av oppgavene i prosjektet har vært å systematisere opplysninger om eksponeringsveiene via ferskvann i Norden. Mye av denne informasjonen er samlet i forbindelse med målinger av nedfallet fra nukleære prøvesprengninger i atmosfæren, og mye av denne informasjonen er relevant til store reaktorulykker, fordi de dominerende nuklider i nedfallet (Sr-90 og Cs-137) også er blandt de viktigste i forbindelse med store reaktorulykker.

Eksponeringen via jordbruksprodukter kan foregå ved at de radioaktive stoffene blir avsatt direkte på plantene og ført med disse til forbrukeren, eller til husdyr og videre til melk og kjøtt, eller ved at de radioaktive stoffene blir tatt opp i plantene fra jordsmonnet. Et slikt opptak kan skje i mange vekstsesonger etter at ulykken har funnet sted.

Den kunnskap vi har om avsetning av radioaktive stoffer på planter og opptak til planter fra jordsmonn, stammer også fra målinger av nedfallet fra atmosfæriske kjernevåpen-sprengninger. Som del av prosjektet har nedfallsmålinger i Norden blitt samlet i en databank, som direkte kan anvendes til beregning av eksponering via jordbruksprodukter.

Nedfallsmålingene gir grunnlaget for beregningene, men i tillegg behøves mer kunnskap for å bestemme de resulterende dosene til befolkningen. Således må man for hver enkelt type husdyr bestemme f. eks. hvor mye av inntaket av radioaktive stoffer som kommer til kjøttet eller melken. Det er tidligere nevnt at vi har bygget inn mer avanserte metoder for beregning av meteorologisk spredning i et risikoanalyse-program. I samme program har vi også satt inn en mer avansert næringsveimodell utviklet i Finland.

Ved et eventuelt reaktoruhell vil det være mulig å redusere de helsemessige konsekvenser til befolkningen ved forskjellige tiltak, f. eks. *evakuering eller pålegg om at folk oppholder seg innendørs*. På lengre sikt kan det være aktuelt å flytte befolkningen fra visse områder, gjennomføre opprenskningsaksjoner eller legge restriksjoner på anvendelse av matvarer eller drikkevann fra bestemte områder. Slike tiltak kan være kostbare for samfunnet, og det trengs retningslinjer for å ta beslutninger. Slike retningslinjer gis ofte i form av såkalte dosekriterier. Prosjektet har utarbeidet en oversikt over den måten dosekriterier og tilsvarende begrep brukes i en rekke forskjellige land pr. idag. Oversikten skal tjene som grunnlag for videre arbeid hos nordiske strålevernsmyndigheter.

Når en risikoanalyse skal utføres, er man ofte tvunget til å bruke parameterverdier "fra litteraturen". Dette betyr at man ofte bruker tallverdier fra bl.a. den *amerikanske reaktorsikkerhetsstudien* (WASH-1400) fra 1975. Men disse tallverdiene gjelder for amerikanske forhold og kan ikke alltid overføres til nordiske forhold. Nordiske tallverdier blir bragt frem etter hvert, og dette prosjektet har også bidratt i den retning. Det finnes imidlertid forhold som er helt spesifikt nordiske, nemlig forholdene som råder under vinteren. Prosjektet har omfattet noen undersøkelser av slike forhold. Vi har undersøkt avrenning av radioaktive stoffer deponert på *sne*, både for tak og for jordbruksmark. Dessuten er det utført et eksperiment, der vi har undersøkt hvor effektiv vanlig snerydding kan være, når man ønsker å fjerne stoffer deponert på et snedekket område. Det ble funnet at for vinterforhold var avrenningen fra jordbruksmark noe høyere, selv om det ikke var vesentlig. På tak ble det som sitter igjen praktisk talt halvert sammenlignet med sommerforhold. Effektiviteten av fjerningen var uventet høy; hele 99%. Radioaktiv forurensning kan følgelig bli vesentlig lavere under vinterforhold enn det man hittil har regnet med.

Prosjektet har bidratt vesentlig til å styrke det faglige samarbeidet innen Norden på feltet risikoanalyse. *Gjennom prosjektets resultater har vi nå et forbedret grunnlag for beregning av konsekvensene av store reaktorulykker under nordiske forhold.* Arbeidet har også vært koordinert med de store internasjonale arbeider som tar sikte på å forbedre kunnskapen om store reaktorulykker.

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1. INTRODUCTION

1.1 General

A joint Nordic safety program was initiated in 1981 as part of the Safety Program of NKA, the Nordic Liaison Committee for Atomic Energy, as planned in (Ref. NO80). The program consisted of several joint projects; it was running over a four year period, and was partially funded by the Nordic Council of Ministers.

The project reported here is entitled "Large reactor accidents - Consequences and mitigating actions". It was administrated by the Steering Group of Radioecology Projects, and had project number REK-1.

The project was subdivided into 12 subprojects, covering a large variety of problem areas. The actual work was carried out in all four Nordic countries, with Institute for Energy Technology, Kjeller, Norway acting as project leader.

The work was carried out at the following institutions:

Denmark: Risø National Laboratory, Box 49, DK-4000 Roskilde.

Finland: Technical Research Centre of Finland, Nuclear Engineering Laboratory, Box 169, SF-00181 Helsinki 18.
Finnish Centre for Radiation and Nuclear Safety, Box 268, SF-00101 Helsinki.

Norway: Institute for Energy Technology, Box 40, N-2007 Kjeller.

Sweden: National Defence Research Establishment, ABC Research Department, S-901 82 Umeå.
Studsvik Energiteknik AB, Fack, S-611 82 Nyköping 1.

Some of the subprojects were carried out at one of the above institutions, while other subprojects were carried out by several institutions in collaboration.

In (Ref. TV83, TV85 and TV85a) the project has already been presented to the international scientific community.

The main purpose of the project was to improve the representation of environmental parameters and conditions in accident consequence assessments. The word "environmental" is used here in a wider sense than usual. In connection with accident consequence assessment, the man-made or -modified parts of the environment are of particular importance. In connection with the possible harmful effects to the population as a whole, houses, roads and domestic animals and plants are more important than birds, mussels, fucus and trees.

A new joint Nordic four-year safety program will be carried out in 1985 - 1989, also partially financed by the Nordic Council of Ministers. The plans for the new safety program were published in November 1984 (Ref. NO84).

1.2 The aim of the project

The safety program has been aimed at research and development, that is expected to find practical application already within the four-year program period, in contributing to maintain the high safety level of nuclear installations in the Nordic countries. The program will also provide decision makers with background information to enable them to realistically judge the impact of nuclear power and the precautions necessary in order to maintain its safety.

The joint effort makes it possible to coordinate resources available in the Nordic countries, which in turn results in an increased efficiency of the research. The results are intended to be applicable also outside the nuclear field.

The specific project, that is the subject of the present report, is closely connected to the ongoing international effort toward improving assessments of the consequences of large reactor accidents. The manner in which these assessments were conducted, was built upon incomplete radioecological models. The aim of the project has been to improve this situation by providing more realistic values of parameters and more realistic calculation models. A large part of the project has been experimental.

1.3 Accident consequence assessment

Most accident consequence assessment (ACA) models consist of the same basic steps, illustrated in figure 1.3.1. A description of the radioactive release is given to the model, together with weather data. Using this information the model calculates atmospheric dispersion,

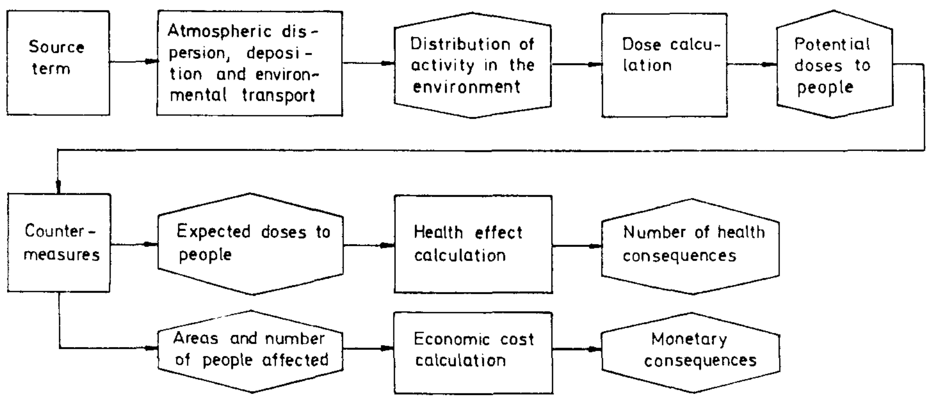

Figure 1.3.1 Schematic outline of accident consequence assessment models.

ground contamination and environmental transport. The dose calculations are performed in two steps. In the first step potential doses to the population are calculated. This information is then used to determine the proper mitigating actions (countermeasures). Then the second dose calculation is performed, in which the mitigating actions are taken into account. These doses and information on the mitigating actions are then used in the final step, calculate the health effects and the economic consequences (cost of the countermeasures). This description is of course a simplification.

Most of the project work described in this report contributes to making this type of assessment more realistic, or provides required data for local conditions. Much of the data previously used in such assessments reflect conditions in the United States. The WASH-1400 report (Ref. NU75) published in 1975 introduced the "modern" way of performing accident consequence assessments, and these methods have been widely adopted, along with much of the data material used in this reference. And because proper data for local conditions have often not been available, these data have been used for conditions quite different from the ones for which they were determined.

1.4 Uncertainties in consequence assessment

At the time when the project was initiated, there were a number of areas in which large uncertainties were suspected or known to exist. The quantitative basis for judging the relative importance of the uncertainties in various parameters or aspects of calculation models was, however, limited.

It was known that local and national variations in nutrition pathway and freshwater pathways parameters were considerable. It was known that the nutrition pathways contribute significantly to total doses, and several publications had indicated that this might also be the case for some freshwater pathways, in particular irrigation.

It was known that the shielding effect and filter effect of buildings was treated in a conservative manner in the assessments, and that natural decontamination was probably underestimated, particularly in urban areas. Furthermore there were strong indications that dry deposition in urban areas was lower than usually assumed.

It was realized that information relating to winter conditions was almost completely lacking. It was expected that some aspects of winter conditions would lead to significantly lower consequences from an accident taking place during winter.

It is not claimed that the project has resolved these issues, but it forms a significant contribution.

2. DATA RELATED SUBPROJECTS

Many parameters used for accident consequence assessments vary significantly from one geographical area to another. It is often necessary to use the data available, though this may be from another country. These subprojects contribute to improving the data base for the Nordic countries.

2.1 Terrestrial transfer factors

2.1.1 The fall-out data bank.

All Nordic fall-out data has been collected in a data bank, which is physically located on a computer in Denmark. The data bank is described in (Ref. NI81). The first compilation was performed in 1981. The bank was updated in 1984 and now contains 1082, 4013, 5438 and 5493 measurements of Cs-137 and Sr-90 from Denmark, Finland, Norway and Sweden respectively. The data cover a time period of more than 25 years. The measurements included in the data bank relate only to Cs-137 and Sr-90 from nuclear weapon test explosions.

Annual mean values in beef, pork, grain, potatoes, carrots, cabbage and apples were collected. For air, precipitation and milk it was decided to register monthly mean values.

Because of the special situation for the Lapps in the Nordic subarctic environment, there exists a vast amount of data for this region. But these special data have no general application, and it was decided not to include data from north of the 64th latitude in the data bank.

Each data point in the bank contains the result, along with seven parameters, that identify the nuclide, time, sample (air, grain, meat etc.), geographical location, relative uncertainty, unit (Bq/m^3, Bq/kg fresh, etc.), the number of measured identical samples. An already existing program system (STATDATA) (Ref. LI75) has been used for registration and treatment of the data.

It was found that in Denmark the variations between locations were so modest that it was sufficient to collect the country-wide-mean (CWM) values. In the other three countries, however, conditions vary so much that CWM values are not useful, and local values had to be registered.

It is rather difficult to make a reasonable set of location codes because different institutions in Finland, Norway and Sweden have used different divisions of their own country. When a sample has been taken at a well defined location the national postal codes can be used. The postal codes in the Nordic countries are in some way useful as location codes, because they increase going north within each country, except for the codes for Stockholm. Within the data registration program there is also the possibility of using the latitude and longitude as a location code. A complete description of the location codes is found in (Ref. NI81).

In Table 2.1.1 is given a survey over the data registered until January 1984.

		Denmark			Finland			Norway			Sweden		
air	Cs	25	CWM*	1958-82	168	CWM	1968-81	1088	8	1958-82	1191	9	1961-83
	Sr	25	"	1958-82	36	1	1968-70	0			0		
precipitation	Cs	0			252	CWM	1961-81	475	10	1962-65	706	8	1961-83
	Sr	125	"	1962-82	252	"	1961-81	0			107	2	1974-83
milk	Cs	247	"	1962-82	1494	7	1960-81	1755	11	1957-83	417	12	1966-83
	Sr	240	"	1962-82	1494	7	1960-81	1757	10	1957-79	1259	12	1965-83
beef	Cs	20	"	1963-82	43	4	1976-81	129	15	1960-72	888	8	1963-75
	Sr	19	"	1964-82	12	3	1976-78	0			0		
pork	Cs	20	"	1963-82	27	4	1976-80	110	11	1960-72	888	8	1963-75
	Sr	19	"	1964-82	8	1	1976	0			0		
grain	Cs	0			0			0			22	4	1962-67
	Sr	0			0			0			0		
barley	Cs	21	"	1962-82	10	3	1975-79	0			0		
	Sr	24	"	1959-82	10	3	1975-79	37	CWM	1957-75	0		
oats	Cs	21	"	1962-82	10	3	1975-79	0			0		
	Sr	24	"	1959-82	10	3	1975-79	0			0		
wheat	Cs	21	"	1962-82	23	6	1975-81	0			0		
	Sr	24	"	1959-82	39	7	1962-81	37	"	1957-75	0		
rye	Cs	21	"	1962-82	24	7	1978-81	0			3	3	1964
	Sr	24	"	1959-82	40	8	1962-81	0			0		
potatoes	Cs	20	"	1963-82	7	3	1974-81	0			4	CWM	1963-74
	Sr	24	"	1959-82	11	4	1974-81	17	"	1959-75	3	"	1969-71
carrots	Cs	19	"	1964-82	7	4	1974-81	0			1	"	1964
	Sr	20	"	1963-82	10	4	1974-81	17	"	1959-75	2	"	1969-71
cabbage	Cs	19	"	1964-82	4	4	1974-81	0			0		
	Sr	20	"	1963-82	5	4	1974-81	16	"	1959-75	0		
apples	Cs	20	"	1963-82	8	4	1976-81	0			1	"	1963
	Sr	20	"	1963-82	9	4	1976-81	0			1	"	1969

Table 2.1.1 List of registered data showing the number of data, the number of localities and the time period.

* CWM = Country-Wide-Mean.

2.1.2 Analytical form of prediction models.

Based upon the data in the bank prediction models for Sr-90 and Cs-137 for various constituents of human diet in the Nordic countries were derived. Data from the data bank were analyzed statistically, and the resulting prediction models and transfer factors are reported in (Ref. NI81a).

The basic model applied is a further development of the UNSCEAR model (Ref. UN77):

where $Y_i = a.d_i + b.d_{i-1} + c.A_{i-2(n)} + d.A_{i-2(R)}$

and

V_i	is the concentration of a given radionuclide in a sample collected in the year i.
d_i	is deposition (Bq/m^2) in the year i.
d_{i-1}	is deposition (Bq/m^2) in the year i-1.
$A_{i-2(n)}$	is the accumulated deposition by the year i-2, with an effective half-life of n years.
$A_{i-2(R)}$	is the accumulated deposition by the year i-2, with a radiological half-life of R years.
a	is the so-called rate factor.
b	is the so-called lag rate factor.
c	is the so-called soil factor
d	is the so-called ultimate soil factor

The effective half-life is shorter than the radioactive half-life, by definition. In addition to radioactive decay, the effective half-life takes account of processes like gradual inaccessability to root uptake, due to chemical processes or migration.

Since root uptake of Cs-137 from soils is often negligible, another model was also applied, in which the terms with accumulated deposition do not appear:

$$Y(i) = a.d_i + b.d_{i-1} + c.d_{i-2}$$

Grain modified models of the following type have been used:

For Sr-90: $Y(i) = a.d_{i(J-A)} + c.A_{i-1(n)} + d.A_{i-1(R)}$

For Cs-137: $Y(i) = a.d_{i(M-A)} + c.A_{i-1(n)} + d.A_{i-1(R)}$

where $d_{i(J-A)}$ is deposition of Sr-90 (Bq/m^2) in July and August of the year i.

$d_{i(M-A)}$ is deposition of Cs-137 (Bq/m^2) in May to August of the year i.

The prediction model for Cs-137 in grain may look somewhat misleading, since the soil terms are included. But they are included only for the case of generality. When numerical values are entered, they disappear.

The different time periods involved for Sr and Cs is due to the fact that Cs is translocated in the plant, while Sr is not. Accordingly, Sr contamination of the grain only takes place after the ears have been formed, in July, while Cs contamination of the plant in May and June can be translocated to the grain, even if the ears were not formed at that time.

2.1.3 Deposition in the Nordic countries.

The prediction models were determined on the basis of data on deposition. For none of the four countries are measurements of deposition quite satisfactory for this purpose, but it has been possible to construct the missing data by using data from other locations together with certain assumptions.

The concentrations of fallout nuclides vary with location on a global scale. However, within the band of latitudes covered by the Nordic countries (55^0 - 67^0N) (arctic area excluded), the deposition may be expected to be proportional to the amount of precipitation.

In Denmark no Cs-137 deposition data were available, and Cs-137 deposition had to be constructed from the Sr-90 data, using the assumption that the amount of Cs-137 deposited is 1.6 times that of Sr-90, as in other locations (Ref. AA79 and UN77). This ratio is, however, a little lower than the ratio between Cs-137 and Sr-90 in air, as measured at Risø from 1962 to 1979.

For Finland the data bank contains only countrywide meanvalues of deposition, since no significant difference between locations was found. One surpricing aspect of the data is that the ratio between deposition of Cs-137 and Sr-90 over the years 1961 to 1978 was as low as 1.29 (compared to 1.6, which is considered typical).

The Norwegian and Swedish data showed significant differences between locations, so that it was decided to divide Norway into four regions and Sweden into three. It is possible that these differences may simply be due to differences in the amount of precipitation. This aspect has, however, not been examined, and the necessary information may not be available.

2.1.4 Prediction models derived from Nordic fall-out data.

The data in the data bank has been used to find the best fit of the parameters in the prediction models, as well as the effective and radiological half-lives. This has been done using the computer program FIT (Ref. LI75). A least squares procedure is used, and the best fit is the set of parameter values for which the highest correlation coefficient between observed and calculated values is found.

2.1.4.1 Grain and vegetables.

Data for grain were available from Denmark, Finland and Norway, while data for vegetables were available only from Denmark and Norway. The Norwegian data were only for Sr-90. One further complicating factor is that the Danish and Finish data are given in units of Bq/kg, while the Norwegian data are given in units of Bq/(g Ca). It is not possible to make a direct conversion since Ca-content per kg grain or vegetable varies with location.

It is unfortunate that such difficulties in interpretation should exist, and they could be avoided in the future if monitoring programs can be run as Nordic projects.

Prediction models were derived for barley, oats, rye, wheat, apples, cabbage, carrots and potatoes.

One prediction model is included here, as an example. It refers to cabbage in Denmark (period 1963-1979), and gives the Sr-90 activity in Bq/kg fresh weight (in the year i):

$$4.7.10^{-4}\ d_{i} + 2.7.10^{-4}\ d_{i-1} + 2.6.10^{-4}\ A_{i-2(1)} + 1.6.10^{-4}\ A_{i-2(28)}$$

where

d_i is deposition (Bq/m^2) in the year i.

d_{i-1} is deposition (Bq/m^2) in the year i-1.

$A_{i-2(1)}$ is the accumulated deposition by the year i-2, with an effective half-life of 1 year.

$A_{i-2(28)}$ is the accumulated deposition by the year i-2, with a radiological half-life of 28 years.

2.1.4.2 Milk.

Both Sr-90 and Cs-137 have been monitored in milk since 1962 or earlier, and are still monitored in all the Nordic countries. More than half of the data in the data bank are milk data.

Again there are some unnecessary difficulties in interpretation, as the Danish and Finnish data are for dried milk, while the Norwegian and Swedish data are for fresh milk.

There are strong variations between years and between locations. These are due to various factors, but are probably dominated by variations in the composition of the fodder. Since composition of the fodder is so important, the prediction models for milk are probably less useful than some of the other prediction models derived. Prediction models for very well-defined situations would have been more useful than the models that can be derived from the data bank.

2.1.4.3 Meat.

The data bank contains only meat data on beef and pork, and it does not contain data from Finland. No Sr-90 data were available from Norway and Sweden. This is not a serious shortcoming, as transfer of Sr-90 via meat is much less important than transfer of Cs-137.

It is interesting to note that the lag rate factors in the prediction models determined for Denmark and Sweden (see the equation in the beginning of chapter 2.1.2 for explanation) are larger for pork than for beef, due to the fact that fodder for pigs consists mainly of barley from the previous year, while grass and beets are the main constituents of cattle feed.

It was impossible to fit the Norwegian beef data to any prediction model. Transfer via beef in Norway must be estimated on the basis of the milk data, together with the assumption that the concentration of Cs-137 in beef (Bq/kg) is 4 - 5 times that in milk (Bq/l), as found in Denmark and Sweden.

2.2 The fresh-water pathways

The purpose was to summarize the information available in the Nordic countries on the freshwater exposure pathways, and to review this information (Ref. TV84a), particularly seen in connection with calculations of large reactor accident consequences.

Off-hand one would expect the freshwater pathways to be of equal importance in the four Nordic countries, because of the similarities in climate. The geographical area is one of abundant precipitation, evenly distributed over the year. But it is found that conditions in Denmark are quite different from conditions in the other three countries, in the following ways: In Denmark lakes are few and modest in size (1.3% of the area). There are no rivers, but a number of streams. Drinking water is ground water pumped up from considerable depth. There is also a very limited supply of freshwater fish; and swimming and recreational use of beaches mostly takes place by the ocean, which is not very far from anywhere in Denmark.

Accordingly, the freshwater pathways are not important in Denmark, and the rest of this chapter concerns only the three other countries.

The Nordic countries are characterized by having a rather severe winter, though the length and severeness varies much with latitude and distance from the ocean. The radiological impact of winter conditions is, however, to a large degree unknown. Some aspects of winter conditions are dealt with in chapters 3.1 and 3.2.

2.2.1 Deposition.

Deposition upon ground from a cloud containing radioactive material is routinely calculated as part of accident consequence assessments. In connection with the freshwater pathways, however, one is also interested in deposition upon water surfaces.

Deposition may take place during wet and dry conditions. During wet conditions the deposition per unit area will be the same upon a water surface as upon a ground surface, since the properties of the surface itself are not important. Dry deposition, however, is quite different on surfaces of different types.

Most measurements of caesium and strontium behavior in the environment are measurements upon nuclear weapons fallout. Deposition of nuclear weapons fallout, at least in climates like the Nordic, is almost entirely wet deposition. This is also shown in work performed at the Norwegian Defence Research Establishment (Ref. HV66 and LI82). This also implies that deposition of fall-out is the same per unit area upon ground and water surfaces.

Dry deposition is expected to be smaller upon a water surface than upon a ground surface, because water is a smooth surface. However, deposition upon a water surface will also depend upon wave characteristics. Higher waves would lead to higher deposition. The highest air concentrations, however, are usually reached under conditions with low wind speed and low waves, so the wave effect may be of limited practical significance.

Available data for dry deposition upon water surfaces are quite limited. Some measurements of the deposition velocity of sulfurdioxide gas on water are, however, referred to in a survey from the Norwegian Air Research Institute (Ref. GO78). The individual measurements are described in (Ref. GA77, GO77, SH74 and WH74). In SH74 it was found that the deposition velocity was 1/1000 of the wind velocity. WH74 reports deposition velocities of 0.16 cm/s for stable, 2.2 cm/s for neutral and 4.0 cm/s for unstable weather conditions. The other two references give deposition velocities of 0.3 and 0.4 cm/s.

2.2.2 Run-off.

Radioactive materials deposited upon the ground will to a certain extent be washed away with run-off water; either in solid or dissolved form, and will eventually reach a lake, a river or the sea. Vertical migration in soil may also take place, leading to release to rivers or lakes via ground water. This process is, however, too slow to be of concern in connection with calculation of severe reactor accidents.

Extensive measurements giving information about run-off have been performed in Finland (Ref. SA84). The catchment areas of five larger Finnish rivers have been studied. Sr-90 and Cs-137 from nuclear weapons tests have been monitored in deposition upon ground, in the five largest rivers, and in certain lake and river systems in Finland since the midsixties. These observations show an even distribution of Sr-90 and Cs-137 deposition over the country, but regional differences are found in their concentrations in surface water. These observations give a basis for determining the total removal from Finnish ground of these isotopes, and also for describing some aspects of their behavior in the environment.

In (Ref. SA84) are reported comparisons of the amounts of Sr-90 and Cs-137 deposited in Finland and removed from Finland up to 1981. It was found that the average removal by water was 41% for Sr-90 and 7% for Cs-137. As the water surface is about 9% of the Finnish area, these results indicate that Sr-90 is significantly removed from ground by run-off, while Cs-137 is not. It is even indicated that part of the Cs-137 deposited directly upon water surfaces is trapped in sediments in rivers and lakes.

In large parts of Finland, the percentage of freshwater surfaces is much larger than in most other parts of the Nordic countries. It will probably not be possible to ignore the contribution from run-off of Cs-137 from ground surfaces in most parts of the Nordic countries. One possible exception in addition to Finland (or much of Finland) is the middle part of Sweden, where there are a number of very large lakes.

There is probably also a link between run-off and the type of terrain. Most of Norway is very hilly or mountainous, and so are the Northern parts of Finland and Sweden. This type of terrain is often combined with special ground conditions. The bedrock is then often covered with only a thin layer of soil and sparse vegetation. Under these conditions one would expect run-off to be much more pronounced than in typical agricultural areas or in dense forest areas.

In (Ref. LU62) are reported some measurements performed in the Møsvatn area, a mountain region in Norway; at altitude from 936 meters to 1100 meters above sea level. The area is dishshaped with lakes at different levels. All water from lakes in the area runs into a single lake at the lowest level. This lake has only a single water outlet. This makes the area exceptionally well suited for studies of deposition and run-off. The total area is 5.91 km^2, and the lakes cover 6.9% of the area.

It was found that during summer, the concentration of Sr-90 and Cs-137 in the lower lake corresponded to deposition onto the water surfaces of the lakes, with no contribution from runoff from the surrounding ground surfaces. In winter and the melting season the conditions were quite different. It was found that there was a peak in the activity concentration in the lake during spring (when the snow is melting). The snow acts as a reservoir of activity. There were also peaks during winter, and these coincided with time periods with above-freezing temperatures. The activity content in the snow-cover of the lake surface itself is not high enough to explain the magnitude of the peaks, and it was concluded that there was significant contribution from run-off from the surrounding ground areas. Under special conditions during winter as much as 50% of the activity deposited in the catchment area could reach the lake. During the spring flood the amount of fallout brought into the lake corresponded to twice the amount contained in the snow on the lake surfaces. These conclusions are drawn for Sr-90 and for Cs-137.

In the same reference are reported measurements performed in another catchment area in the Southeastern part of Norway. This is quite a different type of area, covered with dense forests, and where only 0.4% of the area is water surface. Here it was found that the concentrations in the outlet from the area varied much less than the concentrations in precipitation. This indicates that only a smaller part of the radioactive materials in the outlet water was deposited directly upon water surfaces.

2.2.3 Soil.

The measurements reported in (Ref. SA84) indicate that run-off is quite different for different types of soil, and that run-off is much stronger for strontium than for caesium in all types of soil. The typical soil in South-Western Finland is clay soil. Strontium deposited upon this type of soil does to a large degree run off, while caesium almost completely stays where it is deposited originally. Peat soil is the common type of soil in most other parts of Finland. This type of soil is also the most common in Norway, and there is reason to believe that data from these catchment areas in Finland are valid for much of Norway, and probably for much of Sweden.

Typical of two of the Finnish catchment areas is large proportions of bog. In these areas of low clay content, run-off of Cs-137 was found to be somewhat larger than in the other areas; but still only 6-8% of the total amount deposited in the catchment area.

The vertical distribution of Cs-137 in soil also gives an indication of the mobility, and how easily the material can be washed out. Measurements have been performed in Norway (Ref. LU62 and LI82).

Measurements of the vertical distribution of Cs-137 in soil have also been performed in the other Nordic countries. In (Ref. PR82) are given results from measurement of the vertical distribution in some types of soil in Denmark, Finland and Sweden. In all instances almost all caesium is found in the upper 15 cm, but the distributions differ. In (Ref. AR83) are reported measurements of caesium distribution in different types of soils in Finland, including cultivated pasture. Measurements of the vertical Cs-137 and Sr-90 concentrations in five different Finnish soils are also reported in (Ref. HÄ68).

Uptake in plants can also give an indication of the mobility of Sr-90 and Cs-137 in different soil types (Ref. RA82). Measurements in Finland and Sweden have shown that Cs-137 becomes unavailable to plants after a few years in soils with sufficiently high clay content. Organic matter in the soil, on the other hand, keeps Cs-137 mobile (Refs. ER77 and KU72). High organic matter content in soil reduces root absorption of Sr-90. This may, however, be due to other factors than reduced mobility in the soil. Sr-90 in soil is otherwise available for root uptake for many years after deposition.

2.2.4 Rivers, lakes, sediment.

The content of radioactive materials, originating from an atmospheric release, in a water body will consist of material deposited upon the water surface, run-off of freshly deposited material in the catchment area, and run-off (desorption and erosion) of previously deposited material in the catchment area.

Rivers and lakes contain the radioactive materials in solution or fixed on solid particles. In the latter case the particles may be in suspension or settle on the bottom of the river or lake. A systematic examination of the Cs-137 concentrations in a lake in Southern Sweden was performed up to 1975 (Ref. CA76). Among the measurements performed were some in which the activity concentrations in filtered as well as unfiltered lake water were measured. These were found to be almost identical, indicating that only an insignificant fraction of the Cs-137 in the lake water was fixed on plankton or suspended material. In a river, however, one must expect a much larger portion of the activity to be in suspension. In the following parts of this chapter, it will become evident that a significant part of the deposited activity may be contained in the sediments on the lake bottom. And this seems to indicate that a <u>not</u> insignificant part of the activity is fixed on plankton or suspended material. This apparent conflict has not been resolved at the time of writing.

There is a continuous settling of sediment on the lake bottom. A typical velocity of sediment buildup is somewhat less than 1 cm per year (1-2 mm per year may be a realistic value). The content of radioactive materials in sediments may give some information about the behavior of nuclear weapons fallout in the environment. If the total content in sediment per unit area is more than 100% of the accumulated fallour (corrected for radioactive decay) per unit area, run-off from the catchment area is clearly significant. Such a result would also indicate that resuspension from the sediment probably is insignificant. Measurements in Norway (Ref. AU78) performed upon three sedimentation cores fetched from the bottom of lake Årungen, show that 98%, 87% and 47% respectively of the accumulated Cs-137 fallout per unit area in the district was found in the sediment samples. The estimated turnover of the water in this lake is once every four months. The results seem to indicate that the contribution from run-off from the catchment area is not significant. But this can not be known for certain, without information on how much Cs-137 leaves the lake via the river outlet and evaporation. This information was not collected in this case.

Measurements performed in Finland on sediment indicate that sediment contains much more caesium than strontium. Also, the amount of strontium in the water column above sediment in a lake is roughly equal to the amount of strontium in the sediment; while for caesium, the amount in the water column is much less than in the sediment. In a mass balance, the sediment is often of little importance; meaning that the amount of radioactive material remaining on ground surfaces and the amount going with the freshwater into the sea is much larger than the amount being trapped in the total of all sediments in the freshwater compartments. This is, however, not always true if the fraction of water surface is large. In Finland it is often 10-20%, and the Cs-137 content in sediment per m^2 water surface area may be larger than the amount of accumulated Cs-137 per m^2 on land.

Swedish measurements in lake Ulkesjön (Ref. CA76) gave as result that the caesium content per unit area in sediment was 67% of the cumulative deposition per unit area in the catchment area, which agrees well with the Norwegian measurements. But it is also said in (Ref. CA76) that the value is low compared to other findings (Refs. RI72 and MC73).

Vertical migration of caesium and strontium in the sediments has also been investigated. Norwegian measurements (Ref. AU78) were performed with the intention of using measurement of the Cs-137 content for dating of sediment. This is only possible if the sediment, once it is deposited, remains largely undistrubed. There should be little or no biological disturbance of the sediment, no significant resuspension, and no significant vertical migration of the radionuclide concerned. The measurements indicate that these conditions are fulfilled for the Norwegian lake investigated. It was, however, found in Finnish measurements in a lake beside the town of Tampere, that the vertical distribution was quite different. The Cs-137 was found to be homogenously distributed over the top 20 cm of sediment. Tampere is an industrial town, and there are large releases to the lake as well as considerable traffic on the lake. In undisturbed lakes, measurements in Finland also indicate that all Cs-137 is found in the top 5 - 10 cm, with a strong gradient. Similar Swedish results are found in (Ref. CA76). The vertical Cs-137 distribution in sediment in the lake Ulkesjön has been both measured and calculated by a mathematical model developed in this reference.

From a population exposure point of view, an accumulation of radioactive materials in sediment is an advantage. The sediment will act as a sink. Only in relatively infrequent situations do persons come in contact with sediment. The bulk of the sediment will be situated on the lake bottom, where it is inaccessible. After hundereds or thousands of years, the sediment may, however, once more be exposed, but this is not an important concern in connection with large reactor accidents.

The Cs-137 content in sediment may vary strongly, even quite locally. As could be expected, the concentration is higher in sediment in deep tranquill lakes. The variation in concentration is particularly strong in rivers, where the high concentrations are often found on the inner side of river bends. This aspect is of some significance in connection with population exposure, as the inner side of river bends often have the finest sand, and accordingly are the most popular places for swimming, sunbathing and camping. The concentrations in sediment are in Finland found to vary up to a factor of 100 at the same location.

2.2.5 Drinking water.

The effect of water purification plants upon strontium and caesium concentrations is found in Finnish investigations to be quite unimportant. The strontium concentration is hardly affected, and the reduction of caesium concentration is at the most 50%.

Exposure via drinking water after an accident can be estimated, using rather crude methods. Such an estimation has been carried out in (Ref. TH82).

It has been assumed that a very large release has taken place, and deposition takes place upon the surface of the reservoir, like on the surrounding areas. The atmospheric dilution assumed corresponds to a transport distance from release point of 10-30 km. Depth of the reservoir is assumed to be 10 meters. It is furthermore assumed that the water usage is 1 liter per day.

The doses calculated, using these rather conservative assumptions, are approx. 0.1 Sv (10 rem) to the thyroid from I-131 and 0.001 Sv (0.1 rem) whole body from Cs-137. These doses are very small compared to doses via other exposure pathways when such a severe accident is postulated.

2.2.6 Fish.

Only 1% of the Cs-137 present in a typical lake is found in the fish belonging to that lake (Ref. CA76). On the other hand, the concentration in fish is much higher than in the water. The Cs-137 concentration in fish has been extensively studied, and Nordic measurements are reported in (Refs. CA76, KO66, KO68 and KO70) among others. Typical concentration factors are 200-10000.

In the fish exposure pathway strontium is less important than caesium, as strontium concentrates in the fish bone. Most of the freshwater fish consumed are of species where the bones are easily removed, and it must be valid to assume that less than 10% of the fish bone will be consumed. In (Ref. FL71) it is claimed that aquatic food loses 20-50% of its radioactivity during preparation and processing prior to consumption, without reference to any particular nuclide.

The concentration of Cs-137 in fish depends, among other factors, also on the feeding habits of the fish. This varies with age and size of the fish, as well as with species, and there are large individual differences. The differences in concentration between fish from different lakes are reported in a Finnish investigation to be larger than that between fish from different parts of the Baltic Sea (Ref. SX84).

It is found that the concentration in fish is higher in oligotrophic lakes than in eutrophic ones.

In a Swedish report (Ref. MA70) the contributions to average individual doses in Sweden from various foodstuffs for the years 1962 to 1968 are given. The information is reproduced in table 2.2.1. It was found that in 1968 freshwater fish contributed 15% of the dose. The comsumption of freshwater fish is in this report assumed to be almost 20% of the total fish consumption, a figure which according to the Swedish reviewers of the report is realistic also for presentday conditions.

The Finnish reviewers have found that a 15% contribution from freshwater fish consumption to the nutrition dose may even be somewhat low for Finnish conditions (Ref. RA84).

Similar information has not been found for Norwegian conditions. It is reported, however, that average freshwater fish consumption in Norway is about 10% of the total fish consumption, which is given as 30 kg per person and year (Ref. IN82). This is quite similar to the value given for Sweden for freshwater fish consumption.

Year	1962	1963	1964	1965	1966	1967	1968
Dairy products	23.0 (41%)	35.1 (37%)	33.8 (25%)	23.7 (30%)	13.4 (29%)	9.7 (30%)	7.4 (29%)
Meat	18.2 (33%)	28.8 (30%)	38.4 (28%)	22.1 (28%)	11.0 (24%)	4.3 (13%)	4.3 (17%)
Grain	- -	13.4 (14%)	33.7 (25%)	13.6 (17%)	5.6 (12%)	2.8 (9%)	1.8 (7%)
Lake fish	2.5 (4%)	2.5 (3%)	6.3 (5%)	6.3 (8%)	6.3 (14%)	5.6 (17%)	3.8 (15%)
Salt water fish	1.6 (3%)	1.6 (2%)	1.6 (1%)	1.6 (2%)	1.6 (3%)	1.6 (5%)	1.6 (6%)
Reindeer meat	5.1 (9%)	5.1 (5%)	11.7 (9%)	6.0 (7%)	4.2 (9%)	6.0 (18%)	5.1 (20%)
Total intake, incl. "others"	56	87	137	80	46	32	26

Table 2.2.1 Summary of annual cesium-137 diet intake from various foodstuffs. Sweden. Nationwide averages (nCi/person).

Additional information:

Milk consumption 175 kg (From Nat. Swed. Inst. of Publ. Health)
Cheese " 18 kg (")
Meat " 48 kg (")
Grain " 88 kg (")
(Estimated that 50% in flour and consumption of harvest of the preceding year.)
Reindeer meat consumption 0.3 kg.
Fish consumption 17.6 kg (Estimated that 2.5 kg is lake fish, and the rest salt water fish. About half the lake fish estimated to be from oligotrophic lakes.)
"Others" is estimated to be 10% of the total intake excluding reindeer meat.

2.2.7 Irrigation.

Contamination after an accident in a nuclear power plant via irrigation water is similar to contamination via rain in some respects, and different in other respects. The most important difference is the following: Contamination via rain can only take place during the time it takes the cloud containing radioactive materials to pass over the area in question. Contamination via irrigation water can continue as long as there are radioactive materials present in the water body from which the irrigation water is drawn. Accordingly, although the concentration in irrigation water may be much lower than in rain water, the accumulated deposition may nevertheless approach a comparable level.

A rough estimate of the relative importance of exposure via irrigation is carried out, based upon Norwegian agricultural statistics, as presented in tables 2.2.2 and 2.2.3. The conditions in the other Nordic countries are probably similar enough to make the conclusions drawn valid also in these countries.

Table 2.2.2 gives some information about irrigation in Norwegian agriculture (Ref. IN83). The areas cited here are for the country as a whole. The geographical distribution is very uneven. Table 2.2.3 contains the areas by county, and the areas cover all irrigation plants in existence in 1979 (Ref. CE79). Most irrigated land is found to be located in the central parts of South-East Norway, including the counties along the Oslo Fjord. Table 2.2.2 shows that there has been a sharp decline in the building of new irrigation plants since 1978. This may be due to favorable weather or to less favorable government subsidies.

Year	Number of new installations	Area served by new installations (decares) (1 decar is 1000 sqm)
1974	-	21,198
1975	-	30,439
1976	671	58,980
1977	1112	151,812
1978	1017	105,740
1979	706	75,107
1980	409	35,316
1981	342	29,804
1982	252	17,055

Table 2.2.2 Investments in irrigation equipment in Norway.

County	Number of plants	By capacity in m^3/h				Area covered (decares)
		<10	10-24	25-49	>50	
The whole country	10,624	4,175	3,344	1,885	651	703,030
Östfold	578	123	209	117	48	69,768
Akershus and Oslo	544	219	136	106	51	49,986
Hedmark	1,077	332	287	254	158	138,257
Oppland	2,322	703	856	516	142	148,846
Buskerud	1,014	332	304	224	71	74,016
Vestfold	792	258	345	130	26	56,688
Telemark	506	291	107	50	10	17,931
Aust-Agder	677	392	200	47	15	24,343
Vest-Agder	455	254	124	41	16	17,844
Rogaland	460	118	142	113	51	29,810
Hordaland	411	263	115	11	6	10,961
Sogn og F.	1,180	564	366	140	48	41,250
Möre og Roms	237	144	40	27	2	8,481
S-Tröndelag	86	51	24	4	-	3,234
N-Tröndelag	198	87	68	36	5	8,277
Nordland	51	28	12	8	1	2,338
Troms	24	12	2	1	1	302
Finmark	12	4	7	-	-	698

Table 2.2.3 Irrigation plants in Norway up to and including 1979.

Even with conservative assumptions, it is found that irrigation will add very little to the activity concentration on land, and that irrigation can not be an important exposure pathway. The rough estimate is explained in detail in (Ref. TV84a).

2.3 Comparison of dynamic and static models for saltwater fish

The traditional manner in which the dose via fish is calculated is based on the assumption that there is equilibrium between water and fish, and enrichment factors are used. It is doubtful whether equilibrium is achieved even in relation to releases from normal operation of nuclear power plants, as the releases are not continuous; and equilibrium is certainly not achieved in an accident situation. Uptake in the fish will also be influenced by factors such as water temperature and salinity, but these factors have not been addressed in this subproject (Ref. AG83).

The subproject has consisted of two sets of calculations, using the Swedish computer code BIOPATH (Ref. BE82), assuming static and dynamic conditions respectively in the two calculations. The calculations have been performed for a specific area on the Swedish East coast. The data used in the dynamic calculations were collected at Studsvik, while those used in the static calculations were taken from (Ref. PE73).

This fact made a certain adjustment of the data necessary. Some test calculations, assuming constant concentration in the water, seem to prove that consistency has been achieved. As would be expected, the calculated dose per meal of fish will be the same for the two methods, allowing about half a year for the concentration in the fish to build up to the equilibrium value, in the dynamic calculation. Results from one of these test calculations are shown in figure 2.3.1.

The calculations were performed for the nuclides Cs-137, Co-60 and Zn-65. In the calculations for the accident situation, the concentration in water will decrease, as the water in the area is replaced by uncontaminated water. The results of one of the calculations are shown in figure 2.3.2. In the static model the dose per meal of fish (hereafter referred to as the "dose rate") is by definition proportional to the concentration in the water, and the highest "dose rate" is received directly after the accidental release has taken place. In reality it takes some time before the activity is actually taken up by the fish. This situation is reflected in the results from the dynamic calculations. The maximum "dose rate" from the dynamic calculations is lower and occurs at a later time. Later on, however, the "dose rates" calculated with the dynamic model are higher than the ones calculated with the static model. As mentioned, the "dose rate" calculated with the static model is by definition proportional to the concentration in the water. And from the figure it can be seen that the "dose rate" decreases, as the contaminated water is diluted. But the "dose rate" should not decrease as fast as the concentration in the water. The fish will contain activity taken in previously, when the concentration in the water was higher. The results from the dynamic calculations show this effect.

Although the calculations demonstrated that the "dose rates" calculated by the two methods differed considerably, it was also found that the accumulated doses over one year calculated by the two methods were practically identical. The accumulated dose is, of course, the entity which might have a radiological significance. The "dose per meal" can hardly matter. One conclusion that can be drawn from this subproject is then that in the situation analyzed, the simpler and cheaper of the two methods is acceptable. If this conclusion has a more general validity, it may be possible to save considerable time and effort.

2.4 The shielding effect of buildings

In Denmark, at Risø National Laboratories, shielding factors for buildings had already been calculated before this subproject was planned. And in this connection a computer program for calculation of shielding factors had been developed (Ref. HE81).

It was decided to extend this work, as buildings in the other three Nordic countries will have different shielding factors, due to differences in building traditions. The subproject was carried out in cooperation between Denmark and Norway. The Norwegian contributor collected information on building traditions in Finland, Norway and Sweden, and the Danish contribution consisted of the actual calculation of the shielding factors. The results of the subproject are reported in (Ref. TV82).

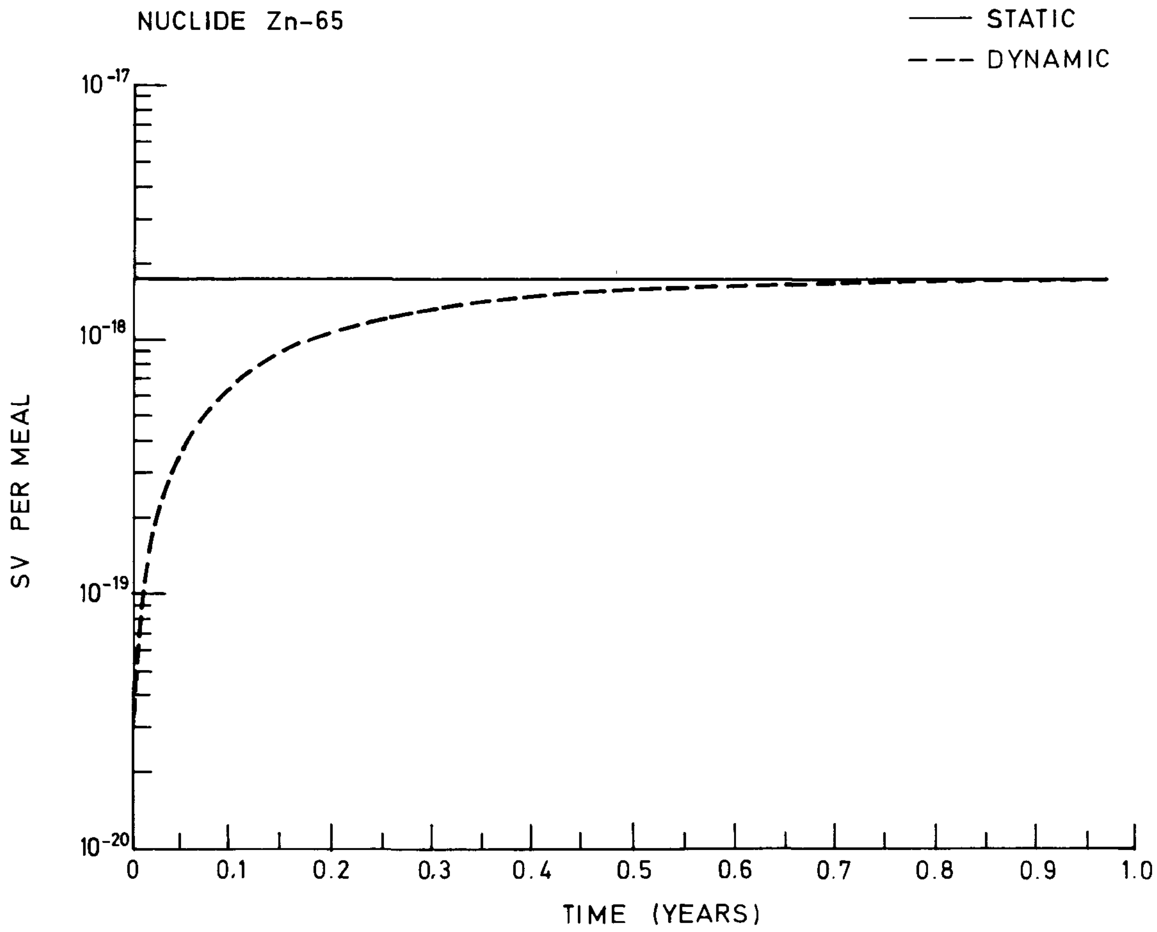

Figure 2.3.1 Individual dose per meal of fish. The Zn-65 concentration in water is assumed to be constant.

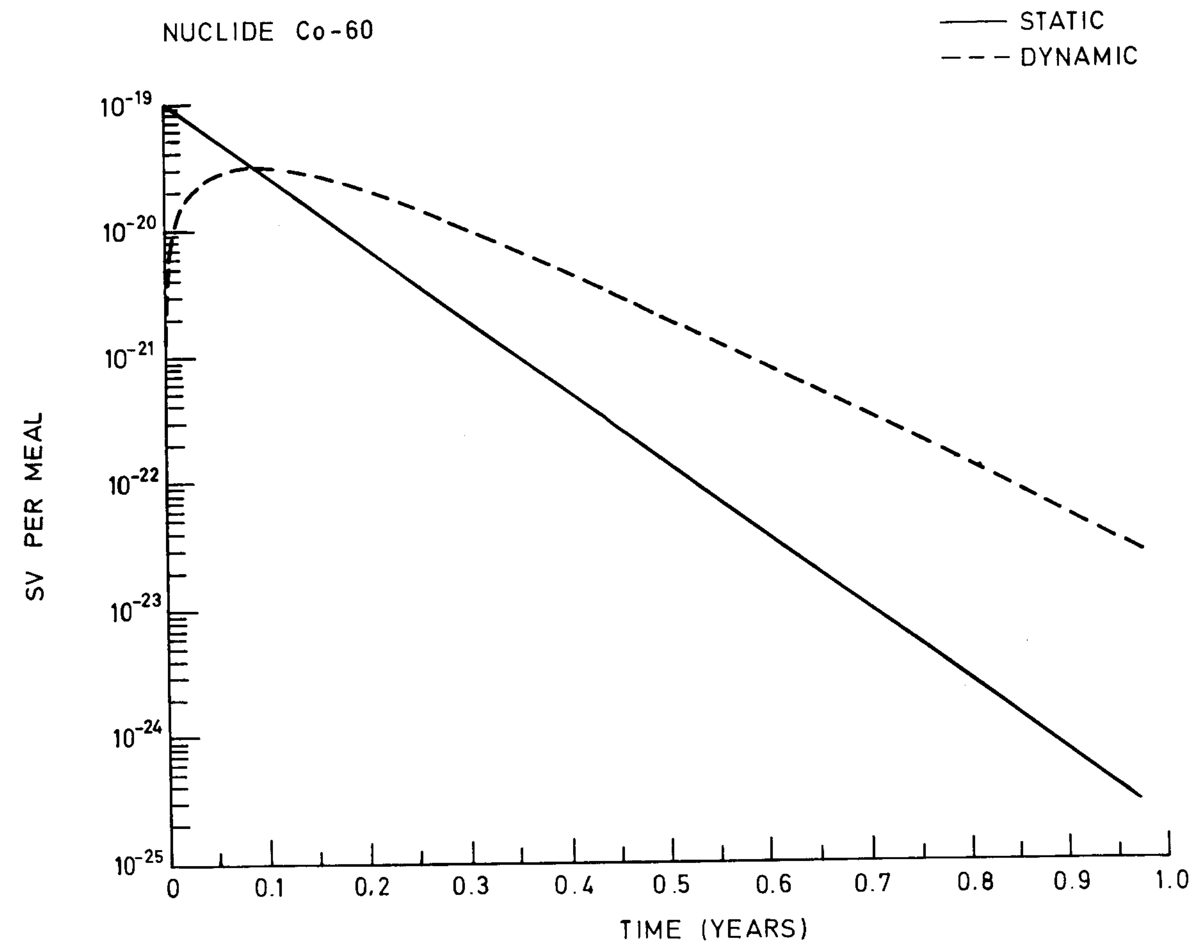

Figure 2.3.2 Individual dose per meal of fish. The Co-60 release is assumed to be instantenious at time zero.

Type of information	Denmark	Finland	Norway	Sweden
Dimension perpend. to length of bldg. (m)	8.0	8.4	7.2	9.5
Width of a room (m)	15.0	4.0	8.0	4.5
Height of one storey (m)	2.5	2.5	2.4	2.7
Area-weight outer wall (kg/m^2)	400	50	36	150
Area-weight inner wall (kg/m^2)	220	40	-	50
Area-weight roof + ceiling (kg/m^2)	110	100	81	100
Percentage windows	25	20*	30	17

\- Not used in the calculations.
* Given as (20 10)%. In calculations used 20%.

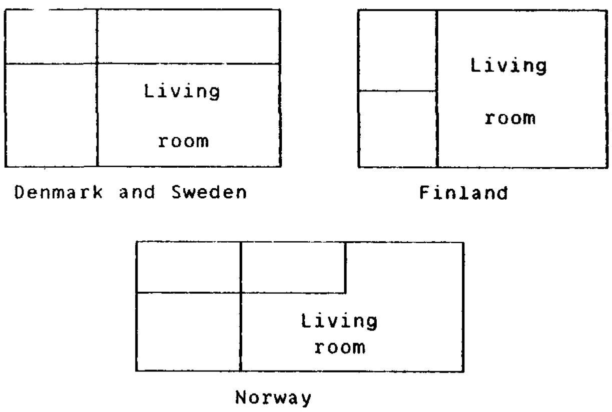

Table 2.4.1 Building data and typical lay-out of one-family house. The lay-out drawings show the house seen from above.

Typical building characteristics were collected for one-family houses and apartment buildings, and one set of characteristics were selected for each of the two types of buildings for each country.

The building characteristics were obtained from the Institute of Radiation Protection, Finland; Statens Planverk, Sweden, and the Norwegian Building Research Institute. Typical buildings differ both in wall and roof materials and thicknesses, typical dimensions and lay-out. The lay-out is important, because of the extra shielding provided by internal walls.

For a multistoried building, the report (Ref. TV82) gives the shielding factors for each floor. The report also contains the separate shielding factors that should be applied to activity deposited upon the road, on the external wall or on the roof. Table 2.4.1 gives data for one-family houses in the four countries. Table 2.4.2 gives the shielding factors calculated for Finnish houses, as example of the results obtained.

The main purpose of the subproject has been to find out if differences in building traditions give significant differences in shielding factors, and the differences found were actually larger than expected. Obviously, shielding factors for two types of buildings per country are insufficient for calculations of risk or collective dose. But extension of this work can easily be performed, when one wants to examine a specific location.

	Single-family house	Number of storeys in apartment building				
	.36	2	3	4	5	6
Dose-point in storey no.	Ground floor	.028	.027	.027	.027	.027
	1st	.042	.011	.0097	.0097	.0097
	2nd		.039	.0082	.0068	.0068
	3rd			.039	.0082	.0068
	4th				.039	.0082
	5th					.039

Table 2.4.2 Shielding factors for typical Finnish houses.

3. EXPERIMENTAL SUBPROJECTS

3.1 Natural decontamination of roofs

In consequence assessments of large, hypothetical reactor accidents, with the assumptions used at present, one of the most important exposure pathways is by radiation from radioactive materials deposited on ground. "Ground" in this connection also means building surfaces; and as most of the collective dose will be received by residents in urban areas, deposition on houses is of particular importance. In the consequence assessments it is usually assumed that deposited radioactive materials stay where they were deposited, but this may be very conservative relating to building surfaces.

The purpose of the subproject is to determine what happens, over a period of several months, to caesium deposited on a roof. The Norwegian Building Research Institute kindly helped choose two types of roof materials that would be typical of houses built in Finland, Norway and Sweden. The small "roofs" were covered with these roof materials, and left to "age" for some weeks. One of the roof materials was tar-paper, and this is typical of apartment buildings. The other roof material was a metal roof with plastic coating, shaped to simulate black cheramic tiles. This roof material is popular on single-family houses. The slants of the roofs were also chosen to appropriately simulate apartment buildings and single-family houses respectively.

It was not attempted to examine roof materials typical of Denmark, since this had already been done (Ref. GJ82) at Risø National Laboratory. The typical Danish roof material, however, is porous tiles, which rarely is found in the other Nordic countries.

A short description of this subproject is given in (Ref. QV84), and a thorough description is found in (Ref. QV84a).

3.1.1 Description of the experiments.

Each of the "roofs" consisted of a supporting structure with a plywood board on top, to which the roof material was attached. The dimensions of the "roofs" were 1.2 m x 2.4 m. One of the short sides is the lower edge of the roof, from which the run-off water is collected. Around the "roofs" were mounted plastic sheet wind shields 0.5 m high. The lowest part of the "roof" was 1.5 m above ground. The set-up is shown in fig. 3.1.1, 3.1.2 and 3.1.3.

The experiments were carried out during the fall and winter of 82/83 and during the winter of 83/84. In the first set of experiments the roofs were contaminated with Cs-134 during dry weather conditions. The rain water running off the roofs was collected, and the content of Cs-134 determined. Later on in the winter of 82/83, contaminant was applied on top of a layer of wet snow on a tar-paper roof. And in February 1984 an experiment on proper sub-zero winter conditions was performed. There was then a stable layer of ice and coarse-grained snow ca. 5 cm thick on the roof, which was a tar-paper roof. In all cases the run-off water was collected and measured, whether it was melt-water or rain-water or both.

3.1.2 Weather conditions.

The weather conditions are of course important for the outcome of these experiments, and the conclusions drawn are basically valid only for the particular sequence of weather conditions we happened to get in each case.

November 1982 was extremely humid, with almost double the normal amount of precipitation. December had about 150% of the normal. Ice was formed on the roof during this period, and somewhat lower concentrations of Cs-134 were observed in the collected water. January, February and March of 1983 were all unusually mild and dry, and the last snowfalls occurred in the beginning of April. During May precipitation was almost normal, while June was drier than normal.

The experiments of winter 83/84 were started in the end of February, during a light snow-fall and a temperature of -7^0 C. The weather during March was changeable and somewhat colder than average, but there were mild periods. Precipitation was about normal, and falling mostly toward the end of the month. Nine days after contamination we had the first run-off, and after 15 days all the snow had disappeared. In April both temperatures and precipitation were lower than normal. May had higher temperatures and precipitation than normal. In June also precipitation was larger than normal.

The pH values of the water from the paper roof and the metal roof were measured on the 1st November 1982 as 5.8 and 4.2 respectively.

3.1.3 Contamination.

The radioactive material used in the experiments was Cs-134 (as cloride) in an ordinary solution with 1-10 mCi/mg Cs. Each roof was contaminated with 1.5 µCi Cs-134, applied in stripes across the roof, using a small spray bottle (50 - 100 ml solution on each roof). The "summer" roofs were completely dry at the time when the contamination was performed. The solution applied on the roofs dried up before run-off started. Contamination under winter conditions was carried out in the same manner, although extra care had to be taken to avoid penetration of the snowcover.

3.1.4 Measuring methods.

It was originally planned to measure the amount of radioactivity on the roofs with two different methods. It was planned to measure directly the activity on the roofs, using a lead-collimated 2"x2" sodiumiodide detector. And it was planned to measure the content of activity in the run-off water. It would then be assumed that the activity not retrieved in the run-off water was still on the roof.

It was soon found that the first method, although it gave satisfactory results for the flat tar-paper roof, did not work for the metal roof, which has a more complicated geometry. This method was accordingly abandoned.

For chemical analysis one liter of the collected run-off water from each period, with 40 mg CsOH added as carrier, and kept acid by addition of HNO3, was evaporated down to a volume of about 200 ml. Then the solution was made alkaline with sodiumhydroxide. The Cs-134 was precipitated using sodiumtetraphenylborate, and the precipitation was measured. In order to find out if any caesium was adsorbed in the container, the container was leached with a nitric acid solution containing caesium. In the leached solution we found ca. 5 nCi. This is about 0.5% of the activity which had been in the containers. A test with three parallels showed that the chemical recovery of the precipitation was close to 100%.

3.1.5 Results of the experiments.

All numerical results here are normalized to refer to inactive caesium. If it is wished to relate the results to specific caesium isotopes, the proper radioactive decay should be applied.

First about the experiment where contamination was applied on the dry roof material. The results show that more than 70% of the Cs-134 added runs off the ironplate roof during one normal rainfall; while on the tar-paper roof, which is covered with fine grains of shale, only 3% is removed during the same rainfall.

When the experiments were concluded, the tarpaper roof which was contaminated under dry weather conditions (in fall 1982), had spent 8 months outdoors. Part of this time it had been covered with ice and snow. The roof where contamination was performed on top of a wet layer of snow, spent only 3 months before the experiments were terminated. The precipitation upon this roof was only half of that on the "dry" roof, and there were only a few days of snow or ice. In spite of this, roughly the same fraction of activity (about 40%) had been removed from both roofs. This indicates that caesium applied to a snow- or ice-covered roof adheres less readily to the roof materials.

From the experiment performed in winter 1984, it was found that the first run-off water (14 liters) contained 16% of the applied activity. The comparative result for the "dry" roof was only 3%. After 15 days after initiation of the experiment there was no more snow or ice left on the roof. The total run-off was 45 liters, equivalent to 15.6 mm precipitation, and 47% of the added activity had run off. More snow fell later on, but the last snow of the season disappeared on 6th April. The experiment was continued for a total of 4 months, during which time a total of 411 liters of run-off water was collected. This is equivalent to 143 mm of precipitation. With this water a total of 65% of the added activity was recovered.

The results can be summed up as follows:

- Ironplate roof, dry application conditions:
  22% remained on the roof at end of experiment.

- Tar-paper roof, dry application conditions:
  61% remained on the roof at end of experiment.

- Tar-paper roof, application on wet snow:

59% remained on the roof at end of experiment.

- Tar-paper roof, application on a stable layer of ice and snow: 35% remained on the roof at end of experiment.

The results will depend not only on weather conditions, and on whether there is snow or ice on the roof. They will also depend upon the general conditions of the roofs. There is reason to believe that the degree of retention is increased if the roofs are dirty, cracked, or if lichen, moss or small plants grow on them, like often is the case with older roofs. This might be worthwhile of further investigation. Another aspect that ought to be examined, is the importance of the size of the mock-up roofs. It is evident that the water flow on a full-sized roof will be larger, and this might result in decreased retention.

This project was designed to investigate natural decontamination. Similar experiments may of course be performed for investigation of forced decontamination.

3.2 Winter conditions

When calculating the consequences of large reactor accidents, it is usually assumed that deposited radioactive materials on ground remain where deposited. And when assumed that decontamination is carried out, the decontamination factor usually used is based upon rather scanty experimental material, determined under conditions very different from the ones to which it is applied. This subproject addresses these two problems under winter conditions, and consists of a run-off part and a snow-removal part. The experiments are summarized in (Ref. QV84) and reported in full in (Ref. QV84b).

3.2.1 Run-off with melt-water in spring.

The run-off part of the subproject is concerned with natural run-off during spring of Cs-134 deposited upon a snow-covered field. Under these conditions one would expect run-off to be larger than under summer conditions, because the ice present may limit contact between the deposited Cs-134 and the soil.

The experiments were performed in a lycimeter kindly lent for this use by the Agricultural University of Norway. The lycimeter has been routinely used for run-off studies of acid precipitation. The lycimeter is 75 m^2 (3.75 m x 20 m), and is isolated from the surrounding soil on all sides and in the bottom. It contains clay soil, and the vegetaion is mainly various types of grass. It is gently sloping (1:20) toward one of the short sides, from which all run-off water is collected.

The Cs-134 was sprayed onto the surface when stable winter conditions had been achieved (14. February 1984). At this time the lycimeter was snow-covered, with 12 - 15 cm coarse-grained snow with a hard, undisturbed top layer of crusted snow. The ground was frozen. The highest air temperature on this day was -3^0C, and the temperatures on the preceding and following nights were below -10^0C.

All subsequent run-off water was collected until all ice, snow and frost in the ground had melted. This winter melting of the snow started on the 5. April (7 weeks after contamination), and lasted for 8 days, when all the snow was melted and the run-off stopped.

Altogether 7890 liters of run-off water was collected, and it was found that caesium run-off under these conditions was roughly 30%. In current accident consequence assessment it is usually assumed that there is no run-off of caesium. Comparable summer experiments have not been found.

3.2.2 Snow-removal experiment.

If a release of radioactive materials takes place during winter conditions, the deposition will usually take place on a snow- or ice-covered surface, and it is probably much simpler to remove the radioactive materials in this case, than when contamination takes place on vegetation, buildings and roads.

In this subproject we made a first attempt at determining just how effective decontamination may be, under winter conditions, using standard snowremoval equipment.

A small parking lot (30 m^2) was used for these experiments. On the asphalt was a layer of solid ice of thickness 1 to 7 cm, covered with ca. 5 cm of undisturbed snow. The snow was a mixture of fine- and coarse-grained.

Cuppersulphate was dissolved in water, and sprayed onto a 15 m x 2 m area. A portable mist spray unit (ordinarily used for spraying pesticide) was used. The reason for using Cu as contaminant is that the required chemical analysis is simple and cheap. The air temperature was -3^0C when contamination was carried out. The temperature in the snow was somewhat lower, as the air temperature had been down around -10^0C the preceding night.

Immediately after contamination, the loose snow was removed from the area, using a tractor with a shovel. The area was then subdivided into 1m x 1m squares. All remaining <u>loose</u> snow and ice was collected in each square separately, put in plastic bags, weighed and the copper content determined, using atomic absorption methods. Samples for background and standard were collected before contamination.

The amount of copper recovered was equivalent to 0.87% of the contaminant; which means that more than 99% of the contaminant has been removed. The decontamination achieved by this experiment was remarkably efficient. But the conditions were in many ways favorable. The smooth, hard ice covering the asphalt surface of the parking lot made decontamination relatively easy. Conditions will not always be as favorable in real situations.

The spray mist particles froze when they came into contact with the cold snow, and stayed on the surface of the snow. If deposition took place with rain, the deposited material would penetrate into the snow layer. But by sub-zero temperatures the materials would be exptected to stay on the snow surface, like they did in this experiment.

One experiment like this is of course not sufficient. Winter conditions vary quite a lot, and "Christmas card" conditions are relatively rare, even in the Nordic countries. Main roads are most of the time almost free of snow, due to frequent removal, mechanical or by spreading of salt. The proper authorities in Norway say that this is done on the main roads as soon as there is "a little snow". Otherwise it is performed when the snow cover is 5 to 10 cm. On smaller roads and on sidewalks there is often a layer of hard-packed snow and ice, that is less easily removed. Partial melting of the snow, due to rain or sufficiently prolonged above-zero temperature periods, will change the situation completely. Under such conditions the contaminant will rapidly penetrate the snow layer, and probably stick to the surface underneath; although it may also be removed with the melted snow. A range of conditions ought to be investigated. If a high decontamination efficiency is found under many typical winter conditions, this fact ought to be taken into account in future evaluations of reactor accident consequences and/or evaluations of the cost and efficiency of mitigating actions.

3.3 Deposition in urban areas

Dry deposition velocities are reasonably well-known for rural areas, while the deposition velocities upon the surfaces of houses and roads are not known to a sufficient degree, as there have been few attempt at determining these. In this subproject the amount of Cs-137 (from nuclear bomb tests) deposited upon urban surfaces has been measured, and gives the basis for determining the deposition velocity. The same type of experiments have also been performed using Be-7, which is contained in the atmosphere, produced by cosmic radiation in the upper layers of the atmosphere. Measurements on this nuclide, which has a much shorter half-life, may more directly be related to accident conditions. The report from this subproject is (Ref. RO85).

3.3.1 Deposition on rough and smooth surfaces.

Pollution in air can have different forms, as liquid drops, reactive and non-reactive gases, and small solid particles. The dispersed pollutant can be removed from the air by different processes. Removal in the absence of precipitation is normally called dry deposition.

A smooth surface is one in which the roughness elements are so small that they do not penetrate the sublaminar layer created at the surface. At rough surfaces the roughness elements are sufficiently large to be able to penetrate the sublaminar layer and this will cause a greater deposition than on smooth surfaces. (Ref. AH79, JO78 and UN83.)

Deposition in urban areas can be dealt with in two ways: By considering the total urban surface as a rough surface with the buildings forming the roughness elements, and from this model find the total deposition to the urban area; or by looking at the deposition processes in the canopy in detail, so that deposition on the separate roughness element are examined.

The first approach has some serious draw-backs: First, the distribution of the deposited material on the different vertical and horizontal surfaces is not found, and knowledge of this distribution is important in dose calculations. Secondly, when this sort of model is used the deposition velocities chosen are often those found from experiments in rural areas with a comparatively dense canopy, characterised by, e.g. friction velocity or roughness length. The use of such deposition velocities in urban areas can greatly overestimate the deposition here.

To describe the deposition in urban areas properly the second approach seems more suited. This type of model is developed for dense canopies by Thom (Ref. TH67). Unfortunately, Thom's model cannot be used for calculating deposition in open canopies such as urban areas.

For describing deposition in urban area canopies it seems necessary to examine the air flows over the surface of the separate roughness elements and from these to calculate the deposition on the different surfaces (roads, walls, roofs, etc.). Although urban areas, as such, must be characterised as a rough surface, many of the separate surfaces within them are generally fairly smooth. The model must be able to cope with these characteristics.

3.3.2 Experiments on dry deposition.

The dry deposition velocity has been investigated in several laboratories and field experiments on different surfaces such as grass, bare soil, and metal.

Nearly all experiments dealing with vertical surfaces have been dedicated to finding the deposition velocities on smooth vertical tube surfaces. (Ref. DA66, SE70, SE73, LI74, SL78, FR57). Only a few of the wind tunnel experiments have dealt with vertical surfaces and a literature study has revealed only one (Ref. RO83) field experiment with deposition on real urban surfaces.

Measurements of the overall deposition in urban areas are also very scarce and those found in the literature review all dealt with total deposition, dry as well as wet (e.g. Ref. AN78).

3.3.3 Dry deposition in reactor safety studies.

The problems of finding a convincing model for dry deposition in urban areas are reflected in the deposition velocities used in recent reactor safety studies.

The Rasmussen Report, WASH 1400, 1975, has used a constant deposition velocity of 1 cm/s independent of the weather situation, pollution form, and type of surface (rural or urban). Later American studies as NUREG/CR-2239 (1982) and the German Safety Study (Gesellschaft für Reactorsicherheit) have also used deposition velocities of 1 cm/s for rural as well as for urban areas. A Swedish study (Ref. RA79) has used a deposition velocity of 0.3 cm/s in both rural and urban areas, while in another Swedish study (Ref. SW83) the deposition velocities were chosen as 0.2 cm/s for particles, 0.005 cm/s for organic iodine and 0.5 cm/s for inorganic iodine. The latest British study (Ref. KE82), used deposition velocities of 0.1 cm/s for particles, 1 cm/s for inorganic iodine, and 0.001 cm/s for organic iodine for rough surfaces (roughness length 30 cm) independent of the density of the canopy, so that in urban as well as most rural areas these values were used.

Until now the only studies where separate deposition velocities for urban and for rural areas have been used, are the Danish ones, performed at Risø National Laboratory.

In the study of potential radioactive contamination of Danish territory following a major accident at the Barsebäck plant (Ref. GJ82), deposition velocities of 2 cm/s for rural and 0.2 cm/s for urban surfaces were used. It must be emphasized that the deposition velocity of 0.2 cm/s is used for all urban surfaces, horizontal, vertical, and sloping, so that the overall deposition velocity for the urban area will be about 0.3 cm/s.

In the study of the consequences of hypothetical large reactor accidents calculated on the basis of empirical data (Ref. GJ83) the values suggested by (Ref. RO82) were used: for rough rural areas 0.2 cm/s for particles, 1 cm/s for inorganic iodine; for urban surfaces 0.04 cm/s for particles, 0.2 cm/s for inorganic iodine. As mentioned above, the deposition velocities for urban surfaces are used on the real surfaces, so that the overall deposition for the urban area will be about 0.06 cm/s for particles and 0.3 cm/s for inorganic iodine.

3.3.4 Experimental part.

3.3.4.1 Caesium-137 measurements.

Some of the surfaces measured were actual vertical building surfaces (the Cs-137 measurements), while some were artificial surfaces mounted on buildings to collect Be-7.

An approximately 5 mm thick layer of the surface of removed bricks was sliced off, pulverised, and later analysed for its content of Cs-137. It was found from these measurements that the Cs-137 was confined to the outermost layer. The concentration in Danish air of nuclear weapons fall-out has been measured continually from 1960 at Risø National Laboratory (Ref. AA84), and this provides the basis for determining the deposition velocity averaged over a long time period. The deposition velocities are given in table 3.3.1 (Ref. RO85).

3.3.4.2 Beryllium-7 measurements.

Plates were mounted on some houses in order to measure conventiently the Be-7 deposited. Two types of plates were fabricated; the surface of one consisted of mortar based on beach sand to insure a low background of Be-7. The other surface was made of rough wallpaper painted with a normal house paint. The surface of each plate was protected against rain by mounting a shed roof over them.

Sample No.	When built (year)	Area of surface (m^2)	Deposition of Cs-137 (Bq/m^2) Sample facing					Deposition velocity (cm/s)
			East	South	West	North	Combined North and South	
Brick wall								
82	1900	0.213	7.86					0.005
83	1900	0.221				25.96		0.015
81	1910	0.188	5.72					0.003
70A	1910	0.464				38.99		0.022
70B	1910	0.534				16.61		0.009
70C	1910	0.158		74.53				0.042
70D	1910	0.341		52.59				0.030
71A	1920	0.094		3.85				0.002
71B	1920	0.231				6.9		0.004
79A	1920	0.181	6.23					0.004
95	1930	0.327				26.72		0.015
80	1956	0.266		56.07				0.036
72	1958	0.132				65.08		0.044
73	1958	0.133					77.00	0.052
Plaster Wall								
97	1900	0.120				58.81		0.034
90	1918	0.277				75.78		0.043
74	1920	0.153		36.71				0.021
75	1920	0.162				16.61		0.010
76	1920	0.115			26.39			0.015
79B	1920	0.061	8.35					0.005
96	1930	0.133	41.76					0.024
78	1940	0.226	20.63					0.011
85	1950	0.195			24.34			0.014
86	1950	0.294	6.75					0.004
87	1950	0.270				45.42		0.026

Table 3.3.1 Depositions on urban surfaces, Cs-137.

After having been exposed to the air and its content of Be-7 particles for some months, the plates were brought to the laboratory and the surface was scraped off and measured for its content of Be-7. In the period of exposure air samples were collected weekly. The deposition velocity for Be-7 was determined, by integrating the air concentration over the time in which the plates were exposed, correcting for decay (Be-7 half-life is 53 days), and determining the total deposition per unit of area on the plates. The determined deposition velocities are shown in table 3.3.2 (Ref. R085).

3.3.5 Summary of results from this subproject.

This subproject was mainly concerned with dry deposition, but some conclusions about wet deposition could also be drawn.

For the samples taken from walls that are well protected from rain, snow and hail the deposited material is due mainly to dry deposition. Samples nos. 71A, 71B, 79A, 81 and 82 were taken from fairly well-protected surfaces, and the calculated deposition velocities are low (0.002-0.006 cm/s). The rest of the samples were taken from houses with small eaves or none.

For the deposition velocities for Cs-137, a major potential source of error is that weathering effects and the effect of cleaning buildings, removel of paint, etc. will eliminate a portion of the deposited caesium. Concerning the cleaning of walls and removal of paint the effect on the deposited caesium is probably small because there is no tradition in Denmark for regularly cleaning outer walls. The likelihood that paint, etc. will be removed is greater for plastered houses; therefore the samples from these surfaces were taken from areas of the walls where the old paint did not seem to have been removed. It could also be concluded from comparing the Be-7 and the Cs-137 measurements that the weathering effect is fairly small.

Two horizontal surfaces have been investigated. Sample No. 107 has been undisturbed, but the other (No. 104) showed clear signs of footprints from children and cats. It is accordingly probable that some of the measured Cs-137 has come from soil contaminated with wet deposited Cs-137. This sample is therefore not representative.

The values of the deposition velocity of Cs-137 obtained in this subproject are in the range from 0.005 to 0.044 cm/s. Some of the samples taken can also have been exposed to some wet deposition. For these samples the determined values are not dry deposition velocities, but rather wet and dry deposition added, and at least it can be concluded that the dry deposition alone can not be larger than the value determined. It may be argued that the weathering effect might invalidate this conclusion, but as already mentioned, it can be concluded from the measurements that the weathering effect is relatively unimportant.

Some of the Cs-137 samples were taken from surfaces well protected against rain, and all of the Be-7 samples were taken from such surfaces. The deposition velocities determined from these surfaces are in the same range, whether Be-7 or Cs-137 is measured.

Sample No.	Surface material	Area of sample (m^2)	Sample exposed from	to	Surface towards	Deposition (Bq/m^2) Vertical	Horizontal	Deposition velocity (cm/s)
89	Plaster	2.98	82.11.16 -	83.03.28	NE	0.635		0.0066
92	Plaster	2.98	83.03.28 -	83.11.24	NE	0.611		0.0049
102	Paint	4.30	83.11.24 -	84.02.09	NE	1.86		0.0158
103	Paint	4.30	83.08.15 -	84.03.26	S	0.193		0.0013
104	Plaster	2.98	83.04.24 -	84.04.09			9.21	0.066
105	Plaster	2.98	83.04.24 -	84.04.09	N	0.305		0.0022
106	Plaster	2.98	83.01.10 -	84.06.06	N	0.212		0.0013
107	Plaster	2.98	83.10.01 -	84.06.06			0.458	0.0028

Table 3.3.2 Depositions on urban surfaces, Be-7.

For the horizontal surface, that seems to have been undisturbed, the deposition velocity determined is in the same range as for vertical surfaces.

The deposition velocities found are very small. This indicates that the deposition velocities in urban areas are considerably smaller than those in rural areas.

3.4 The filter effect of buildings

In consequence calculations as they are carried out at present, it is usually assumed that the inhalation dose indoors is equal to the one outdoors. Experiments carried out in the Nordic countries show that this is not the case. A certain amount of the radioactive materials will be removed from the contaminated air as the air passes through narrow cracks to enter a house in which all doors, windows and ventilation channels are closed. The experiments were carried out by measuring the amount of natural Be-7 inside a house and outside the same house. As building traditions differ substantially between the four Nordic countries, the same type of measurements have been carried out first in Denmark (independent of the project described here), and then in Finland and Norway. The results of the Finnish and Norwegian experiments are reported in (Ref. CH83).

Measurements were carried out in Finland by the Institute of Radiation Protection during the summer of 1982, and at the Institute for Energy Technology in Norway during late fall 1982, and again during early summer 1984.

The experiments were carried out by collecting the Be-7 (fixed on dust particles) on a filter, using a pump to suck the air through the filter. Samples were collected in the house and outside of the house simultaneously. The results of the 1982 experiments in Norway were somewhat unsatisfactory. The reduction in indoor air concentration was markedly less than in the Danish and Finnish experiment, in spite of the fact that the air exchange rate of the Norwegian house was found to be lower. The collection time was 72 hours for each sampling. But over this time period the filters clogged up. The technicians decided to change the filter twice during the sampling time, but they had to enter the house to perform this operation. It is possible that enough outside air has entered the house in this manner, to disturb the measurements. In the 1984 experiments the house was left undisturbed. Deposition velocity indoors was also measured, and found to be quite small (0.008 cm/sec).

The Finnish study includes four apartments in typical Finnish apartment buildings build in 1968 and 1972. The filter factors determined in the Finnish experiments range from 0.23 to 0.45.

The Norwegian study includes one wooden villa built in 1954. Four sets of measurements were performed for this house, and the filter factors determined range from 0.40 to 0.47.

The data material is yet too small to be of real use in accident consequence assessments. More house types have to be investigated.

4. ACCIDENT CONSEQUENCE ASSESSMENT MODELS

The three subprojects described in this chapter were carried out in sequence, and are closely related to each other. There exists, in the Nordic countries and elsewhere, a large number of different models for accident consequence assessment. Although they address more or less the same problem, and to a large extent use the same methods (e.g. atmosphic dispersion model), there are important differences.

On one side it is important to have detailed knowledge of these differences, so that it will be easier to explain differences in calculated results between different assessment models.

Another aspect is that it is important to be aware of the limitations and simplifications in an assessment model, to be able to judge and utilize the results correctly.

In this project these questions have been approached in the following three steps; each of them in form of a subproject:

- What would one ideally want a consequence assessment model to be able to do, and how?
- How do the models available measure up to these wishes?
- Is it possible, with relatively limited resources, to modify an existing model so that it comes significantly closer to the ideal model?

4.1 The "dream" model

This subproject consisted of a generic analysis of reactor accident consequence assessment models. The intention was to describe the "ideal" model, and at the same time describe what capabilities are built into the more widely used existing models. The resulting report (Ref. TH82) contains much useful information on consequence calculation, systematically arranged.

4.1.1 Model development since WASH-1400.

Publication of the American Reactor Safety Study (WASH-1400) in October 1975 (Ref. NU75) introduced a new, and more realistic, way to assess the consequences of reactor accidents. The new ideas were conained in the model CRAC, also described in (Ref. NU75).

The development of accident consequence assessment models in the years to follow was strongly influenced by these ideas, and CRAC itself was also developed and improved, and exists now in a number of versions (e.g. CRACIT, CRAC2, NUCRAC). A new accident consequence assessment code is now being developed at Sandia National Laboratory, as part of the MELCOR program. Although it is closely related to CRAC2, this is a completely new code, not a modified version of CRAC2.

A number of reactor safety studies, carried out by other countries, followed the publication of WASH-1400. The most ambitious study was the German Reactor Safety Study (Ref. DE79). But the first study to be published after WASH-1400 was a study conducted for the Norwegian Government Commission on Nuclear Power (Ref. TV79). The calculations were performed using CRAC, which was kindly made available by the Nuclear Regularoty Commission.

At the time when WASH-1400 was published Denmark, Finland and Sweden had advanced so far in development of their own calculation models, that it was chosen to continue this development.

4.1.2 Content of the report from the subproject.

It is not meaningful to condence or summarize the report from this subproject, so it is chosen instead to summarize the table of contents, and to enlarge upon some few chosen topics.

Release specifications.

Dispersion in the environment. Atmospheric, run-off, infiltration, dispersion in water.

Exposure pathways. General about modelling. Acute (external-cloud, inhalation, external-ground), chronic (external-ground, fishing, swimming, aquatic food, agricultural products, drinking water, irrigation). Environmental description (population, area segmentation, protection factors, terrestric and marine transfer factors).

Health effects.

Mitigating actions (dose criteria, evacuation, iodine tablets, relocation, decontamination, deep plowing, condemnation of agricultural and aquatic foods, alternative agricultural production, restrictions on water usage, protection provided by houses).

Data needed. See chapter 4.1.4.

Desired results from calculations.

Guidelines for modelling. See chapter 4.1.5.

4.1.3 General about exposure pathway modelling.

The equations or relationships describing the physical and chemical processes involved in the various exposure pathways may be very complicated. In consequence models, and especially if the consequence model is to be used in risk calculations, equations of a much simpler type must be used. A fundamental equation in extensive use in this connection is

$$\dot{H} = C \cdot U \cdot D \cdot P$$

where $\dot{H}$ is the dose rate

C is the concentration (in air, water, milk, vegetables, etc.)

U is reffered to as either the usage factor or the concentration factor

D is the dose conversion factor

P is the protection factor

The equation is simple, and accordingly convenient for use in calculations in the inner loops of large computer programs; calculations that are to be repeated perhaps thousands of times. In the form above it is also quite general, and can basically be used in this form for all exposure pathways. (In most of the following, whenever "dose" is mentioned, "dose commitment" is equally appropriate.) At the same time it is so general that the shape of the equation says nothing about the processes involved, and all knowledge of the specific processes must be included in calculations performed to determine the parameters C, U, D and P. Some or all of these parameters must be determined separately for all exposure pathways, isotopes, population groups and body organs.

It is difficult to grasp the meaning of this, without discussing a specific, concrete problem. Consider the pathway (release to atmosphere) -(deposition on ground)-(runoff)-(drinking water)-(man). The equation may be reformulated as

$$\dot{H}_{irg} = C_i \cdot U_g \cdot D_{ir} \cdot P_g$$

where $\dot{H}_{irg}$ is the dose rate to organ r from isotope i in population group g

C_i is the concentration of isotope i in the drinking water reservoir concerned

U_g is intake per unit time (per day) of drinking water from the reservoir concerned by a person in population group g

D_{ir} is the dose conversion factor for isotope i and organ r

P_g the protection factor would not be needed in this case, and would be set equal to one.

U may mean very different things depending upon the exposure pathway, but also depending upon the way in which the particular pathway is analyzed. It is often a composite parameter. For the (grass)-(cow)-(milk)-(man) pathway it may incorporate as different phenomena as wash-off by rain and the cows metabolism. In some cases it is possible and convenient to define the problem in such a way that U is the same for all isotopes. U may for instance be the daily amount of milk or other agricultural products consumed, or it may be the breathing rate. In all cases U should link concentration in some medium with intake per time unit by a person, so that U . C is the intake of a certain isotope by a person via a certain exposure pathway, and the units of U must correspond to the units of C. U may differ from one population group to another.

P is not relevant to all exposure pathways, and when it is not, it is set equal to one. P is meant to correct for certain natural or artificial protecting phenomena, but is occasionally used to correct for simplifications in the other parameters. P is most commonly used as a shielding factor to account for the shielding against external irradiation provided by building structures, or to correct for the rough way in which external irradiation from deposited activity on ground is usually calculated. It might however also be used to account for the filtering effect of buildings in connection with calculation of in-door inhalation doses. Or it might account for the fraction of time a contaminated area is occupied if limited restrictions are imposed. The units of P may vary with the way in which the pathway is examined, but it is often without a unit, expressed as a fraction. It will often differ from one population group to another. It will also usually differ from isotope to isotope; a fact which is still usually neglected, and "average" values are used instead.

4.1.4 Data needed for a complete consequence analysis.

The data needs of a "total" consequence model are immense, when both data entering directly into the consequence model and data used to calculate data entering into the model, are considered. The data also cover many different fields, from meteorology to soil chemistry to dispersion in oceans; from agriculture to medicine, and much more.

A universally valid list of data needed can not be prepared. The needs will vary strongly with purpose of the calculation and sophistication of the model.

Although the present report is meant to be relatively brief, it is felt that a list of the type of data used by a typical accident consequence assessment model, will give valuable insight into the capabilities and limitations of such models.

<u>Release</u>
Release scenario, which means how much of each isotope, when after shut-down of reactor, for how long, and to what medium and where in this medium the activity is released.

<u>Atmospheric dispersion</u>
Dispersion data.
Data on weather or weather statistics, incl. precipitation data.
Deposition data (wet and dry), incl. data on physical/chemical form of release.
Data on run-off.

<u>Transfer from ground to water</u>
Data on infiltration.

<u>Dispersion in water</u>
Dispersion data, usually as transfer coeff. in a compartment model.
Sedimentation data.

Passing cloud
Data for transfer of air concentration into external dose from cloud.
Data for transfer of air concentration into inhalation dose from cloud.

Materials on ground
Data for transfer of ground concentration into external dose from ground.

Aquatic food
Data for transfer from water concentration to concentration in aquatic food.
Data for transfer from concentration in aquatic food to dose to man.
Production data.
Consumption data.

Terrestrial food
Data for transfer from ground concentration to concentration in agricultural products.
Data for transfer from concentration in agricultural products to dose to man.
Production data.
Consumption data.

Drinking water
Data for determining concentration in drinking water.
Data for transfer from concentration in water to dose to man.
Consumption data.
Data on sedimentation.

Irrigation
Data upon irrigation as function of time of season.

Health effects
Data for relation between dose to organ and health effect.

Evacuation
Evacuation data.

Iodide tablets
Protection factor or protection factors to be applied upon inhalation from cloud.

Relocation and decontamination
Relocation dose criteria and data.
Population distribution, possibly subdivided into groups like habitation, work, hospital etc.
Decontamination data.
Data on use of agricultural areas, as a geographical distribution.

Buildings
Data relevant to radiation shielding.
Data relevant to the filter effect of houses as reduction mechanism on inhalation dose.

4.1.5 Guidelines for modelling.

A developed consequence model should not be regarded as a finished product. It will probably be desirable to change or exchange various parts of the model in the future, as new information becomes available or different approaches are chosen. This requires a well organized program. This may be obtained if the theories of structured programming are observed.

To make changes or exchanges possible, computer programming ought to be linear. This means that each small part of the program should perform one operation only, and that the program when executing should pass through these parts in sequence. A program designed in this way will not be of minimal size. Programmed in a more "inter-woven" manner it could be both smaller and faster, but it gets difficult to modify. A change in one function of an "inter-woven" part, might influence another function of the same part, with the most unexpected results. It also gets increasingly difficult for an outsider to understand the way the separate parts function, as the programming gets more "clever".

Two simple ways to make it easier for outsiders to understand the inside of a program are the following. They <u>should be used</u> when new programs are developed:

- documentation
- use variable names that give a clue to the physical identity of the variable, like e.g. ACONC, GCONC, DEPV, for air concentration, ground concentration and deposition velocity
- use <u>plenty</u> of comment cards in the program, and fill them with information. They may give invaluable information, but are excluded from the compiled version of the program, so that no size or running time increases result from abundant use of comment cards.

These rules hold true in general when computer programming is concerned. But they are especially important in a model that will have many different users, and when the desired content in each part of the model (e.g. atmospheric dispersion, evacuation, health effect calculation) may differ from user to user.

Another self-evident requirement, that is however often sinned against, is complete and up-dated documentation. It is regretfully rather the rule than the exception, that programs are changed considerably, while the corresponding modification of reports and/or manuals is neglected.

<u>4.2 Survey of calculation models</u>

Worldwide there are a number of accident consequence assessment programs available. Many of these are designed to solve what seems like exactly the same problems; but closer inspection shows that there are important differences. A complete survey of consequence calculation programs in use in the Nordic countries has been carried out. The exact problems handled by each program have been identified, and differences between otherwise similar programs have been identified.

The subproject was carried out at Institute of Energy Technology, generously helped by the other Nordic experts in this field, who supplied the required information, and reviewed the draft, which later became the report (Ref. TV84) from this subproject. A Nordic seminar on models was also held, in Norway, as part of this effort.

A total of 19 models are included in the survey. All but one have been developed in the Nordic countries. It must be concluded from the survey that the Nordic models have a high level of quality, and that they cover a wide field of applications, from complete risk calculations to calculations of hydrodynamics.

4.2.1 Models included in the survey.

This list was complete when it was written (Summer 1983). No other computer programs were used or under development then in any Nordic country, for reactor accident consequence assessments.

ARANO (VTT, Finland) (Ref. NR80 and KA84)
Atmospheric release. Food pathways. No saltwater or freshwater pathways. Can calculate risk curves.

AQUAPATH-BALTIC (Studsvik, Sweden) (Ref. EV82)
Aquatic release. Compartment model. Does not calculate human intake or doses. Biota not included.

BIOPATH (Studsvik, Sweden) (Ref. BE83)
Source term is deposited activity or release to saltwater or freshwater. Compartment model. Calculates doses and health effects. Can cover all pathways. Flexible. Commercial.

CRAC (IFE, Norway) (Ref. NU75 or RI84)
The only model discussed not developed in the Nordic countries. Atmospheric release. Food pathways. No saltwater or freshwater pathways. Can calculate risk curves.

DETRA (VTT, Finland) (Ref. KO84)
Source term is deposited activity or release to saltwater or freshwater. Compartment model. Calculates doses and health effects. Can cover all pathways. Flexible. Under development.

GEOPATH (Studsvik, Sweden) (Ref. MA80)
Calculates transport with ground water. Does not calculate doses or health effects.

ISAK (FOA, Sweden) (Ref. FI83)
Atmospheric release. Food, saltwater and freshwater pathways are included in compartment model form. Can calculate Gaussian straight-line, Gaussian puff, or a Gaussian timefluctuating plume. Composed of a large number of separate programs, that are integrated into a program system. It is designed for education and training of personnel responsible for initiating mitigating actions after a nuclear power plant accident.

KOMPIS (FOA, Sweden) (Ref. FI83)
Source term is deposited activity or release to saltwater or freshwater. Basically a compartment model, but more useroriented than what is usual in compartment models. Originally developed as a module under ISAK. Under development.

PRISM (Studsvik, Sweden) (Ref. GA83)
Is used in conjunction with BIOPATH to calculate uncertainty, using Monte Carlo methods. Is not by itself a consequence calculation model.

PLUCON4 (Risø, Denmark) (Ref. TH80)
Atmospheric release. No food, saltwater or freshwater pathways. Calculates doses and health effects. Can not calculate risk curves, except by combining results from many computer runs.

PUFFCON-2 (Risø, Denmark) (Ref. MI83)
Atmospheric release. Puff model. Under development.

REG-2000 (Studsvik, Sweden) (Ref. ED82)
Atmospheric release. No food, saltwater or freshwater pathways. Calculates concentrations in air and deposited upon ground. Trajectory model. Is used only for distances more than 50 km from release point. Is operated in conjunction with UNIDOSE, which performs the calculations out to 50 km.

SWEDISH AQUATIC REGIONAL MODEL (Studsvik, Sweden) (Ref. SU82)
Aquatic release. Compartment model. No biota. Calculates doses, but not health effects.

SWEDISH AQUATIC TRAJECTORY MODEL (Studsvik, Sweden) (Ref. SU82a)
Trajectory, but otherwise like the regional.

SWEDISH IODINE MODEL (FOA, Sweden) (Ref. NI82)
Atmospheric release. Iodine only. Specifically developed for calculation of exposure via the milk pathway. Uncommonly high degree of realism in input parameters incorporated.

TERRA (Studsvik, Sweden) (Ref. GY84)
A modified version of BIOPATH. Under development.

TRADOS (FMI and VTT, Finland) (Ref. NR83)
Atmospheric release. Consists of two parts:
TRADOS-TRS, which calculates the trajectories.
TRADOS-DD, which simulates dispersion and deposition, and calculates doses. Food paths under development. No fresh- or saltwater pathways.

TRADOS-DD (VTT, Finland) (Ref. PA83)
Atmospheric release. Simulates dispersion and deposition, and calculates doses.

UNIDOSE (Studsvik, Sweden) (Ref. KA79)
Atmospheric release. No food, saltwater or freshwater pathways. Calculates doses and health effects. Main output in graphical form, such as risk curves, doses and health effect versus distance etc. It is planned to incorporate food pathways. Large efforts have been made on the efficiency of the code, resulting in low computer cost.

4.2.2 Aspects considered.

The information was collected in the following steps:

- Questionnaire, answered by all persons in the Nordic countries involved in the consequence assessment field.
- Additional information collected by telephone or letter.
- Information double-checked and complemented at seminar.

The sort of aspects considered, are best explained by giving some examples:

The release characteristics are the amounts of different nuclides, time after shut-down when release starts, duration of release, heat content, etc. But it is also important in this connection to know if the model in question makes use of nuclide groupings, and if the model is able to take differences in chemical and physical properties into account, when a nuclide can be released in different forms. It is found that the only manner in which differences in chemical and physical properties can be taken into account, is in the values chosen for the wash-out coefficient and dry deposition velocity. Changes in chemical and physical properties during atmospheric transport can only be taken into account by one of the models, because in all the other models deposition velocity and wash-out coefficient for one element have one fixed value each throughout the whole calculation.

The way in which atmospheric dispersion is calculated in the Nordic models, is basically the same for almost all the models. Even worldwide this model is the one in most frequent use at the present time, and it is referred to as Gaussian. A few of the Nordic models, however, are trajectory models, and one is a puff model. Mostly these models are used to assess types of situations different from the ones usually calculated with the Gaussian models.

The atmospheric dispersion part of "Gaussian" Nordic models differ in three respects:

- Calculation of dry deposition.
- Calculation of plume rise.
- Representation of cross-wind concentration distribution.

Nutrition pathways is an aspect in which the Nordic models differ considerably. Some of the models do not contain the nutrition pathways, and among the models that contain nutrition pathways, there are large differences in the level of detail; the number of different agricultural products treated separately, and the sophistication level of the calculation model.

The treatment of mitigating actions (evacuation, iodine tablets, relocation, decontamination, interdiction of agricultural products, drinking water, irrigation, aquatic food products) also differs considerably among the models, and some types of mitigating actions can not be taken into account by some of the models. Probably all the models are too rigid in their treatment of mitigating actions.

It was also found that documentation of the Nordic models generally was weak (with a few exceptions), and that it was accordingly difficult to obtain dependable detailed information about the models.

4.3 Building of an improved consequence model

The aim of this subproject is a consequence analysis tool, in which as much as possible of the results from the other subprojects is incorporated. Three specific ideas for improvements, otherwise not related to the other subprojects, had also been put foreward, and two of these have been carried out.

Among the results from other subprojects it was hoped to incorporate were shielding factors for houses, filter factors of houses, data on natural decontamination, on winter conditions, and on deposition in urban areas. However, the data base in many of these cases needs to be broadened before a general application is possible.

The additional ideas for improvement were concerned with improvement of the methods for calculating exposure via the nutrition pathways and with improvement of the calculation of atmospheric dispersion. The computer program CRAC2 (Ref. RI84) has been used as a basis. This model was chosen, because of good documentation and wide-spread use. The latter aspect gives a certain quality assurance.

The nutrition pathway model in the original version of CRAC2 is rather simplified. To improve this aspect a special version of the Finnish subroutine AGRID was prepared, and can now be inserted in CRAC2, as an alternative nutrition model (Ref. RO85a). The main improvement is that AGRID treats more nutrition pathways separately. CRAC2 in the original version has only two pathways; milk and other agricultural products. AGRID has five nutrition pathways; milk, meat, green vegetables, grain products and root vegetables. Accordingly a more realistic description of the local agricultural conditions is possible.

It was judged to be quite an improvement if the input data to the nutrition pathway model could be in the form of "everyday" quantities, like the area grazed by a cow in one day, or the time delay from the milking of the cow to the drinking of the milk. It is evident that only to a certain extent is it possible to replace transfer factors with such quantities, but in some cases such an option may facilitate variation of the conditions considerably. It was planned to modify a compartment model of nutrition parameters, developed at the Swedish Defence Research Establishment, to provide input data to the nutrition model in the accident consequence assessment model. For various reasons, however, this work has not been carried out.

It is often necessary to take a closer look at certain weather conditions (usually the ones giving the highest consequences), since they may be treated in a unrealistically conservative manner. The atmospheric dispersion calculations built into an accident consequence assessment model are simplified, because they are required to be fast on the computer. These simplified methods are satisfactory for calculating the total risk. But if it is desired to examine specific weather conditions, or to evaluate the upper portion of a risk curve (the maximum consequences), it is highly desirable to employ more realistic methods for calculating atmospheric dispersion.

A more sophisticated atmospheric calculation model has now been built into CRAC2, and can be used as an option. In this manner a more realistic atmospheric dispersion calculation can be carried out when chosen, while the remainder of the accident consequence assessment is still carried out in the same manner as when the sophisticated model is not employed. The more realistic model incorporated is a version of a multi-puff model developed at Risø National Laboratory (Ref. TH85). It is expected that the puff model results will be more realistic than the ones calculated by the Gaussian model (which still will be used in most calculations), even if only a few of the advantages of the puff model can be incorporated at the present stage.

The new version of CRAC2, resulting from the efforts described above, is referred to as NOCRAC (NOrdic CRAC).

5. MITIGATING ACTIONS

One of the subprojects was concerned with various aspects of mitigating actions, and related problem areas. It consists of three parts. The main bulk of the work was concerned with current specifications and utilizations of dose criteria, "action levels" etc. The main purpose of the subproject was to add to the basis for discussions among the Nordic Authorities on these subjects.

As part of this subproject a miniseminar was held in Finland, at which representatives of the Nordic radiation protection institutes were also present.

The report from the subproject (Ref. BL82) has chapters on ICRP, IAEA, Federal Republic of Germany, United Kingdom, United States of America, the Nordic Countries, and also Denmark, Finland, Norway and Sweden separately.

It also has chapters on

Protective measures in a potential or acute accident situation.
Preliminary mitigating actions related to contamination of foodstuff.
Normalizing actions.
Adaption of optimization and cost-benefit analyses.
Dose criteria for emergency personnel.

There is also a report from this subproject which is concerned with the relative importance of the body organs (Ref. SA82). Reactor accident consequence calculations have been performed for two release magnitudes, corresponding to BWR1 and BWR2 in WASH-1400. Acute and long-term doses to a number of body organs have been calculated using the Finnish model ARANO.

A third report (Ref. TV82a) from this subproject reports a sensitivity analysis, performed in order to examine the close connection between dose criteria, cost and health consequences. There is a close connection between the actual value of an action level and economic consequences on one hand, and between action level and health consequences on the other hand. To put it simply, a more severe action level costs additional money, but may save more lives.

The expression "may save more lives" is meant literally. Money spent on mitigating actions is not automatically effective in saving lives. If the situation is not analyzed carefully, money may be spent in reducing exposure via exposure pathways that are not the dominating ones. The best way to analyze such a situation is probably by using a consequence model as a tool in a sensitivity analysis. The model must include all exposure pathways. In this subproject the computer program CRAC, described in the WASH-1400 report, was used.

Two different release magnitudes (BWR1 and BWR4) are assumed. The aspects that are varied are

Ground dose criterion.
Milk interdiction criterion.
Criterion for interdiction of other agricultural products.

For each parameter value the cost of the mitigating action and the health consequence (fatal cancers) are calculated. It is then easy to compare increased cost and saved lives. And it is easy to identify the most efficient manner in which to spend a certain amount of money on mitigating actions. The next step, which would be a cost-benefit analysis, has not been attempted in this subproject.

The subproject was carried out more in order to demonstrate the usefulness of an approach, than in order to analyze a certain situation. One of the limitations is that only one weather condition is considered. Another, and more severe, limitation is that the economic frame of the subproject did not permit proper evaluation of the cost of the various types of mitigating actions. Even though the results of the analysis are believed to be qualitatively valid, this is just a first step in the right direction.

6. CONCLUSIONS

The project has shown that there is ample room for improvement of methods which are to-day available for the assessment of accident consequences. This has been done partly by improving the data to be used in the calculations, and partly by improving the models themselves.

Four data-related questions were dealt with:

- Terrestrial transfer factors.
- The freshwater pathways.
- Comparison of dynamic and static calculation models for fish.
- The shielding effect of buildings.

The work on terrestrial transfer factors has resulted in generation of a Nordic fall-out data bank, and demonstration of how the data could be utilized in calculations of exposure via the nutrition pathways. The present status is that the data collection has already been utilized in Denmark. In the other three countries the need for data from the bank has not been felt so strongly, since data of this type is already in use. The data bank, however, provides a welcome opportunity for increasing the confidence in the results obtained.

In the work on freshwater pathways old and new information has been collected in a systematic setup. The results provide support to the manner in which these pathways are treated as present. They have also cleared some uncertainty as to the possible importance of irrigation with contaminated water. It turns out that in severe accident situations, this is not an important exposure pathway.

The work where static and dynamic calculations for fish are compared is relevant for less severe accident situations, where a significant portion of the release is directly to the sea. The result indicates that it is not neccessary to go to the extra expence of a dynamic calculation, but this conclusion may be valid only for situations similar to the one analyzed.

Typical shielding factors have been calculated for various types of houses in the Nordic countries. Because of differences in building traditions existing factors from one country must be used with care when applied to conditions in another country. Differences between house types at any site must also be taken into account. Additional shielding factors for other house types can easily be obtained in the manner demonstrated in the project.

Four experimental investigations were performed:

- Natural decontamination of roofs.
- Winter conditions.
- Deposition in urban areas.
- The filter effect of buildings.

In the work concerned with roofs it was shown that natural decontamination varies between a factor of somewhat less than two to more than five. The difference depends primarily upon the characteristics of the roof material itself. The weather characteristics are also important, but to a smaller extent.

Accident consequence assessments under winter conditions differ significantly from summer conditions. The health and economic consequences of an accident will be smaller under winter conditions. There are many different reasons for this, two of which have been examined. Firstly it was shown that caesium retention in agricultural areas is reduced when deposition takes place during winter conditions, though only by a factor less than two. The factor will probably vary with type of surface and the steepness of the area. Secondly decontamination of urban areas under winter conditions is shown to potentially be much more efficient than under summer conditions, and to cost a fraction of what it would cost under summer conditions.

Investigation of deposition in urban areas shows that the dry deposition velocities presently used are much too high. Efforts can start immediately to incorporate this information in a realistic manner in future accident consequence assessments.

The filter effect of buildings will typically reduce the inhalation dose by a factor of somewhat more than two to more than four. After a severe accident the total reduction in dose will be less, as the inhalation dose in present assessments is not found to be dominating. There are additional questions that need to be examined. One is deposition indoors. Another is the external exposure from the materials that have been filtered, and accordingly will remain in the walls. The third is possible resuspension of these materials.

The three subprojects on accident consequence assessment models in the Nordic countries were carried out in sequence, and were closely interrelated. First an investigation was carried out on the present status of accident consequence assessment models, and how well these models correspond with the needs. Secondly a survey of accident consequence assessment models in use in the Nordic countries was carried out. Thirdly several improvements were incorporated in one of these models. The treatment of the nutrition pathways was improved, and a more sophisticated atmospheric dispersion model was built in as an option. It was hoped also to incorporate much of the knowledge gained throughout the whole project, but in many cases a broader basis is needed before this can be achieved (e.g. shielding effect of houses, where additional types of houses need to be examined).

Various aspects of mitigating actions, and related problem areas was examined. This work consists of three parts. The main effort was on current specifications and utilizations of dose criteria, "action levels" etc. The main purpose has been to add to the basis for discussions among the Nordic authorities on these subjects. Extensive discussions were held at a miniseminar, which was held as part of the project. Conclusions have not yet been drawn, but further discussions of this type may now be held. The second part consisted of an analysis of the relative importance of the body organs, seen in relation to dose criteria. The third part included a sensitivity analysis, performed in order to examine the close connection between dose criteria, economic and health consequences, when using an accident consequence assessment model. This part indicated the relative importance of dose criteria on different exposure pathways, and can be illustrated by one example: There are situations where a strickter milk criterion gives little reduction in health consequences, although it may be quite costly to enforce this criterion. If the same economic resources were spent on restrictions of other agricultural products one might obtain much higher benefits in the form of reduced health consequences.

The project as a whole covered a very wide field of subjects on a quite limited budget. It has also contributed to giving the Nordic groups employed in this field a reputation for inventiveness and determination.

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