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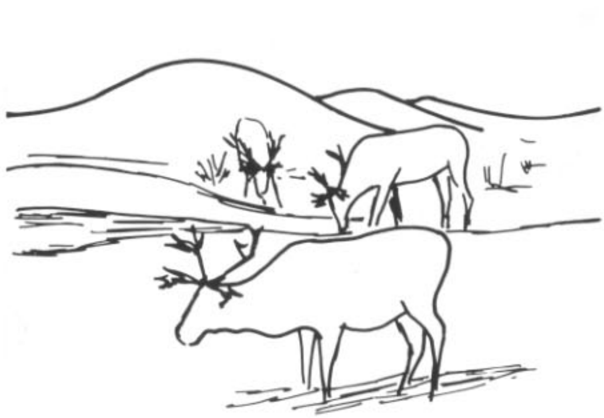
ENVIRONMENTAL CONSEQUENCES OF RELEASES FROM NUCLEAR ACCIDENTS

A NORDIC PERSPECTIVE



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ENVIRONMENTAL CONSEQUENCES OF RELEASES FROM NUCLEAR ACCIDENTS

A NORDIC PERSPECTIVE

Final Report of the NKA Project AKTU-200

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ABSTRACT

The primary purpose of this report is to present the results of a four-year Nordic cooperation program in the area of consequence assessment of nuclear accidents with large releases to the environment. This program was completed in 1989. Related information from other research programs has also been described, so that many chapters of the report reflect the current status in the respective areas, in addition to containing the results of the Nordic program.

Key Words: Chernobylsk-4 reactor; contamination; coordinated research programs; Denmark; deposition; earth atmosphere; environmental impacts; Finland; fission product release; food chains; mitigation; Norway; particle resuspension; radionuclide migration; reactor accidents; remedial action; risk assessment; sensitivity analysis; socio-economic factors; Sweden; weathering

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SUMMARY

An extensive effort is taking place internationally to enlarge our knowledge of environmental consequences of nuclear accidents.

Nordic work in this area has been directed to improve methods for estimation of such environmental consequences, so that greater realism is attained. After Chernobyl, the area of interest was expanded to also include remedial measures in accident conditions and analysis of the experience with mitigating actions.

The area of interest in accident consequence assessment is broad, spanning from meteorology to local agricultural practices. Models for atmospheric dispersion are used to calculate activity concentrations in air and concentrations of deposited activity on the ground. Other information is used to evaluate behaviour in different environments; urban, agricultural, forest or mountain areas. There are also models to calculate the transfer of radioactive materials to man via food and drinking water.

A realistic calculation of radiation exposure to man resulting from a nuclear accident would only be possible if the whole environment could be acurately described and modelled. Present knowledge of meteorology, ecology and numerous other disciplines is far from being at this stage, and such a goal may never be reached. Research efforts are aimed at providing reasonable methods including simplifications and short-cuts that should represent real nature as closely as possible.

Several attempts at quantifying the possible errors in radiation exposure calculations have been made over the last 10 - 15 years. The magnitude of errors depend, of course, upon the calculation methods chosen and the quality of the input data available. In the Nordic program possible errors caused by uncertainties in the numerous parameters that enter the calculations were examined. It turns out that possible errors are quite large, indicating that our knowledge of the manner in which radioactive materials behave in nature is not sufficient for an accurate assessment.

In the following, essential findings of the Nordic AKTU program, performed from 1985 to 1989, are highlighted:

Models describing atmospheric dispersion over medium-range distances have traditionally been based on relatively simple concepts. An advanced model, a so-called multi-puff model has been developed, where weather variation with time can be taken into account in a more realistic manner. The model was verified against experiments carried out across the Øresund sound between Denmark and Sweden. This model is now used in Germany as well as in the UK.

Radioactive materials in an atmospheric plume are mostly fixed on particles of different size. Most atmospheric dispersion models treat the plume as if all radioactive materials were on particles of uniform size, so that only one deposition velocity can be employed in the calculations. Therefore a model has been developed enabling a range of deposition velocities to be taken into account. A procedure has also been established for calculating correction factors, that are to be used if the effect of particle size distribution is to be included in various models used in individual Nordic countries.

Improved knowledge of the processes involved in deposition of radioactive materials on different types of surfaces, as well as the processes taking place after deposition, is crucial to improvement of calculation of radiation doses to man from accidents. The dominating modes of radiation exposure in connection with large accidents are determined by such processes. In addition, representation of the conditions in urban areas needs to be made more realistic. Most calculation models at present address transport of radioactive materials in the environment as if the only environment of interest were agricultural areas. However, the larger part of the population in the Nordic as well as almost all European countries live in urban areas, and radiation directly from radioactive materials deposited on the ground and other surfaces often represent the most important modes of radiation exposure.

The tracer used in these experiments has changed over the time of the project. Initially cesium from weapons fall-out was used. As measurement techniques were improved, Be-7 (a radioactive nuclide continously produced in the upper atmosphere) was used as tracer. Since late April 1986 the Chernobyl fall-out has been the obvious choice, and it has been possible to measure a number of different nuclides, although work has been concentrated around I-131, Cs-134 and Cs-137.

When the radioactive materials are deposited in dry weather, the deposition depends upon the characteristics or the surface upon which deposition takes place. They are of no importance in connection with deposition during rainfall (wet deposition).

Dry deposition on smooth surfaces like paved surfaces and tiles was found to be roughly a factor of ten lower than on rough surfaces like mowed grass and some types of rough roof materials (e.g. non-glazed ceramic tiles). Dry deposition on grass was found to be proportional to the density of grass. Dry deposition in a forest was higher than on grass. Dry deposition on vertical surfaces was found to be about a factor ten lower than on horizontal surfaces.

As a result of the investigations it appears that deposition upon trees, divided into vertical sections, is proportional to the mass of the measured section. This implies that deposition on a forest will be proportional to the mass density much in the same manner as observed for grass.

<u>Run-off</u> or <u>wash-off</u> are expressions used in connection with wet deposition, to describe removal of radioactive materials with the rain water from the place of deposition. It is a process that takes place at the same time as the deposition, or in connection with the same rainfall. The radioactive materials do not, of course, disappear but in many cases they follow the rain water to brooks, rivers, lakes or the ocean; and in most cases the potential doses to humans are considerably lower if run-off has taken place.

The expression <u>weathering</u> is used to describe more long-term removal of radioactive materials from a surface by natural processes (weather).

Experiments indicate that run-off and weathering from lawns is negligible, and it is also very low from non-glazed roof tiles, and from tar-paper, which is frequently used as a roof material in the Nordic countries. Another common roof material is lacquered metal panels, from which run-off and weathering is found to be much larger. Actual run-off from roofs, the process which takes place with wet deposition, is found to be much larger than weathering after dry deposition. And the removal during the first rainfall after dry deposition is much larger than later on.

Another aspect of accident consequence assessment where the uncertainties have traditionally been large is the aspect of shielding against radiation, either against radiation from a plume containting radioactive materials, or against radiation from radioactive materials deposited on the ground. Exposure models used in accident consequence assessment models often use extreme simplifications for this purpose.

The situations where a person is exposed to radiation from radioactive materials deposited upon ground was the subject of a previous Nordic project. In the present report a computer program is described dealing with the situation where a person is exposed to radiation from radioactive materials in a plume. The program is based on the relatively simple so-called point kernel method. Since the geometry of a house can not be accurately described by point kernel, the simple method is tested against the much more accurate Monte Carlo method. This, however, is so time-consuming on the computer that only a limited number of cases can be run.

Inhalation doses have traditionally been calculated in accident consequence assessment models as if the person spends all his time outdoors. Experiments in the Nordic countries show that indoor air concentrations were 2 - 5 times lower than those outdoors, resulting in correspondingly lower inhalation doses. There are two possible explanations. One is that particles containing the radioactive materials will be retained in narrow fissures through which outdoor air enters a house (filtering effect of houses), the other that radioactive materials in the indoor air will deposit in the rooms. Up till now it has been thought that the filtering effect is the most important. Earlier indoor measurements gave deposition velocities that were very low compared to deposition velocities on outdoor surfaces. The surfaces measured in these cases were smooth and plane. However, interiors of dwellings contain many surfaces where deposition could be expected to be much larger, like rugs and fabrics. Measurements in furnished rooms show that the total deposition in this case is much larger, and from these experiments it may be concluded that almost the whole reduction of indoor air concentration is due to deposition, and not to the filtering effect.

Winter conditions are, of course, of particular interest in the Nordic countries. Consequences of a reactor accident may be different under winter conditions for a number of reasons. Deposition with snow will be different from deposition with rain. Deposition on snow-covered surfaces will be different from deposition in the same areas during summer. Weathering in winter and spring will be different from summer conditions, and decontamination may be cheaper and much less destructive than under summer conditions.

Such problems have been addressed in previous and in the present Nordic program. The experiments performed are to a large degree the first of their kind.

Decontamination of snow-covered areas can be performed with ordinary snow-removal equipment. Between 25 and 99% of the contaminant was removed in this manner, with the lowest efficiency in cases with a hard-packed, thin snow-cover.

When the different exposure pathways have been compared in the past, it has generally been found that resuspension (lifting of deposited material into the atmosphere, e.g. by wind or agricultural activities) is a pathway of no particular importance. After Chernobyl resuspension tests were carried out, using air-filter stations originally intended for measurement of radioactive materials from nuclear weapons tests. It was found that resuspension was higher in urban than in rural areas. After snowfall one would expect resuspension to be reduced. But the concentrations stayed at the same level. The probable explanation of this phenomenon is that neighboring forest areas are important sources of resuspended activity. It is expected that resuspension will be reduced over the years, but it was found that the variation over seasons is much larger than the gradual reduction over the few years followed up till now. The seasonal variation has peaks in spring and autumn.

The physico/chemical form of radioactive materials is important in connection with deposition, with migration of the activity after deposition, and with uptake by plants. Several methods of investigation have been utilized. Hot particles have been identified; some containing only few radionuclides and some containing mixed fission products. It was found that much of the radiocesium from Chernobyl fallout is in colloidal form, and accordingly less available for uptake by animals. Three years after the accident more than 80 - 90% of the cesium nuclides are present in the upper soil layer. The cesium nuclides are strongly associated with organic and mineral matter, while Sr-90 is more mobile. It has also been found that the relative uptake of radiocesium to animals from grass doubled from 1986 to 1987 or 1988. The explanation is that in 1986 the radiocesium was mostly deposited externally on grass, while in the following years it was taken up via the roots into the plants, and thus became more available for uptake in animals.

Actions to mitigate the effects of an accident are linked to the "action levels" at which eventual mitigation actions are initiated. This has an impact on health consequences and economy. In the past, assessments of accidental consequences have often been performed using parameters and assumptions from the American Reactor Safety Study from 1975 or insufficient modifications of these.

The cost of alternative types of mitigating actions may be very different, and so may be their efficiency measured in saved lives. The most costly mitigating action is not necessarily the most efficient. The relationships also depend upon the magnitude of the release and upon various site characteristics, including weather conditions. In the Nordic program it was found that better sheltering is very efficient in reducing the short-term consequences of an accident. The only practical type of mitigating actions to reduce long-term consequences are restrictions on consumption of various types of foodstuffs.

In the days and weeks following Chernobyl various mitigating actions were actually adopted in the Nordic countries. The experience in Finland shows that in actual practice it was not possible to carry out the risk comparisons and cost-benefit analyses implied in the principles stated by the International Commission on Radiological Protection (ICRP). A number of recommendations were given by the authorities, but it was not necessary to impose any restrictions.

In Norway large-scale practical mitigating actions that have been employed since 1986 have to a large extent been developed within the Nordic program. The benefits have been substantial. Several types of mitigating actions were investigated. The simplest and most obvious is to put the animals on "clean" fodder before slaughtering. In many cases this is achieved simply by allowing sheep and cattle who have been grazing in the mountain areas to graze in the valleys (where the soil type is different from that in the mountains, so that uptake of radiocesium to plants is lower) for two to eight weeks. In addition the animals are given concentrate containing 5% of a clay mineral, bentonite, that prevents uptake of radiocesium. This type of mitigating action is quite inexpensive.

Another efficient cesium binder is referred to as Prussian Blue. Two practical methods of application have been developed: salt-lick stone and bowel tablet, both containing Prussian Blue. A salt-lick stone is placed in a suitable position in the grazing area. It is a practical solution for animals being relatively stationary, even when they are up in the mountain areas. The bowel tablet dissolves slowly in the stomach of the animal, and is meant to last all through a season.

Much information has been collected that will contribute to improve future predictions of uptake in animals. One method, which makes it possible to measure intake of cesium directly, rather than by sampling in the environment, utilizes "fistulated animals". These animals have a metal tube operated into their gullet in the throat, from which samples of the intake of vegetation is collected. Thus transfer factors from the actual intake to content in meat can be determined properly. Reindeer, sheep and goats have been used in such experiments.

<u>Mitigating actions</u> to reduce the cesium content in <u>fresh-water fish</u> have also been studied. Fish with a high content of radioactive cesium were placed in "cesium-clean" pools over the winter of 1986/87. It appeared that there is very little biological activity over the winter period, and correspondingly very little reduction in cesium level in the fish.

Fish were also studied in their natural habitat, in twin mountain lakes. Potassium was added to the lower lake, and the impact upon the cesium content in the fish was examined. The "effective half-life" (the time until the content in fish is reduced to half the original value) was found to be about 250 days in the lake to which potassium was added, and about 400 days in the upper lake.

The <u>cost</u> of long-term mitigating actions including the cost of discarded food after Chernobyl has been analyzed in Finland, Norway and Sweden. In Denmark there were no significant economic consequences. The analyses incorporate different types of costs and restrictions, and are for that reason not directly comparable. In Norway the value of condemned food and cost of related mitigating actions has been somewhat less than 300 million NOK (43 million USD), while the value was close to 400 million SEK (52 million USD) in Sweden. In Finland the value of condemned food products was quite modest, amounting to roughly 300,000 FIM (20,000 USD) in total.

The cesium content in various food products can be reduced in the course of food preparation. A large number of foodstuffs, like fish, meat, berries, vegetables, potatoes and other root vegetables was examined. The methods of food preparation that most efficiently reduced the cesium content were salting, boiling in ample water, and preparing of juice by steaming berries, as well as parboiling mushrooms and soaking of dried mushrooms.

Soon after the accident a collection of Chernobyl measurements in the Nordic countries was initiated in the so-called Nordic Chernobyl Data Base. A group of contact persons in the Nordic countries has been formed to assure that data submitted from their country is of satisfying quality. Institutions that submit data have had to take part in an intercalibration program, again to assure proper quality of submitted data. At present 14 Swedish, 4 Norwegian, 3 Danish and 4 Finnish laboratories participate.

The AKTU program has contributed considerably to the continued development of methods and improvement of assumptions and parameters utilized by the methods within the area of nuclear accident consequence assessment, so that consequences of nuclear accidents can be assessed with higher certainty. It has also contributed to improved understanding of the problems involved in more general terms. Particularly rewarding have been the opportunities provided by the program of investigating problem areas where very little work had been done previously, and problem areas of particular interest in the Nordic countries, such as winter conditions, uptake in animals grazing in the mountain areas and the various types of mitigating actions developed. An extremely important secondary effect of the program has been to maintain the good personal contacts between the scientists in the Nordic countries involved in this area, and of having the possibility of establishing contact with scientists in related areas.

SAMMENDRAG (ABSTRACT IN NORWEGIAN)

En omfattende innsats er gjort internasjonalt for å utvide vår kunnskap om konsekvensene av utslipp til omgivelsene fra ulykker i nukleære anlegg.

Nordisk arbeide på dette området har vært rettet mot å forbedre metodene for estimering av omgivelses-konsekvenser, slik at mer realistiske analyser kan utføres. Etter Tsjernobyl ble interesseområdet utvidet til også å omfatte eventuelle mottiltak etter et utslipp av radioaktive stoffer til omgivelsene, og hvilke erfaringer man etter Tsjernobyl har høstet med slike mottiltak.

Interesseområdet på feltet ulykkes-konsekvens-analyse er vidt; det spenner fra meteorologi til lokal jordbrukspraksis. Atmosfæriske spredningsmodeller blir brukt til beregning av konsentrasjoner av radioaktive stoffer i luften og av hvor mye som deponeres på bakken. Andre typer informasjon blir brukt til å evaluere de radioaktive stoffers oppførsel i forskjellige typer omgivelser: i tettbebyggelser, jordbruksområder, skog- og fjellområder. Det finnes også modeller som brukes til å estimere overføringen til mennesker via matvarer og drikkevann.

En helt korrekt beregning av stråle-eksponering av mennesker som følge av en nukleær ulykke kunne bare utføres hvis hele naturen kunne beskrives og modelleres i alle detaljer. Vår kunnskap om meteorologi, økologi og på diverse andre felt ligger langt fra dette nivået, og et slikt mål vil vel aldri nåes. Forskningsinnsatsen er derfor rettet mot å utvikle "rimelig bra" metoder, inneholdende diverse forenklinger og "snarveier", som så godt som mulig beskriver naturen.

Over de siste 10 - 15 år har det vært gjort diverse forsøk på å kvantifisere usikkerhetene i beregninger av stråle-eksponering. Usikkerhetenes størrelse avhenger selvfølgelig av beregningsmetodene som benyttes og kvaliteten på input-data som er tilgjengelig. I dette nordiske samarbeids-programmet har de feil som skriver seg fra usikkerheter i de tallrike parametre som inngår i beregningene blitt analysert. Det viser seg at de mulige feil kan være betydelige, hvilket tyder på at vår kunnskap om måten radioaktive stoffer oppfører seg på i naturen fremdeles er utilstrekkelig.

I det følgende oppsummeres de vesentligste resultater av det nordiske AKTU-programmet, som ble utført i årene 1985 til 1989:

Modeller som beskriver atmosfærisk spredning over middels lange avstander har tradisjonelt vært utført med modeller bygget på ganske enkle grunn-prinsipper. En avansert modell har blitt utviklet innenfor rammen av det nordiske programmet, der værets variasjoner med tiden kan tas hensyn til på en mer realistisk måte enn det som er mulig i de enklere modellene. Modellen er av en såkalt multi-puff type. Modellen har blitt utprøvet mot resultater fra eksperimenter som ble utført tvers over Øresund, fra Sverige til Danmark, og omvendt. Modellen er nå i bruk både i Vest-Tyskland og i Storbritannia.

Radioaktive stoffer som finnes i en radioaktiv sky er stort sett festet på partikler av forskjellig art og av forskjellig størrelse. I de fleste atmosfæriske spredningsmodeller antas det som en forenkling at alle partikler har samme størrelse, slik at det bare er påkrevet å bruke én såkalt deposisjons-hastighet i beregningene. Derfor har det blitt utviklet en modell der det er mulig å anvende en rekke forskjellige deposisjonshastigheter i samme beregning. En prosedyre har også blitt utviklet for beregning av korreksjonsfaktorer, som kan brukes hvis man ønsker å ta hensyn til virkningen av bestemte partikkel-størrelses-fordelinger i andre beregningsmodeller i de nordiske land.

Det er viktig å forbedre kunnskapene om deposisjon av radioaktive stoffer på forskjellige typer overflater, så vel som de prosesser som finner sted etter deposisjon. Disse prosesser er viktige i forbindelse med de eksponeringsveier som dominerer stråleeksponeringen fra store utslipp. Dessuten er det viktig å få en mer realistisk representasjon av forholdene i tettbebygde områder. De fleste beregningsmodeller behandler nå transport av radioaktive stoffer i omgivelsene som om jordbruksområder var den eneste type omgivelse som er av interesse. Imidlertid oppholder største-delen av befolkningen seg i tettbebygde områder, i Norden akkurat som i resten av Europa, og stråling direkte fra radioaktive stoffer deponert på bakken og andre overflater er ofte den dominerende eksponeringsmåte.

Spormaterialet som er brukt i disse eksperimentene har skiftet i løpet av den tiden denne typen eksperimenter har blitt utført. Opprinnelig ble cesium fra kjernefysiske våpen-prøver anvendt. Men måleteknikken ble gradvis forbedret, og det ble mulig å bruke

Be-7 (en radioaktiv nuklide som kontinuerlig genereres i de øvre lag av atmosfæren). Etter april 1986 har nedfallet fra Tsjernobyl vært det opplagte valg av spormateriale i slike eksperimenter; og det har vært mulig å undersøke et antall radioaktive nuklider, selv om arbeidet har vært konsentrert om Cs-134 og Cs-137.

Når radioaktive stoffer deponeres i tørt vær, avhenger deposisjonen av egenskapene til den overflate deponeringen finner sted på. Det er ikke tilfellet i forbindelse med deposisjon når det er nedbør (våt deposisjon).

Tørr deposisjon på jevne overflater som asfalt og heller av stein eller sement ble funnet å være omtrent en faktor ti lavere enn på ujevne overflater som gress og noen typer grove takmaterialer (f.eks. uglasert takstein). Tørr deposisjon på gress ble funnet å være proporsjonal med gressets tetthet (vekt pr. flateenhet). Tørr deposisjon i en skog var høyere en på gress, mens på jevne vertikale flater var den omtrent en faktor ti lavere enn på jevne horisontale flater.

Undersøkelser indikerer også at deposisjon på trær, delt i vertikale seksjoner, er proporsjonal med vekten av hver seksjon, og dette igjen innebærer at deposisjon i en skog vil være proporsjonal med tettheten av trær (vekt pr. flateenhet) tilsvarende det som ble observert for gress.

To uttrykk som brukes i forbindelse med våt deposisjon, for å beskrive fjerning av radioaktive stoffer med regnvann fra det sted hvor de ble deponert, er avrenning (run-off) og avvaskning (wash-off). Disse uttrykkene brukes (litt om hverandre) for å beskrive fjerning på det tidspunkt når deposisjonen finner sted, eller i forbindelse med samme regnvær. De radioaktive stoffene "forsvinner" selvfølgelig ikke, men følger som regel regnvannet til bekker, elver, sjøer og havet, og potensielle doser til mennesker vil nesten alltid være langt lavere enn om avrenning ikke hadde funnet sted.

Et uttrykk som brukes for å beskrive en mer langsiktig fjerning av radioaktive stoffer fra deponerings-stedet ved naturlige prosesser (været) er weathering (det finnes dessverre ikke noen skandinavisk versjon av dette uttrykket).

Eksperimentene tyder på at både avrenning og weathering er helt uten betydning på gressplener, og dessuten er veldig lav på uglaserte takstein og fra tjærepapp. Effekten er vesentlig viktigere på lakkerte metallplater (av den typen som er laget slik at det ser ut som glaserte takstein). Avrenning (den prosessen som finner sted samtidig med regnværet som de radioaktive stoffene kommer ned med) er mye kraftigere enn weathering etter tørr deposisjon, og det første regnvær etter tørr deposisjon har meget større virkning enn etterfølgende regnvær.

En annen side ved ulykkeskonsekvensberegninger der det tradisjonelt har vært store usikkerheter gjelder skjerming mot stråling, enten fra en sky som inneholder radioaktive stoffer eller fra radioaktive stoffer som er deponert på bakken. De modeller som finnes, bruker ofte ekstreme forenklinger i skjermingsberegningene.

I et tidligere nordisk prosjekt ble det utviklet en modell for beregning av skjerming mot stråling fra radioaktive stoffer deponert på bakken. Innen det nordiske program som beskrives i denne rapporten ble det utviklet en modell for den situasjonen hvor de radioaktive stoffene finnes i en passerende sky. Programmet er basert på den relativt forenklete såkalte "point kernel" metode. Ettersom det ikke er mulig å beskrive geometrien til et hus ved denne metoden, så har den forenklete metoden blitt testet mot den mye mer nøyaktige Monte Carlo metoden. Disse datakjøringene er imidlertid så tidskrevende at bare et begrenset antall beregninger har kunnet utføres.

Inhalasjonsdoser har tradisjonelt i feltet ulykkeskonsekvensanalyse blitt beregnet som om personen tilbragte hele tiden utendørs. Eksperimenter i de nordiske land har vist at innendørs luftkonsentrasjoner er 2 - 5 ganger lavere enn utendørs. Dette innebærer en tilsvarende reduksjon i inhalasjonsdoser. Det finnes to mulige forklaringer på dette fenomenet. Den ene er at partiklene med de radioaktive stoffene holdes igjen i de utettheter hvorigjennom luften trenger inn i huset (dette refereres til som filter-effekten), og den andre er at radioaktive stoffer i luften innendørs deponerer på overflater i rommene. Hittil har man ment at filter-effekten var den viktigste av disse to effektene. Målinger av innendørs deposisjon som tidligere er utført ga som resultat at deposisjonshastighetene var langt lavere enn hva som var målt for utendørs overflater. Imidlertid var de innendørs overflater som ble målt jevne og glatte, mens et interiør inneholder mange typer flater der deposisjonen kunne ventes å være langt større, slik som tepper og andre tekstiler. Målinger utført i møblerte rom viser at total deposisjon virkelig er langt større enn på de tidligere målte glatte flater, og konklusjonen er nå at den største del av reduksjonen i innendørs luftkonsentrasjoner skyldes innendørs deponering.

Vinterforhold er selvfølgelig av særlig interesse for de nordiske land. Konsekvenser av en nukleær ulykke som finner sted under vinterforhold kan være svært forskjellige fra sommerforhold av en rekke forskjellige årsaker. Deposisjon med sne vil være forskjellig fra deposisjon med regn. Deposisjon på en snekledd flate vil være forskjellig fra deposisjon i de samme områder om sommeren. Weathering om vinteren/våren vil være forskjellig fra om sommeren, og dekontaminering kan under mange forhold om vinteren utføres mindre kostbart og mindre destruktivt enn om sommeren.

Mange av disse problemstillingene har blitt undersøkt i dette og de foregående nordiske programmene. Mange av de eksperimentene som er utført er de første i sitt slag.

Dekontaminering av snedekte områder kan utføres med vanlig snerydnings-redskap. Mellom 25 og 99% av forurensningen ble fjernet, med lavest effektivitet i de tilfelle hvor sneen var i et tynt, hard-pakket lag.

I tidligere utførte ulykkesanalyser har man som regel funnet at resuspensjon (den effekt at deponerte stoffer av vind eller annet kan løftes opp i luften igjen) har vært en eksponeringsmåte av svært liten viktighet. Resuspensjonsmålinger har blitt utført etter Tsjernobyl, ved bruk av luftfilter-stasjoner som egentlig er montert for å måle eventuelle luftbårne radioaktive stoffer fra sprengninger av kjernefysiske våpen. Det er funnet at resuspensjon er sterkere i tettbebygde enn i andre områder. Etter snefall ville man vente at resuspensjon ville bli redusert, men det viste seg at luftkonsentrasjonene forble omtrent på samme nivå. Den sannsynlige forklaring er at den største delen av de resuspenderte stoffene kommer fra nærliggende skogsområder. Det ventes at resuspensjon vil reduseres gradvis med tiden, men det ble funnet at variasjonen med årstider er meget større enn den eventuelle gradvise reduksjon over de få årene i hvilke målingene hittil har vært utført. Årstidsvariasjonene har topper på våren og høsten.

De radioaktive stoffers fysisk/kjemiske form er viktig i forbindelse med deposisjon, med transport av stoffene etter deposisjon, og med opptak til planter. Flere metoder er anvendt i de undersøkelsene som er utført innenfor rammen av dette nordiske programmet. "Hete" partikler har blitt funnet; noen inneholdende et fåtall radionuklider, noen inneholdende blandete fisjonsprodukter. Det ble funnet at mye av radiocesiumet i nedfallet etter Tsjernobyl er i kolloidal form, og følgelig mindre tilgjengelig for opptak til dyr. Tre år etter nedfallet finnes fremdeles 80 - 90% av cesium-nuklidene i det øvre jord-lag. Cesiumnuklidene er sterkt assosiert med organiske og mineralske stoffer, mens Sr-90 er mer transportabelt. Det er også funnet at det relative opptak av radiocesium til dyr er dobbelt så høyt fra gress fra 1987 eller 1988, som fra gress fra 1986. Dette forklares ved at i 1986 var det meste av radiocesium deponert på utsiden av plantene, mens i de følgende år det meste av radiocesium i gresset var tatt opp gjennom røttene, og derfor var mer tilgjengelig for opptak til dyr.

Tiltak som iverksettes for å lindre virkningene av en ulykke er knyttet til såkalte tiltaksgrenser, og dessuten til størrelsen på både helsekonsekvenser og økonomiske konsekvenser. Tidligere har vurderinger av størrelsen av disse konsekvensene ofte bygget på forutsetninger og parametre fra den amerikanske reaktorsikkerhetsstudien fra 1975 eller utilstrekkelige tilpasninger av disse til nordiske forhold.

Alternative typer mottiltak kan ha meget forskjellig kostnad, og meget forskjellig effektivitet, målt i sparte liv. Og det er ikke nødvendigvis samsvar mellom kostnad og effektivitet. Disse forholdene vil også være funksjon av utslipps-størrelse og diverse forhold i området som er av interesse, slik som værforhold. I arbeidet utført i det nordiske programmet har man funnet at god skjerming er en meget effektiv måte å redusere korttids-konsekvensene på etter en ulykke. Når det gjelder langtids-konsekvensene er det bare forskjellige restriksjoner på forbruk av matvarer som er aktuelle mottiltak.

I dagene og ukene etter ulykken i Tsjernobyl, ble forskjellige typer mottiltak faktisk iverksatt i de nordiske land. Erfaringene fra Finland viser at det i realiteten ikke var mulig å utføre de risikosammenligninger og "cost-benefit"-analyser som det er forutsatt skal utføres, i prinsippene til den internasjonale strålevernskomité (ICRP). En rekke anbefalinger ble gitt av strålevernsmyndighetene i Finland, men det var ikke nødvendig å iverksette spesifikke restriksjoner.

I Norge har stor-skala mottiltak vært iverksatt fra og med 1986, og disse har til stor grad blitt utviklet innenfor rammen av det nordiske programmet. Innsparingene har vært betydelige. Det enkleste og mest opplagte mottiltak er å sette dyrene på "rent" fòr før slakting. I mange tilfelle ble dette opnådd bare ved å la dyrene (sau og storfe) som hadde vært på fjellbeite, gresse nede i dalen (hvor jordsmonnet er forskjellige fra det i fjellet, slik at opptaket til gress blir lavere) i to til åtte uker. I tillegg blir dyrene gitt kraftfor tilsatt 5% av et leirmineral (bentonitt) som motvirker opptak til dyrene av cesium. Denne typen mottiltak er ikke kostbar.

En annen effektiv cesium-binder er såkalt Berlinerblått. To praktiske anvendelsesmetoder har blitt utviklet: slikkestein og vomtablett. En slikkestein skal dekke dyrenes saltbehov, og er en gammel tradisjon. I den nye versjon inneholder den i tillegg til salt også Berlinerblått. Den blir plassert i en hensiktsmessig posisjon i beiteområdet, og passer for dyr som er relativt stasjonære. En vomtablett løses langsomt opp i vommen til dyret, og skal vare hele beitesesongen.

Mye informasjon har blitt samlet som vil bidra til å gjøre fremtidige prediksjoner av opptak til dyr mer pålitelige. En metode, som har blitt anvendt i det nordiske programmet, og som gjør det mulig å måle opptak av cesium direkte, er bruk av såkalte "fistulerte" dyr. Disse dyrene har et metallrør operert inn i spiserøret. Prøver kan så samles når ønsket gjennom dette røret. På denne måten kan overføringsfaktorer fra inntak til innhold i kjøtt bestemmes med større sikkerhet enn ved andre metoder. Reinsdyr, sauer og geiter har blitt brukt i slike eksperimenter.

Mottiltak for å redusere innholdet av radiocesium i ferskvannsfisk har også blitt undersøkt. Fisk med høyt innhold av radioaktivt cesium ble plassert i bassenger med "cesium-rent" vann gjennom vinteren 1986/87. Det viste seg at det var meget lav biologisk aktivitet gjennom vinteren, og følgelig ubetydelig reduksjon av cesium-nivået i fisken.

Fisk ble også studert i sitt naturlige miljø, i to "tvillingsjøer" i fjellet. En kaliumforbindelse ble tilsatt den nedre sjøen, og virkningen på cesiuminnholdet i fisken ble studert. Den effektive halveringstiden (den tid det tar før nivået har blitt halvert) var rundt 250 dager i den sjøen der kaliumtilsetning var foretatt, og rundt 400 dager i den øvre sjøen.

Kostnadene av lang-tids mottiltak, inkludert verdien av kasserte matvarer, har blitt analysert i Finland, Norge og Sverige. Tsjernobyl-nedfallet førte ikke til økonomiske konsekvenser av betydning i Danmark. Analysene for de forskjellige land inneholder forskjellige typer økonomiske konsekvenser og forskjellige restriksjoner, og kan derfor vanskelig sammenlignes. I Norge har verdien av kassert mat og kostnader forbundet med relaterte mottiltak vært litt under 300 millioner NOK. I Sverige var omtrent motsvarende kostnader 400 millioner SEK (ca. 450 millioner NOK). I Finland var det bare beskjedne mengder matvarer som ble kassert, til en verdi av omtrent 300.000 FIM (vel 500.000 NOK).

Cesiuminnholdet i forskjellige typer matvarer kan reduseres under matlaging. Mange typer matvarer, som fisk, kjøtt, bær, grønnsaker, poteter og andre rot-grønnsaker ble undersøkt. De mattilberedningsmetodene som mest effektivt reduserte cesiuminnholdet var salting, koking i rikelig med vann, dampsafting av bær, samt forvelling av sopp og bløtlegging av tørket sopp.

Kort tid etter ulykken i Tsjernobyl tok man innen rammen av det nordiske program initiativet til å samle "Tsjernobyl-målinger" i de nordiske land i en Nordic Chernobyl Data Base. En gruppe kontaktpersoner har ansvaret for at data som inkluderes i databasen har tilfredsstillende kvalitet. De institusjoner som ønsker å bidra med data til basen, må først delta i et interkalibreringsprogram, for å sikre at kvaliteten opprettholdes. Hittil deltar 14 svenske, 4 norske, 3 danske og 4 finske laboratorier.

AKTU-programmet har bidratt vesentlig til fortsatt utvikling av metoder og forbedring av antagelser og parametre som brukes ved analyse av ulykkeskonsekvenser. Det har også bidratt til en generelt bedre forståelse av hele området. Det har vært særlig verdifullt å ha anledning til innen dette programmet å undersøke problemområder der svært lite arbeide har vært gjort tidligere, og problemområder som er av særlig interesse i de nordiske land. En sekundær virkning har også vært gjennom samarbeidet å ytterligere styrke de faglige kontakter innen det nordiske fagmiljøet, og å knytte nye kontakter til spesialister innen relaterte fagkretser.

CONTENTS

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1. INTRODUCTION

Nordic cooperation in the area of Accident Consequence Assessment is performed within the Nordic Safety Program. The countries involved are: Denmark, Finland, Norway and Sweden. This work is partially funded by the Nordic Council of Ministers, via the Nordic Liaison Committee on Atomic Energy. The first of the Nordic Safety Programs started in 1977, and the second in 1981. The present Program started in 1985, and was completed in 1989. The overall Safety Program encompasses work within five principal areas:

- Release of radioactivity from severe reactor accidents, its dispersion and environmental impact.
- Nuclear waste management.
- Risk analysis and safety philosophy.
- Materials research.
- Advanced information technology.

The first of these areas is referred to as AKT (abbreviation of the Nordic word "aktivitet"). It is subdivided into two separate sub-areas. The sub-area referred to as AKTI (I for "inside" or "innenfor" in Nordic language) addresses the relevant phenomena inside the containment. The other sub-area addresses the relevant phenomena outside the containment and is referred to as AKTU (U for "utenfor" in Nordic language, which means "outside"). AKTI encompasses identification of accident sequences leading to release of radioactivity from the reactor core, various release-, transport- and recombination mechanisms, and investigation of accident sequences that may lead to rupture of the containment. AKTU encompasses, based upon release time and other release characteristics, analyses of the manner in which radioactive materials disperse through the environment, eventually leading to health consequences to man; or alternatively adoption of mitigating actions, with accociated socio/economic impact.

The main purpose of the present report is to present and put into perspective the work performed within the AKTU program area. It also contains descriptions of some closely related Nordic research that has not been part of the AKTU program. Where appropriate, research in non-Nordic countries has also been referred to at some length. However, throughout the report there is special emphasis upon the problems addressed in the 17 AKTU projects. The aim has been for the present report to contain all important information on all AKTU projects, so that it will only be necessary to aquire the project reports when quite detailed information is desired. Some of the shorter project reports are included in extenso, with only slight modifications.

The following Nordic institutions have been directly involved in the AKTU program: Risø National Laboratory (Denmark). Technical Research Centre of Finland. Finnish Centre for Radiation and Nuclear Safety. Finnish Meteorological Institute. Ministry of Agriculture and Forestry (Finland). Institutt for energiteknikk (Norway). Ministry of Agriculture (Norway). Agricultural University of Norway. Meteorological Institute of Norway. Studsvik (Sweden). National Defence Research Laboratory, Stockholm (Sweden).

Typically 1-2 persons at each institution are engaged in research and application efforts within the actual area of concern. They normally belong to research groups of a size of 5-20 members working with related problems (normal operation, waste storage, nonnuclear releases).

2. HISTORICAL BACKGROUND

Until 1986 and Chernobyl there had been no accidents at nuclear power plants where large amounts of radioactive materials had been released to the environment. The releases resulting from the only other serious accident at a nuclear power plant, the Three Mile Island in 1979, were very small, and so were the resulting doses. In addition there has been a serious accident at a reactor for plutonium production, at Windscale in 1957. Information has also very recently been made available, through the IAEA, on a large accident in a reprocessing plant in Ural in 1957. The releases from these accidents were orders of magnitude below the release from Chernobyl.

Accordingly, there is no statistical material upon which the risk of nuclear power plants can be based. Risk assessments in this connection have been performed in other ways, by means of development of models and collection of data that make the assessment possible. The models should imitate nature as closely as needed for the particular purpose for which the models should be used. The aim is not unconditionally to imitate nature as closely as possible, since a model that is too detailed may become impractical to use. As for data to be used by the models, it is imperative that they are representative of the geographical location. On the other hand, some uncertainty in the numerical values of specific types of data may be acceptable. The condition is that the data should be sufficiently accurate for the purpose for which they shall be used.

The first official report of importance in which a risk assessment of a severe nuclear power plant accident was described, is WASH-740, published by the U.S. Atomic Energy Commission in 1957. The report was prepared for a specific purpose, namely to assess nuclear accidents seen from an insurance company point of view. A number of crude assumptions had to be made, and the report dealt mainly with the consequence aspect of risk, while calculation of the related probabilities was not attempted.

A publication from 1967 by Farmer and Beattie is a classic in the field of risk assessment. Here it was pointed out that consequences and probabilities belong together. It is meaningless to present the estimated consequences of an accident without also presenting an estimate of the probability. It was also emphasized that it is not sufficient to look only at the severest accidents. A complete picture of the risk is obtained only when all potential accident sequences, large and small, are taken into consideration.

In the years to follow many institutions developed models, collected data and performed reactor accident calculations. In the Nordic countries one should especially mention the Swedish "Urban Siting Study", initiated in 1970 and published in 1974; and the Norwegian "Nuclear Power Plant at Brenntangen. A Preliminary Assessment of Safety Issues related to Atmospheric Release", published in 1971. An informal Nordic group for exchange of information in this field, called SNODAS, was formed in 1975. In the years to follow a couple of code intercomparison studies were also performed within this group. The experience gained here proved very valuable in the international Benchmark Study, carried out 1980 - 1983 by an ad hoc expert group under OECD/NEA (Organisation for Economic Co-operation and Development/Nuclear Energy Agency).

A milestone in the field was reached with the publication in 1975 of the report WASH-1400 (the U.S. Reactor Safety Study) by U.S. Nuclear Regulatory Commission (ref. WA75). There is probably no later work in this field that is not influenced by the work performed within the Reactor Safety Study. A number of other studies soon followed, from Denmark, Norway, Sweden, Finland, Netherlands and Federal Republic of Germany. An "updated version of WASH-1400", NUREG-1150 is about to be published by the U.S. Nuclear Regulatory Commission.

3. ACA, PCA, PSA AND ERAS (OR REAL-TIME)

These are expressions one comes across all the time in this field.

PSA means Probabilistic Safety Assessment, and encompasses the whole risk assessment for the plant, including assessment of the accident sequences and the release characteristics (See Figure 3.1). One often talks about the three levels of a PSA:

- Level 1: Analyses of accident sequences and probabilities leading to core melt.
- Level 2: Analyses of (for core melt accidents) the sequences and probabilities leading to releases to the environment.
- Level 3: Analyses of the consequences and probabilities of consequences from environmental releases.

ACA means Accident Consequence Assessment, and signifies that a deterministic approach has been used in the assessment. It usually means that the consequences of a particular combination of release and weather conditions have been assessed. The results should, however, be accompanied by a probability, composed of the probabilities of the particular release and weather conditions.

PCA is Level 3 of a PSA, and the abbreviation means Probabilistic Consequence Assessment, and signifies that a probabilistic approach has been chosen. The results are usually presented as curves of CCDF´s (Conditional Cumulative Density Function) for the various types of consequences.

Figures 3.2 and 3.3 apply more or less both to an ACA and a PCA. In Figure 3.2 is shown the more general structure of an assessment, while Figure 3.3 shows in greater detail the dose calculation, broken down into the usual exposure pathways, and also including the various types of mitigating actions that are typically taken account of in the assessment. One particular feature shown in Figure 3.2 and neglected (for reasons of simplification) in Figure 3.3, is that potential doses are usually calculated as a first step (all types of mitigating actions neglected); these potential doses are tested against various dose criteria to find the need for mitigating actions; and then the calculation of expected doses and economic consequences, depending upon the choice of mitigating actions, is performed.

ERAS means Emergency Response Assisting Systems, also known by the name of Real-Time Assessment. This is meant to be used, not for ACA or PCA of potential accidents, but for assessment of the consequences of a release that has actually taken place, more or less simultaneously with the development of the situation. The models in an ERAS are basically the same as in an ACA or PCA, although somewhat simpler models are often used. The main difference, however, is in the mode of application. Since the assessments by an ERAS are performed while the situation is developing, it is possible to check the assessments against measurements, with a certain time delay. As a next step it is then possible to adjust unknown quantities, used in the assessment, to make the results of the assessment fit the measurements. And then as the next step to make new assessments (predictions) for distances to which the release has not yet been dispersed. And the next step again, after some time, is to make additional adjustments of the assessment, as additional measurements become available. The whole process is interactive. The results of the assessments are used to determine where in the surroundings to take measurements. The results of the measurements are used to improve the assessments, which are then repeated to determine where to take additional measurements, what countermeasures to adopt, etc.

The AKTU program has not been concerned with ERAS, but at the time when this is written, there are strong indications that there will be a *considerable emphasis on this type of models within a following* four-year Nordic safety program, 1990 - 1993.

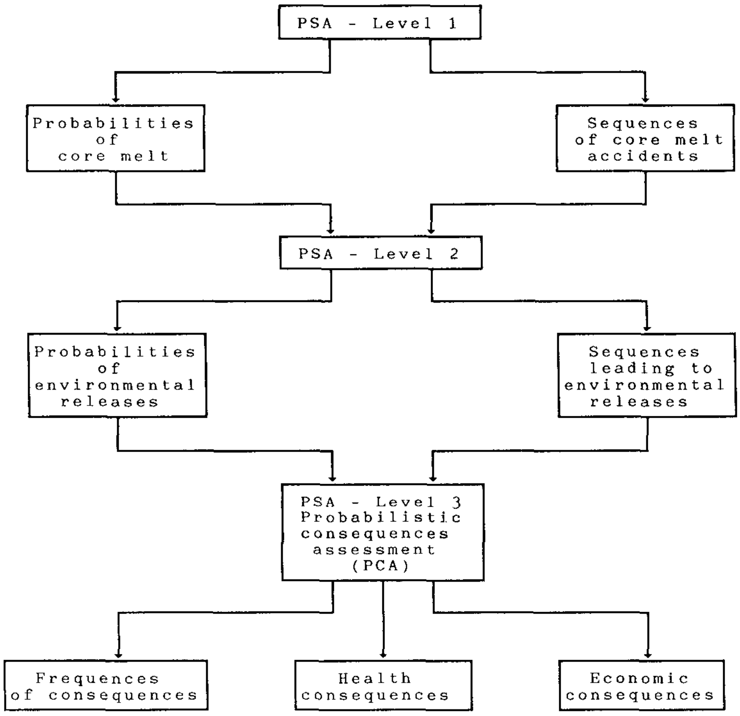
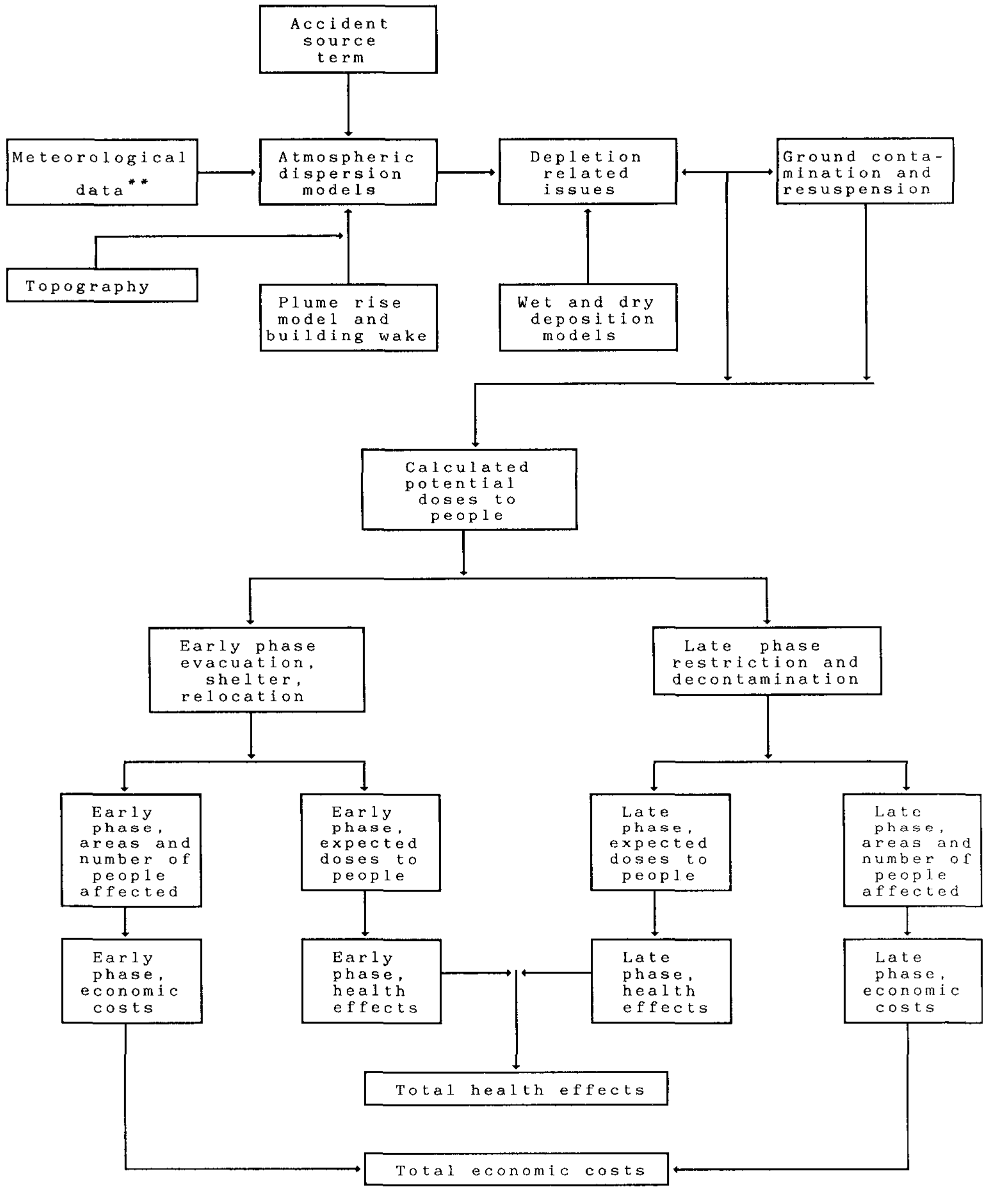


Figure 3.1 General structure of a PSA (Probabilistic Safety Assessment) for a nuclear installation.



** When the calculations are repeated for many statistically selected weather sequences, an assessment of the risk can be obtained.

Figure 3.2 Example of the structure of a PCA (Probabilistic Consequence Assessment) code.

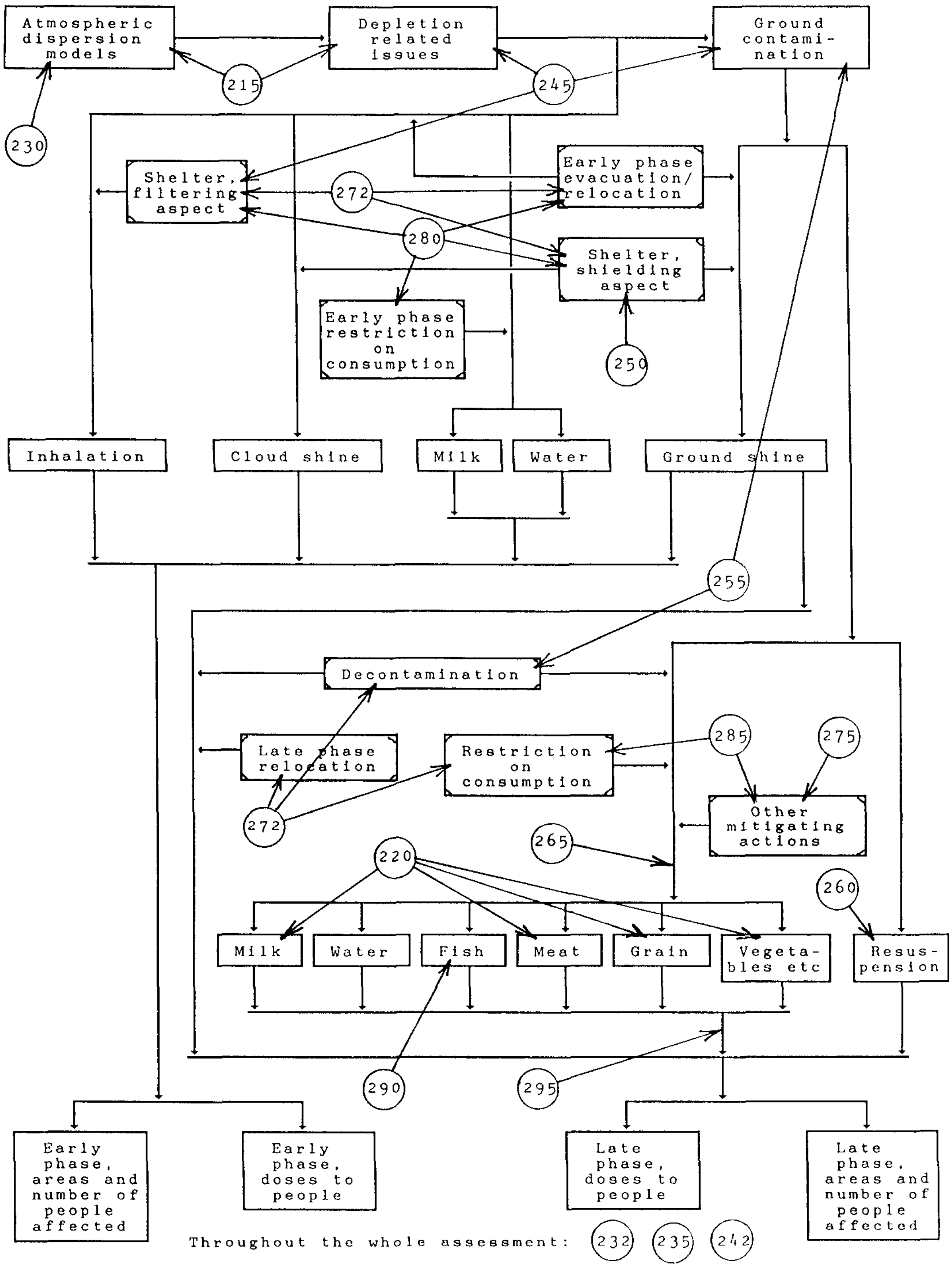


Figure 3.3 Scheme of dose calculations. In addition the "positions" of the AKTU projects are shown.

4. GENERAL INFORMATION ABOUT AKT, AKTI AND AKTU

The third period, 1985 - 1989, of the Nordic cooperation in Nuclear Safety covers a broad spectrum of questions related to nuclear safety. The area AKT is specifically directed to large accidents involving core melt in nuclear power reactors. The acronym AKT an abbreviation for "activity release" in Nordic language. The area roughly corresponds to Levels 2 and 3 in PSA studies.

4.1 The coupling between AKTI and AKTU

The AKT program area involves a very broad scientific field with sometimes very loose or no coupling between the disciplines. Examples of areas involved are thermal phenomena, aerosol behaviour etc at elevated temperatures inside the containment on one hand, and weather dependent atmospheric dispersion and uptake of specific radionuclides in plants and animals etc on the other hand. This made it necessary to divide the program area into two sub-areas AKTI and AKTU, each with its own scientific coordinator. AKTI has dealt with phenomena occuring inside the containment - i.e. PSA Level 2 questions - whereas AKTU has dealt with phenomena outside the containment - i.e. PSA Level 3 questions.

The AKT program area has had a common steering group for AKTI and AKTU, and joint annual seminars have been organized. The broad scientific field has sometimes made this structure somewhat difficult to handle because of conflicting interests or very narrow common scientific ground between experts involved. It has, however, also provided opportunities for giving a broad scientific perspective of the area "severe reactor accidents", which stands in the focus of the nuclear safety issue.

It is difficult to tell whether the coupling between the sub-areas in AKT has been an advantage, a disadvantage or immaterial. Similar coupling between these two areas of research has been utilized elsewhere (e.g. in OECD/NEA, where it was, however, recently made considerably looser). The value of such a constellation is obvious in connection with actual application to specific assessments for specific plants (e.g. WASH-1400 and NUREG-1150).

The direct coupling between AKTI and AKTU is the release data from the containment or the "source term". AKTI should provide a source term for AKTU. There is, however, no such thing as a generic "source term". They are all very plant specific. This caused lengthy discussions within the program. AKTU requested release specifications for some reference accidents for use in two projects. One of these is concerned with uncertainty/sensitivity analyses, and the other with the efficiency of ex-plant mitigating actions. Only after several discussions between representatives of AKTI and AKTU one reached agreement on a set of specifications on "source terms" to be used in these two projects.

4.2 Specifically about AKTU

The name AKTU is derived from the Scandinavian full name of the program area: "Aktivitetsfrigjørelse, -spredning og miljøpåvirkning. Utenfor inneslutningen", which means: Release of radioactivity, its dispersion and environmental impact. Outside the containment. A full list of the AKTU projects, project leaders and institutions involved is found in chapter 4.3.

The primary aim of the AKTU program area has been to improve ACA and PCA. Some of the results from the projects will also be relevant to Real-Time calculations, but improving Real-Time calculations is not a primary purpose. Some of the results of the projects have no direct application in connection with ACA or PCA at this time, but they give valuable additional insight and can probably find application in the future.

In Figure 3.3 are shown the "positions" of the AKTU projects in the ACA scheme. The remainder of this report will more or less follow the structure in this figure. Two of the AKTU projects can not be assigned a specific position in the scheme in Figure 3.3, as they address a wide range of items: Project AKTU-235 deals with parameter variation analysis and uncertainty analysis of consequence assessment models, while project AKTU-242 is concerned with the Nordic data bank for measurements after Chernobyl and a related Nordic intercalibration.

4.3 List of the AKTU projects

Main project 210. Methods and data related to transport through the environment.
Søren Larsen, Risø National Laboratory, Denmark.

Project 215. Improvement of consequence assessment models.
Seppo Vuori, Technical Research Centre of Finland.

Project 220. Dynamic models.
Sven Poul Nielsen, Risø National Laboratory, Denmark.

Project 230. Development and testing of multi-puff model for atmospheric dispersion.
Søren Thykier-Nielsen, Risø National Laboratory, Denmark.

Project 232. Pre-project for planning of emergency preparedness at the Nordic meteorological institutes, in relation to nuclear power accidents.
Anton Eliassen, Meteorological Institute of Norway.

Project 235. Parameter variation analysis and uncertainty analysis of consequence assessment models.
Olof Karlberg, Studsvik, Sweden.

Main project 240. Problems particularly related to urban areas or winter conditions.
Agneta Rising, Ringhals Power Plant, Sweden.

Project 245. Urban areas, deposition weathering and indoor doses.
Jørn Roed, Risø National Laboratory, Denmark.

Project 250. Shielding factors.
Per Hedemann Jensen, Risø National Laboratory, Denmark.

Project 255. Winter conditions.
Ulf Tveten, Institutt for energiteknikk, Norway.

Project 260. Resuspension in urban areas.
Björn Bjurman, Swedish Defence Research Establishment.

Project 265. The physical/chemical form of radionuclides in the fall-out after Chernobyl.
Brit Salbu, Agricultural University of Norway.

Main project 242. Nordic data bank for measurements after Chernobyl, and Nordic intercalibration.
Ole Walmod-Larsen, Risø National Laboratory, Denmark.

Main project 270. The relationship between health consequences, economic consequences, mitigating actions and dose criteria.
Seppo Vuori, Technical Research Centre of Finland.

Project 272. Analysis of the relationship between mitigating actions and consequences, calculated with the ARANO code.
Seppo Vuori, Technical Research Centre of Finland.

Project 275. Investigation of the actual efficiency of mitigating actions.
Knut Hove, Agricultural University of Norway.

Project 280. Analysis of recommended mitigating actions during a shorter time period after Chernobyl.
Leif Blomqvist, Finnish Centre for Radiation and Nuclear Safety.

Project 285. Investigation of the realistic costs of mitigating actions.
Ulf Tveten, Institutt for energiteknikk, Norway.

Project 290. Cesium i fresh water fish. Efficiency of mitigating actions. Biological half life.
Gordon Christensen, Institutt for energiteknikk, Norway.

Project 295. The influence of food preparation methods on the cesium content in various types of food.
Aino Rantavaara, Finnish Centre for Radiation and Nuclear Safety.

5. ATMOSPHERIC DISPERSION

Typical of an accident release is the limited duration of the release, on the order of hours, days or at the most weeks. This is the most important difference from releases taking place during normal operation, as far as assessing the consequences is concerned. For this reason the models for atmospheric dispersion to be used in connection with accidents, have to be on a higher level of detail than for dispersion calculations in connection with normal operation releases.

Most ACA models are, moreover, not able to treat properly releases of duration more than a couple of hours, mainly because the most frequently used type of model is still the relatively simple Gaussian plume model. Descriptions can be found in numerous reports and textbooks, for example (TI83 and IA87). The basic aspect of the Gaussian plume model is that the distribution of pollutant within a plume, perpendicular to the plume axis, is described by a normal distribution both in the horizontal and vertical directions. The problem of assessing dispersion for releases of duration longer than some few hours is addressed in project AKTU-215.

Techniques have been developed so that both dry and wet deposition, and also changes in the weather during dispersion, can be taken into account. These techniques have, however, definite weaknesses, and some of these weaknesses are addressed projects AKTU-215 and AKTU-230.

In connection with a preproject, partially funded also by the Nordic Council of Ministers via NKA, and carried out in 1986, Helen ApSimon prepared an overview of types of models in use, which is included in the following subchapter. The overview was primarily prepared with long-range dispersion in mind, but it really covers the short-range aspect as well. The purpose of the preproject was to plan an international intercomparison/validation study of long-range dispersal programs, using the measurements after Chernobyl, and the plans for the project are described in (ref. TV87a). A similar project is now coordinated in cooperation between IAEA, WMO and CEC.

5.1 Types of models in use for long-range dispersion

Models available for simulating atmospheric transport and deposition of radionuclides released in a nuclear accident vary widely in their complexety. They may be divided into broad classes.

a) At the simplest level "Gaussian plume" models, traditionally used for modelling dispersion at short distances, are extrapolated to follow longer estimated trajectories across the map area. Empirical formulae are used to describe average spreading about these trajectories.

b) "Lagrangian puff" models treat the release as a sequence of pollutant puffs each of which is tracked across the map area individually, with depletion and deposition determined according to assumed meteorological conditions along its path. Such models are simple yet flexible, but are generally limited in their treatment of a fully 3-dimensional windfield.

c) "Eulerian grid" models integrate the diffusion equation to simulate the dispersal of the release, considering the volume of air over the map area divided into a rectangular array of grid cells. These models can use 3-dimensional wind fields if suitable data exist, but are more complex and can be subject to numerical problems such as numerical diffusion.

d) Alternatively dispersion in 3-dimensional windfields can also be estimated by treating the release as an assembly of particles using "Monte-Carlo" or "particle-in-cell" techniques (in the Monte- Carlo methods each particle is followed independently through series of time steps using random walk techniques to simulate turbulent displacement relative to advection with mean winds. Particle-in- cell techniques also follow particles in a series of time steps, but advection velocities are compounded with effective diffusion velocities deduced from the concentration gradients to dilute the cloud. These diffusion velocities are determined from the particle densities in different cells of a rectangular grid at any time). Since large numbers of particles are required to represent dispersal of a radioactive release adequately, such models are again more demanding on computer resources than the simpler Lagrangian puff models.

Apart from calculating air concentrations during passage of the cloud, estimation of spatial distributions of deposited actitivty is also important. This is a more demanding task and not all general-purpose models include deposition, particularly wet deposition; although models designed for accident consequence assessment generally do. Ignoring deposition also reduces model utility to calculate air concentrations as the air packet depletion by deposition during transportation is important in this connection. Some models which have been developed for non-radioactive pollutants may not allow for radioactive decay, but are potentially suitable for treatment of radioactive releases with fairly minimal adjustments.

5.2 Improved dispersion model, project AKTU-230

Project *AKTU-230*, carried out in Denmark, has consisted of development of an advanced model for dispersion calculations. A so-called multipuff model has been develped, and the resulting computer program is called RIMPUFF (Ref. TH85). The project also included verification against experiments carried out across the Sound between Denmark and Sweden. RIMPUFF now also operational at Kernforschungszentrum Karlsruhe, Federal Republic of Germany, where it is one of two alternative short-range (up to about 50 km) atmospheric dispersion models incorporated in their new program system UFOMOD (Ref. PA89). A version of RIMPUFF is also in operation at the National Radiological Protection Board, United Kingdom. Project *AKTU-230* was completed in 1987, and documentation has been written throughout year 1989. The most pertinent references are (Ref. MI84, MI87, TH87, TH88a, TH88b and TH89).

5.2.1 Characteristics of the puff model RIMPUFF

In a puff model the release is simulated by a large number of individual puffs. In RIMPUFF these puffs are Gaussian-shaped. That means, that the concentration distribution inside the puff is Gaussian, like shown in Figure 5.1, part b), at the right-hand side, where the distribution is called "short-term average prediction". The individual puffs are symmetrical in all directions. In the same figure can be seen that a large number of puffs have been released. The time interval between release of two successive puffs is chosen so that they are frequent enough to give a satisfactory representation of the dispersion, but not so frequent that computer time expenditure becomes unreasonable. The amount of material in a puff is the release rate (of the release which shall be simulated) multiplied by the time elapsed between puffs; which means that longer time intervals between puff releases results in higher initial puff pollutant concentration than a shorter time interval.

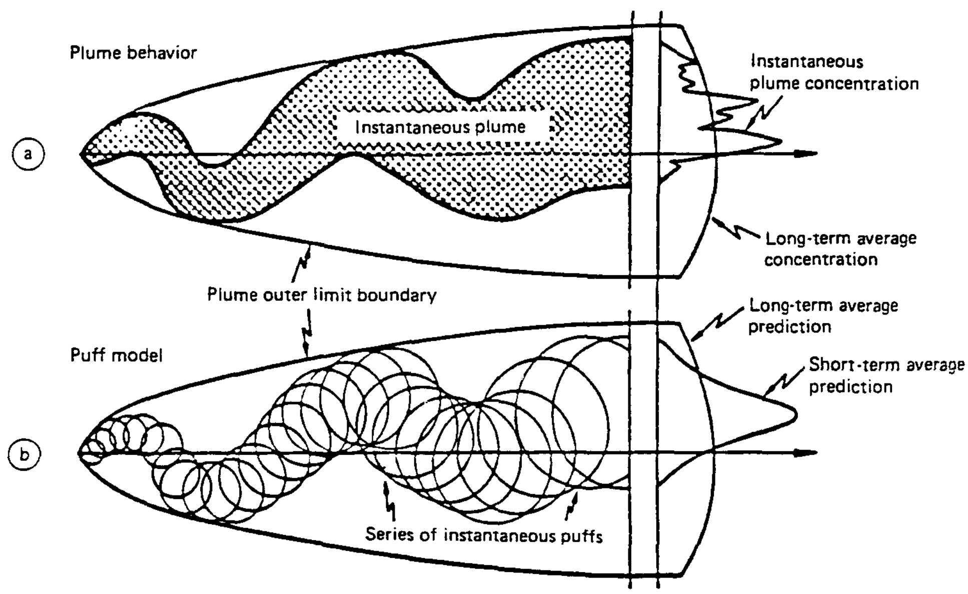


Figure 5.1 Instantaneous behaviour of a typical plume and a series of puffs from a puff-diffusion model (frem ref. RI82).

In the puff model it is possible to monitor multiple sources of puffs, and a puff source can be located anywhere on the grid and have a unique release rate, starting and stopping of release time and heat production. A source with time-dependent release rate can be simulated by a time-dependent function, or alternatively it can be approximated by placing a number of sources at the same location, each with different starting and stopping times and release rates.

Calculations take place in a grid, covering the area of interest. The movements of the individual puffs are followed in finite time steps over the grid, and they are advected by horizontal wind. The wind input may be either the measured wind from single points, a wind simulation, or a combination of both. At the same time the puffs grow in size, as time since release increases. The grid distances may vary from meters to kilometers. The model calculates the concentration at each grid point by summing the contributions from surrounding puffs for each advection step. When a puff leaves the grid, it is droppped from the calculation, and it is also possible to set a minimum grid concentration of interest, to reduce computer time. In Figure 5.1 is shown the shape of a plume, in part a), at a certain point in time, and the simulation of this plume by using a series of puffs, in part b). The "plume outer limit boundary" and "long-term average prediction" refer to the time integrated plume behaviour.

Wind shear, plume rise, reflection at ground surface or an inversion lid, dry and wet deposition can be taken into account. Both individual and collective doses can be calculated. Calculation of doses via the nutrition pathways is not incorporated.

RIMPUFF is flexible and has the advantage of being able to handle situations of large instationarity and inhomogeneity. The model is rapid, is operational on Burroughs and VAX computers, as well as on personal computer. Results can be output as a range of different types of graphic representations.

5.2.2 Model validation, simulation of the 1984-Øresund experiments

In 1984 tracer experiments using SF_6 were carried out over an inhomogenous region (land/water/land), across the Sound between Sweden and Denmark (Øresund), at distances typical of mesoscale. The tracer was in some experiments released on the Swedish side and in other experiments on the Danish side of the Sound. Two arcs of samplers were used on both sides; one close to the coast and one a few kilometers inland.

Nine tracer experiments were carried out. Of these two were insufficiently covered by the sampling positions, and one experiment was cancelled due to unfavorable weather conditions. The remaining six experiments have been simulated with the RIMPUFF model, for validation purposes. In the simulations the weather conditions measured at the release point was used to control the calculations over the Sound, and from there on the weather conditions from a station at the inland arc of samplers were used.

The agreement between simulated and measured concentrations and positions of the plumes was found to be good.

5.2.3 Use of RIMPUFF for calculations over complex terrain

In complex terrain atmospheric dispersion is influenced, not only by turbulent mixing and diffusion of the cloud itself, but also by topographically induced wind shear, channeling, local slope flows and hill-curvature speed-up effects etc. A technique has been developed whereby RIMPUFF in conjunction with a fast diagnostic mean-flow model LINCOM can be used for calculations over complex terrain. This technique has been applied in a climatological study of possible release hazards from the Vandenberg Air Force Base in California.

A typical situation in complex terrain is that a plume may split and pass on both sides of a terrain feature. In a puff model the whole puff will move in the direction in which the center of the puff would move. To better represent a situation like this, a technique has been incorporated in RIMPUFF where each puff splits into five smaller puffs when the puff reaches a certain size, and these five puffs may move in individual directions. RIMPUFF (combined with a flow model) has been validated using experiments performed at Vandenberg.

5.2.4 Use of RIMPUFF in ACA and in real-time

The RIMPUFF model can be used both in accident consequence assessments and in real-time mode. The Vandenberg Study mentioned in the preceding subchapter is an example of real-time application. The most recent applications in connection with accident consequence assessment is as a module in the German UFOMOD calculation system, where RIMPUFF is one of two alternative near range (<50 km) models built into the system. Furthermore a CEC project has started with the aim of creating a PC-based on-line version of RIMPUFF for use in emergency preparedness situations.

5.3 Atmospheric dispersion of particles, project AKTU-215

In Finland another aspect of atmospheric dispersion calculations has been improved. Modifications have been incorporated in the computer program ARANO, developed jointly by the Finnish Meteorological Institute and the Technical Research Centre of Finland, that make it possible to perform dispersion calculations for a release containing a spectrum of particle sizes. The development of the ARANO program has not been part of the AKTU program.

The characteristic aspect of the Finnish model is a special treatment of dispersion in the vertical direction. A model of this type is usually referred to as a K_z model; a term which is explained in the following.

In the traditional Gaussian dispersion models the source depletion approach is employed and it is assumed that the dry deposition does not affect the relative shape of the vertical concentration profile and the inventory in the plume is depleted with the same relative rate at all elevations. In the surface depletion approach, which is used in the ARANO code, the dry deposition is accounted for as a boundary condition in solving the partial differential equation describing the vertical diffusion. In short the employed method is referred to as the K_z-approach. To avoid unnecessarily long computer simulations a special data file structure is used in ARANO: complete data files describing the vertical distribution and depletion by dry deposition in the plume for several stability conditions and release heights are calculated once by a special code, and in actual ARANO simulations interpolations between the pre-calculated results are done. Similar data files are also produced for external cloud shine dose calculations.

References from project *AKTU-215* are (ref. NO86 and RO88f). The first reference deals with the theoretical studies when a K_z-approach is employed to describe atmospheric dispersion of particles with different sizes, especially those on which gravitation has an influence. The latter paper describes the incorporation of four different particle size categories in the consequence model ARANO. Effects on early health effects and long-term collective doses are also presented.

During dispersion radioactive matter of an airborne release is gradually removed from a plume by dry and possibly by wet depostion. Aerosols are removed by impaction on obstacles on the ground, by precipitation scavenging and by sedimentation under gravitation. For smaller particles and gases, the processes of impaction or precipitation scavenging are dominant. Wet and dry removal processes depend on several factors as for instance: characteristics of the released material, meteorological conditions, precipitation type, rate, particle size distribution, humidity and surfaces involved. A value of 0.01 m/s for the dry deposition velocity is usually employed in computer codes.

In co-operation with the Finnish Meteorological Institute (FMI), the ARANO computer program has been modified to make it possible to treat four different particle size categories. In the two largest particle size groups gravitational settling is also taken into account. The following sub-chapters describe the dependence of atmospheric dispersion on particle size and the subsequent effects on individual and collective doses. Finally the impacts on early and late health effects are studied, based upon several large, postulated "source terms". In particular the differences in results are studied, between the cases when a single value (of 0.01 m/s) or a spectrum of discretised values are used for the dry deposition velocities.

5.3.1 Model description and modifications

The ARANO model (ref. SA77 and NO79) is described here only as regards to the recent modifications. In the ARANO model the vertical mixing is described with a K_z-model (gradient transfer theory) and the crosswind dispersion with a Gaussian distribution. The plume scavenging is considered with the surface depletion model. The numerical solutions of the diffusion equations for different v_d-values have been calculated by the Finnish Meteorological Institute (FMI) into data files, that are subsequently utilized in the consequence code. Dispersion calculations for different particle sizes are described in detail in (ref. NO86).

The dispersion data files provided by FMI consist of vertical concentration profiles and fraction remaining in the plume for four release heights (20, 60, 120 and 180 m), three stability classes (stable, neutral and unstable) and four dry deposition velocity values (0.001, 0.01, 0.013, 0.02 m/s). A zero deposition value is applied for noble gases and other chemically less active gases. The two highest dry deposition values are the sums of a dry deposition value of 0.01 m/s plus gravitational settling velocities of 0.003 and 0.01 m/s representing particle sizes of 10 and 20 µm (aerodynamical diameter) respectively. A separate code (ref. VU87), which carries out the required 3-dimensional numerical integrations for calculation of normalized submersion gamma dose, is used to generate data files containing values for 4 release heights, 6 stability categories and 5 v_d-values. The ARANO consequence code interpolates from these files the relative air concentration and the normalized gamma dose values separately for each particle size category in the dispersion condition involved.

For utilization of the particle size categorisation, the released particle matter can be distributed into four classes at most. Each of these groups has one dry deposition value depending on the particle diameter. Assuming that radioactivity is proportional to mass, the release fractions of each group are determined by multiplying the total release fraction by mass fractions in each size class. Based on the results of the containment aerosol studies in severe light water reactor accident studies a most probable release discretisation may be calculated from a log-normal distribution. This distribution is fully determined by the mass median radius (r_m) and by the geometric standard deviation of the number distribution (σ).

5.3.2 Data for dispersion and dose calculations

Most of this sub-chapter describes the ways in which various types of input data to ACA and PCA calculations are typically handled by the ARANO computer program. These aspects are also described, perhaps more fully, in sub-chapters 12.1 and 13.1 of the present report.

5.3.2.1 Release and environmental data for the site

Calculations are performed employing meteorological statistics and demographic characteristics applicable to the Loviisa nuclear power plant site. There are two VVER-440 type reactors each of 465 MW output capacity, on which the employed radioactivity inventory is based. The release fractions of the core inventory for each assumed release category are presented in Table 5.1. The duration of each release is assu-

Release category	Time of release (h)	Xe-Kr	Nuclide groups			
			I	Cs-Rb	Te-Sb	Ba-Sr-Ru-La*
FK2(ref. RI83)	10	1	$6.4\cdot10^{-3}$	$6.9\cdot10^{-3}$	$5.6\cdot10^{-3}$	$6.9\cdot10^{-5}$
A	0	1	0.1	0.1	0.1	0.01
B	0	1	0.01	0.01	0.01	0.001
C	0	1	0.001	0.001	0.001	0.0001
A_{mod}	10	1	0.1	0.1	0.1	0.01

* includes also Rh, Co, Mo, Tc, Y, Zr, Nb, Ce, Pr, Nd, Np, Pu, Am, Cm

Table 5.1 Fraction of the reactor core radioactivity inventory released to the environment in the considered reference accidents.

med to be 3 hours and the σ_y-values are corrected to correspond to this value. The release height is 20 meters and no plume buoyancy is assumed. For one release case the effective release height of 100 m was used in two dispersion conditions to illustrate variation in individual doses.

The discretisation of the log-normal distributions used for the releases are presented in Table 5.2. A value of 4 gcm^{-3} was adopted for particle density. Because the r_m-value is strongly dependent on the reactor accident sequence and containment failure type, it should not be strictly connected to the release categories considered in this study. The r_m-values of 0.6 and 1.0 µm with a standard deviation of 2.0 are used.

dr(µm)	v_d(cm/s)	f(r_m=0.6µm)	f(r_m=1.0µm)
< 1.3	0.1	0.84	0.63
1.3-2.4	1.0	0.14	0.28
2.4-3.8	1.3	0.017	0.07
> 3.8	2.0	0.003	0.02

Table 5.2 Discretised log-normal particle size distribution employed in calculations. Fraction (f) of total mass within stated radius range. Mass median radius (r_m), geometric standard deviation of the number distribution σ = 2.0.

Site specific weather data corresponding to one whole year were used to generate the probability distributions. The weather data are given in 12 sectors for wind direction, Pasquill stability categories A - F, seven wind speed classes and rain frequencies. The washout coefficient values assumed in the consequence model are $5 \cdot 10^{-3}$ s^{-1} for unstable and $7 \cdot 10^{-5}$ s^{-1} for stable conditions. When collective doses are calculated, the population and agricultural production statistics up to 100 kilometers around the Loviisa site are used, and thereafter average values of southern Finland are employed up to the distance of 250 km in all dispersion directions.

5.3.2.2 Data for dose calculations

Early health effects are calculated on the basis of the bone marrow dose from cloud shine, ground shine and inhalation. The particle size variation is assumed not to affect inhalation dose. The ground shine dose is integrated until the assumed evacuation timepoint at 8 hours up to 20 km and at 24 hours at longer distances. The population is divided into two shielding factor groups, of which the first one is assumed to be out of doors and the second one indoors when a plume is passing. Both groups spend on an average 10% of the time out of doors and the rest of the time in houses. The inhalation dose is integrated fully for the seven first days and half of the dose between the 8th and 30th days is taken into account.

The early health effects assessed are: radiation illness cases and early fatalities. The bone marrow dose threshold of 1 Gy has been employed for assessment of the number of radiation illness cases. The number of early fatalities is calculated with a dose response relationship in which a bone marrow dose of 2 to 5 Gy causes the probability of early fatalities to increase from 0 to 1. The relevant data are presented in Table 5.3.

Long term collective doses are calculated on the basis of exposure via the cloud shine, inhalation, ground shine and ingestion pathways. The ingestion pathways include milk and meat from cattle, green vegetables, grain and root vegetables. The population is assumed to consume

	Early health effects Group 1	Group 2	Long-term doses
Fraction of population	15%	85%	100%
Shielding factor for direct radiation from			
- plume	1.0	0.6	0.5
- ground	0.4	0.4	0.25
Integration time for ground shine exposure	8 h; 0-20 km 24 h; 20-100 km		24 h; 0-250 km, thereafter relocation criterion 0.1 Sv/30 a
Breathing rate	10 m³/8 h		20 m³/24 h
Individual food consumption (kg/a) for intervention criteria			
- milk			250
- meat			35
- green vegetables			35
- grain			70
- roots			30

Table 5.3 Shielding factors, inhalation rate and individual consumption of foodstuffs employed in intervention considerations.

the food products cultivated in the fallout area. The interdiction criterion for the food pathways is (0.1 Sv/30 a) for each pathway. The ground shine dose is integrated fully for the first 24 hours, but thereafter a population relocation criterion of (0.1 Sv/30 a) is employed. Because the collective dose is accumulated from a large area, the effect of the better shielding of multistorey houses is taken account. The relevant data are gathered in Table 5.3. Latent fatal cancers are calculated by multiplying the total collective dose with a risk factor of 0.01/manSv.

5.3.3 Results

The effects of different particle sizes (and corresponding deposition velocities) are considered regarding both atmospheric dispersion and individual doses. In addition the effect of two different choices for particle size distribution upon early health effects as well as long term collective doses is presented. The probability distributions are conditional; it is assumed that the release has taken place.

5.3.3.1 Effect of particle size on dispersion and individual doses

Relative ground level air concentrations for four dry deposition velocities at plume centerline as a function of distance are shown in Figures 5.2 and 5.3. Two stability conditions are considered and the release altitute of 20 m is used.

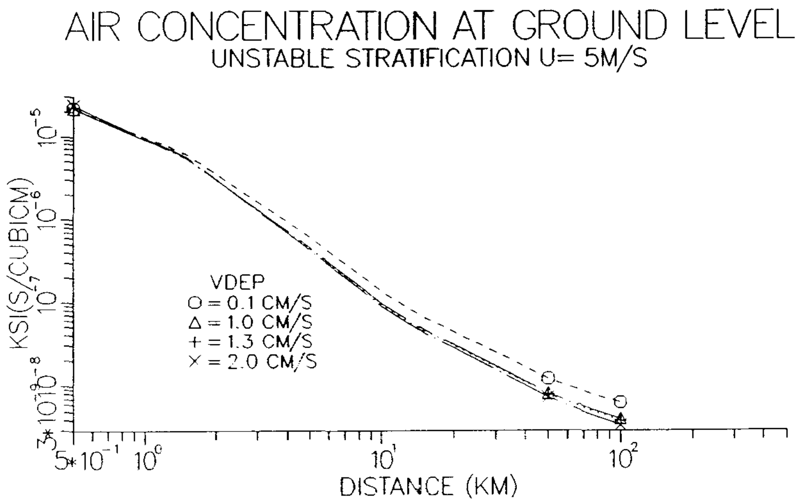


Figure 5.2 Relative ground level air concentration (KSI;s/m³) as a function of distance for particles with different dry deposition values. Stability B, wind speed 5 m/s, release height 20 m.

Differences in ground level concentrations between various particle sizes are clearly largest in stable weather conditions, where the differences between the largest particle size and the smaller sizes are several orders of magnitude. For the largest particle size gravitational settling is significant. Variation in ground concentrations (fall-out) as a function of deposition velocity can be read from the same curves, as the ground concentrations will be proportional to ground level air concentrations. Table 5.2 shows what particular particle size range each of the deposition velocities in Figures 5.2, 5.3, 5.4 and 5.5 correspond to.

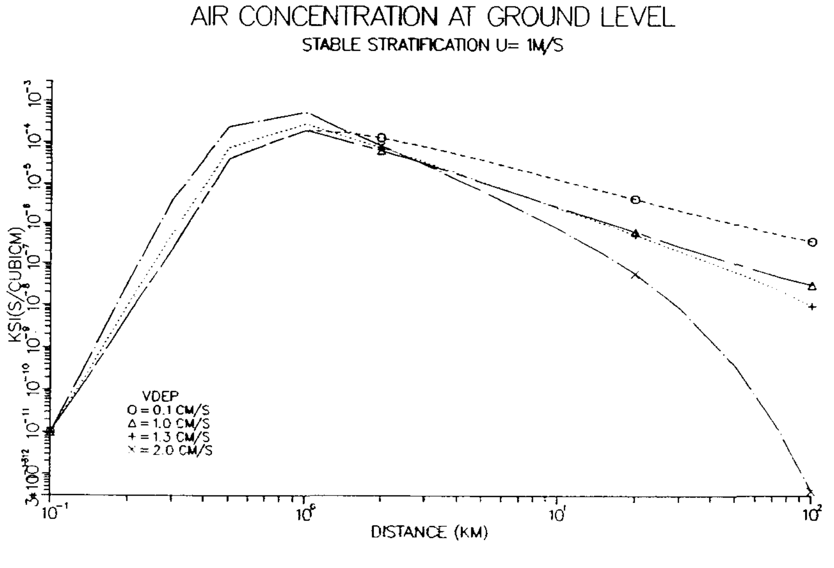


Figure 5.3 As in Figure 5.2, but stability F and wind speed 1 m/s.

Figures 5.4 and 5.5 show the results of calculations of total dose (sum of doses via the inhalation, cloud shine and ground shine pathways, as described in subchapter 5.3.2.2) as function of distance from release point. The doses are effective dose equivalents. The release in this case was assumed to be the revised FK2 release (ref. RI83) (Table 5.1), the release height of 20 m was used, and the calculations were performed for two different weather conditions. Calculations were performed for four different deposition velocities, for comparison.

Here no shielding is assumed against external radiation, and the inhalation dose is calculated employing dose factors of 50 years integration time. Noble gases have a strong effect on the dose commitment due to the release fraction of 100%. In neutral weather conditions (stability D) the results do not vary much with deposition velocity, whereas in stable weather conditions (stability F) the largest difference is nearly an order of magnitude.

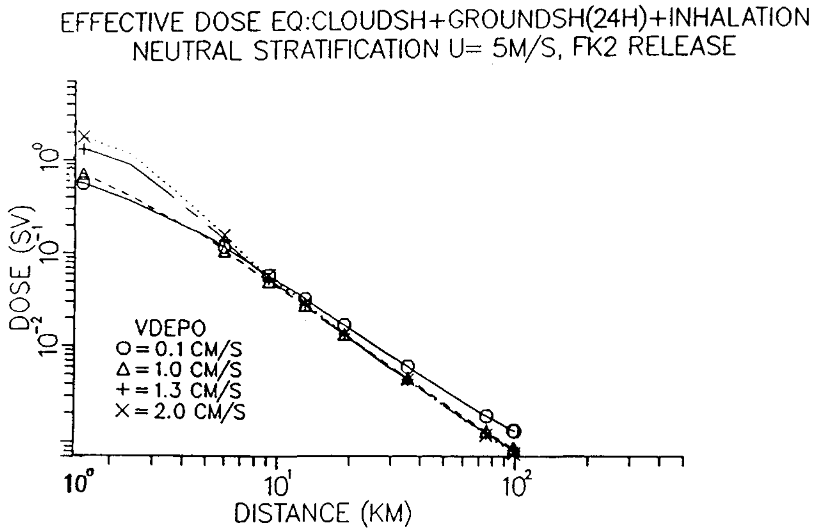


Figure 5.4 Total individual dose as a function of distance due to the revised release FK2, for particles with different dry deposition values. Stability D, wind speed 5 m/s, release height 20 m. Shielding is not taken into account.

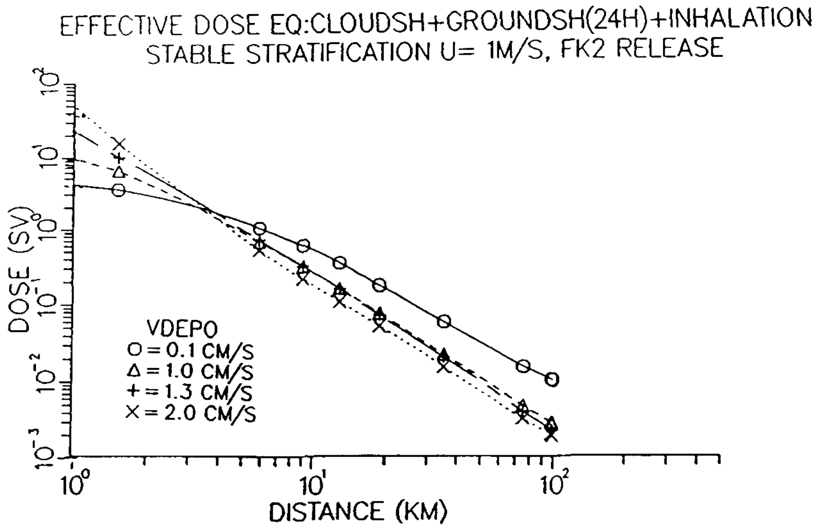


Figure 5.5 As in Figure 5.4, but F; 1 m/s.

5.3.3.2 Early health effects

Figures 5.6 to 5.9 shows the variation of the magnitude of early health effects in the form of complementary cumulative probability distribution functions separately for the releases A, B, C and A_{mod} employing for the dry deposition velocity either two discretised distributions or a single value of 0.01 m/s. The corresponding expectation and peak values are presented in Table 5.4.

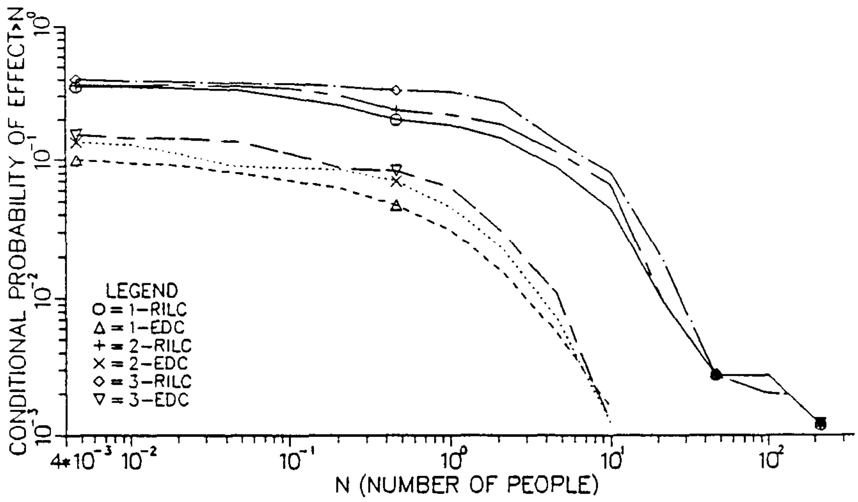


Figure 5.6 Complementary cumulative probability distribution for radiation illness cases (RILC) and early fatalities (EDC) due to the release A. Curves marked 1 are for particle distribution discretisation with r_m = 0.6 μm, curves marked 2 correspond to r_m = 1.0 μm, and curves marked 3 correspond to a deposition velocity of 0.01 m/s.

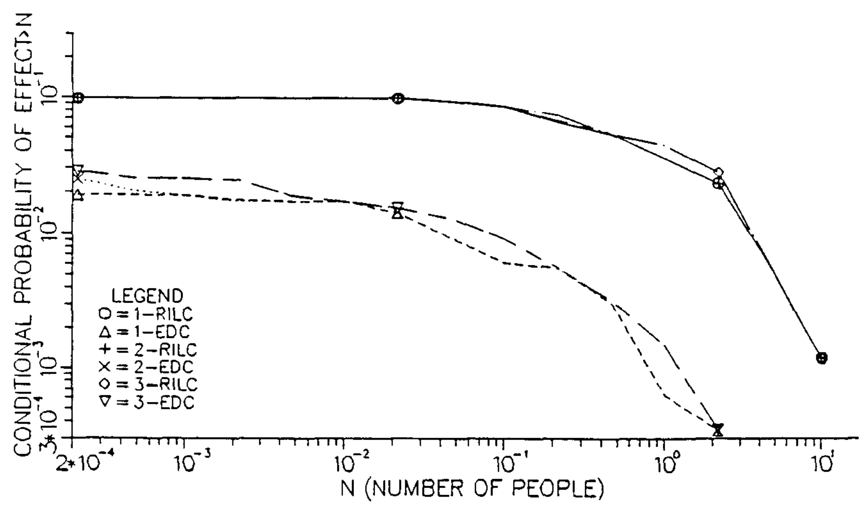


Figure 5.7 As in Figure 5.6, but due to the release B.

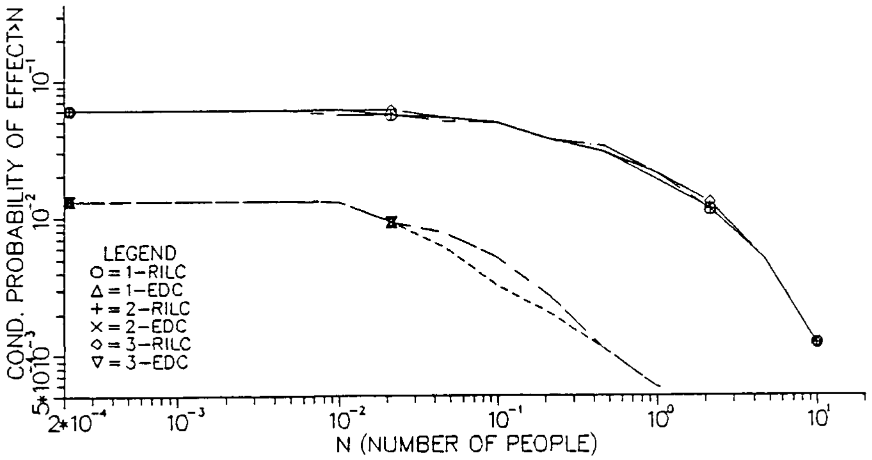


Figure 5.8 Complementary cumulative probability distribution for radiation illness cases (RILC) and early fatalities (EDC) due to the release C. Curves marked 1 are for particle distribution discretisation with r_m = 0.6 µm, curves marked 2 correspond to r_m = 1.0 µm, and curves marked 3 correspond to a deposition velocity of 0.01 m/s.

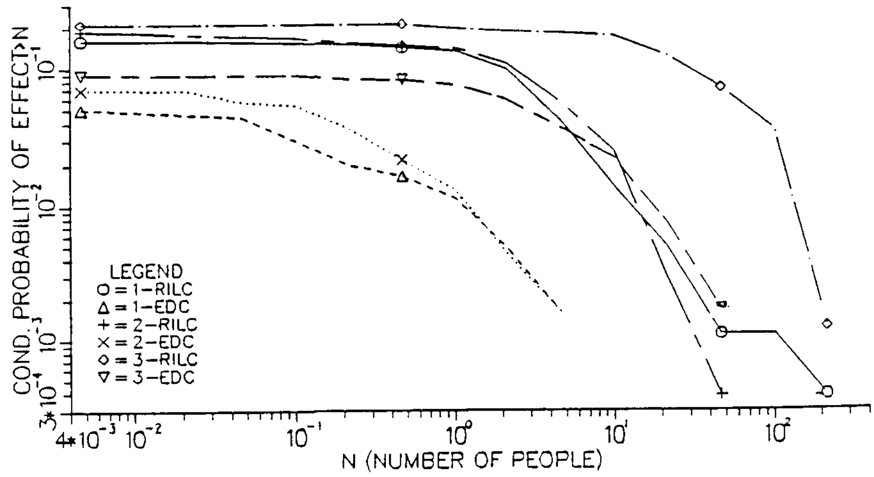


Figure 5.9 As in Figure 5.8, but due to the release A_{mod}.

The differences in early health effects between the different particle size distributions and the case where a single value for deposition velocity is employed, due to the releases B and C, seem to be insignificant. This is because the total dose is dominated by the noble gases. However, in the case of the release A, a factor of nearly 2 is found in probabilities of early fatalities and radiation illness cases. The delay of 10 hours assumed for the release in Figure 5.9 increases the differences between the particle size distributions and the case where a single value for deposition velocity is employed. This increased variation is caused by the decreasing contribution due to cloud shine (which is mostly caused by noble gases). The expectation values of early health effects are largest for the largest r_m-value, and even larger for the deposition velocity value of 0.01 m/s, which is the value generally used in models at present. The maximum values behave conversely. These results are in accordance with those presented for the non-buoyant plume in (ref. NI87).

r_m(μm) or v_d(m/s)	Radiation illness cases for release categories				Early fatalities for release categories			
	A	B	C	A_{mod}	A	B	C	A_{mod}
r_m = 0.6	2.12 (450)	0.14 (15)	0.08 (13)	0.86 (295)	0.11 (14)	0 (3)	0 (2)	0.04 (10)
r_m = 1.0	2.35 (390)	0.15 (15)	0.08 (13)	0.90 (250)	0.15 (14)	0.01 (3)	0 (2)	0.05 (10)
v_d = 0.01	2.73 (251)	0.16 (15)	0.08 (13)	0.96 (30)	0.20 (14)	0.01 (3)	0 (2)	0.07 (10)

Table 5.4 Conditional expectation values of early health effects and maximum values (in parentheses) for two discretised log-normal distributions and for v_d = 0.01 m/s due to release categories A, B, C and A_{mod}.

5.3.3.3 Long-term collective doses and late health effects

Table 5.5 contains the expectation values of fatal cancers and peak values obtained by applying a risk factor of 0.01/manSv to the estimated total long-term collective doses caused by the releases A and C.

Differences in results between the two distributions and a single v_d-value are unimportant as regards the release C. Also for the release A there seems to be a minor difference in expected values. However, a difference by a factor of 2 in peak values exists between the two distributions and a single value for v_d. Both the expectation value of of fatal cancers and the peak values are decreased when r_m-value is increased. Also here a similarity is found when compared to the results of (ref. NI87).

r_m(μm) or v_d(m/s)	Expected fatal cancers for release categories	
	A	C
r_m = 0.6	320 (2510)	7 (47)
r_m = 1.0	300 (2160)	7 (46)
v_d = 0.01	250 (1280)	7 (45)

Table 5.5 Conditional expectation values of late health effects (latent fatal cancers) and maximum values (in parentheses) for two discretised log-normal distributions and for v_d = 0.01 m/s for the release categories A and C.

5.3.4 Conclusions about impact of particles size distribution

The consequence model ARANO has been developed to include a possibility for considering four different particle size categories. The employed particle size categorisation has been determined on the basis of the preliminary dispersion calculation results and recent aerosol studies for containment conditions during a severe light water reactor accident. Especially modelling of the gravitational settling of larger particles has been incorporated in the code.

Calculations indicate that in stable weather conditions there may be a difference of orders of magnitude in consequence values, such as air concentration, fallout and doses, corresponding to different particle sizes. When dispersion conditions are changing towards more unstable stratification, differences in consequences between particle sizes decrease.

The dry deposition value of 0.01 m/s generally used in computer codes seems to result in an overestimate of less than a factor two for expectation value of early health effects, compared to results calculated on the basis of release categorisation into different particle size groups. However, this depends on the release magnitude. For the smaller releases calculated in this project cloud shine exposure from noble gases dominate the doses. This is because the release of noble gases is assumed to be 100% for all the releases. There is accordingly no difference between the different particle sizes distributions.

On the other hand the use of the dry deposition value of 0.01 m/s instead of particle size distributions results in underestimation of late health effects in the case of the large release. For smaller releases the differences are insignificant.

The maximum values of early as well as late health effects are underestimated with a factor of up to two, when the dry deposition value of 0.01 m/s instead of the release categorisation is used. In the case of small releases there is no difference in peak values.

In this context one should emphasize that numerous other factors may have interrelated influences on consequences, when effects of different particle sizes are studied. For instance one can refer to magnitude of source term, composition of the release, release height and countermeasures such as evacuation, criteria for relocation and interdiction.

5.3.5 Correction factors for other codes

The revised methodology to account for the more realistic description of the dry deposition process (by K_z-model approach) and different particle sizes (i.e. different v_d-values) can be transferred for use in other consequence models in approximate manner by deriving appropriate correction factors. Because ARANO applies a data file structure for vertical concentration profiles at different dispersion times for a number of discrete release heights and stability categories and similar data for cloud depletion factors and normalized cloud gamma doses, the other code, where one wishes to use correction factors, should be slightly modified accordingly. The following steps have to be followed in deriving correction factors corresponding to either the complete contents of VTT's data files or only individual dispersion conditions:

a)
VTT gives detailed specifications of the contents, structure and formats of the pertinent data files (parameters include release height, stability, wind speed class, dispersion times (distances), deposition velocity).

b)
The laboratory requesting correction factors calculates the ground level value for vertical concentration profile, cloud depletion factor and normalized cloud gamma dose employing same parameter specification as above using their own code.

c)
VTT derives correction factors (i.e. ratios between code predictions) and supplies it back in the same data file format.

5.4 Long duration releases

Originally it was planned to include in project *AKTU-215*, in addition to studying the importance of particle size on dispersion, studies of wind shear and of the effect upon doses of long duration releases. Proposal for studying this aspect originated in Finland, and were to be performed there, in cooperation between the Finnish Meteorological Institute and Technical Research Centre of Finland. However, partially due to the additional work-load after Chernobyl, and partially due to the fact that the particle size task was more time-consuming than originally anticipated, the task concerned with wind shear within AKTU-215 had to be abandoned; while work on the long duration release aspect was reduced to preparation of an overview of the subject matter (ref. NO89), of which the following is a translation:

5.4.1 In general about long duration releases

In the beginning of the 1980's an intercomparison of ACA (Accident Consequence Assessment) models was performed under the auspicies of Organisation for Economic Co-operation and Development, Nuclear Energy Agency (OECD/NEA) (ref. OE84). In these comparisons release durations of 1 hour and 10 hours respectively were assumed. For the 1 hour release dispersion was treated roughly identically by the participating institutions. However, for the release of 10 hours duration, there was no generally accepted method for calculating the horizontal dispersion. Several of the participants did not take the longer release duration into consideration, while those who did, did so by introducing a correction of the σ_y values to account for the increased horizontal spread.

The OECD/NEA intercomparison shows that one is not sure about the best approach to calculating dispersion for releases of more than a couple of hours duration. These difficulties were clearly exposed even when attempting to calculate a release of 10 hours duration, and as shown by the Chernobyl accident, it is quite realistic to expect release durations of several days. How to treat this problem, is a question upon which too little effort has been spent till now.

5.4.2 Dispersion for instantaneous and continuous releases

A really instantaneous release hardly exists, and even dispersion experiments that really represent instantaneous releases are rare. Based upon diffusion theory it has been shown that dispersion of an instantaneous release (a puff) ought to be proportional to the time (t) since the time of release, at small distances from the release. At longer distances it should be proportional to $t^{2/3}$, and at distances where the puff has grown to be larger than the turbulence, it should, according to theory, be proportional to $t^{1/2}$ at larger distances. For continuous releases the theory shows that dispersion should be proportional to t near the release point, and proportional to $t^{1/2}$ further out (ref. HA82).

The above should be valid if diffusion only took place on the turbulent scale. Diffusion on the synoptic scale, diffusion caused by inhomogeneities in the wind field, cause an additional horizontal spread. For long sampling times, months or years, it is possible to estimate the horizontal synoptic scale diffusion by using trajectory statistics (ref. BO75) or simply by assuming that the concentration distributions are homogenous within certain sectors. The width of the sectors is then widths determined by the discretization of the wind directions; e.g. 30^0, if twelve directions are used. If the sampling time is from several days to several weeks, it can not be assumed that the different synoptic situations should be represented according to their climatological frequencies of distribution.

As neither statistics nor class division can be used to calculate diffusion on the synoptic scale for time intervals from some days to some weeks, a handy approach is to divide the release into "puffs". The spread in the trajectories of the puffs then give an approximation of the diffusion on the synoptic scale. However, since the interval between puffs possibly can not be made sufficiently short, one can lose some of the diffusion; namely the portion that takes place in the puff intervals. In the Danish RIMPUFF model (project AKTU-230) horizontal diffusion is calculated for a sampling time equivalent to the time interval between the puffs according to (ref. MI84)

$$\sigma_y = 0.22 \, bx$$

Here x is distance from the release point, and b is the turbulence intensity, which is determined from the wind variation over the sampling time. This approach, to take diffusion on both the turbulence and the synoptic scales into consideration, can be used if one has continous measurements of wind variation over the whole area in which transportation takes place. While a synoptic diffusion part is integrated in the turbulence intensity (b), it is nevertheless not possible to determine values for b based upon routine synoptic weather observations.

Empirical values of puff diffusion over large distances are relatively rare. Thus, plume σ_y values have frequently been used also for puff models. If the empirical plume σ_y values are from measurements with sampling times of several minutes or more, they will be too large to represent the horizontal spread of a puff. From instantaneous photographs of plumes it is possible to estimate σ_y values that are applicable also for puff diffusion. However, photographs showing plumes at more than 100 km from the source are almost non-existant.

A possible approach to the problem of diffusion on the synoptic scale, is to calculate trajectories with a certain time interval, and assume that the spread of the trajectories gives the diffusion on the synoptic scale. In order not to lose the portion of meandering that takes place within the trajectory interval, it is possible e.g. to assume that the diffusion due to meandering would cover the area between the trajectories. This method has been used in the Finnish model TRADOS (ref. KA87c). If the trajectories are reasonably close together, this assumption is probably quite realistic. If, however, the trajectories go in quite different directions, it may well be that part of the area between is in reality not hit by any part of the release. By reducing the time interval between the trajectories, additional knowledge of the situation is of course gained.

One advantage of the trajectory method compared to the puff method is that it is easier to estimate diffusion on the synoptic scale from the spread in trajectories rather than from the spread of the single puffs. The use of empirical σ-values for plumes in a trajectory model is also better founded than the use of theoretically estimated σ-values in a puff model. In a puff model it is easier, on the other hand, to take into account rapid changes in the transporting wind.

5.4.3 Conclusions concerning long duration releases

The subchapters above contain nothing new for persons actively engaged in the areas of diffusion theory and dispersion calculations. When results of dispersion calculations shall be applied in a wider context, however, it is useful to be aware of the basic facts contained in these subchapters.

The choice of the approach to estimation of horizontal diffusion for long duration releases depends upon the purpose of the dispersion calculations. If the dispersion calculations are performed as part of a risk analysis it is important to obtain a proper frequency distribution of the consequences resulting from the different dispersion situations. If the release is assumed to last for e.g. five days, it is possible to obtain an acceptable estimate of the frequency distribution of the radiation dose at a certain position, by carrying out the dispersion and radiation exposure calculations over a large number of five-day periods. Eventual over- or under-estimation of the dose exposures in each single dispersion situation will have little impact upon the frequency distribution.

If the dispersion calculations are performed in order to estimate the doses resulting from specific radioactive releases of short duration, the largest difficulty is actually to have access to weather data of sufficient time- and space resolution over the area of interest. If sufficient meteorological data are available, there are mostly no difficulties in finding a model to estimate the dispersion with sufficient accuracy.

The uncertainties in the dispersion models due to their mathematical/ physical design is generally of less importance than the uncertainties that are due to inexact data on weather, and on release characteristics. If the dispersion model is linked to an advanced numerical prognostic weather model, the difficulties of obtaining weather data can to a certain extent be eliminated. In this case the advanced numerical model ought to have a mesh sufficiently fine to satisfy the demands for accuracy of the dispersion calculation. In the dispersion model that the Nordic meteorological institutes proposed to construct (project AKTU-232), it was planned to use the Nordic fine-scale prognostic weather model HIRLAM, for obtaining the needed analyzed and forecasted weather data.

5.5 Dispersion over long distances and emergency preparedness, project AKTU-232

The dispersal and fallout of the radioactive release from the reactor accident at Chernobyl demonstrated the importance of good and reliable calculational tools for long-range atmospheric transport. In September 1986, as a direct result of the impact of the Chernobyl accident upon the Nordic countries, a meeting was called of the informal Nordic group SNODAS, mentioned in chapter 2 of the present report. At this meeting, held at Kjeller, Norway, representatives of all four Nordic countries expressed interest in development of an inter-Nordic model for this purpose.

The next step leading up to initiation of project *AKTU-232* was taken at a meeting of the Directors of the Meteorological Institutes of the four Nordic countries, where the Directors proposed that a common Nordic meteorological model for use in emergency preparedness in situations involving radioactive releases ought to be developed. A contact group, with one member from each of the countries, was established, and the Norwegian delegate was given the task of seeking the needed financial support from the Nordic Council of Ministers and other organizations engaged in environmental protection.

This contact group applied for funds for carrying out a pre-project as a project in the AKTU program area. The application was submitted at the yearly AKTU seminar in November 1986, and the application was approved by the AKT Steering Group. The primary purpose of the project was to prepare an application to the Nordic Council of Ministers. The contact group also expressed interest in continued association with the AKTU program, even after eventual approval of their project proposal to the Council.

5.5.1 Status by March 1987

The contact group held its first meeting in the beginning of March 1987, where they discussed international emergency preparedness and models for atmospheric transportation of radioactive materials.

Concerning emergency preparedness the group observed that the Vienna Convention on Early Notification of a Nuclear Accident had just been established, in which the duties of a country in case of an accident are described. It is described in the Convention in some detail what type of information shall be given, including actual and predicted meteorological and hydrological conditions needed to predict atmospheric transportation of the radioactive materials across international boundaries. The World Meteorological Organisation (WMO) also proposed that the information concerning radiological and meteorological aspects should be sent via the meteorological telecommunication system GTS. The contact group also observed that a "point of contact" with the International Atomic Energy Agency (IAEA) shall exist in each country. The official warning and information shall be sent from there to the IAEA, from where it will be sent to the other countries.

Concerning atmospheric transportation of radioactive materials, the group referred to three initiatives that had been taken; the first two described in (ref. TV87a):

a) A Nordic project to validate models for long-range atmospheric transportation, using Chernobyl data for validation.

b) An international project for the same purpose.

c) WMO and IAEA has decided to carry out a cooperation project referred to as "Establishment of a data base for the validation of radionuclide transport models and subsequent preparation of a technical report on the modelling of transport of radionuclides".

The first two of these initiatives had to be abandoned (before initiation of project AKTU-232), as the funds applied for from the Nordic Council of Ministers were not available. Both initiatives fell as a result of this, since the plan was to perform them in sequence.

The group suggested continued activity in the Nordic countries in the following areas, of concern to the meteorological institutes:

Distribution within each country of accident information received over GTS at the meteorological institutes.

Accident information and prognoses of dispersion and deposition of radioactive materials shall immediately be communicated to the national "point of contact" and other institutions, according to guidelines supplied by the national authorities.

Prognostic model calculations of dispersion and deposition of radioactive materials.

According to the group the most suitable system for predicting dispersion and deposition over the Nordic countries could be obtained by constructing a dispersion model based upon the Nordic fine-scale numerical weather prognosis model HIRLAM (which was under development). The group suggested that this effort should be undertaken as a Nordic cooperation project, and that financial support should be sought from the Nordic Council of Ministers. A project proposal to accompany this application would be put together by the group in the course of 1987.

The group made a further proposal: That in the case of an accident, dispersion and deposition calculations ought to be performed by as many Nordic countries as possible, and the results are then to be exchanged.

Network of measurement stations for "early warning", and Nordic system for rapid distribution of information on increased levels of radioactivity.

The group pointed out that an "early warning" system should react even to quite low levels of radioactivity of artificial origin. But it also stressed the importance of the ability of the system to identify and not react to variations in natural radioactivity levels.

The group stressed the importance of rapid information exchange; both of measured radioactivity levels and of the results of dispersion calculations. The meteorological institute ought to receive information on increased levels of radioactivity from the national radiation protection authority or other relevant institution as soon as these are received, in order to initiate dispersion calculations. And information on weather and on the results of dispersion calculations should be transmitted from the meteorological institute to the national radiation protection authorities as soon as possible.

The group expressed the opinion that information to mass media were most conveniently handled by the radiation protection authorities or some other national authority or institution responsible for radiation measurements, and not by the meteorological institutes.

The line of action for the contact group was approved by the Directors of the Nordic meteorological institutes at their meeting in Helsinki in late August 1987.

5.5.2 Status by November 1987

At the second meeting of the contact group, held in the end of November 1987, the following items were discussed:

Project proposal concerning development of a Nordic prognostic model for dispersion and deposition of radioactive materials.

A project proposal had been written by the Norwegian delegate, and was approved by the group with some minor adjustments.

Evaluation of two project proposals submitted to the AKT Steering Group.

The AKT Steering Group had received two project proposals within the AKTU program area, and had asked the AKTU-232 contact group to give an evaluation of the proposals, as they were concerned with subjects of major interest to the members of the contact group.

One proposal had been submitted by Studsvik, Sweden, and was titled: "Development and evaluation of a puff model, based on SODAR data, for adjustment to meso scale calculations." The group was quite positive to the idea of using data from the Øresund experiment (an experiment that was performed a few years earlier, across the sound between Denmark and Southern Sweden) to validate dispersion models. The proposal, however, suggested development of a model on the European scale, partially based upon a network of SODARs. The group doubted that the future SODAR network would cover areas of this size. For this reason and the additional reason, that SODARs only rarely gives information about atmospheric conditions at levels above 500 meters, the group was somewhat sceptical to the proposal.

The second proposal had been submitted by Risø National Laboratory and the Meteorological Institute of Denmark, and was titled: "Regional scale dispersion model." In the groups opinion this proposal was quite close to the proposal the group itself was working at. The group was positive to the proposal, but believed that the financial resources required were somewhat underestimated. The group proposed that the Danish proposal and the Nordic proposal could support each other, in particular because Risø National Laboratory will then also be brought into the cooperation between the Nordic meteorological institutes in this connection. The group, however, stressed that they did not wish the Nordic project to be obliged to base their development work on the puff model concept, assumed in the Danish proposal, at such an early stage in the planning.

IAEA/WMO project for testing of dispersion models for radioactive materials.

IAEA and WMO are organizing a three year project to test dispersion models for radioactive materials against various data; primarily Chernobyl data. The USSR has indicated that they are willing to put their radiological data at disposal for the project. A data base for such data is being compiled. The Commission of the European Communities has expressed willingness to support the IAEA/WMO project.

The plans of IAEA/WMO of distributing accident warnings on the WMO telecommunication network GTS.

The Norwegian delegate informed the group that the cooperation between the two organizations was in the process of being established (an aim which has later been accomplished).

Cooperation between the Nordic meteorological institutes and other relevant Nordic institutions.

In cooperation with the national authorities, and in order to improve emergency preparedness, the meteorological institutes are interested in making dispersion progonses of radioactive materials available for other relevant institutions in the Nordic countries; in particular to nuclear industry, to institutions involved in the radiation surveillance programs and the authorities responsible for evaluating mitigating actions.

The meteorological institutes also believe that total emergency preparedness would be enhanced if observations of increased levels of radioactivity could rapidly be communicated to these institutes from the institutions making the observations. Efforts to identify the point of release and prediction of the future development of the situation could then be initiated with minimum delay. The group pointed out that cooperation of this sort was not actually operating, and that there was a need for evaluating the situation and identifying means of carrying out such cooperation, involving the meteorological institutes as well as numerous other institutions in the Nordic countries.

5.5.3 Termination of project AKTU-232

As planned, the project led to submittal of an extensive project proposal to the Nordic Council of Ministers. The project, however, ended rather abruptly, when funds were not granted. Work within the contact group was terminated and no further meetings have been held beyond the two referred to in this subchapter.

6. DEPOSITION AND WEATHERING, PROJECT AKTU-245

One of the AKTU projects has been concerned with experimental investigations of many different aspects of deposition and weathering, and various other phenomena. This project, *AKTU-245*, had the largest budget of the AKTU projects. The project was run by Risø National Laboratory, but some experiments were also performed at the Institutt for energiteknikk in Norway. Studsvik, Sweden joined the project in 1988.

A separate report about this project will be published in the NORD report series (ref. R090); the same report series as the one in which the present report is published.

6.1 About urban and rural areas

The initial aim of the *AKTU-245* project was to make representation of the conditions in urban areas more realistic. Urban areas are of particular importance, because most of the population lives in such areas, even in the Nordic countries. Nevertheless, at least in an ACA context urban areas were completely overlooked, until some ten years ago. Calculations in urban areas were performed using parameter values (e.g. for deposition) for rural areas. Investigation of various aspects of urban areas were initiated at Risø National Laboratory in the early 1980s, and somewhat later also received support via the Nordic Safety Program, during the program period 1981 - 1985. In the present program period the work concerned with these problems has been significantly extended. This is also in part due to the Chernobyl accident. Along with all its negative aspects, the accident has presented an opportunity for finding out more about the behaviour of radionuclides in the environment. For this reason the accident has had impact upon many of the AKTU projects.

Ecology is, as a rule, not concerned with conditions in urban areas, just as if these are not to be regarded as part of the environment. Certainly urban and rural areas are very different, but urban areas also have their own ecology; even involving plant life. And, as mentioned before, urban areas are particularly important, since the major part of the population dose, at least via the non-nutrition exposure pathways, comes from these areas.

6.2 Tracers used in the experiments

The tracer used in these experiments has changed over the time period of the project. Initially cesium from weapons fall-out was used. As measurement techniques were improved, Be-7 (a radioactive nuclide continuously produced in the upper atmosphere) was used as tracer. Since late April 1986 the Chernobyl fall-out has been the obvious choice, and it has been possible to measure a number of different nuclides, although work has been concentrated around I-131 (to a smaller extent), Cs-134 and Cs-137.

6.3 Dry deposition

Dry deposition in an urban area may be dealt with in two ways:

In the first approach, the total urban area is considered as a rough surface with the buildings providing the roughness elements, as basis for estimation of the total deposition velocity to the area. This approach has serious disadvantages because the source distribution for the dry deposition of radionuclides to roofs, walls and areas surrounding the structure (e.g. streets and gardens) and deposits to building interiors due to aerosol and gas infiltration are not included. Knowledge of this distribution pattern is necessary for consequence calculations.

The second approach is to define local deposition velocities on the different surface elements and by integration to find the total deposition velocity. This integration, however, is not straightforward, as it calls for the definition of surface elements which are representative of the urban area and then individual deposition velocity for each surface. The total horizontal projected flux per unit area must then be calculated. The total deposition velocity is found from this flux and a knowledge of the air concentration above the urban canopy.

Deposition on many different types of surfaces typical of urban areas has been measured in project *AKTU-245*, as well as on grass, bare soil and a forest area. The results have been reported in a number of publications, e.g. (Ref. GJ86, RO85b, RO86a, RO86b, RO86c, RO86d, RO87a, RO87b, RO87c, RO87d, RO88a, RO88b and RO88d). Important references from earlier Nordic projects and from outside the Nordic countries are (ref. JE84, UN84, UN87). The results from project *AKTU-245*, will be published separately in the NORD-series (ref. RO90), But nevertheless deposition velocities for I-131, Cs-137 and Cs-134 are summarized here in Table 6.1. Deposition velocities for Ce-141, Ce-144, La-140, Ru-103, Ru-106, Zr-95 and Nb-95 have also been measured, but these results are not included here.

Type of surface	Sample number	Deposi. velocity (cm/s)	
		I-131	Cs-134
Asphalt	184	0.077	0.011
	152	0.075	
	157	0.054	
	186	0.030	0.0045
	187	0.033	0.011
Concrete flagstones	183	0.025	0.008
	185	0.073	0.005
	145	0.022	0.005
	147	0.029	0.006
Plastered house walls	138	0.030	0.002
	140	0.030	0.001
	141	0.031	0.001
	142	0.027	0.001

Table 6.1 Deposition velocities for some urban surfaces.

In general the measurements indicated that dry deposition on smooth surfaces like paved surfaces and tiles was roughly a factor of ten lower than on rough surfaces like mowed grass and some types of rough roof materials (e.g. non-glazed ceramic tiles). Dry deposition on grass was found to be roughly proportional to the density of grass (kg/m^2). Dry deposition in a forest was higher than on grass. Dry deposition on vertical surfaces was found to be about a factor ten lower than on horizontal surfaces.

Deposition upon trees has been examined in detail. Two trees were divided into sections (height above ground), and twigs, needles, branches and the cortex measured separately. In these cases it was possible to measure quite a number of nuclides. Similar measurements have been performed on smaller trees taken from a suburban area. It seems that deposition is proportional to the mass of the measured section; which also implies that deposition on a forest will be proportional to the mass density much in the same manner as observed for grass.

6.4 Run-off

Run-off is an expression used in connection with wet deposition, and means the amount of rainfall that is not retained by the surface on which it falls. The total run-off consists of surface run-off and infiltration. There is some confusion concerning this expression, since it is sometimes taken to mean the amount of contaminant removed from the place of deposition with the rain water flowing away (running off) from the place. In connection with radioactive materials the use of the definition of total run-off also seems to be inconvenient. In connection with radioactive materials that may be contained in the water, it will make a lot of difference whether rainfall is removed from the surface by surface run-off or by infiltration. In urban areas the distinction is not so important, since most surfaces in urban areas are sufficiently impervious to prevent infiltration.

A related expression is wash-off, which is used for the process by which rain falling on a surface removes a contaminant which has been intercepted and retained earlier. This process is also referred to as weathering, and is discussed in the next subchapter.

In (ref. RI78) a model for run-off was proposed, assuming that run-off from artificial surfaces would be 100% for all rain falling after the first 3 mm. If the surface was already wet, run-off would occur sooner. Furthermore it was assumed that the concentration of radioactive materials in the run-off water was equal to the concentration in the rain water.

Experiments performed within project *AKTU-245* showed that the concentration of pollutant in the run-off water could be much smaller than the concentration in the rain water (ref. RO85b). It was also found that the extent to which the pollutant is retained in the run-off water depends both upon the type of surface and upon the type of pollutant. It was found that a pollutant was less readily intercepted by silicon-treated material (like some typical roof materials) than for e.g. clay tiles. Cesium was intercepted less readily than beryllium, and it was found that new tiles retained cesium more readily (50 - 69%) than old tiles (14 - 64%). Measurements after Chernobyl (ref. RO87c) confirmed the earlier cesium measurements. The post-Chernobyl measurements were on red tile, cement tile, corrugated eternite and silicon-treated eternite; all as roof material.

6.5 Weathering, wash-off

The radioactive materials intercepted by and retained on surfaces are subsequently exposed to the weather, and the depletion of the contamination by this mechanism is referred to as weathering or wash-off.

Weathering of permeable and impervious surfaces differ considerably. Only impervious surfaces have been investigated in project *AKTU-245*. Experiments on weathering of permeable surfaces are reported in (ref. GA64, PA86 and JA87).

The experiments indicate that run-off and weathering from lawns is negligible, and it is also very low from non-glazed roof tiles, and from tar-paper, which is frequently used as a roof material in the Nordic countries. Another common roof material is lacquered metal panels, from which run-off and weathering is found to be much larger. Actual run-off; the process which takes place with wet deposition, from roofs is found to be much larger than weathering after dry deposition. And the removal during the first rain-fall after dry deposition is much larger than later.

Weathering of cesium and ruthenium from various types of roof materials has been measured both within the AKTU program and in a preceding project period in the Nordic program. The main references are (QV84a, QV90 and RO87c).

The experiments performed in Norway used cesium-134, sprayed directly upon the roof, as contaminant. The Danish experiments have used radioactive materials naturally contained in the rainwater that has fallen upon the roof.

In the early set of Norwegian experiments it was found that after 2 days and only 7 mm rainfall (it did not rain during application of the contaminant) 3% of the applied cesium had been washed off from a tar-paper roof and 71% from a roof of plastic-coated steel. After 15 days and 51 mm rainfall, 15% of the cesium had been washed off from the tar-paper roof and 78% from the steel roof. A later set of experiments carried out in 1988 was performed only on a tar-paper roof. After 2 months 8% of the cesium had been washed off.

In the Danish experiments there were other roof material, reflecting differences in building traditions in the two countries. The Danish materials were cement tile, silicone-treated eternite, red tile and corrugated eternite. From corrugated eternite there seemed to be virtually no wash-off of cesium, and from the other materials only about 1% in 6 months. Wash-off of ruthenium was also measured, and found to be about a factor 2 higher.

The Swedish experiments were concerned with other surfaces, namely paving stones, asphalt and concrete (ref. KA87) (see also subchapter 6.6). Halflives, referring to primary photo fluence, of cesium was 140 days on paving stones, 105 days on asphalt, and 110 days on concrete. These halflives do not directly describe retention of cesium or exposure from cesium. Later studies (ref. KA89b) have shown that the long time removal on urban surfaces (except for roads with heavy traffic) is small.

Weathering on roads seems to be much more pronounced than on other surfaces. Measurements were carried out on a road with heavy traffic, and after 2 years there was only 5 - 10% of the original deposited activity left. On house walls weathering seems to have no effect.

6.6 Work on conditions in urban areas coordinated by SSI, Sweden

The Swedish National Institute of Radiation Protection (SSI) coordinates extensive research in the same areas as those of concern within the AKTU program. Short summaries of these related projects have been prepared by SSI for inclusion in the present report. These projects are not part of the AKTU program.

Dry and wet deposition, weathering and run-off effects etc.

Weathering and migration of Chernobyl fallout.
Olle Karlberg, Bjørn Sundblad, Studsvik

The objectives of this study have been to study the weathering effects on typical urban surfaces and migration on porous surfaces. Measurements were made in May, July and September 1986. The average remaining fraction after five months was around 0.6 (exc. decay). The report from the project contains mainly raw data.

6.7 Measurements of weathering performed in Kuopio, Finland

In addition to radioactive decay, ground shine is reduced with time via processes which remove radioactive material from roofs and ground surfaces such as lateral washout, downward migration, and resuspension. The overall reduction depends on the physical and chemical properties of the fallout, on the properties of the surface, on the time of the year (frozen soil), and on the time elapsed from the fallout. Typically, the environmental decay of a long lived nuclide is faster in fresh fallout than later on.

Because the fallout from the Chernobyl accident all settled within about two weeks, studying its behaviour has provided a unique opportunity to determine the environmental decay rates of the most significant artificial radionuclides in the urban environment, where most people live.

The environmental decay measurements were done with open field gamma spectrometry. The methods and results are described here in some detail.

6.7.1 Measuring sites and methods

The measurements were performed in two small towns in Eastern Finland, Kuopio and Suonenjoki. Eleven measuring sites were chosen in Kuopio and three in Suonenjoki with fall-out up to 30 kBq/m² of Cs-137. The sites were selected to represent different types of surfaces, like asphalt, concrete, cobblestone, sand, grass and forest. Characteristics of the sites are summarized in Table 6.2.

Site	Description and location
1	Parking lot, asphalt, Kuopi
2	Barren park forest, Peräniemi, Kuopio
3	Lawn in a park, Väinölänniemi, Kuopio
4	Sand covered walk in a park, Väinölänniemi, Kuopio
5	Market place covered with cobblestone, Kuopio
6	Open passage on the ground floor of a store, Kuopio
7	Asphalt covered pavement, Savonkatu, Kuopio
8	Parking lot on a roof, asphalt, Kuopio
9	Gravel field in a harbor, Kuopio
10	Sand covered children's playground in a park, Kuopio
11	Yard of a block of flats, grass, Kuopio
12	Parking lot, asphalt, Suonenjoki
13	Sand field for sports, Suonenjoki
14	Lawn in a park, Suonenjoki

Table 6.2 The measuring sites for environmental radiation.

The environmental gamma radiation was measured with a Ge(Li) semiconductor detector. The detector crystal on the dewar system was side-looking. On the measuring site the detector was placed on a support so that the distance from ground level to the crystal was approximately one meter. A van was used for transport. When measuring, the van was behind the detector so that the detector crystal pointed away from the van. The same geometry was used for all measurements. Figure 6.1 presents the setup.

The measuring apparatus consisted of the detector, preamplifier, main amplifier, 4096-channel analyzer and a personal computer. The spectra were measured from gamma energy range 0.1 - 2 MeV. The measuring time per site varied from one to two hours. The measured spectra were read by the computer and recorded on floppy disks. The final results calculated from the peak areas are given as exposure rates.

The calibration was done according to (ref. BE72), assuming an exponential distribution of radioactivity in the soil. The parameters for the exponential distribution were estimated by measuring soil layer samples from some sites.

The environmental half-times T_{half} were calculated using the following equation of linear regression:

$$\ln J = -((\ln 2)/T_{half}) \cdot t + C$$

where J is the measured exposure rate, t is time and C is constant.

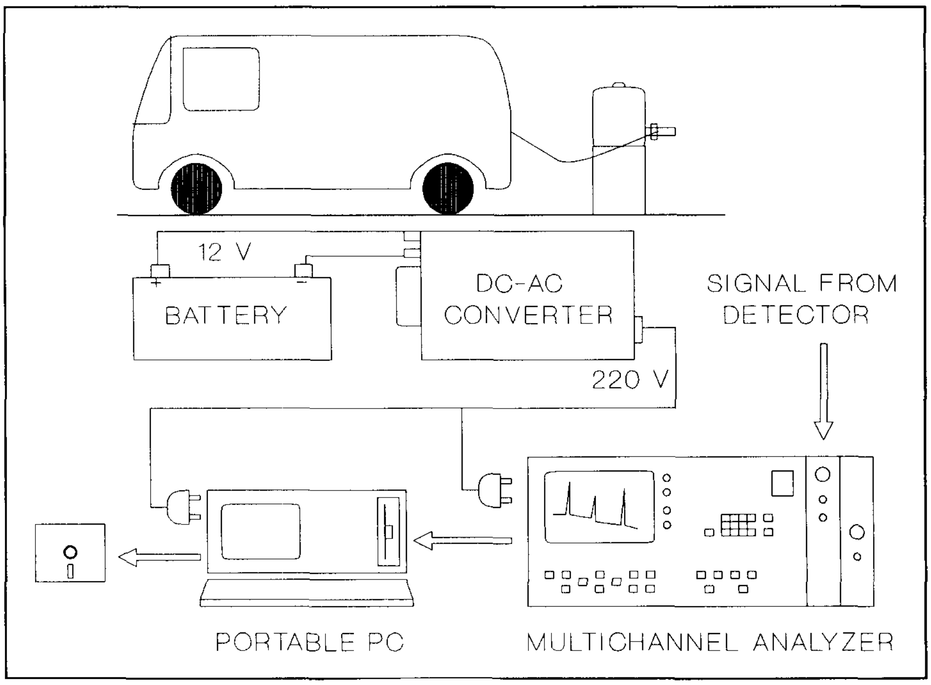


Figure 6.1 The measuring facility.

6.7.2 The measurements and results

The level of environmental radiation from the Chernobyl fallout decreased rapidly in the beginning, but already in autumn 1986 the decrease slowed. Decrease is fastest on hard cobblestone and asphalt surfaces in the centers of the towns, slowest on park lawns and in forests. On hard surfaces the radiation level had decreased also during winther, while in parks and forests the exposure rates of Cs-137 were in spring 1987 and 1988 approximately the same as in the preceeding autumns.

Before the half-times were calculated, the exposure rate values of both measured Cs-isotopes (134 and 137) were first corrected for radioactive decay to the same reference date and then added. Figure

6.2 shows the decrease of Cs exposure rates on different surfaces. It shows the ratio of measured exposure rate to exposure rate in the first measurement.

Environmental half-times for other nuclides than Cs-isotopes could be calculated only for those measuring sites, where the nuclides were detected more often than only once or twice.

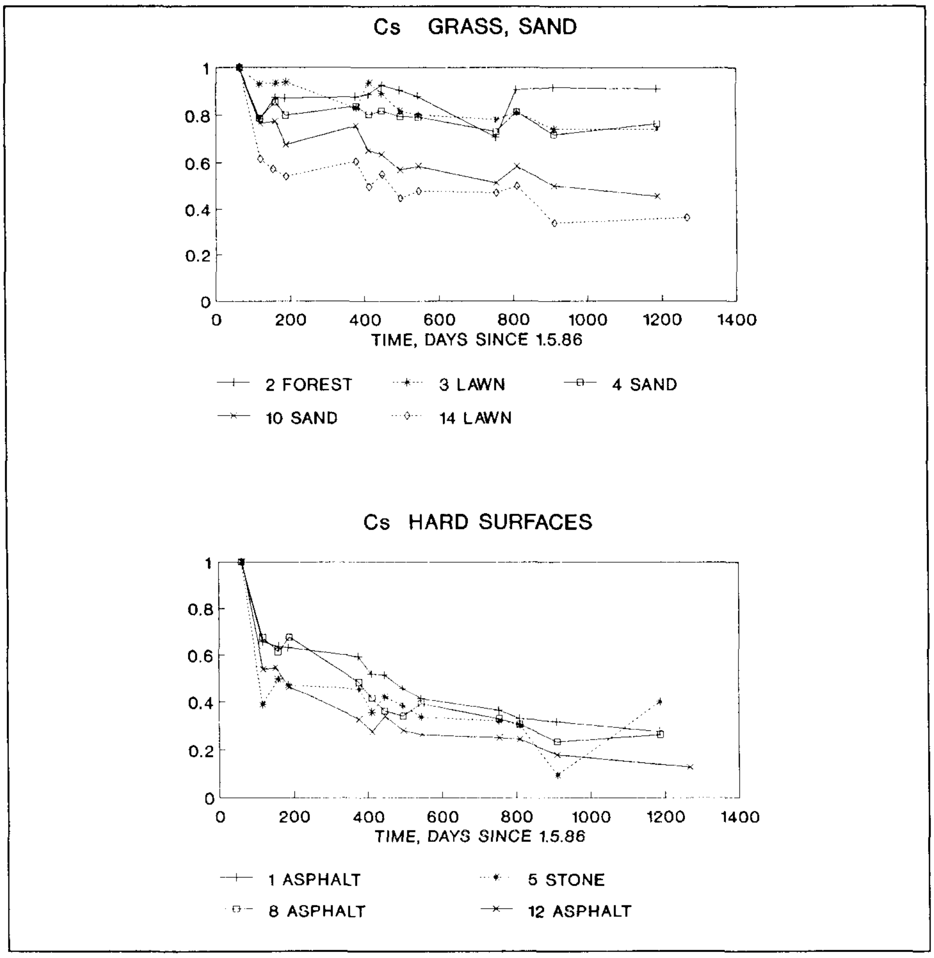


Figure 6.2 The decrease of Cs exposure rates.

The half-times were calculated separately for the year 1986 and for the period October 1986 - August 1989. The half-times calculated from the year 1986 measurements are given in Table 6.3 and those for the latter period in Table 6.4. The half-times that are statistically insignificant, are not included in Table 6.4. It means that the radiation levels on those sites have not significantly changed during the latter period.

Site		T_{half}(d) Cs	+-%	Ru-103	+-%	Nb-95
1	Asphalt	190	37			240
2	Forest			88	5	270
3	Lawn	1400	55	103	12	170
4	Sand	470	63	860	28	140
5	Stone	130	72			
6	Covered					
7	Asphalt	80	24			
8	Asphalt					190
9	Gravel	220	32	100	10	150
10	Sand	250	26	106	11	130
11	Lawn	500	72	125	12	65
12	Asphalt	120	29			79
13	Sand	130	30	64	15	54
14	Lawn	140	27	80	23	45

Table 6.3 The physical decay corrected environmental half-times for the first period (year 1986) given in days. The standard error for the half-times are also given.

Site		T_{half}(d) Cs-134	+-%	r	n
1	Asphalt	770	8	0.98	11
2	Forest				11
3	Lawn	2800	18	0.88	11
4	Sand	6000	40	0.69	11
5	Stone				11
6	Covered				6
7	Asphalt	430	12	0.95	11
8	Asphalt	740	16	0.90	11
9	Gravel				11
10	Sand	1400	16	0.91	11
11	Lawn				10
12	Asphalt	590	10	0.96	11
13	Sand				5
14	Lawn	1600	25	0.82	11

Table 6.4 Physical decay corrected environmental half-times (T_2) for the latter measuring period (October 1986 - August 1989) given in days. The standard error for the half-times, correlation coefficient (r) for the exponential fit and number of data points are also given. The statistically insignificant half-time values are not included.

7. SHELTERING IN HOUSES

The fact that houses actually provide protection against radiation has to a certain degree been taken into account in accident consequence assessment models. But the importance of this aspect has often been underestimated. The obvious protection mechanism, the direct shielding effect of the house materials, is discussed in subchapter 7.1. The sheltering aspect actually covers two quite different situations: shielding against radiation from radioactive materials in a passing plume (which is the subject matter of project *AKTU-250*), and shielding against radiation from radioactive materials deposited upon the ground and exterior surfaces of houses.

A quite different type of protection provided by houses is referred to as the filtering effect. The presence of such an effect is rather less obvious and usually it is still not incorporated in accident assessments. It has been known for quite some time that the indoor concentration of pollutants (of outdoor origin) is lower than the outdoor concentration, but it has been believed that for a short-duration pollution situation (like a passing plume containing radioactive materials) the time integral of the indoor and outdoor concentrations would be equal. The reason is that although the indoor concentration will be lower, the time before the indoor concentration decreases will be longer. Even now, when experiments have shown that the indoor time integral is indeed lower, it is discussed whether this is due to a filtering effect or to deposition on indoor surfaces (including furniture, carpets, draperies etc.). These aspects, investigated within project *AKTU-245*, are treated in subchapter 7.2.

7.1 Shielding effect, project AKTU-250

Exposure models used in accident consequence assessment models often use extreme simplifications. One example is the often used assumption of the flat field of infinite size, when exposure to radioactive material deposited upon ground shall be calculated. It is assumed that the exposed person is standing in the middle of the flat field - day and night. Admittedly, a reduction factor of 0.3 - 0.5 is often applied, as a rough representation of the shielding effect of buildings and roughness of the terrain. In urban areas this representation is especially wrong. The population in urban areas spend most of their time indoors, and the houses provide shielding. As a matter of fact, the houses even provide mutual shielding; that is shielding against radiation from e.g. an area of contaminated ground behind neighboring buildings. Furthermore, the building materials in urban areas are to a large degree heavier than building materials in rural areas, and accordingly provide a better shielding.

There are actually two different shielding situations that have to be addressed. One is the case when a plume containing radioactive materials is passing a house. The other is the case when deposition on the areas surrounding the house and on walls and roof has taken place. These two situations are often referred to as cloud shine and ground shine, respectively. Ground shine was the subject of a project in a previous project period of the Nordic Safety Program, and a computer program was developed (ref. HE81 and TV82). Ground shine has been studied in great detail, both theoretically and experimentally. Cloud shine is more complicated situation, and extensive work has so far not been performed. The shielding factors for cloud shine used in ACA calculations up till now are based upon simplified calculations, mainly for American housing conditions.

In past ACA calculations it has usually been concluded that improvement of the methods for calculating cloud shine is not imperative, since this exposure pathway is not dominant for the releases investigated. However, for releases more dominated by the noble gases this will not be true, and it is felt that more realistic shielding factors will be needed in future safety studies. Cloud shine is addressed in project AKTU-250, in which a computer program for this situation is developed. The program is based on the relatively simple point kernel method. Since the shielding geometry of a house can not be accurately described by the point kernel, the simple method is tested against the much more accurate Monte Carlo method. This, however, is so time-consuming on the computer that only a limited number of cases can be run.

7.1.1 Calculation model

The fundamental calculation method used is based on the socalled exponential point attenuation kernel that links the radiation flux density at a given detector position to the point source strength. Attenuation resulting from geometrical spreading with increasing distance from the source as well as exponential attenuation and scattering of the photons are taken into consideration.

The assumptions for the point kernel method are that both the source and the shielding medium are isotropic, i.e. that the source radiates uniformly in all directions, and that the medium has the same attenuating properties in all directions. As far as scattering of photons are concerned it is assumed that both source and detector are located within an infinite homogeneous medium. The scatter contribution is calculated by the use of a dose build-up factor which depends on both the photon energy and the atomic number of the absorbing medium.

Building types are described as a composite of cubic boxes. The thickness of outer walls, inner walls, partition walls and floors/ceilings can be varied independently of each other. Attenuation and scattering in the different walls and floors having projected thicknesses $t(x,y,z)$ along the direction from each plume sub-volumen to the detector point is accounted for in the numerical integration over the relevant part of the plume volume. Attenuation of photons passing through window apertures is neglected.

7.1.2 Calculated shielding factors for Nordic houses

The shielding factor S for plume radiation is defined as the ratio of the dose rate (or kerma rate) at a given indoor (or outdoor) location to an outdoor reference dose rate. The reference dose rate is here defined as the dose rate 1 meter above ground level without buildings present and with the same horizontal position to the plume as that of the indoor position. Shielding factors have been calculated for Nordic single-family houses and multistory buildings. The dimensions and the building data for the single family houses are shown in Table 7.1 and Figure 7.1.

	Denmark	Norway	Sweden	Finland
Outer wall	400	34	155	51
Inner wall	220	20	51	43
Roof	200	82	100	100
Window fraction	25%	25%	25%	25%

Table 7.1 Structure dimensions in kg per square meter for selected Nordic single-family houses.

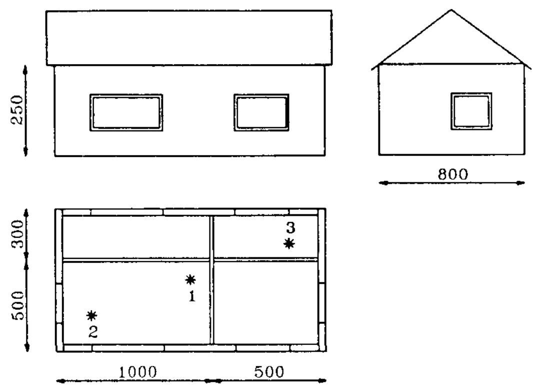


Figure 7.1 Dimensions of Nordic single-family house in cm.

The dimensions and building data for the multistory buildings are shown in Table 7.2 and Figure 7.2. The density of the walls, roofs and floors is assumed to be 1.7 kg/liter for all the buildings.

	Denmark	Norway	Sweden	Finland
Outer wall	595	390	305	360
Inner wall	220	30	100	255
Partition walls	255	360	325	440
Roof + ceiling	255	80	255	100
Floors	170	440	425	440
Window fraction	20%	20%	20%	20%

Table 7.2 Structure dimensions in kg per square meter for selected Nordic five-storied buildings.

The window fraction is the fraction of the total outer wall area that is covered by windows. The dimensions and the placing of the windows can be selected freely, exemplified in Figures 7.1 and 7.2.

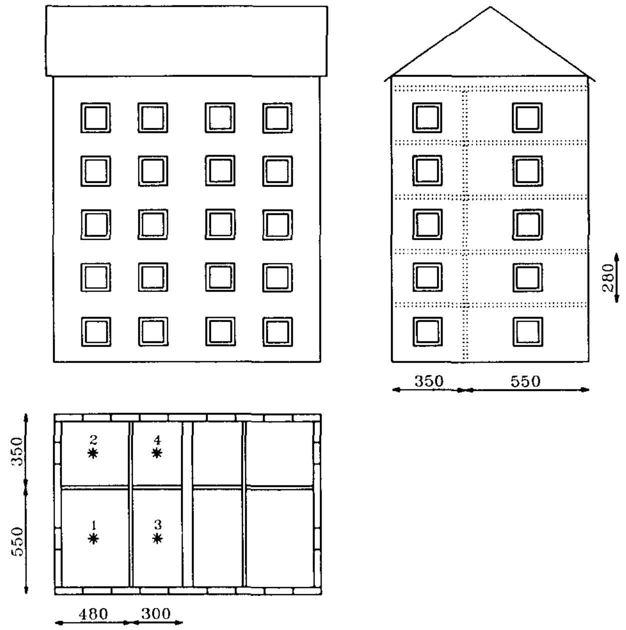


Figure 7.2 Dimensions of Nordic five-storied building in cm.

For the single-family houses shielding factors have been calculated for three different photon energies and for the three different locations shown in Figure 7.1. The results are shown in Table 7.3. The variation in the shielding factors indicate the dependence of the position in the house.

	0.20 MeV	0.68 MeV	1.68 MeV
Denmark	0.03-0.04	0.05-0.09	0.07-0.14
Norway	0.46-0.58	0.56-0.66	0.63-0.71
Sweden	0.11-0.21	0.17-0.30	0.22-0.36
Finland	0.19-0.42	0.27-0.52	0.34-0.59

Table 7.3 Shielding factors for Nordic single-family houses.

The shielding factors shown in Table 7.3 are calculated without contribution from the indoor cloud. This would increase the figures with approximately 2-8 % for the given structures.

For the multistory buildings shielding factors have also been calculated for three different stories and four different locations at each story as shown in Figure 7.2. The results are shown in Table 7.4 for a photon energy of 0.68 MeV. The contribution from the indoor cloud is excluded. The variation indicate the dependence of the location at the given story.

	Bottom story	3´rd story	5´th story
Denmark	0.013-0.056	0.032-0.097	0.081-0.130
Norway	0.031-0.069	0.065-0.110	0.270-0.360
Sweden	0.026-0.083	0.050-0.120	0.150-0.290
Finland	0.016-0.054	0.024-0.120	0.170-0.330

Table 7.4 Shielding factors for Nordic five-storied buildings.

7.1.3 Parameter variations

Some of the parameters were varied in the calculations to identify their influence on the shielding factors. For a simple blockhouse without windows and submerged in a semi-infinite cloud the outer wall thickness and the photon energy were both varied within relevant ranges. The blockhouse geometry is shown in Figure 7.3. In the calculations the density of the walls is 1.7 kg/l.

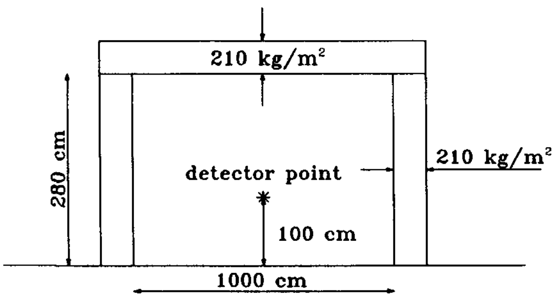


Figure 7.3 Geometry for a simple blockhouse.

Figure 7.4 shows the calculated shielding factors for the blockhouse for different wall thicknesses and photon energies. Both parameters have a significant influence on the calculated shielding factors.

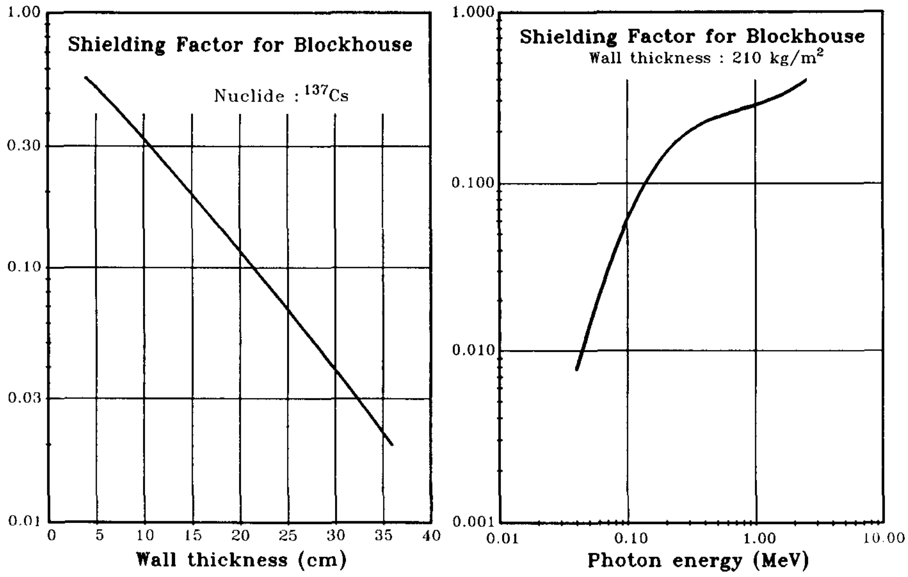


Figure 7.4 Shielding factors for a simple blockhouse for different outer wall thicknesses and photon energies.

For the Danish traditional single-family house the plume/building geometry was varied. Two different atmospheric stability categories (Pasquill D and F), two different heights (0 m and 100 m), four different downwind distances and two different crosswind distances were used. The photon energy was 0.68 MeV. The results are shown in Table 7.5 for location no. 1.

For elevated plumes at short downwind distances the shielding factor is increased compared to the semi-infinite plume due to the less effective shielding of the roof, which constitutes the main part of the shielding in these situations. For ground level plumes at short distances an opposite effect is observed. The reason is that the dimension of the plume still is small compared to the semi-infinite plume and that the fraction of the total radiation penetrating the roof therefore is proportionally smaller. A similar effect can be observed for narrow ground level plumes (Pasquill F) with the house positioned at some crosswind distance from the plume. At large downwind distances the results become close to those for the semi-infinite cloud.

Downwind distance (meters)	Pasquill D	Pasquill F
(crosswind distance = 0 meters)	(plume height = 0 meters)	
200	0.04	-
500	-	0.03
1000	0.06	0.06
10000	0.09	0.05
(crosswind distance = 0 meters)	(plume height = 100 meters)	
200	0.22	-
500	-	0.20
1000	0.15	0.19
10000	0.09	0.11
(crosswind distance = 500 meters)	(plume height = 0 meters)	
10000	0.09	0.04

Table 7.5 Shielding factors for a Danish traditional single-family house for different plume/building geometries.

For a Danish single-family house submerged in a semi-infinite cloud, variations of the structure dimensions have been made. The thicknesses of the outer walls, inner walls and the roof/ceiling were varied in a range from 5 cm to 30 cm (85 kg/m^2 to 510 kg/m^2). The results are shown in Figure 7.5.

The main contribution to the indoor dose rate from a semi-infinite cloud originates from that part of the plume lying above the roof plane and only a minor contribution from the part of the plume below the roof plane. In a single-family house of standard dimensions, more than 90% of the indoor dose rate originates from the could above the roof plane.

This situation is reflected in Figure 7.5. It appears that the shielding factor is rather insensitive to variations in outer and inner wall thickness, typically less than 30% for the given range. For the roof/ceiling, on the other hand, there is a strong dependence of the thickness, up to a factor of 10 for the given range.

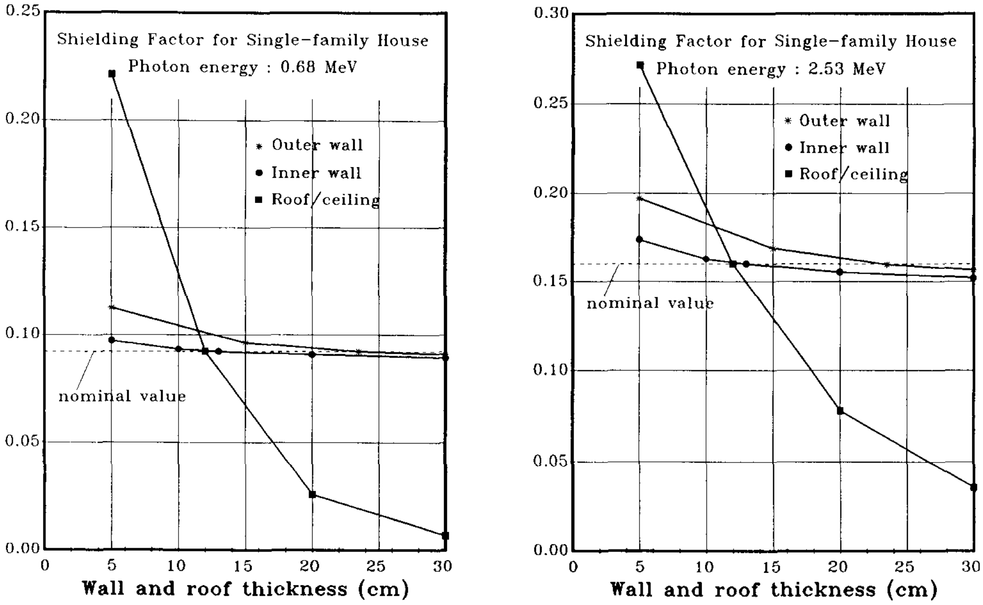


Figure 7.5 Shielding factors for a Danish single-family house for different wall and roof thicknesses at photon energies of 0.68 MeV and 2.53 MeV.

7.1.4 Shielding factors for outdoor residence

A person staying out-of-doors in an urban area will be exposed only to radiation from a part of the plume, because the surrounding buildings will act as a shielding. The geometry is shown in Figure 7.6.

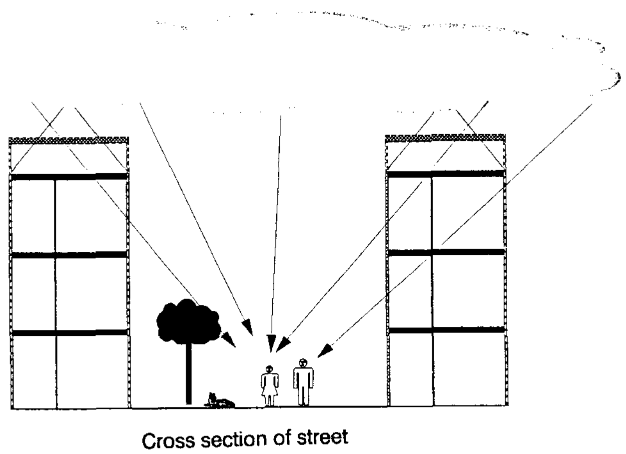


Figure 7.6 Geometry for outdoor exposure from a plume in an urban area with multistory buildings at both sides of a street.

The shielding factor will decrease for increasing building height and decreasing street width. Calculations have been made for building heights and street widths of 10 m and 20 m and the results are shown in Table 7.6.

	Street width 10 m		Street width 20 m	
	Height 10 m	Height 20 m	Height 10 m	Height 20 m
Shielding factor	0.33	0.13	0.48	0.24

Table 7.6 Outdoor shielding factors for urban areas with buildings at both sides of the street, for photon energy 0.68 MeV.

7.1.5 Monte Carlo calculations

The assumptions for the validity of the point kernel method are not always fulfilled, and in certain situations where the main part of the indoor dose rate originates from scattered photons the point kernel results can be misleading. Therefore, some calculations have been made with the Monte Carlo code MCNP (Monte Carlo Neutron Photon) supplied by the Oak Ridge National Laboratory. The purpose of these calculations was in the first place to investigate the influence of window and door apertures.

Shielding factors have been calculated for a simple blockhouse for different photon energies and for three different window fractions, 0%, 15% and 50%. The dimension of the plume was assumed to be semi-infinite. The MCNP results are shown in Figure 7.7 together with the corresponding point kernel results.

The results indicate that window apertures could be more important than found by the point kernel method, especially at lower photon energies. On the other hand, the MCNP values tend to be lower than the corresponding point kernel values at higher energies.

7.1.6 Comparison with other work

Several authors have made both simplified and more detailed calculations of shielding factors for semi-infinite clouds containing different gamma-emitting nuclides.

Kocher (ref. KO80) has made calculations for "igloo"-shaped buildings without windows. He finds shielding factors in the order of 0.7-0.02 for an outer wall thickness ranging from 5-20 centimeter concrete. The method of calculation is the point kernel method.

Burson and Profio (ref. BU77) have used the same method and find a representative shielding factor of 0.6 for masonry houses, 0.4 for basements in masonry houses and 0.2 or less for large office buildings.

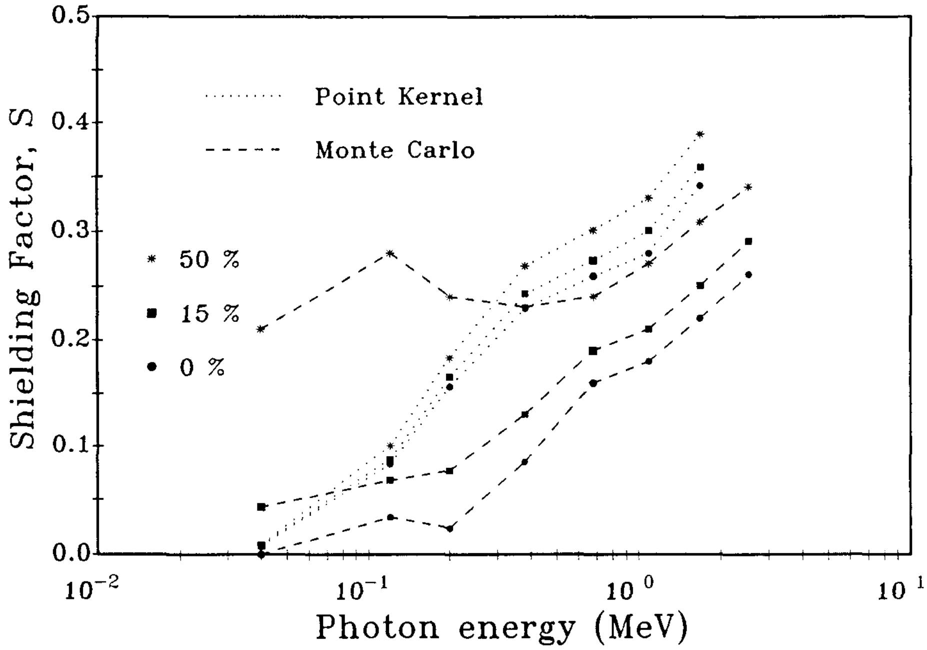


Figure 7.7 Shielding factors for simple blockhouse with dimensions as shown in Figure 7.3 and with a window fraction of the outer walls in the range 0 - 50%.

Peter Jacob et al. (ref. JA86) have made Monte Carlo calculations for a semi-detached house and find shielding factors in the range 0.1-0.3 in the ground floor, 0.2-0.5 in the first floor and 0.001-0.02 in the basement for photon energies between 0.2-3.0 MeV. The outer wall thickness is 23 centimeter brick.

Le Grand et al. (ref. LE85) have calculated shielding factors for dwellings in the order of 0.3-0.4 for 0.5 MeV photons using Monte Carlo technique. The indoor doses are calculated to a cylindrical phantom. The thicknesses of the outer walls and roof are 10 centimeter and 3 centimeter concrete, respectively.

7.1.7 Conclusions about shielding

The shielding factors for plume radiation are generally larger than the corresponding shielding factors for ground radiation from deposited activity. For Nordic buildings this difference constitutes a factor of 2 - 3.

Based on calculations on different building types and parameter variations the following shielding factors for plume radiation are recommended as conservative values covering a wide range of Nordic building types.

	Single-family	Multistory
Denmark	0.10	0.05
Norway	0.60	0.10
Sweden	0.30	0.10
Finland	0.50	0.10

Table 7.7 Recommended shielding factors for Nordic single-family and multistory buildings.

Outdoor residence in urban areas with multistory buildings will give smaller cloud shine doses than in open areas. Outdoor shielding factors for urban areas in the range 0.3 - 0.8 are realistic, depending upon how densely built-up the area is. A value of 0.5 can be recommended as reasonably conservative for urban areas and 0.8 for sub-urban areas.

7.2 Filtering effect and indoor deposition, project AKTU-245

Inhalation doses have traditionally been calculated in accident consequence assessment models as if the person spends all his time outdoors. Danish, Norwegian and Finnish experiments show that indoor air concentration is lower than outdoor air concentration by a factor of 2 - 5, resulting in correspondingly lower inhalation doses. There are two possible explanations. One is that particles containing the radioactive materials, partially will be retained in narrow fissures through which outdoor air enters a house. This is often referred to as the filtering effect of houses.

The other possible explanation is that radioactive materials in the indoor air will deposit in the rooms. Up till now it has been thought that the filtering effect is the most important of the two processes. Measurements of indoor deposition gave deposition velocities that were very low compared to deposition velocities on outdoor surfaces (0.008 cm/s in a Norwegian experiment and 0.0001 - 0.002 cm/s in Danish experiments). The surfaces measured were, however, smooth, plane surfaces. An interior contains many surfaces where deposition could be expected to be much larger, like rugs and fabrics. Within project *AKTU-245* Danish measurements of total deposition in furnished rooms show that total deposition in this case is much larger, and from these experiments it is concluded that almost the whole reduction of indoor air concentration compared to outdoor air concentration is due to deposition indoors, and not to the filtering effect.

7.2.1 Indoor/outdoor concentration ratio

The ratio between indoor and outdoor air concentration has been measured in Denmark (ref. RO85), Finland and Norway (ref. CH87). The Danish experiments were performed in all rooms in 17 selected buildings, the Finnish experiments were for four different flats in apartment buildings built between 1968 and 1972, and the Norwegian experiments were performed in one typical villa built in 1954. Air exchange rates were measured in all experiments. In all the experiments naturally occuring Be-7 was used as tracer.

In the Danish experiments the following mean value between indoor and outdoor concentration was found:

$$C_i \;/\; C_o \;=\; 0.32$$

It was also found that the ratio between indoor and outdoor concentration was a function of the air exchange rate, with the following expression relating to the living rooms investigated:

$$C_i \;/\; C_o \;=\; 0.24 \cdot \lambda_r \;+\; 0.21$$

In the Finnish experiments the ratio between indoor and outdoor air concentration was found to vary from 0.23 to 0.45. The air exchange rate was about 0.2 times per hour in all the flats. The results from a first set of experiments could not be used, as some of them indicated that the indoor concentration was larger than the outdoor concentration. The outdoor concentration in this set was measured at an air filter station about one kilometer from the building where the indoor measurements were performed. Evidently variations in outdoor air concentration are sufficiently large to make it imperative to measure outdoor air concentrations immediately outside the building where the indoor concentration is measured. In each measurement a collection period of four days was used.

In the Norwegian experiments ratios between indoor and outdoor air concentration of 0.5 to 0.9 in the first set of experiments, and 0.4 to 0.47 in the second set of experiments. In the first set of experiments the collection period was three days, but the filters were changed every 24 hours. It is believed that entering the house every day disturbed the measurements, and that the results from the first set of experiments are not representative. During the second set nobody entered the house during the collection periods. There were four collection period, but these varied in length from two to seven days. The length of the collection period did not seem to have any impact on the results of the measurements. The air exchange rate was found to be 0.13 per hour. Some doubts were expressed as to the correctness of this measurement, but this will not be discussed further here.

7.2.2 Indoor deposition

A measurement of indoor deposition was performed in the same house in Norway, as the one in which the experiments described in previous subchapter were performed. The surface upon which deposition was measured was a horizontal perspex plate, which was allowed to collect deposited material 12 days. Outdoor air concentration was measured throughout this time period. Indoor air concentration measurements were, however, not performed, so as not to disturb deposition conditions. The deposition velocity onto the plate was found to be 0.00003 cm/s, which is a very low value compared to e.g. grass.

Much more extensive measurements of indoor deposition have been performed in Denmark (ref. R087d and R088e). In these experiments the deposition velocity on some surfaces have also been measured. The results for cesium-134 were on the average 0.00015 cm/s for wallpaper (vertical), 0.0011 for vinyl floor and 0.00065 cm/s for wooden floor. These values are considerably higher than the values from the Norwegian experiment. The difference may be explained by differences in the experimental conditions; primarily that in the Norwegian experiment there was only natural air exchange in the house, while in the experiment conducted in Denmark an overpressure was maintained in the house by blowing outside air into the house through a duct.

The other aspect of the experiments performed in Denmark is measurement of total indoor deposition. To express indoor deposition, an "average local deposition velocity" has been defined. This quantity is, however, not comparable to deposition velocities as usually defined. Unlike other deposition velocities it will also be a function of the volume of the house, as well as the sizes and types of surfaces in the house. The "average local deposition velocities" determined (ref. R087d) are about a factor ten larger than the deposition velocities determined for specific surfaces. "Average local deposition velocities" have also been measured in unfurnished, partly furnished and wholly furnished rooms. No difference was found between the two first cases, but deposition in the wholly furnished room seemed to be a factor two higher. For further information, including the exact definition of the "average local deposition velocity", see (ref. R090).

8. WINTER ASPECTS, PROJECT AKTU-255

Winter conditions are, of course, of particular interest in the Nordic countries. The possible consequences of a reactor accident may be different under winter conditions for a number of reasons.

- Deposition with snow will be different from deposition with rain.
- Deposition on snow-covered surfaces will be different from deposition in the same areas during summer.
- Weathering (may often be referred to as run-off) during winter and throughout spring, will be different from during summer conditions both in urban and rural areas.
- Decontamination may be much cheaper and much less destructive than under summer conditions.
- Snow provides additional shielding against ground shine.

Some of these aspects have been addressed in the project *AKTU-255*, but by no means all. The investigations were performed at Institutt for energiteknikk, Norway, and were actually initiated during the preceding program period of the Nordic Safety Program. All the results from all the types of winter experiments were presented at the 1988 Summer Meeting of the American Nuclear Society (ref. TV88a).

8.1 Winter conditions

Just like during summer, weather conditions change from one day to another during the winter. The snow conditions also change, and a contamination situation may be quite different if there is light feathery snow, coarse-grained snow, snow packed solid from traffic, or wet snow.

It has not been possible at present to obtain data on typical frequencies of different weather and snow conditions, but some data on snowfalls in a typical inland climate, not far from Oslo, have been provided by the Meteorological Institute of Norway. The average number of days with snow per month is:

September:	0.2 days
October:	2.6 "
November:	8.8 "
December:	12.5 "
January:	12.0 "
February:	8.0 "
March:	6.2 "
April:	6.2 "
May:	1.3 "
Sum:	65.7 days

At this station ca. 20% of the days of the year have some snow-fall. But a fact that is more relevant is that roughly one third of the days of the winter months have some snow-fall. This is an average over the years 1957 to 1977, and the total precipitation per year falling as snow is equivalent to 150 - 200 mm precipitation as rain.

Some additional information has been obtained for another meteorological station, at Blindern in Oslo. These data, which are averages over the years 1941 to 1980, concerns the increase in thickness of the snow cover from one day to the next:

Increase is 1 - 5 cm in 76.39% of 24 hour periods with snowfall.
Increase is 6 -10 cm in 17.26% of 24 hour periods with snowfall.
Increase more than 10 cm in 6.35% of 24 hour periods with snowfall.

According to this information, there is relatively often a layer of fresh snow. But it would be equally important to know how long that layer stays fresh; that is, when the next above-freezing-point situation occurs. This is of particular importance since previous investigations, both in connection with weapons fall-out and acid precipitation, indicate that a pollutant may stay on the snow surface as long as there are stable cold conditions, but that it quickly moves to the bottom of the layer at temperatures close to or above the melting point for snow.

8.2 Weathering from roofs under winter conditions

The investigation of natural decontamination (weathering) of roof materials reported here, was performed as part of the preceding program period within the Nordic Safety Program, at the Institutt for energiteknikk, Norway. Although it is not an AKTU project, a rather full presentation is given here, since there are so few experiments at all that are concerned with winter conditions. Similar experiments during summer conditions were performed as part of the same project, and these are also described in this chapter to provide the necessary basis for comparison between summer and winter conditions.

8.2.1 The roofs used in the experiments

In order to decide what could be considered fairly typical roof materials, the Norwegian Building Research Institute was contacted, and gave advice on what should be considered typical roof materials for relatively modern buildings in Norway and Sweden (ref. GA81).

Typical roof materials in Norway were said to be:

- One-family houses: Tar-paper shingles, concrete "tiles", metal (steel covered with plastic paint), and corrugated aluminum.
- Apartment buildings: Tar-paper and metal.

In Sweden the typical materials are said to be:

- One-family houses: Mostly metal, but also concrete "tiles" and ordinary tiles, while tar-paper is rare.
- Apartment buildings: Metal, but also tiles and tar-paper.

Each experimental roof consisted of a supporting structure with a plywood board on top. The dimensions of the board were 1.2 meters width and 2.4 meters heights (area 2.88 squaremeters). This was covered with ordinary tar-paper and a steel roof material respectively. Around the roof was mounted a plastic foil wind shield of 0.5 meters height. The lowest part of the roof was 1.5 meters above ground.

The tar-paper was of a type covered with crushed shale. The type of shale may be quite important in connection with retention. The roof was mounted at an angle of 5 degrees, which is typical of an apartment building.

The metal roof was of a type that is made as an imitation of glazed tiles, made from corrugated steel sheet. The upper surface was covered with a layer of primer and a top layer of 25 micron PVF2 80/20 (polyvinylidenfluoride). This roof was mounted at an angle of 23 degrees, typical of a one-family house.

The water running off the roof was collected by a plastic gutter, and led to a polyethylen container.

The metal roof was mounted about 2 weeks, and the tarpaper roof 4 days, before contamination. Both roofs experienced some heavy rainfall before contamination. The roof used for contamination upon a snow layer was mounted about 1 month before contamination. The distance between the roofs was about 5 meters.

The roof used for the winter 1984 experiment was one of those used during the previous experiments, but the tar-paper had been replaced with new tar-paper. The new paper was put on three weeks before the activity was added. The roofs were of course always left out in the open.

8.2.2 Contamination of the roofs

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, and the solution applied dried quickly.

Like in the "summer" experiments, 1.5 µCi of Cs-134 was sprayed onto the roof in the winter experiments. The spraying technique was the same, although extra care had to be taken to avoid penetration of the snowcover.

On 29 October 1982 each of the two roofs was contaminated with 1.5 µCi of Cs-134. There was still "summer" conditions. After 2 days and a precipitation of 7 mm (19.2 liters), the remaining activity on the metal roof was 29% and on the tar-paper roof 97%. After 15 days with altogether 51 mm precipitation, the concentration of Cs-134 in the water from the metal roof was below the limit where detection was possible. At this time the remaining activity on the metal roof was 22% and on the paper roof 85%. By the 29th June 1983 (some 8 months after contamination) about 61% remained on the tar-paper roof after a total of 589 mm precipitation (1207 liters). At this time measurements were terminated.

In one of the experiment performed in this period the contamination was done during winter conditions. 1.5 µCi Cs-134 was sprayed upon an 8 cm thich cover of wet snow on a tar-paper roof. This experiment was performed in late March. After 3 days 38 liters of water had been collected (equivalent to 13 mm precipitation). In the water was found 10%

of the Cs-134, meaning that 90% had been adsorbed to the roof. By this time all the snow on the roof was melted. After 3 months and 191 mm precipitation the remaining activity on the roof was 59%.

It was planned to perform experiments upon roofs with a stable cover of snow and ice, typical of Norwegian winter conditions. However, the winter of 82/83 decided not to be typical, and the experiment could not be carried out. But the following winter it was performed. The experiment was started on 20 February 1984. There was a stable layer of ice and coarse-grained snow on the roof, approximately 5 cm thick. There was light snow-fall at the time when activity was added. During the time needed to add the activity, roughly 1 cm of additional snow accumulated, increasing to another 5 cm during the following night. The temperature, when the activity was added, was -7^{0}C. After 9 days came the first day with run-off (about 10 liters). The following day the first sample was collected. The run-off had then increased to 14 liters, and 16% of the added activity was recovered. After 15 days, when all snow had disappeared from the roof, 47% had been recovered; and when the experiment was terminated, after 4 months, the cumulative recovered fraction was 65%.

8.2.3 The weather conditions

Weathering depends, as the name implies, upon the weather, which may differ substantially from one year to another. For this reason, the weather conditions during the period over which the experiments were performed are summarized here.

November 1982 was extremely humid, with almost double the normal amount of precipitation. December had about 150% of the normal. The ice which was formed on the roof during this period may explain the somewhat lower concentrations of Cs-134 in the collected water. January, February and March were in 1983 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 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. One did not attempt to explain this difference. The pH of the actual precipitation might be more important, but was not measured. If warranted, it could be obtained from the meteorological institute.

In the winter 83/84 the weather during March was changeable and somewhat colder than average. Precipitation was about normal, and falling mostly toward the end of the month. 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.

8.2.4 Summary of the results

All numerical results here and in the tables of this chapter are normalized to refer to inactive cesium. If it is wished to relate the results to specific cesium isotopes, the proper radioactive decay should be applied.

The results show that more than 70% of the Cs-134 added during "summer" conditions runs off the ironplate roof during the first normal rainfall; while on the tar-paper roof, which is covered with fine grains of shale, only 3% is removed during the same rainfall. There is evidently a marked difference in the decontamination time of these two types of roof material. Measurements on run-off water from the ironplate roof were only performed for a couple of weeks. At that time the content in the run-off water was very low. Measurements on the tarpaper roof, however, were continued for a long time, and were discontinued 8 months after contamination. Part of this time the roof had been covered with ice and snow. The time sequence of run-off is shown in Table 8.1.

The winter experiments included two tarpaper roofs that were contaminated while covered with snow. In the first case contamination was performed (in early spring 1983) on wet snow. In the second case (in winter 1984) contamination was performed on a stable layer of ice and snow.

For the roof contaminated in spring 1983 the measurements were continued only 3 months. The precipitation upon this roof was only half of that on the roof in the "summer" experiment. In spite of this, roughly the same fraction of the contaminant (ca. 40%) had been removed from the "summer" as well as the "winter" roof when the experiment was terminated. This indicates that cesium applied to a snow- or ice-covered roof adheres less readily to the roof materials. The fact that the "summer" experiment extended into the winter season, offers an alternative explanation. In the last part of the "summer" experiment, an icelayer was formed upon the contaminated surface. This may have prevented contact between the contaminant and the runoff water, thereby preventing its removal.

From the experiment performed in winter 1984, it was found that the first run-off water contained 16% of the applied activity. On 16th March (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 removed from the roof.

Summary of the results:

- Ironplate roof, application during "summer" conditions:
  22% remained on the roof at end of experiment.
- Tar-paper roof, application during "summer" 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.

Time period	Steel roof: Vol. run-off (l)	Steel roof: Sum pre-cip. (mm)	Steel roof: Remai-ning activ. (%)	Tar-paper roof: Vol. run-off (l)	Tar-paper roof: Sum pre-cip. (mm)	Tar-paper roof: Remai-ning activ. (%)
29/10 to 31/10	19	7	29	19	7	97
31/10 to 1/11	21	14	25	24	15	93
1/11 to 9/11	22	22	24	22	23	91
9/11 to 10/11	14	27	24	13	27	90
10/11 to 12/11	20	34	24	20	34	89
12/11 to 13/11	24	42	24	24	43	86
13/11	>25	51	24	>25	51	85
13/11 to 22/11	Experiment			>76	78	85
22/11 to 25/11	discontinued			50	95	84
25/11 to 28/11				71	120	82
28/11 to 8/12				>103	155	79
8/12 to 16/12				22	163	79
16/12 to 22/12				>46	179	78
22/12 to 10/1				46	195	77
10/1 to 25/1				60	216	76
25/1 to 24/3				82	244	72
24/3 to 7/4				79	272	71
7/4 to 21/4				78	299	68
21/4 to 12/5				79	326	65
12/5 to 16/5				88	357	63
16/5 to 30/5				>100	391	62
30/5 to 29/6				80	419	61

Table 8.1 Results from the experiment where application took place during "summer" conditions.

Time period	On wet snow: Vol. run-off (l)	On wet snow: Sum pre-cip. (mm)	On wet snow: Remai-ning activ. (%)	Time period	On stable ice and snow: Vol. run-off (l)	On stable ice and snow: Sum pre-cip. (mm)	On stable ice and snow: Remai-ning activ. (%)
21/3 to 24/3	38	13	91	20/2 to 1/3	14	5	84
24/3 to 7/4	79	41	84	1/3 to 6/3	31	16	53
7/4 to 21/4	84	70	77	6/3 to 21/3	18	22	50
21/4 to 12/5	80	98	68	21/3 to 2/4	38	35	46
12/5 to 16/5	88	128	65	2/4 to 9/4	76	61	40
16/5 to 30/5	100	163	61	9/4 to 1/6	59	82	38
30/5 to 29/6	80	191	59	1/6 to 4/6	60	103	37
	Experiment			4/6 to 15/6	65	125	36
	discontinued			15/6 to 22/6	50	143	35

Table 8.2 Results from the experiments where application took place during winter conditions.

The decontamination time depends upon the weather conditions when contamination is performed, the weather conditions afterwards, upon whether there is snow or ice on the roof. It 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.

Another aspect that ought to be examined, is that the run-off may be linked to rain intency. An examination of the detailed data in this chapter does not show such a dependency, but from these data one can only estimate the average precipitation over the time periods given. Data on rain intensity must be collected in a different manner. Intense rain occurs over time periods of the order of an hour or less. Data collected over some days may contain unidentifiable periods of intense rain.

Some losses of water have occurred due to overflow of the containers. Almost without exception, the precipitation happened to fall during the weekends, and it was not always possible to empty them in time to prevent overflow. On one occasion some water was lost because one of the containers split in the bottom due to frost. These occurrences, however, have little impact on the trend of the results.

8.3 Run-off from a meadow

8.3.1 About migration in snow

It is a well-known fact that cesium has a strong tendency for being trapped in soil, and particularly in clay rich soils. Many different investigators have found that cesium deposited upon ground is still found in the upper (10 cm) layer of soil even after extended time periods, indicating slow vertical and horizontal migration. This has been observed for weapons fall-out during the late 50s and the 60s. There are strong indications by now that vertical migration of Chernobyl cesium in soil is also very slow.

A large part of the year there are winter conditions in the Nordic countries; there is frost in the ground, and the ground is covered with ice and snow. Conditions and parameter values used in accident consequence assessments represent summer conditions, and the calculated consequences may be too high, if deposition takes place during winter conditions.

Dry deposition during winter conditions or deposition with snow will leave the deposited materials in the upper layers of snow. But previous experiments indicate that the first melt water will bring the materials down to ground level. Since the ground is frozen, however, it is not sure that the cesium will be trapped in the soil, like under summer conditions.

These aspects of winter conditions have been investigated (although referring to other chemical elements) e.g. in project SNSF (The

Norwegian Interdiciplinary Research Program: Acid Precipitation - Effects on Forest and Fish). Relevant information can be found in (ref. DA78, DA79, DA80, JO78, OV80 and SE80).

Behaviour of the nuclides Sr-90 and Cs-137 from weapons fall-out has been studied rather extensively in Norway, particularly in the late 50s. Some relevant references are (Ref. BE59 and LU62). One of the observations made is that the deposited materials in the different layers of snow were relatively stable as long as no melting of snow occurred. But as soon as melting or rain happened, the activity was rapidly washed down through the snow, increasing the concentrations in the bottom layers.

In (Ref. LU62) is also described an experiment in which carrier-free Sr-90 and Cs-137 was sprayed onto a 60 cm thick layer of snow. After a few days, the activity had migrated through the snow layer. This is said to be under conditions where "no obvious melting took place", but this implies that the temperature probably was approaching the melting point.

In connection with the aforementioned SNSF project an experiment was performed, in which a color compound was sprayed on snow. By cutting out vertical samples, migration of the color in the snow could be followed. The horizontal migration was also observed. It was found that the color could migrate sideways several meters, along the horizontal layers of icy snow in the snowpack.

The rapid migration of impurities in snow under melting conditions can be explained in the following way. When deposition takes place in cold weather, the impurities will deposit on the surface of the snow crystals. When melting starts, the outer layers of the crystals will melt first, and the impurities follow this water down through the layers of snow. The melting process is found to sometimes take place even if the air temperature is below the freezing point, and it is reasonable to assume that the presence of the impurities has caused a local lowering of the melting point.

8.3.2 Description of the lycimeter experiment

These experiments are described in (ref. QV84b and QV86). The cesium experiment was performed as part of the previous project period within the Nordic Safety Program, and the strontium experiment was performed as part of project *AKTU-255*. The experiments were performed in a lycimeter belonging to the Agricultural University of Norway, and situated at their facilities at Ås, roughly 30 km Southeast of Oslo. This lycimeter is routinely used for run-off studies. The plot is 75 m^2 (3.75 m x 20 m). It is isolated from the surrounding soil on all sides and in the bottom. It contains clay soil, and the vegetation is mainly various types of grass (mostly timothy, but also some meadow fescue and clover). The slant of the plot is 1:20. The lowest part of the plot is at one of the short sides, where all run-off water is collected in a large container.

The measurements of content of radioactive materials were performed at the Institutt for energiteknikk, Norway.

8.3.2.1 The cesium experiment

The first experiment was initiated on the 14. February 1984. The lycimeter was then covered with 12 - 15 cm coarse-grained snow with a hard, undisturbed top layer of crusted snow. The ground was frozen. Highest air temperature on this day was -3°C, and the temperatures in the preceding and following nights were below -10°C.

The plot was contaminated with 100 µCi Cs-134 dissolved in 5 liters of water. About 0.01 mg inactive Cs was used as carrier. The solution was sprayed onto the surface using a garden sprayer with fog nozzle. The lower 2 meters of the plot were not sprayed, for fear of contaminating the sampling position. The sprayer was subsequently cleaned, using two times one liter of water, which was also distributed over the plot.

Melting of the snow started on the 5. April (7 weeks after contamination), and lasted for 8 days, till 13. April, when all the snow was melted and the run-off stopped. The run-off water was collected, and one or two samples of 0.4 - 1 liter were taken daily.

8.3.2.2 The strontium experiment

The second experiment was initiated on the 18 March 1986. The lycimeter was then covered with about 35 cm coarse-grained snow with a relatively thin top layer (1 - 2 cm) of light, new snow. The ground was frozen. The air temperature was +2°C, calm, overcast and some mist.

The plot was contaminated with 18.5 MBq (500 µCi) Sr-89 dissolved in 5 liters of slightly acid (nitric acid) water. The solution was sprayed onto the surface using a garden sprayer with fog nozzle. The lower 2 meters of the plot were not sprayed, for fear of contaminating the sampling position. The sprayer was subsequently cleaned, using one liter of water, which was also distributed over the plot.

Melting of the snow started on the 21 March and lasted to 5 April, followed by somewhat colder weather. All the snow had then melted. The next precipitation came 23 April, and was also collected. It contained very little activity, and the experiment was terminated.

8.3.3 Run-off from meadow in spring

All numerical results in this chapter are normalized to refer to inactive cesium and strontium. If it is desired to relate the results to specific cesium or strontium isotopes, the proper radioactive decay should be applied.

The migration experiments reported here represent, of course, only the particular sequence of weather and snow conditions in the respective springs during which they were performed. Such experiments ought to be repeated under varying conditions. The run-off can be expected to depend not only on the weather conditions after, but also upon the conditions before contamination was carried out. One important aspect is probably the absence or presence of frost in the ground, and of an ice layer on the ground, under the snow cover. There are a great number of combinations, and in these experiments one just has to accept the conditions of one winter and spring as they happen to be in that particular year.

Both of these experiments demonstrate that accident consequence calculation assumptions, as they are currently applied, tend to overestimate the consequences resulting from an accident taking place under winter conditions; but concerning this particular aspect, the overestimation is not very large.

8.3.3.1 Results for cesium

Altogether 7890 liters of run-off water was collected. This is equivalent to 105.2 mm precipitation on the plot. About 30% of the distributed cesium was contained in this water. The results of the actual measurements are shown in Table 8.3, to illustrate the time sequence of removal.

The migration experiment shows a significant, although modest, difference between summer and winter conditions. Roughly 30% of the Cs-134, with which the area was contaminated, ran off with the melt water, which is significantly more than would be expected during summer conditions. Following an accident situation, this would have given a corresponding reduction in the longterm doses.

Date (1984)	Run-off water (mm)	Conc. (nCi/l)	Recovered Cs-134		
			(μCi)	(% of added activity in run-off)	(% of added activity, cumulative)
April 5	3.5	13.9	3.64	3.6	3.6
April 6	5.1	9.6	3.67	3.7	7.3
April 7	6.9	6.0	3.12	3.1	10.4
April 8	16.3	6.5	8.04	8.1	18.5
April 9	6.9	5.2	2.71	2.7	21.2
Apr. 10	12.9	2.4	2.29	2.3	23.5
Apr. 10	13.7	2.0	2.02	2.0	25.5
Apr. 11	7.7	2.1	1.22	1.2	26.7
Apr. 12	13.7	2.0	2.06	2.1	28.8
Apr. 12	12.0	1.4	1.28	1.3	30.1
Apr. 13	6.5	1.2	0.59	0.5	30.6
Sum	105.2		30.64		30.6

Table 8.3. Collected data on cesium from run-off experiment.

8.3.3.2 Results for strontium

Altogether 15450 liters of run-off water was collected. This is equivalent to 206 mm precipitation on the plot. About 54% of the distributed strontium was contained in this water. The results of the actual measurements are shown in Table 8.4.

The migration experiment shows a significant, although not very large, difference between summer and winter conditions. Roughly 54% of the Sr-89, with which the area was contaminated, ran off with the melt water. Following an accident situation, this would have given a corresponding reduction in the longterm doses.

Date (1986)	Run-off water (mm)	Recovered Sr-89		
		(MBq)	(% of added activity in run-off)	(% of added activity, cumulative)
March 21-22	22	5.13	27.77	27.77
March 23	17	0.53	2.84	30.61
March 23	20	0.62	3.35*	33.96
March 23	14	0.42	2.28	36.24
March 25	17	0.66	3.59	39.83
March 27	17	0.36	1.96	41.79
March 29	9	0.19	1.05	42.84
March 30	17	0.30	1.60	44.44
March 31	10	0.25	1.36	45.80
April 2	26	0.75	4.08	49.88
April 3	17	0.44	2.39	52.27
April 5	17	0.29	1.56	53.83
April 23	3	0.03	0.16	53.99
Sum	206	9.97		53.99

* Overflow on the night between 22 and 23 March.

Table 8.4. Collected data on strontium from run-off experiment.

8.4 Decontamination during winter

The experiments on decontamination were performed several years apart. The earlier experiments were part of the preceding project period of the Nordic Safety Program, and are reported in (ref. QV84b). The later experiments were performed in project *AKTU-255*, and will be described in a report to be published (ref. QV90).

When consequences of atmospheric releases caused by a large reactor accident are calculated, it is usually found that the major contributions to the dose are via nutrition and exposure to radiation from radioactive materials deposited on ground. Accordingly, one of the ways in which the doses resulting from a large accident can be reduced, is by decontamination of urban and/or agricultural surfaces. When decontamination cost and feasibility has previously been assessed, it has up till now never been considered that the accident may take place when the ground is snow-covered. In the Nordic countries we have winter conditions 4 to 6 months a year. There are good reasons to believe that decontamination may be both simpler to perform and less costly during winter conditions.

Both for economical and practical reasons, the number of experiments that can be perform during one winter are limited, while the possible combinations of weather and surface conditions are very numerous. A complete survey of all possible conditions is almost impossible to achieve, but the experiments conducted thus far, represent a relatively wide range of different conditions.

The experiments have been performed using standard snowremoval equipment and routines, although they have been limited to rather small-scale equipment. It would have been interesting to carry out experiments using the much larger machines ordinarily used on roads, but this could not be handled with the modest funds available. A non-radioactive contaminant was used for the decontamination experiments.

8.4.1 Experiment carried out in 1984

The experiment was carried out 21. March 1984. A small parking lot of ca. 30 m² at the Institutt for energiteknikk, Norway was used. 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.

10.5 grams of Cu as $CuSO_4 \cdot 5H_2O$ was dissolved in 5 liters of 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°C when contamination was carried out. The temperature in the snow was somewhat lower, as the air temperature had been down around -10°C 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 · 1m squares. All remaining <u>loose</u> snow and ice was collected in each square separately, put in plastic bags and measured. The division into squares was actually not important for the outcome of the experiment. It gives information on how evenly the remaining contaminant is distributed over the area, and it may perhaps give some indication of how evenly the contaminant was distributed before decontamination started. The concentrations in the individual samples actually varied over quite a wide range (from 0.1 to 12 mg/l). But the very high values are found only in two of the 30 squares, and these are adjoining, probably indicating that in these squares the snow left after shoveling was from the original top layer of snow.

100 ml from each sample was removed for determination of Cu content, using atomic absorption methods. Samples for background and standard were collected before contamination.

In total 91.5 mg Cu was recovered (contained in 41.8 liters of water). This is equivalent to 0.87% of the contaminant; which means that <u>more than 99% of the contaminant had been removed</u>. 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 it relatively easy to remove much of the loose snow with the shovel. Conditions can not be expected to be as favorable in a real situation.

The spray mist particles froze when they came into contact with the cold snow, and stayed on the surface of the snow. If deposition takes place with rain, or if rain occurs after deposition, the deposited material will probably penetrate the snow layer. By sub-zero temperatures, however, the deposited material is expected to stay on the snow surface, and the conditions of the experiment gives a reasonable representation of this situation.

8.4.2 Experiments carried out in 1988

Two winter decontamination experiments were carried out in March 1988. In the experiment carried out in 1984, conditions were such that one could expect a quite efficient removal. The surface was covered with un-packed snow on top of solid, smooth ice. But such conditions are not typical of urban areas. The main roads will most of the time be almost free of snow. Smaller roads and sidewalks will be partly snow-covered. This snow, however, will most of the time be quite hard-packed. Of the two experiments carried out in 1988 one was carried out on a road where there was only a quite thin layer of hard-packed snow, while the other case perhaps was typical of a situation on side-walks; with relatively compact snow, topped by a layer of new-fallen snow a few centimeters thick.

In the experiment carried out in 1984, the surface was ice-covered underneath the snow, so that it was easy to collect the snow remaining after the snow-shovel had been run over the area. In the experiments carried out in 1988 it would have been impossible to be sure to collect all or most of the remaining snow. Therefore the measurements had to be made on the snow removed by the snow-shovel instead of on what remained. This again means that all snow shoveled had to be collected and melted.

The first of the two experiments carried out in 1988 was performed the 8. March on an asphalt-covered road at the Institutt for energiteknikk, Norway. There is regular, though not heavy, traffic on the road. There was a roughly one half centimeter thick layer of ice, covered with 1 to 1.5 centimeters of hard-packed snow. Three to four days had passed since the last snow-fall, and the road had been used regularly since then, both by pedestrians and cars. The snow and ice cover was quite irregular, and there were some completely bare patches.

Like in the previous experiment cuppersulphate was used as contaminant. 400 mg was dissolved into 100 ml of water and sprayed upon the chosen surface, using a simple hand sprayer. The surface contaminated was a square of $2 \cdot 2$ meters bordering the edge of the road, with no bare patches. The road was closed for traffic during the experiment.

The air temperature was $-6.5^{0}C$ and the temperature in the snow was $-8^{0}C$. The experiment was performed in the early forenoon, under sunny and calm weather conditions.

Immediately after contamination the snow was removed using a tractor with a shovel. The shovel was narrower (1.8 meters) than the width of the contaminated area (2 meters). Accordingly the tractor was driven twice over the stretch, with overlap. The tractor collected the snow from a much longer stretch (starting roughly 4 meters below and stopping about 8 meters above) than from the area actually contaminated. At the end of this stretch some snow, which had been pushed in front of the shovel, was still lying on the road in front of it. This heap of snow, and the the snow in the shovel, were put in two different buckets. Some snow was pushed to the side of the road by the shovel, but this was just left there, and was neither collected nor measured. What was pushed towards the middle of the road during the first run over the contaminated area, was collected the second time the shovel ran over the area.

The water volume of the snow collected by the shovel was 13.6 liters, and of what was pushed in front of the shovel 10.5 liters. Three parallel water samples from each bucket were measured. The average cupper concentration was 5.45 mg/l and 1.9 mg/l respectively. The variation between the individual samples was very small. A background for cupper of less than 0.01 mg/l was also determined from collected non-contaminated snow.

Expressed in percentage of the original amount of contaminant, this means that 18.5% of the contaminant was removed with the shovel, and about 5% was in the heap of snow pushed in front of the shovel. If pushed further, more of this snow would of course eventually end up inside the shovel, so that the conclusion is that this technique <u>can remove roughly 25% of the contaminant under these conditions</u>.

The second of the experiments carried out in 1988 was performed shortly before noon of 25. March. In this experiment the surface was also a road, but with less traffic than the one in the other experiment. It was covered by a few centimeters of rather compact, but somewhat porous snow. There had been a number of days with above-freezing temperatures. On top of the old snow layer was a roughly 3 centimeter thick layer of fresh, slightly humid snow.

Cuppersulphate was used for contaminant, just like in the preceding experiments, but the amount in this case was 1000 mg. The procedure was the same as in the other 1988 experiment, both for application and removal.

The air temperature was $+1.5^0$ and the temperature in the snow -0.5^0. There was some wind, but not enough to make the snow drift.

The amount of snow collected was much larger than in the preceding experiment. When melted, the volume of the snow in the shovel was about 59 liters, and of what was pushed in front of the shovel 24.5 liters. The average cupper concentrations were 15.0 mg/l and 4.5 mg/l respectively.

Expressed in percentage of the original amount of contaminant, this means that 88.5% of the contaminant was removed with the shovel, and about 11.0% was in the heap of snow pushed in front of the shovel. From this can be concluded that by this technique and under these conditions one can <u>remove roughly 99.5% of the contaminant</u>.

8.4.3 Conclusions from the winter decontamination experiments

The three experiments performed, under different snow conditions, gave as results that more than 99%, about 25% and 99.5% of the contaminant was removed, using ordinary snow removing equipment.

Two of the experiments were performed under quite favorable snow conditions, that may be rare in urban areas. On smaller roads in cities and on sidewalks there is often a layer of hard-packed snow and ice, that can not be removed except by use of different and more time consuming methods.

The weather conditions were also favorable under these experiments. Strong wind will spread the contaminant together with the snow, and distribute it more evenly throughout the snow layer. Snowfall during decontamination will not matter; but 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. It has, however, not been possible to examine these conditions within project *AKTU-255*.

These experiments were conducted under a few out of a large number of different possible conditions, both regarding ice, snow and weather conditions. 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.

9. RESUSPENSION OF DEPOSITED ACTIVITY, PROJECT AKTU-260

The population may, after a release of radioactive material into the atmosphere, be exposed in several ways. Direct exposure by the passing cloud and inhalation of passing radionuclides are of short duration, while direct exposure from deposited radionuclides, intake of radionuclides through food and inhalation of resuspended radionuclides may contribute to the dose for many years after an accident. The relative dose contribution from each exposure pathway is very much dependent on released radionuclides, weather conditions during the release and the environment where the radionuclides are deposited. In accident earlier consequence assessments (like ref. WA75) the main exposure of the population was usually found to be due to external radiation from the ground and intake of radionuclides with the food, while resuspension was a pathway of little importance compared to the others.

The data upon which these evaluations of the resuspension pathway were based were from weapons tests sites in the Nevada desert area. Resuspension under other conditions may be rather different. In particular the suitability of these data for calculation of resuspension in urban areas could be doubted. In urban areas one can imagine that there may be considerable resuspension even at low wind speeds, as an effect of the traffic. Cars may resuspend activity both as a result of the air turbulence caused, and as a result of the wear on the road surface caused by the wheels.

A project on resuspension was proposed from the beginning of the AKTU project period. Possible approaches were discussed with the Norwegian Air Research Institute and with Risø National Laboratory. One proposed approach involved the use of gold as tracer, due to the extremely low ambient air concentrations, and because extremely small amounts can easily be measured by activation analysis. Both financial and manpower resources, however, were insufficient at that time, and no institution was sufficiently interested to take on responsibility for performing this project.

After Chernobyl the situation was completely different. The tracer was suddenly there, to be used, and the National Defence Research Establishment, Sweden has performed project AKTU-260 on resuspension, using their air filter stations. The results of these investigations are reported in (ref. BJ87 and BJ88).

9.1 Earlier experiments on resuspension

As already mentioned, most information on resuspension up till now describes the behaviour of nuclear weapons fall-out in desert areas. Some additional work has, however, also been performed. A survey of these projects was prepared as part of the previous Nordic Safety Program (ref. JE84), and is briefly presented in this subchapter, along with some rough evaluations of the results of these projects and comparison with the conditions concerning resuspension that were used in the US Reactor Safety Study (ref. WA75).

In the US Reactor Safety Study a time-varying resuspension factor was used. The time dependence reflected downward migration of cesium in soil or other processes with similar effect upon the availability of cesium for resuspension. The resuspension factor varied from 10^{-5} m^{-1}

for freshly deposited material to 10^{-9} m^{-1}. It is said in the Safety Study that these values probably are too low for urban areas, but alternative values are not given.

The resuspension factor is defined as the ratio between air concentration and ground concentration. What is easily forgotten in this connection is that the air concentration is not a welldefined quantity. It will obviously vary both with the height above ground and with the weather conditions. The interesting height above ground is of course some 150 to 170 cm, where breathing usually takes place. Resuspension should be expected to be more pronounced under dry weather conditions. The effect of the wind speed, however, is difficult to predict. Higher wind speeds will lift more material from the ground, but will at the same time give increased dilution. The conclusion in (ref. JE84) is that higher wind speeds give higher concentrations of resuspended material. The size of the area from which resuspension takes place is also important, if ground concentrations are non-uniform.

Among the many references cited in (ref. JE84) is one of particular relevance in connection with traffic and resuspension (ref. SE76). In the experiment described there, a powder is distributed on a surface and cars drive across it. Among the results, it is found that for the fast driving the fraction of material resuspended per vehicle pass is about 10^{-2}. However, this piece of information is too unqualified to be compared with resuspension in a strict manner. If one does not know the concentration in air, the resuspension factor can not be determined. In order to "translate" this fraction of resuspended material, one needs information on how long the material will stay resuspended; and not least, one needs information on the vertical concentration distribution. If we assume that the resuspended materials will stay in suspension somewhat longer than the time interval between two car passes, and that the resuspended material is evenly distributed in a two meter thick layer of air (somewhat higher than a car), the corresponding value for the resuspension factor will be ca. 10^{-2} m^{-1}.

Accordingly, the value for resuspension factor estimated from the experiments in (ref. SE76) is very much higher than the values previously used in accident consequence assessments. As a matter of fact, it is so high that it seems it would be quite an efficient method for decontamination. However, it is probable that cesium deposited upon a road surface will behave rather differently from a powder distributed on a road. The results of the experiment of (ref. SE76) are probably not relevant in the present context.

There are additional interesting references to work on resuspension, but they will not be discussed in the present report. For more information, see (ref. JE84).

9.2 Pre- and post-Chernobyl resuspension

Resuspension of already deposited radioactive material has before the Chernobyl accident only influenced the air concentration in Sweden of Cs-137 slightly. It was only during the last years before the Chernobyl accident that any indications were observed concerning resuspension of fall-out cesium. After the Chernobyl accident though, the air concentration of radioactive Cs-137 in Sweden is dominated by resuspended radionuclides, while the contribution from the stratosphere is almost neglectable at all sites. Prior to the Chernoby accident no Cs-134 was present in the environment, as nuclear weapons do not produce significant amounts of this isotope. The releases from Chernobyl accident did not penetrate the tropopause, so the presence of Cs-134 in ground level air is solely due to resuspended activity, or new releases.

9.3 Air concentration measurements

Air samples are collected all over Sweden, as shown by Figure 9.1. The sampling stations marked with • belongs to the surveillance network of high volume (1100 m^3/h) air sampling devices designed for verification of nuclear test bans (ref. VI82). The stations in the Gävle area, marked □ , are mobile air sampling devices specially made for this study. These mobile stations have a capacity of filtrating about 330 m^3/h of air through a $0.58 \cdot 0.58\ m^2$ glass fiber filter. Filters are changed two or three times a week at all stations, and measured weekly on Ge-detectors, giving a detection limit for both Cs-134 and Cs-137 of less than 0.1 and 0.5 $\mu Bq/m^3$ for the stationary and the mobile air sampling stations, respectively. This means that the air concentration of both the Cesium isotopes has been above detection limit for all the sampling stations at all time.

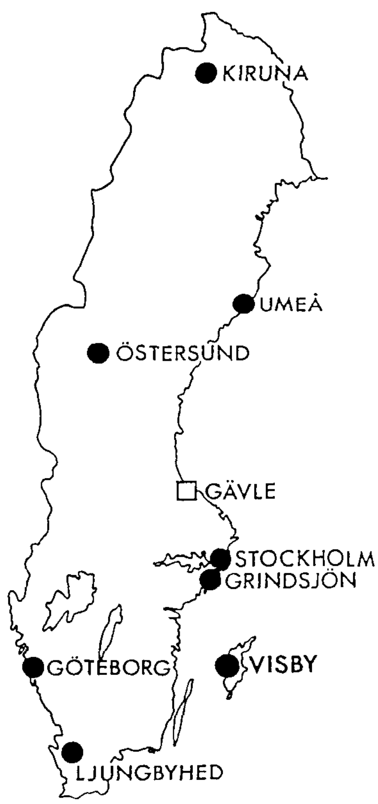


Figure 9.1 Air sampling stations in Sweden.

For the permanent air sampling stations (except Visby), belonging to the surveillance system, the air concentration has been measured continuously since long before the Chernobyl accident. The Visby fan was started in the middle of May 1987.

In Gävle the air sampling stations were originally put up to study differences in resuspension due to the local environment, and especially in urban areas. From the start in May 1987 eight air sampling stations were running in the Gävle area, with two samplers each in city, forest, grass and farming areas, respectively. From October 1987 four of the air samplers were stopped, but all types of environments are still reprecented by one air sampler each.

9.4 Deposition measurements

Two types of measurements have been performed for the purpose of estimating both deposited activity concentration and activity depth distribution in the soil. The total deposition at each site has been measured by taking at least 3 soil samples at each site to a total depth of 10 cm. These soil sample were dried, crushed and homogenized and thereafter measured on Ge-detectors in 180 ml plastic tubes. The uncertainty in the measured total deposition is totally dominated by the variability of the deposition, and the uncertainty of the activity concentration in each soil sample can be disregarded.

On each air sampling site in situ gamma-ray spectrometric measurements have been performed with a 35% HpGe-detector one meter above ground. The germanium detector was calibrated with point sources at great distances, as normally done, and the surface equivalent ground concentration was calculated for each site, assuming the relaxation length to be zero. By comparing these ground contamination results with the soil sample results, effective relaxation lengths were determined. These are valid only for the in situ gamma-ray spectrometric measurements as they also include a roughness correction. This means that the calculated relaxation length slightly overestimates the actual depth distribution of cesium in the ground, depending on the roughness of the area measured.

The total amount of contaminant per unit area, Sa, can be calculated from the following equation:

$$Sa = \varrho\int_0^\infty Sm(z)\exp(-\alpha z)dz = \frac{\varrho}{\alpha} Sm(0)$$

where Sm(z) is the activity concentration per unit mass of soil at depth z, ϱ is the soil density, and $1/\alpha$ is the relaxation length. From this equation we can see that ϱ/α is a rough index of surface activity.

9.5 Resuspension factor

The simplest resuspension factor Rf (ref. LA71) is defined by:

$$Rf = X\ (Bq/m^3)/Ssurf\ (Bq/m^2)$$

where X is the air concentration and Ssurf is the surface activity concentration. The air concentration causes little problem, but the definition of surface activity is difficult. The surface activity concentration should only include the part of the ground contamination available for resuspension. A time dependent resuspension factor approach has therefore been applied (ref. AN75):

$$Rf(t) = X(t)/Stot$$

where Stot is the total amount of contaminant per unit area. Stot can in many cases be considered as constant with time, which makes the formula easier. The time dependent resuspension factor has been proposed as:

$$Rf(t) = Rf(0)\exp(-\lambda t)$$

For plutonium the halflife that corresponds to λ has been calculated from environmental measurements to between 35 (ref. LA69) and 45 days (ref. KA68) during the first weeks post deposition. The resuspension factor at time of deposition Rf(0) has been estimated to $10^{-4}\ m^{-1}$. The models above give unrealistic low values for the resuspension factor 10-20 years after deposition. Measured resuspension factor 17 years after deposition was $10^{-9}\ m^{-1}$ (ref. AN74), and therefore a model that reflects this value better was proposed by (ref. AN75).

$$Rf(t) = 10^{-4}\ \exp(-\lambda\sqrt{t}) + 10^{-9}\ (m^{-1})$$

where λ is 0.15 ($1/\sqrt{\text{days}}$).

9.6 Particle size measurements

Four weeks during September 1988 two air samplers equipped with multistage impactors were running at two sites in the Gävle area (city and Lillgården). The airflow (53 m^3/h) used, was not the calibration flow for the impactors, so the correct cut off sizes are not accurately known. By comparison with the size distribution of Be-7 and roughly estimated cut-off particle sizes, the results can help us to get a better understanding of the resuspension processes. The calibration of the air samplers to the proper air flow will be performed during 1989, together with additional measurements.

9.7 Results and discussion

During the first weeks or months after the Chernobyl accident considerable amounts of fission products were floating around in the atmosphere. It is therefore difficult to separate between the contributions due to resuspension and fresh radioactivity, not yet deposited, during May and June 1986. The first two months post Chernobyl, has therefore been excluded from this study.

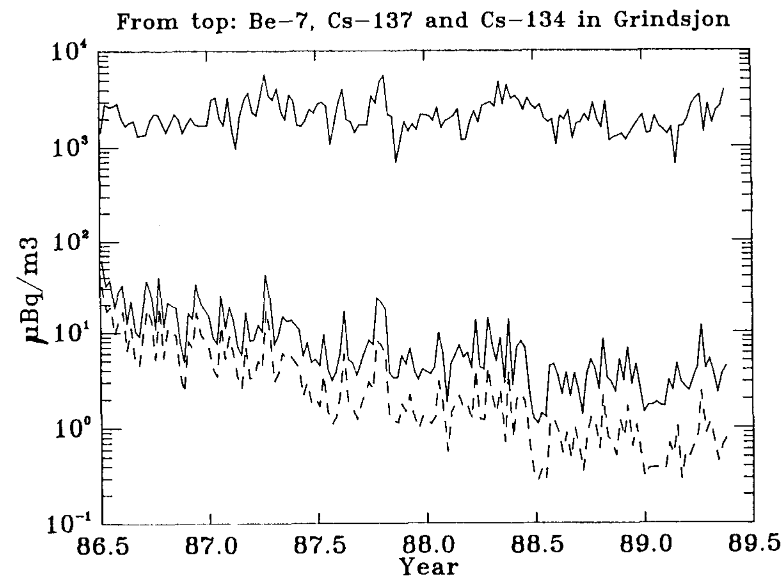


Figure 9.2 Air activity concentrations of Be-7, Cs-134 and Cs-137.

The activity concentrations of Cs-137 and Cs-134 will follow each other, Figure 9.2. As Cs-134 is detectable at all sampling stations during the whole period, and as Cs-134 is not present after the nuclear weapons tests, only Cs-134 is studied in this project. All resuspension factors and conclusions are also applicable to Cs-137.

The air concentration of Be-7 is presented in Figures 9.3, 9.4, 9.5 and 9.6 for the 12 sites where air sampling has been performed continuously for at least 1.5 years. As can be seen from these figures, both the air concentration and the seasonal variation of the cosmogenically produced Be-7, is very similar for all sites. For "neighbouring" sites, like the stations in the Gävle area, there is a striking resemblence in Be-7 concentration. Even if Stockholm and Gävle (> 100 km) is compared, Figure 9.7, only small variations can be seen. This was expected, and is typical for a globally distributed radionuclide. It can also be noticed from these measurements, that Be-7 seems to be an extremely good normalizing factor for air samples taken over rather long distances (< 100 km) and long time periods (weeks).

If we now regard the air concentration of Cs-134 (Figures 9.8, 9.9, 9.10 and 9.11), this can be rather well correlated to the deposition after the Chernobyl accident (Figure 9.12) in the area surrounding each air sampling station. Seasonal variations occur somewhat like for Be-7, but due to different reasons. The seasonal variation for Be-7, with one peak in spring and a smaller one in autumn, comes from the change of the tropopause, while the variation for Cs-134 is due to weather conditions, and the lack of binding vegetation during autumn, winter and spring. Stations that were extremely similar for Be-7, as Stockholm and all the sites in the Gävle area, are not at all similar regarding the cesium isotopes. The contribution over longer distances (> 100 km) can be regarded as small except during extreme circumstances with high wind speeds. Only occasionally, at times with strong wind from the North, has the air concentration in Stockholm been significantly increased due to resuspension from the high deposition areas starting 50-60 km North of Stockholm (Figure 9.12). Some indications of much longer transport of resuspended radionuclides, have been noticed in Visby and Grindsjön. At some occasions the nuclide concentration has been increased significantly during periods with strong South-Easterly wind, coming from the Chernobyl area. Most of the time though, the air concentration of the cesium isotopes at all sites can be expected to represent the local resuspension.

As described above, the resuspension factor can either be correlated to the surface activity, So, or to the total deposited activity concentration with a correlating half-life. In Table 9.1 the total deposited activity concentration of Cs-134 is given together with the surface equivalent activity concentration (surface equivalent concentration is the activity concentration given by in situ gamma ray spectrometry assuming surface deposition) measured with in situ gamma ray spectrometry. From these measurements relaxation lengths can be calculated (Figure 9.13). The relaxation lengths can be regarded as factors showing the fraction of the total deposited activity available for resuspension. This relaxation length will change with time, and can be incorporated in the timedependent resuspension factor. We will therefore start with calculating the resuspension factors correlated to the total deposition (Table 9.2).

Sample site	Soil sample (kbq/m^2)	In-situ measurement* (kbq/m^2)
Grindsjön	739	320
Visby	1270	1060
Kiruna	164	154
Göteborg	1480	497
Ljungbyhed	634	529
Umeå	13710	7110
Esplanaden Gävle	79394	24600
Kastvallen Gävle	89580	33700
Östersund	1196	821
Stockholm	352	350
Avan Gävle	99900	39170
Lillgården Gävle	43580	19100

* Surface equivalent activity.

Table 9.1 Deposition of Cs-134 measured by soil samples and in-situ gamma-ray spectrometry.

Year Quarter		Site							
		G	I	K	O	T	U	X1	X2
86	3	236.4		206.6	45.1	146.1	45.8		
86	4	149.8		97.4	19.4	103.2	26.3		7.3
87	1	97.2		104.0	28.4	106.4	12.8		5.1
87	2	104.1	53.1	96.2	26.2	132.2	29.0	17.4	15.5
87	3	43.5	39.5	57.9	13.2	40.9	16.7	9.1	4.8
87	4	64.3	34.7	77.0	16.1	74.1	9.7	4.3	2.5
88	1	44.7	27.7	63.7	11.1	53.1	7.3	2.7	1.9
88	2	51.5	34.9	64.2	14.0	51.5	15.3	17.4	10.4
88	3	19.0	12.4	79.9	5.7	20.4	16.5	4.7	5.1
88	4	33.0	17.8	37.2	6.4	32.9	6.7	3.1	2.6
89	1	17.2	11.9	51.9	6.3	27.9	9.8	5.3	2.9

Continuation of the Table above

Year Quarter		Site				
		Y	A	X4	X5	Average
86	3	109.3	406.0			170.8
86	4	85.1	271.6			95.0
87	1	77.9	235.7			83.4
87	2	44.9	243.7	5.1	19.0	65.5
87	3	29.7	110.1	5.4	13.7	32.0
87	4	24.1	126.9	2.2	6.4	36.9
88	1	28.6	99.5	1.7	4.8	28.9
88	2	23.7	128.0	9.5	13.5	36.2
88	3	29.0	58.6	5.3	11.7	22.4
88	4	22.9	103.3	2.2	7.7	23.0
89	1	25.9	45.1	3.0	6.7	17.8

Table 9.2 Resuspension factors ($\times 10^{10}$).

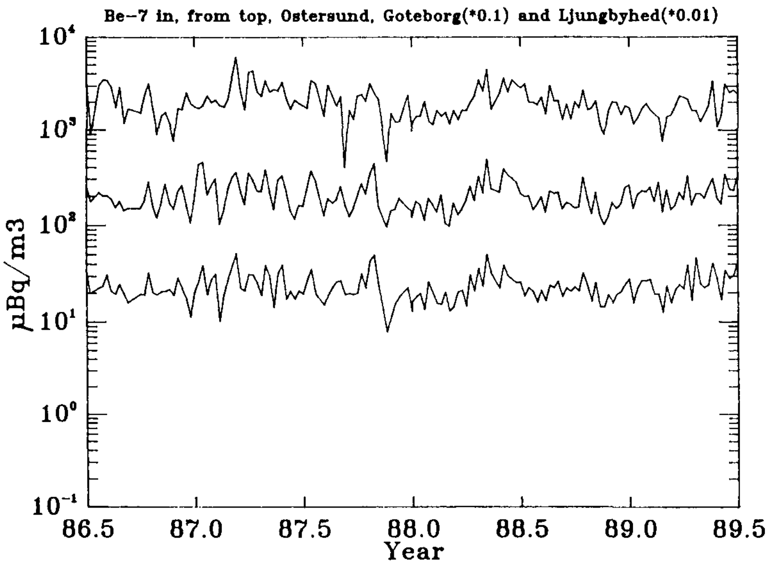


Figure 9.3 Air activity concentrations of Be-7 for sites Östersund, Göteborg and Ljungbyhed.

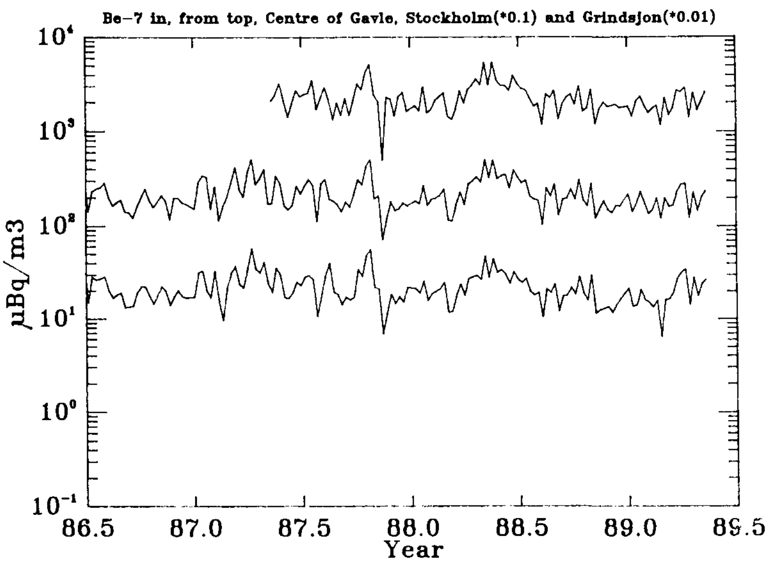


Figure 9.4 Air activity concentrations of Be-7 for sites Center of Gävle, Stockholm and Grindsjön.

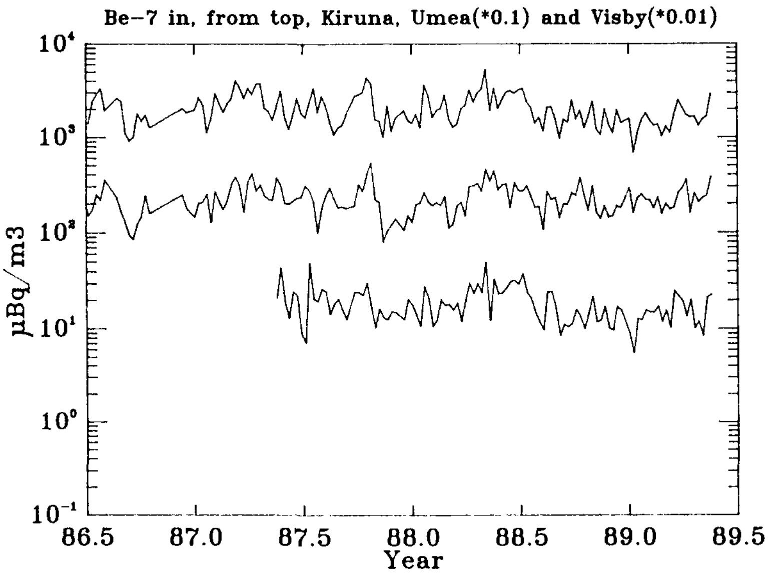


Figure 9.5 Air activity concentrations of Be-7 for sites Kiruna, Umeå and Visby.

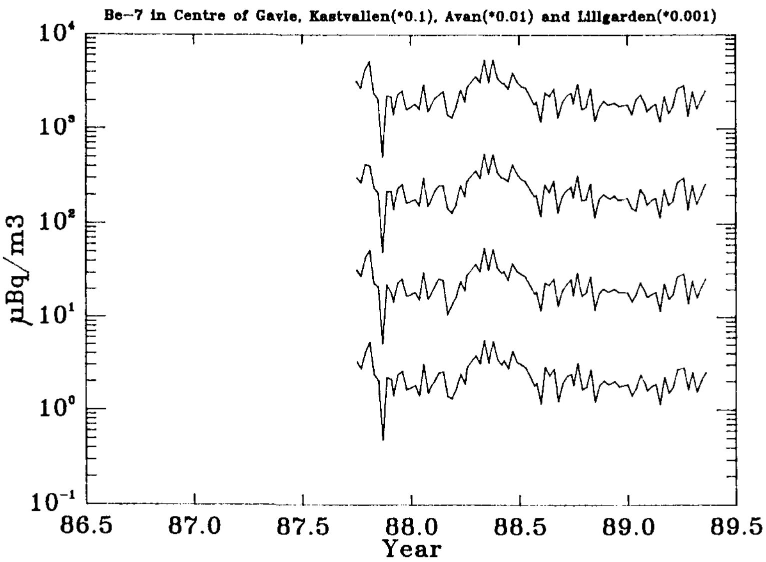


Figure 9.6 Air activity concentrations of Be-7 for sites Center of Gävle, Kastvallen, Avan and Lillgården.

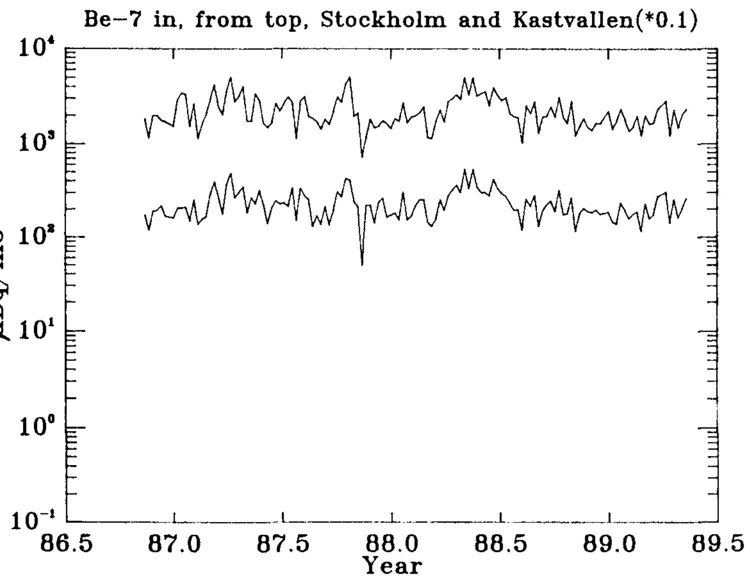


Figure 9.7 Air activity concentrations of Be-7 for sites Stockholm and Gävle.

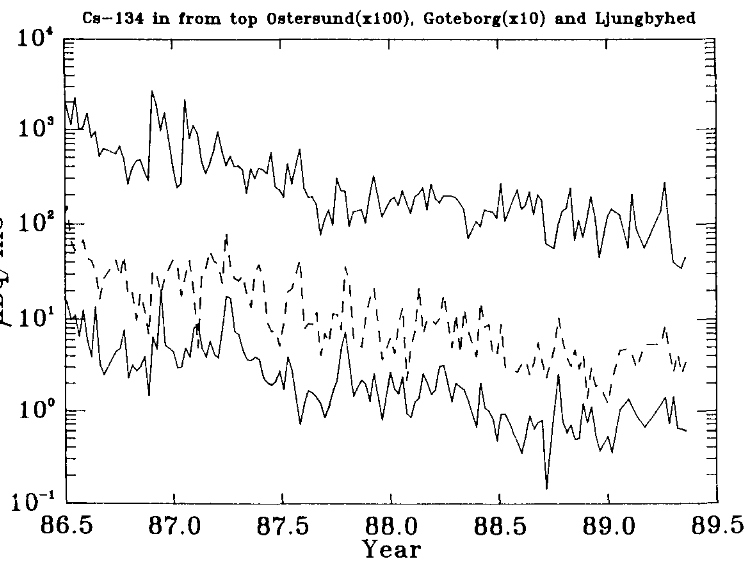


Figure 9.8 Air activity concentrations of Cs-134 for sites Östersund, Göteborg and Ljungbyhed.

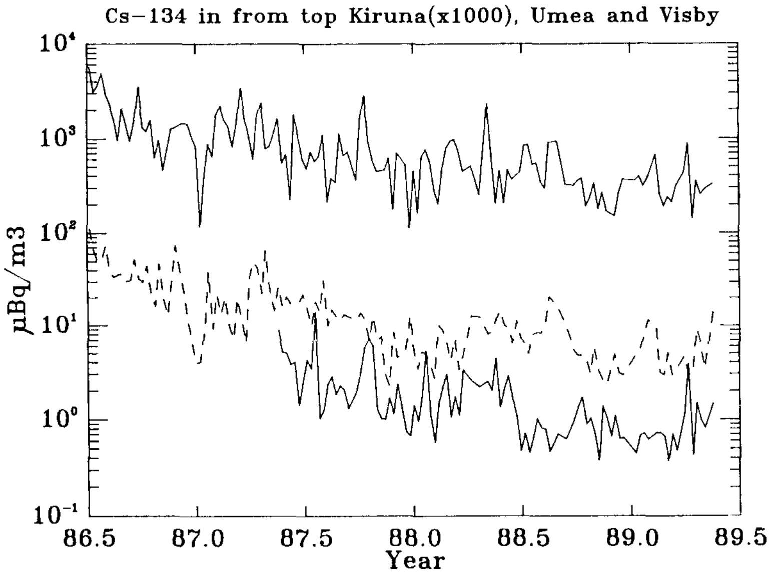


Figure 9.9 Air activity concentrations of Cs-134 for sites Kiruna, Umeå and Visby.

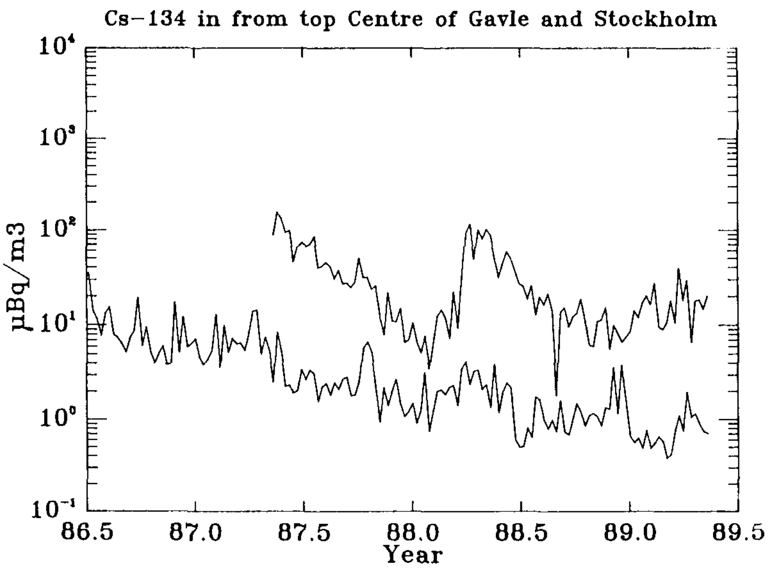


Figure 9.10 Air activity concentrations of Cs-134 for sites Center of Gävle and Stockholm.

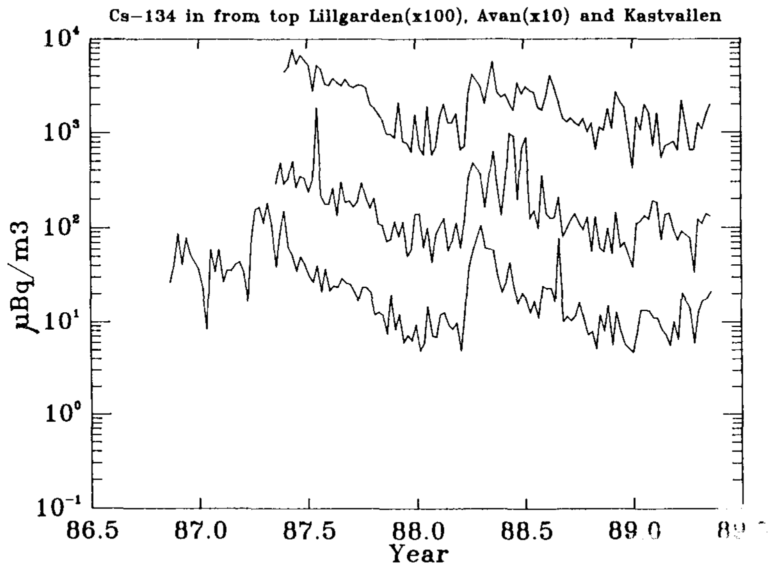


Figure 9.11 Air activity concentrations of Cs-134 for sites Lillgården, Avan and Kastvallen.

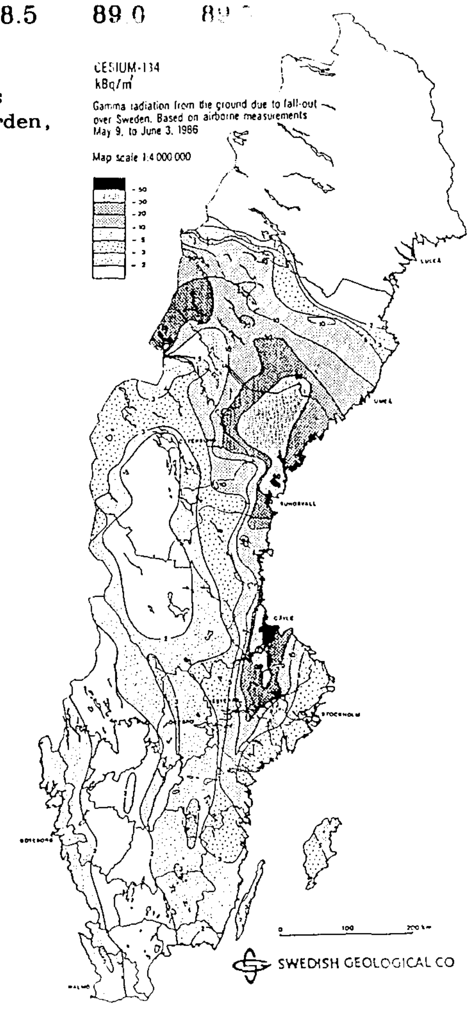


Figure 9.12 Deposition after the Chernobyl accident around sampling stations.

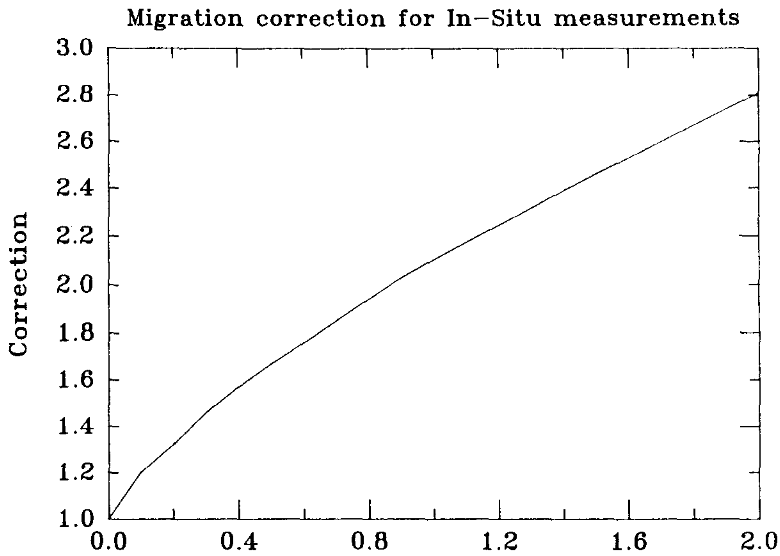


Figure 9.13 Relaxation lengths.

The highest resuspension factor, Figure 9.14, occured in Stockholm during the third quarter of 1986, thus 2 to 5 months after the Chernobyl accident. This resuspension factor $4.1 \cdot 10^{-8}$, though the highest measured, is still at least a factor 1000 lower than previous investigations have indicated (ref. OK77), or calculated from the models described above. On sites where the deposition occured as dry deposition (Stockholm, Grindsjön, Ljungbyhed, Kiruna, Östersund and Visby) significantly higher resuspension factors have been measured than on sites where the dominating activity came down with rain (Göteborg, Gävle and Umeå). In Table 9.1 the same type of relation can be noted concerning the relaxation length. Thus in wet deposition areas, the relaxation length is higher than in dry deposition areas. By comparing Tables 9.1 and 9.2 we can see that the longer relaxation length the lower resuspension factor, as could be expected. The resuspension is almost proportional to the $1/\alpha$.

Due to the great seasonal variaton in resuspension, a good and useful half-life to be used for resuspension is very difficult to obtain. The seasonal variation is much greater than the decline in resuspension. At present the best rough estimate is a half-life of about 0.9 years.

The seasonal variations with peaks in spring and autumn, can be explained by less vegetation binding the top soil during these periods. Higher windspeeds, especially at ground surface, is also a factor that will influence the resuspension factor positively during spring and autumn. At times with dry weather and high windspeeds much higher resuspension factors than average can be expected. Statistical analysis between resuspension and weather parameters have yet not been made, but will be made during 1990.

The size distribution of resuspended activity, will influence how far a resuspended particle will travel before re-deposition. The inhalation doses will also be influenced by the size distribution. Some measurements were therefore performed in the Gävle area during 1988. The cascade impactor was not calibrated, but by comparing with Be-7 it can be seen that Cs-134 is present in the air as significantly larger particles than Be-7, (Figure 9.15). A rough estimate of the cut-off sizes during our sampling, indicates that about 50% of the activity of cesium isotopes are resuspended as particles greater than 2 µm. This means that during normal weather conditions resuspension will only influence the air activity concentration significantly up to some kilometers away.

Using a simple exponential model the resuspension factor can be given by

$$Sf = Sf(0) \exp(\ln 2 \cdot t/T\ 1/2)$$

where $Sf(0) = 2 \cdot 10^{-8}$ and $T\ 1/2 = 0.9$ years. Comparing these results with the models described above (Figure 9.16) we can see that the resuspension factor is significantly lower, and also has a slower decline.

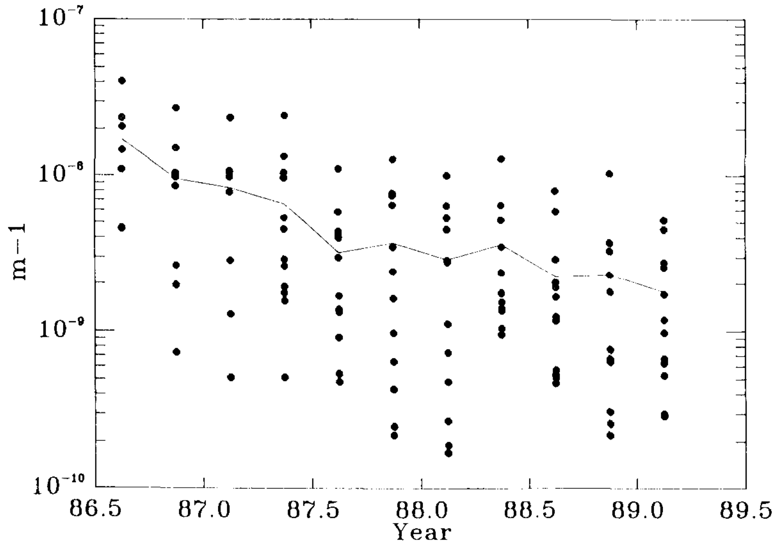


Figure 9.14 Resuspension factors.

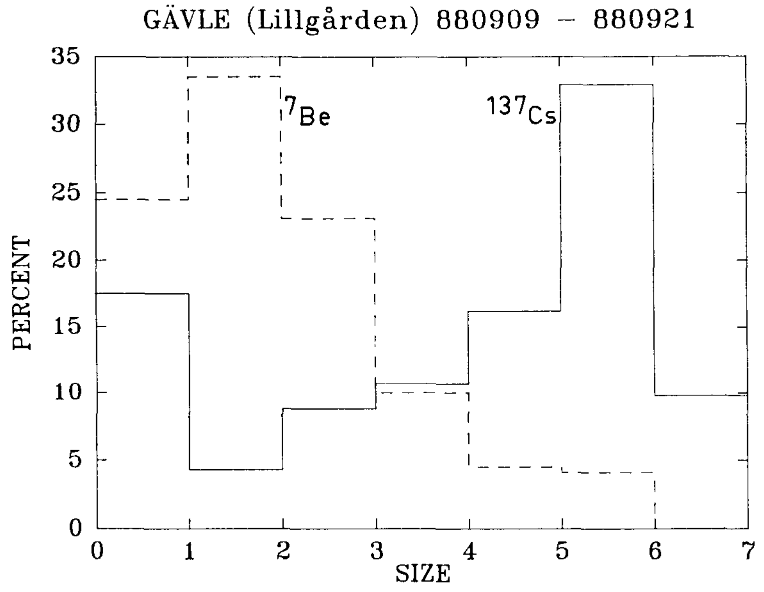


Figure 9.15 Size distribution of resuspended activity.

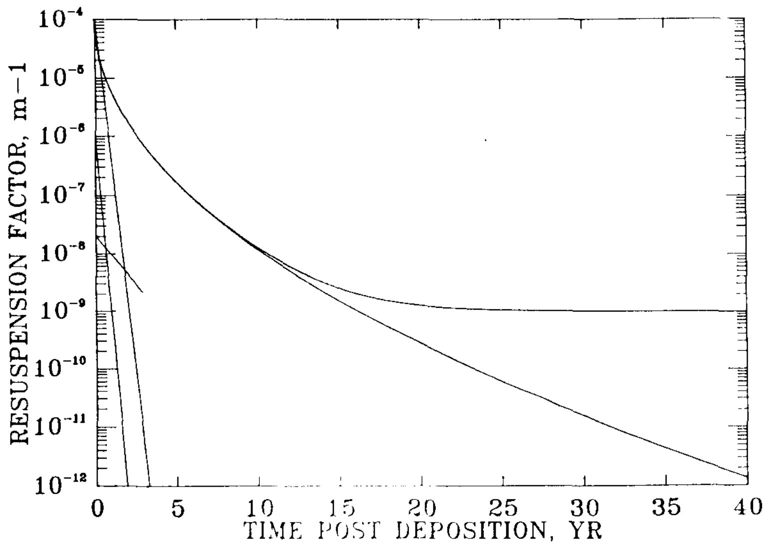


Figure 9.16 Resuspension factors predicted by different models.

10. NUTRITION PATHWAYS

10.1 Dynamic model for transfer via grass-cow-milk, project AKTU-220

From the time of the intensive atmospheric nuclear weapons testing around 1960 a large amount of information on the transfer of radioactive material in the environment has been accumulated. That information has been used for modelling purposes permitting radiological assessments to be made. The first models were equilibrium models describing for the steady-state situation the ratios of radionuclide concentrations in one compartment relative to the concentrations in another compartment. The end point of these models is generally collective dose commitment which provides a useful index of potential health risk from the radioactive contamination.

Now dynamic models have become more common, not least because these models are now easily implemented on personal computers. Dynamic models are generally based on the same information as the steady-state models. But instead of giving time-integrated values they produce predicted time-histories of the radioactive concentrations in the various compartments, also of the resulting dose rates. This dynamic feature naturally requires additional information for the modelling, and this need of data is one of the major drawbacks for these models since sufficient information often is not available.

After the Chernobyl accident in 1986 a large amount of data has been collected on the transfer of radioactive material in the environment. These data are very useful for modelling purposes because they, in contrast to the data from the weapons test fallout, are due to a contamination event of very short duration. The data thus permits the time response of the environmental systems to be recorded, which is of great importance for dynamic modelling.

Project AKTU-220 has been performed at Risø National Laboratory. The initial version of the dynamic model for cesium-137 developed in the project (ref. NI88) was implemented via the TAMDYN model simulation code (ref. KA88a). This code is a further development of the TAM3 code (Ref. GA88a). The TAMDYN code was later replaced with the TIME-ZERO system (ref. KI89), since the latter provides a more flexible and user friendly modelling environment. All of the results in the following are derived from models implemented with the TIME-ZERO system (ref. NI89 and ØH89).

10.1.1 Description of the model for iodine-131

The schematic model structure is shown in Figure 10.1. The model comprises an atmospheric compartment, three soil compartments, and four grass compartments. The basic structure of the soil and grass compartments is similar to that used in the Farmland model developed by the National Radiological Protection Board (ref. SI85). The soil compartments are divided into three soil layers 0-1 cm, 1-5 cm and the deeper soil. The grass compartments comprise two external and two internal plant compartments. One external compartment accunts for direct deposition and initial resuspension and the other for surface soil contamination. The levels of radioiodine in the soil and the grass compartments are calculated from first-order differential equations as a function of time based on the atmospheric inputs and rate constants indicated by the arrows in Figure 10.1.

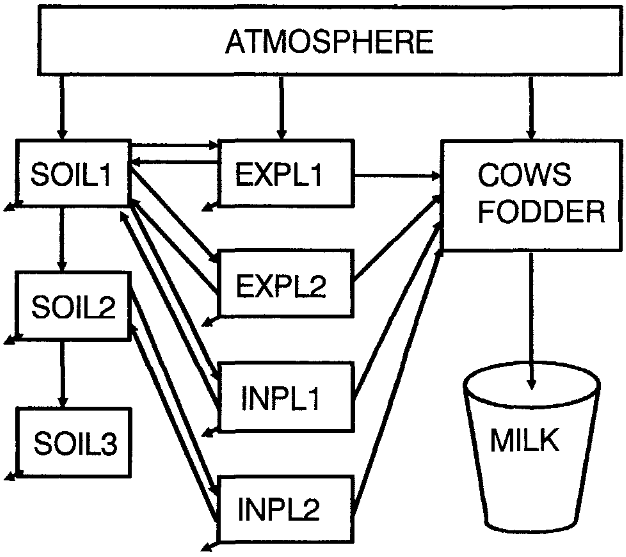


Figure 10.1 Schematic box structure of the model TAMDYN.

The daily intake of radioiodine by the cows are calculated from the fodder composition where the pasture component represents the major intake of radioiodine when the cows are not stabled. The intake also includes the inhalation pathway calculated from the breathing rate and the atmospheric concentrations. The milk concentrations are calculated from a simple multiplicative factor that accounts for the fraction of the daily intake of radioiodine by the cow that appears in each litre of milk.

The model dynamics uses a time step of one day, and as inputs to the model, atmospheric concentrations of I-131 are used together with data on precipitation and fodder composition, all given on a daily basis. The atmospheric iodine is assumed to comprise an elemental fraction, a particulate fraction and an organic fraction. The deposition processes are modelled by dry deposition velocities and volumetric washout ratios that are different for each fraction. The values used are given in Table 10.1.

The fraction of the total deposition that is retained initially on vegetation is modelled using a constant interception factor at a value of 0.3. The grass yield is assumed to be 0.14 kg dw m^{-2}. A weathering half-life of 10 days is used to account for the field-loss of initially retained iodine on the grass surface. When the cow is on a full grass diet the daily intake is 22 kg dry weight. The concentration of iodine in the milk is calculated from the daily intake using a factor of 0.001 Bq/l per Bq/day.

Fraction	Dry deposition ($m\ d^{-1}$)	Washout ratio ($Bq\ m^{-3}$ per $Bq\ m^{-3}$)
Elemental	1000	200
Particulate	100	300
Organic	10	10

Table 10.1 Parameter values used for modelling dry and wet deposition for different iodine fractions.

10.1.2 Description of the model for cesium-137

The schematic structure of the model is shown in Figure 10.2. The pathways between the different compartments are indicated. The transfers between compartments are described by rate constants in units of reciprocal time. The air, soil and plant compartments, all refer to a surface of 1 m^2. The size of the time step, used throughout the model, is one day.

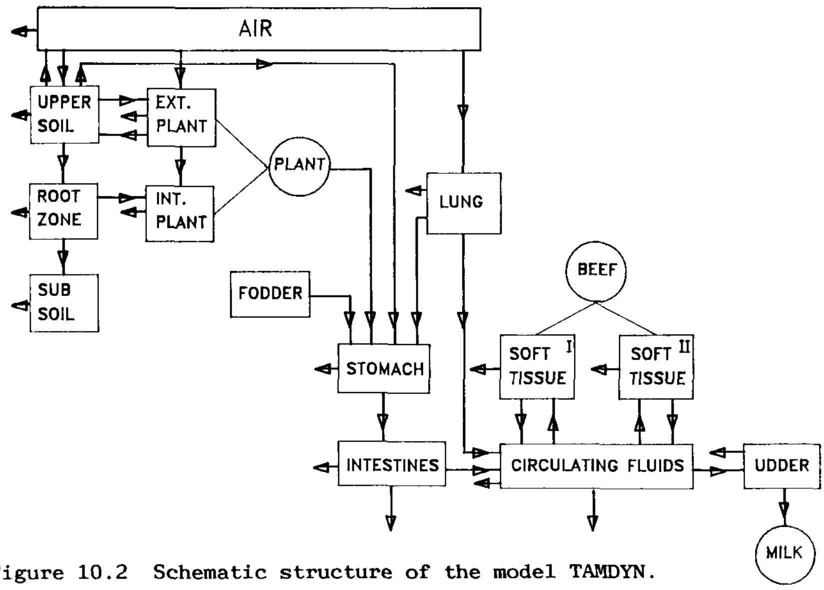


Figure 10.2 Schematic structure of the model TAMDYN.

The inputs to the model are the daily Cs-137 concentrations in the air after an accidental release. The transport processes from the air to the soil-plant system are the traditional ones. In order to calculate the wet deposition, average values for Danish precipitation are used; however, when data from the Chernobyl accident are used as input to the model, the actual amount of daily rain in 1986 is included in the model.

The deposition is partitioned between the soil and plant surfaces, using a common foliar interception factor. The plant used in this model is pasture grass, and a value of 0.3 has been used for the interception factor, based on Danish conditions. The plant growth rate is calculated from experiments made by the National Institute of Animal Science, Denmark (ref. KR87) and provides for the above-ground biomass and dilution of the concentration of Cs-137 in the plant during the growing season. The loss of Cs-137 from the surface of the plants to the soil is described by two weathering constants (ref. NI81). The value used for the first 30 days after the contamination is chosen to be 0.099 d^{-1} which gives an effective half-life of 7 days. The value used for the rest of the simulation time is chosen to be 0.023 d^{-1} which gives an effective half-life of 30 days.

Three soil compartments are used in order to account for the intensive cultivation of the Danish soil. Resuspension of contaminated particles and soil ingestion by the livestock remove activity from the surface of the soil. Thus, an upper soil compartment of 1 cm thickness is considered. Plant uptake via roots is described by an expression similar to that used in the Pathway model (ref. WH87). The effect of ploughing is simulated by converting the inventories of the external and internal parts of the plant into the root zone compartment.

Cow's diet is main determinant of the Cs-137 levels in milk and beef. The diet specified in the model is based on common Danish agricultural practice. The cow model used here is based on a model previously published (ref. CR84 and SI85) and adjusted to Danish conditions.

10.1.3 Results and discussion for iodine-131

The model was used to simulate the I-131 contamination of the Roskilde area in Denmark after the Chernobyl accident. As input values daily averages of measured levels of I-131 in the air were used. Since only the particulate fraction was measured an assumption was made of the iodine composition: 30% particulate, 30% elemental and 40% organic.

The predictions of the I-131 levels in grass are shown in Figure 10.3 together with the observed levels. The predicted maximum level and the predicted time-integrated value are in relatively good agreement with the observations giving predicted-to-observed (P/O) ratios of 1.2 and 1.3, respectively. However, the dynamic time variation of the predictions appears to be significantly out of phase in comparison with the observations, which show that the deposition is inadequately simulated by the model. It is noted that wet deposition (from day no. 127) does not produce higher levels in the grass than does dry deposition, which contrasts with the modelpredictions. Considerable uncertainty is associated with the assumptions of a constant iodine fractionation, where higher levels of elemental iodine in the initial phase could account for the higher grass levels in this period.

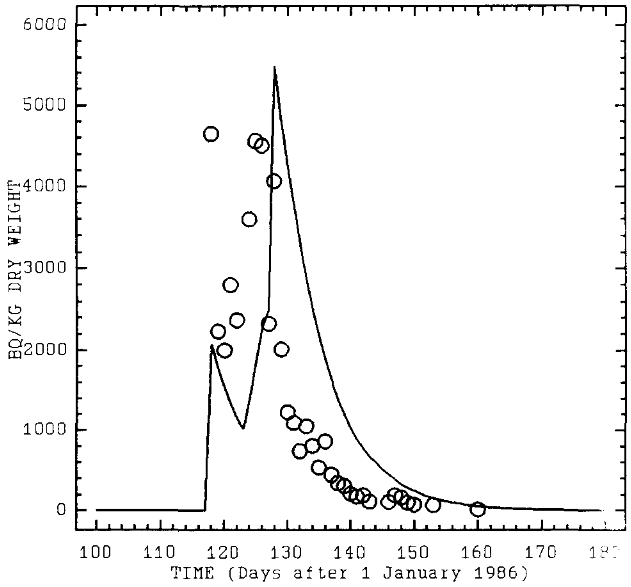


Figure 10.3 Predicted and measured levels of I-131 in grass at Roskilde after the Chernobyl accident (Bq kg^{-1} dry weight).

The predictions of the I-131 levels in milk were calculated from the fodder composition for the cows from where milk samples were obtained. The farm is located about 4 km from the site where the air and grass samples were collected. The cows were kept stabled until day no. 140 prior to which they were fed concentrates (mash and silage), which were stored outdoors and thus contaminated with I-131. The model predictions of the milk levels are shown in Figure 10.4 together with the observed values. The predicted milk levels from the stabling period are in good agreement with the observations, but when the cows are sent to pasture, the model significantly overestimates the milk levels (P/O=5.7). This is caused mainly by the overestimate of the grass levels during the same period (P/O=3.8).

10.1.4 Results and discussion for Cs-137

In order to simulate the conditions in Denmark after the Chernobyl accident, the input to the air compartment are the Cs-137 concentrations (Bq/m^3) measured at Risø (ref. AA88). The levels in grass, milk and beef predicted by the model are shown in Figures 10.5-10.7, and the measured levels are given in comparison. It can be seen that the model predictions for the levels in grass and beef are consistent with the measured levels; on the other hand the predicted levels for the milk are underestimated, primarily due to the lack of knowledge of the metabolism of cows.

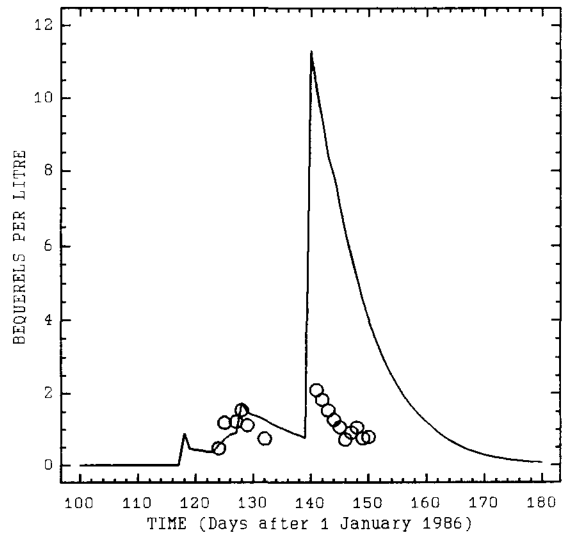


Figure 10.4 Predicted and measured levels of I-131 in milk collected near Roskilde after the Chernobyl accident (Bq/liter)

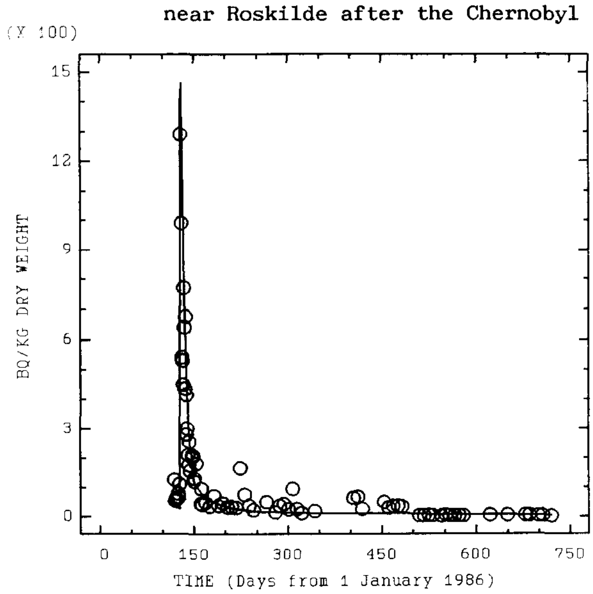


Figure 10.5 Cs-137 in grass at Risø. Predicted and measured concentrations.

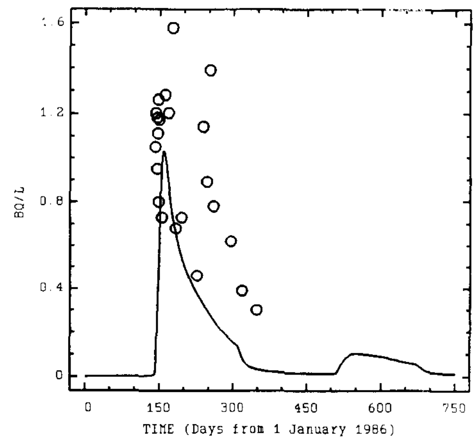


Figure 10.6 Cs-137 in milk. Predicted and measured concentrations.

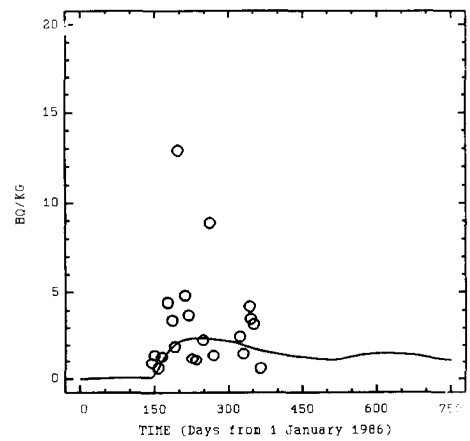


Figure 10.7 Cs-137 in beef. Predicted and measured concentrations.

The model has been used to demonstrate the influence of the season following an accidental release at different times of the year. In addition the effects of applying countermeasures have been investigated. The effect of reducing the contamination levels in animal produce by increasing the stabling period for the cows is demonstrated.

10.1.5 Conclusion

Two dynamic models have been developed to simulate the transfer of radioactive iodine and cesium through the grass-cow-milk pathway. The two models are of identical structure, each containing an atmospheric compartment, soil compartments and grass compartments. The cesium model additionally comprises a metabolic-cow submodel whereas the iodine model uses a simple transfer factor from forage to milk. The basic model structures are similar to those of food-chain models developed by the National Radiological Protection Board in the UK. The cesium model includes features to account for seasonal variation of the growth of vegetation in addition to a varying composition of the cows fodder throughout the year.

The models have been used to simulate the contamination in Denmark near Risø National Laboratory of I-131 and Cs-137 from the Chernobyl accident in 1986. As input values were used daily averages of measured levels of radionuclides in the air and data on percipitation. The model output values were compared with the corresponding observed levels.

The modelling of the transfer of radioiodine from the atmosphere to pasture is particularly uncertain due to the lack of knowledge of the chemical composition of the atmospheric iodine, where the elemental fraction plays an important role. The model predicts the transfer of radioiodine from pasture to milk within a factor of two compared with the observations.

Due to the predominant occurrence of particulate radiocesium in the air, the modelling of the transfer of this nuclide from the atmosphere to pasture performs better than for radioiodine. The comparisons between predicted and measured concentrations of Cs-137 in grass, milk and beef demonstrate agreement within a factor of two. Effects of season and countermeasures have been examined.

The United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) reports for their steady-state models a transfer factor for I-131 from total deposition to milk of 0.23 Bq-day/liter per Bq/m^2, which compares reasonably well with a value of 0.33 Bq-day/liter per Bq/m^2 predicted by the dynamic iodine model. The UNSCEAR transfer factor for Cs-137 from total deposition to milk is 2.1 Bq-day/liter per Bq/m^2 compared to a value of 1.5 Bq-day/liter per Bq/m^2 calculated from the dynamic cesium model using assumptions similar to those of the UNSCEAR model.

10.2 Norwegian experiment on cesium decline in freshwater fish, project AKTU-290

In early September 1986 about 50 trout were captured in the river Nordåa, located in one of the areas in Norway where the Chernobyl fall-out was highest, and where very high concentrations of radioactive cesium in freshwater fish had been observed in the summer of 1986. The purpose of the investigation described in this subchapter, which was part of project *AKTU-290*, was to follow the reduction of cesium concentration in the fish if it was placed in an environment with low concentrations of radiocesium.

The captured fish was placed in clean containers in a test-station near Lillehammer, Norway. Over the winter 1986/87 fish was collected for analysis at the following dates: 2 and 25 September, 27 October, 20 November, 23 December, 22 January, 26 February and 24 March.

During the experimental period the fish was not fed. The reason for this is that throughout the experiment the water temperature was so low that the fish would not have taken nutrition even under normal circumstances. The fish in the experiment varied much in size, from more than 300 grams to less than 20 grams freshweight. Concentrations of both cesium 134 and cesium 137 were determined, and only the fish filets (muscle tissue with skin) were measured.

The cesium-137 concentrations in the measured fish tissue are given in Table 10.2. Each value is for one specific fish. The difference between individual fish is large, and evidently not related to body weight. The average values were calculated by letting each fish have equal "weight", regardless of the actual body weight of the fish. These averages do not indicate that there has been any decrease in cesium-137 concentrations throughout the winter. This is not the conclusion that was expected, although it does not seem unreasonable, when it is considered that the water temperature was so low throughout the experiment that there was little or no biological activity in the fish. When there is no excretion, there is no reduction of cesium con-

Capture date							
2 Sept		25 Sept		27 Oct		20 Nov	
Weight	Bq/kg[1]	Weight	Bq/kg	Weight	Bq/kg	Weight	Bq/kg
300	1330	152	1410	299	565	311	1580
294	970	149	800	240	640	285	1000
164	960	136	1050	161	970	213	1130
49	1390	114	870	130	1550	159	1310
42	1500	85	1140	108	1450	103	1980
34	1500					96	1650
39	1360					64	1520
31	1280						
30	1590						
Average 1320		Average 1054		Average 1035		Average 1453	

Capture date							
23 Dec		22 Jan		26 Febr		24 Mar	
Weight	Bq/kg	Weight	Bq/kg	Weight	Bq/kg	Weight	Bq/kg
267	860	342	1150	212	880	177	1050
239	2020	137	1000	159	705	145	2020
185	1340	88	1390	155	980	75	1380
153	1380	84	1610	120	1620	69	2050
117	1480	55	1140	112	1430	39	2190
107	2370	53	1390	61	1060	19	1820
102	1640	36	690				
Average 1584		Average 1196		Average 1113		Average 1752	

[1] Per kg freshweight.

Table 10.2 Concentration of cesium-137 in individual fish.

Date	18 Sept	12 Oct	14 Oct	28 Oct	4 Nov
Temperature	10.5°C	7°C	5.5°C	4°C	3°C
Date	9 Nov	18 Nov	30 Nov	6 Dec	24 Dec
Temperature	2.5°C	2°C	3°C	0.5°C	0°C
Date	31 Dec	10 Jan	20 Jan	2 Feb	28 Feb
Temperature	0°C	0°C	0.5°C	1°C	1°C
Date	14 Mar	28 Mar			
Temperature	0.5°C	0°C			

Table 10.3 Temperature in containers throughout the experiment.

centration either. It would have been interesting to continue the experiment during summer conditions, but funds were unfortunately not allocated to continuation of this experiment.

The cesium-134 concentrations were also measured in this experiment. The cesium-134 concentrations behave in parallel to the cesium-137 concentrations, like would be expected, and give no additional information. There was therefore no reason to include the measured values here.

10.3 Work on various nutrition pathways coordinated by SSI, Sweden

The Swedish National Institute of Radiation Protection (SSI) coordinates extensive research in the areas covered by the AKTU program, and the Appendix of this report contains a list of these SSI projects. The short summaries below of selected projects have also been prepared by SSI. These projects are not part of the AKTU program.

Radionuclides in agricultural land in the northern county of Upland and Västerbotten.

K.J. Johansson, Å. Eriksson. The Department of Radioecology, Swedish University of Agricultural Sciences, Uppsala.

The Department of Radioecology has presented various results presented in the reports below:

- Depth distribution of radiocesium in agricultural soils in Chernobyl fallout areas in Sweden in 1987-1988.
- Transfer of cesium to grassland crops in the Chernobyl fallout areas in Sweden in 1986-1987.
- Use of Chernobyl field data in modelling cesium transfer to grassland crops.
- The transfer of radiocesium from pasture to milk.

Some of the results may be summarized from the reports:

1) The variation of the transfer coefficients determined for grass, hay and ensilage are as a whole important. The quotient between the highest and the lowest value is of the order of 100 within all the areas studied.

2) At the Swedish University of Agricultural Sciences a well defined experiment for estimation of the transfer coefficient was performed in 1986. Milk cows were fed on green-cut forage with a known concentration of Cs-137. Using forage cut with 5 cm stubble height the transfer coefficient was found to be 0.0019 d kg^{-1} (day per kg), with 15 cm stubble height 0.0067 d kg^{-1}. The mean transfer coefficient estimated in the 1987 study is 0.052 d kg^{-1}.

3) The transfer of cesium to cereal crops was considerably smaller than the transfer to the grass.

4) Root uptake has contributed, but to a very small extent, to the cesium content of the vegetation.

5) The results of the investigation of radiocesium in the Chernobyl fallout, when deposited on permanent grasslands rich in organic matter in the surface layer, after 1.5 year show that up to 80% or more was retained in the upper 20 mm of the soil profile. In grasslands, which will not be ploughed, the activity is expected to migrate very slowly into the soil profile, and it will take several years before the mean depth has increased from 10 to 20 mm.

6) From 1986 to 1987 the cesium transfer to the grass crops on the mineral soils is reduced by a factor 4-6. For the organic soil this factor is 4 or lower. This reduction may indicate that one phase with a high cesium transfer from the contaminated plant community to new grass is ended and that another process with transfer by uptake from the shallow root system starts to dominate.

7) Allowing for the time of growth, for the productivity of the land and for the residence time of the contaminant, it should be possible to give fair estimates of the contamination level of grass crops in situations similar to that after the Chernobyl fallout.

The turnover of cesium in forest ecosystems.

R. Bergman et al. The National Defence Research Establishment, Umeå.

Uptake, turnover and transport of radioactive caesium (Cs-137 and Cs-134) in a boreal forest ecosystem.

This work has been focused on the presence of radioactive cesium in moose and its food plants in northern Sweden during the first two years after the Chernobyl accident.

The objective of the project has been to illustrate the importance of foodstuffs from the boreal forest ecosystems for the exposure of certain groups or the total population and to improve the knowledge about the behaviour of cesium in this ecosystem with different time perspectives.

Results are presented concerning:

- the initial behaviour i.e. retention and run-off of cesium after wet deposition over snow-covered areas
- the distribution of radioactive cesium in vegetation, small rodents, fox and moose, and
- the time dependent turnover of cesium

Comparison of the cesium content before and after the Chernobyl accident

The cesium content in moose can be regarded as an integral measure of the contributions of different components in the food during long time. During 1986 and 1987 the Cs-137 content in meat of calves was as an average 250-300 Bq/kg, while corresponding values for older animals were 150-200.

During the autumn 1986 about ten samples of meat from the hunting period 1985 were sampled. For these samples a cesium content of 30±20 Bq/kg was observed.

Even for two types of plants it has been possible to compare the Cs-137 levels before and after Chernobyl.

A sample of bilberry wires from 1985 had a concentration level of Cs-137 of about 50 Bq/kg d.w., which shall be compared with 1,500 Bq/kg after Chernobyl.

For pine-twigs sampled 1984 the content varied between 10 and 175 Bq/kg. In the autumn 1986 the Cs-137 content in pine shoots was about 500 Bq/kg.

The cesium contamination in a longterm perspective

According to data given by UNSCEAR and the Swedish Defence Research Establishment the Cs-137 fallout up to and including 1984 from atmospheric weapon tests in the county Västerbotten is of the order of 1.8-2.6 kBq/m². After correction for decay about 1.2 - 1.5 kBq is left in 1985. A typical value for the cumulative deposition after Chernobyl is about 15 kBq/m².

Based upon these results it is concluded that the reduction of the cesium content in important food chains, including vegetation and game, can be expected to be remarkably slow.

Dose contributions

The collective dose from consumption of products from food chains in the forest ecosystem (moose and wild berries) is compared to that due to beef, milk and milk products. The two pathways are estimated to be of equal importance for the Swedish population over a period of twenty years.

The cesium content in moose

A report called "The moose and its fodder-plants" (In Swedish) has been prepared by Karl J. Johansson (The Department of Radioecology, The Swedish University of Agricultural Sciences) and Roger Bergström (The Research Centre of the Swedish Hunting Society).

The cesium content has been determined for meat as well as for plants being the main part of the moose´s food. The concentration in meat has been about the same 1986-87-88, somewhat lower in 1987, somewhat higher in 1988.

The study has indicated a relationship as expected between the activity deposited and the content in moose. But also other circumstances causing variations have been recognized, which are of about the same importance.

The nutrient supply from the soil where the moose is living seems to be of even greater importance than the deposit. Thus, it has been observed that moose which had been living within a certain area, all had an activity level <400 Bq/kg, while other hunting-parties within other areas with the same deposition have shot moose with a cesium activity >1 500 Bq/kg.

The reason most often is the presence of agricultural land within the area where the moose had been grazing, but even in forest such differences have been found. Barren areas including bogs or moorlands often promote high concentrations of cesium, both in moose and in roe-deer.

Cesium content in roe-deer

During the summer 1988 concentrations in roe-deer meat of about 100 Bq/kg were found. In the middle of August the content had increased to 5000-12000 Bq/kg. The reason seems to be cesium intake via consumption of mushrooms. Roe-deer can consume large quantities of mushrooms, up to 20% of the intake of food, and mushrooms often contain very high concentrations. Normally the content was about 40 000 Bq/kg d.w. within the area studied.

It is estimated that an intake per day of about 10 000 Bq, which corresponds to consumption of 1250 g dry mushrooms, might result in a cesium content in the muscle of 10,000 Bq/kg.

This problem will remain for several years and be especially pronounced in years with a climate favourable for mushrooms.

Radioactive cesium in the reindeer fodder

Radioactive cesium in grazing areas for reindeer (In Swedish). Olof Eriksson, The Swedish University of Agricultural Sciences.

The objective of this inventory has been to elucidate the activity content in certain plants, sampled mainly from different parts of the reindeer winter grazing land.

If the sequence, atmospheric deposition, reindeer pasture, man is considered, it can be concluded that it is the lichens which have the longest residence time for Cs-137. Lichens have a great capacity in absorbing, store and eventually even react under the influence of substances of several types, e.g. sulphuric acid, heavy metals and radionuclides. Substances, which attached to the air humidity penetrate the plant body, will remain for a long time as the plant lacks a root system, which to some extent could promote removal. The slow growth and declination - lichens can be several tens of years old - also contribute to the slow decay.

The plant sampling program has included:

- ground lichens
- tree lichens
- plants, including water and berry plants
- leaves

<u>Mechanisms for transfer of radioactive nuclides to trouts and chars via nutritional organisms in "natural" and regulated mountaint lakes</u>

G. Neumann and M. Notter (The National Environment Protection Board, Stockholm) and J. Hammer (The National Board of Fisheries, Gothenburg) (Report in Swedish).

The cesium analyses by the Swedish National Food Administration very early indicated that even the mountain areas in the upper regions of the rivers had received large quantities of radioactive fallout.

The aim of the investigation has been to quantify the amount of Cs-137 in the limnic environment as well as in different trophy levels in a series of 8 mountain lake systems in NW Jämtland and to define the reasons most important for the transport of the radioactivity via the entire nutrient chain to fish.

Three of the lakes are incorporated in the regulation system; in two of these lakes and in a "natural" one the new nutrient organism <u>Mysis relicta</u> has been introduced.

The investigation has given preliminary results summarized below:

- The Cs-137 content in sediment has been doubled between October 1986 and October 1987.
- The activity contamination in the regulated lakes has been considerably greater and faster than in natural lakes.
- The radioactive content increased rapidly during 1986, both in trout and in char. It has been reduced in char during 1987, while the content in trout went on to increase to August 1987 in some of the lakes. After that the tendency is towards lower values, but as late as October 1987 the average level in trouts in some lakes was >20 000 Bq/kg d.w.
- The bottom feeding fish populations as a rule have higher activity values than the plankton eaters.
- The content in Mysis relicta, which was much greater in zoo plankton, was reduced during the winter 1986/87. During the later part of 1987 the values again began to rise.
- The hypothesis that Mysis relicta via its special ecology should promote and maintain high values in fish is still valid. The con-

centration values are, however, even affected by differences in the ground deposition, the degree of regulation, the nature of the sediment etc.

Marine ecosystems Gauss expedition III

Per-Olof Agnedal, Studsvik

Cs-137 and other radionuclides in the benthic fauna in the Baltic Sea before and after the Chernobyl accident.

The purpose of the cruises was to collect water, sediment and benthic fauna and to determine the radioactivity in the samples and calculate the inventory of radionuclides in the Baltic Sea, especially of Cs-137.

The Cs-137 concentration in water showed great variation in 1986 with low values in the northern part of the Bothnian Bay and in the Baltic Proper but high values in the southern part of the Bothnian Sea.

An increase of the Cs-137 concentration in sediment and benthic fauna has occurred in the Baltic Sea especially in the parts of the Bothnian Sea and Bay outside the most contaminated land areas.

The presence of nuclides such as Mn-54, Ag-110m, Ru-103, Ru-106, Cs-134 and Cs-137 from the Chernobyl fallout has been observed in the benthic fauna in the Danish Strait, in the northern part of the Baltic Proper and in the Bothnian Sea.

The results obtained during the third cruise in October-November 1986, after the Chernobyl accident, are compared with those from the first two cruises in 1983 and 1984. The highest concentration of Cs-137 in sediment in October 1986 was 450 Bq/kg d.w. compared with 51 Bq/kg d.w. in 1983 at the same locality.

The mean value for Cs-137 in Saduria entomon has increased about 18 times and the highest value in 1986 was 250 Bq/kg d.w. Saduria entomon seems to be a good bioindicator for long-lived nuclides as it reacts fast and is rather long-lived.

Comparison of predicted and measured Cs-137 concentrations in a lake ecosystem.

Ulla Bergström and Sture Nordlinder, Studsvik

The study is organized with the aim to receive long-term series which can be used for model validation of a total drainage area. To a certain extent independent data sets regarding the transfer of radionuclides in the biosphere are now available due to the Chernobyl fallout.

In the report from this project comparison is made between calculated and measured data of concentrations of Cs-137 in water, sediment and fish from the Swedish lake, Hillesjøn, situated in the contaminated area in Sweden.

This investigation is part of a scenario in the international BIOMOVS Study (Biological Model Validation Study). Two different approaches have been applied for modelling the uptake by fish. The differential equation are solved using the BIOPATH code and the uncertainties in the results, due to uncertainties in the model parameters, have been calculated by use of the PRISM code (the PRISM code has also been used for calculation of uncertainties in project AKTU-235).

Reasons for similarities and discrepancies between calculated and measured values are identified and discussed. The validity of applying data for steady state conditions to dynamic systems is addressed.

11. PHYSICO-CHEMICAL FORM OF RADIONUCLIDES, PROJECT AKTU-265

The presence of different physico-chemical forms of radionuclides in fallout is of major importance for the transport, mobility and biological uptake.

Project *AKTU-265* has been focused on:

- *The presence of hot particles and colloids (soil, vegetation,* rainwaters).
- The association of radionuclides with different components in soil (binding mechanisms).
- Electrodialysis for distinguishing low molecular weight charged species of Cs in aquatic solutions.
- The transfer from vegetation to animals.

Figure 11.1 illustrates that biological uptake of radionuclides will depend on their physico-chemical forms. Low molecular weight species are believed to be of major importance for active biological uptake while hot particles and colloids represent more inert species. Their presence should imply a reduced uptake for instance in the vegetation. However, weathering effects may transfer inert forms into available species and the uptake may increase with time. Furthermore, mobile species may become inert upon the interaction with components in the soil.

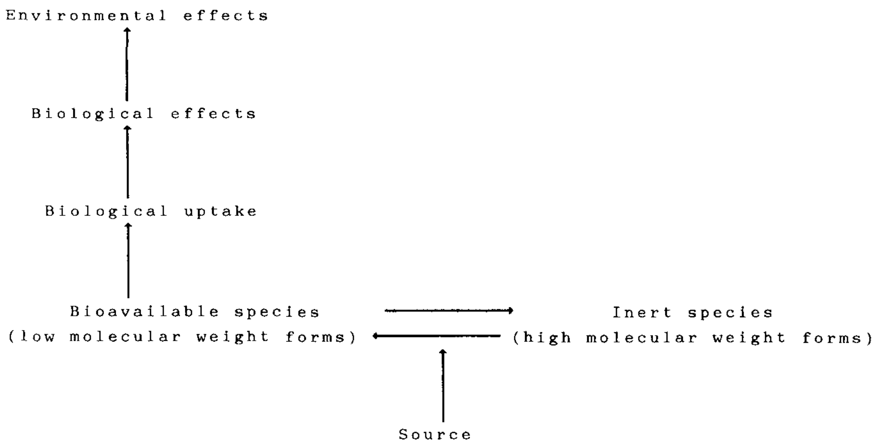


Figure 11.1 The relationship between molecular weight forms of radionuclides and biological uptake. Transformation processes are indicated.

Information on low and high molecular weight species of radiocesium as well as their association with components in soil and vegetation should therefore be essential for improving the predicting power of models estimating transport, mobility and biological uptake.

11.1 Hot particles

Hot particles (nuclides of Sr, Zr, Nb, Ru, Ce and U) were found in Central Norway, which was hit by the first part of the plume from Chernobyl in the time period April 28 - May 1 (ref. SA88b, SA88c and RI90). In October 1987 GM-tubes were used for field measurements of "hot" particles in Central Norway. Several particles were identified on the ground and collected for electron microscopic and gamma spectrometric analysis. Secondary electron imaging (SEI) was used for structure analysis, while backscattered electron imaging (BEI) was applied to identify areas with differences in elemental composition. Hot particles were also found in Southern Norway, deposited from the second part of the plume in May 5 - 10.

Some of the electron micrographs obtained are included in the present report. Figure 11.2 shows a sample containing Ru-106 (no gamma), with daughter Rh-106 as the only gamma-emitting nuclide at the time of measurement (ref. SA88c). Based on X-ray micro analysis (XRMA), and Scanning electron microscopy, Ru, Mo and Te were found to be associated with a spherical structure which was attached to a larger, irregularly shaped particle (Figure 11.2a) containing Si and small amounts of K and Cl. Due to the high average atomic number of the elements, the spherical structure appears bright compared to the large particle, in BEI mode (Figure 11.2b). Higher magnification revealed that this structure was composed of an aggregate of smaller grains (Figure 11.2c).

The oblong particle shown in Figure 11.3 was identified in herbage collected by a fistulated goat (samples of feed intake are collected via a tube operated into the throat) grassing in a mountain birch forest in June 1987. The particle contained Ce-144 as the only gamma emitting nuclide at the time of measurement. Based on XRMA the Ce seemed to be evenly distributed over the particle, whereas the dark areas seen in BEI mode (arrows in Figure 11.3b) indicated that the particle also contained low atomic number elements.

Figure 11.4a shows part of a large particle containing a mixture of fission products. These products were probably localized to the small area (framed) where XRMA showed the presence of U, Zr and Nb, among other elements. At higher magnification in SEI mode it was not possible to relate these element to a distinct structure (Figure 11.4b), while their presence was indicated by the bright area seen in Figure 11.4c, in BEI mode.

Radionuclides associated with particles and colloids in rainwater were also observed in rainwater collected in Oslo 6 May 1986 (ref. SA88a). These colloids are referred to as "hot" colloids. The samples were fractionated by ultrafiltration, using a hollow fibre cartridge with a nominal molecular weight cut-off of 10,000 Dalton. It was found that 50 - 90% of the gamma emitting radionuclides were retained by the cartridge. For Cs-137 the retention was 75%, while for a standard solution of Cs-137 as cloride it was less than 5%. This indicates that a major fraction of the Chernobyl cesium was associated with high molecular weight forms. Desorption experiments did not alter these results significantly, indicating that the colloidal fraction was relatively inert. According to these results only a small part of cesium as fallout in rain water is available for uptake in the biosphere, as bioavailability and molecular weight forms are linked, with the higher weight forms less available.

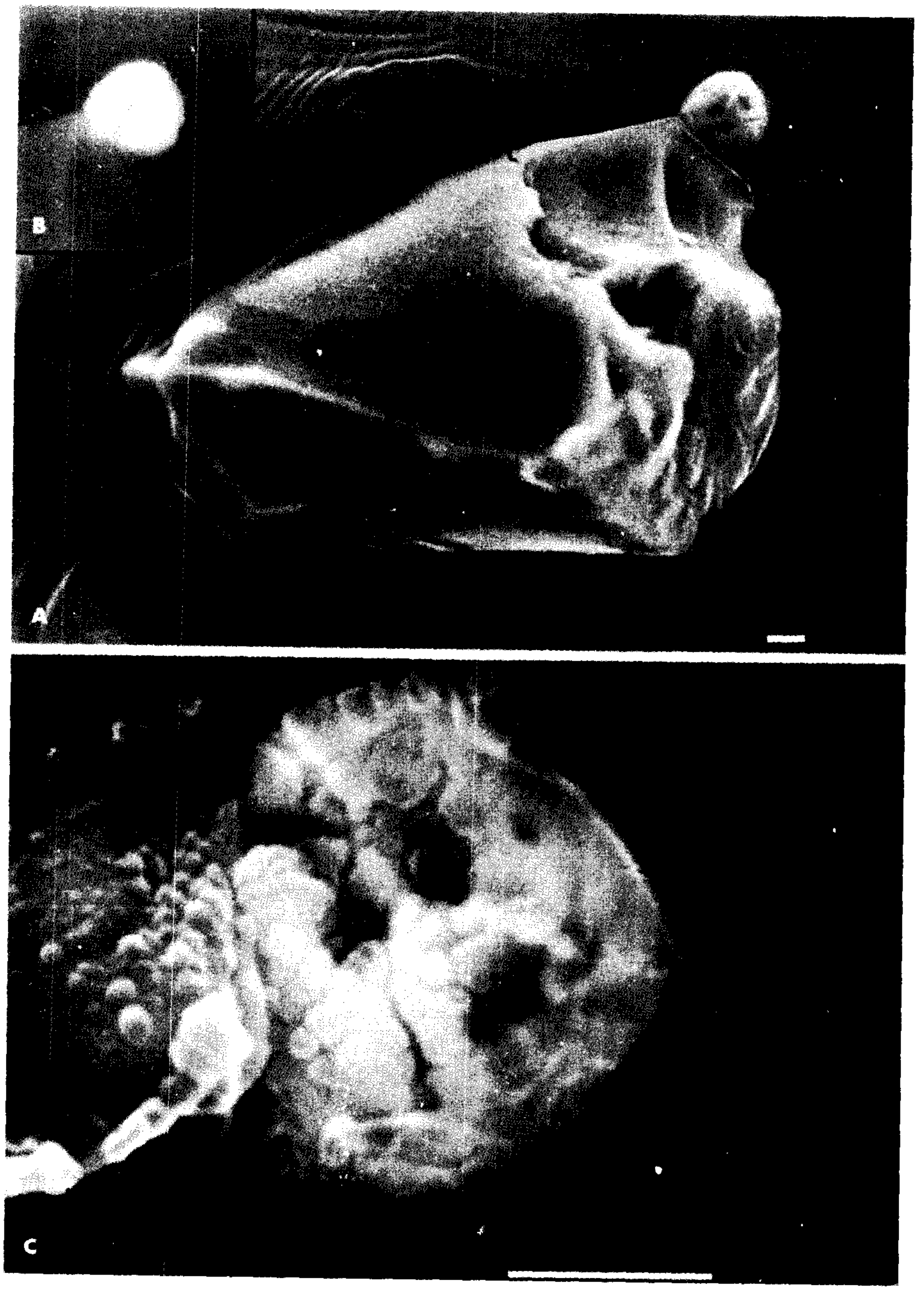


Figure 11.2 Scanning electron micrographs of a Ru-106-containing spherical structure attached to a larger particle. Bars are 1 μm.

Figure 11.3 Scanning electron micrograph of a Ce-144-containing particle. Bars are 1 µm.

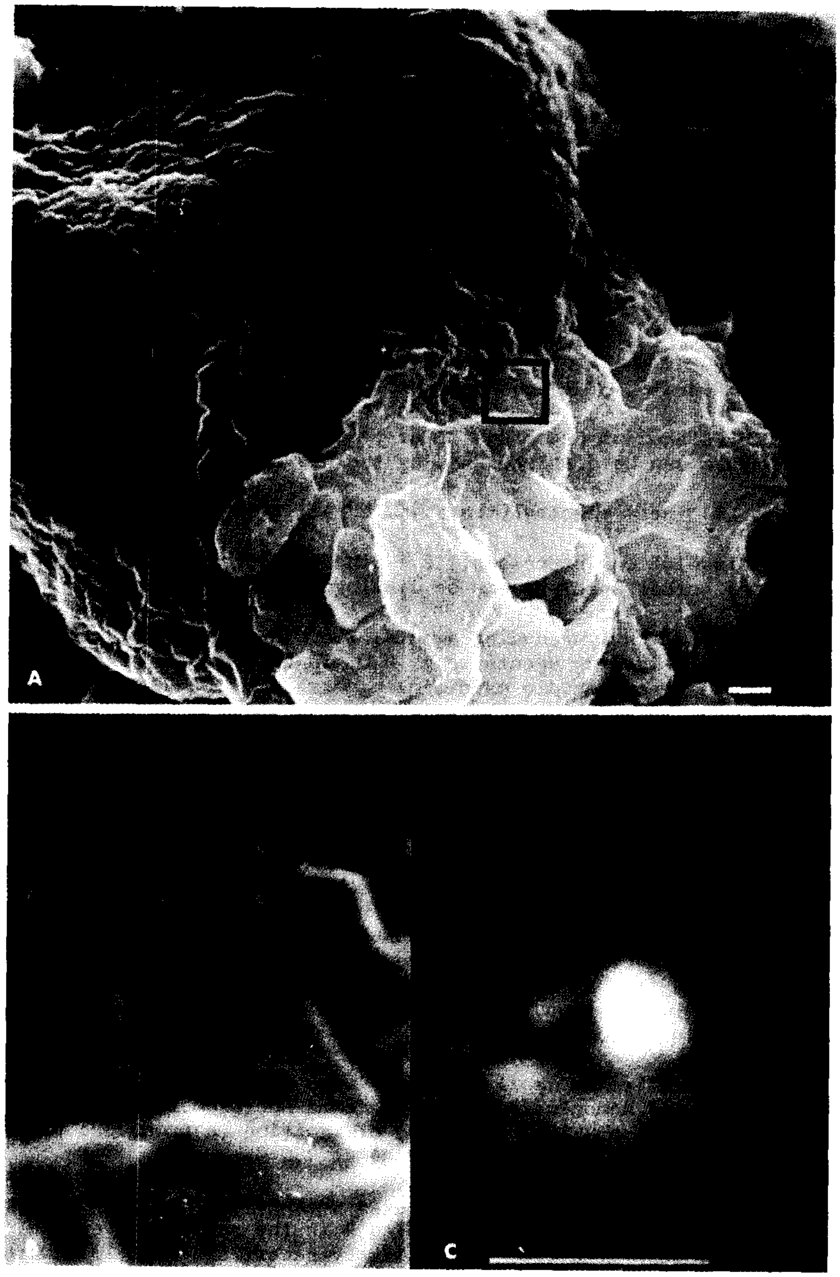


Figure 11.4 Scanning electron micrograph of a "hot" particle containing a mixture of fission products. Bars are 1 μm.

11.2 Sequential extraction

After the Chernobyl fall-out, it was found in Norway that transfer coefficients from vegetation to soil showed large variations. The reason for these variations are probably connected to differences in the physico-chemical forms of the radionuclides concerned. It is possible to obtain information on the binding mechanisms of radionuclides, i.e. primary ad/ab-sorption processes, from desorption experiments. In project *AKTU-265* a certain procedure for sequential extraction has been used in order to identify how much of the radiocesium is easily available for uptake, how much is associated with the organic fraction of the soil, and how much is unavailable for biological uptake (ref. RI90). The project has been performed by G. Riise, H.E. Bjørnstad, H.N. Lien and B. Salbu of the Isotope and Electron Microscopy Laboratory of the Agricultural University of Norway, and D. Oughton of the University of Manchester, UK.

Displacement processes, i.e. reversible processes without binding rupture can be distinguished by the use of

1) inert electrolytes, e.g. NH_4Ac

Dissolution processes, i.e. the change of the physico-chemical forms of sorbed species, can be distinguished by the use of:

2) slightly reducing agents, i.e. NH_2OH in HAc
3) oxidizing agents, e.g. H_2O_2 in dilute HNO_3
4) strong acidic oxidizing agents, e.g. 7M HNO_3
5) chelating or complexing agents, and
6) micro-organisms

11.2.1 Sequential extraction experiments

In project *AKTU-265* the association of radioactive Cs-nuclides and Sr-90 with components in soil has been investigated. Soil cores, including litter, with a diameter of 10 cm were collected at different sites in Central Norway. The soil samples (including litter) were subjected to a sequential extraction procedure according to the scheme 1-4 given above. This procedure is a modified version of that described in (ref. TE79).

The collected soil samples were sectioned in 2 cm layers, dried at 60°C and homogenized prior to gamma-spectrometry. Dry weight was determined at 105°C and loss of ignition (a procedure to determine the organic content of the sample) at 450°C.

Dried, homogenized samples were weighed (ca 2 g), and sequentially extracted with the reagents 1 to 4 in the scheme above. The extractants were separated from the solids by centrifugation (10 000 rpm), and the residue was then washed with Milli-Q-water and centrifuged. Soil samples, liquids and solid fractions were subjected to gamma spectrometry (Cs-137 and Cs-134) or liquid scintillation counting (Sr-90), while stable cesium content was determined by instrumental neutron activation analysis.

The following extraction procedure was utilized:

F1 - Exchangeable fraction
20 ml of 1 M NH_4Ac (pH 7) was added to the soil (ca 2 g). The suspension was shaken for two hours prior to separation.

F2 - Slightly reducible fraction
20 ml NH_2OH in 25% (v/v) HAc was added to the residue. The suspension was shaken for six hours at 80°C prior to separation.

F3 - Oxidized fraction
20 ml of 30% H_2O_2 and HNO_3 was added to residue. The suspension was shaken for six hours at 80°C prior to separation.

F4 - Acidic oxidized fraction
20 ml of 7M HNO_3 was added to residue. The suspension was shaken for six hours at 80°C prior to separation.

F5 - Residue
Contains the remaining activity (F5 = total activity - (F1 + F2 + F3 + F4)).

11.2.2 Results from the extraction experiments

Figures 11.5 and 11.6 illustrate the enrichment of Cs-137 and Cs-134 in the upper 2 cm layer of the soil (including litter), indicating that migration into deeper layers has been rather slow.

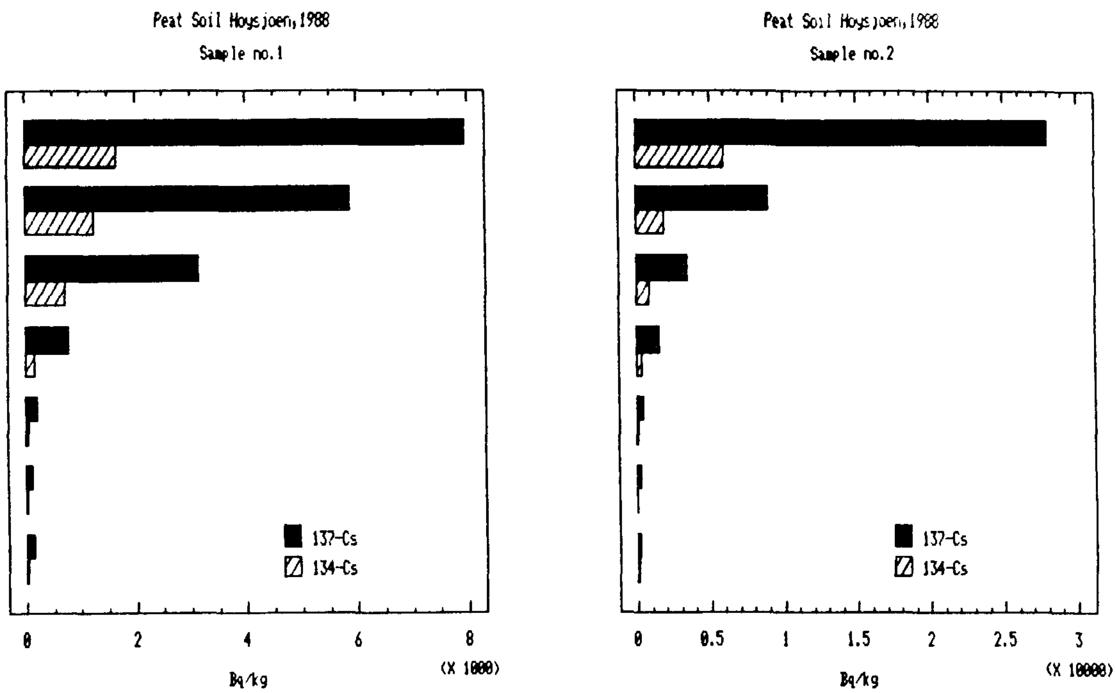
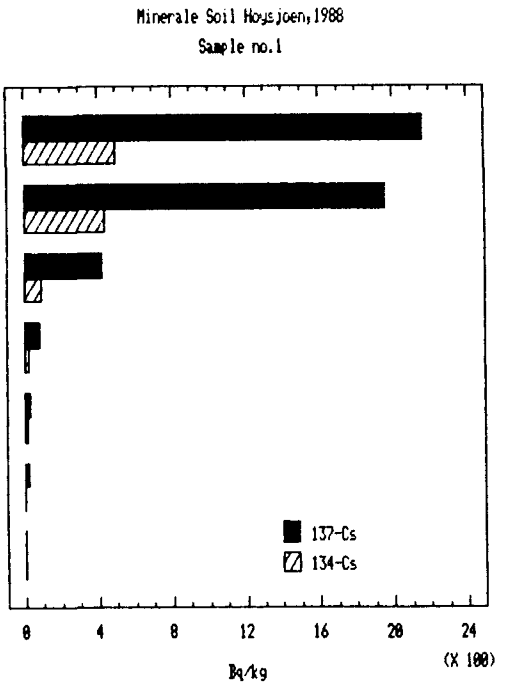


Figure 11.5 Vertical distribution of radioactive cesium nuclides (Bq/kg) in peat soil, Central Norway, September 1988.



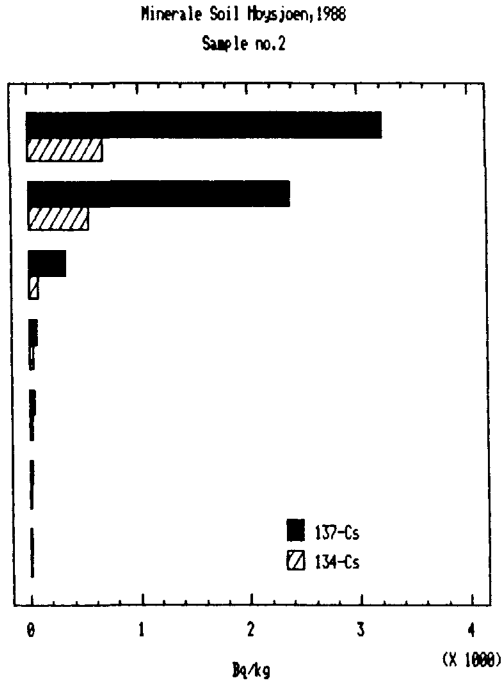


Figure 11.6 Vertical distribution of radioactive cesium nuclides (Bq/kg) in mineral soil, Central Norway, September 1988.

In Table 11.1 are summarized some results from the sequential extraction experiments (ref. RI90). There are large variations in the content of radiocesium between the different fractions. Fraction F1 represents Cs-137 and Cs-134 in simple ionic form, easily exchangeable with competing ions (ref. TE79). This fraction is important in connection with root uptake to plants. In fraction F2 is used a reagent that is effective dissolving the amorphous phases of Mn and Fe oxides/hydroxides and associated microcomponents (ref. FO87 and WI85). Fraction F3 represents primarily oxidized organic material; while in fraction F4 mineral phases are stripped to a certain extent in 7M HNO_3 (ref. FO87 and WI85). It should be stressed that the fractions obtained are operationally defined. However, the reproducibility is remarkably good.

Sample (site)	Year	Whole soil (Bq/kg Cs-134+Cs-137)	% in 0-2 cm	Cs-134/Cs-137 (ratio)
Heimdalen	1987	37700	97.9	0.68
Heimdalen	1989	38400	ND[1]	0.66
Vestre Slidre	1987	31800	ND	0.49
Vestre Slidre	1988	265000	ND	0.53
Kjølastøl	1987	16300	95.7	0.64
Beitostølen	1987	12800	86.7	0.47
Vinstern	1987	34200	93.9	0.45

Sample (site)	Year	Fractions (Bq/kg Cs-134+Cs-137) F1	F2	F3	F4	F5
Heimdalen	1987	1200	600	4800	21000	10100
Heimdalen	1989	4700	2600	4800	20200	15000
Vestre Slidre	1987	1500	800	2800	15600	7600
Vestre Slidre	1988	4800	3800	7800	129000	119000
Kjølastøl	1987	1800	800	700	6100	6900
Beitostølen	1987	600	400	700	8600	2500
Vinstern	1987	1100	800	4300	19200	8800

[1] ND = Not Determined

Table 11.1 Radioactive cesium in different fractions of sequential extraction of soil samples.

It is found that, for all samples investigated, radiocesium is strongly associated with components in the soil (fractions F3, F4 and F5), and particularly mineral matter, and less that 10% is readily mobile (fractions F1 and F2). Accordingly, one should expect a rather low transfer factor from soil to vegetation.

In Figure 11.7 the results of the neutron activation measurements are shown. The figure gives the ratio between Cs-137 and Cs-133 (stable cesium), and this ratio is remarkably constant for all fractions. This indicates that, to a large extent, isotopic exchange with naturally occurring stable cesium appears to have taken place during the three year period since deposition took place. In fraction F4 the Cs-137 content seems to be somewhat higher relative to stable cesium than for the other fractions. From Table 11.1 it is seen that, for most of the soil samples, fraction F4 (strong acidic oxidizing agent) is effective in desorbing the radiocesium. As the diffusion through double layers into mineral lattices is a slow process, the Cs-fixation may not yet have reached equilibrium, as indicated by this fraction.

For some of the samples, association of radiocesium with oxidisable material present in the soil, indicated by fraction F3, is significant.

Figure 11.8 summarizes the results concerning Sr-90. The easily mobile fractions (F1 and F2) are significantly higher than the strongly bound fractions (F3, F4 and F5). Compared to radiocesium, a substantially higher fraction of Sr-90 is mobile and available for plant uptake.

The present experiments indicate that the main part of the deposited radiocesium can be represented as a relatively inert pool or reservoir. However, transformation processes taking place in the environment (e.g. weathering) may influence the mobility both of the radiocesium nuclides and of other radionuclides (i.e. Sr-90). It is therefore important to have available well established techniques for measuring mobility. These experiments indicate that the sequential extraction technique is useful for obtaining such information. However, proper standardization of the technique is needed in order to obtain more chemically defined fractions.

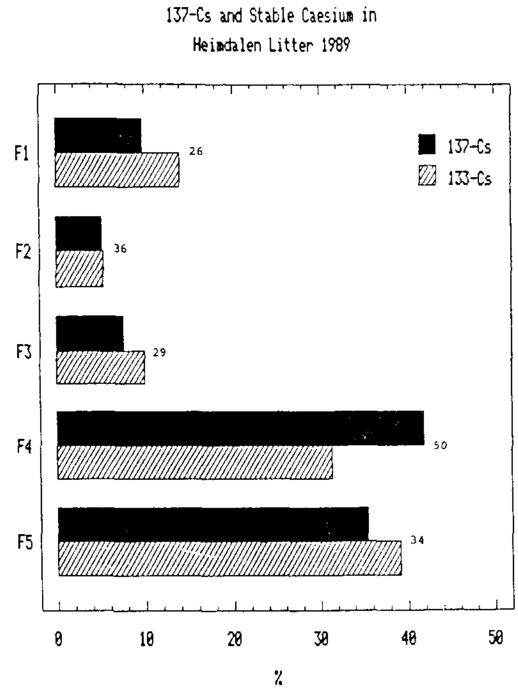


Figure 11.7 Distribution of Cs-137 and stable cesium in soil sample (0 - 2 cm). 137/133 ratio is given after the bars (Bq/µg). Central Norway. 1989.

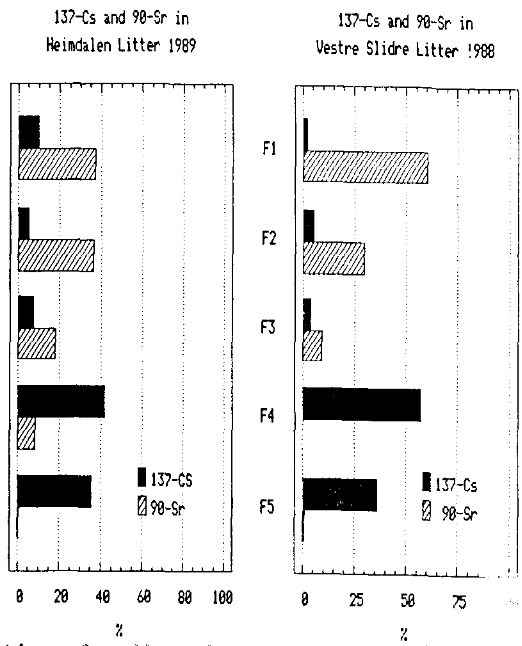


Figure 11.8 Relative distribution of radioactive cesium nuclides a Sr-90 in the different fractions (0 - 2 cm sample) Hei- dalen 1989 and Vestre Slidre 1988.

11.3 Electrodialyse cell

For Cs-nuclides, the strong interaction with components in the soil implies a low concentration of Cs^{+} in the soil solution. In order to distinguish low molecular weight positively charged radio-Cs (Cs^{+}) from colloidal Cs an experimental method using electrodialysis has been developed, as part of project *AKTU-265*, whereby the Cs-nuclides

are fractionated according to size and charge properties simultaneously. The apparatus constructed is called an electrodialysis cell. The work in this project has been performed by J.P. Rambæk of Institutt for energiteknikk, Norway and Brit Salbu of the Norwegian Agricultural University. So far, the technique has been standardized for aquatic solutions containing Cs-134 tracer.

In aquatic solutions radionuclides as well as trace metals may be present in different physico-chemical forms influencing the transport, distribution and biological uptake. Information on size properties (e.g. molecular weight distribution) can be obtained for instance by dialysis, or hollow fiber ultrafiltration. Information on charge properties is, however, obtained from ionic exchange chromatography, extraction techniques or electro-chemical methods. By combining dialysis with electrolysis, information on the size and charge properties can be obtained simultaneously.

11.3.1 The electrodialysis experiments

Humic waters were collected in June 1989 from Hellerudmyra, located in a forest area outside Oslo. Concentrated water samples were obtained by low temperature evaporation at low pressure. The low molecular weight fraction of the sample from Hellerudmyra were obtained by hollow fiber ultrafiltration (ref. SA85 and LY87).

Electrodialysis cells (Figure 11.9) were made from teflon tubes having diameters of 25 and 43 mm, respectively. Each of the cells were separated by ultrafiltration membranes into three chambers having a volume 50 ml each. Pt-wires serve as electrodes and are mounted at the walls of the outer chambers. The membranes used have molecular weight cut-off levels of 500, 1,000 and 10,000 Dalton, respectively. Different tests solutions (deionized water, 0.1 M NH_4Ac and ammonium iron (III) hexacyanoferrates (II) in 0.1 M NH_4Ac (0.1 mg/ml)) were used in the dialysis cell.

For solutions high in radioactivity a gamma camera was used for following the migration of the tracer within the electrodialysis cell. For low activity solutions information on the migration was obtained by withdrawing samples of the solution from each of the cells during the experiment.

The aqueous solution (50 ml) to be investigated was transferred to the central chamber while deionized water (50 ml) or 0.1 M NH_4Ac was added to the outer chambers. During electrolysis low molecular weight charged species move through membranes towards opposite charged electrodes in the outer chambers. Neutral low molecular weight species diffuse through both membranes into the outer chambers with time. Species having molecular weight higher than the cut off levels of the membranes are, however, retained in the central chamber.

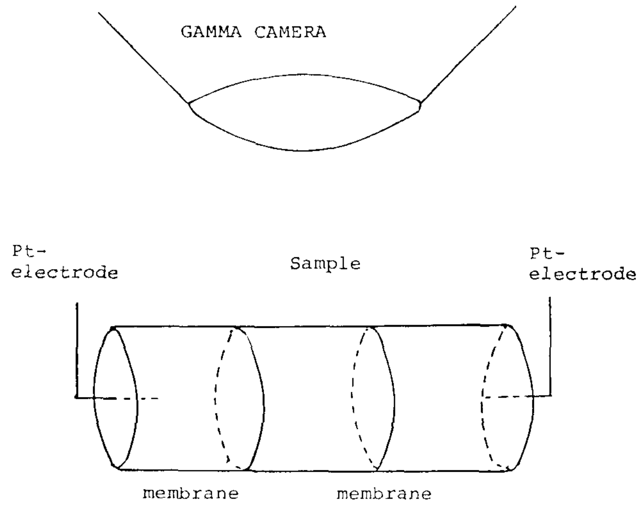


Figure 11.9 Teflon electro-dialysis cell separated with ultrafiltration membranes. Pt-wires serve as electrodes.

A radioactive tracer (Cs-134 as cloride) was used for standardization of the technique. The tracer was added to the solution 3 - 5 days prior to the experiments. The migration of 0.37 MBq 134-CsCl was followed during 10 hours of electrodialysis using the gamma camera. For lower activities (0.25 kBq Cs-134 per 50 ml solution), 1 ml aliqouts were withdrawn from the chambers during the electrodialysis, for measurement of the content of radioactive material.

Figure 11.10 illustrates the electrodialysis of Cs-134 in 0.1 M NH_4Ac solution using membranes with cut off levels of 1000 Dalton. At t=0 the tracer solution is present in the central chamber. Then migration of Cs-134 through the membrane towards the cathode takes place due to the applied potential (330 V). After 10 hours electrodialysis more than 90% of the tracer has moved to this chamber, and is accordingly present in a low molecular weight positively charged form. Smooth curves describing the increase in the cathode chamber and the decrease in central chamber are also obtained from the gamma camera system. Furthermore, it is found that sorption to the membrane surface and the equipment walls is of no significance. Therefore, the migration of Cs-134 does not seem to be affected by any built up potential across the membrane.

In order to investigate the association of Cs-134 with aquatic humic substances, the tracer was added to the concentrated samples of humic water from Hellerudmyra, mentioned earlier in this subchapter. Ultrafiltrate of water from Hellerudmyra with molecular weight <10,000 Dalton served as standard. Dialysis for 3 days (without potential) indicated retention of about 0.4% per mg total organic carbon per liter.

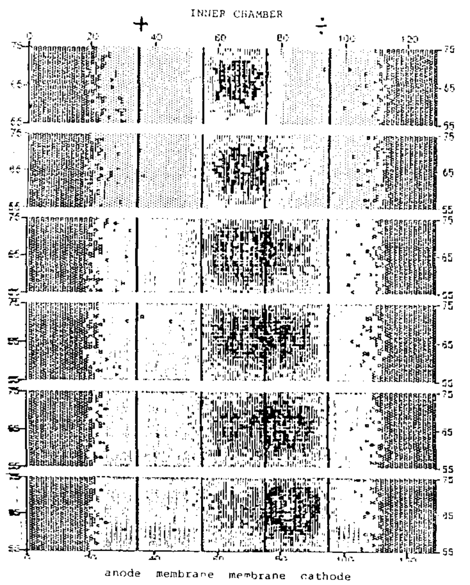


Figure 11.10 Migration of Cs-134 during electro-dialysis (10 hours) followed by gamma camera.

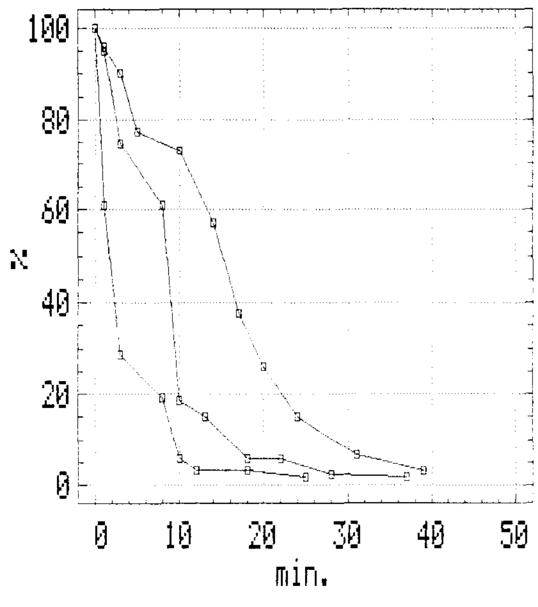


Figure 11.11 Cs-134 in waters rich in humic substances. Lower curve is ultrafiltrate, middle curve concentrate factor 2, and upper curve concentrate factor 20.

Electrodialysis, as shown in Figure 11.11, indicates that the rate of migration increases with the amount of aquatic humus. However, the association with Cs is rather weak, as binding rupture occurs at the applied potential.

The experiments demonstrate that the migration is strongly dependent on the electrolyte (ionic strength) and on potential applied, while no differences are seen when changing membranes (Mw 500-10,000 Dalton). At a potential of 330 V, chosen for use in these experiments, migration in NH_4Ac is slow (t > 3 hours), while it is fast in deionized water (t < 2 min). Therefore, the standard solution applied should have a composition of potential determining ions similar to that of the sample investigated.

11.3.2 Results of the electrodialysis experiments

The investigation of humic waters show that in such an environment Cs-134 is predominantly present as low molecular weight positively charged species and only a weak interaction with high molecular weight organics is observed.

Experiments were also performed with addition of Prussian Blue (ammonium ironhexacyano ferrates) to a solution containing Cs-134. Prussian Blue is one of the substances that have been adopted as countermeasure in Norway, as it inhibits uptake of cesium from fodder to animals. In this solution a rapid interaction of Prussian Blue with Cs-134 was seen. During electrodialysis the Cs-134 migrated towards the anode, but was retained at the membrane surface. No significant amounts of Cs-134 were observed in the outer chambers. Thus, Cs-134 was present as high molecular weight negatively charged forms in the solution. These experiments give additional proof that Preussian Blue transforms low molecular weight bioavailable forms of cesium into high molecular weight inert species.

The technique investigated seems to be promising for fractionation of low molecular weight charged species, but more standardization of the electrolyte is needed before separation of cesium nuclides in soil solutions can be achieved.

11.4 Transfer from vegetation to sheep

The transfer from vegetation to sheep depends on degradation processes in the GI tract as well as on the physico-chemical forms of radiocesium. As part of project *AKTU-265* experiments have been performed in which the bioavailability of radiocesium has been investigated by measuring the release of radiocesium from plant material directly to the rumen liquid (stomach juice) of sheep, both in sacco (in vivo) and in vitro. This work has been performed by G. Østby, G. Riise and B. Salbu of the Isotope and Electron Microscopy Laboratories and T.H. Garmo and K. Hove of the Department of Animal Science of the Norwegian Agricultural University.

Information on the transfer of radio-Cs from fodder to animals is usually obtained from feeding experiments. However, this technique is extremely time consuming and expensive. Furthermore, transformation processes taking place in the litter layer may change the physico-chemical forms of radio-Cs with time, and make it necessary to determine "transfer factors" or similar quantities repeatedly. Therefore, a more rapid and inexpensive technique for estimating the relevant available fraction of cesium-nuclides in the vegetation is urgently needed.

Accurately weighted fresh or dried (40^0) samples of grass, herbage, mushrooms, lichens and litter collected in Central Norway were used in the experiments, both the in sacco and in vitro experiments (Table 11.2).

Sample type	Year	kBq/kg	In sacco (% released)	In vitro (% released)
Grass	1986	10	78	15
Grass	1987	10	98	65
Grass	1988	5		65
Grass	1989	20		90
Herbage				
Rumex acetosa	1986	12	98	
Rumex acetosa	1987	7	98	60
Deschampsia flexnosa	1989	7		85
Deschampsia caespitosa	1989	10	85	
Mushroom				
Tricoloma album	1988	150	100	75
Rozites caperata	1988	259	100	75
Lichen				
Cladonia stellaris	1988	25	71	32
Litter	1988	40		1
Litter	1989	48		10

Table 11.2 Characteristics of the contaminated vegetation, and release of radiocesium (%) from vegetation after 48 hours incubation in rumen liquid.

11.4.1 The "in sacco" (in vivo) experiments

24 nylon bags (porediameter of 35-40 mm) containing 1 g samples of contaminated vegetation each were placed in the rumen of a fistulated sheep. Fistulation makes it possible to place the samples directly into the stomachs of the sheep. Four bags were withdrawn after incubation times of 1 h, 2 h, 4 h, 8 h, 24 h, and 48 hours, respectively. The activities of Cs-134 and Cs-137 in each sample bag were measured prior to experiments. After withdrawal of the bags the activity of the Cs-nuclides, as well as dry weight of the sample material were determined.

11.4.2 The "in vitro" experiments

Six aliquots of 2 g plant material each were added to 50 ml rumen liquid withdrawn from fistulated sheep. After incubation at 30°C at different time intervals (1 h, 2 h, 4 h, 8 h, 24 h, 48 h) the aliquots were ultracentrifuged (40 000 rpm) in order to remove high molecular weight components (> 50 nm). The activities of Cs-134 and Cs-136 were determined prior to the experiment and after the centrifugation.

11.4.3 Results from the "in sacco and vitro" experiments

The in sacco measurements show, as in Figure 11.12, that the radioactive Cs-nuclides are released very rapidly from the nylon bags containing the contaminated plant material. For all samples except lichens about 90-100% of the Cs-nuclides are released within 1-2 hours even though significant amounts of plant material was retained in the bag after 48 hours of incubation (20-90% on dry weight basis). The cumulative releases are also shown in Table 11.2.

No significant differences are seen between the release of Cs-nuclides from mushrooms, grass and herbage. The pore diameter of the nylon bags is, however, rather large (35-40 mm) and radiocesium released may be associated with high molecular weight components in the rumen liquid suspension.

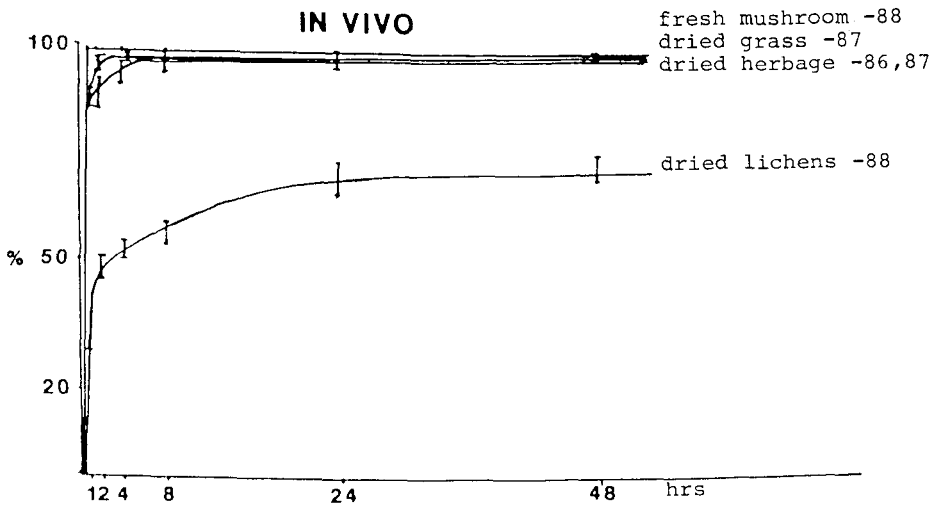


Figure 11.12 Release of radiocesium (%) from contaminated vegetation during 48 hours incubation in sacco.

From the in vitro experiments (Figure 11.13), it is seen that about 60 - 70% of the radiocesium released from plant material during 1 - 2 hours incubation is present as low molecular weight species (<50 nm). The results indicate that the low molecular weight fraction of Cs-nuclides depends on type of vegetation and is not significantly influenced by the drying prodedure. The release from litter is extremely low, which indicates that contamination of vegetation due to resuspension from the litter layer should not be considered important for uptake in sheep.

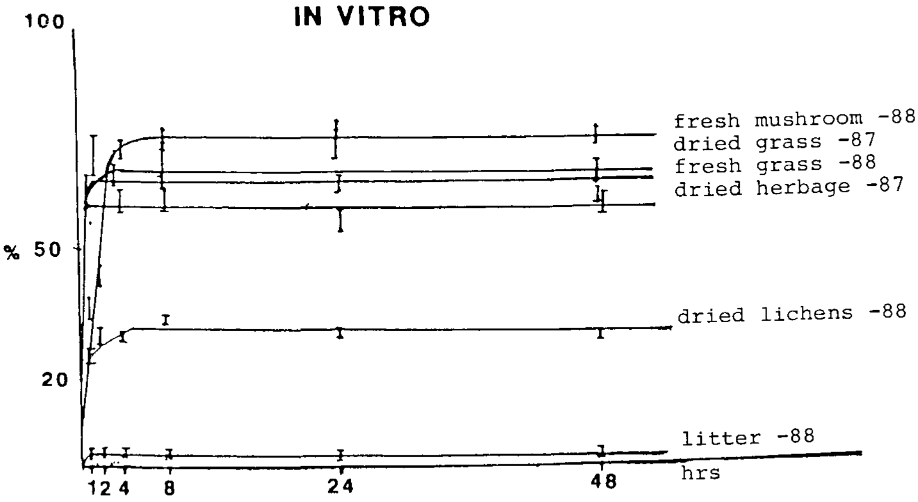


Figure 11.13 Release of low molecular weight radiocesium (%) from contaminated vegetation during 48 hours in vitro incubation with rumen liquid.

Figure 11.14 shows a comparison between experiments on samples from different years. It was found that in grass collected in 1986 about 15% of the total activity of the Cs-nuclides was present as low molecular weight forms after incubation. For grass collected in 1987 and 1988 this fraction was about 65%, while 90% is obtained for grass collected in 1989. The probable explanation for these differences is connected to surface contamination of vegetation by resuspension. From Table 11.2 it is seen that most of the cesium in litter is in inert forms (high molecular weight forms). Resuspension of cesium from Chernobyl would be expected to be most pronounced in 1986, and decrease over the following years. Thus, the results from these experiments indicate that the major portion of cesium nuclides in grass collected in 1986 was associated with resuspended litter/soil, while this portion was much smaller in 1987 and 1988. In grass collected in 1989 almost all cesium in the grass was due to root uptake.

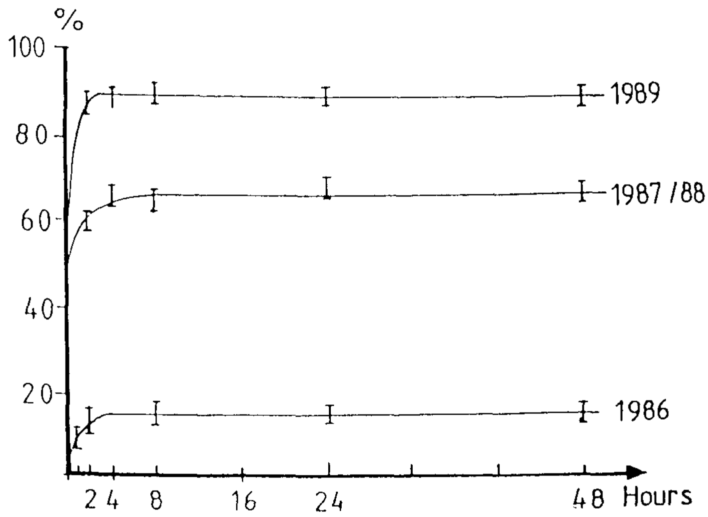


Figure 11.14 Release of low molecular weight radiocesium (%) from contaminated grass collected in 1986, 1987, 1988 and 1989. Central Norway.

The difference in bioavailability found in these experiments, and described in the preceding paragraph, is also found in experiments where the transfer from grass contaminated with Cs-134 to milk is measured from feeding experiments (ref. HO88a and HO88b). For grass collected in 1986 the transfer was low (3-4%), but increased in 1987 (8.8%) and in 1988 (10%). For Cs-134 plus tracer (CsCl) the transfer was 12%.

Compared with results obtained for the in vitro incubation experiments remarkable good agreement between these and the feeding experiments is seen for 1987 (60-65%). For grass collected in 1986 and 1988 the in vitro incubation seems to underestimate the transfer to some extent. However, the differences may also be attributed to variation in the surface contamination (e.g. resuspension) of the grass investigated.

The results from project *AKTU-265*, as well as those from the feeding experiments, imply that, even though the total activities of the cesium nuclides in vegetation may decrease with time, the relative fraction of easily available Cs-nuclides may increase with time.

From the in vitro experiments it appears, in contrast to the in sacco experiments, that the low molecular weight fraction varies considerably between the different types of vegetation.

The in vitro incubation technique described here seems therefore to be a promising technique for obtaining information on the potential availability of radiocesium in vegetation. Compared to feeding experiments, in vitro incubation with rumen liquid is a fast technique, giving results in 4 - 8 hours, and reproducible results are obtained at low expenses.

12. SHORT-TERM MITIGATING ACTIONS

Mitigating actions have one purpose: to reduce exposure to radiation of members of the population or the population as a whole. The actions fall into two main categories; short-term and long-term actions. In this report these are treated in two separate chapters, as they can be discussed quite independent of each other.

Health consequences, economic consequences, mitigating actions and action levels are all linked; in the real world, as well as in an accident consequence assessment model. Although it has been recognized that the manner in which these items are coupled, and the magnitude of parameters involved must be different from one country to another (and even from one area to another), assessments are often performed using parameters and assumptions from American Reactor Safety Study from 1975 or insufficient modifications of these. The initial purpose of the AKTU projects concerned with mitigating actions was to amend this situation for Nordic conditions, in particular concerning the economic parameters. Chernobyl, however, changed the direction of the projects; because the time after the accident was a time when both short-term and long-term mitigating actions were considered and activated in some of the Nordic countries.

Two projects addressing mitigating actions were under way before the accident in Chernobyl took place. But the new situation caused a delay in one of these projects, changed the content of the other project, and four new projects concerning various aspects of mitigating actions were added to the AKTU program area. Two of the new projects were performed in Finland and two in Norway.

The short-term (or early phase) mitigating actions usually considered in ACA models are the following:

- evacuation
- sheltering
- use of stable iodine

Traditionally the position has been that short-term mitigating actions are aimed at prevention or mitigation of possible acute health effects; effects encountered only above certain threshold doses. Post-Chernobyl experience, however, is that mitigating actions may be warranted also at far distances, where the doses are far below these thresholds. But at these lower doses there are other types of short-term mitigating actions that may have to be considered; not the ones listed above. The possible types of mitigating actions are so many and of such diverse character that a list like the one above has not been prepared.

This chapter contains descriptions of two AKTU projects, *AKTU-272 and AKTU-280*. The first of these is an investigation, using an ACA computer model, of the effect upon the consequences (health and economic) of various alternative mitigating action strategies. This project has addressed the problem from the point of view usually adopted in ACA models, and the types of mitigating actions considered are the ones listed above. The second project is concerned with the situation in Finland shortly after the Chernobyl accident took place. Various types of short-term mitigating actions (some concerned with quite specifically Finnish aspects of the situation, like the advicability of using rain water on the hot stones in the sauna) were then considered by the authorities. In this project the various recommendations given are summarized and re-evaluated.

Project *AKTU-272* is equally concerned with long-term and short-term mitigating actions. In the present report it has been chosen to present the short- and long-term mitigating actions in separate chapters. The main reason is that it was considered interesting to view the mitigating actions both from a theoretical and practical point of view in the same chapter, rather than place the two projects in two chapters, each covering all types of mitigating actions. The parts of the two projects concerned with the long-term aspects are described in the appropriate chapter. The part of project *AKTU-280* concerned with long-term mitigating actions is, however, quite modest; simply because the post-Chernobyl situation in Finland was such that rather limited and not very costly long-term actions were warranted.

12.1 ARANO-calculations of short-term mitigating actions, proejct AKTU-272

Project *AKTU-272* has been concerned with investigating the impact upon the consequences of alternate mitigating actions. Both short-term and long-term mitigating actions have been examined in the project, and are reported in the appropriate parts of the present report. Health and economic consequences, mitigating actions and action levels are linked. The cost of different types of mitigating actions may be very different, and so may be the efficiency of different mitigating actions, measured in saved lives. And it is not necessarily the most costly mitigating action that is the most efficient.

The analyses of these problems have been performed using the Finnish accident consequence assessment code ARANO. The importance of various mitigating action parameters have been examined for a range of release characteristics, and various combinations of the parameters. The costs of mitigating actions, however, have only been calculated for long-term mitigating actions, and cost-effectiveness analyses have been performed separately based either upon ground shine or ingestion doses, and the results of the latter are considered in the present report.

12.1.1 The types of mitigating actions

The types of short-term mitigating actions considered are:

Evacuation

Evacuation means that the population is removed from an area. Evacuation is an immediate action and is considered to last only a few days. It shall be carried out in areas where some predetermined dose criterion will most probably be exceeded.

An evacuation can not happen at the very moment when an accident takes place. There is delay, for several reasons. First of all it takes some time before the degree of seriousness of an abnormal behaviour of the plant can be determined, it takes some time before the proper authority can issue an evacuation order/recommendation for specific geographical locations, and it takes some time before the evacuation can actually be effected.

Two quantities are often used in this connection: "Warning time", which has some relation to the delay before an evacuation can be carried out, and "time of release".

The warning time has been given quite varying definitions. In (ref. WA75) it is defined as "the interval between awareness of impending core melt and the release of radioactive material from the containment building". This definition, however, does not seem to be a particularly useful one. What is really important is to know when evacuation can take place relative to the time when release from the containment building takes place, as illustrated by the two extreme situations: evacuation is completed before the plume reaches the area; the plume reaches the area just when evacuation starts, and exposes the whole population group, outdoors, to the full length of the plume. In the German ACA model UFOMOD (ref. EH88) the quantity "warning time" is not used, and one has instead defined a series of important points in time and time intervals (some of these are related to sheltering only, and are not listed below):

- time of accident (end of chain reaction)
- time of end of release
- initial delay (time interval between time of accident and start of mitigating action (evacuation or sheltering))
- driving time to leave the area

In the ARANO computer program the quantity "warning time" is defined in the same manner as in (ref. WA75), but the concept is actually not used in the AKTU-272 project, as there are other parameters in the ARANO program to vary the integration time for ground shine. In ARANO the user can freely select the time point(s) of evacuation events as calculated from the time of arrival of the plume to a certain location. This delay time can include several factors, but the final aim is to describe how long population will be exposed before evacuation can take place safely and in an organized manner.

In the project the sensitivity of the results to the length of the delay time is studied by using two alternative values; 8 or 24 hours. The full plume passage has always occurred before evacuation starts, since the delay time is reckoned from the time when the plume reaches a certain position. In the report from the AKTU-272 project it is also referred to a delay time of one week, which for all practical purposes means that there is no evacuation. One week is chosen as the boundary between short-term and long-term considerations. A population movement after one week will be referred to as relocation.

This means that the following three situations are compared in the sensitivity study:

- Being sheltered for a while, and being evacuated 8 hours after plume passage.
- Being sheltered for a while, and being evacuated 24 hours after plume passage.
- Staying sheltered until the timepoint of relocation (or the decision time whether relocation is appropriate).

The evacuation area is given alternative sizes of 5, 10 or 20 km downwind distances in the affected dispersion sector, in order to study the sensitivity of the results to this parameter. Some calculations are also performed where the population is moved out of an area reaching 100 km from the release point. This case is only calculated with the longer delay time of 24 hours, and should be considered an early relocation rather than an evacuation.

It is also assumed, concerning evacuation, that it is an instantaneous measure, and no evacuation speed construction method is included. Also it is assumed that the evacuation efficiency is 100%. Because evacuation always take place after plume passage, the population has been in shelter before evacuation takes place, and evacuation and sheltering strategies are interconnected. The strategies considered in the project are given later in this chapter.

The assumption that evacuation is instantaneous; that the population somehow "disappears in an instant" at the time when evacuation starts; makes it necessary to be a little careful in drawing conclusions concerning the length of staying in the shelter before evacuation. In reality a population group will have to pass through an area of contaminated ground during evacuation. This situation, however, has been evaluated, and it is concluded that omission of ground shine exposure during the evacuation process will not affect the conclusions drawn significantly.

Sheltering

Sheltering means that people staying outdoors transfer themselves to the closest buildings such as houses of their own, multistory houses, cellars, or an air-raid shelter, if one is in the vicinity. This countermeasure is aimed at reducing the direct external radiation from the passing radioactive plume and external radiation from deposited radioactive materials. In models the sheltering provided by different buildings is considered with the shielding factor, which may vary widely depending on construction material, shape and size of the building. In project AKTU-272 two "types" of sheltering are considered: normal and active (local sheltering) sheltering. The latter implies that people seek better shielded places in their houses (like basements) or go to better shielded buildings in the vicinity. Sheltering also may have the effect of reducing the inhalation doses, due to the so-called filtering effect of the buildings. In the calculations in project AKTU-272, however, this effect is disregarded, partially because mistakes in ventilation strategy may easily occur, e.g. opening of windows or ventilation at the wrong time; and partially because the inhalation exposure pathway is usually not the dominant exposure pathway.

Use of stable iodine

Inhaled or ingested radioiodines are quickly absorbed into blood and concentrated in the thyroid. This may result in large doses in this gland and cause severe thyroid damage. There are a few different compounds of stable iodine that are effective in blocking the thyroid for uptake of radioiodine. If potassium iodine (KI) is taken prior to the exposure, or up to about 30 minutes after exposure, a blocking effectiveness of about 99% is obtained. If the potassium iodine is taken longer time after exposure, the blocking effectiveness is reduced. Potassium iodine does not reduce the dose from other radionuclides inhaled or ingested. For this reason, use of iodine tablets is not an alternative to sheltering or evacuation, but may be a supplement. And since the aim of use of iodine tablets is to reduce thyroid doses, while the aim of sheltering and evacuation is primarily to reduce doses to other body organs (except if the release consists only of or predominantly of iodine), these types of mitigating actions are evaluated independently of each other. According to earlier studies the risks associated with use of tablets containing stable iodine are very small.

12.1.2 Criteria for short-term mitigating actions

The dose level at which a certain mitigating action is intended to be introduced must be specified for each type of mitigating action. Otherwise, of course, the computer program will not know when to activate the different types of mitigating actions. Except, that is, if one has chosen an alternative method for instructing the computer program. In ARANO a combination of these two methods has been utilized. In connection with long-term mitigating actions dose criteria are specified, and this is described in the appropriate chapter. Concerning short-term mitigating actions, the simpler solution is chosen, by specifying e.g. that evacuation is effected 8 hours after plume passage out to a distance of 20 km. The time and the distance can be changed from one computer run to the next, and the impact upon the results of this variation be investigated. The conditions pertaining to sheltering are specified in a similar deterministic (not dose-dependent) manner.

National practice concerning dose criteria is a different matter. The authorities set predetermined action levels for different types of mitigaing actions, based upon recommendations from ICRP and IAEA. A consistent international practice, however, is missing. Nordic default values for short-term mitigating actions are collected in Table 12.1. Norway is not included, as there are no established criteria relating to short-term countermeasures, and in Denmark only a criterion concerning sheltering is available. This information is given here in order to provide som background, but is not actually utilized in this project.

Countermeasure	Dose-criterion/distance Denmark	 Finland	 Sweden
Evacuation	-/-	100 mSv/20 km	-/15 km
Sheltering	1 mSv/-	-/100 km	-/15 km
Iodine tablets	-/-	200 mSv/20 km (Infant)	-/15 km

Table 12.1 Applied intervention levels.

12.1.3 About the calculations and the results

Calculations were carried out for seven different sets of hypothetical release characteristics. There were three release magnitudes involved (but all with 100% release of noble gases). In addition release height was varied, and also start time of the release (but only for the largest of the releases). The actual release specifications are shown in Table 12.2.

Release category	Start of release (h)	Release duration (h)	Release height (m)	Nuclide groups Xe-Kr	I-Cs-Rb-Te-Sb	Sr-Ba-etc.
A_1	1	3	20	1.0	0.1	0.01
A_2	10	3	20	1.0	0.1	0.01
A_3	1	3	100	1.0	0.1	0.01
B_1	1	3	20	1.0	0.01	0.001
B_2	1	3	100	1.0	0.01	0.001
C_1	1	3	20	1.0	0.001	0.0001
C_2	1	3	100	1.0	0.001	0.0001

Sr-Ba-etc. also includes Rh, Co, Mo, Tc, Y, Zr, Nb, Ce, Pr, Nd, Np, Pu, Am and Cm.

Table 12.2 Source term categorization; release fractions of the core radioactive inventory with a net output capacity of 1000 MWe.

A number of other parameters have impact upon the results of calculations. The values used in the calculations in project AKTU-272 are shown in Table 12.4.

The so-called filtering effect of houses is not taken account of in these calculations, as mentioned previously in this chapter. This effect has been investigated in another project, *AKTU-245*.

The calculations can not be performed without input of weather data and population data. Since the purpose of the calculations in *AKTU-272* is not to evaluate the risk for any specific plant or site, but only to see how variations in mitigating actions employed impact upon the consequences in a generic sense, the weather and population data might have been constructed in an artificial manner, like for instance letting the population be homogenous over the whole area and add a few population concentrations of varying size and position. However, when constructing input data of this kind, there is a danger that the constructed data may become so different from any real site, that the conclusions drawn from the calculations are also irrelevant in relation to any real site. In order to avoid this situation, one chose in this project to use weather and population data for the Loviisa nuclear power plant in Finland.

The weather data are given as frequency distribution within a system of 12 directions, stability classes A - F, seven wind speed classes, and rain occurence/non-occurence. The weather statistics have been measured in a mast at the Loviisa plant. The actual data are not given in the present report, but are included in an annex to the report from the project. This information is not needed in order to understand the results of project AKTU-272 properly.

The population data used are given in Table 12.3 out to a distance of 100 km for the 12 directions. Further away than 100 km it is assumed that there is a uniform population density of 30 persons/km².

Sector	Distance interval (km)							
	<5	5-10	10-20	20-30	30-40	40-60	60-80	80-100
1	36	32	1216	1627	2277	37533	38814	3088
2	53	23	1526	2527	3538	14996	3554	5738
3	33	10	217	22984	32178	18821	4592	2738
4	14	65	31	11	15	10	0	0
5	10	0	0	0	0	0	0	0
6	3	0	1	0	0	0	0	0
7	0	0	0	0	0	0	0	0
8	0	4	4	88	124	0	0	0
9	0	1	116	1392	1949	6794	340849	399193
10	46	55	707	11260	15765	12944	82638	83770
11	37	1603	6711	1130	1581	11466	19053	25537
12	43	32	1999	1536	2150	6245	79612	41175

Table 12.3 Population data. An average population density of 30 persons per km² is used in the distance interval 100-300 km.

	Short-term		Long-term
Population fraction	0.2	0.8	1.0
Shielding factors:			
- Normal shielding, cloud shine	1.0	0.6	-
- Active shielding, cloud shine	0.6	0.2	-
- Normal shielding, ground shine	0.4	0.4	0.3
- Active shielding, ground shine	0.2	0.02	-
Breathing rate	(10 m³)/(8 hour)		
Integration time of inhalation dose	30 days*		

* The portion of the inhalation dose accumulated within the body during the time interval 7 - 30 days is accounted for with a weighting factor of 50%.

Table 12.4 Shielding data and various other parameter values.

Local sheltering is taken into account by application of two alternative sets of shielding factors (normal and active sheltering), given in Table 12.4. The shielding factors and the population fractions involved are valid for the whole distance interval out to 100 km, thereafter shielding conditions according to normal living habits are used.

The different evacuation strategies considered in the calculations are listed in Table 12.5.

Evacuation strategy	Evacuation time	out to distance	Relocation* time	in distance interval
Reference case	-	-	7 days	0 - 100 km
A	24 hours	20 km	7 days	20 - 100 km
B	8 hours	20 km	7 days	20 - 100 km
C	24 hours	10 km	7 days	10 - 100 km
D	8 hours	10 km	7 days	10 - 100 km
E	24 hours	5 km	7 days	5 - 100 km
F	8 hours	5 km	7 days	5 - 100 km
G	8 hours	20 km	24 hours	20 - 100 km
H	8 hours	5 km	24 hours	5 - 100 km

* This is either the time at when it is decided whether relocation is warranted, or (for G and H) the time when early relocation is carried out.

Table 12.5 Evacuation matrix in the ARANO calculations. The evacuation area consists of the 30^0 sector in the wind direction and the adjoining sectors. Evacuation is not considered outside of 20 km. The evacuation time is the time period between time of arrival of the plume and the time at which the momentary evacuation occurs. The reference case is, of course, the case of no evacuation anywhere.

The use of stable iodine is taken into account simply by reducing the iodine inhalation dose to the thyroid by a factor of 100. The use of stable iodine is not included in results obtained in the context of sheltering and evacuation considerations. Early health effects are based on the bone marrow dose in this project. The effect of stable iodine on thyroid doses via inhalation is considered separately in section 12.1.5 of this report.

The results from the calculations are presented in a couple of different ways. Some examples will be given here; but apart from these only few of the results will be included in the present report.

One type of result is simply the dose as function of distance from the release point for a specific weather condition. A sample is shown in Figure 12.1 for the largest release with release height at 20 meters, and weather conditions Pasquill D and wind speed 5 meters per second. The doses in this figure are presented without any shielding factors, and consequently the relative importance of different dose pathways is dependent on sheltering conditions.

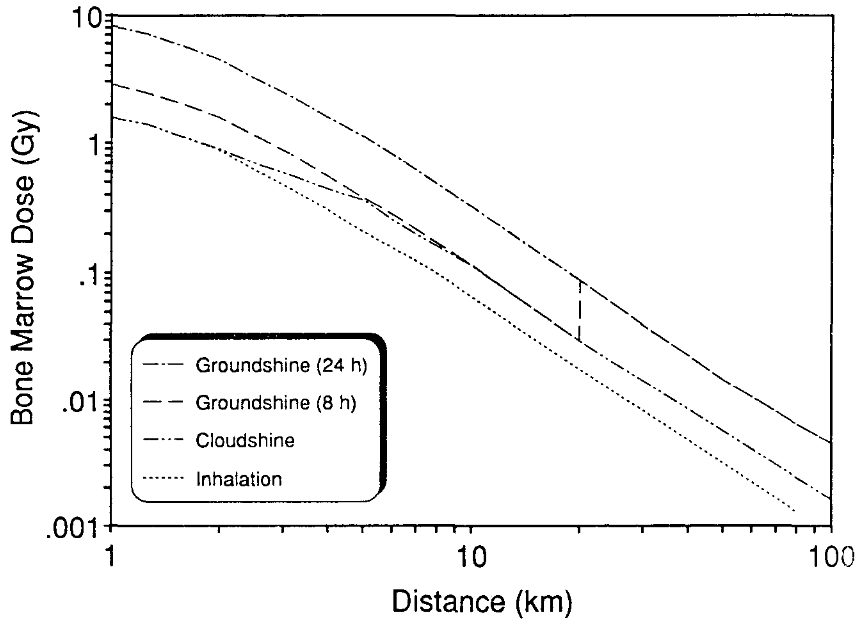


Figure 12.1 Individual bone marrow dose for an unsheltered person as a function of distance in case of release category A1 (large release, H=20 m) and stability class D, wind speed 5 m/s. Two integration times are used for the ground shine up to the distance of 20 km.

The different components of the short-term dose are shown: Cloud shine, inhalation, ground shine for integration times 8 hours and 24 hours. For a release of this type the inhalation dose is always lower than the doses via the other exposure pathways. Ground shine is usually the most important pathway, but this depends upon the time of exposure. For the two exposure times shown in the figure, ground shine dominates absolutely when the integration time is 24 hours, but ground shine and cloud shine are almost equal for distances longer than about 5 km when the integration time is only 8 hours.

The different types of mitigating actions have different impact upon the components of the short-term dose. Sheltering has no impact upon the inhalation dose, according to the assumptions used in the project AKTU-272. It is also assumed that evacuation always takes place after plume passage, and it has accordingly no impact upon either inhalation dose or cloud shine dose. Said in a different way:

- The inhalation dose can neither be avoided, nor reduced.
- Sheltering reduces both the cloud shine and the ground shine doses; the latter more than the former.
- Evacuation reduces the ground shine dose.

This also implies that when the cloud shine and ground shine doses have been reduced to a certain level, little is gained from even better sheltering or earlier evacuation, since the level has been reached ter sheltering or earlier evacuation, since the level has been reached where the inhalation dose is dominant, for the population group in question. However, it should be remembered that when active sheltering is assumed, it is always in these calculations assumed that 20% of the population only has normal sheltering. For this group and the sheltering scenarios discussed here, it is unlikely that the inhalation exposure pathway will ever be dominant, and it is probable that this population group will dominate the total consequence pattern.

In Figure 12.2 another way of presenting the results is shown. The calculations are for the reference case concerning evacuation strategy, and for the basis for judging the benefit from alternative evacuation strategies. The figure also demonstrates the efficiency of sheltering in reducing the number of persons exceeding certain threshold values (0.1 Gy corresponds to an evacuation dose criterion for Finland; see table in the criteria chapter, and 1 Gy corresponds to threshold assumed for radiation sickness in AKTU-272), and the number of early deaths (probability of early death assumed to increase from zero to one over the range 2 to 5 Gy) respectively. As could be expected, for integration time one week, the ground shine dose will dominate completely, and improvement of sheltering is of course very efficient in reducing doses.

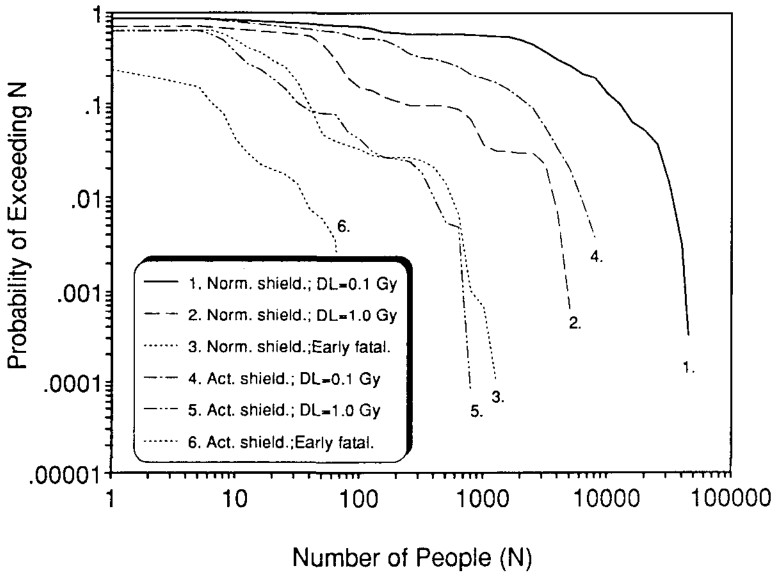


Figure 12.2 The CCDFs for the number of persons exceeding different dose threshold values (0.1 and 1.0 Gy) and of early fatalities under normal and active shielding conditions. No evacuation is assumed and relocation takes place after a week. Release category A_1.

The preceding figure presents the results in the form of so-called CCDF curves, which is a traditional manner in which to present results of risk-related calculations. CCDF means Complementary Cumulative Distribution Function. The curves are read in the following manner:

The vertical axis gives the probability that the function on the horizontal axis exceeds a specific value. In the above figure the function on the horizontal axis is the "number of people exceeding the threshold value". If we look at the upper of the curves and probability 0.1, then the corresponding point on the curve means that there is a 0.1 (10%) probability that more than ca. 12,000 persons exceed the threshold value under the conditions for which this curve is valid. It is "more than", since the CCDF shows the probability of an effect being equal to or larger than a specific value.

The way of presenting the results that is used to the largest extent in the report from project AKTU-272 is shown in Figure 12.3, and the main purpose here is to compare different evacuation strategies directly. This is also a CCDF type curve. Curves like these have been prepared for all releases and for both types of sheltering.

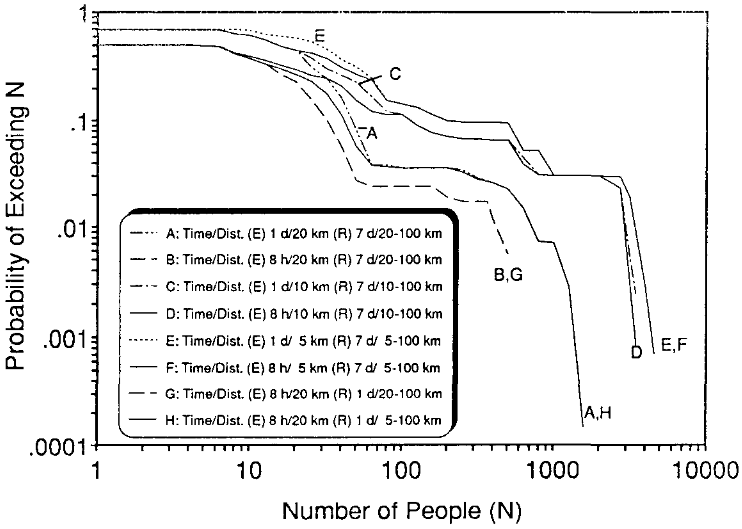


Figure 12.3 The CCDFs for the number of persons who are exposed to doses exceeding the threshold value of 1.0 Gy in different evacuation/relocation strategies (cf. Table 12.5). Release category A_1.

12.1.4 Impact of mitigating actions upon the consequences

Conclusions can be drawn by comparing all the different curves of the type presented last in the previous subchapter, but it is easier to compare the various mitigating action strategies if only single results of some type are considered. Table 12.6 contains the expectation values and the maximum values for early illness cases resulting from the largest release with release height 20 meters and start time for release at 1 hour.

In order to prevent or mitigate possible acute health effects, the dose critera should be based upon radiation illness cases (dose criterion of 1.0 Gy). This, of course, is only one of several possible ways to evaluate the situation regarding mitigating actions, but it has been used in the following, where the conclusions drawn are based upon the calculated numbers of early illness cases when alternative mitigating action strategies are employed.

Evacuation strategy	Expectation values		Maximum values	
	Normal sheltering	Active sheltering	Normal sheltering	Active sheltering
No evac.	195	23	4,830	781
A	41	7	2,010	226
B	19	5	563	162
C	143	15	4,330	545
D	128	13	3,940	501
E	187	20	4,820	778
F	181	19	4,810	777
G	19	5	563	162
H	36	7	2,000	225

Table 12.6 Conditional expectation values and maximum values of early illness cases due to release category A_1 for the various combinations of sheltering and evacuation strategies.

It is clearly seen that better sheltering is efficient in reducing the early illness cases; both expectation and maximum values for all evacuation strategies.

As for evacuation, it can be seen that when the evacuation area is the smallest (these are evacuation strategies E and F, where the radius is 5 km) little is gained by performing the evacuation fast (at 8 hours; F) as compared to later (24 hours; E). This indicates that a large fraction of the population group at risk (contributing to the consequence) is located outside of the evacuation area in this case. The same is more or less true when the evacuation area is of 10 km radius (strategies C and D). For strategies A and B, there is a significant difference in consequences between the two alternative evacuation times. This result is an indication that an evacuation area of only 5 to 10 km is too small.

By comparing the results for the different evacuation strategies, one can conclude that it is important to perform evacuation of the population group in the 10 to 20 km interval for the release examined in this case (A_1), since increase of the evacuation area from 10 to 20 km (from strategy C to A or from D to B) results in a significant decrease in early illness cases, both expressed as expectation values and maximum values. The possible importance of the area beyond 20 km is seen by comparing strategy G with strategy B (the difference between these strategies being the relocation time beyond 20 km), and since the results are identical, the area beyond 20 km is quite unimportant in all cases.

To summarize the conclusions concerning the effectiveness of measures to mitigate early illness cases:

- Sheltering is very important, and there is a large difference between active sheltering and normal sheltering.

- For release A_1 it seems that the evacuation area should have a radius of between 10 and 20 km. Decrease of the evacuation area to 10 km results in a significant increase in consequences, while increase beyond 20 km or decrease beyond 10 km gives little additional reduction or increase, respectively, in consequences.

- When evacuation is performed earlier in the interval 10 to 20 km, there is a significant impact upon the consequences.

It is of particular interest to note that if conclusions are drawn on the basis of the calculation results for early fatalities, the conclusions are almost identical to the ones above. This analysis has been performed, although it is not shown in the present report.

Table 12.7 contains the expectation and maximum values of early illness cases for the same release magnitude as above, but with a 10 hour delay before release (release category A_2). The evacuation strategies included in the table have been reduced to the ones with the short delay time, and the strategies involving the outer evacuation area are also not included.

Evacuation strategy	Expectation values		Maximum values	
	Normal sheltering	Active sheltering	Normal sheltering	Active sheltering
No evac.	164	14	4,350	604
B	10	3	342	64
D	98	5	3,320	398
F	149	11	4,330	598

Table 12.7 Conditional expectation values and maximum values of early illness cases due to release category A_2 for the various combinations of sheltering and evacuation strategies.

The all-over reduction in consequences is due to radioactive decay during the extra delay before release. Otherwise the conclusions regarding mitigating action strategies are just like for release A_1.

Table 12.8 contains the expectation and maximum values of early fatalities for the same release as above, but with elevated release and normal release time.

Evacuation strategy	Expectation values		Maximum values	
	Normal sheltering	Active sheltering	Normal sheltering	Active sheltering
No evac.	26	1	5,320	679
B	0.7	0.09	438	46
D	12	0.4	4,030	485
F	23	1	5,300	667

Table 12.8 Conditional expectation values and maximum values of early illness cases due to release category A_3 for the various combinations of sheltering and evacuation strategies.

The expectation values are significantly lower than for the non-elevated release A_1, but the maximum values are (except for evacuation strategy B) slightly larger. That the maximum values are larger is not in itself strange. Elevated release gives slower deposition, accordingly slower removal of radioactive materials from the plume, and this might result in larger maximum values for the consequences. Or said in a different way; a larger part of deposition will take place further from the release point, where the population density is larger.

That the maximum values for release A_3 are smaller than for release A_1 when evacuation strategy B is applied, is really the same effect as above; but gives the opposite result concerning consequences. For an evacuation time of 8 hours for the larger population group (between 10 and 20 km) the result is probably that this group does not reach the threshold for early illness in the 8 hours integration time, while at least parts of this population group will reach this threshold in the longer integration time of 24 hours used in the other evacuation strategies in this distance interval.

The conclusions regarding mitigating action strategies are the same as for releases A_1 and A_2:

- For release A_3 and active as well as normal sheltering the evacuation area should have a radius of between 10 and 20 km.

Now, to turn to the smaller releases. Both for releases B and C the number of early illness cases (or acute fatalities) are quite small or none. The results for these releases are also available, but will not be given here in the present report. The impact of sheltering can still clearly be seen. However, nothing is gained by increasing the size of the evacuation area beyond the smallest radius (5 km) calculated. On the other hand, the case where no evacuation is assumed (integration of ground shine dose over one week) results in a significantly higher number of early fatalities than when either of the other evacuation strategies investigated (B, D and F) are employed. This means that only a small number of persons quite close to the release point receive doses above the lower threshold for early illness cases. For the smaller elevated releases and active sheltering there are no early fatalities.

12.1.5 Effect of taking stable iodine

Calculation of thyroid doses with and without intake of iodine tablets to block uptake of radioactive iodine has been performed only for the largest release. It is shown that stable iodine is absolutely effective in avoiding thyroid ablations. As mentioned previously, it is assumed that the iodine inhalation dose is reduced by 99% by the intake of stable iodine. As the filtering effect of houses is not taken into account and evacuation is assumed to take place after plume passage, the other mitigating actions have no effect upon inhalation doses. Use of stable iodine was considered completely separate from calculations of early health effects on the bone marrow dose. Although there might be some interactions, the threshold dose for acute thyroid effects is, however, so high that it is obvious that the bone marrow will suffer first.

Comparison with the results of the calculations of other early health effects, however, show that intake of stable iodine is not an effective mitigating action for releases of the type investigated here, since it is the bone marrow and not the thyroid which is the critical organ. The early fatalities resulting from bone marrow doses are at least of the same magnitude as fatal thyroid health effects, even for the most efficient evacuation and sheltering strategies. From the results it is not possible to tell whether these bone marrow and thyroid health effects affect exactly the same persons. If this is not so, there might still be reason for stable iodine intake when the best sheltering and evacuation strategies have been employed.

12.2 Mitigating actions in Finland shortly after Chernobyl, project AKTU-280

Mitigating actions after a nuclear reactor accident have now, after the Chernobyl accident, been experienced in real life in the Nordic countries. Even though the Nordic countries are far from Chernobyl, both short-term and long-term mitigating actions have been imposed over various time periods and in certain areas; in some connections quite limited areas and in other connections (mainly very short time after the accident) whole countries.

The short-term mitigating actions in Finland after Chernobyl have been the subject of project *AKTU-280* (ref. BL90). The purpose of this project has been to summarize the various actions, recommendations issued by the Finnish authorities etc. and to give an evaluation. The project also cover the long-term aspects, although the main emphasis has been on the short term.

The report (draft version only is available at the present time) from project *AKTU-280* is relatively short, so that almost the full text of the parts concerned with short-term mitigating actions is included, slightly modified, in the present subchapter. The parts of project AKTU-280 concerned with long-term mitigating actions and economic considerations are incorporated in appropriate places of the present report. Project *AKTU-280* was carried out in cooperation between the Finnish Centre for Radiation and Nuclear Safety and the Finnish Ministry of Agriculture and Forestry.

In the days and weeks following Chernobyl various mitigating actions were adopted in the Nordic countries, but mostly in the form of recommendations. The situation in Finland can serve to explain the various types of mitigating actions considered, how they were adopted, and possibly also give information on how efficient and how expensive the mitigating actions were.

In Finland, the radioactive fallout from Chernobyl arrived at a time when the ground was still frozen. Northern Finland had a snowcover and even in the southernmost parts of the country there were patches of snow on the ground. The growing season had not started. Thus the short-lived isotopes, such as iodine 131, did not enter food chains effectively. From July 1986 onwards, the only significant radionuclides in foodstuff were cesium-137 and cesium-134.

The Chernobyl accident occurred on 26 April 1986. Preliminary information on the fallout situation in Finland was based on measurements at fixed monitoring stations of the Ministry of the Interior and the Finnish Defence Forces (ref. ST86a, ST86b and ST86c), the first slight increase in environmental gamma radiation being observed in the evening of 27 April. The rainfall on 29 April and in the following days created an uneven fallout distribution, which effected the southern half of the country. The Finnish Centre for Radiation and Nuclear Safety (STUK) made measurements using a mobile gamma spectrometer and vehicle-mounted sensitive Geiger-counters. The measurements in the first few weeks indicated that not only the intensity but also the composition of the fallout differed significantly in various parts of the country. Notably, the proportion of short-lived radionuclides such as Te-132/I-132 was higher in areas of early, heavy rainfall. This presented a problem for the evaluation of later dose rate levels and accumulated doses.

In the autumn of 1986, when the environmental dose rate was caused mainly by Cs-137 and Cs-134, STUK carried out a more detailed survey of the area affected by the Chernobyl fallout. A total of 16000 road kilometers was measured. Figure 12.4 from (ref. AR89) presents the results of the survey in map form, giving the dose rate on 1 October 1986.

12.2.1 Advice on emergency protective action

At no stage did the radiation situation require actual protective action, such as taking shelter indoors or in civil defence shelters. Civil defence plans for emergency situations include a warning level at 200 μSvh^{-1} (population has to stay indoors) and an alarm level at 2000 μSvh^{-1} (population has to seek shelter immediately). Both levels are "at the latest" levels, given as guidance in case regional or local authorities have to make the decision. The highest comfirmed gamma radiation reading in Finland following the Chernobyl accident was 5 μSvh^{-1}.

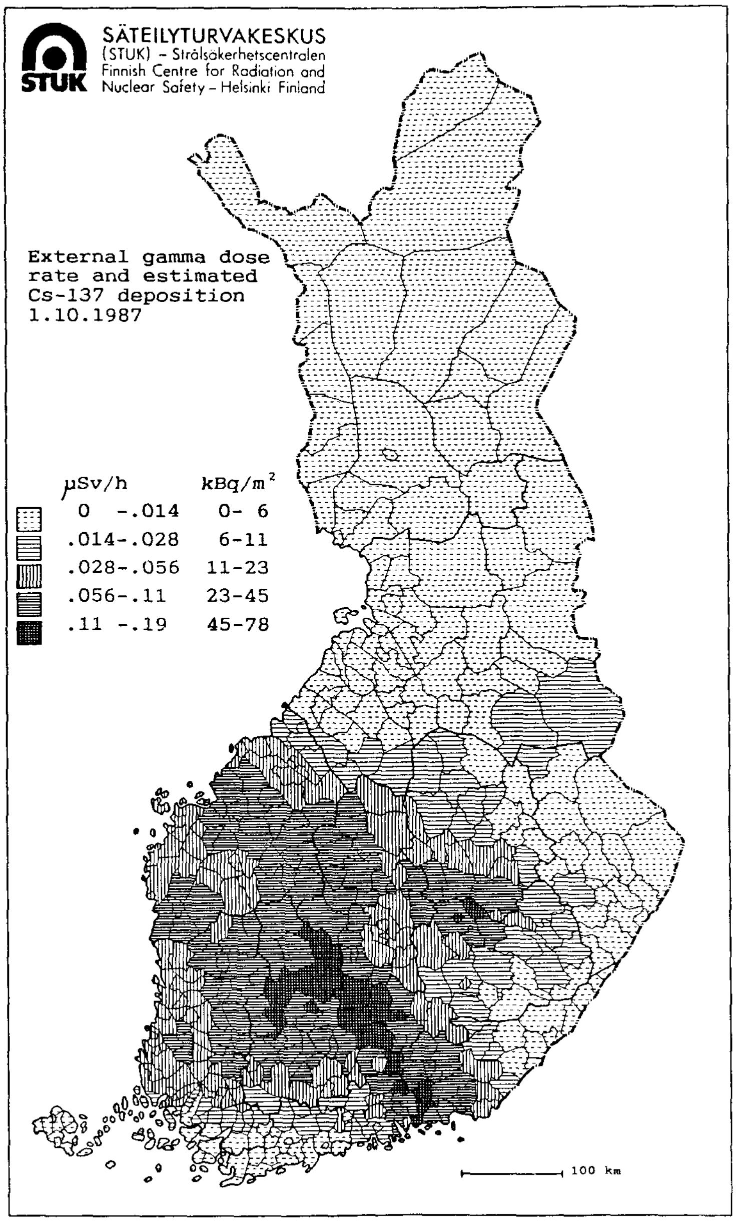


Figure 12.4 External dose rate (µSv/h) and estimated Cs-137 surface activity caused by the Chernobyl fall-out in Finland. 1 October 1987.

12.2.2 Recommendations involving intervention

The following is a list of recommendations given by STUK and other competent authorities in the period 28 April to 20 May 1986 (ref. ST86b).

Travel abroad

30 April. Voluntary evacuation of Finnish citizens working or studying in the Kiev area was recommended. (A special Finnair flight to Kiev was scheduled for this purpose).

30 April. Unnecessary travel to the Ukraine and eastern parts of the East European countries was discouraged, awaiting further information about the situation.

6 May. The recommendations about travel abroad were reformulated. Travelling to an area within 50 kilometers from Chernobyl was discouraged altogether, whereas for travel to an area within 500 kilometers from Chernobyl certain precautions were recommended, including taking a 200 mg single dose of potassium iodide.

16 May. The travel restrictions were in force until this date.

Rain water

2 May. Cows should not be watered with rain water.

2 May. Rain water should not generally be used as drinking water. This was specified on 3 May to apply also to cooking and e.g. use on the stones in the sauna.

7 May. Children's play in rain water puddles should be discouraged. This was specified on 9 May to include gutters and, in general, places where rainwater had accumulated.

15 May. The restrictions on rain water use were cancelled on this date.

16 May. The advice on childrens's play was withdrawn on this date.

Plants and mushrooms

7 May. Recommendation to postpone the sowing of lettuce, spinach and other fast-growing vegetables.

7 May. The consumption as such of plants (e.g. nettles) and mushrooms (morels) collected in the wild was discouraged. Use after boiling was approved but restricted to once or twice a week.

15 May. These recommendations were in force until this date.

Pasture

7 May. It was recommended not to let cows out on pasture for the time being.

12 May. The scope of this recommendation was further specified on this date.

26 May. The recommendation was in force de facto until this date.

12.2.3 Recommendations involving non-action

In the situation caused by the Chernobyl accident the demand for information on behalf of the public went beyond the official recommendations. Therefore, the following explicit reassurance was included in the recommendations given on 7 May:

- water from wells and public waterworks is safe to use,
- no restrictions on staying outdoors are warranted,
- pregnant women need not take any special precautions,
- breast feeding is not restricted.

12.2.4 Recommendations concerning protection of occupational groups

10 May. Recommendation to farmers to use respiratory protection in dusty soil cultivation work.

12 May. Instructions concerning the replacement of filters in large air conditioning facilities.

12.2.5 Derived intervention levels applied in Finland

Control of the activity levels in foodstuffs and interdiction of foodstuffs that do not satisfy the criteria (the intervention levels) are mitigating actions that are equally relevant both as short-term and long-term actions. The part of the report from the project AKTU-280 concerned with the Finnish intervention levels is included here, in the chapter on short-term mitigating actions, although it is equally relevant to long-term mitigating actions.

During the first days of the Chernobyl fall-out it became evident that no large scale restrictions for use of foodstuffs were needed in Finland. During these first months after the accident no internationally agreed intervention levels for foods were available. Dose levels presented by the ICRP (ref. IC84) were used to determine the derived intervention levels. In this publication the projected dose equivalent in the first year was set for control of foodstuffs as:

1) Whole body, upper level 50 mSv and lower level 5 mSv.
2) Individual organ, upper level 500 mSv and lower level 50 mSv.

Taking 50 mSv as a reference level for thyroid, and taking one tenth of this for infant, an intervention level for I-131 in milk and drinking water was set to 2000 Bq/liter. The total intake of Cs-137 in a year was set to 200,000 Bq. This also took into account other radionuclides, especially Cs-134, to keep the dose below 5 mSv in the first year. From this total intake peak values for Cs-137 in milk, beef and pork were calculated as 1000 Bq/liter and 1000 Bq/kg respectively.

12.2.6 Short-term impact on food production

As the growing season had not yet commenced at the time of the accident, no direct contamination of unprotected leafy vegetables did take place. However, the demand for fresh domestic lettuce went down sharply as a consequence of television- and newspaper-reports on condemnation of large amounts of lettuce in Central Europe. The consumers obviously were not aware of the fact that fresh domestic early-season lettuce is grown in greenhouses, where the products are not exposed to fall-out. The price of domestic lettuce dropped in price, and did not return to normal price for quite some time, thus causing economic loss to the producers. It is estimated that some 50 - 100 producers were affected by this situation. By Mid-June 1986, as the free grown lettuce entered the market, the price was back on the normal level.

In general it seems that during the latter part of the summer 1986 the production and price level of fresh vegetables and fruits were not affected. However, the consumers were intensively interested in the possible contamination of their home-produced products, resulting in numerous requests for measurement of radioactive content and for other advice.

Radioactive iodine was detected in milk on the third day after the accident. In fact, milk was the first food affected by the negative publicity around the all-out situation. Because cows were not let out on pasture until 26 May, the farmers had to feed the animals on feed from the previous year, if available. Some farmers suffered from shortage of feed, and therefore had to buy fodder, with accordingly increased production costs.

The contamination of agricultural products in the whole of Finland was low. The concentrations of radioactive materials were well below the intervention levels. In most cases the Cs-137 levels were below one tenth of the intervention levels. Some products, like game meat, reindeer meat, freshwater fish and mushrooms, were contaminated to a higher degree. The situation regarding these products will be described in the chapter on long-term mitigating actions.

12.2.7 Reevaluation of recommendations and decisions by competent authorities

When the fallout situation was detected in Finland on 27-28 April, the bulk of the radioactive cloud was at a height of one to two kilometers Before the onset of rain on 29 April in central and southern Finland, only slightly increased radiation levels were reported. Thus, the authorities did not anticipate at first that there would be any need for countermeasures or recommendations affecting the general public.

However, after the rain showers had washed radioactive material from the cloud, resulting in comparatively high levels of iodine-131, cesium-137 and cesium-134 deposited on the ground, it became clear that food chains were going to be affected, and advice by the competent authorities would be needed fairly quickly.

The principles stated by the International Commission on Radiological Protection (ICRP) in its Publication 40 for planning intervention in the event of an accident are the following:

a) individual dose should be limited to levels below the thresholds for serious non-stochastic effects.

b) the risk from stochastic effects should be limited by introducing countermeasures which achieve a positive net benefit to the individuals involved.

c) the overall incidence of stochastic effects should be limited as far as reasonably practicable, by reducing the collective dose equivalent.

The condition a) was inherently satisfied for both external and internal exposure in the situation prevalent in Finland.

As for conditions b) and c) the ICRP further states that the decision to introduce any countermeasure should be based on a balance of the risks and disadvantages to the individuals affected. It is realized that the implementation of any countermeasure to reduce the exposure of the public carries with it some detriment to the people concerned, including social and economic disruption. The experience from handling the Chernobyl situation in Finland showed that in actual practice it is not possible to carry out the risk comparisons and cost-benefit analyses implied in ICRP 40 principles b) and c). Because of limited expert resources and the time scales involved, most of the recommendations given by STUK and other authorities had to be decided upon with haste, based on the expert´s impression of the situation. The experts were frequently confronted with novel radiation exposure pathways for which transfer factors and other essential parameters were not available, but had to be crudely estimated.

The way of applying the ICRP´s basic principles in the event of an accident is also in question. At the International Atomic Agency (IAA) conference on Radiation Protection, Sydney, Australia in 1988, several experts expressed the opinion that in order to apply the linear non-threshold concept in practice, a lower bound value for individual dose is needed. According to a report by the Swiss Nuclear Safety Inspectorate, the approach of the Swiss national emergency organisation in the Chernobyl situation was as follows:

- measures that would reduce the dose to members of the critical group by 0.1 mSv only, were considered useless,
- measures that might bring a dose reduction between 0.1 mSv and 0.5 mSv were discussed and led in some cases to recommendations,
- protective measures which might possibly reduce person doses by 0.5 mSv should be adopted, with a view of limiting the effective doses to 5 mSv.

The result of the Swiss approach was that there were very few official steps actually taken in Switzerland.

The experience in Finland proved that a restrictive and absolutely consequent line is equally hard to carry out in actual practice. The demands for information on the part of the public was enormous. Many issues were raised in public that were allegedly neglected in the official recommendations. Such issues were reconsidered by STUK and other authorities and, where appropriate, taken into account in formulating subsequent recommendations. In some cases, where countermeasures with negative net benefits were advocated, it was considered necessary to intervene in the spirit of the basic ICRP principles. Therefore, also explicit non-action advice was included among the recommendations.

13. LONG-TERM MITIGATING ACTIONS

The long-term (or late and intermediate phase) mitigating actions usually considered in ACA models are the following:

- relocation
- decontamination
- interdiction of areas
- ban of nutrients

Long-term mitigating actions are aimed at prevention or mitigation of possible stochastic health effects. The three first types of mitigating actions are limited to the relative vicinity of the release site, but interdiction of foodstuffs may be warranted at very far distances, as shown after the Chernobyl accident. Post-Chernobyl experience, moreover, is also that the nutrition exposure pathways must be evaluated in much finer detail than what has been customary in ACA models, where all different foodstuffs often were divided into two coarse groups: milk and everything else.

Six of the AKTU projects have been concerned with various aspects of long-term mitigating actions. Projects AKTU-272 and AKTU-280 have been concerned with both short- and long-term actions, and the parts of these projects concerned with short-term mitigating actions are described in the appropriate chapter. AKTU-272 is an analysis of mitigating actions using the computer program ARANO as an analytical tool. The other projects are concerned with aspects of the actual situation in the Nordic countries after the Chernobyl accident.

13.1 ARANO-calculations of long-term mitigating actions

Project *AKTU-272* has been concerned with investigating the impact upon the consequences of alternate mitigating actions. Both short-term and long-term mitigating actions have been examined in the project, and are reported in the appropriate parts of the present report. Health and economic consequences, mitigating actions and action levels are linked. The cost of different types of mitigating actions may be very different, and so may be the efficiency of different mitigating actions, measured in saved lives. And it is not necessarily the most costly mitigating action that is the most efficient.

The analyses of these problems have been performed using the Finnish accident consequence assessment code ARANO. The importance of various mitigating action parameters have been examined for a range of release characteristics, and various combinations of the parameters. The costs of mitigating actions, however, have only been calculated for long-term mitigating actions, and cost-effectiveness analyses have been performed separately based either on ground shine or ingestion doses. The results of the latter analysis are considered in the present report.

13.1.1 Types of long-term mitigating actions

The types of long-term mitigating actions considered are:

Relocation

Relocation is not the same type of mitigating action as evacuation. It is a long-term action. Relocation may be initiated a few days after a release and it may continue for months or years. When doses are below the level where one would perform evacuation, but above the level where decontamination is warranted, relocation would be performed. The risks due to relocation can be smaller than in connection with evacuation, because of the longer time period available for planning. On the other hand the social and economic losses are significantly higher.

Admission to an area from which the population is either evacuated or relocated must be prevented. This control of access is aimed at reducing spreading of contaminated materials. Interdiction of contaminated land is simple to carry out, but on the other hand it may result in high economical losses.

Decontamination

Decontamination means cleanup, removal or redistribution of radionuclides. As examples of decontamination methods one can mention removal of contaminated surfaces, deep plowing and sand blasting of surfaces. Decontamination of persons may be warranted when they are moved from a contaminated to a clean area. The decontamination factor is defined in project *AKTU-272* to mean generally the ratio of exposure from untreated contaminated surface to the exposure after decontamination. The risks from decontamination can be effectively reduced by proper protective equipment. Costs may vary very much, depending on the location and the methods applied.

The decontamination factor should be determined separately for different exposure pathways (e.g. ground shine and ingestion doses). In project AKTU-272, however, decontamination by a factor of 3 has been assumed to apply only to the ground shine exposure.

Ban of nutrients

The practical control of foodstuffs should be based on measured (in project AKTU-272 they are of course calculated) concentrations of contaminated food. Intervention criteria of different foodstuffs are determined based upon the annual consumption according to the diet and on the ingestion dose reference level.

13.1.2 Criteria for long-term mitigating actions

The calculations of the impact upon the consequences of relocation/ decontamination, and of interdiction of foodstuffs are performed independently. Both types of calculations are performed for alternative values for the criteria as well as for the case where no action is imposed. Table 13.1 contains the default predetermined action levels used at present in the Nordic countries. The criteria actually used in the calculations are summarized at the end of this subchapter. The derived intervention levels are derived from the recommendations of ICRP and IAEA. In Sweden and Finland the following food control criteria are also formulated: As lower reference level 5 mSv in the first year, and then 1 mSv per year; and as upper reference level 50 mSv.

Countermeasure	Derived intervention levels (Bq/kg or Bq/liter) or (dosecriterion)/(distance)			
	Denmark	Finland	Norway	Sweden*
I-131 in				
- milk	-	2000	1000	2000
- meat	-	-	1000	300
- green veget.	-	-	1000	300
Cs-134 + Cs-137 in				
- milk	370	1000	370	300
- meat	600	1000	600	300
- green veget.	-	1000	600	300
Relocation	-/-	-/20 km	-/-	-/20 km

* The Swedish criteria are based upon Cs-137 only.

Table 13.1 Intervention levels.

The actual ARANO calculations have been performed with the following criteria:

For relocation the alternative criteria used are: 0.03, 0.1, 0,3 and 1.0 Sv of ground shine dose over 30 years (first week excluded). When the collective dose is calculated, the individual dose is truncated to the level of the dose criterion. Decontamination is only taken into account then economic consequences due to ground shine are calculated (chapter 13.1.4 of the present report).

The whole area around the release site is divided into sector segments, 12 sectors and 18 distance intervals, and each sector segment is considered separately in these calculations, and the population doses from all sector segments are summed to give the collective dose.

For the nutrition pathways the alternative criteria used are:

0.001 and 0.01 Sv the first year for any single nutrition pathway; or 0.007 and 0.02 Sv over 30 years for any single nutrition pathway.

(The criteria, as stated in the report from project AKTU-272 are five times higher than the above values. However, the manner in which ARANO tests compliance with the nutrition pathway criterion, interdiction is effectuated if a nutrition pathway exceeds one fifth of this criterion.)

13.1.3 About the calculations and the results

The calculations of the nutrition and the non-nutrition pathways are performed completely independent of each other. This implies also that the mitigating actions (food interdiction and decontamination/relocation) are independent of each other. It is assumed in the calculations that decontamination of an agricultural area (from ground shine exposure considerations) is performed when the criterion is exceeded by occupancy one third of the time; corresponding to a contamination level three times higher than in the other types of areas. It may be discussed whether the different types of mitigating actions can be regarded as being independent of each other in a real situation. If relocation of a specific area has been carried out, it may be that agricultural use will also be terminated, both for practical and psychological reasons. The coupling the other way is, however, quite weak, as shown by actual experience after Chernobyl. Even though interdiction of certain food products has been imposed in certain areas, there have been no restrictions on use for living and industrial purposes.

When it is said that the calculations of the nutrition and non-nutrition pathways are independent, this also means that the results of the calculations concerning the non-nutrition pathways do not include doses via the nutrition pathways and v.v.

13.1.3.1 Relocation

Relocation is calculated in ARANO according to the following procedure: At the checkpoint moment (which is at 7 days), it is checked whether the dose during the following 30 years will exceed the specified dose limit. If it is exceeded, the program calculates the time point at which return to the area will be permissible (the time when the dose during the following 30 years is equal to the dose limit). When this procedure is followed, the total dose to each person in the population groups that undergo relocation will be equal to the dose limit.

In ARANO short-term doses and long-term doses, divided at the time point 7 days, are calculated separately. This is also true in the events when an evacuation continues directly as a relocation. In the report from AKTU-272 the calculated collective doses from the first week as well as the long-term collective doses (the first week excluded) are presented, and these doses are compared, as shown in Table 13.2.

Release category	A_1	A_2	A_3	B_1	B_2	C_1	C_2
<u>Short-term (first week)</u>							
0 - 100 km, normal sheltering, evacuation at 24 h in 0 - 20 km	10,500	9,910	7,420	1,200	880	350	230
0 - 100 km, active sheltering, evacuation at 8 h in 0 - 20 km	8,300	8,080	5,690	920	630	190	120
100 - 300 km	5,730	5,550	7,140	660	800	150	170
<u>Long-term (first week excluded)</u>							
Relocation criterion							
0.03 Sv / 30 years	13,100	13,100	14,400	2,590	2,820	320	315
0.1 Sv / 30 years	20,800	20,800	23,300	2,980	3,060	340	320
0.3 Sv / 30 years	25,900	25,900	28,200	3,220	3,150	350	320
No limit	35,000	35,000	31,900	3,510	3,190	350	320

Table 13.2 Expectation values of collective doses (manSv) for a selection of sheltering and relocation strategies. (Short-term doses include contributions from cloud shine, inhalation and ground shine. Long-term doses include only the contribution from ground shine.)

Here it is proper to remind the reader that release categories A, B and C are the large, intermediate and small release, and that subscript 1 means ground release and 2 means 100 meters release height (except for the largest release, where 3 means 100 meters release height, while 2 means ground release and long delay before release takes place). In all categories the release of noble gases is 100%.

It is seen from the table that the short-term dose (which is the sum of the doses in the two distance intervals; where there are two alternatives for the inner interval) and the long-term dose is of the same order of magnitude, if the strictest long-term countermeasure is assumed.

Comparison of the different long-term doses calculated shows that, at least regarding expectation values, there is considerable population dose to be saved when the release is the largest one. For the intermediate and smallest releases, however, the savings are quite modest and quite insignificant, respectively. The reason is simply that for the two smaller releases the criterion is exceeded in only small population groups, or none at all.

The full report from project AKTU-272 includes CCDF's corresponding to the results in above table, but the main conclusions can already be made based on the expectation values presented in the table. It is, of course necessary to check the validity of these conclusions, using the CCDF curves, and they support the conclusions above. They show, however, by the irregular shape of the "high-consequence" part of the curves, that this part is dominated by quite few weather sequences, and that one should be careful of drawing more detailed conclusions than those that can be drawn from the table of expectation values.

From the table of expectation values it is also observed for the largest of the releases that each strengthening of the criterion brings a considerable reduction in population dose. It is accordingly possible or probable that additional lowering of the criterion would give further significant lowering of the population dose. However, each step down on the criterion will be increasingly expensive, since it will involve an additional area that will increase even faster than the criterion decreases. A balance between cost and savings in population dose is involved, and this aspect is examined later in this chapter.

13.1.3.2 Interdiction of food

Interdiction of nutrients is calculated using two different sets of criteria; one applied upon the first year dose (strictly the committed effective dose from the first year's intake), and one upon the summed dose over 30 years (the committed effective dose from intakes during 30 years). The alternative criteria are:

First years dose:

- 5 or 50 mSv

30 years dose:

- 35 or 100 mSv

The nutrition interdiction criteria are applied in ARANO in such a manner that each of the five nutrition pathways (milk, beef, green vegetables, grain, root vegetables) is evaluated separately, possibly resulting in different sizes of the interdiction areas for each of the pathways.

<u>First year criteria</u>

Table 13.3 shows the size of interdiction areas for three of the release categories and all five classes of foodstuffs, for the first year. A, B, and C means the largest, intermediate and smallest release respectively, and all are for start time for release at 1 hour, and for elevated release.

Criterion and type of foodstuff	Expectation values of interdiction areas (km^2) in different release categories A_3	B_2	C_2
1 mSv/a			
Milk	16,100	7,580	3,380
Meat	13,800	4,970	230
Vegetables	2,990	1,180	40
Grain	4,420	2,700	920
Roots	160	5	0.06
10 mSv/a			
Milk	7,600	3,370	230
Meat	4,980	230	10
Vegetables	1,290	40	1
Grain	2,750	920	25
Roots	5	0.06	0

Table 13.3 Expectation values of interdiction areas (km^2), when limitation criteria for the first year´s intake are applied.

As expected, the milk pathway has the largest expectation values for size of interdiction areas. However, summer/winter variations are included in these calculations, and there might be exceptional situations (release during winter) where the areas of milk interdiction may not be the largest ones.

The following figures and table are for the three magnitudes of releases, all for start time for release being 1 hour, and for elevated release. The doses shown are only the doses via the nutrition pathways. Doses from non-nutrition pathways are excluded from this consideration, but included in the separate consideration of long-term ground shine dose.

The "saddle"-shape of these curves is caused by the fact that in some cases deposition takes place in the growth season, and in other cases during the dormant season, with a corresponding marked difference in the dose during the first year.

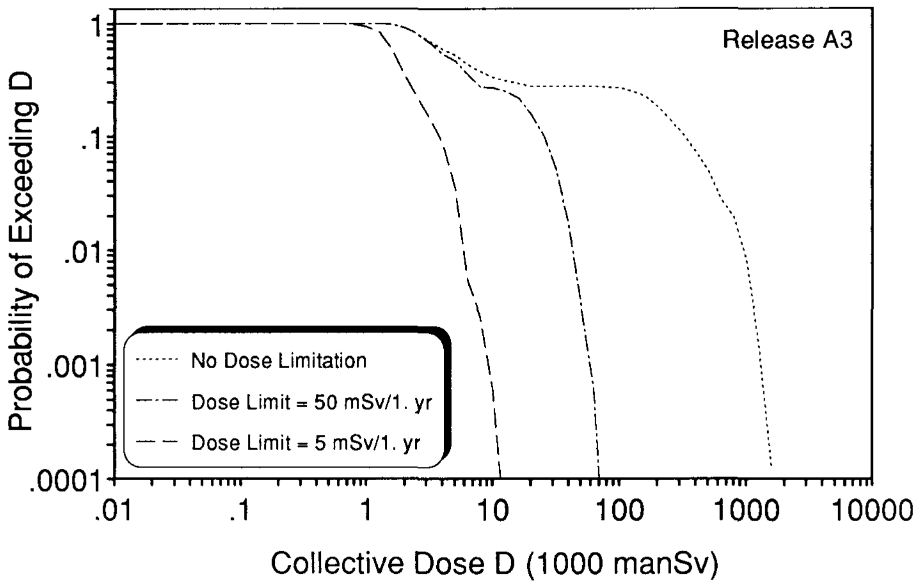


Figure 13.1 The CCDFs of the collective doses from nutrition in the first year for release category A3 (large release, H=100 m) applying different dose criteria (5 or 50 mSv in the first year) for food interdiction.

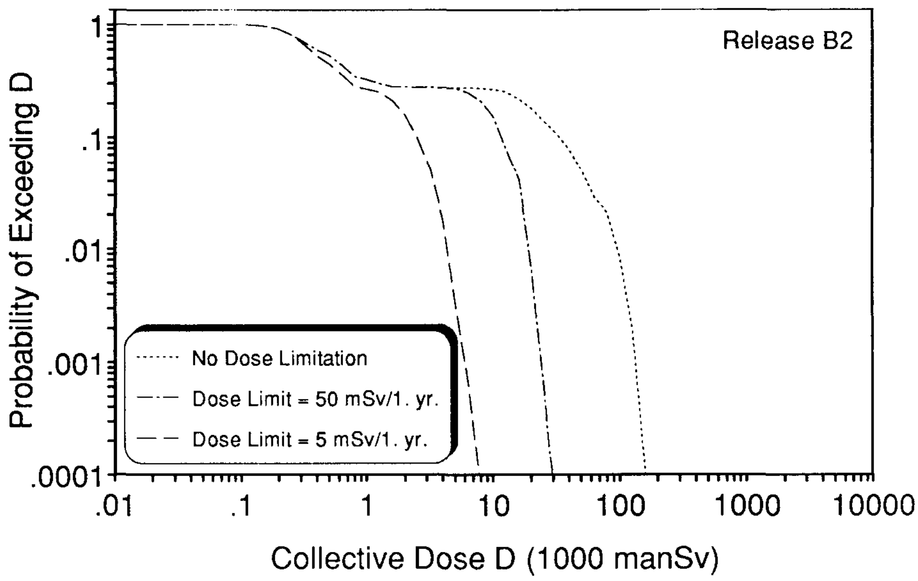


Figure 13.2 The CCDFs of the collective doses from nutrition in the first year for release category B2 (medium release, H=100 m) applying different dose criteria (5 or 50 mSv in the first year) for food interdiction.

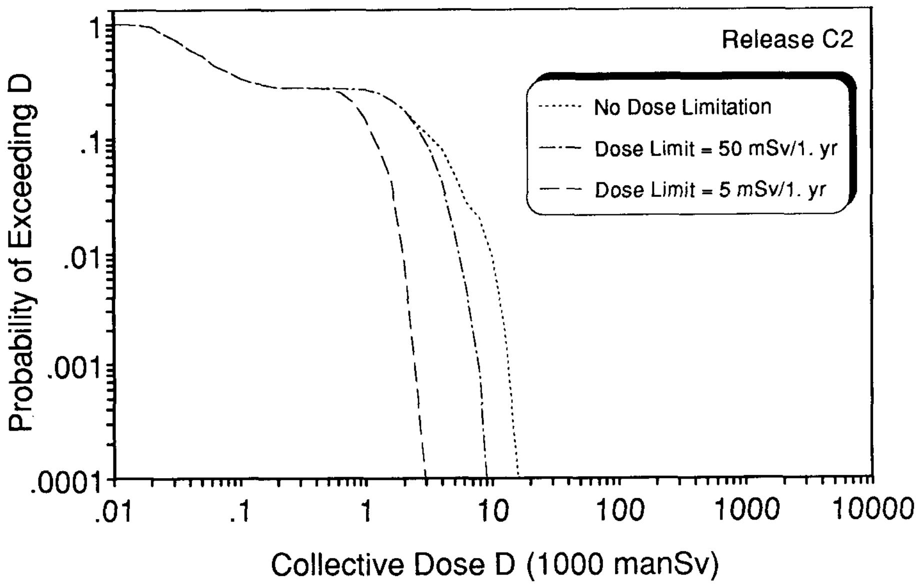


Figure 13.3 The CCDFs of the collective doses from nutrition in the first year for release category C2 (small release, H=100 m) applying different dose criteria (5 or 50 mSv in the first year) for food interdiction.

Conclusions are more easily drawn from the table of expectation values, shown in the following:

Release category	Dose criterion (Sv/1 year) 0.005	0.05	No dose limit
A_3	1,970	8,800	93,800
B_2	880	3,300	9,380
C_2	330	740	940

Table 13.4 Expectation values of the ingestion collective doses (manSv) due to release categories A_3, B_2 and C_2.

It is seen that for the large and intermediate releases interdiction of foodstuffs leads to significant reduction of population dose. For the smallest release the reduction is quite small, due to the fact that the interdiction areas, and the fraction of the total population dose that is received in these areas, will be quite small. For none of the releases is it possible to judge what would be the optimal criterion, although it seems that the benefit from going from the higher to the lower value for the criterion is relatively modest. A proper evaluation must also take the cost of the mitigating actions into consideration. This is done at the end of this subchapter.

The doses without interdiction of foodstuffs are, since all other conditions are the same, proportional to the amount of radioactive materials (noble gases excluded) released.

30 year criteria

These calculations have been performed for the same three magnitudes of releases, with release time at 1 hour, and for elevated release. The doses in the following are, like for the 1-year considerations above, only the doses via the nutrition pathways. Doses from non-nutrition pathways are excluded from this consideration, but included in the separate consideration of long-term ground shine dose.

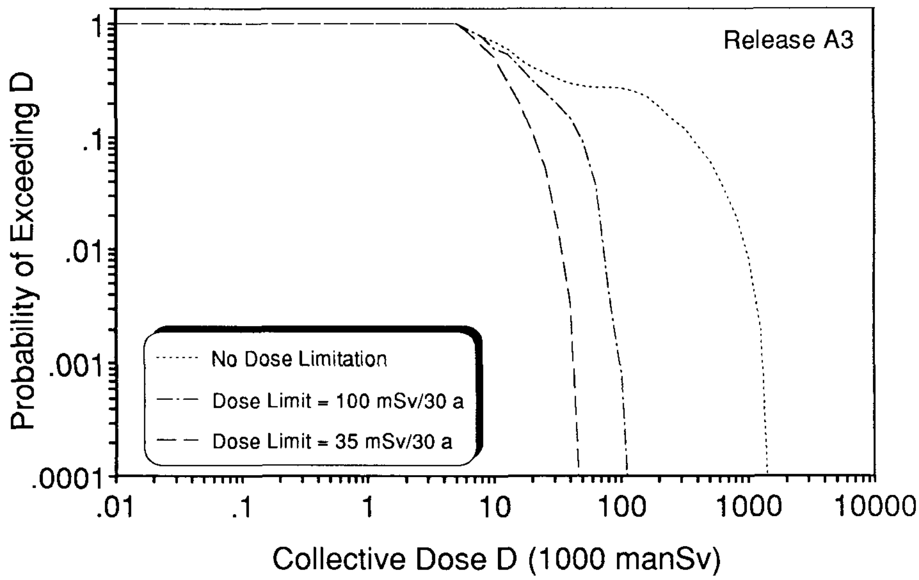


Figure 13.4 The CCDFs of the collective doses from nutrition in 30 years for release category A_3 (large release, H=100 m) applying different dose criteria (0.035 or 0.1 Sv in 30 years) for food interdiction.

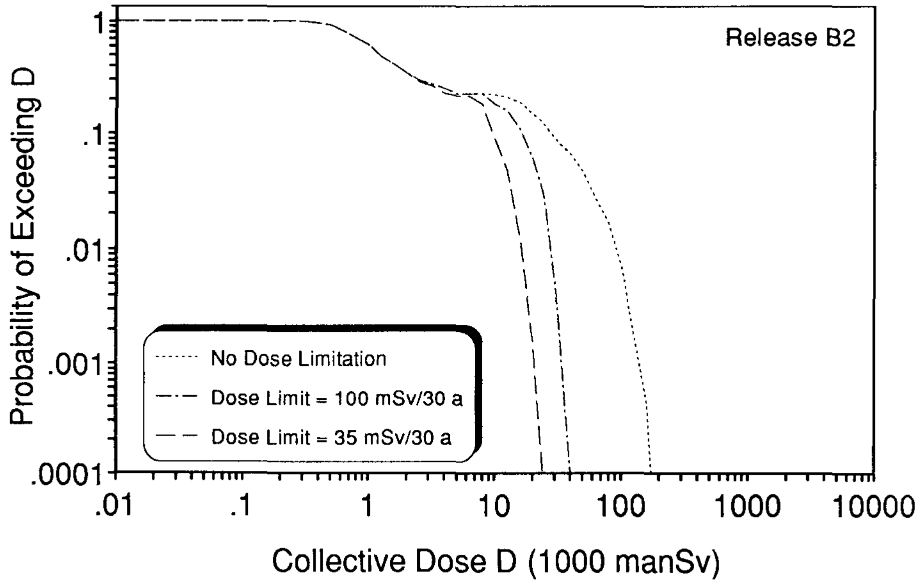


Figure 13.5 The CCDFs of the collective doses from nutrition in 30 years for release category B_2 (medium release, H=100 m) applying different dose criteria (0.035 or 0.1 Sv in 30 years) for food interdiction.

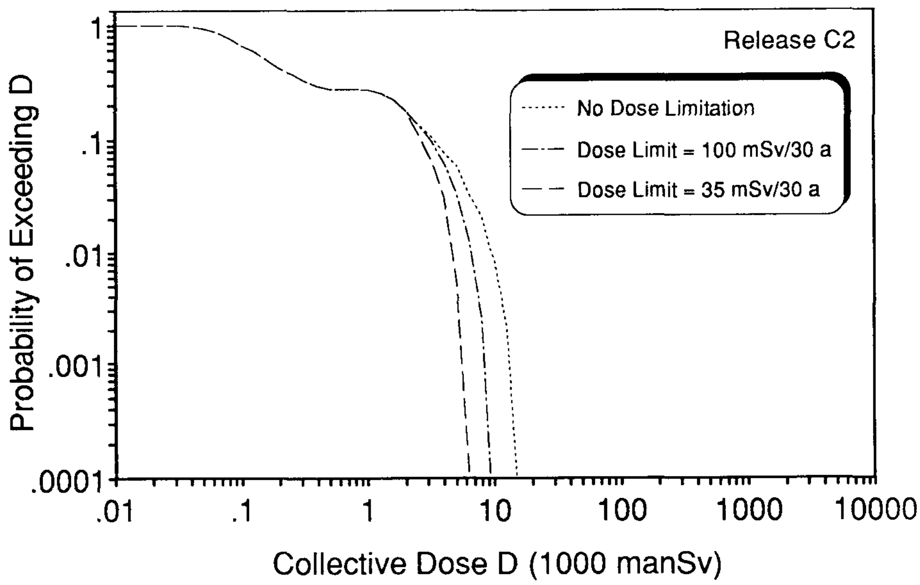


Figure 13.6 The CCDFs of the collective doses from nutrition in 30 years for release category C_2 (small release, H=100 m) applying different dose criteria (0.035 or 0.1 Sv in 30 years) for food interdiction.

The "saddle"-shape of the curves due to the season variations can still be observed, although less marked than on the first-year curves.

Release category	Dose criterion (Sv/30 years) 0.035	0.1	No dose limit
A_3	11,100	19,000	103,000
B_2	3,430	5,400	10,300
C_2	780	900	1,030

Table 13.5 Expectation values of the ingestion collective doses (manSv) due to release categories A_3, B_2 and C_2.

The conclusions that can be drawn from the above are similar to the ones that could be drawn from the calculations of first-year-doses.

Another interesting point can be seen by comparing parts of the tables of expectation values for 1-year respectively 30-year calculations (Tables 13.4 and 13.5). Comparing the cases where no interdiction is imposed, it is seen that the long-term ingestion dose is mainly caused by the first year dose. Similar comparisons for the cases when interdiction is imposed is not possible, since there is no actual relationship between the 1-year and 30-year criterions.

13.1.4 Cost of long-term mitigating actions

A proper cost-effectiveness analysis has actually been performed separately for relocation and for interdiction of foodstuffs. The results of the analysis for foodstuffs are considered in more detail here, in the latter part of this subchapter.

The basis for the relocation/decontamination calculations is as follows: When the criterion is exceeded, decontamination is assumed to be performed, and the decontamination is assumed to give a general dose reduction of a factor 3. The dose calculations are then performed, and if they indicate that the criterion will be exceeded in course of the coming 30 years, relocation is performed (for a more detailed explanation see chapter 13.1.3.1.

Cost of decontamination and relocation

The bases for these calculations are given in the following table:

Cost group	Cost per person inhabiting the affected area (land decontamination, relocation and housing) or per employee (the remaining items)
Land decontamination	330
Farmland decontamination	10,200
Relocation	5,000
Housing	30,700
Agriculture and forestry	61,400
Manufactoring and construction	71,900
Service	73,100

Table 13.6 The unit cost of investments and countermeasures in 1988. (Values given in \$, assuming that 1 \$ = 4.2 FIM). All values are per person.

The results of the calculations are given in the following figures. In the first of these figures the different components of the cost are shown. The case calculated is the largest release with normal release time and non-elevated release, and the criterion used was 0.1 Sv per 30 years.

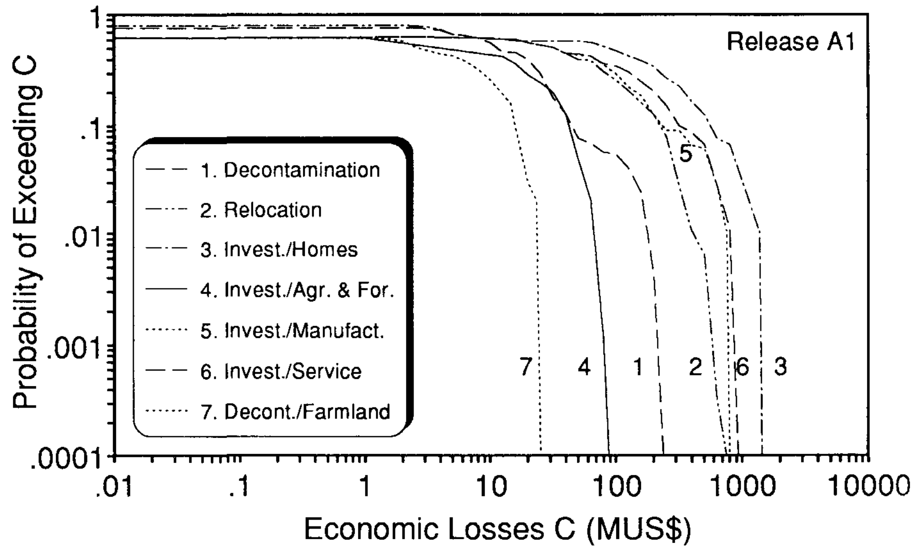


Figure 13.7 The CCDFs of the economic losses (million US\$) due to release category A_1 (large release, H=20 m). The components of the total loss are presented. The relocation criterion applied is 0.1 Sv in 30 years (groundshine dose).

The following three figures show total cost numbers for the largest, intermediate and smallest release magnitudes, normal release time and non-elevated release. The effect of varying the criterion is shown.

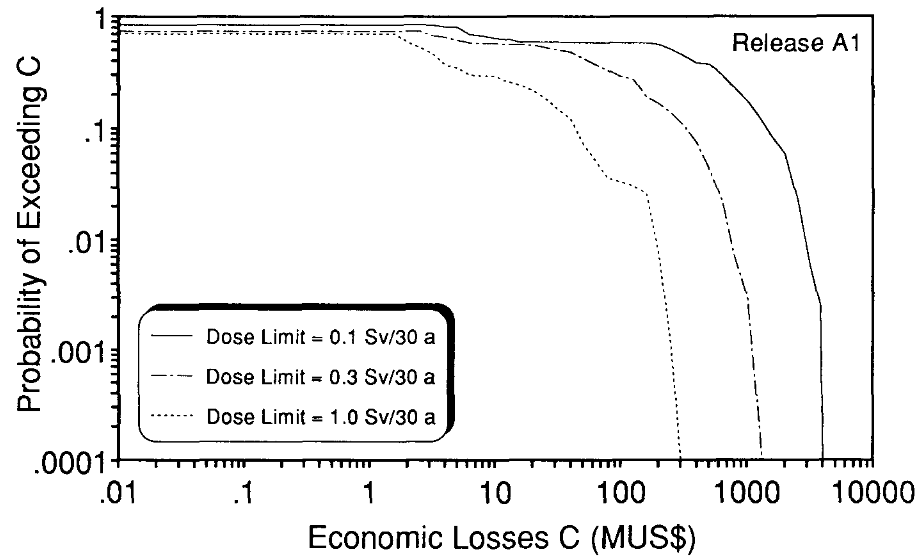


Figure 13.8 The CCDFs of the economic losses (million US$) due to release category A_1 (large release, H=20 m), when alternative dose criteria for relocation are employed. Decontamination by a factor of three is included.

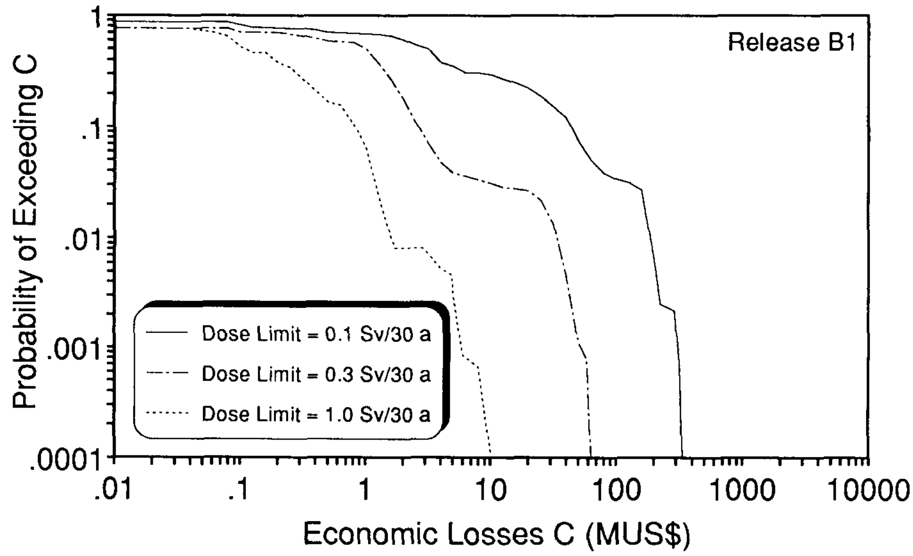


Figure 13.9 The CCDFs of the economic losses (million US$) due to release category B_1 (medium release, H=20 m), when alternative dose criteria for relocation are employed. Decontamination by a factor of three is included.

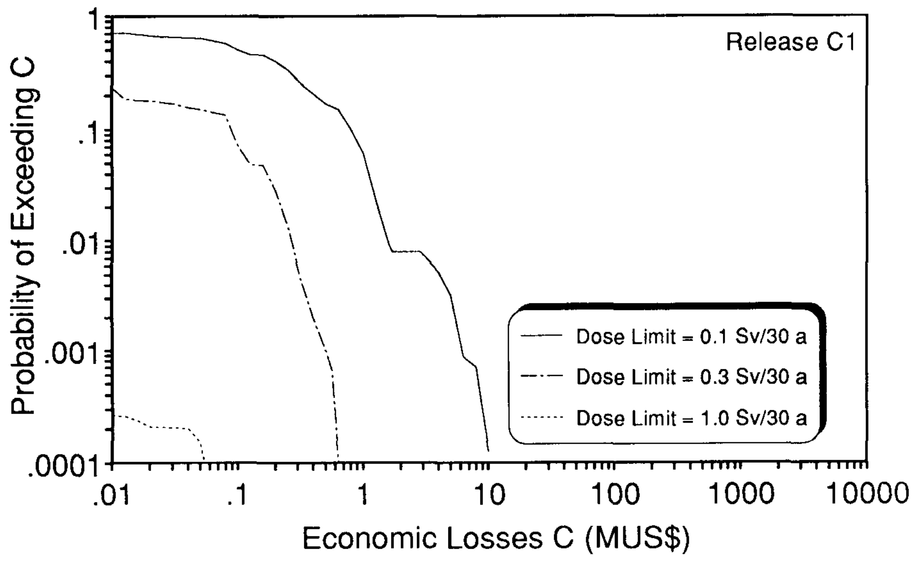


Figure 13.10 The CCDFs of the economic losses (million US$) due to release category C_1 (small release, H=20 m), when alternative dose criteria for relocation are employed. Decontamination by a factor of three is included.

The results show that loss of investments dominate the cost, and housing is the most important component, but that can be seen even from the basic data upon which the calculations are based. The calculations demonstrate that there is rapid increase in cost with introduction of more severe criteria. The optimal criteria can not be determined from these calculations.

Cost of food interdiction

The bases for these calculations are given in the following table:

Product	Cost
Milk	1.4
Beef	16.9
Green vegetables	5.2
Grain	2.0
Root vegetables	1.5

Table 13.7 Cost of interdicted agricultural products ($/kg). (This cost is the sum of production cost and price to the consumer; which is equivalent to the cost from destruction of contaminated food plus replacement by clean food.).

The calculations for the nutrition pathways are performed for the three release magnitudes, for normal release time and for elevated release. The calculations have been split, so that deposition in the growing season and the dormant season can be evaluated separately. The results are presented as expectation values in the following tables:

	Dormant season			Growing season		
Dose criterion (mSv/a)	5	50	None	5	50	None
Collective dose (manSv)	730	2,300	4,500	3,070	12,500	360,000
Dose savings (manSv)	1,610	2,160		9,430	347,500	
Cost of interdict. (mill $)	440	85	0	1,700	1,200	0
Cost increment (mill. $)	355	85		500	1,200	
Cost per unit collective dose saved (mill. $/manSv)	0.2	0.04		0.05	0.003	

Table 13.8 Expectation values of the collective nutrition doses for different interdiction criteria, and the corresponding cost of discharded foodstuffs. These are ingestion doses from the first year's intake of contaminated foodstuffs. Largest release category.

	Dormant season			Growing season		
Dose criterion (mSv/a)	5	50	None	5	50	None
Collective dose (manSv)	230	400	460	1,280	7,760	37,000
Dose savings (manSv)	164	64		6,480	29,000	
Cost of interdict. (mill $)	85	3.3	0	1,200	460	0
Cost increment (mill. $)	81.7	3.3		750	460	
Cost per unit collective dose saved (mill. $/manSv)	0.5	0.05		0.1	0.02	

Table 13.9 Expectation values of the collective nutrition doses for different interdiction criteria, and the corresponding cost of discharded foodstuffs. These are ingestion doses from the first year's intake of contaminated foodstuffs. Intermediate release category.

	Dormant season			Growing season		
Dose criterion (mSv/a)	5	50	None	5	50	None
Collective dose (manSv)	39	45	46	775	2,460	3,700
Dose savings (manSv)	6	1		1,685	1,200	
Cost of interdict. (mill $)	3.3	0.03	0	460	29	0
Cost increment (mill. $)	3.27	0.03		431	29	
Cost per unit collective dose saved (mill. $/manSv)	0.5	0.03		0.3	0.02	

Table 13.10 Expectation values of the collective nutrition doses for different interdiction criteria, and the corresponding cost of discharded foodstuffs. These are ingestion doses from the first year's intake of contaminated foodstuffs. Smallest release category.

The Nordic radiation protection authorities have recommended as guidance that if a mitigating action can save one manSv at a cost smaller than 20,000 $, then the mitigating action ought to be carried out. The results contained in the preceding tables then indicate that if deposition takes place during the dormant season, a criterion of 50 mSv is too severe to be cost-effective. If the deposition takes place during the growing season, the cost-effective criterion for the largest release is somewhere between 5 and 50 mSv, and for the other two release magnitudes the cost-effective criterion is quite close to 50 mSv.

13.1.5 Summary of conclusions concerning short- and long-term mitigating actions

In project *AKTU-272* the effectiveness of environmental countermeasures carried out to mitigate possible detriments as a consequence of a radioactive release from assumed reactor accidents are investigated. The purpose of the protective measures is to prevent prompt health effects and to reduce late health effects as low as reasonably achievable by minimizing the collective dose. However, individual dose limits shall not be exceeded. There are international recommendations of intervention dose levels for introduction of different countermeasures. Often those recommendations and national protection practices differ from each other due to local conditions or other factors. At the beginning a review on the different countermeasures was performed and international and Nordic recommendations of action levels were compared. In this concept an effort was made to use a range of dose criteria to indicate variations on consequences.

Effectiveness of different countermeasures is studied in the case of various hypothetical source terms assumed at the Loviisa nuclear power plant site. The core inventory of net output capacity of 1000 MW was assumed, the release fraction of noble gases was 100% in all releases and three alternative release category fractions of 10, 1 and 0.1% for iodine, cesium and tellurium and with a factor of 10 less for other substances were employed. The site specific environmental data as well as annual weather statistics were used. All but one source term started after a delay of one hour at two alternative altitudes. In the case of a longer delay time there exists possibilities to execute protective measures so that exposures are fully avoided. This kind of possibility was omitted and evacuation for example is calculated to occur after the plume has reached the area involved.

The computer code ARANO was employed to obtain radiation doses, short- and long-term health effects, interdicted land and farmland areas, and further, economic losses as a consequence of ground contamination and denial of areas. The code is suitable for probabilistic consequence assessment and the CCDFs and further expectation values of consequences can be obtained. Cost-effectiveness of alternative mitigation measures is assessed by combining costs with alternative countermeasures.

Short-term exposure

Calculations indicate that the effectiveness of countermeasures is basically source term dependent. However, the conclusions that can be drawn from the calculations are surprisingly similar for the different release categories examined. The noble gases contribute relatively more to the short-term exposures than the long-term exposures. Also, the significance of the cloud shine increases when the release fractions of other nuclides decreases relative to the noble gas release fraction, which has been 100% for all the release categories examined. Sheltering seems to reduce the number of early health effects approximately with an order of magnitude in all release categories considered. When sheltering is improved, the time of evacuation becomes correspondingly less important. Evacuation also becomes a less important mitigating action when release categories with smaller release fractions (except for noble gases, have release fraction 100% has been assumed for all release categories) are considered.

The following conclusions are based upon the results of the calculations of early illness cases and release category A_1. Very similar conclusions can be drawn from the results for the two other A-categories, and also from the results of calculations of early fatalities:

- Sheltering is very important, and there is a large difference between active sheltering and normal sheltering.

- For release A_1 it seems that the evacuation area should have a radius of between 10 and 20 km. Decrease of the evacuation area to 10 km results in a significant increase in consequences, while increase beyond 20 km or decrease beyond 10 km gives little additional reduction or increase, respectively, in consequences.

- When evacuation is performed earlier in the interval 10 to 20 km, there is a significant impact upon the consequences.

Iodine tablets may be very efficient in reducing thyriod doses. Comparison with the results of the calculations of other early health effects, however, shows that intake of stable iodine is not an effective mitigating action for releases of the type investigated here, since it is the bone marrow and not the thyroid which is the critical organ. The early fatalities resulting from bone marrow doses are at least of the same magnitude as fatal thyroid health effects, even for the most efficient evacuation and sheltering strategies. From the results it is not possible to tell whether these bone marrow and thyroid health effects affect exactly the same persons. If this is not so, there might still be reason for stable iodine intake when the best sheltering and evacuation strategies have been employed.

Long-term exposure

The short-term exposures are included when the long-term effects are assessed. In the case of low release fractions (of all nuclides except noble gases) the contribution of the short-term exposures to long-term doses may be important due to the 100% release of noble gases. In this case relocation has little effect on long-term exposure. For the larger releases contaminated foodstuffs may give a significant contribution to the collective dose. Furthermore restrictions on food consumption may dominate the economic consequences, relative to other losses of investments, particularly if low intervention levels are set, and the release has taken place during in summer conditions.

Losses of contaminated agricultural production depend strongly on the season when the accident takes place. If consumption of smaller amounts of contaminated food is approved, and a distribution pattern is chosen to assure that the contaminated foodstuffs are evenly consumed within a large population group, in order to fullfill the individual dose restrictions, abandonment of contaminated food is avoided in the case of the two smallest release categories considered. For the largest releases (with the release fraction of 10% of iodine, cesium and tellurium) a decision making process utilizing a cost-effectiveness technique will clearly be needed.

- For the largest type of release examined in this project there are considerable collective dose savings to be gained by using a more strict relocation criterion. For the smaller releases the savings are insignificant.

- For the largest and intermediate releases interdiction of foodstuffs leads to considerable decreases in collective dose. For the smallest release the reduction is quite small.

Figure 13.11 illustrates the cost-effectiveness ratios obtained. Effectiveness of food control decreases when the release magnitude decreases. The best cost-effectiveness ratio is attained in the case of releases occurring during the growing and pasturing seasons. The Nordic radiation safety authorities have recommended a target value of 20,000 USD per saved manSv as a reasonable aim. This target level is roughly achieved for all release categories if the limit value of 50 mSv for the first year's individual ingestion dose is applied. For a lower intervention level, 5 mSv in the first year, the cost-effectiveness ratio clearly exceeds the target level.

It is illustrative to see how much the different exposures contribute to the total dose, when short-term exposure from cloud shine, ground shine and inhalation are also taken into account. Figure 13.12 illustrates the relative contributions of short-term exposures, long-term ground shine and nutrition dose from the first year's intake to the total expected population dose. In the short-term calculations local sheltering, followed by evacuation, was taken into account. The long-term contributions presented are based on either the unlimited collective dose, or a relocation criterion of 0.1 Sv in 30 years combined with two alternative food interdiction criteria.

If no long-term countermeasures are applied, the dose from contaminated food is clearly the largest component. If the long-term ground shine is reduced to the level of the background radiation (0.1 Sv in 30 years), and furthermore the individual nutrition dose is limited to 50 mSv, the ground shine dose will dominate for the largest and the medium release categories. For the smallest release category the three contributions will be approximately equal when the ground shine dose is reduced to the background level and the most severe nutrition dose limit is applied.

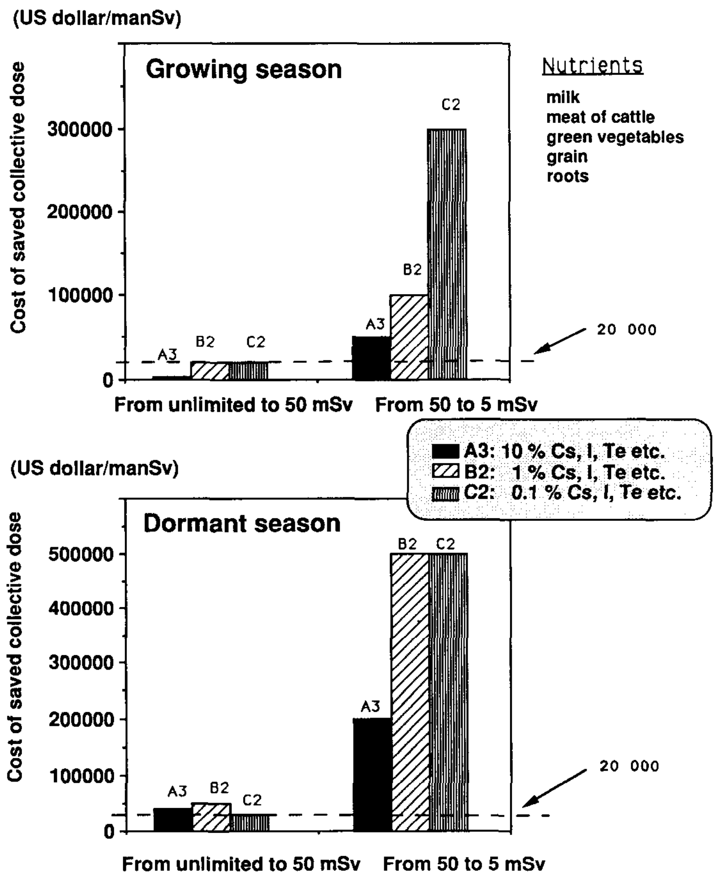


Figure 13.11 Cost-effectiveness ratios of food control from first year's intake.

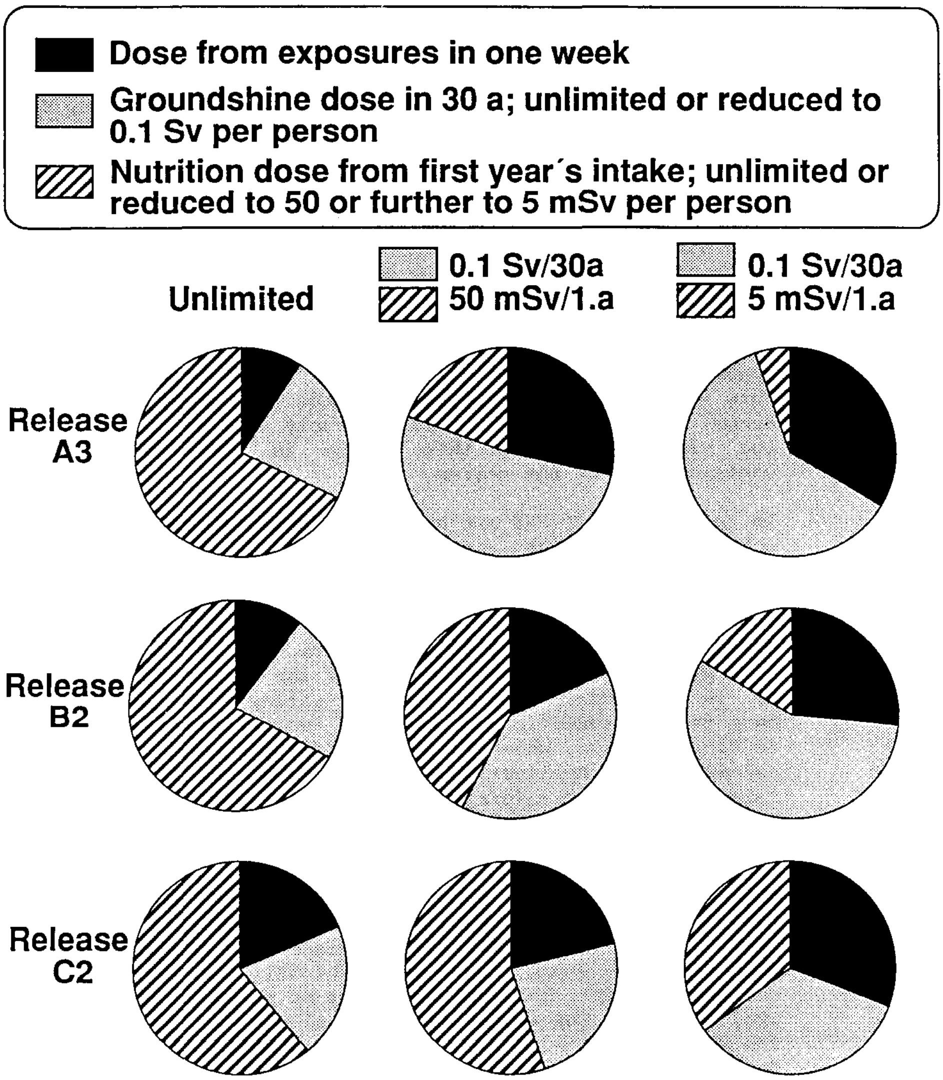


Figure 13.12 Relative contributions to the total long-term collective dose for unlimited long-term exposure, as well as for different relocation and food control limits.

13.2 Long-term mitigating actions in Norway after Chernobyl, project AKTU-275

The fallout from the Chernobyl accident swept central parts of Norway and Sweden, where especially semi-natural mountain areas, but also some agricultural land areas were heavily contaminated.

Only 3% of Norway is farmland. The rest is to a large degree mountain areas. But this does not mean that the mountain areas are not used for agricultural purposes. The sheep spend all the summer up in the mountain areas, going and grazing where they like. The reindeer, more or less wild, spend most of their time in these areas, summer and winter. And some of the cattle are also brought up to the chalets over summer, although on a much smaller scale than in earlier times. Many farmers now prefer to bring the rich mountain grass down to the valley instead of bringing the cattle up. Unfortunately the soil in the mountain areas do not bind cesium the way clay-rich soils in the valleys do, and uptake in vegetation, and ultimately animals, will be correspondingly high. The reindeer have additional problems. Lichen is an important part of their diet, especially in winter; and lichen has a tendency to cover the ground almost completely where it grows, and therefore collects all deposited activity. It also lives for years, as opposed to grass. Goats milk is also important in this connection. This milk is mostly used for production of goat cheese; a reddish-brown, sweet cheese, which is extremely popular in Norway. The part of the milk from which this cheese is produced unfortunately also contains most of the cesium.

In project *AKTU-275* one has investigated several types of mitigating actions. The simplest and most obvious is to put the animals on "clean" fodder before slaughtering. In many cases this is achieved simply by allowing the sheep and cattle to graze in the valleys for two to eight weeks when they are brought down in fall. In addition the animals are given concentrate containing 5% bentonite. The bentonite, a clay mineral, prevents uptake of the radioactive cesium that might be in the fodder. This type of mitigating action will give much lower cost than interdiction of the meat. Another cesium binder that is used is Prussian Blue, which is a very efficient cesium binder. Two practical methods of application have been found: saltlick stone and bowel tablet, both containing Prussian Blue. Saltlickstone with Prussian Blue is the least expensive og the mitigating actions that have been in use in Norway.

A saltlick stone is practical with animals being relatively stationary, even when they are up in the mountain areas. There will, however, be individual differences, since some animals seem to need much less salt, and almost never go to the salt-lick stone. The other method, bowel tablet, has been tried out on reindeer during the summers of 1987 and 1988. The tablet dissolves slowly, and is meant to stay in the stomach of the animal all through a season. The test in 1987 was a failure, since the tablets dissolved too fast. Other production methods have now been developed and tested. In 1987 other and more cumbersome mitigating actions had to be adopted. Enclosures were erected, in which the animals were "down-fed" before slaughtering. With reindeer this method is much more complicated, since the animals are wild. It is quite a task to make them adjust to living in an enclosure and getting fodder different from their regular feed.

Cattle and goats grazing in the mountains, that are milked daily, have been given concentrate with Prussian Blue. In this case this is a practical solution, since these animals return "home" each evening.

Early slaughtering has also been used as mitigating action for reindeer. The diet of the reindeer varies strongly with the season, resulting in corresponding variations in concentration of radiocesium in the animals. The concentrations are lowest in late summer and early autumn, but will start to rise during the autumn, as the diet contains increasing amounts of lichen. The normal slaughtering time is in late fall or early winter.

13.2.1 Countermeasures to be used under practical conditions

Pollution of grasslands and natural pastures by airborne pollutants has become increasingly important for the health and well being of grazing animals, as well as for the quality of the products harvested for human consumption from such animals. Grazing animals utilize the plant growth of relatively large areas, and are therefore susceptible to hazards from direct surface depostion, as well as from pollutants transferred into the vegetation by biological transport.

In the more affected areas in Norway, levels up to 700 kBq/m² were measured. In contrast to the fallout from the nuclear weapons tests 20-30 years earlier, the fallout from Chernobyl was very patchy with high concentrations of activity in areas with precipitation. The fallout further contained a considerable portion of the radioactivity as colloidal particles, with ruthenium, cerium and cesium as radioactive constituents. The impact of these particulate pollutions in the long term behaviour of the Chernobyl fallout is not presently appreciated.

Parts of the heavily polluted areas are important in Norwegian animal production, being grazed by reindeer all year, and by sheep, goats and to a certain extent also by cattle in the summer. The main radioactive pollutant of animal products from the fallout in Norway was radiocesium, initially with 60-70% as Cs-137 and 30-40% as Cs-134. These radionuclides appeared in feeds shortly after the accident and the levels increased throughout the summer. Both meat and milk became contaminated with maximum levels in 1986 of about 3000 Bq/kg in beef, 8000 Bq/kg in sheep meat and 90,000 Bq/kg in reindeer meat. The highest levels in goat and cattle milk were about 1000 and 500 Bq/l, respectively. Grass silage with radiocesium contents of 5-10,000 Bq/kg dry matter (DM) were produced in the most heavily polluted areas. Since 1986 radiocesium levels in animal products has varied greatly. Major causes of variation include differences in the radiocesium levels of grazed pasture, and the use of cesium binders to reduce transfer of radiocesium to farm animals. Because of the time scope expected for the duration of the problems of cesium contamination in agricultural production in natural ecosystems (20-30 years) it has

been important to focus on mitigating actions which might alleviate the changes in farming practices used to minimize radiocesium contamination today. The action limits for foodstuffs vary between countries (Norway: 600 Bq/kg (Cs-134 + Cs-137) with the exception of 360 Bq/l in milk and 6000 Bq/kg for reindeer meat, game and freshwater fish).

The work carried out under the AKTU-275 program at the Agricultural University of Norway was especially concerned with:

- Defining halflives of radiocesium excretion in farm animals contaminated by fallout from the Chernobyl accident.

- Development and testing feeds, cesium binders and modes of treat-treatment which allow normal grazing practices to be used as much as possible.

Two additional areas of research has in part been funded from the *AKTU-275* program, namely:

- Studies of the transfer of radiocesium from pastures to goat milk.

- Comparison of the levels of radiocesium from the nuclear weapons tests with that of the Chernobyl fallout in animal products and fungi.

Experiments and some of the results from these activities will be reported in the following. The work was carried out as part of a broad program on effects of radiocesium pollution from the Chernobyl accident on animals and animal products by research workers in the Department of Animal Science and the Department of Zoology, assisted by the Isotope Laboratory, all at the Agricultural University of Norway.

13.2.1.1 Use of uncontaminated feeds for decontamination of farm animals

The chemical pattern displayed by the nuclear fallout in 1986 was different from that following the nuclear arms tests in the 1960's. The colloidal/particulate form may influence the absorption of cesium from the intestinal tract as well as the effects exterted by cesium binders. The experimental data obtained earlier with cesium salts and with spontaneously contaminated animals in the sixties had therefore to be verified in experiments with animals naturally contaminated from the Chernobyl accident. The purpose of the experiments undertaken was to determine the biological half life for Cs-excretions on uncontaminated feeds, and to study the potential effect of substances which may affect the halflife.

Feeding experiments with lambs

Three to four months old lambs were taken from contaminated mountain pastures in early august 1986 after a grazing period of about 45 days. Average muscle radiocesium content was 2000 Bq/kg. The animals were divided into six groups and fed different diets (Table 13.11). Roughages were either ammonia-treated straw ad libitum or hay harvested before the Chernobyl accident. Group 5, however, was given hay of the 1986 harvest containing 80 Bq/d. Hay (0.2 kg/d) and concentrate (0.5 kg/d) were given in equal parts twice daily. The sheep in Group 4 had free access to treated straw and the intake gradually increased to about 0.5 kg/d.

Although it is known from the literature that radiocesium release from contaminated animals best can be described by a multi compartment model, for practical reasons we decided to use as simple models as possible in explaining the results obtained. This assured that the data results could be used under practical feeding conditions. The excretion of radiocesium from the blood could be satisfactorily explained as a first order process, since the residual variance was 10% or less in all feeding groups. Radiocesium halflife in Group 1 fed a high concentrate diet was 18 days. Addition of bentonite or potassium chloride to the diet did not significantly influence the rate of cesium excretion, although somewhat shorter half lives were observed.

Other scientists have observed an increase in the cesium excretion by a factor of about 2 in reindeer and rats, respectively, when potassium intakes were increased 10 to 20 times from a very low level. Hay providing 80 Bq/d (Group 4) did not measurably influence the halflife. Group 5 which was fed straw ad libitum consumed 2.5 times more roughage than the other groups. The halflife was somewhat shorter than in Group 1 which received 0.2 kg/d of hay. Only 3 animals were given a combination of bentonite and potassium chloride (Group 6). A half life of 11 days was observed both for blood cells and for muscle. The halflife of cesium in the blood cells were 3-4 days shorten than that observed in muscle in all the feeding groups.

Group	Diet	n	Halflife (d+SD)		RBC/Muscle ratio
			Muscle	RBC	
1	Hay + conc.	7	18±4	14±2	5.7±1.1
2	Hay + conc. + bentonite	5	17±5	14±2	5.4±2.0
3	Hay + conc. + KCl	5	15±3	11±2	6.9±1.4
4	Straw + conc.	5	15±2	12±1	5.7±0.6
5	Hay (80 Bq/d) + conc.	5	17±3	13±2	6.3±1.3
6	Hay + conc. + bentonite + KCl	3	11±1	11±4	5.4±4.4

Table 13.11 Feeding of different groups of lambs. Diets and halflives for disappearance of radiocesium from red blood cells and from muscle tissue. n=number of animals. conc.= concentrate. SD=Standard Deviation. RBC=Red Blood Cells.

"On the farm" experiments with sheep and lambs

In order to study the disappearance of radiocesium under farm conditions and to follow the Cs activity for a more extended period two "on the farm" experiments were carried out at the end of the 1986 grazing season. Groups of lambs and sheep were slaugthered after being removed from contaminated pasture and offered clean feeds (indoor feeding or grazing lowland pasture). Biological halflives of 19.5 days in lambs and 24.0 days in sheep during indoor feeding and 18.6 days during grazing in lambs and were similar to the results obtained during more defined experimental conditions (Table 13.11). The halflives of radiocesium observed in the Norwegian experiments with sheep contaminated by grazing natural vegetation were somewhat longer than those reported in British studies of sheep contaminated with radiocesium from Chernobyl. The values resemble results from the literature obtained with use of inorganic radiocesium salts and show that the cesium absorbed from the feed must be in ionic form, i.e. as an ordinary salt.

13.2.1.2 Transfer of radioactivity to milk

Experiments with dairy cows

The transfer of radiocesium from hay to milk was studied in an experiment designed to test the efficacy of bentonite supplementation in reducing milk radioactivity in dairy cows (Table 13.12). The cows were fed 6 kg hay of the 1986 harvest, the daily intakes of radiocesium being 19-22 kBq/d. Silage (12 kg/d) and concentrates had a negligible radiocesium content. As calculated from the last 9 d of the 28 d feeding period 0.31% of the ingested radiocesium was recovered per liter of milk in the control group. Totally 8.0% of the daily dose was transferred to milk. The transfer coefficient on the 1986 hay was at the low end of those observed following the nuclear fallout in the 1960's. In Swedish experiments transfer coefficients as low as 0.5 to 0.11 was reported using grass harvested shortly after the Chernobyl accident.

Experiments with dairy goats

Considerable interest is focused on the production of goats milk during the summer grazing period. Goats milk is preferentially used for the production of whey cheese, and the evaporation of water during the production process concentrates the radiocesium about 10 times. The upper limit for radiocesium in goats milk therefore is about 50 Bq/l.

The transfer of radiocesium from hay grown in 1986, 1987 and 1988 were compared to the transfer from $^{134}CsCl$ in aqueous solution (ref. HA89). Diets were 1 kg/d of hay (1986: 6700 Bq/d, 1987: 1790 Bq/d, 1988: 1330 Bq/d), and 1 kg/d of dairy concentrates. The $^{134}CsCl$ solution contained 2000 Bq/d. The hay used for the experiment was grown in an area where the ground activity was 150-200 kBq/m². Milk yields were 1.0-1.5 l/d. Transfer factors of radiocesium from feed to milk were calculated from milk radioactivity levels obtained when the curves had leveled out (days 10-28). With hay from 1986 3.9% of the ingested radioactivity was transferred per liter milk, as compared to 8.8% and 12.8% with hay harvested in 1987 and 1988, respectively. Values for cesium chloride were 10-12%. Thus the transfer factors for radiocesium from hay of the 1986 harvest to milk were about 1/3 of those observed with ionised cesium both in our cow and goat experiments. An average transfer coefficient of 8.8% for the hay from 1987 is in contrast to this. One likely explanation to this discrepancy is that the activity in the hay during the first season after the accident contained particles with poorly absorbable radiocesium. This could either be particles originating directly from the Chernoby reactor, or less likely, cesium adsorbed to soil and clay particles which had been redistributed to the vegetation following exposure to rain and wind. The much larger transfer of cesium activity from the 1987 and 1988 harvests indicate that the cesium appeared in the plants following root uptake, and thus is more readily digestible by the animal. The test of transfer coefficients involve considerable experimental work and facilities for keeping of animals. It should therefore be mentioned that the availabilities observed in the present study resemble data obtained by in vitro incubation techniques within project AKTU-265, performed at the Isotope laboratory at the Agricultural University.

13.2.1.3 The use of cesiumbinders to reduce cesium contamination

Clay minerals like bentonite, vermiculite and natural and synthetic zeolites have fixed binding sites for cations which bind cesium with a high affinity. The minerals have proved to be helpful for reduction of cesium transfer to animals both in experimental situations and during feeding with roughages contaminated with radiocesium fallout. Special interest has also focused on the monovalent metal salts of the ferrihexacyanoferrate complex, since these salts exchange an alkali-metal ion with cesium and binds it in the crystal lattice with high affinity.

Experiments with bentonite

Experiments with cows and goats showed that bentonite reduced the absorption of radiocesium from roughages contaminated with fallout from Chernobyl. A daily ration of 500 g (2.5%) of bentonite reduced the transfer with about 75% (Table 13.12). In goats a 5% supplementation of bentonite to the concentrate was used during the grazing season in 1987. This proved, however, to be of too low efficiency in areas where the vegetation contained 1-3 kBq/kg dry matter, and increased concentrations of bentonite and salt licks with Preussian Blue had to be given to control radiocesium absorption.

There are indications in the literature that supplementation of bentonite may significantly affect digestion and mineral balances in ruminants. Therefore balance studies were carried out with four sheep during two periods each of four weeks. The daily diets were kept constant throughout consisting of 500 g hay of the 1986 harvest (3100 Bq/kg) and 500 g Dairy mix with 15% digestible crude protein. During period two the animals were supplemented with 50 g bentonite mixed with the concentrate.

Bentonite	Milk		Radiocesium	
g/d	kg/d	Bq/l	%dose/l	%dose/d
0	26.7	51	0.31	8.0
250	23.7	28	0.21	4.7
500	25.0	16	0.07	1.8
750	23.9	16	0.10	2.2

Table 13.12 Transfer of radiocesium from hay grown in 1986 to milk. Effect of addition of bentonite during a 28 d feeding trial. Daily intake of hay was 6 kg (19-22 kBq radiocesium), cesium values are averages of milk concentrations during the last 9 days of the experiment.

Faeces and urine were collected daily for radiocesium determination. Ration digestibility and mineral balances were determined for the last 16 days of each period. A small decrease in organic matter digestibility (0.70 vs 0.68) was observed with bentonite supplementation. No difference in protein digestibility (0.68) was observed (Table 13.13). There were no indication of lower balances of phosporous, as has been previously reported from other investigations, or of other minerals after bentonite supplementation (Table 13.14). However, due to the high content of aluminium in bentonite, intake, faecal excretion and measured balances were much higher during period 2 when bentonite supplements were given.

Animal no.	Control Organic matter	Control Protein	+ Bentonite Organic matter	+ Bentonite Protein
22	71	71	69	69
23	69	66	65	64
25	69	65	67	65
27	71	65	69	68
Mean	70	68	68	68

Table 13.13 Effect of addition of 50 g of bentonite on the digesti-digestibility (% digested) of a hay/concentrate ratio (50/50) in sheep.

	Intake[A]		Faeces		Urine		Balance	
	-B	+B	-B	+B	-B	+B	-B	+B
P	3.76	3.49	3.31	2.96	0.09	0.20	0.36	0.33
Ca	7.14	7.26	8.13	7.86	-	-	-0.99	-0.60
Mg	2.18	2.29	1.71	1.85	0.15	0.21	0.32	0.23
K	8.61	7.65	0.76	0.64	4.74	3.59	3.11	3.42
Na	1.32	1.81	0.40	0.31	1.45	0.49	-0.52	1.01
Al	0.12	1.96	0.09	1.12[B]	-	-	0.03	0.85[B]
Fe	147	473	144	445[B]	-	-	3	28
Cu	8.4	9.3	10.6	4.1[B]	-	-	-2.2	5.2[B]

[A]: Values in g/d except for Fe and Cu (mg/d)
[B]: Significant difference ($P < 0.05$) between +B and -B

Table 13.14 Intake, excretion and balance of minerals in sheep fed a hay/concentrate (50/50) diet supplemented with 50 g bentonite.

In experiments with contaminated lichens fed to reindeer bentonite has likewise been effective in reducing the absorption of cesium. Moist bentonite sticks strongly to the hairs on the muzzle. Therefore bentonite should preferentially be incorporated in other feeds and made into pellets.

Experiments with cesium binders of the Preussian blue type

Various forms of iron(III)hexacyanoferrates(II) have proved effective as cesium binders in vivo both in laboratory and domestic animals, and in man. Interest in these substances as cesium binders has been renewed following the Chernobyl accident. Recent studies (ref. AR88) in West Germany showed that the ammonium-iron hexacyanoferrate (AFCF) compound is unabsorbable, stable in the digestive tract of ruminants, and apparently non-toxic in doses required to control cesium absorption. According to the manufacturer AFCF and related substances will reduce the transfer of radiocesium from feeds to milk or meat with 90 to 95% when given in doses of 1-2 g to sheep and 3 g/d to lactating cows. No reports appear to exist on the dose/response relationship for AFCF in binding radiocesium in ruminants. The doses required daily for reduction in absorption becomes critical when grazing animals infrequently handled by humans are concerned. In order to study the possibilities of supplying the hexacyanoferrates by means of a salt lick or a rumen depot tablet experiments were undertaken to determine the lowest efficient doses of AFCF in reindeer.

In spite of the fact that the dilution in the stomachs of the ruminant is considerable, the dose/response experiments showed that AFCF in daily doses of as little as 100-200 mg would reduce the absorption of cesium by 80% or more in small ruminants. The chemical amounts of radioactive cesium is very much lower than the amounts of stable cesium in the feed. It can therefore be assumed that a major part of the binding sites for cesium in the supplied cesiumbinders will be occupied by stable cesium. We may thus expect that any cesiumbinder will bind a constant proportion of the radioactivity in feed or vegetation, and that an 80-90% reduction in radiocesium transfer may be achieved by similar doses of binder also in areas with much higher levels of pollution than those experienced in Norway after the Chernobyl accident. In our experiments the hexacyanoferrates were 2-500 times more active than bentonite on a weight basis. They are therefore especially useful as cesiumbinders for grazing animals.

Effects of AFCF on transfer to lactating animals under field conditions

Dairy animals usually receive some concentrate also during the grazing season. Cesium binders (bentonite and ammonium-iron hexacyanoferrate, AFCF) are conveniently dosed by addition to the concentrate. Addition of 5% bentonite led to a reduction in milk radiocesium of about 50-60%. In 1989, bentonite was replaced by AFCF (1g kg^{-1} concentrate). Reductions of up to 90-95% of the radiocesium content of the goat milk were obtained in animals fed 0.5 kg concentrate per day.

Salt licks for grazing animals

In Norway, salt licks with AFCF were developed in 1986/87 and have been in general use in the grazing season of 1989. During the summer of 1988 experiments to test the effect of sodium chloride blocks of 10 kg containing 2.5% AFCF were carried out. The body burdens of radiocesium in sheep were measured monthly in 10-30 sheep in three districts contaminated by fallout from the Chernoby accident. Sheep from a neighbouring area where ordinary sodium chloride salt licks were given served as a reference group. A significant reduction (25-75%) in the accumulation of radiocesium was observed in the groups which had access to the AFCF stones. Care must, however, be used in the interpretation of quantitative difference between the control and the AFCF treated groups since the sheep did not graze the same lands. The results from the 1988 grazing season indicated that salt licks with AFCF will be a simple and cost effective method to reduce the accumulation of radiocesium in grazing ruminants. Present data shows that a reduction of approximately 50% in radiocesium levels may be expected in mountain areas where the sheep make use of salt licks.

Sustained release boli delivering AFCF to grazing animals

A slow release bolus containing AFCF provides an opportunity to continuously deliver the cesiumbinder to rumen liquid for a period of weeks or preferably months after treatment. We have developed and tested boli of varying size and compositions for use in goats, sheep and reindeer.

The boli are presently under test, but promising results have been obtained.

In goats: The transfer of radiocesium to milk was reduced from 10% of the daily ingested dose to 1.5-2% for a period of 45 days in treated animals.

In lambs: Five sets of twin lambs grazing contaminated pasture (3000 Bq/kg dry matter) for 4 weeks acquired radioactivity levels of 1400 Bq/kg meat. Then one of the twins was kept as a control whereas the other was treated with 2 boli (5 g AFCF). The ewe did not receive cesium binder. In treated lambs radiocesium concentration decreased during the three first weeks to about 700 Bq/kg^{-1}, i.e. a 50% reduction in the body burden of radiocesium. The same degree of reduction has so far been observed in field trials including several hundred sheep which are under way the present grazing season. This reduction is equivalent to what can be achieved by feeding sheep uncontaminated feeds for three weeks and will be a cost effective alternative to this feeding in preparation for slaughter.

In reindeer: The radioactivity levels in reindeer have been successfully reduced from 25 kBq/kg meat to 3-4 kBq/kg in the course of 3 months by two treatments with the sustained release boli. Further studies are, however, required to solve problems connected with the application of the boli in this animal species before treatments can be used routinely.

13.2.1.4 Studies on the role of fungi in the transfer of radiocesium to grazing ruminants

In connection with the studies of radiocesium uptake by sheep, goats and reindeer in the highly contaminated Jotunheimen area we observed that fungal fruit bodies had a very large effect on the radiocesium levels at the ordinary time of slaughter. The results are presented in a paper to be published in a short time, and the main points are given in the following abstract:

Radiocesium fallout from the Chernobyl accident transferred easily to grazing ruminants. Studies of the dynamics of radiocesium transfer from vegetation to milk in dairy goats grazing heavily contaminated mountain pasture in Southern Norway have been performed in the years following the accident. Radiocesium levels in milk and green vegetation remained stable throughout 1986 and 1987. In 1988, a sudden 3 - 5 fold increase in milk levels occurred during the second half of the summer. Levels in sheep also increased rapidly. This coincided with an abundant growth of fungal fruit bodies with radiocesium levels up to 100 times higher than green vegetation. Fungal radiocesium was highly available. Milk radioactivity levels could be accounted for by consumption of as little as 20 - 100 g/d of funghi (dry weight). Fungal fruit bodies, when abundant, were the major source of radiocesium for ruminants. This ability of common fungal species to mobilize radiocesium from natural soils and enhance the transfer into the human food increases the vulnerability of food production in natural ecosystems to pollution from nuclear industry.

In a recent reevaluation of the data of radiocesium in natural ecosystems (sheep and goat production in Norway) (ref. HO89) calculated an efficient halflife for radiocesium of about 20 years. It is not known to what extent the activity of fungi plays part in determining the long halflife of the radiocesium which was deposited after the nuclear weapons tests in the 1960's. It should, however, be noted that concentration ratios of radiocesium of bomb origin between soil and fruit bodies of Rozites caperata were similar to those observed in 1988 with radiocesium from the Chernoby accident. In conclusion, we have shown that fungal transport and recycling of radiocesium can be a major pathway of transfer for radiocesium from natural soils to grazing ruminants and their products.

13.2.2 Mitigating actions adminstrated by the Department of Agriculture

The actual mitigating actions carried out in Norway in the years following the Chernobyl accident are described in (ref. TV87b, TV88b, TV88c and ST89), and in this subchapter. It is planned to continue publishing reports summing up the situation each year as long as the Chernobyl fall-out constitutes a problem in Norway. The mitigating action program has not been part of the AKTU program. However, since it is one of the most comprehensive mitigating action programs that have been carried out, permission has been granted to present it here. Summing up of the costs of the mitigating action program and publication of this information has been performed within project *AKTU-285*, in cooperation between the Institt for energiteknikk, Norway and the Department of Agriculture in Norway.

During the summer 1986 the National Institute of Radiation Hygiene collected soil samples from all municipalities in Norway; and it was found that Norway was probably the most heavily contaminated country in Western Europe. In large areas the deposited activity was above 100 kBq/m^2, and the total amount of radiocesium deposited over Norway is estimated to be 6% of the radiocesium released from the reactor. The areas where ground concentrations are highest are mostly in sparsely populated mountain areas. These areas are, however, important in connection with several nutritional pathways; notably sheep, goats, reindeer and wild freshwater fish.

The mitigating action program described in this subchapter is organized by the Ministry of Agriculture. The sampling program is approved by the National Institute of Radiation Hygiene.

13.2.2.1 Milk

Cattle had not started feeding outdoors at the time of the accident, and iodine in milk was never high. The cattle was let out at the normal time. Cesium was found in the milk from late May 1986, and the content increased for some time. Only in one limited area were the action levels exceeded. The milk from this area was used as animal fodder. Following ancient agricultural practices, much cattle is still brought up in the mountain areas in the summer months, to graze at random in these rich natural pastures. The milk with high content came from these herds.

13.2.2.2 Sheep

It was decided not to put restrictions on the use of grazing areas for sheep. In most of Norway sheep are brought up to the mountain commons in early summer, where they are free to move around until early fall. Towards the end of July 1986 it was found that the content in mutton was above the action level in some areas, and the 31. July 1986 it was decided by the Ministry of Agriculture, in cooperation with regional agricultural/veterinarian officers, to initiate a program for surveillance, mitigating actions and economic compensation. Information material on feeding etc. was sent to about 30,000 farmers both in 1986, 1987 and 1988. The Ministry of Agriculture, in cooperation with regional veterinary officers, was responsible for dividing the country into three types of zones. Roughly 70% of the sheep were in free zones (average content below 600 Bq/kg). Roughly 3% of the sheep were in so-called ban zones (average content above 2000 Bq/kg). Slaughtering proceeded as usual, but the meat was classified as unfit for human consumption. Initial plans for burying the interdicted meat was later changed to using the meat as fur animal fodder. The remaining areas, containing 27% of the sheep, were referred to as special measures zones. Here the cesium content was reduced below the action level by the use of cesium-free fodder for periods of 4 to 8 weeks. In addition concentrates containing bentonite have also been given to the animals.

The problem areas in 1987 were the same as in 1986, though somewhat smaller in size. No areas were classified as "forbidden". Roughly 77% of the sheep were in free zones, and 23% in the special measures zones. The "down-feeding" program, covering a total of ca. 280,000 sheep, has been very successful. The conditions in 1987, however, were especially favorable, since the autumn was unusually mild and the first snow-fall unusually late. In the down-feeding program a half-life in sheep of 21 (intially 24 days) was used. In 1986 the division into zones mainly was built upon measurements on meat samples. The problems were easier to handle in 1987, in part because live animal measurements were adopted. The National Institute of Radiation Hygiene and the Division of Veterinary Services of the Ministry of Agriculture have cooperated in the testing and calibration of equipment for measuring levels in live reindeer, cattle and sheep. The equipment has also been tested and calibrated for direct measurement of radioactivity in carcasses from the species at slaughter.

Because 1988 was an exceptionally good year for mushrooms, one observed levels of radiocesium in sheep 3 - 4 times the levels of the previous year in many parts of the country; and roughly 30% of the sheep were in special measures zones, and 70% in free zones. The down-feeding program involved a total of about 360,000 sheep. Sheep with levels up to 40,000 Bq/kg before down-feeding were successfully included in the program. The conditions in 1987 and 1988 were, however especially favorable for carrying out the down-feeding program, since the years were unusually mild and the first snowfall unusually late.

13.2.2.3 Reindeer

Levels up to 150,000 Bq/kg were found in reindeer from mountain areas in Southern Norway, as summer 1986 proceeded. Only in three areas was the content below the action level. The content decreased from June to August, but increased in September 1986, with further increase through the winter, since the reindeer then feed mostly on lichen. 31. July 1986 it was decided by the Government to initiate a program for economic compensation. In November 1986 it seemed that 85% of the production for 1986 could not be used. Because of the severity of this impact (which also mainly hits a minority population group), and because reindeer meat is not an important part of the diet of the average Norwegian, the Health Directorate decided 20. November 1986 to increase the action level for reindeer meat by a factor ten. With this action level much of the production was saved from interdiction. However, about 560 tonnes of reindeer meat were interdicted, converted into meat/bone flour and buried.

However, in 1987 several different types of mitigating actions were employed. This program involved individual measurement of the animals before slaughtering, early slaughtering, use of less contaminated areas for feeding, use of saltlickstone with Prussian Blue, and down-feeding. A special bowel tablet containing Prussian blue (a cesiumbinder) was also developed at the Norwegian Agricultural University, but successful large-scale use was not attained until 1988. Early attempts failed because the tablets, when mass-produced, dissolved too fast to last through the required time period. In 1987 almost 14% of the production (312 tonnes) of reindeer meat was saved in the early slaughtering program. The amount of animals with access to saltlickstone is not known, nor is the number of animals using areas with lower fallout levels. The number of animals in both cases is undoubtedly high. On the other hand, the number of animals in the down-feeding program is small. Even after these mitigating actions 10% of the production (216 tonnes) had to be interdicted.

Also in 1988 these mitigating actions were employed. About 6% of the production (126 tonnes) had to be interdicted in 1988.

13.2.2.4 Beef and horse meat

In some samples of beef and horse meat from the most affected areas one measured cesium content above the action level in 1986. These products were from animals that had been grazing in the mountain areas. Accordingly it was decided that only meat from animals that had been grazing in cultivated fields for a period of at least 4 weeks, or that had been fed indoors for at least 4 weeks, would be approved for slaughtering. This restriction was issued the 8. September 1986. In addition to the cesium-free fodder, cattle in the affected areas were fed concentrates containing bentonite, a clay mineral proven to be a cesium-binder in ruminants.

The approach in 1987 was somewhat different. Some areas were classified as "precautionary measures zones" or "special measures zones" respectively, while most parts of the country had no restrictions. In the precautionary measures zones the cattle were fed cesium-free fodder, to the extent this was possible, and in addition concentrates that had about 5% bentonite added. Compensation was not granted in these areas. Live cattle from these areas were measured at arrival to the slaughtering house, but were returned to the farm, if it was found that the content was above 600 Bq/kg. The mitigating actions in the special measures zones were the same, but a compensation was given of 8 kr per day and animal. Compensation equivalent to the value of the animal was given in addition to this, if downfeeding was not successful, and the animal had to be discarded. Cattle with levels of radiocesium up to 3000 Bq/kg were measured in 1987. The number of animals in the down-feeding program is not known exactly, but is estimated at 45,000 for the whole country.

In 1988 the problems were of the same character as in 1987. Cattle with levels up to 6000 Bq/kg were measured. The number of animals in the downfeeding program in 1988 is assumed to be of the same order of magnitude as in 1987, although it is not known exactly.

13.2.2.5 Plowing and use of fertilizer

In 1987 the Division of Agriculture of the Ministry of Agriculture described how the activity levels in vegetation could be reduced by simple mitigating actions like e.g. additional plowing and administration of fertilizer. These practices have probably been effective in reducing the activity levels in the animal feed from the home farm, leading to corresponding reduction in activity levels in the milk and meat from cattle. The available information on these aspects are, however, much too sparse to allow estimation of the resulting activity reduction, but there is reason to believe it is considerable, and that it will continue to be so for a number of years to come.

13.2.2.6 Dietary advice

In addition to the types of mitigating actions that are aimed at reducing the levels of radioactive materials in various foodstuffs, there are other types of mitigating actions that function by aiming at reducing the intake of specific foodstuffs. To population groups with a particularly large intake of reindeer meat and/or wild freshwater fish, dietary advice was given (ref. HE87) containing recommendations upon how frequently one could consume foodstuffs with activity levels above the action levels. In 1987 a survey of the changes in diet caused by the Chernobyl accident was carried out in the municipality of Sel in Norway, covering two population groups: one consisting of specially selected persons and one chosen at random (ref. BO88). This municipality is in one of the areas of Norway where the fall-out level was high. The survey indicated that a reduction in annual intake per person of 30,000 Bq was attained the first year after the accident, compared to what the intake would have been if there had been no change in the diet.

13.2.3 The addition of potassium chloride to a lake and its effect on the concentration of radiocesium in trout, project AKTU-290

This work was performed as part of project *AKTU-290*, and is presented in (ref. CH89).

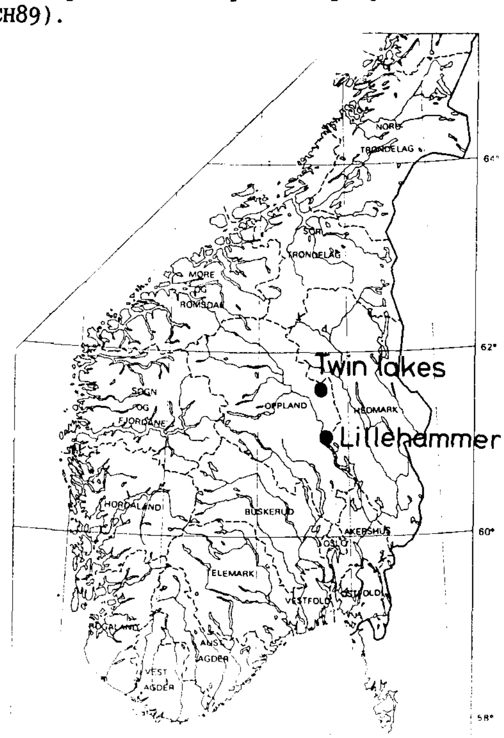


Figure 13.13 *Location of the twin Åsdalstjern Lakes.*

The experiment was done in two twin mountain lakes 1130 m above sea level, called Inner and Outer Åsdalstjern. They are located about 60 km north of Lillehammer in an area rather heavily contaminated by the Chernobyl fallout, and the water from the inner lake (L1) flows into the outer lake (L2). The lakes have about equal surface areas and mean depths, and trout is the only fish species in both. L1 has served as a reference lake, while potassium chloride (KCl) has been added to L2.

The number of original wild trout was estimated to about 600 in L1 and 100 in L2. The content of radiocesium in these trout was rather high. To each lake an additional 250 tagged trout were stocked the 24 June 1987. These trout were of very low radiocesium content, and being tagged they were easily recognizable from the wild ones. Thus each lake now had two trout populations that could be studied: the tagged one for radiocesium build-up study, and the wild one for reduction rate study.

Potassium was added to L2 in September 1986, July 1987, and July 1988 in a quantity of two metric tons each time. Each addition of potassium lead to an increase in the K^+-concentration of the water in L2 by a factor of about 10. The concentration had been allowed to come back to the normal low value before the next addition. Several sediment profile samples were taken in each lake in October 1988.

Samples of trout (5 - 8 fish of each population) were taken simultaneously in L1 and L2 in August 1986, June, July, August and September 1987, and July and August 1988.

The results of the reduction rate study up towards the end of 1987 (actually at ca. 350 days in the figure) show that the effective half-life of cesium-137 in the trout was about 250 days in L2 compared to

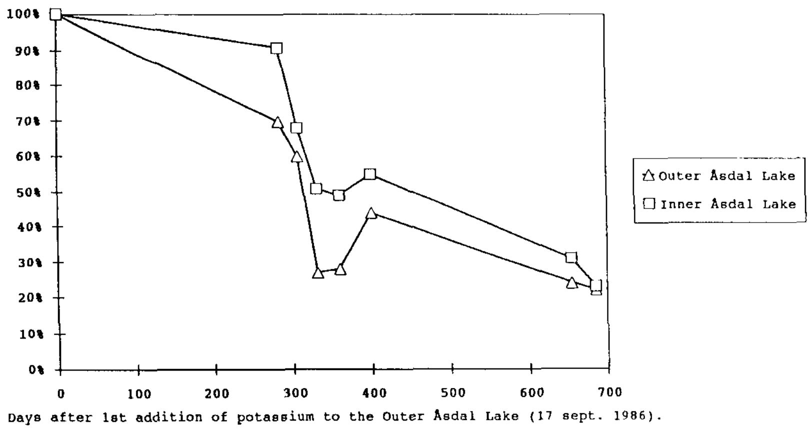


Figure 13.14 Relative concentrations of Cs-137 in wild trout from L1 and L2 as a function of days after the first addition of KCl to L2.

about 400 days in the reference lake L1. Figure 13.14 actually contains measurements beyond this date, but they appear rather confusing. The last day at which samples were collected shows the same reduction in the two populations. Analysis of these samples was unfortunately delayed, and before these results were available, it had been decided not to allocate further funds to these experiments. Accordingly it has not been possible to determine whether the latest results were perhaps unrepresentative.

13.3 Long-term mitigating actions in Sweden after Chernobyl

13.3.1 Measures to reduce the content of cesium-137 in fish

Cesium-137 in perch from lakes in northern Sweden after Chernobyl (1988)

Lars Håkansson et al.; The National Environment Protection Board, Stockholm

The aim of the project is to make systematic studies and test various possible means and combinations of means, like liming of lakes, of wet land and of entire drainage areas, fertilization with commercial fertilizers and with nutrients from fish cage farms, intensive fishing and the adding of potash.

This project is carried out within the framework of a major project called "Liming - Mercury - Cesium" in close cooperation with the county administrations of Gävleborg and Västernorrland.

The actual situation

Surveys have shown that there exists an area especially affected, where the cesium content in fish exceeds the action level recommended by the Swedish National Food Administration.

In some parts of the counties Gävleborg and Västernorrland there are two environmental problems. These lakes also constitute a high priority group in mercury contexts. The problems of cesium and mercury in water and ground are closely related to the energy production and the gradually increasing acidification.

Investigations

Comprehensive investigations are now being performed in 41 lakes in these counties. Perch is used as biological test organism.

Remedial measures

There are several possible remedial measures to speed up the natural decrease of Cs-137 in fish, e.g.:

- adding of potash (K) which is "similar" to cesium and could cause a biological dilution of radioactive cesium in the biota in the lakes

- adding of lime, which alter many chemical and biological characteristics of the lake

- adding of nutrients (P and N) which could increase the bioproduction and hence, in this sense, give rise to a "dilution" of a given cesium dose

- extensive fishing, which would change the fish community/predation

Today there are no scientifically tested and ecologically valid guidelines on "diagnosis and medication" of the various possible measures to reduce the amount of Cs-137 in fish.

Results

- Correlations

  There is a cluster with strongly internally correlated parameters which include alkalinity, conductivity, K-concentration, Ca-concentration and water hardness (Ca+Mg).

  The intention has been to show that some very few simple key parameters in a representative manner are able to reflect even other variables which hardly can be determined.

  A formula is outlined by which values in the entire cesium range can be predicted. However, this formula may not be used uncritically, since it is based on an empirical set of data which covers only certain lake types.

  By the primary dose received from the ground contamination about 70% of the cesium variation in perch can be explained. The degree of explanation is increased to > 80% if the conductivity of the lake water is included, and to as much as 85% if the phosphoros content of the lakes is considered.

- Cesium content reduction

  Cesium in perch from all 41 lakes has been reduced as follows:

The median value from	4 000	Bq/kg w.w.	1986	to	2 700 Bq/kg
The maximum "	" 131 000	"	1986		24 800 "

  That is by 33% and 81%, respectively.

  No specific factor has been found which can be linked to the decrease in perch between the two years. The main part of the reduction depends on natural causes and cannot be related to mitigating actions.

Recovery

A preliminary recovery prognosis has been made, which indicates that the recovery will take a rather long time if no specific measures are taken to speed up the process.

13.4 Long-term mitigating actions in Finland after Chernobyl

The contamination levels in Finland were far below levels where relocation or decontamination would be considered.

The contamination of agricultural products in the whole country was also low. The radioactivity concentrations were well below the intervention levels. In most cases cesium 137 levels were less than one tenth of the intervention levels.

Some non-agricultural products like game meat, reindeers, inlake fishes and mushrooms were occasionally contaminated to a higher degree. Game-meat, especially moose, was relatively highly contaminated in May - June 1986. Values over 1000 Bq/kg were found. However, during the hunting season cesium 137 values were already approximately 300 Bq/kg. Also in 1988 some higher values were found, due to a good mushroom year. Anyhow, no special advice were needed for moose.

Mushrooms have shown very variable cesium 137 concentrations. During the past three years some advice has been given. The most important kind of mushroom is Lactarius and it has shown the highest cesium 137 levels. These mushrooms are always parboiled before eating. It was recommended that these mushrooms should be boiled in two waters. The cesium content will decrease with this treatment to about 5 percent from the original. For people using very much mushrooms it was recommended to restrict the consumption of mushrooms to one kilogram per week.

The greatest concern for high cesium 137 concentrations was, and still partly is, in lake fish. Some of the smaller lakes having a large catchment area were collecting water from large areas and, if they additionally were nutrient poor, fishes in these lakes contained up to thousands of bequerels per kilo of cesium 137.

In the first two years the recommendation was that fish taken from lakes in the highest fallout areas should not be eaten as a main dish more than two to three times a week. In 1988 this recommendation was modified to concern only predatory fishes. By this recommendation the aim was to restrict doses, even to people who ate inlake fish with the highest contamination every day, to less than 5 mSv per year.

13.4.1 Recommendation on use of horticultural peat

In the growing season of 1986, it could safely be said that greenhouse vegetables were free from radioactivity. However, the horticultural peat produced during the summer of 1986 could be expected to contain widely varying amounts of cesium 137 and cesium 134 from the Chernobyl fallout. Thus the Finnish Centre for Radiation and Nuclear Safety (STUK) decided in December 1986 to undertake a survey of the radioactivity content of peat intended for use in vegetable growing in the 1987 season (ref. RA89). The survey covered commercial producers of horticultural peat as well as peat storages of greenhouse growers. A simple transfer model was used to relate the radiocesium concentration in peat to the resulting actitivity in fresh tomatoes, cucumbers and lettuce. Some growing experiments in pots were quickly undertaken in order to estimate applicable transfer factors.

Recommendations were decided upon jointly by STUK and the State Institute of Agricultural Chemistry. The recommendations were distributed together with the results of the horticultural peat radioactivity survey, and went into effect in January 1987. The results were graded into three categories based on the cesium 137 content, as follows:

A Less that 1500 Bq/kg: The peat is suitable for vegetable growing without any further measures.

B 1500 - 3000 Bq/kg: The peat is suitable for vegetable growing, but measures to reduce the radioactivity of the peat are recommended when possible at reasonable cost.

C More than 3000 Bq/kg: Measures to reduce the radioactivity should be undertaken.

The recommendations identified measures to reduce the radioactivity content, such as replacing or diluting the peat with non-contaminated material, or, conditionally, watering the peat thoroughly with clean water. The use of contaminated lots of peat was approved in flower-growing, soil improvement in the fields and in landscaping.

The stated objective of the recommendations was to keep the cesium 137 concentration below 100 Bq/kg in the major part of the greenhouse vegetables in 1987. This goal was achieved quite easily, indicating that the transfer factors used did not account for all the factors that reduce the transfer of radiocesium to plants in commercial vegetable production.

13.4.2 Recommendation on use of fuel peat

Combustion of fuel peat yields about 4% of the total primary energy consumption in Finland. In 1987, the total fuel peat burned amounted to 13.5 x 10^6 m^3 (4.3 x 10^6 tons). In the fallout region where the Cs-137 concentration was larger than 5 kBq m^2, the total amount of fuel peat harvested in 1986 was about 2.2 x 10^6 metric tons dry weight (the water content of milled peat is about 50%). In a survey undertaken at six peat-fired power plants in different parts of Finland throughout the heating season 1986-87, the radiocesium 137 concentrations in fuel peat varied between 30 and 3600 Bq/kg dry weight. Differences in radionuclide concentrations between the power plants were great and also the radionuclide composition in fuel peat varied regionally. Regarding the production and handling of fuel peat, there was no need to give any special recommendations from a radiation protection point of view. In reality, the production and burning of fuel peat may have a net positive radiological benefit since the contaminated surface layer of peat bogs is removed and, in the combustion, the activity is concentrated into a small amount of peat ash.

13.4.3 Recommendation on use of fuel peat ash

The mean ash concentration of dry milled peat is about 5%, so the total ash amount of fuel peat produced in the fallout region where Cs-137 concentrations were larger than 5 kBq m^{-2}, was in 1986 about 110,000 tons. The survey of concentrations of fallout radionuclides in precipitator peat ash was started in September 1986 and it was continued until June 1989. Variations in activity concentrations between the studied power plants were great. During the first heating season, 1986-87, the monthly averages of Cs-137 concentration, inside the fallout region, varied from 3000 to 67000 Bq/kg dry weight. Outside the fallout region the Cs-137 concentrations were usually below 1,000 Bq/kg.

Before the Chernobyl fallout, peat ash was used as a fertilizer in silviculture (foresty) and in agriculture, as a land-fill material at building sites, and as a filler in concrete industry. As a general instruction about the use and handling of the contaminated peat ash after the Chernobyl fallout, the STUK recommended that peat ash should be removed to dumps. In dumps, peat ash will be covered by other dumping materials and, in addition, public dumps are places which are under public surveillance as long as the peat ash is contaminated by the fallout radionuclides. This recommendation did not concern the administrative districts of Karelia, Oulu and Lapland.

In a few cases the use of contaminated peat ash as a land-fill material and as a filler in certain concrete products (not concrete for housing construction) was accepted once its radiological consequences had been estimated by the STUK and certain radiation protection measures had been taken into account.

In the heating season 1988-89, the activity levels in peat ash were significantly lower than during the previous heating seasons. For that reason the STUK gave more detailed recommendations on utilization of peat ash. In these recommendations, peat ash with a Cs-137 concentration greater than 10 kBq/kg was still recommended to be removed to dumps. Below this level, peat ash was accepted to be used as a fertilizer in silviculture and in agriculture, although in amounts smaller than 10 tons per hectare. Peat ash with a Cs-137 concentration below 10 kBq/kg was also accepted for use as land-fill material provided certain measures minimizing the radiation dose were taken into consideration. Peat ash with a Cs-137 concentration below 1 kBq/kg was accepted for use as land-fill material without any special radiation protection measures. When cesium concentration in ash is greater than 1 kBq/kg, the ash is not accepted to use as a filler in concrete for housing construction, and any time when peat ash is used in concrete, the activity of ash has to be measured.

13.4.4 Recommendation on use of sewage sludge

In Finland a recommendation dating from 1977 given by the medical board, stipulates that sludge may be used as a soil improvement agent on fields, with a maximum amount of 20 tons (dry weight) per 10,000 m^2 at intervals of at least five years. Sludge is not recommended for use in the growing of vegetables or on grazing.

If sludge accumulated in 1986 were spread on fields in the maximum amount allowed, the increase in the amount of Cs-137 activity might be at most 24 kBq/m^2. Cultivation experiments performed in 1987-88 indicated that the addition of sludge equal to 10 kBq/m^2 of Cs-137 would result in 2-3 Bq/kg in spring wheat and barley (ref. PU89). Thus, it would seem that the use of sewage sludge within the recommended quantity limits could not cause significant contamination of grain. Even if concentrations in the sludge were very high, the overall significance of this route, compared to direct transfer from deposition, may not be so great.

After the accident at Chernobyl, the Finnish Centre for Radiation and Nuclear Safety recommended that in order to avoid unnecessary doses, sludge dating from May 1986, containing the greatest amount of radioactive material, should not be used on fields as a soil improvement agent. According to this recommendation, sludge could still be used for landscaping. No special activity limits were applied to use of sludge in Finland. Transportation to fields of sludge from the treatment plants with the highest concentrations of Cs-137 activity was avoided voluntarily by the treatment plants during spring and summer of 1986. Studies are under way to evaluate the transfer factors from sludge to grain. These factors are needed when calculating doses and giving recommendations regarding the utilization of sludge.

13.4.5 Reevaluation of recommendations and decisions by competent authorities

When the fallout situation was detected in Finland on 27-28 April, the bulk of the radioactive cloud was at a height of one to two kilometers. Before the onset of rain on 29 April in Central and Southern Finland only slightly increased radiation levels were reported. Thus, the authorities did not anticipate at first that there would be any need for countermeasures or recommendations affecting the general public.

However, after the rain showers had washed radioactive material from the cloud, resulting in comparatively high levels of iodine-131, cesium-137 and cesium-134 deposited on the ground, it became clear that food chains were going to be affected and advice by the competent authorities would be needed fairly quickly.

The principles stated by the International Commission on Radiological Protection (ICRP) in its Publication 40 for planning intervention in the event of an accident are:

a) individual dose should be limited to levels below the threshold for serious non-stochastic effects,

b) the risk from stochastic effects should be limited by introducing countermeasures which achieve a positive net benefit to the individuals involed,

c) the overall incidence of stochastic effects should be limited, as far as reasonably practicable, by reducing the collective dose equivalent.

The condition a) was inherently satisfied for both external and internal exposure in the situation prevalent in Finland.

As for conditions b) and c) the ICRP further states that the decision to introduce any countermeasure should be based on a balance of the risks and disadvantages to the individuals affected. It is realized that the implementation of any countermeasure to reduce the exposure of the public carries with it some detriment to the people concerned, including social and economic disruption. The experience from handling the Chernobyl situation in Finland showed that in actual practice it is not possible to carry out the risk comparisons and cost-benefit analyses implied in ICRP 40, principles b) and c). Because of limited expert resources and the time scales involved, the recommendations given by STUK and other authorities in many cases had to be decided upon with haste, based on the experts' impression of the situation. The experts were frequently confronted with novel radiation exposure pathways for which transfer factors and other essential parameters were not available but had to be crudely estimated.

The way of applying the ICRP's basic principles in the event of an accident is also in question. At the International Atomic Energy Agency (IAEA) conference on Radiation Protection, Sydney, Australia, in 1988, several experts (ref. AL88 and IL88) expressed the opinion that in order to apply the linear non-threshold concept in practice, a lower bound value for individual dose is needed. According to a report by the Swiss Nuclear Safety Inspectorate (ref. SW86a), the approach of the Swiss National Emergency Organisation in the Chernobyl situation was as follows:

- measures that would reduce the dose to members of the critical group by 0.1 mSv only, were considered useless,

- measures that might bring a dose reduction between 0.1 mSv and 0.5 were discussed and led in some cases to recommendations;

- protective measures which might possibly reduce person doses by 0.5 mSv should be adopted with a view of limiting the effective doses to 5 mSv.

The result of the Swiss approach was that there were very few official steps actually taken in Switzerland.

The experience in Finland proved that a restrictive and absolutely consequent line is equally hard to carry out in actual practice. The demand for information on the part of the public was enormous. Many issues were raised in public that were allegedly neglected in the official recommendations. Such issues were reconsidered by STUK and other authorities, and, where appropriate, taken into account in formulating subsequent recommendations. In some cases, where countermeasures with negative net benefits were advocated, it was considered necessary to intervene in the spirit of the basic ICRP principles. Therefore, also explicit non-action advice was included among the recommendations.

13.4.6 Example of cost-benefit evaluation

Use of peat ash for various applications

In the recommendations mentioned in the above, only two intervention levels (10 kBq/kg and 1 kBq/kg) were selected for simplicity. As a base for these intervention levels was a justification and cost-benefit analysis applied on restriction of peat ash utilization after the Chernobyl fallout. This analysis gave the activity levels in peat ash, at which the interruption of peat ash use for vaious applications is justified and optimized. The optimized intervention levels were pretty high, except in use in concrete industry: Cs-137 concentration of 290 kBq/kg in fertilization use in agriculture, 450 kBq/kg in silviculture, 200 kBq/kg in use as landfill material and 1.4 kBq/kg in use as filler in concrete industry. Corresponding committed individual dose equivalents from annual use of peat ash were estimated to be 0.73 mSv, 0.72 mSv, 110 mSv and 3.0 mSv respectively The committed dose from annual use of peat ash as a landfill material, with a Cs-137 concentration of 200 kBq/kg was too high to be acceptable (110 mSv). 5 mSv was considered as an acceptable individual dose from annual use of peat ash, and this lead to the intervention level of about 10 kBq/kg of Cs-137 in peat ash. This was selected in the STUK recommendation. The other intervention level, 1 kBq/kg in the STUK recommendation was got by rounding the intervention level 1.4 kBq/kg of peat ash use in concrete industy to 1 kBq/kg.

13.5 Reduction in radionuclide content during food preparation, project AKTU-295

The project *AKTU-295*, carried out in Finland, has analyzed possible reduction in cesium content in various food products during food preparation. A large number of different foodstuffs have been investigated, like fish, meat, berries, vegetables, potatoes and other root vegetables. A total of 220 samples were examined. One example of the type of results obtained, is that when boiling meat most of the cesium goes over into the liquid part. Similar experiments have also been performed in Norway and Sweden, although not as part of the AKTU program. A cooperative report, presenting the results from all these projects, is being prepared, and will soon be published.

13.5.1 Japanese investigations on spinach and komatsuna

Work of this type has also been performed previously, for instance in Japan (ref. OH84). The Japanese experiments were performed on spinach and komatsuna (a leafy vegetable), and for the contaminants cesium, strontium and iodine. Traditional cooking methods were employed, involving washing for 5 minutes, then boiling in salted water for 5 minutes, and washing again for 5 minutes (to remove harshness).

The reduction observed was largest for cesium; by 86% and 92% respectively for kotsuma and spinach. For strontium the reduction was by 49% and 32% respectively, and for ionic iodine 82% and 70% respectively. For spinach CH_3I and I_2 were also tested, and the reductions were by 60% and 30%.

13.5.2 Finnish experiments on radionuclide reduction, project AKTU-295

When the internal radiation dose has to be restricted in a fallout situation, it is important to know the efficiency of different methods of reducing the contents of radionuclides in food. Foodstuff contaminated by Chernobyl fallout were used to study the effect of both normal cooking methods and processes in the food industry.

Since autumn 1986 a set of cooking experiments has been planned by the Finnish Centre for Radiation and Nuclear Safety and the Work Efficiency Institute. The experiments covered the most usual domestic methods of cooking and conserving and they were performed in 1987-1988 under the project AKTU-295. The reduction of radiocesium during preparation of food was determined (ref. RA89). As early as spring 1986, some experiments were performed with surface-contaminated vegetables and morels. Examination of fish and mushrooms continued in 1988-1989.

Outside the project AKTU, studies of the same type as in Finland were carried out by the National Food Administration in Sweden. In addition to losses of radiocesium and potassium, changes in the contents of vitamin B_6 and sodium were determined in connection with Lapp cooking courses (ref. DA89). Decontamination with substances which can bind radiocesium chemically has also been tested in Norway and Sweden.

13.5.2.1 Material and methods

The experiments comprised cooking the most usual foods on a kitchen stove, e.g. boiling potatoes, root vegetables, meat and fish, and also baking meat and whole fish in an oven. Different accompaniments were also examined such as sauerkraut, sprouts and pickled beetroots. The conserving methods tested were preparing purees and juice of berries and salting fish. The recipes, taken from cookery books and textbooks, were those in general domestic use. The pourpose was to study treatments of importance on acocunt of their effect on the cesium contents, not to prepare complete meals.

The factors probably having an effect on the cesium content of the food were varied. For example, different amounts of water and salt were used in boiling, the boiling time was varied and different types of puree presses and cooking equipment were used. Combinations of these variables were examined with two to four replicate experiments. Samples were taken at different stages of preparation in order to follow the overall change in the cesium and potassium contents as compared with the levels in the raw material (Figure 13.15). Changes in the mass were also checked during preparation.

The homogenized samples were analysed for gamma emitting nuclides, using a low-background semiconductor spectrometer. Potassium was determined via K^{40}.

13.5.2.2 Results

Figure 13.16 shows the results of the Finnish experiments. The diagrams present the percentage of the original mass in the edible food and the cesium-137 and potassium contents as percentages of the original levels. When both the solid and liquid parts of the food are edible, the percentages are given for the solid part.

The methods of cooking that reduced cesium most effectively were salting, boiling in ample water and preparation of juice by steaming berries. Baking meat or fish in the oven or cooking them in a frying pan did not reduce radiocesium, as very little liquid was formed.

Peeling of uncooked potatoes and root vegetables was a stage of preparation in which the losses of radiocesium exceeded those of potassium. In boiled, unpeeled vegetables as well, cesium was still found to be more concentrated in the peels than potassium. During the preparation of puree or juice, the changes in the contents of the two substances were the same and relative to the mass discarded. When juice was made by steaming berries, the ratio of its radiocesium content to the content of the raw material was smaller than the ratio juice/berries. The reduction varied with the species of berries.

Sauerkraut, pickled beetroots and sprouts contained almost the same radiocesium activity as the raw materials. The liquid of sauerkraut contained less than 10 % of the total radiocesium.

The results on mushrooms are shown in Tables 13.15 and 13.16.

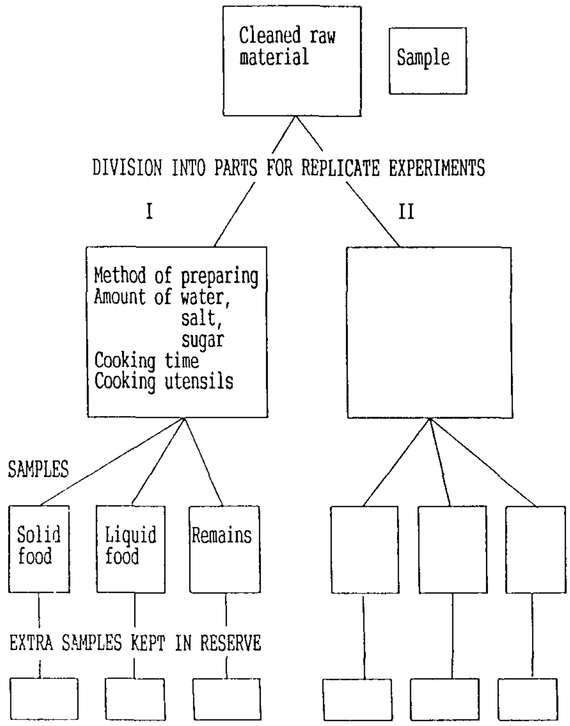
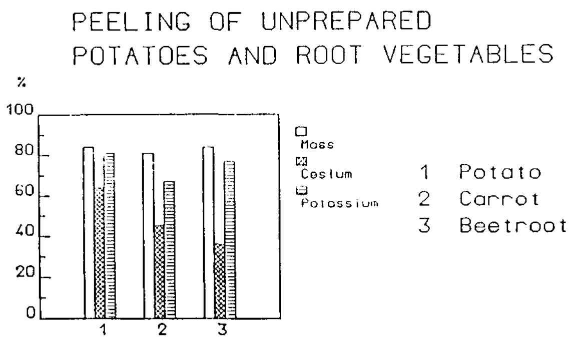


Figure 13.15 Scheme for cooking experiments

Species	Percentage of original amount left in edible part Cs-137	K
Cantharellus cornucopioides	18	21
Cantharellus tubaeformis	15	15
Hydrophorus camarophyllus	16	16
Boletus	20	27
" 1)	9	23

1) Treated in small pieces

Table 13.15 Soaking of dried mushrooms.



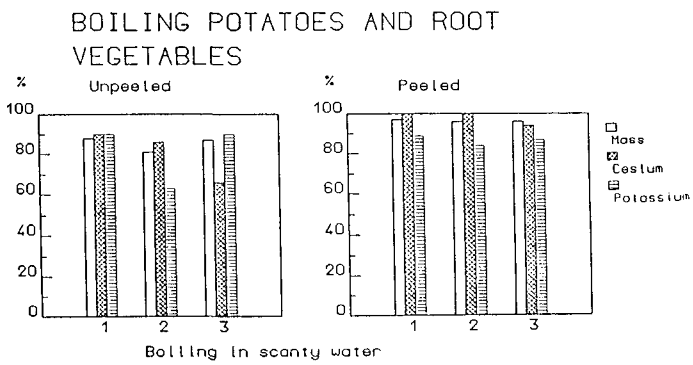


Figure 13.16 Percentage of original mass, Cs-137 and potassium left in edible (solid) part of food.
(Figure is continued next page)

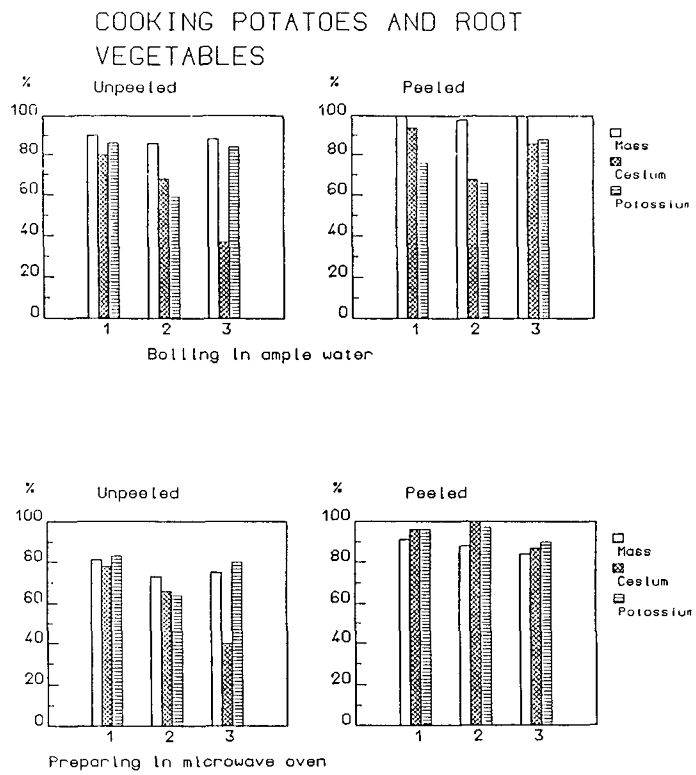
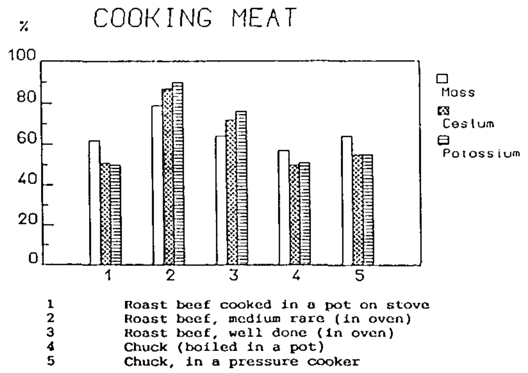


Figure 13.16 Percentage of original mass, Cs-137 and potassium left in edible (solid) part of food.
(Figure is continued next page)



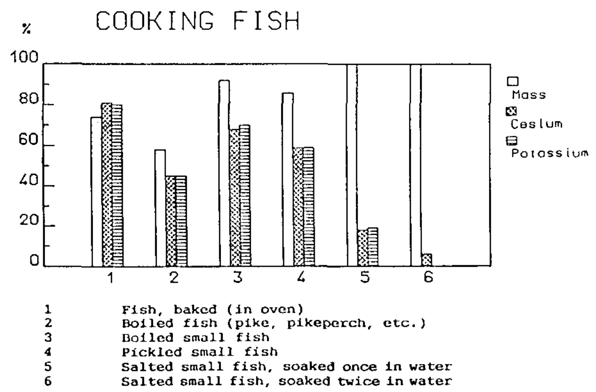


Figure 13.16 Percentage of original mass, Cs-137 and potassium left in edible (solid) part of food.
(Figure is continued next page)

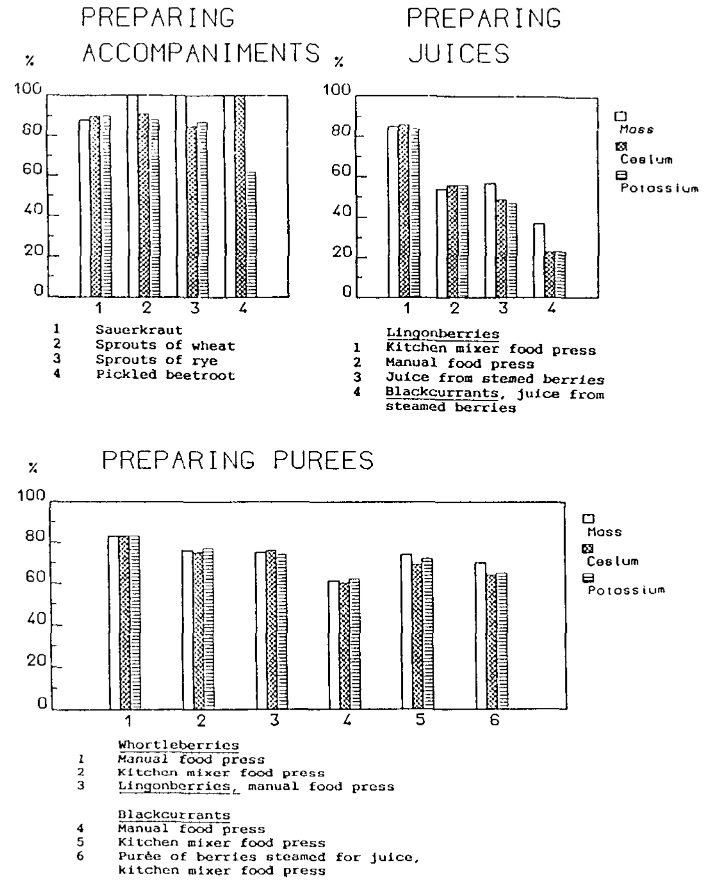


Figure 13.16 Percentage of original mass, Cs-137 and potassium left in edible (solid) part of food.
(Figure continued from preceding pages)

13.5.2.3 Discussion

The Finnish and Swedish results were practically the same, considering the slightly different experimental conditions. In Sweden chiefly the Lapp cooking courses were studied (ref. DA89).

Although salting followed by soaking in unsalted water was the most efficient method of reducing radiocesium in fish, the method is recommended with reservations. Significant losses of potassium, vitamins and water-soluble proteins occur during this treatment. The increased sodium content of the food is also a disadvantage.

The contents of radionuclides in foodstuff are in practice determined both by the amounts of radioactive substances deposited directly on the plants during the growing season and by the amounts of radionuclides available to the plants in the soil. The contributions of these

Species	Percentage of original Cs-137 left in edible part	
	once	twice
Lactarius trivialis 1)	14	
" 1)	18	5.4
"		2.5
Lactarius torminosus	12	
"		2.6
"		1.8
Lactarius rufus		3.2
Lactarius necator 1)	43	21

1) Parboiling started in boiling water (otherwise cold water).

Table 13.16 Parboiling mushrooms.

pathways to the radionuclide contents of food vary with time, as does also the importance of different foodstuff as a source of dietary radiocesium. The need for reduction of the radionuclide contents in foodstuff thus also varies.

During the first year after a new fallout, it is probable that the intake of radiocesium in the Nordic countries occurs predominantly with milk and beef. The contribution of vegetables and cereals to the intake varies according to the season of the fallout. After the first year, there is an increase in the contribution of foodstuff of natural origin, such as freshwater fish, game meat, wild berries and mushrooms. Restrictions of the dose will then affect rather limited groups of consumers.

The amounts of different foodstuff consumed in Denmark, Finland, Norway and Sweden are comparable (ref. OE88). The most distinct differences from the average European-type diet (ref. WH88) are the rather high consumption of milk and relatively low consumption of cereals.

Freshwater fish may be more important in Finland and Sweden than in Denmark and Norway. Mutton is consumed most in Norway, and vegetables are used more in Denmark than elsewhere in the North.

If the fallout is only regional, foodstuff produced outside the fallout area may be used instead of the contaminated foodstuff. Stocks of uncontaminated food may also be available. The most difficult situation is when the basic foodstuff have to be decontaminated before use or during preparation of food. In such case, intake of the necessary nutrients has to be checked. Losses of nutrients connected with reduc-

tion of radionuclide contents can in some cases be compensated for by increased use of less contaminated foodstuff. Efficient instruction of the consumers is needed if private households must take responsibility for the reduction of radionuclide intake on a major scale.

13.5.2.4 The final stage of the project AKTU-295

In addition to normal cooking methods, industrial processing of foodstuff and methods of decontamination of meat and fish have been studied in the Nordic countries since 1986. In Norway a boiling method is being examined, in which Prussian Blue is used as a cesium-binding agent (ref. BE89). Boiling with zeolites has been studied in Sweden (ref. FO89a). A short description is found in the following subchapter. For more information on the Norwegian work, the reader is referred to the references given. A rapid salting procedure for meat and fish has been developed in Finland (ref. PE88 and PE89). Reduction of radiocesium in meat by salting has also been studied in Norway (ref. BE88).

A general review paper is under preparation on all the Nordic studies dealing with processing and decontamination of food performed after spring 1986.

13.5.3 Use of zeolites for reduction of the content of radiocesium in meat and fish at cooking

The Swedish National Institute of Radiation Protection (SSI) coordinates extensive research in the same areas as those of concern within the AKTU program. Short summaries of these related projects have been prepared by SSI for inclusion in the present report. These projects are not part of the AKTU program.

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Mordenite, one of the zeolites, has been used for reduction of radiocesium in meat and fish in the cooking. Both artificial and natural zeolites have been tried, yielding very similar results.

In finely divided meat the content of radio-cesium has been reduced to about one tenth after boiling for 1/2 hour with 10 g zeolite per 100 g of meat. The cesium content of the broth is reduced by the zeolite. It can be further reduced by continued boiling with the zeolite after removal of the meat.

According to the experiments the total residue of radio-cesium in meat and broth can be reduced to about 1/10 of its original value by the use of moderate amounts of zeolite, probably without any negative effect to the resulting dish.

With the dosage of zeolite mentioned above and a price of 5 SEK/kg for natural mordenite, this way of reducing the radio-cesium intake can be performed at a cost for the zeolite of 0.50 SEK/kg of meat.

14. ECONOMIC CONSEQUENCES IN EARLIER ACCIDENT ANALYSES, PROJECT AKTU-285

The economic consequences calculated as part of an accident consequence assessment are usually only the cost of the mitigating actions. It was in the mid-70s that it became customary to perform these calculations of economic consequences, usually following the concepts of the US Reactor Safety Study (ref. WA75). Often even the values of the economic parameters (cost of agricultural products, land value etc) were also copied directly from this reference, even though this was in many instances quite inadequate.

The overall impact of an accident in monetary terms would include costs incurred both on and off the nuclear power plant site. Although on-site costs, which include those associated with plant clean up, replacement power and capital costs, may dominate the total cost, especially of small releases, only off-site costs are considered in this chapter.

In 1976 a Government Commission on Nuclear Power was appointed in Norway. The Institutt for energiteknikk (then the Institute for Atomic Energy) was one of the main consultants for the Commission, and one of the tasks performed by the Institute was performance of an accident consequence assessment for a site for a nuclear power station in Norway. The calculations were performed with the CRAC computer program, the program which was also used in the US Reactor Safety Study. But it soon became evident that the economic parameter values used in the American calculations were not representative of Norwegian conditions. More appropriate values were sought, and for many of the parameters found. But in some cases it was probable that the answer found (and also the values used in the American calculations), did not reflect the whole truth. Economic inter-relationships are very complicated.

When the AKTU program was initiated in 1985, project *AKTU-285* had as objective to determine more appropriate economic parameter values for Nordic conditions. The natural starting point was to assemble values used in some relatively recent assessments. This was done, and all values adjusted to the monetary value of year 1985, in order to be comparable.

The plan was then to try to take a more "complete" approach to the economic consequences than had been done till then. A reactor accident will also have economic consequences in the form of social (e.g. the social effects of unemployment and moving away from homes might be anxiety and possibly civil disobedience), political (for instance loss of confidence in the government and the acceptability of a nuclear power program) and environmental effects (radiation exposure of the biosphere may affect other organisms besides man). It is extremely difficult and perhaps impossible to convert these effects into monetary units.

A more achievable improvement of economic consequence assessment would, however, be to take links between the different sectors of the economy into consideration. This has not been taken into account in most reactor accident consequence assessments. Some attempts at performing more realistic assessments of economic consequences along these lines have been performed.

For instance, the US Bureau of Economic Analysis (BEA) has developed an economic cost model, the regional industrial multiplier system (RIMS) based on economic production. RIMS is an established technique for estimating the effect on a regional economy of a change in demand for goods in one sector of the economy (ref. US82). A change in one industrial sector inevitably affects the entire economy and each sector will adjust to achieve a new equilibrium in the region. This method can also be modified to predict the economic effects of input shortages in specific economic sectors. For reactor accident impacts the regions under consideration are divided into "physically affected" and "physically unaffected" areas. Estimates of the amount of output lost are needed for the analysis. The model can then calculate the effects of compensating adjustments within the economy. However, the economic effects are only calculated for the first year after the accident.

The BEA model is unsuitable for direct use in accident consequence assessments because it requires a very detailed data base and a great deal of computational effort. In addition, the resolution of the areas for which the calculations are made are fairly coarse, and more importantly, only the costs in the first year are considered. However, despite these disadvantages, the complex RIMS model has uses in accident consequence assessments, since it can be used to test the results of the simpler models in the first year following the release. One example of this is that RIMS can calculate costs in areas which are physically unaffected by the release, but which may be affected by changes in the economic climate within the contaminated area. Results from RIMS indicate that costs in the physically unaffected areas are relatively small compared with those in the physically affected area. Since neither of the simpler models referred to elsewhere in this chapter include costs outside the affected area, this indicates that such an omission is not unreasonable.

In a report from the United Kingdom National Radiological Protection Board (NRPB) (ref. CL82 and DI84) another interesting approach is taken. The liberty is taken here of quoting a considerable portion of the method description from the first of these references, since it is well written, and certainly gives increased understanding of the problems invovled.

The basic assumption of the model is that the cost of countermeasures will be a function of the contribution to Gross Domestic Product (GDP) from the area prior to its contamination. This contribution will be assumed lost when countermeasures involving relocation are necessary or when a food ban is imposed. GDP is a measure of output which can be assessed directly, in terms of output, as income or as expenditure. It is a standard measure used in National Accounts Statistics and is described in detail in literature.

The concept of GDP can be explained using very simple models of domestic economies. For example, consider a closed economy totally concerned with the production and consumption of bread. Farmers grow wheat, millers produce flour, and bakers bake and sell bread. No other goods or services enter or leave the economy. Suppose the farmers sell their grain to the millers for 1000 pounds, the millers produce flour and sell it to the bakers at 2000 pounds and finally, the bakers bake bread and sell it to the consuming public who, in this case, are themselves, the millers and farmers, for 3000 pounds.

In terms of income, the GDP of this economy is 3000 pounds, or 1000 pounds each for farmers, millers and bakers. The value of the final output is also 3000 pounds because 1000 pounds is added at each transaction. Finally the total expenditure on the final product, bread (the wheat and flour are classified as intermediate products), is 3000 pounds. Hence the GDP is 3000 pounds, however assessed.

In an analogous manner, the GDP of the UK economy can be assessed in terms of income, output or expenditure, although the precise methods and practice of national accounting are necessarily complex.

The total income approach measures the sum of all the incomes of UK residents, earned in the production of goods and services in the UK. This income is usually divided into the classifications of employment, self-employment, profits and rent. These are incomes earned from production; they do not include transfer payments such as pensions and sickness benefits and, because UK domestic produce is being assessed, incomes from investments abroad are excluded. (These latter incomes are included in Gross National Product (GNP).)

GDP may be measured from the production side by either summing the value added by the production of various firms and public enterprices in the country, or by simply taking the value of final production, net of intermediate products to avoid double counting. As noted in the simple illustration above, wheat and flour are intermediate products in the production of bread. Also, if a firm uses imported goods the values of these are subtracted from the value of the product as the objective is to assess the domestic product.

GDP is measured from expenditure statistics by summing the spending in four main categories - consumer spending, public authority spending, spending by firms and spending by foreign nationals in the UK. In theory, GDP measured by this route should give the same result as that derived from income or production, but in practice there are small residual errors arising from the use of different sources of statistics.

Whichever route is used to assess GDP, it is simply a measure of economic activity derived from national statistics. While a rise in GDP is usually taken as an indicator of a rise in overall living standards in a country, the statistics do not include all activities having an economic benefit, and do not take into consideration many real factors which can add to or detract from the quality of life. A loss of GDP cannot therefore be used as a measure of the total cost of countermeasure; it is an estimate of the loss of production; there will be other costs.

The contribution made to GDP in various parts of the UK is not uniform, and therefore relocation will be more costly for some locations than others. In addition, a projected loss of GDP may not be permanent. For example, the loss of production by a factory in a contaminated region could be substituted either by factories elsewhere, or by subsequent investment in a new factory in a different region. Such readjustment can explain why, even after natural disasters or war, the GDP of countries can regain previous or higher rates of growth. The use of GDP as a measure of loss could therefore be an overestimate if the loss was assumed to be constant over time. On the other hand, a projected loss of GDP may only reflect a potential loss of value of final production, and not directly account for losses of intermediate production outside an affected region. For example, steel being supplied from elsewhere to an industry in a contaminated region as an intermediate product, may not have an immediate market elsewhere in the economy. Similarly a farmer may lose more than just the value added by his production (i.e. his net income) if he has already invested in seed and fertilizer, or in animals and their feed, and then is unable to sell his product due to a food ban. Hence, in the short term GDP could underestimate potential losses, but in the longer term it could be an overestimate.

14.1 Sets of economic parameters compared

The values of the economic parameters used in the following reactor accident assessments or reports upon the economic aspects of nuclear accidents have been compared, and are presented in subchapter 14.3. Although this chapter contains much details, it is thought useful to have such a compilation available.

- US Reactor Safety Study (ref. WA75) (called WA in the tables in the following subchapters).
- Norwegian Government Commission on Nuclear Power (ref. NO78) (called NO in the tables). (Although the year of publishing is 1978, the values are for year 1977).
- Sandia National Laboratory (ref. BU84 and OS85) (called SA in the tables). (Although the years of publishing are 1984 and 1985, the values are for year 1982).
- National Radiological Protection Board (ref. CL82) (called NR in the tables). (Although the year of publishing is 1982, the values are for year 1978).

14.2 Adjustment of monetary values

All costs were converted to monetary value of the year 1985 (except when specifically stated to be otherwise), using the Norwegian consumers price index, from the Central Office of Statistics. The values of the index on the 15 May of the relevant years are:

1975: 72.9	1982: 138.2
1977: 87.6	1985: 169.9
1978: 94.3	

The parameters also had to be converted to the same currency, to ease comparison. Norwegian crowns (NOK) was chosen. The other currencies involved were US dollars and British pounds. The British pound had been relatively stable relative to NOK over the time period concerned. US dollars, however, had fluctuated from 1$ = 5 NOK to 1$ = 10 NOK.

The exchange rates in 1985 were 1$ = 8 NOK and 1 pound = 10 NOK roughly, and these exchange rates are used throughout this chapter.

The values of economic parameters used in the Norwegian report are a mixture of values actually assessed for Norwegian conditions and values taken directly from the US Reactor Safety Study. The latter are labeled "As WA" in the following subchapters.

14.3 Values of economic parameters

As mentioned in the above, the first task performed within project *AKTU-285* was compilation of the values of economic parameters used in several previous accident consequence assessments. This compilation was never published. The Chernobyl accident happened, and the project took a completely different direction from then on. This compilation contains much details, but as a similar compilation does not exist elsewhere, to our knowledge, it is thought useful to include it here in extenso, in spite of the level of detail.

The table of values of economic parameters is not complete. Some values were not to be found in the reports, and would have to be obtained through personal contact. The values from NRPB are not directly comparable, as a different approach to assessment of the economic consequences has been adopted.

	WA	NO	SA	NRPB
Evacuation (NOK/evacuee-day)	260.-	As WA	117.- to 246.-	
Agricultural product interdiction(NOK/dekar)	50.- to 1,900.-	2,900.-	No numerical information	
Yearly consumption of non-diary products (NOK/person)	4,500.-	Not used	No numerical information	
Yearly consumption of diary products (NOK/person)	690.-	Not used	No numerical information	
Decontamination cost, resident., business and public areas (NOK/person)	32,000.-	As WA	73,000.-	
Decontamination cost farm-land (NOK/dekar)	1,100.-	As WA	1200.-[1]	
Depreciation rate (per year)	0.2	0.2	0.2	0.05

[1] This may be 1984 dollars, but it is assumed here that it is 1982 dollars.

Table continued next page.

	WA	NO	SA	NRPB
Relocation cost (loss of personal and corporate income, and moving costs) (NOK/person)	54,000.-	As WA	37,000.-	43,000.-[2]
Value of residential, business and public areas (NOK/pers)	330,000.-	200,000.-	No numerical information	
Value of farm areas (NOK/dekar)	300.- to 10,000.-	6,800.-	No numerical information	

[2] Excluding moving costs.

14.3.1 Detailed economic information on short-term mitigating actions

Sheltering

SA It is assumed that there is only negligible cost.

NR It is assumed that there is only negligible cost.

Evacuation

WA Cost of food, lodging and transportation (1 week duration) is NOK 1,800.-/evacuee = NOK 260.-/evacuee-day.

NO Direct conversion from WASH-1400, using 1$ = 5 NOK.

NR "A few tens of pounds" roughly equal to NOK 200.-/evacuee. It is assumed that the evacuation cost is equivalent to relocation cost, and is the contribution to the GDP per person for the time period involved (contribution to the commercial and industrial sector of the economy).

SA National average per-capita personal plus corporate profits and interests is estimated at NOK 260/person-day.

SA Ratio of region-specific to national average personal incomes, is not quantified in the report.

SA Contains references to evacuation data, and TMI evacuation data.

SA Cost of food, lodging and transportation for each evacuee is given in the following table. (Are expressed in 1982- Norwegian Kroner)

Commercial Care Facilities:	
Housing	NOK 170,-
Food	" 52,-
Transportation (Private)	" 24,-
	NOK 246,-/evacuee-day
Mass Care Facilities:	
Housing	NOK 68,-
Food	" 36,-
Transportation (Mass)	" 13,-
	NOK 117,-/evacuee-day
Evacuation Personell (~2% of total # of evacuees):	
Compensation	NOK 570,-/day
Food, Housing, and Transportation	Same as evacuees
Total Weighted Cost - (E) (Based on 80% commercial care, 20% mass care facilities)	=NOK 230,-/evacuee-day

NR According to NRPB compensation costs (evacuation must belong in this group) do not represent a loss to society, as it is simply transfer of funds from one part of the economy to another.

Bastien, M.C., Dumas, M., Laporte, J. and Parmentier, N. (to be published in Risk Analysis, at the time when this compilation of information was first prepared; 1985/86) reports about the risk of road accidents in evacuation situations. It appears from experience from evacuations (vulcanic eruptions, hazardous chemicals, tornadoes etc.) involving 1.5 million persons, that the risk during evacuations is at least a factor ten lower than normal traffic statistics both for fatalities and injuries.

Emergency phase relocation

The economic parameters are the same as for evacuation.

Intermediate phase relocation

The difference between emergency phase and intermediate phase relocation is rather hazy. The economic parameters are as for evacuation.

14.3.2 Detailed economic information on long-term mitigating actions

Agricultural product interdiction

SA No numerical information.

WA Yearly production of total agricultural products varies very much, from NOK 50.-/dekar (New Mexico and Nevada) to NOK 1,900.-/dekar (Connecticut). The fraction which is dairy production varies from 0.022 (Wyoming) to 0.811 (Vermont), but it is not clearly stated whether this is by area or value.

WA Yearly consumption of non-diary products NOK 4,500.-/person. Yearly consumption of diary products NOK 690.-/person. These two are used in a rather unfortunate part of the program, which is used in order to calculate the number of persons exposed via milk and via other agricultural products. E.g. subsidies may throw this entirely off balance. This part of the program is replaced in the Nordic version with a part based upon number of persons that can receive supplies from one unit of area.

NO Yearly production of total agricultural products is 2,900.-/dekar.

NR Annual contribution to GDP by agriculture is determined from census data on a 5x5 km grid. National statistics gives annual contribution per animal and per hectare of crops.

Decontamination of land and property

SA: Decontamination cost and effectiveness values for farm areas are given in the following table.

Dose Rate Reduction Factor After Decontamination	Approximate Costs (NOK/dekar)	Fraction of Cost for Paid Labor	Worker Dose Reduction Factor (Estimated Worker Dose/Dose from Continuous Exposure)
3	390	.30	.10
15	1100	.35	.25
20	1200	.35	.33

SA: Decontamination cost and effectiveness values for non- farm areas are given in the following table.

Dose Rate Reduction Factor After Decontamination	Approximate Costs (NOK/person)	Fraction of Cost for Paid Labor	Worker Dose Reduction Factor (Estimated Worker Dose/Dose from Continuous Exposure)
3	26 000	.7	.33
15	68 000	.5	.33
20	73 000	.5	.33

SA Labor cost is based upon military personnel.

WA Decontamination cost residential, business and public areas is NOK 32,000.-/person.

WA Decontamination cost farm-land is NOK 1100.-/dekar.

NO All decontamination costs are equal to the ones from WASH-1400, but the exchange rate 1$=5NOK was used to convert the costs into Norwegian Croner in the report for the Government Commission.

NR No decontamination cost included. Assumed only natural decontamination of a factor three.

Land area interdiction

SA Here is used the concept of wealth to estimate the total present value of land and tangible assets in an area.

SA Rather few numerical values given. Depreciation rate is 0.2/year for improvements both in farm and non-farm areas. Farm land and structures values are available in an Agricultural Cencus from 1978. Bureau of Economic Analysis (BEA) within the U.S. Department of Commerce is in the process of completing a study to estimate the total tangible wealth of the U.S.. Permanent relocation costs from personal income losses for a 100 day transition period and corporate income losses for a 180 day period is about NOK 37,000.- and is small compared to the wealth loss predictions.

WA Personal income losses for a 90 day transition period about NOK 21,000.-/person. Corporate income losses for a 6 month transition period is NOK 17,000.-/person. Moving costs, sum of personal, business and public (the last two 10% of value), is NOK 16,000.- per. person. In the calculations are used a total relocation cost of NOK 54,000.-, or the sum of the above.

WA Depriciation rate 0.2. Value of residential, business and public areas NOK 330,000.-/person. Value of farm areas varies from NOK 300.-/dekar (New Mexico) to NOK 10,000.-/dekar (New Jersey). (1 dekar = 1000 m^2). 75% of this is the value of the land itself.

NO Depreciation rate 0.2. Value of residential, business and public areas NOK 200,000.-/person. Value of farm areas NOK 6,800.-/dekar. NOK 4,800.-/dekar of this is the value of the land itself.

NR ECONO-MARC uses as a measure of loss the lost contribution to the nation's Gross Domestic Product. This is a standard quantity used in economics and simply measures the value added by all sections of the economy.

NR Annual GDP contribution by housing is based upon rent or mortgage payments. A computerized data base of housing stock in each 1 km square of the UK is available from census data.

NR The annual contribution to GDP by commerce and industry can be determined in two ways. One is upon census data in 1x1 km grid, giving the land usage (bottom lefthand corner represents the whole square). The other is by contribution to GDP per person. The latter is used, although the census data refers to where persons live, and some might work in the area and live elsewhere.

Health effects cost

SA Takes into account the direct health effects costs: Medical care costs and human capital costs (life-time loss of earnings).

15. ECONOMIC CONSEQUENCES OF CHERNOBYL IN THE NORDIC COUNTRIES

The following subchapters contain information on the economic consequences in Norway, Sweden and Finland respectively. There were no significant economic consequences of the Chernobyl fall-out in Denmark. The types of consequences included in the surveys differ somewhat between the countries. The Swedish data are the most complete, including e.g. the costs of various Chernobyl-related measurement and information tasks, while the Norwegian data are more concentrated directly on the cost to the agricultural community, and excluding costs related to work performed by e.g. staff from the Ministry of Agriculture.

All three surveys were partially funded via project *AKTU-285*.

15.1 The cost of mitigating actions in Norway after Chernobyl

The cost of mitigating actions carried out in Norway in the years following the Chernobyl accident are described in (ref. TV87b, TV88b, TV88c and ST89). The Department of Agriculture in Norway has the intention of continuing to publish similar reports summing up the situation each year as long as the Chernobyl fall-out constitutes a problem in Norway, in cooperation with Institutt for energiteknikk. The collection of information concerning cost of the mitigating actions, presented in this subchapter, has primarily been performed by the Department of Agriculture, and preparation of the reports has primarily been performed at the Institutt for energiteknikk, Norway, as part of project *AKTU-285*.

The Chernobyl accident created serious problems for various parts of Norwegian agriculture; in particular in connection with sheep, goats, reindeer and freshwater fish.

In this chapter is summarized the economic consequences of the deposited radioactive materials to Norwegian agriculture in the years 1986, 1987 and 1988, or more correctly in the slaugthering periods 86/87, 87/88 and 88/89.

The current action levels are 370 Bq/kg in milk and baby-food, and 600 Bq/kg for all other foodstuffs; except for meat from domestic reindeer, game and freshwater fish, where the level is 6000 Bq/kg.

The economic loss in the time soon after the Chernobyl accident was limited to lettuce from a specific area in Mid-Norway, where this was grown outdoors, even though it was very early in the season. It was decided by the authorities that this lettuce should be destroyed. The value is reported to be roughly NOK 300,000 ($ 45,000) by the Department of Agriculture. The cost may, however, have been larger, as local farmers complained that it was difficult to sell any type of agricultural product from this area, at least in the early part of summer, because of general fear of radiation. There has been no similar loss in 1987 or 1988.

Concerning cow´s milk, cattle had not yet started feeding outdoors at the time of the accident. Later, when cattle were brough up to the mountain pastures, the cesium levels in milk increased, but exceeded the action level only in one limited geographical area, where the milk was interdicted. In 1987 and 1988 the levels were somewhat similar.

Although down-feeding (feeding with cesium-free or -low fodder) had to be carried out in some areas, in some cases including administration of bentonite; cows milk was not interdicted.

In goats milk the cesium content increased from mid-June 1986, when the goats were brought to the mountain grazing areas, but the content rarely exceeded the action level. However, goats milk is mainly utilized in the production of a special and very popular type of brown cheese, in which the content will be much higher than in the milk. It was therefore decided to use goats milk from certain areas as fodder instead of in cheese production. The production of brown goat cheese has been affected even in 1987 and 1988 in the same manner as in 1986, but to a somewhat smaller extent, because of special measures taken. Loss of production of goats cheese amounts to a value of NOK 10 mill. ($ 1.4 mill.) in 1986 (the value of the interdicted cows milk is included here), and NOK 7 mill. ($ 1 mill.) in 1987, after taking into consideration the savings because the goats milk was used as animal fodder.

The levels in butter and ordinary cheese have been consistently low. Pork and poultry are primarily fed with grain products, and have accordingly also very low content of radioactive cesium.

Although the levels in salt-water fish have not been higher than normal, the levels in wild fresh water fish have in some cases been quite high, and levels up to 60,000 Bq/kg have been reported. Summer 1986 the Health Directorate decided to forbid sale of fresh-water fish from altogether 37 municipalities. The levels in Spring 1987 had not decreased significantly compared to the levels measured in Fall 1986, and actually rose to a peak in June 1987 (although lower than the peak in 1986). Fresh-water fish is not important commercially, and the economic consequence has not been estimated. No compensation has been offered to fresh-water fishermen.

Sheep has turned out to be one of the largest problems. Between two and three thousand tonnes of mutton was interdicted in 1986, representing a value of roughly 100 mill. NOK ($ 14 mill.). The down-feeding program cost 4 NOK/animal-day. The total cost of this mitigating action is estimated to 30 to 35 mill. NOK ($ 5 mill.), and involved ca. 320,000 animals. The sale of mutton turned out to be 10% below normal, which was equivalent to the amount that was interdicted. This indicates that the consumers generally felt that the protective measures adopted were adequate. In 1987 it was estimated that only about 8000 animals were from the areas with content exceeding 5000 Bq/kg, and only 73 tonnes of mutton was interdicted, representing a value of roughly NOK 2.5 mill ($ 350,000). The down-feeding program had the same basic cost as in the slaughtering season 86/87 (4 NOK/animal-day), but the compensations was NOK 8 NOK/animal-day if the animals had to be moved. Generally the animals were moved within the same district, and mostly they were moved from areas in the mountains to areas in the valleys. The total cost of the down-feeding program was roughly NOK 40 mill. ($ 6 mill.), and involved ca. 280,000 sheep. In addition there were costs of control and surveillance (including instruments), and administration of the program; estimated at NOK 13 mill. ($ 1.8 mill.). Because 1988 was an exceptionally good year for mushrooms, one observed levels of radioactive cesium three to four times the levels of the previous year in many parts of the country. Roughly 30% of the sheep were in special measures zones in 1988, and 70% in free zones. The down-feeding program covered a total of ca. 360,000 sheep and cost NOK 60 mill. ($ 8.5 mill.) in 1988, and in addition there were costs of control and surveillance of the same order of magnitude as in 1987.

The value of interdicted reindeer meat (about 560 tonnes) in 1986 was roughly NOK 20 mill. ($ 2.9 mill.). This was after the authorities decided, in November 1986, to raise the raised action level on domestic reindeer meat from 600 Bq/kg to 6000 Bq/kg. The cost of the whole program related to control and surveillance of reindeer etc. was roughly an additional NOK 8 mill. ($ 1.1 mill.). The value of interdicted reindeer meat (216 tonnes) in 1987 was roughly NOK 8.5 mill. ($ 1.2 mill.), and the cost of the whole program related to control and surveillance of reindeer was roughly an additional NOK 10 mill. ($ 1.4 mill.). This includes a special compensation given when early slaughtering is carried out, of NOK 5 and NOK 2 per kilo in addition to the regular price of the meat when slaughtering was carried out before 20 October and 27 November respectively. In 1988 the value of the interdicted reindeer meat was roughly NOK 5 mill. ($ 0.7 mill.).

The cost of the mitigating actions to obtain content in beef below the action level was roughly NOK 5 mill. ($ 710,000) in 1986, but the costs in 1987 were insignificant. Levels up to 3000 and 6000 Bq/kg respectively were measured in these two years.

No compensations have been offered to game hunters or fresh-water fishermen, although claims have been sent to the Department of the Environment.

It is very difficult to estimate the economic consequences of an accident like Chernobyl, e.g. concerning tourist trade or boycot of products from specific areas of high contamination, even if the content in the product is below the action levels. Most of the costs quoted in this report are costs of the product to the producer and direct costs of down-feeding programs. Otherwise, however, the major portion of the costs of research activities, planning and administration of the programs, and the measurement programs in Norway are not included in the costs quoted in this subchapter.

15.2 The cost of mitigating actions in Sweden after Chernobyl

The purpose of this chapter is to summarize the economic consequences in Sweden of the Chernobyl accident in the years 1986/87. The summary has been prepared by Christina Gyllander on behalf of the Swedish National Institute of Radiation Protection, partially financed by AKTU project funds. The material is collected from the following references: (ref. AG86, AG89, BA89, CO89a, CO89b, CO89c, CO89d, FA86, FO89b, NA89, OS89, RI89, SA89, SW86b, SW88a, SW88b, SW89 and TE89).

During May 1986 the Swedish government decided that compensation should be paid for products from agriculture and horticulture, products that will not be allowed for sale because the activity content exceeds the applied action level.

In June 1986 this decision was extended to include reindeer breeding, in September 1986 even game (moose).

In this financial summary the economic consequences are specified for the departments involved:

- The Ministry of Defence
- The Ministry of Agriculture
- The Ministry of Industry
- The Ministry of Environment and Energy

15.2.1 Economic impact

15.2.1.1 The Ministry of Defence

By an act of Parliament of 17 December 1986 the Swedish National Rescue Services Board has been assigned 11.5 MSEK for improvement of the mitigation actions prepared and for preparation and publication of a handbook containing the information necessary.

Measurements performed by the Swedish Air Force have been included in ordinary activities and thus without extra cost.

Measurements made by the National Defence Research Establishment (FOA), Umeå, covering a large part of the country, have been paid by the National Institute of Radiation Protection.

For activities within the framework of the Ministry of Defence no additional costs caused by the Chernobyl accident have arisen.

15.2.1.2 The Ministry of Agriculture

Mitigating actions caused by the Chernobyl fallout have been taken, considering the the following

- agriculture and horticulture	218.7 MSEK
- reindeer breeding	137.6 "
- fish	4.3 "
- game (moose)	6.4 "

The total cost of these mitigation actions during the years 1986 and 1987 (up to 88-02-25) has been 367 MSEK. 1.5 million liters of milk from cows released to grazing before the areas were classified as free, had to be discarded, corresponding to a value of 4 MSEK.

<u>Reindeer breeding</u>

Total down-feeding included < 2000 reindeer.

Supplementary down-feeding: 16 villages were offered this mitigation action including additional feeding with fodder enriched by potassium chloride and bentonite (up to May 87). This possibility was used to a very great extent. 7 villages within the most affected area have been offered transportation of maximum 15000 reindeers to only slightly contaminated areas within the district Dalarna (Dalecarlia). The government has paid SEK 700 per reindeer approved. Two villages have accepted this offer and transferred 12000 reindeers to other grazing areas. In connection with the budget proposition 1987 the Parliament accepted an increase of the subsidy to the reindeer breeding with 4.7 MSEK.

Fish

Up to January 1988 an amount of about 79000 SEK has been granted to professional fishermen.

Game

Compensation has been allowed with SEK 2200 for a full-grown moose, SEK 1000 for a calf. The total value for 4060 animals was 6.4 MSEK.

Berries and mushrooms

Up to February 1988 the county administrations have granted 4.2 MSEK.

15.2.1.3 The Swedish National Food Administration:

Analyses performed by external laboratories: 3 MSEK.

Internal work by the staff (estimated costs) comprising specification of basic material intended for decision making: 5-10 MSEK.

Information: about 800-1000 copies containing actual advice and instructions have been released each week.

15.2.1.4 The Ministry of Industry

After acceptance by the Parliament the government issued a government decision (1986:1424) regarding compensation to enterprises, covering additional costs and losses on account of the impact of the Chernobyl accident, arising by the ban or restrictions put on their basic products.

Compensation has been granted to about 190 enterprises with about 55-60 MSEK. Most of these funds have been used as a support to the industries related to reindeer breeding.

15.2.1.5 The Ministry of Environment and Energy

The National Institute of Radiation Protection has received 38.968 MSEK: 39 MSEK. Internal work by the staff (estimated costs): 10 MSEK. These funds should be disposed of for mitigation actions specified below:

- training, measuring techniques
- measurement devices, acquisition costs
- automatization of 25 gamma measuring stations
- preparation and distribution of information
- research and development

15.2.2 Summary

15.2.2.1 Direct costs

The numerical information from the preceding subchapter is summarized below:

	MSEK (for each item)	MSEK (summed over groups of items)
The Ministry of Defence:		11.5
The Ministry of Agriculture:		
- Agriculture and horticulture	218.7	
- Reindeer breeding	137.6	
- Fish	4.3	
- Game	6.4	367
Further costs are expected to arise during the following years regarding reindeer meat and fresh-water fish, as the action levels might be exceeded for some time.		
The Swedish National Food Administration:		
- analyses, external work	3	
- estimated costs: internal work by staff	5-10	8-13
The Ministry of Industry:		
- compensation to about 190 enterprises, most of which to reindeer breeding		55-60
The Ministry of Environment and Energy:		
(The National Institute of Radiation Protection)		
- additional funds for specified actions	39	
- estimated costs: internal work by staff	10	49
(The additional funds should be used for:		
- investment in measuring device		
- training		
- information		
- research and development)		
Summed over all items of direct costs:	491-501	

15.2.2.2 Additional indirect costs, estimated

To the costs specified above additional indirect costs might be added. These will of course be extremely difficult to quantify even on a very uncertain basis. According to people deeply involved in different activities it cannot be excluded, however, that these costs will amount to the following:

	MSEK (for each item)	MSEK (summed over groups of items)	
Reduction of tourist trade to Sweden		400-500	(ref. SW89)
Consumers´ resistance			
Fresh-water fish reduction:			
- commercial fishing (licences)	7.5		(ref. SW86b)
- private fishing: All counties affected by radioactive fallout (5.5 MSEK of this in the counties Jämtland and Väster-Norrland)	15.5		(ref. SW86b)
- wild berries, excl. transport costs, estimated approximately to	100		(ref. AG89)
- mushrooms	10-15		(ref. AG89)
- horticulture, private consumption:			
vegetables	0.4		(ref. AG86
berries	7.2	133-138	and FA86)
Internal staff work			
At the County Administrative Boards: Total costs assessed to the order of (3.5 to 4 MSEK of this at the County Adminstrative Boards of the following counties:	8		
- Västerbotten			(ref. CO89a)
- Västernorrland			(ref. CO89b)
- Gävleborg			(ref. CO89c)
- Jämtland)			(ref. CO89d)
- The National Board of Agriculture	16	24	(ref. NA89)

To this will be added other costs not yet quantified, for example costs for data evaluation and analyses and information actions to the public performed by almost all urban and rural districts.

Nuclear industry

Extra costs for participating in measurements and information actions:

- Barsebäck	no costs		(ref. BA89)
- Forsmark	1-1.5		(ref. FO89b)
- OKG	0.25		(ref. OS89)
- Ringhals	0.318		(ref. RI89)
- RKS	1-1.5	2.6-3.6	(ref. OS89)

Summed over all items of indirect costs: 557-663

15.2.2.3 Total costs

	Million Swedish Kroner
Direct costs	491-501
Indirect costs	557-663
	1048-1164

15.3 The cost of mitigating actions in Finland after Chernobyl

This chapter should be read in context with chapter 13.4, which deals with the actual mitigating actions in Finland.

15.3.1 Costs of intensified surveillance

Decisions to intensify surveillance are not protective actions as such, and do not lend themselves to be scrutinized by cost-benefit analyses. If the fallout situation is not known, the full use of relevant measurement capacity must be regarded as cost-effective by definition. All the same, decisions about surveillance programs can have considerable economic consequences.

STUK used a special code for Chernobyl-related costs in its bookkeeping. The direct costs of radiation monitoring and laboratory analyses during the first year after the accident was close to one million FIM, including travels, laboratory and other materials and services purchased. The indirect costs, including salaries for normal work and overtime added 7.5 million FIM.

15.3.2 Costs related to agriculture and food production

Due to the well-developed monitoring of environmental radioactivity and domestic foods the authorities had enough data for reasonable decision making. The selective and less rigorous policy in fixing the derived intervention levels contributed to the cool public reactions, which never developed to the stage of strong concern. As it was mentioned earlier the producers of fresh lettuce suffered from the negative publicity in May 1986. It has been estimated that the total losses of these producers amounted to 2.5 - 3 million FIM. However, this is only an estimate, since the producers never made any claim for an economic compensation for their losses.

More generally, it can be stated that very few rejections of domestic agricultural products were made. In fact the Finnish government paid economic compensation only for reindeer meat, which was withdrawn due to high concentration of radioactive cesium in meat.

The reindeer breeding area covers the northern part of Finland, which was not seriously affected by the Chernobyl fallout. However, due to the fallout from the nuclear tests in the 1960s and sporadic contamination of lichen pastures, the radioactivity of reindeer meat became a problem in the fall of 1986. The cesium levels in reindeer meat began to go up as soon as the reindeer began to feed on lichen, i.e. mid-September 1986. However, there were considerable variations in the cesium concentration, which necessitated an extensive monitoring of reindeer activity. The strongly intensified monitoring was, however, the only measure which seemed to be necessary in the existing situation. By the time the reindeer slaughter season started, i.e. mid-October 1986, it became evident that a limited area in the southeast part of the reindeer breeding area was heavily exposed. Carcasses from that area were individually monitored, whereas elsewhere representative samples were taken. An interim intervention level of 2000 Bq/kg of Cs-137 was agreed. Altogether 354 carcasses were condemned, and a compensation of 164,000 FIM was paid to the company which had purchased the carcasses from the reindeer poducers. In fact this amount of 164,000 FIM is the only direct compensation paid to the producers due to condemned food in 1986.

In 1987 the monitoring of reindeer meat was continued. However, the interrim intervention level for reindeer meat was elevated to be 4000 Bq/kg, because the dose contribution from other food went down. The application of this level resulted in condemnation of 127 carcasses, resulting in a total compensation of 110,000 FIM to the reindeer producers. In 1988 the number of condemned carcasses was 54 resulting in an economic compensation of 41,000 FIM. Thus, the cumulative compensation paid by the Finnish government has been ca 315,000 FIM.

Since there is no legislation in Finland on such a compensation, the relevant regulations of the law on the prevention of the spread of communicable animal diseases were applied. According to this law the owner of an animal may be granted economic compensation due to communicable diseases. The concept of communicable animal disease also includes radioactive contamination. If any food of vegetable origin had to be condemned, this law were not applicable, but an ad hoc decision must have been done. The radioactivity to some extent affected the demand of reindeer meat thus causing marketing problems to the producers and the meat processors.

15.3.3 Costs related to restrictions on use of peat and peat ash

The summer 1986, because of favourable weather conditions, was an exceptionally good summer for peat production. It caused that the fuel peat harvested in 1986 was burned even in the heating season 1988-89. When estimating the economic impact of the recommendations given for the peat ash utilization, only approximate figures can be derived. In this estimation we can assume that about half of the total ash amount produced in Finland during the time period from September 1986 to March 1989, was removed to dumps because of the recommendations. The remaining ash was either produced outside the fallout region or it was piled up elsewhere according to the directions of the STUK. This time period corresponds to about two and a half years of consumption of fuel peat, i.e. 2.7×10^5 tons of peat ash. Half of that is estimated to have been removed to dumps, i.e. 1.4×10^5 tons.

The cost of the removing of ash to dumps consists of the dump charges (29 FIM per ton of ash) and of the cost of soil-improving lime when peat ash in silviculture and in agriculture has to be compensated by the lime (30 FIM per ton of ash). If all the ash had been used in silviculture or in agriculture, the economic impact of the interruption of the use during these years can be estimated to about 8 million FIM. If, on the other hand, all the ash had been used as a landfill material, the cost of the interruption would consist of the dump charges (29 FIM per ton of ash), of the cost of the soil material compensated by the ash (5 FIM per ton of soil) and of the transport charges of the soil material (7 FIM per ton of soil). In this case the economic impact could be estimated to about 10 million FIM.

The use of peat ash in the concrete industry in Finland before the Chernobyl fallout was quite small. Only one factory used peat ash in its concrete products, and its ash consumption was about 16000 tons per year. The cost of the interruption of peat ash use consists of the dump charges (29 FIM per ton of ash), of transport charges of ash to dumps (48 FIM per ton of ash), (there was no transport cost when ash was used in the concrete factory because the transmission set of pipes for peat ash was earlier built directly from the power plant to the factory), and of the cost of fine grained filler sand compensated by the peat ash (47 FIM per ton of filler sand). With these figures we can estimate the upper limit for the economic impact of interruption of peat ash utilization in the concrete factory, being about 10×10^6 FIM during these three years. In reality, the cost has been smaller because the use of peat ash was not interrupted wholly.

15.3.4 Costs related to tourism and transportation

The State Board of Tourism estimated that the cancellation of trips to Finland during the summer 1986 season caused the tourism industry a loss of about 100 million FIM. The actual loss is smaller since there are costs involved in offering the tourism services. For the national economy another loss-reducing factor might have been increased domestic tourism, as Finns cancelled vacations abroad to some destinations. The national airline, Finnair, calculated a Chernobyl-related loss of revenue of about 40 million FIM.

15.4 Summary of economic consequences in the Nordic countries

The following table contains the information from the previous subchapters related to interdiction of food-stuffs, and for Finland also including losses related to interdiction of peat ash, which would otherwise have been used in production of concrete.

Type of economic consequence	Norway 1986	Norway 1987	Norway 1988	Finland 1986	Finland 1987	Finland 1988	Sweden[1]
Salad	0.045 M$	0	0	0.7 M$	0	0	
Goat cheese	1.4 M$	1.0 M$	-[2]				
Sheep interdiction	14 M$	0.35 M$	0				
Sheep down-feeding	5 M$	6 M$	8.5 M$				
Sheep control and surveillance	-	1.8 M$	-				
Reindeer interdiction	2.9 M$	1.2 M$	0.7 M$	0.04 M$	0.02 M$	0.01 M$	
Reindeer down-feeding, control and surveillance	1.1 M$	1.4 M$	-				35 M$
Beef down-feeding, control and surveillance	0.71 M$	0	0				
Fish							0.7 M$
Moose							1.0 M$
Berries and mushrooms							0.7 M$
Peat ash				4.5 M$	0	0	
<u>Consumers resistance:</u>							
Fish							3.7 M$
Berries							16 M$
Mushrooms							2.4 M$

[1] Sum of the years 1986, 1987, and part of 1988 in some cases.
[2] "-" means "information not available".

Table 15.1 Summary of information from the other parts of this chapter. (It is stressed that the values for the different countries, in this table are not directly comparable, as also stated in the text.)

It is seen that the types of consequences differ considerably between the countries, and that the approach to evaluating the economic impact has also been somewhat different. The economic consequences cited for Norway and Finland are more or less direct. The Swedish numbers also include losses attributed to consumer's resistance, which must necessarily be based upon estimates; and items like losses due to reduction in the extent of collection of berries and mushrooms in the forests for private consumption.

16. SENSITIVITY AND UNCERTAINTY ANALYSES, PROJECT AKTU-235

The program includes a sensitivity/uncertainty analysis that is performed in Sweden (project *AKTU-235*) using the computer programs PRISM and UNIDOSE. The results of the project are reported in (ref. KA89a).

Mathematical models are important tools for summarizing current knowledge and information in a variety of scientific fields. Models provide a rapid means of testing new ideas and, where reliability allows, provide predictions about the long term results of system modification. As model based decisions become more common, the methods of evaluating their reliability also become important.

In this project uncertainty and sensitivity analysis are applied to the consequence assessments of some hypotetical reactor accident scenarios. The primary goal has been to study the general behaviour and usefulness of such an analysis, but also to get an idea of the overall uncertainty in a typical assessment and to identify those submodels and/or input parameters that significantly contribute to this uncertainty.

Similar studies and more comprehensive ones have been going on the last decade, and today uncertainty analysis is recognized as an important tool in consequence assessment.

The use of the terms uncertainty and sensitivity analysis is sometimes unclear. In project *AKTU-235*, uncertainty analysis means the quantification of the uncertainty in a model respons in terms of statistical quantities like standard deviations etc, and sensitivity analysis is the identification of important parameters and the more general behaviour of the model when input parameters are varied.

This study involves only the uncertainty of the model response caused by uncertainty in the input parameters. Other sources of uncertainty, i.e. inability of the model itself to describe the physical/chemical events must be studied with experimental tools or with comparisons with other models. However, model uncertainty can be simulated in a simplified way by introducing "model uncertainty factors" in critical parts of the model.

The methodology used in this study is, in a summarized form, Latin Hypercube sampling from given input parameter probability density functions (pdf´s) and correlation and regression analysis of the model responses against the sampled parameter values. The methodology is implemented in a suite of codes called PRISM (Ref. GA83).

The study has been carried out in three phases.

- Phase 1 Evaluation of existing techniques of uncertainty analysis and calculations of some test cases (Ref. KA85).

- Phase 2 Identification of significant parameters, scenarios and source terms. Estimation of probability distribution functions for selected parameters (Ref. KA88a).

- Phase 3 Calculation of consequences and uncertainties for the selected scenarios. Identification of important parameters.

In Phase 1, different methods for uncertainty analysis, such as Simple Monte Carlo, Discrete Variable approach, Moment Matching and Response Surface Techniques, were studied in a literature survey (Ref. HO83 and AL85). No indications were found, that any of the named methodologies is more suitable then the PRISM one for this type of study, although the latter methodology had to be used by practical and budget reasons.

The most relevant parameters, based on the test calculations in Phase 1, were selected in Phase 2, and assigned to uncertainty pdf's by a Nordic group of experts according to a standardised procedure. A selection of scenarios was also carried out.

Finally, in Phase 3, the consequences and their output pdf's as well as the ranking of the most important parameters were calculated with the UNIDOS and PRISM codes for these scenarios.

UNIDOSE (Ref. KA79) is used to model the consequences of accidental airborne releases. It calculates dispersion in the air, deposition on the ground, doses and health effects to individuals as well as to populations. The model was used by the Swedish participant in the CSNI Benchmark study (Ref. OE84).

An important part of an uncertainty analysis consists of choosing the ranges within which each parameter shall be allowed to vary, and what probability distributions to use within these ranges. This part of the project has been carried out by several different groups in the Nordic countries. The conclusions drawn from these evaluations are summarized in Table 16.3.

Preliminary analyses are reported in (Ref. KA87a), in which 22 parameters were included. Two approaches were chosen; one with small variations of the parameters around their best estimates and one with variation over their full ranges (from a preliminary estimation). These two approaches did not quite give the same parameters as the ones contributing most to uncertainty. Relative to calculation of doses the first approach gave the following parameters as the most important ones (by ranked regression): run-off, sigma-y, release. The second approach gave: sigma-y, wet deposition, release time. Relative to calculation of acute fatalities, there were also different parameters that were the most important. For the first approach: LD-50, wet deposition, release magnitude. For the second approach: Release time, wet deposition, sigma-y.

Other publications relating to this project are (ref. KA85, KA87b and RE88)

16.1 Uncertainty in model responses

Figures 16.1, 16.2 and 16.3 show the CCDF's for individual dose at a certain distance, collective dose and some health consequences. The uncertainty bands, denoted 1-99%, is the 5th lowest and highest value at each point of the curve out of 500 runs. These bands thus represent the extreme outer bands of the distributions, but it is difficult to say whether they really correspond to the true 1-99% levels. The reason to choose the 5th highest and lowest values is computational only. Figure 16.1 also shows the contribution to individual dose via the four exposure pathways included in the calculations.

Figure 16.4 shows the results in a somewhat different way. Here, the lowest value, the 5, 25, 50, 75 and 95 percentages and the highest value (the 7 columns within the vertical lines) are plotted for the mean and 95 percentage of the CCDF for each quantity.

Finally, the actual distributions of the mean and 95 percentage are shown in Figures 16.5 and 16.6. The distribution of the most probable values, i.e. the most probable values of all parameters are used in the computations (no parameter variation at all), is also plotted in these figures. The probability distribution in these cases (most probable values) represents consequently the "uncertainty" from the naturally varying parameters, i.e. the wind speed, stability etc. For this case one can see that this "uncertainty" is quite similar to the uncertainty from the analysed parameters.

All results shown in the figures in the present report are for the largest release magnitude. Similar results have been calculated for several different release magnitudes, but the results are not included in the present report. Results for other types of health effects are also available, though not included here.

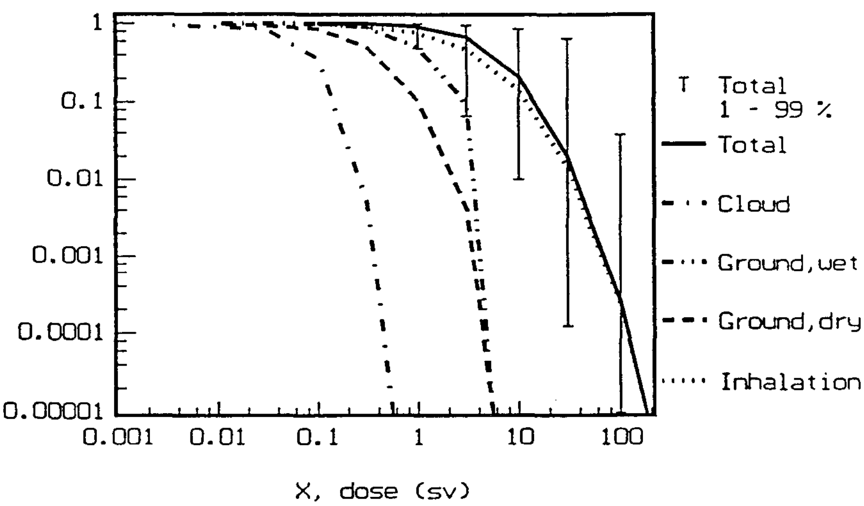

Figure 16.1 CCDF for release category A1. Plume passage and 24 hour doses only. Probability of exceeding x. Individual dose.

Note: The results presented represent fictive source terms and site assumptions, and have therefore no significance to any real sites.

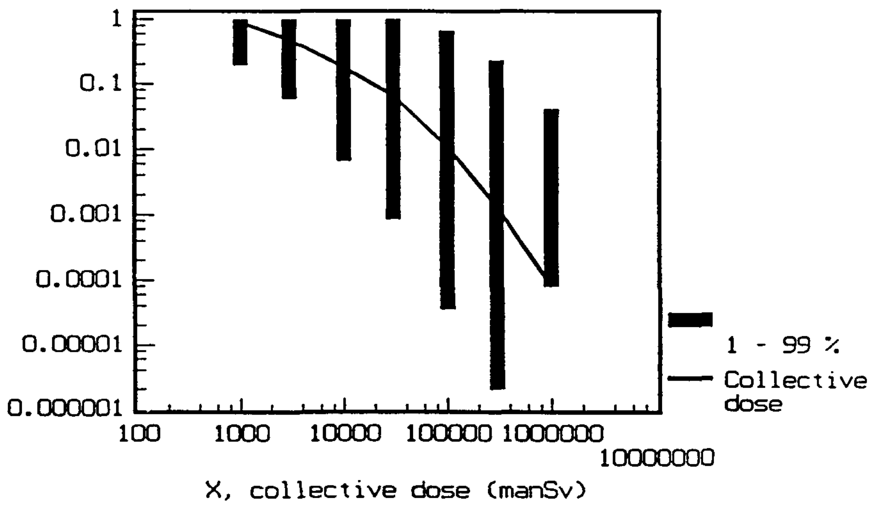

Figure 16.2 CCDF for release category A1. Plume passage and 24 hour doses only. Probability opf exceeding x. Collective dose.

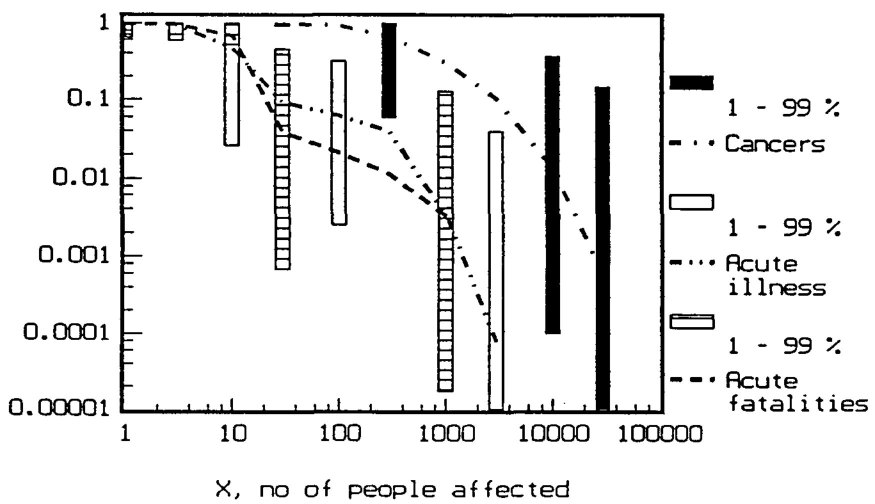

Figure 16.3 CCDF for release category A1. Plume passage and 24 hour doses only. Probability opf exceeding x. No. of persons.

Note: The results presented represent fictive source terms and site assumptions, and have therefore no significance to any real sites.

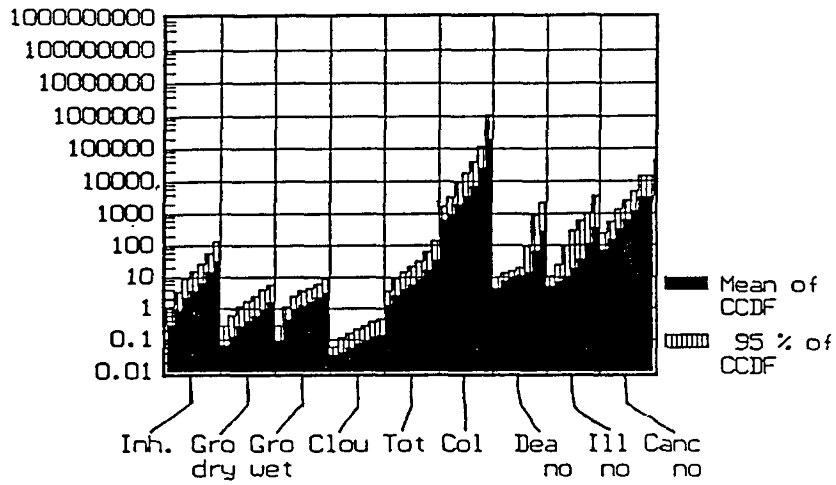

Figure 16.4 Release category A1. Lowest, 5, 25, 50, 75, 95% and highest value of the mean and 95 percentile of CCDF. (Dose (Sv)) or (No of persons).

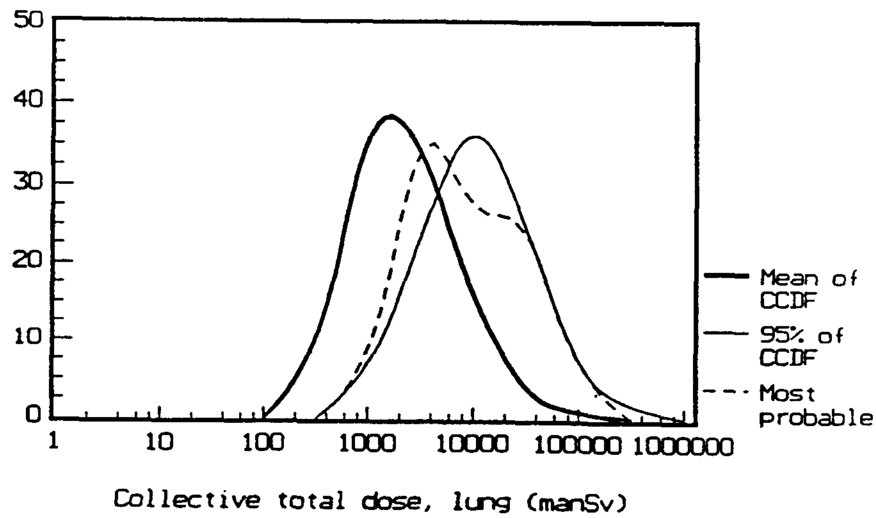

Figure 16.5 Release category A1. Distribution of output responses. Plume passage and 24 hour doses only. Frequency (%) per half decade.

Note: The results presented represent fictive source terms and site assumptions, and have therefore no significance to any real sites.

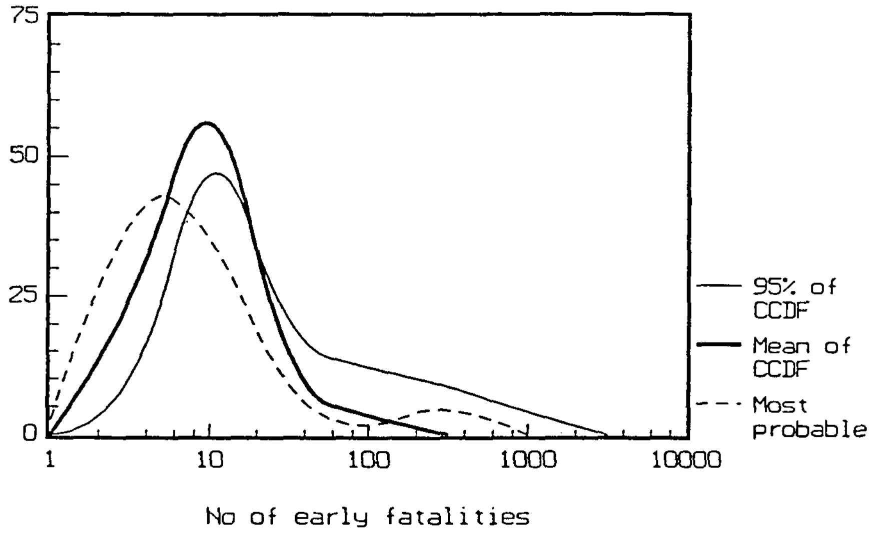

Figure 16.6 Release category A1. Distribution of output responses.

Note: The results presented represent fictive source terms and site assumptions, and have therefore no significance to any real sites.

Some of the conclusions from project *AKTU-235* regarding the magnitude of the uncertainties are:

In release cases A_1 and A_3 (see subchapter 12.1.3 for description of the releases) the dominating pathway of exposure is inhalation during plume passage due to the relatively high release fractions of Tellurium and Ruthenium. Release case C_1 is completely dominated by the ingestion doses, but all contaminated foodstuffs are assumed to be consumed in these calculations.

The uncertainty in the probability that a consequence will be exceeded (Figures 16.1 to 16.3) is growing very largely at the low probability end of the curves, up 4 - 5 orders of magnitude, and there is no significant differences between the pathways of exposure. At a probability level of 95% (0.05 on the ordinate) the uncertainty is 1 - 2 orders of magnitude.

The uncertainties of the mean and 95% level of consequences (Figure 16.4) are smaller. The quotient between highest and lowest, which roughly corresponds to the 1-99% values in the CCDF curves, varies between 1 order of magnitude (cloud shine dose) to 3 orders of magnitude (inhalation dose, collective total dose and the health effects). The variation within one standard deviation (16% to 84%) is a factor 3 to 5 for most of the analysed consequences.

Note also the long tails of the distributions for the acute fatalities release case A_1 (Figure 16.6).

16.2 Identification of important parameters

The relative importance of the parameters utilized in the calculations has been investigated by regression analyses. The relative importance will differ depending upon the calculation result concerned; e.g whether one is interested in the impact of the parameter upon plume dose or ingestion dose. Table 16.1 shows the results of the regression analyses and the identification of important parameters for the small magnitude release C_1. The table summarizes the output from PRISM3, as an example. Explanations of the code names found in Table 16.1, and others that are used in the computer runs, but happen not to be included in this table, are found in Table 16.2. In this table is also summarized the results of the parameter uncertainty evaluation. The standard deviation, lower limit, upper limit, most probable value and type of distribution are given for each of the parameters.

The % cov(ariance) column indicates the amount of variance that is explained by the uncertainty of the listed parameters. Differences in the results of the ranked and unranked regression indicates that there is a non-linear dependence between the parameter and the model response. Only covariances greater than 1% are included in the tables. The high accuracy of these values (the large number of digits in the tables) are only due to the printout format and have no real significance.

The difference between the analysis of the 95% value and the mean of the CCDF is not so large. Tendencies found for the mean values seem to be enlarged for the 95 percentages.

The overall most significant parameter for release category A_1 and A_3 is the wet deposition parameter, which is both negatively (plume depletion) and positively (increased wet deposition) correlated to the exposure.

The results of the regression analyses for three different release categories are summarized below, although only the results for release category A_1 are included in the present report (Table 16.1). Releases A_1 and A_3 are large releases with release heights 20 and 100 meters respectively, and C is a much smaller release with release height 20 meters.

For each item in the list is given the calculation result of concern and the parameters that contribute most to the uncertainty of this type of result, due to the inherent uncertainty of these parameters:

Release category A_1 and A_3:

- Inhalation lung dose at 2 km: wet deposition, inhalation dose conversion factor and house filter factor.
- 24 hour ground bone marrow dose from dry depos. activity at 2 km: dry deposition, ground shielding and wet deposition.
- Like above, from wet deposited activity: wet deposition and ground shielding.
- Cloud shine bone marrow dose at 2 km: plume shielding.

(The list is continued after the tables)

Unranked regression Parameter	% cov	Ranked regression Parameter	% cov
Ground-shine bonemarrow dose, wet deposition, at 2 km, mean of CCDF			
WETD,PART	25.2506	WETD,PART	35.6696
SHIELD,G,R	24.2486	SHIELD,G,R	23.6780
SY,A-D,X<1	11.6433	SY,A-D,X<1	11.1658
RUNOFF,R	5.16925	RUNOFF,R	3.59451
MEAND, FAC	1.91384	DRYD,PART	2.08305
DRYD,PART	1.71492	SY,E-F,X<1	1.64661
SY,E-F,X<1	1.34337	MEAND, FAC	1.00861
Cloud-shine bonemarrow dose at 2 km, 95 % of CCDF			
SHIELD,P,R	71.4186	SHIELD,P,R	73.6403
MEAND, U0	17.1812	MEAND, U0	14.9722
WETD,PART	1.98478	WETD,PART	1.82081
DRYD,PART	1.70649	DRYD,PART	1.62714
Cloud-shine bonemarrow dose at 2 km, mean of CCDF			
SHIELD,P,R	80.5163	SHIELD,P,R	83.3083
MEAND, FAC	5.51001	SY,A-D,X<1	4.11802
SY,A-D,X<1	4.94983	MEAND, FAC	4.17908
MEAND, U0	2.39069	WETD,PART	2.22584
WETD,PART	2.24094	MEAND, U0	2.02320
DRYD,PART	1.02950		
Total lung dose at 2 km, 95 % of CCDF			
DOSOMVANDL	27.1272	WETD,PART	32.5914
WETD,PART	23.4769	FILTER	27.7188
FILTER	21.4642	DOSOMVANDL	26.2888
SY,E-F,X<1	1.48253	SY,E-F,X<1	1.45969
SZ,E-F,X<1	1.49032	MEAND, U0	1.43137
DRYD,PART	1.25809	SHIELD,G,R	1.20051
Total lung dose at 2 km, mean of CCDF			
DOSOMVANDL	25.7521	WETD,PART	28.2316
WETD,PART	20.8347	FILTER	27.8767
FILTER	21.6340	DOSOMVANDL	25.9846
SZ,A-D,X<1	1.92166	SY,A-D,X<1	3.41985
SY,A-D,X<1	1.78766	SZ,A-D,X<1	1.91492
SZ,E-F,X<1	1.31041	SHIELD,G,R	1.82197

Table 16.1 Identification of important parameters. Release category A_1. Doses from plume passage and 24 hours. (cov means co-variance).

(Table continued on next page)

Unranked regression Parameter	% cov	Ranked regression Parameter	% cov
Collective total lung dose out to 500 km, 95 % of CCDF			
WETD,PART	15.9439	WETD,PART	67.4849
DOSOMVANDL	7.29676	DOSOMVANDL	13.7987
FILTER	3.88326	FILTER	12.7940
DRYD,PART	1.36090		
SZ,E-F,X<1	1.05407		
Collective total lung dose out to 500 km, mean of CCDF			
WETD,PART	17.9752	WETD,PART	68.2258
DOSOMVANDL	6.62303	FILTER	13.6715
FILTER	4.19835	DOSOMVANDL	12.5331
DRYD,PART	1.37491		
Acute fatalities, 95 % of CCDF			
WETD,PART	17.4283	WETD,PART	42.8702
DOSOMVANDL	12.5319	DOSOMVANDL	19.3123
FILTER	10.7992	FILTER	17.5218
DRYD,PART	2.04619	LD-50	1.65917
SZ,E-F,X<1	1.52924		
RUNOFF,R	1.05988		
Acute fatalities, mean of CCDF			
DOSOMVANDL	11.4564	DOSOMVANDL	27.6012
WETD,PART	10.6151	FILTER	25.0527
FILTER	9.49622	WETD,PART	8.66026
DRYD,PART	1.91458	SHIELD,G,R	5.06591
RUNOFF,R	1.32954	LD-50	2.71780
SZ,E-F,X<1	1.27676	SZ,A-D,X<1	1.41633
Acute illness, 95 % of CCDF			
WETD,PART	34.7362	WETD,PART	46.3250
DOSOMVANDL	10.3472	DOSOMVANDL	8.67572
FILTER	6.16428	SHIELD,G,U	6.71943
SHIELD,G,U	3.82823	FILTER	6.46569
DRYD,PART	1.86889	SY,A-D,X<1	3.22695
SZ,E-F,X<1	1.77199	LD-50	2.84677
SY,A-D,X<1	1.12993	RUNOFF,U	1.55299
		MEAND, FAC	1.40337
		SZ,E-F,X<1	1.20323
		DRYD,PART	1.14445

Table 16.1 Identification of important parameters. Release category A_1. Doses from plume passage and 24 hours.

(Table continued on next page)

Unranked regression Parameter	% cov	Ranked regression Parameter	% cov
Acute illness, mean of CCDF			
WETD,PART	31.2920	WETD,PART	57.4569
DOSOMVANDL	9.79495	DOSOMVANDL	6.74108
FILTER	6.79287	SHIELD,G,U	5.50483
DRYD,PART	2.91960	FILTER	5.35413
SZ,E-F,X<1	1.56585	LD-50	2.54461
SHIELD,G,U	1.33216	SY,A-D,X<1	2.53805
		DRYD,PART	2.09432
		MEAND, FAC	1.76947
		RUNOFF,U	1.19116
Late cancers, 99% of CCDF			
WETD,PART	14.4920	FILTER	21.5950
DRYD,PART	14.0450	WETD,PART	21.8624
DOSOMVANDL	12.7615	DRYD,PART	19.8863
FILTER	11.5076	DOSOMVANDL	18.0702
THYR CANC	3.35422	THYR CANC	10.5436
Late cancers, 99% of CCDF			
WETD,PART	16.3166	WETD,PART	25.2148
DOSOMVANDL	13.9342	FILTER	21.4299
DRYD,PART	13.8739	DRYD,PART	19.5996
FILTER	12.5699	DOSOMVANDL	17.7169
THYR CANC	4.14670	THYR CANC	10.4666

Table 16.1 Identification of important parameters. Release category A_1. Doses from plume passage and 24 hours.

(Table continued from preceding pages)

DEPENDENT VARIABLE		I.LU.1.95		UNRANKED	REGRESSIONS	
		MEAN = 2181.73				R2 = 75.16
INDEPENDENT VARIABLES	DF	MEAN	SEQUENTIAL SUM OF SQS	SLOPE	PRODUCT	R2 IMPROVE
INTERCEPT	1			619.958		
DOSOMVANDL	1	1.55330	5.069344E+08	1522.86	2365.45	26.6944
WETD,PART	1	7.01000	4.606886E+08	-213.811	-1498.82	24.2592
FILTER	1	1.00884	4.007125E+08	2080.58	2098.97	21.1009
DRYD,PART	1	2.64340	3.372605E+07	-183.030	-483.823	1.77596
SZ,E-F,X<1	1	1.15001	2.516635E+07	-799.997	-920.008	1.32522
ERROR	494		4.718009E+08	MSE =	955063.	
TOTAL	499		1.899029E+09			

DEPENDENT VARIABLE		I.LU.1.95		RANKED	REGRESSIONS	
		MEAN = 250.500				R2 = 91.66
INDEPENDENT VARIABLES	DF	MEAN	SEQUENTIAL SUM OF SQS	SLOPE	PRODUCT	R2 IMPROVE
INTERCEPT	1			168.824		
WETD,PART	1	250.500	3.716802E+06	-.598759	-149.989	35.6814
FILTER	1	250.500	2.906143E+06	.519932	130.243	27.8991
DOSOMVANDL	1	250.500	2.793207E+06	.517482	129.629	26.8149
DRYD,PART	1	250.500	132059.	-.112604	-28.2073	1.26777
ERROR	495		868414.	MSE =	1754.37	
TOTAL	499		1.041663E+07			

Table 16.2 Selected parameters and estimated uncertainties. (Explanation of distribution types: (T = triangular, N = normal, LN = log normal).

Code name	Distr type	Std. dev	Low. lim.	Upp. lim.	Most prob.	Parameter description
'SY,A-D,X<1'	'T'	0	0.33	3.0	1.0	Error of sigma-Y, A-D, X<10km
'SY,E-F,X<1'	'T'	0	0.6	1.6	"	" E-F "
'SY,A-D,X>1'	'T'	0	0.3	3.3	"	" A-D X>10km
'SY,E-F,X>1'	'T'	0	0.5	1.7	"	" E-F "
'SZ,A-D,X<1'	'T'	0	0.27	4.0	"	Error of sigma-Z, A-D X<10km "
'SZ,E-F,X<1'	'T'	0	0.55	1.9	"	" E-F "
'SZ,A-D,X>1'	'T'	0	0.5	2.0	"	" A-D X>10km
'SZ,E-F,X>1'	'T'	0	0.24	4.4	"	" E-F "
'MIXINGH, A'	'T'	0	0.27	2.0	1500	Mixing height (m), Pasquill A
'MIXINGH, B'	'T'	0	0.30	2.5	1200	" B
'MIXINGH, C'	'T'	0	0.2	2.2	1000	" C
'MIXINGH, D'	'T'	0	0.1	2.9	750	" D
'MIXINGH, E'	'T'	0	0.25	1.9	400	" E
'MIXINGH, A'	'T'	0	0.24	1.9	400	" F
'MEAND, FAC'	'T'	0	0.5	1.5	-	Meandering factor, see text
'MEAND, U0 '	'T'	0	0.6	1.5	2.0	Upper wind speed for meandering
'DRYD,PART '	'T'	0	0.33	6.6	0.3	Dry dep. for particles, (cm/s)
'DRYD,EL. I'	'T'	0	0.2	3	1.0	" el. Iodine "
'WETD,PART '	'T'	0	0.03	20	1.5E-4	Washout coeff., particles (s-1)
'WETD,EL. I'	'T'	0	0.5	5	2.0E-5	" el. Iodine "
'SHIELD,P,R'	'N'	0.5	0.5	1.7	0.12	Plume shielding, rural area
'SHIELD,P,U'	'N'	0.5	0.5	2.0	0.01	" urban "
'SHIELD,G,R'	'N'	0.25	0.5	1.5	0.10	Ground shielding, rural "
'SHIELD,G,U'	'N'	0.30	0.6	1.6	0.05	" urban "
'FILTER '	'LN'	0.4	0.4	2.0	0.5	Filter factor for houses
'RUNOFF,R '	'N'	0.1	.75	1.25	0.2	Run off, rural areas
'RUNOFF,U '	'N'	0.1	.67	1.33	0.4	" urban "
'WEATHERING'	'N'	0.5	0.5	2.0	0.3	Fraction of short time weathering
'T1/2 WEATH'	'LN'	0.4	0.5	2.0	300	Corresponding, $T_{\frac{1}{2}}$ (days)
'DOSOMVANDL'	'T'	0	0.33	3.33	W1400	Dose conv. fact., inhal. and inges.
'LD-50 '	'T'	0	0.8	1.2	Diff	50% affection level, early effects
'LEUKEMI '	'T'	0	0.1	2.5	0.002	No of cancers per manSv, leukemi
'OTHER CANC'	'T'	0	0.16	1.66	0.03	" , other canc
'LUNG CANC '	'T'	0	0.33	3.33	0.006	" , lung canc
'THYR CANC '	'T'	0	0.16	1.33	0.03	" , thyroid
'INTFOOD '	'T'	0	.2	3.33	Diff	Food concentration factor (m2/kg)
'CONSUMPT. '	'T'	0	.5	2	Diff	Individual consumption, (kg/year)
'AREAFRACT.'	'T'	0	.5	2		Fraction of area used for growth
'PRODUCT. '	'T'	0	.5	2		Production of foodstuff, (kg/m2*y)
'T1/2 W LT '	'T'	0.4	0.5	2.0		Long time weathering $T_{\frac{1}{2}}$, (s)
'WINTER SH '	'T'	0	0.5	1.5		Shielding factor for winter cond.

Assumed correlations
'SHIELD,P,U' 'SHIELD,G,U' 0.5
'SHIELD,P,R' 'SHIELD,G,R' 0.5

- Total lung dose from plume passage and 24 hours at 2 km: same as for inhalation dose.

- Total collective lung dose out to 500 km from plume passage and 24 hours: wet deposition.

- Early fatalities: wet deposition.

- Early illness: wet deposition.

- Late cancer cases: wet deposition.

Release category C_1:

- Total plume passage bone marrow dose at 2 km: plume shielding.
- 50 years ground bone marrow dose at 2 km: ground shielding.
- 50 years individual ingestion bone marrow dose at 2 km: food transfer factor.
- Total 50 years bone marrow dose: food transfer factor.
- Total collective plume passage bone marrow dose: plume shielding.
- 50 years collective ground bone marrow dose: wet deposition.
- 50 years collective ingestion bone marrow dose: food transfer factor.
- Total collective 50 years bone marrow dose: food transfer factor.

16.3 General aspects of the analyses

The number of iterations to be carried out in order to achieve the desired accuracy are crucial for the computer costs. Therefore, three runs with 50, 100, and 500 iterations were carried out, and the results are shown in Table 16.3. As can be seen, the 50 and 100 iteration runs detected some false dependencies, even from parameters not present in the calculations (VAKANT, which means that the parameter has not been assigned any value), but the conclusions made from these runs would be similar. Especially for the ranked regresssion, the differences are small.

16.4 Conclusions drawn from the uncertainty/sensitivity analyses

The goals for this project have been to answer or at least try to answer the following questions:

1) How large are the uncertainties in the results of a typical assessment?

2) Can these uncertainties be reduced, and if so, to which extent?

3) Do uncertainty/sensitivity analyses improve consequence analyses?

4) Which methods can be used, how easily can they be implemented into existing codes and how expensive are these methods to use?

5) What parameters contribute most to the uncertainty of the different types of results that can be obtained from a consequence analysis?

The answer to the first question depends on which measure of uncertainty is to be used, i.e. 1 - 99% limits or variations within one standard deviation, and which quantity is used as result, i.e. expectation values or 95 percentage. The uncertainty grows rapidly with decreasing probability of occurrence to several orders of magnitude at the low probability end of the CCDF curve. If one looks on the expectation value of the CCDF and variation within one standard deviation, the uncertainty is a factor 2 - 5 up and down.

The sensitivity analysis indicates which parameter causes the uncertainty. For some of these, such as the wet deposition coefficient or the dose conversion factors, it is not an easy task to reduce the uncertainty related to them. For others, like the shielding factors, improvements are more feasible. Sensitivity analyses could therefore be used in a cost/benefit way.

It is also possible to compare the uncertainty in the results due to uncertainty in the parameter values, to the "natural" variation (due to varying weather conditions, differences in day and night population distribution etc.). From a decision makers point of view it is illogical to reduce the first type of uncertainty to a level very much lower than then the latter, the natural, which can not be reduced. In this study, these two different types of uncertainties turned out to be fairly equal.

To the third question the answer is probably yes, for both scientist and decision-maker. Without uncertainty analyses, the estimates of parameter values must be made conservative in order to get an upper limit of the consequences, sometimes leading to overconservative and almost useless results. With uncertainty analyses it is possible to use the most probable values in connection with a probability distribution, which is a more natural way to quantify the influence of a parameter. For the scientist, the cost/benefit approach to sensitivity analyses is very useful.

The decision-maker wants to know the range of consequences for a given accident scenario, disregarding the sources of this variation, and with uncertainty analyses he/she can achieve this information. One problem is, however, to find ways of presenting him/her with the result in an understandable form.

To the fourth question an answer could only be given for the methodology implemented in the PRISM codes.

With a dynamic structure as in the PRISM codes, it is very easy to perform uncertainty analyses on any code that reads parameter values and output model responses.

The computer costs are directly related to the necessary number of runs, and is therefore depending on the type of analysis that is to be performed. However, 50 to 100 runs seems to be adequate for most applications.

The answers to the fifth question are given in the various tables in this chapter.

17. NORDIC CHERNOBYL DATA BASE, PROJECT AKTU-242

Immediately after the Chernobyl accident the AKT Steering Group initiated this project, funded by re-allocating funds from the other AKTU projects. It was decided that the Nordic Chernobyl Data Base (NCDB) should be an AKTU project and be coordinated from Risø National Laboratory. The already existing Nordic Weapons Fall-out Data Base had been developed at Risø in the 70's and early 80's. The latest updatings were performed as part of the "predecessors" to AKTU in the preceding project periods.

Quite soon contact was taken with a similar effort that was initiated by the Commission of the European Communities. This work was to be performed in Ispra, Italy, and the data bank is called REM (Radioactivity Environmental Monitoring). Contact with this project has been upheld, and although the two data bases will not be identical in structure, it is intended that it shall be simple to transfer of data from one to the other, and that participation in one will give access to data in the other. It is not quite sure, however, what decision has been or will be taken as regards the accessibility.

A group of contact persons in the Nordic countries was formed. The contact persons were responsible for assuring that data submitted from their country were of satisfying quality. Institutions that submitted data had to partake in the intercalibration program.

The Nordic Chernobyl Data Base is established for scientific purposes. The aim is to collect validated data resulting from measurements made in Nordic laboratories in the period following the Chernobyl accident.

All information is stored in the C_base data handling system. This system is developed at Risø especially for such types of environmental data in order to facilitate:

- Easy input from a variety of sources.
- Easy handling of multiple information on each sample.
- Easy output to other computer programs for further data treatment.

The main technical reference is (ref. LI89), which contains descriptions in detail of the structures, codes and methods used.

17.1 Overview of the development

Following the decision by the AKT Steering Group, a pre-project was initiated by the Swedish National Institute of Radiation Protection (SSI), until the actual project AKTU-242 was established. The SSI has continued this work, in parallel to project AKTU-242, and has chosen to collect the Chernobyl data from Sweden in a Swedish data base, before transmittal to the NCDB. A short description of this Swedish work, which has not been actual part of the AKTU program, is given in subchapter 17.9.

The main aim of project AKTU-242 has been to collect as much as possible of the large amount of data resulting from the environmental investigations in the Nordic countries following the Chernobyl accident.

The variety of information on each sample can be very large. It depends on sample type, location, method of treatment and applied measurement techniques. A sample may vary from a simple measurement of dose rate over a gamma spectrum to a long earth profile, sampled for determination of different isotopic concentrations versus depth of the profile with microchemic conditions as essential parameters.

Furthermore it was the aim to utilize the Nordic Weapons Fall-Out Data Base, established at Risø in 1973, which contains such data since the late 50's.

It was also the task to take initiatives in order to enhance a unified approach to the interpretation of measurements, e.g. through intercalibration exercises.

A final consideration was to facilitate cooperation with international agencies doing parallel work, like the REM- bank of the Commission of the European Communities. This work was performed at the Ispra establishment in Italy.

An initial search showed, that existing data base systems- mainly used for administrative purposes - were not well suited for handling of the large data amounts to be expected from the Nordic national programmes concerned with the Chernobyl consequences.

Therefore it was decided to develop a special data base system based on experience with similar tasks, which would be able to give:

- a fast and flexible input/output.
- a high degree of economy of storage space.
- possibility for collection and treatment of almost all kinds of experimental results and documentary messages especially from the environmental field.

Large emphasis was put on getting a presentation system as versatile as possible, giving treated data in time- or geographical scales as well as a handling system for any statistical analysis.

It was further decided, that the NCDB should be developed for use on modern Personal computers as well as on a VAX-computer.

Data quality has been a constant concern for the working group. This lead to the execution of an intercalibration exercise between 21 of the participating laboratories. Sets of calibration samples were produced and distributed for measurements, and the results were then evaluated. The results of the exercise proved to be very satisfactory, as it showed that all cesium measurements made by the 21 laboratories were satisfactory.

Classification of the validity of the data available has been discussed at length. Means have been defined for stating of

- measurement uncertainty.
- sampling quality.
- calibration quality.

In cases where these are not given, the authority to add a classification has been delegated to the national representatives in the working group.

17.2 Working group

The working group has consisted of the following national representatives and functionaries:

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17.3 Intercalibration exercise

As the amount of measurement data after Chernobyl was vast and with a very large variation in quality (in sampling method, sampling treatment, measurement technique etc.), it was already in the initial stages of project AKTU-242 clear that an intercalibration exercise between the participating laboratories was desireable. This exercise was initiated in early 1987.

17.3.1 Intercalibration samples

Six types of samples have been used. Two sets of each six samples, containing different, relevant isotopes in differing amounts, were produced for each of the four Nordic countries. The samples were:

1) Dried milk.
2) Lichen, collected in the Härnösand area, northern Sweden.
3) Seaweed, collected close to the Ringhals nuclear power plant.
4) Water sample, synthetic with short lived isotopes added.

5) Low activity, deep layer sample, (background sample, 2 m deep clay layer, Risø area).
6) Swedish surface soil from the Gävle area, Sweden.

The lichen samples were produced by SSI and the remaining samples by Risø. All the samples were - after check of their homogeneity - distributed at the end of 1987.

17.3.2 Intercalibration procedure

The intercalibration program between the participating laboratories was successful. In spring 1988 most of the calibration data were received by the Secretary, and a preliminary analysis was made. Some corrections had to be made, especially with respect to reference date. A new analysis was made by Risø in spring 1989. Interest for participation was later expressed by Iceland and Färoe Islands, and sets of samples have been delivered for their measurements and results are expected.

17.3.3 Analysis

A detailed, preliminary list of all received measurements (randomly numbered) has been distributed to all participating laboratories. Each participant was informed only about the identification number of their own measurements, so that they easily could identify their own measurements and compare them to all the other measurements; but at the same time be unable to compare them to the results submitted by any other specific institution.

The main conclusions from this intercalibration analysis are:

- All cesium measurements (Cs-134, Cs-137) are acceptable.

- All background measurements (sample no.5) from the 12 laboratories who gave results, are acceptable.

- With respect to other specific isotopes the results were not quite so satisfactory. This indicates a need for further intercalibration exercises with radionuclides other than cesium.

A detailed report on the intercalibration exercise has been distributed to the participating laboratories (ref. NY89).

17.3.4 Participants

In the final analysis, the following 21 laboratories participated (3 from Denmark, 3 from Finland, 4 from Norway and 11 from Sweden):

- Risø National Laboratory
- State Radiation Hygiene Institute, Denmark
- Laboratory for Electrophysics, Danish Technical Highschool, Copenhagen

- Dept. of Radiochemistry, University of Helsinki
- Finnish Centre for Radiation and Nuclear Safety, Helsinki
- Dept. of Environmental Hygiene and Toxicology, National Public Health Institute, Kuopio

- Isotope Laboratory, Agricultural University of Norway
- Institutt for energiteknikk, Kjeller
- National Institute of Radiation Hygiene, Norway
- Institute for Anorganic Chemistry, Technical University of Norway

- Swedish National Institute for Radiation Protection
- Ringhals Nuclear Power Plant
- Oscarshamn Nuclear Power Plant
- Forsmark Nuclear Power Plant
- National Defence Research Establishment, Umeå
- National Environment Protection Board, Stockholm
- Institute for radiophysics, Umeå University
- Institute for radiophysics, Sahlgrenska Hospital, Göteborg
- Institute for radiophysics, The Hospital, Lund
- Studsvik
- Swedish University of Agricultural Sciences, Uppsala

17.4 Introduction to the C base system

The software foundation - named the C_base system - of the NCDB is developed at Risø as a continuation of an earlier system that has been available for several years. A detailed review of the of the C_base system is given in the technical report (ref. LI89).

The C_base system consists of:

- a basic binary file system.
- a number of modules handling input/output in text-form.
- modules for various calculations, selection, sorting, plotting etc.

To use these there are several user-interface programs, i.e.:

FILEHANDler - to perform all normal data and file operations including a special keyboard window module and a macro facility (ref. LI90a).

MAPHANDler - to draw maps and show labels and/or values from records according to coordinates in the record (ref. LI90b).

The two examples shown in Figures 17.1 and 17.2 give an idea of the illustrative possibilities of the C_base system:

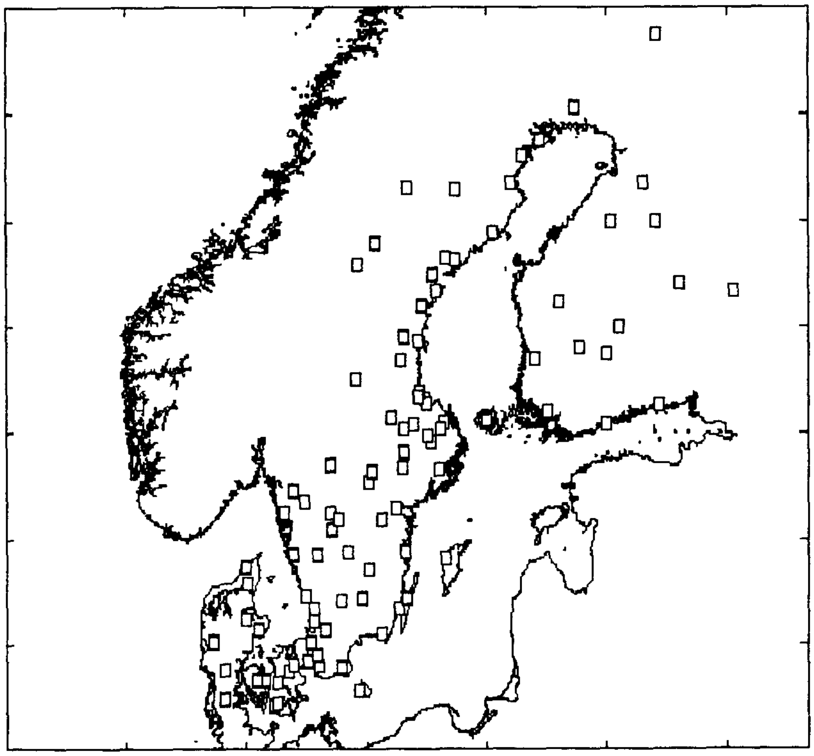

Figure 17.1 Milk sampling locations May - August 1986 (except Norway).

17.5 Computer application

The programs and data of the NCDB are developed for and stored on different computer installations:

PC-version: to be used on any IBM-PC compatible machine. Data and code is interchanged using ordinary diskettes, primarily 3.5" - 720 kb and 5.25" - 1.2 Mb. Problems may arise for the graphical parts due to the many different types of displays. The FILEHANDler uses no graphics. The MAPHANDler is available in a preliminary version.

VAX-version: may be used on any DEC-VAX - machine; data and code is moved on tapes. Access to the machines may be possible via telephone lines or public networks. The FILEHANDler may be used on VAX-machines running the VMS operating system. The MAPHANDLer is not implemented in the VAX-version.

UNIX-version: not implemented presently. An implementation on UNIX/386 is foreseen.

A special transfer file method may be used to move data between different systems.

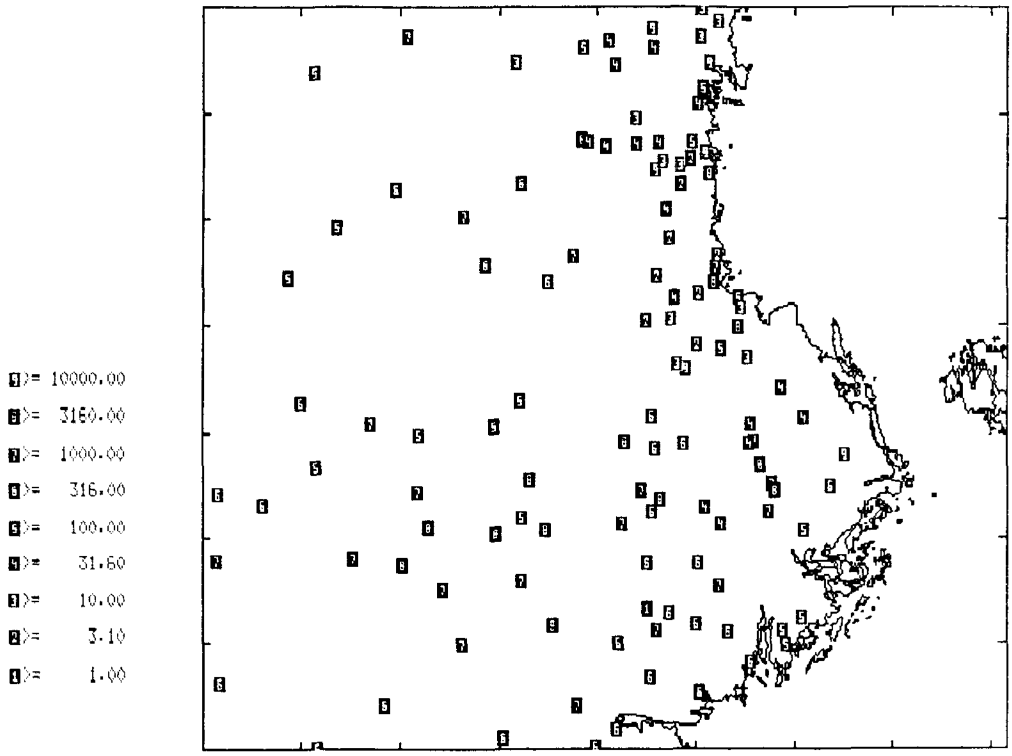

Figure 17.2 I-131 measurements, Central Sweden, May 1986.

17.6 The file structure of the NCDB

The data belonging to the NCDB will be stored in C_base binary files entered by the contributors of the data and assembled into national files. All such files will be interchanged by the national NCDB working group members. The final details of storage and distribution is to be defined. A coming version of the technical report will describe the actually entered data in more detail.

17.7 Access to the NCDB

The working group of the NCDB decided that only institutions that contribute data to the data base shall have access to the base. The working group also decided to delegate the responsibility for the national data to the national working group members. They will be assisted technically from Risø where the final accumulation into a master copy of the NCDB and subsequent distribution will take place. It is the intention to keep parallel versions on VAX and on PC's, from which tapes and diskettes may be copied to other users. The necessary software will be distributed in the same manner. On line access to the VAX'es may be provided through the national working group members.

17.8 Status of the NCDB

The present status of the software development part of the project is such that operating versions of the programs have been distributed to the national working group members and some instructions have been given locally at their laboratories. The development process is continued through improvements to the exsisting functions and introduction of some new help-functions.

Further additions, like some statistical analysis, plotting of time functions and correlations etc. will be made. Also improvements of the MAPHANDler program, used to plot data on maps are planned.

Below follows a list of some of the tasks that remain to be performed:

Sample types: The list in Risø-M-2832, appendix D may need some additions to be agreed upon.

Location names or area codes exist for most of the data. They have to be defined and put into the data base.

Location coordinates are an essential part of the NCDB. Unfortunately they only exist presently for Danish, some Finnish and some Swedish locations.

References to origin of data. A list of participants in the measurements and their publications should be collected.

Measurement results. Values exist, and are going to be entered into the NCDB in many different forms. Most are without any measured uncertainty. As it has been decided to include uncertainty and sample collection and preparation quality in the records, some additional work must be done to fulfill this requirement.

The process of actually inserting data into the NCDB is now about to begin. The easiest way to do this will be to arrange the data into some machine readable form and use one of the input functions in the FILEHANDler to transfer the figures. Additional treatment may be necessary to make the data comply to the intended standard representation.

The foreseen relations to the REM-bank at Ispra have to be updated. In principle the technical problems should be only small.

18. CONCLUSIONS

The conclusions that can be drawn from the results of the AKTU projects have impact upon many facets of accident consequence assessment, ranging from e.g. new computational tools developed within the program to e.g. recommendations concerning food preparation methods to be utilized in a fallout situation. The full conclusions are, of course, to be found in the separate chapters of this report. Some of the important conclusions are, however, summarized here.

The computational tools developed are mainly the developed RIMPUFF program for atmospheric dispersion and a program for calculation of shielding factors of houses.

A computational tool of a different type, developed as part of the AKTU program, is the Nordic Chernobyl Data Base. Data handling and evaluation procedures have been developed; and although data compilation has not progressed as far as had been hoped, the Data Base will be an important source of information in the future.

Some of the projects have approached areas where little or no work has been done previously, like the project on winter conditions, the project on physico/chemical form of radionuclides in the Chernobyl fallout, and the project on resuspension. The conclusion from the first of these projects is that the impact of an accident or fallout situation occuring during winter may be considerably smaller than of a similar situation during summer conditions. The most important conclusion from the second of these projects is that bioavailability of radiocesium in soil (and also in resuspended radiocesium) is much less than of radiocesium in plant material, taken up via the roots. In this project an instrument for simultaneous measurement of particle size and electrical charge of particles/colloids in solution has also been developed. In the third of these projects it was found that the resuspension factor is several orders of magnitude lower than the values traditionally cited, and that for most weather conditions resuspension is a local phenomenon (order of magnitude some kilometers).

The development and large-scale testing of mitigating actions to prevent uptake of radiocesium to animals in a fallout situation is also one of the projects where new ground has been broken sucessfully. Practical and efficient methods have been developed, and these have already had an economic benefit far exceeding the cost of the project, and in a longer perspective probably the whole AKTU program. Another attempt at developing mitigating actions, in this case to prevent radiocesium uptake to fish, did not lead to any efficient approach, but the project has nevertheless added to the knowledge of such situations.

Altogether the AKTU program has been very successful, adding much new knowledge and insights to the area of accident consequence assessment and related areas. Through the Nordic project work numerous aspects of the methods have been improved, so that consequences of nuclear accidents can be assessed with higher certainty. Some of the aspects examined are typical for problems encountered in the Nordic countries, such as winter conditions, uptake in animals grazing in the mountain areas and the various types of mitigating actions developed. However, most of the results are universal in character.

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APPENDIX

RESEARCH PROJECTS INITIATED BY THE NATIONAL INSTITUTE OF RADIATION PROTECTION, STOCKHOLM, SWEDEN, ON ACCOUNT OF THE CHERNOBYL ACCIDENT.

DISPERSION ROUTES

The Chernobyl Accident
A meteorological analysis of how radionuclides reached Sweden.
Christer Persson, Swedish Meteorological and Hydrological Institute (SMHI), Henning Rodhe, Meteorological Institue of Stockholm University (MISU), Lars-Erik De Geer, Swedish Defence Research Establishment (FOA).
SSI P 356.86 *

Measurements of the initial levels of I-131 and Cs-137 in air, pasture and milk from the Studsvik area after the Chernobyl accident.
Ulla Bergström, Rune Axelsson, Yvonne Stiglund, Studsvik.
SSI P 373.86 *

Comparison of predicted and measured Cs-137 concentrations in a lake ecosystem.
Ulla Bergström, Sture Nordlinder, Studsvik.

Chernobyl fallout studies in the area of Studsvik.
Björn Sundblad and Sverker Evans, Studsvik.
SSI P 381.86 *
SSI P 381.87

PROBLEMS RELATED TO THE CONDITIONS IN POPULATED REGIONS

Weathering and migration of Chernobyl fallout.
Intermediate results of in situ measurements in the Gävle and Studsvik areas.
Olle Karlberg, Björn Sundblad, Studsvik.
SSI P 345.86 *
SSI P 354.86 *
SSI P 369.86 *

Dry deposition velocities on grass at Studsvik after the Chernobyl accident, Part 1.
Anders Appelgren, Studsvik.
SSI P 345.86 *

Analysis of particles in the fallout from Chernobyl, Part 1.
Jan Chyssler and Lennart Devell, Studsvik.
SSI P 345.86

Analysis of particles in the fallout from Chernobyl, Part 2.
Jan Chyssler, Studsvik.
SSI P 345.86

Fallout nuclides in the snow cover.
Björn Sundblad, Studsvik.
SSI P 345.86

Iodine forms in fallout from Chernobyl **
Bård Normann, Lennart Devell, Studsvik.
SSI P 345.86 *

An estimate of the nuclide inventory in the Chernobyl reactor core.
Ove Edlund, Erik Johansson and Lennart Devell, Studsvik.
SSI P 354.86 *

Deposition velocity and fractioning of different radionuclides **
Variability of deposition.
Influence of topography and micrometeorology on deposition and redistribution of fallout.
Analysis of particle activity and particle size.
R. Finck et al., FOA.
SSI P 359.86

Resuspension of radionuclides **
I. Vintersved, FOA.
SSI P 359.86 x)

Shielding factors - Buildings **
R. Finck et al., FOA.
SSI P 359.86 x)

Retention and migration of Chernobyl fallout in Sweden.
Intermediate results after one year of in-situ measurements in the Gävle and Studsvik areas.
Olof Karlberg, Studsvik.
SSI P 379.86 * x)
SSI P 424.87

x) Organized as AKTU project

NORDIC CHERNOBYL DATA BASE

Swedish Chernobyl data base **
The National Institute of Radiation Protection, Stockholm.
SSI P 360.86

MEASUREMENTS, SURVEYS, ANALYSES

Terrestrial ecosystems

Measurements from aircraft of the cesium fallout at the wings of Söderhamn and Uppsala **
H. Mellander, SGAB, Uppsala.
SSI P 442.87 *

Measurements from aircraft of the cesium fallout within the Gävle area **
H. Mellander, SGAB, Uppsala.
SSI P 443.87 *

Measurements of fallout by using high-resolution gamma spectroscopy **
R. Finck, FOA.
SSI P 375.86 *

Plutonium, americium and curium isotopes in the Chernobyl fallout**
E. Holm, The Department of Radiation Physics, University of Lund.
SSI P 417.87

Analysis of fallout data: Preparation, documentation etc for the GRECA meeting, 87-01-14--15 **
Lennart Devell, Studsvik.
SSI P 395.87 *

Nuclide composition of Chernobyl hot particles.
Lennart Devell, Studsvik.
SSI P 412.87 *

Characteristics of the Chernobyl release and fallout.
Lennart Devell, Studsvik.
SSI P 458.87 *

Evaluation of Chernobyl data obtained at Studsvik **
Lennart Devell, Studsvik.
SSI P 459.87

Effect of atmospheric deposition processes and surface contamination on accident consequences, their mitigation and emergency response planning.
Lennart Devell, Studsvik.
SSI P 509.88

Urban storm water transport and wash-off of cesium-137 after the Chernobyl accident.
Sven Halldin et al., Department of Physical Geography, University of Uppsala.
SSI P 348.86 *

Measurement of radioactive substances in sludge from Gothenburgh and Lund **
B. Erlandsson, The Department of Radiation Physics, University of Lund
S. Mattsson, The Department of Radiation Physics, University of Gothenburg.
SSI P 344.86

Measurement of deposition of radioactive substance in Gothenburg and Lund **
B. Erlandsson, The Department of Radiation Physics, University of Lund
S. Mattsson, The Department of Radiation Physics, University of Gothenburg.
SSI P 344.86

Agricultural research after Chernobyl.
The Department of Radioecology, The Swedish University of Agricultural Sciences, Uppsala.
SSI P 353.86 *
SSI P 472.88

- Depth distribution of radiocesium in agricultural soils in Chernobyl fallout areas of Sweden in 1987-88.
  H. Lönsjö

- Transfer of cesium to grassland crops in the Chernobyl fallout areas in Sweden in 1986 and 1987.
  Å. Eriksson, H. Lönsjö and K. Rosen.

- Use of Chernobyl field data in modelling cesium transfer to grassland crops.
  Åke Eriksson

- Behaviour of fallout Cs-137 in grassland experiments.
  Åke Eriksson

- The transfer of radiocesium from pasture to milk.
  Karl J. Johansson et al.

- Observations on the transfer of Cs-137 from soils to barley crops in Sweden after the Chernobyl fallout in 1986.
  Åke Eriksson and Klas Rosen.

- The importance of potassium fertilization for control of the cesium transport from polluted fiords in the Northern Sweden **
  Klas Rosen

- The radioecology of the pasture land **
  Gunnel Karlén and Karl J. Johansson.

Sampling at experimental farms **
Measurements of hay, ensilage and meat samples.
L. Hagblad, SCB, and Gammadata Mätteknik AB.
SSI P 452.87
SSI P 453.87

Measurements of pasture plants **
Å. Eriksson, The Department of Radioecology, The Swedish University of Agricultural Sciences, Uppsala.
SSI P 454.87.

Transfer of Cs-134, Cs-137 and I-131 from grass to cow's milk. A field study after the Chernobyl accident.
E. Håkansson et al., The Department of Radiation Physics, University of Gothenburgh.
SSI P 350.86 *

Transport through important foodchains in Northern Sweden. Uptake, turnover and transport of radioactive cesium (Cs-137 and Cs-134) in a boreal forest ecosystem **
(Abstract in English)
R. Bergman et al., The National Defense Research Establishment, Umeå.
SSI P 423.87

Levels of Cs-137 in Northern Swedish moose during the first year after the Chernobyl accident.
Kjell Danell and Per Nelin, Department of Wildlife Ecology, Swedish University of Agricultural Sciences, Umeå.
Göran Wickman, Department of Radiation Physics, University of Umeå.
SSI P 376.86 *

Radiocesium in a forest ecosystem **
F. Andersson, The Swedish University of Agricultural Sciences.
SSI P 406.87 *

Radiocesium in Swedish forests and wild food.
A pilot study after Chernobyl.
Jerry S. Olsson, University of Uppsala.
SSI P 406.87 *

Measurement of the concentration of radioactive elements in different materials appropriate for bio-energy production, including waste **
B. Erlandsson, The Department of Radiation Physics, University of Lund
S. Mattson, The Department of Radiation Physics, Universtiy of Gothenburg.
SSI P 397.87

Transfer coefficients for uptake of Cs-137 in berries **
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The external dose distribution in Sweden after Chernobyl **
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The elk and his fodder-plants **
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Cesium-137 measurements in elk **
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L. Albanus, The Swedish National Food Administration, Uppsala.
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Follow-up of the Chernobyl research during the time interval May-September 1987 **
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Mechanisms associated with transfer of radionuclides to shars and trouts via different food chains in lakes with natural as well as water regulated regimes **
G. Neumann and M. Notter, The National Environmental Protection Board.
J. Hammar, The National Board of Fisheries.
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Transport of fallout in food-chains in a lake **
P.-O. Agnedal, Studsvik.
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Evert Andersson and Anders Broberg, The Limnologic Institution. University of Uppsala.
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Transport in food-chains in a lake **
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Radioactivity in freshwater fish immediately after the Chernobyl accident. Turnover of Cs-137 in lake ecosystems **
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Lars Brydsten and Mats Jansson, Department of Physical Geography, University of Umeå.
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The distribution of radioactive substances and their exposure routes in the ecosystems of the Bothnian Sea **
Distribution, re-distribution and retension time in sediment and biota in Örefjärden.
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Per-Olof Agnedal, Studsvik.
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Inventory and radioactivity studies in the Baltic Sea after the Chernobyl accident in USSR **
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Participation in the DHI expedition in the Baltic Sea, August 1987 **
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Accumulation of radiocesium in the bottom fauna in the Baltic Sea **
S. Evans, Studsvik.
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S. Forberg, The Royal Institute of Technology, Stockholm.
B. Jones, The Swedish University of Agriculture Sciences, Uppsala.
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S. Forberg, B. Carell and Torbjörn Westermark, The Royal Institute of Technology, Stockholm
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Radiocesium in perch from lakes in the Northern Sweden after Chernobyl. The actual situation, causal relationships and the future. Methods to reduce the content of mercury and cesium in fish **
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S. Mattsson, The Department of Radiation Physics, University of Gothenburg.
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The Chernobyl accident - transport of radionuclides to humans living in Northern Sweden.
Lennart Olofsson and Hans Svensson, Radiation Physics Department, University of Umeå.
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Whole-body measurements - Farmers **
The National Institute of Radiation Protection, Stockholm.
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Statistics, Sweden, and R. Falk, The National Institute of Radiation Protection.
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Ulf Bäverstam and Rolf Falk, National Institute of Radiation Protection, Stockholm.

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Örjan Hultåker, Skandinavisk Opinion AB, Stockholm.
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Britt-Marie Drottz, Lennart Sjöberg, Psykologisk Metod AB, Stockholm.
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The reactions of the Swedish people following the Chernobyl accident**
Örjan Hultåker, Skandinavisk Opinion AB, Stockholm
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"What did they really say", in radio and TV regarding the Chernobyl accident, and how did we meet with the news? **
Olle Findahl et al., The Psychological Institution, University of Stockholm.
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The reaction of young people following the Chernobyl accident **
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* The project is completed
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