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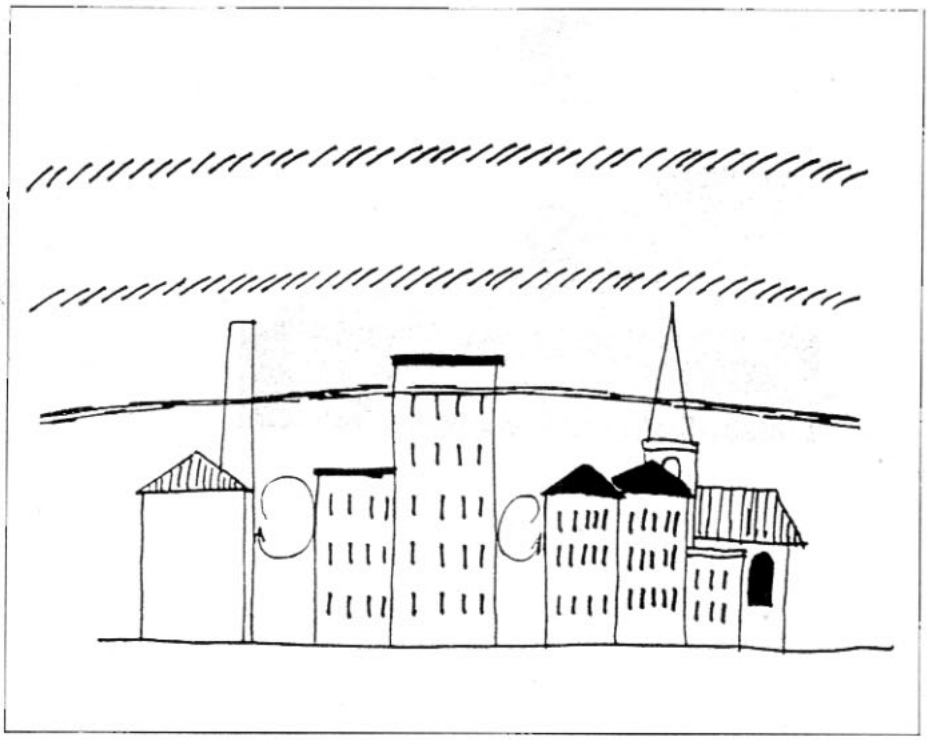

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DEPOSITION AND REMOVAL OF RADIOACTIVE SUBSTANCES IN AN URBAN AREA

Final Report of the NKA Project AKTU-245

Jørn Roed
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October 1990

ABSTRACT

Radiation doses received by the population of a contaminated urban area have been estimated. Possible dose reduction measures and their cost-effectiveness are investigated. Potentially important parameters influencing the doses have also been studied. They include distribution of contamination following both wet and dry deposition, run-off, weathering, shielding, resuspension, indoor deposition, the relative airborne concentrations indoors and outdoors, and forced decontamination. It is shown that contamination of the green areas in an urban complex is generally a major contributor to dose. A study of the cost-effectiveness of different clean-up procedures indicates that decontamination of green areas and streets are relatively cost-effective and would rank highly in a list of priorities. Following a contamination due to a reactor accident, the dose rate to an individual will generally be less in an urban area than in a rural environment.

INIS Descriptors:

ATMOSPHERIC PRECIPITATIONS; BUILDINGS; COST BENEFIT ANALYSIS; DECONTAMINATION; DOSE RATES; FALLOUT DEPOSITS; PARTICLE RESUSPENSION; PLANTS; RADIATION DOSE DISTRIBUTION; REMEDIAL ACTION; SHIELDING; SURFACE CONTAMINATION; URBAN AREAS; WASHOUT; WEATHERING.

SUMMARY

It is more than 30 years since the consequences of a severe nuclear accident were described in the US Reactor Safety Report WASH-740. Many of the shortcomings of WASH-740 were overcome in WASH-1400 (also known as the Rasmussen report) which dealt with both the probability and conseqences of severe nuclear accidents. The report showed to what extent the reactor type and the siting could influence the consequences of an accident. Similar risk studies were subsequently conducted, notably in Germany.

From these risk studies it appeared that the worst credible accident would be where the core melted completely and the containment (if any) ruptered and released fission products to the atmosphere. In this event, iodine-131 and to a lesser extent ruthenium would present the major short term external radiation hazard, and isotopes of caesium the major long term external radiation hazard. Caesium isotopes together with strontium-90 would be the main sources of internal radiation through ingestion of contaminated agricultural produce.

Until about 5 years ago, when studying off-site consequences of nuclear accidents, little or no account was taken of the special conditions encountered when the contamination reached urban areas. This was a remarkable oversight in view of the fact that the great majority of the population of Western Europe lives in towns. Perhaps the reason for this was that virtually no input data was available to permit reasonable calculations and predictions. In order to make the calculate the effects of radioactive deposition in urban areas some of the important factors to take into account are

- the relative distribution of deposited radionuclides following both wet and dry deposition
- the effect of run-off to surface water drains
- the effect of weathering, especially for roofs and paved areas

- the decontamination effected by road traffic, and street cleaning
- the degree of resuspension, i.e. the return of deposited material from the ground to the atmosphere
- the effectiveness of man's effort to reduce radiation, through decontamination and reclamation

The accident at Chernobyl in 1986 presented an opportunity to study the behaviour of radioactive fallout in the urban area. This opportunity was used to study many of the above parameters, mainly through real measurements in Scandinavia as well as in Germany.

It appears that very few of the data applicable to the contaminated rural area are also valid for the urban area. Furthermore, the distribution pattern of fallout in an urban area is very dependent on whether the deposition took place in dry or wet weather.

It was found that the degree of interception of the fallout varies according to the type of urban surface and the physico-chemical form of the radionuclides. Only little iodine is retained on hard surfaces, while the retention of caesium and ruthenium is very significant on these surfaces.

In-situ measurements following wet deposition have shown that in the first few days after deposition about 60% of the caesium is removed from asphalt, concrete and granite pavements by runoff and street cleaning.

Dry deposition velocities of caesium to vertical walls has been found to be very low while they were 5-10 times greater on roads. On rough surfaces such as corrugated roofs and grass the deposition velocities are even higher.

Deposition indoors depends on furnishing in the room. Deposition indoors has now been shown to be a relatively important source of radiaton in houses that are well-shielded by its walls towards radiation from the outside.

People present in contaminated areas are shielded by various obstacles. The shielding factor, relative to the doserate measured one meter above a grassed area can range from 0.5 to 10,000. The low figure refers to a case where trees and bushes are present. The highest figure should be used for well shielded basements. This range illustrates the importance of using accurate shielding factors in dose calculations.

The investigations have confirmed that a marked reduction in inhalation dose during the passage of a radioactive cloud can be achieved by remaining indoors. For modern Scandinavian houses, this reduction is about a factor of 3.

Using numercial values for the various parameters studied, it has been possible to illustrate how different surfaces contribute to dose rates to people present in an urban area. Generally, in the case of dry deposition, the garden with trees and bushes present the main source of radiation. Roofs are also important, especially in an environment dominated by small houses. In the case of wet deposition, the contribution to dose comes mainly from the ground (garden surface and street surface), but roofs are also important.

The efficiency of different reclamation procedures has been investigated and their costs studied. The cost-effectiveness of appropriate methods for clean-up of various surfaces and environments could also be demonstrated. From these considerations, a list of decontamination priorities has been drawnup. In the case of dry deposition, cutting back trees and bushes, removing top soil and digging gardens would be given top priority. Cleaning streets would also rank highly. Cleaning roofs will carry a lower priority on account of cost rather than effectiveness. For wet deposition, removal of soil and digging of gardens would carry the highest priority.

In practice, for many urban contamination scenarios a dose reduction factor of about 4 can be achieved.

The overall conclusion is that realistic input data are now available for use in calculation of dose and assessing the reduction of dose that could be achieved through different clean-up procedures.

In general, doses received by individuals in urban areas due to a reactor accident will be smaller than in a rural area at the same atmospheric conditions. Thus, earlier calculations of doses received by urban populations have been grossly overestimated. This is of particular importance in Western Europe as the vast majority of the population here spends most of its time in builtup areas.

Summary in Danish

SAMMENFATNING

For mere end 30 år siden blev de mulige konsekvenser af et alvorligt reaktoruheld beskrevet for første gang i den amerikanske reaktorsikkerhedsrapport WASH 740.

Mange af de svagheder, som denne rapport havde, blev først korrigeret af WASH 1400, den såkaldte Rasmussen-rapport, som beskæftiger sig med såvel konsekvenser som sandsynligheder i forbindelse med alvorlige reaktoruheld. Rapporten viste, hvorledes såvel reaktortypen som reaktorens beliggenhed er af stor betydning for konsekvenserne af et reaktoruheld.

Tilsvarende reaktorsikkerhedsundersøgelser blev også foretaget andre steder. Ikke mindst en stor tysk undersøgelse har været af betydning for at klarlægge de ricici, som er forbundet med uheld på kernekraftværker.

Alle undersøgelserne viser, at det værst trolige uheld vil være en kernenedsmeltning kombineret med et brud på reaktorindeslutningen (hvis en sådan eksisterer) med udslip af store mængder radioaktive stoffer til atmosfæren til følge.

Ved et sådant uheld vil jod-131 og i mindre grad ruthenium udgøre den største fare for ydre og indre stråling i den første fase af uheldet. Cæsium-isotoperne vil på længere sigt udgøre den største fare for ydre stråling, og sammen med strontium-90 vil de være hovedansvarlige for den indre stråling forårsaget af indtagelse af fødevarer, som er radioaktivt forurenet.

Indtil for blot fem år siden tog man meget lidt eller ingen hensyn til de specielle forhold, som gør sig gældende, når man skal beregne doser til befolkningen i et forurenet byområde. Dette er især bemærkelsesværdigt, når man betænker, at langt de

fleste mennesker i Vesteuropa bor i byerne.

Grunden til denne før så utilstrækkelige behandling af byområdernes radioøkologi var måske de meget få relevante data, som fandtes, til hjælp for dosisberegningerne her.

For at foretage sådanne beregninger korrekt, må man have kendskab til en række forhold (parametre)

- den relative fordeling af de deponerede radioaktive stoffer, som følge af såvel våd som tør deponering,
- virkningen af at det forurenede regnvand kan løbe væk fra hårde overflader og ende i kloaksystemet,
- vejrets eroderende virkning, herunder ikke mindst regnvandets afvaskning af tage og veje,
- den afslibende og rensende virkning af trafik og gadefejning,
- virkningen af resuspensioner, d.v.s. den proces at allerede deponeret materiale igen hvirvles tilbage i luften,
- virkningen af de bestræbelser som gøres for at reducere dosis gennem en oprensning af det forurenede område.

Uheldet i Chernobyl-reaktoren i 1986 gav os mulighed for at undersøge det radioaktive nedfalds skæbne i et byområde. Denne mulighed blev udnyttet især i de nordiske lande og desuden i Tyskland.

Undersøgelser viser at kun få af de data, som er indsamlet i agerbrugs og skovområder, kan bruges i byerne. Det har også vist sig, at det mønster i hvilket det radioaktive nedfald fordeler sig er helt afhængigt af om det regner eller om det er tørvejr.

Det viser sig, at den mængde af det deponerede materiale, som tilbageholdes på forskellige hårde overflader, er afhængig af overfladernes beskaffenhed samt af den fysisk-kemiske form af det deponerede materiale. Således bliver meget lidt jod tilbageholdt, mens en stor del af det deponerede cæsium og ruthenium

bliver bundet på forskellige hårde overflader.

Målingerne viste også, at omkring 60% af vådt deponeret cæsium bliver fjernet fra asfalt, cement og brolægning i løbet af de første få dage som følge af regnens afvaskning og gadefejning.

Deponering i tørvejr viser overraskende lave hastigheder på lodrette vægge. På veje er de 5-10 gange højere, og på ujævne overflader som tage og græsarealer er de væsentlig højere end på veje.

Indendørs deposition har vist sig at være stærkt afhængig af møbleringen. I tidligere undersøgelser har man anset det materiale, som deponeres indendøre, for at være uden betydning for dosisbidraget. Vore beregninger viser imidlertid, at det indendørs deponerede radioaktive materiale kan give et ikke uvæsentlig bidrag til dosis, ikke mindst i huse som er velafskærmet fra den stråling, som kommer udefra.

Mennesker, som befinder sig i byområder, er skærmet fra den radioaktive stråling på forskellig måde. Afskærmningsfaktoren i forhold til dosishastighed 1 m over en stor græsplæne kan gå fra en halv på steder med træer og buske til 10.000 i velafskærmede kælderlokaler. Denne store variation i afskærmningsfaktor viser, hvor væsentlig det er at anvende de rigtige skærmningsfaktorer i dosisberegninger.

Undersøgelserne har endvidere bekræftet, at der kan opnås en væsentlig reduktion i indåndningsdosis ved at opholde sig indendøre under passage af en radioaktiv sky. Denne reduktion er fundet til at være ca. 3 for moderne skandinaviske huse.

Ved at anvende de forskellige parameterværdier, som vore undersøgelser har frembragt, har det kunnet demonstreres i hvilket omfang de forskellige overflader i byerne bidrager til dosishastigheder, som befolkningen udsættes for. Når det radioaktive

materiale er deponeret tørt, vil træer, buske og græs være de største bidragsydere til dosis. Hustage er også vigtige ikke mindst i områder med mange enfamiliehuse. Ved deponering med regn vil hovedbidraget til dosis komme fra horisontale overflader som haver, græsplæner, veje og pladser, men hustage kan også her være vigtige bidragydere til dosis.

Forskellige oprensningsmetoders effektivitet er blevet undersøgt, og omkostningerne anslået. Omkostnings-effektiviteten af en række rimelige metoder er også testet for en række overflader i forskellige byområder. Ud fra disse beregninger har det været muligt at opstille en prioritetsliste for de forskellige metoders anvendelse i forskellige situationer.

Når man ser på tørdeponering, vil beskæring af træer og buske, fjernelse af det øverste lag jord og havegravning stå højest på prioritetslisten. Gadefejning vil også have høj prioritet. Afrensning af tage vil have en noget lavere prioritet, mere som følge af de store omkostninger end som følge af manglende effektivitet. Er de radioaktive stoffer deponeret med regn, vil afskrabning af det øverste jordlag og havegravning have topprioritet.

I praksis vil den dosisreduktion, man vil kunne opnå ved en fornuftig oprensningsprocedure, være omkring en faktor 4.

Hovedkonklusionen, som kan drages af dette arbejde, er at realistiske parameterværdier nu er tilgængelige til beregning af dosis til bybefolkningen, eller hvor man ønsker at finde frem til den mulige dosisreduktion som følge af en oprensning.

I almindelighed vil den dosis, en person i et byområde får som følge af et reaktoruheld, være mindre end den han ville få, hvis han befandt sig på landet under samme vejrforhold. Tidligere beregnede doser til bybefolkningen har således været stærkt overvurderede. Dette er af speciel interesse i Vesteuropa, hvor størsteparten af befolkningen tilbringer det meste af deres tid i byerne.

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1. INTRODUCTION

In the early years of nuclear power, the safety of nuclear power plant, was assessed in terms of the maximum credible accident which would result in the notional release of a small quantity of radioactivity, the effects of which would be limited to a few kilometers from the plant and the only countermeasures called for, would be to agriculture and agricultural products. However, from risk studies conducted in the 1970's, it became apparent that the worst credible accident would be where the core melted and a large amount of radioactive material was released to atmosphere. Much of the released radioactivity would eventually be deposited on the ground surface and present a potential radiation hazard to man.

The potential for widespread contaminations was tragically realized by the accident in Chernobyl in 1986 when some 135.000 had to be evacuated from a zone within 30 km of the plant.

In view of the fact that most of the populatiom of Western World spends the majority of its time in built-up areas, it is remarkable that it is only about 5 years since due attention was given to the special problems of the urban radioecology.

The accident at Chernobyl presented a unique opportunity to study the behaviour of radioactive fallout deposited in urban areas. We have taken this opportunities to determine how dose to the population of a contaminated urban area can be assessed and how to reduce that dose through decontamination and reclamation. We have studied those parameters which we believe to be necessary for such calculation used the data to demonstrate what can be achieved in terms of dose reduction for various urban contamination scenarios. The important factors which we have studied to satisfy the input requirement for our calculations were

- The relative distribution of deposited radionuclides following both wet and dry deposition
- The effect on run-off to surface water drains
- The effect of weathering, especially for roofs and paved areas
- The decontamination effected by road traffic, and street cleaning
- The degree of resuspension, i.e. the return of deposited material from the ground to the atmosphere
- The effectiveness of man's effort to reduce radiation, through decontamination and reclamation

Most of the parameter values has been obtained through field measurements in Denmark, Sweden and West Germany.

2. DRY DEPOSITION

2.1 Introduction

Pollution in air takes the form of liquid drops, reactive and noncreative gases, and aerosols. Dispersed pollution can be removed from the air by various processes. Removal in the absence of precipitation and fog is normally called dry deposition; when precipitation is present, it is called wet deposition. In between, we have deposition under foggy conditions.

All three forms of deposition are of particular interest when assessing the consequences of reactor accidents in the context of risk assessment. The reason for this is that external gamma doses delivered by the deposited material are often major contributiors to acute effects, and doses from deposited long-lived contaminants are usually the major long-term hazards.

In risk assessment it is extremely important to deal with deposition in urban areas as this is where most of the population of the European Community lives.

But it is enough to know the total deposition in urban areas in order to make realistic dose calculations. The spatial distribution of the deposition in the urban area must also be known.

It is important therefore to know the deposition on the most common surfaces in the area, e.g. roofs, walls, streets, bushes, trees, gardens and lawns.

2.2 Definitions

To describe dry deposition, Gregory [**1945**], and Chamberlain and Chadwick [**1953**] introduced the concept of deposition velocity

(2.1)

where Z_d is the distance from the surface at which V_d is determined, $F(Z_d)$ the flux of the contaminant towards the surface considered at a distance Z_d from the surface, and $\chi(Z_d)$ the concentration of the contaminant at the same distance.

The applicability of the concept of deposition velocity in the urban complex was discussed by Underwood [**1987**]. He pointed out the convenience of relating the downward flux of contaminant to the mean concentration near the surface via a known coefficient of proportionality, namely the deposition velocity. This enables the problem to be factored into (1) dispersion outside the influence of near surface phenomena, and (2) behaviour near the surface where the two regions are separated by an imaginary boundary z. The V_d concept provides a relationship at z, which thus acts as a boundary condition on the equation representing dispersion in the outer region.

2.3 The Urban Area

In the case of an urban area, the V_d may vary not only as a function of pollutant characteristics, meteorological variables, and surface characteristics, but also as a function of such variables as the downwind distance from the rural-urban transition or other local transitions in the urban complex, such as that from a building cluster or a park.

It is suggested that one way of solving these problems might be to use "local deposition velocities" V_d^1

V_d^1 is defined as

where F(i) is the flux towards a local surface, e.g. a roof or a wall), and $\chi(z)$ is the air concentration at the imaginary boundary surface well above the roughness elements of the city that are also above the city canopy. These "local deposition velocities" can then be used for calculating the total flux to the area and then the deposition velocity over the urban surface. Such a simplified model was proposed by Roed [**1987c**].

The surface types, i.e. the local surfaces, can be assigned their own invidual deposition velocities, each obtained as the result of experiments or calculations. Thus the ratio of the deposition velocity of the urban canopy to the area as a whole is the weighted aggreate of the local deposition velocity, i.e.

$$V_d \text{ (urban)} = \Sigma A_i \cdot V_d^1$$

where A_i is the total surface type "i" in a horizontally projected area of the city.

The simplified model contrasts with the usual one which makes use of the overall aerodynamic roughness length of the urban complex (the macrosurface roughness). In the former case the spatial proximity of various microsurfaces plays no part,whereas in the latter case is very important.

However, the total deposition in both cases is dependent on the density of bluff bodies such as buildings, the simplified model giving a higher deposition velocity because of larger integrated area per projected horizontal area.

2.4 Measurements before Chernobyl

In order to find the local velocities onto selected urban surfaces, Roed [**1983 and 1985**], measured the deposition of ^{137}Cs - mainly bomb fallout accumulated over many years - on the surface of a building; he then related it, after applying a correction

for radioactive decay, to the known time-intergrated air concentration of ^{137}Cs. Also, he measured the deposition of naturally produced ^{7}Be on artificial plates placed against vertical walls.

This type of measurements has the advantage that the surfaces studied have been immersed in an actual turbulent environment generated by wind flow on an array of buildings and that the deposition velocity is averaged over a time enough to include a wide variety of weather conditions.

The measurements also have a number of drawbacks such as:
1) The areas of plane surfaces chosen in the experiment may not be representative for a number of reasons: deposition could be highly non-uniform spatially, for example with enchancement occurring near edges, discontinuities, projections, etc. This calls for measurements of large surface areas at different types of locations.

2) The ^{137}Cs deposited on walls had an unknown contribution from wet deposition for some of the samples, whereas others were well protected from rain. Weathering can diminish the deposition. Roed [**1983**] presented an argument to explain why weathering was not expected to have a dominant influence on the results, and the ^{7}Be results bear this out.

3) The characteristics of the aerosols associated with the deposition of ^{137}Cs are not known in detail, whereas those associated with ^{7}Be have a mean aerodynamic size of about 0.4 μm.

The values of local deposition velocities obtained were notably low. Values for ^{137} Cs onto vertical surfaces largely protected from the rain were below 10^{-4} m S^{-1}. The ^{7}Be results for vertical surfaces not exposed to rain were below 1.6×10^{-4} m S^{-1} and horizontal surfaces below 7×10^{-1}.

2.5 Measurements after Chernobyl

There is a paucity of experimental data on dry deposition on urban surfaces.

Roed's measurements [**1987b, 1987c and 1988**], however, have provided some insight into how various isotopes are distributed on different surfaces. These despoition measurements were made during the passage of the first radioactive cloud from the Chernobyl over the Roskilde area. The measurements were carried out at noon on Sunday 27 April; the cloud cleared the area sometimes during the follcwing week. When the deposition took place the weather was not changeable: the mean wind speed was 3 m S^{-1} at 8 m above the ground and the Pasquille stability category was B-C.

The measurements were taken in the city as well as in suburban and rural areas.

The measured deposition velocities are listed in Tables 2.1 and 2.2. Table 2.1 shows the deposition velocities for different isotopes orginating from the Chernobyl accident and Table 2.2 shows the deposition on different urban surfaces relative to deposition on roads.

Table 2.1 Deposition Velocity in 10^{-4} m S^{-1}.

Isotope	I	Cs	Ru	Ba	Ce	Zr
Paved areas	4.6	0.7	3.5	4.6	8.1	3.5
Walls	3.0	0.1	0.4	0.4	0.9	1.3
Windows	2.3	0.05	0.1	0.2		0.1
Grass (clipped)	22	4.3	4.1	5.8	7.7	7.1
Trees	8.0	7	25	26	39	45
Roofs	33	2.8	3.4	53	40	

Table 2.2 Deposition on Various Urban Surfaces Relative to Deposition on Paved areas.

Isotope	I	Cs	Ru	Ba	Ce	Zr
Paved areas	1	1	1	1	1	1
Walls	0.6	0.2	0.1	0.1	0.1	0.2
Windows	0.5	0.1	0.04	0.040		0.02
Grass (clipped)	5	6	1.1	1.2	1.0	1.0
Trees	17	10	7	6	13	6
Roofs	7	4	1	12	13	

There is no obvious indication that the deposition velocity changed from one area to another. It clearly differed for various isotopes, however. Particle-bound caesium had the smallest values, with a mean V_d of about 1×10^{-4} m s^{-1} for road surfaces. The next group consisting of particulate ruthenium, lanthanum, and elementary iodine had deposition velocities of around 5×10^{-4} m s^{-1}. The highest deposition velocity, 10×10^{-4} m s^{-1}, was found for particulate cerium and zirconium. The deposition velocity of iodine was similar to that on road surfaces. For caesium, however, it was one order of magnitude lower. The wall surface samples were identical, as they had been prefabricated in our laboratory for deposition velocity measurement purposes. However, the walls of which they were part were situated at very different locations, varying from very open areas to very dense city areas. Nevertheless, the deposition velocities were suprising similar. The deposition velocities of caesium, lanthanum, and cerium were some 5-10 times higher than on roads.

Only ruthenium had the same deposition velocity on both roads and walls.

The deposition velocities of the volatile group of elements (I, Te, Cs, Ru) are lower than those of the refractory group (La, Ba, Ce, Zr). As shown by Rulik et. al. [17], these two groups have different particle sizes: the first group has an AMAD of about 0.4 μm.

Dry deposition velocities reported by Magua et. al. (1987) for ^{137}Cs and ^{131}I on grass are shown in Table 2.3.

Table 2.3 Deposition Velocities for ^{137}Cs and ^{131}I, derived from measurements at the RWTH Aachen after the Chernobyl accident. (Magua et. al. 1987)

Nuclide	Remarks	$v_{d.grass}$ (cm/s)
^{137}Cs	all samples considered	0.03 - 0.15 mean: 0.07
	calculated with fitted curves	0.05 ± 0.01
^{131}I	total iodine:	
	mean for daytime minimum	0.15
	overall mean	0.2
	iodine soecies*):	
	elemental:	
	mean for daytime minimum	0.5
	overall mean	0.8
	particle bound:	0.1

*) calculated with 30% elemental, 30% particle bound and 40% organic iodine.

Nicholson (1987) reported dry deposition velocities for vertical surfaces and roofs, and these values are shown in Table 2.4.

Sehmel (1980) showed the importance of gravitational particle settling as a deposition mechanism. He suggested that the settling velocity for 1 μm diameter particles is of the order of 10^{-4} m^{-1} while those for 3, 5, and 10 μm particles can be of the order of 10^{-3}, $3x10^{-3}$, and 10^{-2}m s^{-1}.

Tabel 2.4 Deposition velocities (cm s^{-1}).

Location	Deposition Velocities (cm s^{-1}) ^{134}Cs	^{137}Cs (Total)	^{137}Cs (Weapons fallout)
Building Briks			
Norwich	$<5 \times 10^{-4}$	2×10^{-3}	$>4.4 \times 10^{-3}$
Harwell Lab.	$<4 \times 10^{-3}$	1×10^{-2}	$>1.2 \times 10^{-2}$
Clay Roof Tiles (Building 1)			
North: Upper	6×10^{-2}	6×10^{-2}	6×10^{-2}
Lower	9×10^{-2}	11×10^{-2}	11×10^{-2}
Mean	8×10^{-2}	8×10^{-2}	8×10^{-2}
South: Upper	8×10^{-2}	12×10^{-2}	13×10^{-2}
Lower	7×10^{-2}	8×10^{-2}	9×10^{-2}
Mean	7×10^{-2}	10×10^{-2}	11×10^{-2}
Clay Roof Tiles (Building 2)			
East	4×10^{-2}	5×10^{-2}	5×10^{-2}
West	5×10^{-2}	6×10^{-2}	6×10^{-2}
South	3×10^{-2}	7×10^{-2}	8×10^{-2}
East: Upper	$<4 \times 10^{-2}$	7×10^{-2}	8×10^{-2}
Middle	9×10^{-2}	8×10^{-2}	6×10^{-2}
Lower	5×10^{-2}	6×10^{-2}	6×10^{-2}
Mean	6×10^{-2}	7×10^{-2}	7×10^{-2}
Concrete Roof Tiles (Building 3)			$^{137}Cs/^{134}Cs$ (Surface Activity)
East Upper	4.2×10^{-2}	5.5×10^{-2}	2.0
East Lower	4.5×10^{-2}	5.8×10^{-2}	2.0
West Upper	3.9×10^{-2}	5.0×10^{-2}	2.0
West Lower	3.2×10^{-2}	4.7×10^{-2}	2.2
Roofing Felt (Building 4)			
Flat Roof	8×10^{-2}	18×10^{-2}	3.4

* Deposition velocities could be up to 50% greater.

2.6 Deposition on trees and grass.

The deposition on trees in a forest area can be important as an attenuation mechanism, but for a forest close to an urban areas and in which institutions have been placed, dosemetric aspects must also be taken into account. Boserup forest in which we have made measurements is one of this type; it is placed close to Roskilde, a town of 80.000 including suburbs. Boserup forest is only about 8 km from the center of Roskilde, and is a popular area for excursions and sports activities for Roskilde as well as Copenhagen 30 km away. Besides this, a large mental hospital with about 2000 patients and a large staff is located within this forest. In order to calculate the dosemetric effect of deposition in woods it is necessary to know the distribution of the activity on the trees. For trees in a city or suburban area the local deposition velocity and its distribution are needed as input parameters for making dose calculation inside and outside houses.

2.6.1 Experimental conditions.

The first cloud from the Chernobyl release arrived under dry weather conditions at the Roskilde area, where the measurements were carried out at noon Sunday, April 27th, 1986; the cloud cleared the area sometimes during the following day. The dry weather conditions persisted throughout the folowing week.

In the time interval during which the deposition took place the weather continued unchanged with a mean speed of 3 m/s at 8 meters height and Pasquill stability category of B-C.

The airborne radioactivity was measured by sucking air through a Whatman glass-fibre paper and measuring the material collected using gamma spectroscopy. Such filters provide an efficiency close to 100% for particulate pollution. Thus, for isotopes existing only in particulate form, representative deposition velocities can be calculated based on the airborne activity

collected on the glass-fibre filters.

For iodine, however, a problem arises as this element can be present in the atmosphere in three forms: (i) attached to particles, (ii) as elemental iodine vapour, and (iii) gaseous organic compounds of iodine. Organic iodine is deposited neither on glass-fibre filter, nor significantly on surfaces, so it can be excluded from further consideration. Of the remaining forms of iodine, only the partculate fraction is found in the filter, whereas the major fraction of the deposition may arise from the more rapidly deposited iodine vapour. Calculated deposition velocities are therefore unrepresentative of either form. However, some measurements made in Germany (Schwibach, 1986) indicate that the levels of elemental iodine in the initial Chernobyl cloud were about equal to those of the particulate fraction. Thus the deposition velocities given here provide an approximate value for the elemental iodine component (asuming the composition of the cloud reaching Roskilde to be similar to that observed in Germany). The measured deposition velocities can therefore be considered as those of elementary iodine.

The investigation was carried out in the position of Boserup forest 5 km South-west of Risø, consisting mainly of common spruce with an average height of about 6.4 m. Two trees chosen at random were felled and were cut into sections, one in 8 and the other 4. The branches and needles were then chopped into pieces and the deposition on each section was measured separately, as well as the cortex of each section. To find the total deposition the number of trees per m^2 forest area were found and samples of the forest soil were taken.

In the case of trees from the suburban area, only the local deposition velocity was of the interest; a yew tree 2.5 m high was measured in two sections.

The material deposited on the two common spruces chosen from the

Boserup forest was very evenly distributed per unit mass of bulk material (small branches, twigs, and needles). Besides the total deposition velocity, it is therefore interesting to know as well the amount of bulk mass per unit forest area, as the even distribution indicates that the total deposition velocity is proportional to the bulk mass per unit forest area within the limitation of the ability of the atmosphere turbulence to carry enough material to the boundary layer at the canopy of the forest.

Tables 2.5 and 2.7 shows the distribution of the deposited material on branches, twigs, and needles with height above ground. The bulk deposition with height is shown in Tables 2.6 and 2.8. In Tables 2.9 and 2.11 the distribution of the deposition on the cortex of each tree is shown, and Tables 2.10 and 2.12 show the deposition per unit area of cortex. The total deposition velocity of the forest is given in Table 2.13. It is calculated as the total deposited material on the trees and on the forest soil per unit area divided by the integrated air concentration. A yew and juniper berry tree have been cut in the urban environment. They were part of a tight hedge in a suburban front garden. Table 2.14 shows the local deposition velocity, i.e. the deposited material material per unit horizontal projected area covered by the tree. The mean bulk deposition constant, i.e. the deposited material per unit mass of small branches, twigs, and needles divided by the time-integrated air concentration for all the trees are shown in Table 2.15. It is seen that the bulk deposition constant is about the same for both trees in suburban area and equal to that of forest trees.

In Table 2.16 is given the local deposition velocity and a bulk deposition constant Bd for grass, is given as the deposited material per unit mass of grass, divided by the time integrated air concentration. When modelling deposition on trees and on grass it seems that the important parameters to be used are the mass of the bulk material and the bulk deposition constant.

Table 2.5

Deposited Material on Branches, Twigs and Needles of Common Spruce at Different Heights, in Bq

Tree no. 1

Height	cm	0-75	75-135	135-215	215-315	315-405	405-472	472-545	545-654	0-654
^{7}Be	Bq	6.3	8.1	11.6	11.2	19.3	11.2	26.4	7.9	102
^{95}Nb	"	19.4	72.2	57.0	28.0	95.2	52.2	69.7	63.1	456.7
^{95}Zr	"	10.4	37.4	36.0	16.6	59.5	30.4	40.2	36.2	266.8
^{103}Ru	"	4.7	14.0	10.6	17.9	16.8	17.3	26.8	19.5	127.5
^{106}Ru	"	0.71	4.6	3.1	4.0	3.8	5.1	8.7	7.8	37.7
^{131}I	"	226.7	226.2	282.2	294.7	-	182.6	585.5	116.5	1916.4
^{134}Cs	"	0.97	1.5	2.0	3.2	3.6	2.2	5.4	2.9	21.6
^{137}Cs	"	2.7	3.8	4.4	7.2	8.9	5.0	11.7	7.4	50.9
^{141}Ce	"	7.7	30.0	52.7	22.7	49.3	39.0	57.4	32.3	291.2
^{144}Ce	"	5.1	22.8	25.2	14.4	34.6	27.0	42.7	16.7	188.5
^{152}Eu	"	-	-	-	0.55	-	0.16	0.60	0.40	1.71
^{154}Eu	"	0.07	0.31	0.18	0.23	0.73	0.09	0.15	0.68	2.44

Table 2.6

Deposited Material on Branches, Twigs and Needles of Common Spruce in Bq per m^2 of Cortex

Tree no. 1

Height	cm	0-75	75-135	135-215	215-315	315-405	405-472	472-545	545-654	mean 0-654
^{7}Be	Bq/kg	7.4	6.5	6.6	6.6	8.1	7.5	10.4	4.9	7.3
^{95}Nb	"	22.8	57.7	32.6	16.5	39.7	34.8	27.3	39.4	33.9
^{95}Zr	"	12.3	30.0	20.6	9.8	24.8	20.3	15.8	22.6	19.5
^{103}Ru	"	5.5	11.2	6.1	10.5	7.0	11.5	10.5	12.2	9.3
^{106}Ru	"	0.84	3.7	1.8	2.3	1.6	3.4	3.4	4.9	2.7
^{131}I	"	266.7	181.0	162.4	173.4	-	121.7	229.6	72.8	172.5
^{134}Cs	"	1.1	1.2	1.2	1.9	1.5	1.5	2.1	1.8	1.5
^{137}Cs	"	3.1	3.0	2.5	4.2	3.7	3.3	4.6	4.6	3.6
^{141}Ce	"	9.1	24.0	30.1	13.3	20.5	26.0	22.5	20.2	20.7
^{144}Ce	"	6.0	18.2	14.4	8.5	14.4	18.0	16.8	10.4	13.3
^{152}Eu	"	-	-	-	0.32	-	0.11	0.24	0.25	0.2
^{154}Eu	"	0.052	0.25	0.10	0.14	0.30	0.06	0.059	0.43	0.1

Table 2.7

Deposited Material on Branches, Twigs and Needles of Common Spruce at Different Heights, in Bq

Tree No. 2

Height of tree	cm	0-125	125-275	275-390	390-610	0-610
^{7}Be	Bq	14.2	19.5	15.9	56.1	105.6
^{95}Nb	"	67.5	150.7	44.6	243.7	506.5
^{95}Zr	"	51.3	56.3	27.0	153.6	288.2
^{103}Ru	"	-	22.2	9.1	78.5	109.7
^{106}Ru	"	4.5	7.5	-	17.2	29.2
^{131}I	"	148.2	197.0	339.9	792.3	1477.3
^{134}Cs	"	1.8	3.6	3.5	13.0	21.8
^{137}Cs	"	3.9	8.4	7.7	29.0	48.9
^{141}Ce	"	42.9	49.9	24.8	171.6	289.2
^{144}Ce	"	27.3	35.9	20.1	110.4	193.6
^{152}Eu	"	0.71	0.56	0.13	1.92	3.3
^{154}Eu	"	0.16	0.42	0.93	0.81	2.3

Table 2.8

Deposited Material on Branches, Twigs and Needles of Common Spruce at different Heights, in Bq per Kg (Branches, Twigs and Needles)

Tree No. 2

Height of tree	cm	0-125	125-275	275-390	390-610	mean 0-610
^{7}Be	Bq/kg	10.9	11.5	8.1	13.2	11.0
^{95}Nb	"	57.9	88.7	22.9	57.3	56.7
^{95}Zr	"	39.5	33.1	13.9	36.2	30.7
^{103}Ru	"	-	13.0	4.7	18.46	12.1
^{106}Ru	"	3.5	4.4	-	4.1	4.0
^{131}I	"	114.0	115.8	174.0	186.4	147.6
^{134}Cs	"	1.4	2.1	1.8	3.0	2.1
^{137}Cs	"	3.0	5.0	3.9	6.8	4.7
^{141}Ce	"	33.0	29.4	12.7	40.4	28.9
^{144}Ce	"	21.0	21.1	10.3	26.0	19.6
^{152}Eu	"	0.55	0.33	0.067	0.45	0.35
^{154}Eu	"	0.12	0.25	0.48	0.19	0.26

Table 2.9
Deposited Material on Cortex of Common Spruce
at Different Heights, in Bq
Tree no. 1

Height	cm	0-75	75-135	135-215	215-315	315-405	405-472	472-545	545-654	0-654
^{7}Be	Bq	1.6	1.7	2.1	1.7	1.9	1.2	0.93	2.5	13.6
^{95}Nb	"	8.3	0.27	0.19	0.19	-	0.51	0.31	2.2	12.0
^{95}Zr	"	6.0	0.37	0.16	0.16	-	0.25	0.49	1.4	8.8
^{103}Ru	"	2.5	0.13	0.18	0.24	1.5	0.22	0.22	0.49	5.5
^{106}Ru	"	1.1	0.0030	0.38	0.15	-	0.069	-	0.14	1.8
^{131}I	"	15.6	10.1	14.2	13.8	-	-	248.8	117.7	420.2
^{134}Cs	"	0.19	0.052	0.017	0.11	0.053	-	(0.0080)	0.056	0.49
^{137}Cs	"	0.45	0.22	0.27	0.27	0.18	0.060	0.030	0.16	1.6
^{141}Ce	"	4.8	0.090	-	-	0.44	-	-	0.75	6.1
^{144}Ce	"	3.4	0.20	0.13	-	-	0.26	-	0.75	4.7
^{152}Eu	"	0.040	-	-	0.015	-	-	-	-	0.055
^{154}Eu	"	0.12	0.089	0.17	0.13	-	0.098	0.064	0.030	0.70

Table 2.10
Deposited Material on Cortex of Common Spruce
at Different Heights, in Bq per m^2 of Cortex
Tree No. 1

Height	cm	0-75	75-135	135-215	215-315	315-405	405-472	472-545	545-654	mean 0-654
^{7}Be	Bq/m^2	5.5	8.3	8.4	6.1	8.9	9.4	8.5	23.1	9.8
^{95}Nb	"	29.4	1.4	0.78	0.68	-	3.9	2.9	20.9	8.6
^{95}Zr	"	21.2	1.9	0.66	0.61	-	1.9	4.5	13.1	6.3
^{103}Ru	"	8.8	0.67	0.71	0.88	7.2	1.7	2.0	4.6	3.3
^{106}Ru	"	3.8	0.016	1.5	0.53	-	0.52	-	1.3	1.3
^{131}I	"	55.3	50.7	56.9	50.0	-	-	2284.3	1675.0	695.4
^{134}Cs	"	0.68	0.26	0.069	0.22	0.25	-	(0.074)	0.53	0.30
^{137}Cs	"	1.9	1.1	1.1	1.0	0.83	0.47	0.25	1.5	1.0
^{141}Ce	"	17.0	0.47	-	-	2.1	-	-	7.1	6.7
^{144}Ce	"	12.1	1.0	0.54	-	-	2.0	-	7.0	4.5
^{152}Eu	"	0.20	-	-	0.050	-	-	-	-	0.13
^{154}Eu	"	0.42	0.44	0.66	0.47	-	0.74	0.59	0.28	0.51

Table 2.11
Deposited Material on Cortex of Common Spruce
at Different Heights, in Bq
Tree No. 2

Height of tree	cm	0-125	125-275	275-390	390-610	0-610
^{7}Be	Bq	1.7	1.1	1.7	3.5	8.0
^{95}Nb	"	0.35	7.3	3.0	-	10.7
^{95}Zr	"	0.45	4.4	1.7	0.068	6.6
^{103}Ru	"	0.23	0.94	0.33	0.22	1.7
^{106}Ru	"	0.13	0.073	0.086	0.36	0.65
^{131}I	"	263.0	103.7	28.9	9.6	405.2
^{134}Cs	"	0.11	0.13		0.083	0.33
^{137}Cs	"	0.40	0.44	0.17	0.23	1.2
^{141}Ce	"	-	2.1	0.24	0.57	2.9
^{144}Ce	"	0.13	1.7	0.93	0.26	3.0
^{152}Eu	"	-	0.074	-	0.13	0.13
^{154}Eu	"	0.065	0.074	0.21	0.052	0.40

Table 2.12
Deposited Material on Cortex of Common Spruce
at Different Heights, in Bq per m^2 of Cortex
Tree No. 2

Height of tree	cm	0-125	125-275	275-390	390-610	mean 0-610
^{7}Be	Bq/m^2	4.4	3.1	8.3	20.0	9.0
^{95}Nb	"	0.91	20.7	14.4	-	12.0
^{95}Zr	"	1.2	12.5	8.3	0.39	5.6
^{103}Ru	"	0.59	2.7	1.6	1.2	1.5
^{106}Ru	"	0.32	0.21	0.42	2.1	0.76
^{131}I	"	678.1	293.4	139.9	54.6	291.5
^{134}Cs	"	0.28	0.37	0.046	0.47	0.29
^{137}Cs	"	1.0	1.2	0.82	1.3	1.1
^{141}Ce	"	-	5.9	1.2	3.3	3.5
^{144}Ce	"	0.32	4.7	4.5	1.5	2.8
^{152}Eu	"	-	-	-	0.72	0.72
^{154}Eu	"	0.17	0.21	1.0	0.29	0.42

Table 2.13

Deposition velocity in a forest: (units: 10^{-4} ms^{-1})
40.5 trees per 100 m^2, average tree height = 6.4 m

Isotope	Common spruce
^{134}Cs	7.3
^{131}I	89
^{103}Ru	28
^{106}Ru	53

Table 2.14

Local deposition velocities: (units: 10^{-4} ms^{-1})

	Yew trees	Juniper berry
Isotope	Height 2.5 m	Height 2 m
^{134}Cs	9	3
^{131}I	105	32
^{141}Ce	46	23
^{144}Ce	46	28
^{140}La	32	14
^{103}Ru	32	13
^{106}Ru	47	28
^{95}Zr	55	26
^{95}Nb	58	26

Table 2.15

Bulk Deposition Constant, B_d: (in kg^{-1} m^3 $s^{-1} \cdot 10^{-4}$)

	Yew trees	Juniper berry	Common Spruce	Common Spruce
Isotope	Height 2.5 m	Height 2.0 m	Height 6.5 m	Height 6.1 m
^{137}Cs	2.8	3.2	1.8	2.3
^{134}Cs	2.2	2.7	1.4	1.9
^{131}I	24.5	26.5	19.4	16.6
^{141}Ce	12.2	21.9	13.9	19.4

Table 2.16

Deposition velocity V_d: 10^{-4} m.s.$^{-1}$,
bulk deposition B_d: 10^{-4} m^3.s^{-1}.kg^{-1}, for grass.

Sample No.		^{137}Cs	^{134}Cs	^{131}I
1384	V_d	4.3	4.4	22
	B_d	21	21	110
1387	V_d	1.8	1.5	18
	B_d	10	8.7	100
1388	V_d	8.8	7.2	93
	B_d	10	8.5	110
1391	V_d	6.0	6.6	86
	B_d	7.9	8.7	110
1392	V_d	7.4	9.9	120
	B_d	9.1	12	140

3. WET DEPOSITION

Precipitation scavengeing or washout of particles and gases from the atmosphere can be a significant contributor to ground deposition. In the case of Chernobyl fallout it has led to areas of very high deposition even at large distances (>2000km) from the reactor site and in those areas wet deposition far overweighed the dry deposition.

Run-off is a term used to describe the deposited rainwater which is not retained on the area receiving the rainfall. This phenomenon has been extensively studied in the context of hydrology.

As run-off water can retain and carry away some of the radioactive material falling on impervious surfaces such as roads and roofs it is clearly of importance in consequence assessments. Since materials deposited in urban areas may not be retained there and the radiation dose to the local populace would therefore be reduced.

The total run-off can consist of surface run-off and infiltration, where infiltration is the flow of water through the soil surface. Generally construction materials in the urban environment are sufficiently impervious to prevent infiltration. For these surfaces the following equation is valid.

$$Q = P - I_a$$

where Q is the direct run-off in mm, P the total rainfall in mm, and I_a the initial accumulated rainfall in mm prior to run-off. Ritchie et. al. (1976) assumed that for an urban area the run-off from artificial surface would be essentially 100% for all rainfall above an initially accumulated 3 mm. Had rain fallen within the previous hour the run-off will occur sooner.

In these model, Ritchie et. al. assumed that the concentration of radioactive material in the run-off water was equal to that in rainwater. Roed (1987) however, showed that the amount of run-off from roofs was very sensitive to the type of construction material. For a rainfall (P) of 9.2 mm shortly after the Chernobyl accident he found for roofs with a slope of 45^{o}, that I_a values were 1.8 mm for cement tile, 4.2 mm for red clay tile, 1.4 mm for eternite (an asbestos type of material) and $\sim$ 0 mm on silicone treated surfaces. If Ritchie's model represented the true situation, all the radioactive material on the silicone treated roof should have remained in the run-off water. This was clearly not the case, however, as the concentration in the run-off water was considerably less than that in the rainwater for three of the four elements. Table 3.1 shows the concentrations in run-off water relative to these in precipitation.

Table 3.1. Concentration of radioelements in run-off water relative to that in rainwater for a precipitation of 9.2 mm.

	Isotope			
Surface	Cs	I	Ru	Ba
Cement tile	0.49	1.24	0.56	0.40
Red tile	0.55	1.05	0.65	0.58
Eternite	0.14	1.18	0.30	0.37
Silicone treated eternite	0.74	1.00	0.52	0.67

In some recent experiments at Risø, a similar situation was observed for run-off of caesium on road surfaces. For rainfall of 6 mm I_a of 3.8 were observed for asphalt and 3.4 mm for concrete. The ratio of the concentration of radioactive ^{137}Cs in run-off water to that in rainwater was 0.16 for asphalt and 0.21 for concrete.

The retained wet deposition is defined as the amount of radioactive material retained on a given surface after cessation of the precipitation which carried the material. Wet deposition measurements in urban areas have been made at Risø, Munich and in Gävle, Sweden.

The distribution of wet deposited material is very dependent on run-off and of windspeed and direction when considering the walls, as the amount of rainwater deposited on the walls is dependent on the angle of the direction of the precipitation to that of the wall surfaces.

At Risø the retained wet deposition on different roofs relative to the deposition on a grassed area were measured after a rainfall of 9.2 mm. The measurements are shown in Table 3.2 It can be seen that of the roof material, only red clay tiles retained a measurable amount of radioiodine. The silicone treated roofs retained 25-35% of the caesium and the lanthanum, but the retention of ruthenium was similar for all the roof material.

Table 3.2

Relative wet deposition on different roofs.

	Slope	^{137}Cs	^{134}Cs	^{131}I	^{106}Ru	^{103}Ru	^{140}La
Grassed area	0^{o}	1	1	1	1	1	1
Cement tile	45^{o}	0.58	0.62	0	0.42	0.27	0.68
Red tile	45^{o}	0.68	0.71	0.43	0.60	0.53	0.69
Corrugated eternite	45^{o}	0.87	0.88	0	0.65	0.64	0.68
Silicon treated eternite	45^{o}	0.12	0.29	0	0.63	0.41	0.33
Corrugated eternite	30^{o}	0.80	0.81	0	0.59	0.55	0.63
Silicon treated eternite	30^{o}	0.18	0.25	0	0.49	0.47	0.22

3.1 Weathering

Following the initial interception of fallout by a surface, that surface will be exposed to the weather. Weathering, the action of rain, snow, frost etc. will tend to deplete the adsorbed radioactive material. Sometimes, man's activities such as routine street cleaning are also included under this heading.

Two years after wet deposition of fallout from Chernobyl on the town Gävle in Sweden Roed and Sandalls (1989) determined the distribution of radiocaesium on walls, roads and paved and grassed areas. The measurements were made in the town centre and in an industrial area. The relative levels of contamination are given in Tables 3.3 and 3.4.

Table 3.3. Distribution of retained wet deposited caesium 137 after two years of weathering in town centre of Gävle.

Surface	Relative distribution
Grassed area - 85.000 Bq/m^2	1
Plastered wall no. 1	0.009
Plastered wall no. 2	0.007
Concrete paved area no. 1	0.008
Concrete paved area no. 2	0.11
Concrete paved area no. 3	0.08
5.8 m wide road	0.02

Table 3.4. Distribution of retained wet deposited caesium 137 after two years of weathering in industrial area in Gävle.

Surface	Relative distribution
Grassed area (130.000 Bq/m^2)	1
Brick Wall no. 1	0.030
Brick wall no. 2	0.013
Brick wall no. 3	0.008
Brick wall no. 4	0.003
Plastered walls	0.009
Asphalt (car park)	0.026
Asphalt (crossroad)	0.009

Wash-off has been investigated for roof materials at Risø and the findings are shown in Table 3.5, both the caesium isotopes and the ruthenium isotopes were retained very efficiently: between 4.5% and 0% being removed from the roofs depending on the roof material. The corrugated eternite roof still retained virtually all the caesium eight months after deposition. The contamination was much more readily removed by weathering of the concrete tile: between 2 and 4.5 per cent of the ruthenium being removed per month.

The fraction removed by wash-off had fallen from about 1% of the retained material for caesium in August to 0.7-0.8% in December. The corresponding figures for ruthenium are 1.4 and 1.2-1.3%. The wash-off effect per unit of rainfall for caesium

Table 3.5 Comparison of the wash-off from roofs in September and in December 1986. Figures expressed as % of retained material and as % of retained material per mm rainfall.

Isotope	Roof	August 1986				December 1986			
				Mean				Mean	
		(%)	(%mm^{-1})	(%)	(%mm^{-1})	(%)	(%mm^{-1})	(%)	(%mm^{-1})
^{137}Cs	1	1.4	0.044	1.0	0.031	2.1	0.051	0.8	0.017
	2	1.8	0.056			0.9	0.020		
	3	0	0			0.1	0.001		
	4	1.5	0.048			0.8	0.020		
	5	0	0			0	0		
	6	1.3	0.040			0.6	0.015		
^{134}Cs	1	1.3	0.040	0.9	0.026	2.0	0.046	0.7	0.016
	2	1.6	0.050			0.7	0.018		
	3	0	0			0.1	0.001		
	4	1.2	0.037			0.7	0.016		
	5	0	0			0	0		
	6	1.0	0.031			0.5	0.013		

Key to roof type:

1. Cement tile 45^0 slope
2. Red tile 45^0 slope
3. Corrugated eternite 45^0 slope
4. Silicon-treated eternite 45^0 slope
5. Corrugated eternite 30^0 slope
6. Silicon-treated eternite 30^0 slope

fell from 0.028% per mm rainfall in August 1986 to 0.016% in December. For ruthenium the corresponding figures were 0.045% in August and 0.026% in December.

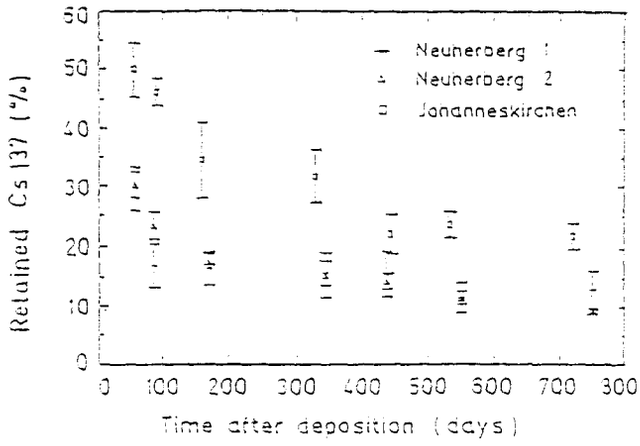
The amount of caesium retained on roofs has also been measured in and around Munich (Roed and Jacob 1990). In Neuherberg shortly after the Chernobyl accident, 70% of the deposited caesium was retained after a brief precipitation of 5.6 mm of rain. The event was followed by a relatively dry week with less than 0.7 mm rain on each of the days, contributing a total of 20% to the total retained. In Neuherberg the exact deposition was not known, but since the deposition was 30% less than in Neuherberg, it can be assumed, that the contribution of the wet deposition was also less in Neuherberg. The characteristics of the roofs are given in Table 3.6.

Table 3.6. Characteristics of the roofs and houses, for which measurements have been performed in Munich area.

Designation	Number of floors	Tile	Pitch of roof
Neuherberg 1	3	clay, hard burnt surface	45°
Neuherberg 2	1	concrete	22°
Johanneskirchen	2½	concrete	20°

Figure 3.1 shows that in Johanneskirchen, due to the smaller amount of precipitation during the deposition, relatively more was retained on the roof than in Neuherberg. In agreement with the result from Risø, the weathering in the period 6 months to 2 years after the deposition was small. On the other hand, in the period 2 to 5 months after the deposition, about 40% of the activity was removed from roofs by weathering, in contrast to the results from Risø. (Roed 1987a)

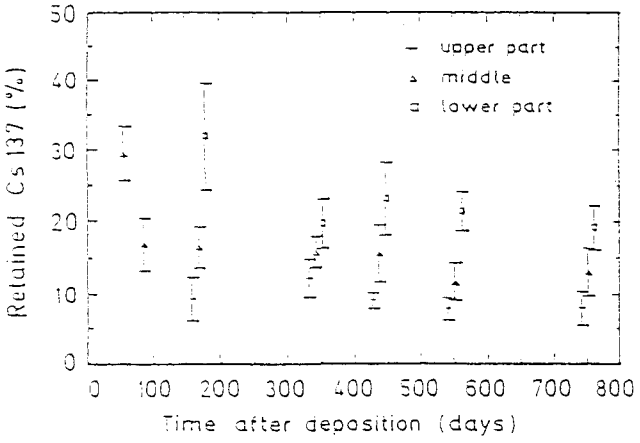
Figur 3.1



Amount of Cs 137 retained on roofs in Munich area after the Chernobyl fallout. Mean value and standard deviation of the results for 5 tiles are given for each measurement.

Figure 3.2 suggest, that the distribution of caesium on the roof may be inhomogeneous. On the roof "Neuherberg 2" a more homogeneous distribution was found.

Figure 3.2

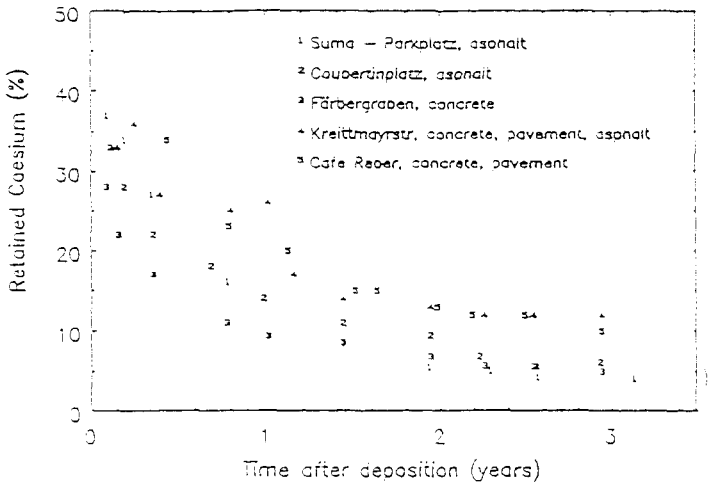


Amount of CS 137 retained on different parts of the roof "Neuherberg 1" after the Chernobyl fallout.

In situ gamma-ray spectrometric measurements were performed at 48 sites in Southern Baveria (Jacob et al 1987). Results for impermeable surfaces (it is assumed that the migration of the cesium into the surface can be neglected in these cases) are given in Figure 3.3. The sites are car parks, pedestrian zones and narrow streets without heavy traffic. It is evident from Figure 3.3, that due to the gaps between the cobble stones,

caesium is better retained at these sites than at sites with asphalt surfaces or where the pavings consists of large (50 x 50 cm) concrete slabs.

Figure 3.3



Caesium retained at 5 sites in Southern Bavaria as inferred from in situ gamma-ray spectrometry.

Together with the measurements performed at Risø (Roed 1987a and Roed and Sandalls 1989), and the measurements performed by Karlberg (1990), it is possible to give an approximate distribution for wet deposited caesium and iodine immediately after deposition and for caesium about two years later. These data are given in Tables 3.7 and 3.8.

Table 3.7. Relative distribution of wet deposited caesium (5-10 mm of precipitation).

Surface	Immediately after deposition	2 years after deposition
Grassed area	1	1
Paved areas		0.01 - 0.05
heavy traffic	0.4 - 0.8	
light traffic		0.05 - 0.2
Roofs	0.3 - 0.9	0.1 - 0.7
Walls	0.01 - 0.03	0.01 - 0.03

Table 3.8. Relative distribution of wet deposited iodine (5-10 mm of precipitation).

Surface	Immediately after deposition
Grassed area	1
Paved area	0 - 0.03
Roofs	0 - 0.04
Walls	0.01 - 0.03

4. RESUSPENSION

The processes that may bring deposited material to be suspended in the air is called resuspension.

Resuspension can cause inhalation hazards, recontaminated surfaces, redistribution of the deposited material, and contamination of crops.

Allthough after the Chernobyl accident especially many measurements has been performed, it is still difficult to predict resuspension (Garland and Pattenden 1989).

The resuspension factor is defined as:

$$R(z) = \frac{\chi(z)}{G}$$

where $R(z)$ is the resuspension factor at a height z (unit length) over the surface and is measured in (unit length)$^{-1}$. $\chi(z)$ is the airborne resuspended radionuclide concentration at height z (Bq per unit volume) and G is the concentration of the radionuclide in Bq per horizontal plan-projected unit area.

As most of the people in Western countries live and work in suburban or urban areas, resuspension in the urban environment is of great importance, especially for calculations of inhalation dose.

Measurements made by Sehmel (1973) showed the importance of vehicular traffic in the resuspension of small particles. He observed that the fraction of particles resuspended per vehicle passage on a contaminated road was in the range 10^{-5} to 10^{-2} m^{-1} on the day the tracer was deposited on the road surface. The fraction resuspended was a function of the speed of the vehicle.

Linsley (1978) found a resuspension factor of 5×10^{-5} - $5 \times 10^{-4} m^{-1}$ by dividing the air concentration of dust by the amount of dust on unit area of road surface. Using the same approach, Kaul and Roberts (1983) estimated a value of 10^{-6} m^{-1}. Bjurman et al. (1987) using Chernobyl fallout measured resuspension of radioactive particles at four different locations in Sweden during the second half of 1986. For ^{137}Cs and ^{134}Cs, the resuspension was found to be in the range $4 \times 10^{-9} m^{-1}$ to $3 \times 10^{-8} m^{-1}$ depending on the surroundings. The highest resuspension factors were measured in Stockholm and the lowest values were measured in a rural area over a surface of grass and exposed rock. They found that the degree of resuspension probably depends on local conditions in an area of the order of few tens of kilometers radius from the measuring point. They did not find a marked reduction in the resuspension factor at the end of December when the area was covered with snow. This seems to indicate that the contribution from field deposition is low and that the source for resuspension may be trees. Other possible sources, such as particles from higher air layers, or directly from the area around Chernobyl, can be ruled out as major contributors since there is a clear correlation between the air concentration and the surface concentration when the latter varies by more than a factor of one hundred.

The resuspension factors for the four areas studied by Bjurman are given in Table 3.1. The values of the resuspension factors in May 1986 may have been up to a factor of five higher than

in June 1986 but the values are uncertain because of difficulties in separating the resuspended material from the as yet undeposited material in the atmosphere.

Table 4.1. Resuspension in Sweden in second half of 1986 (Bjurman et al 1987)

Local environment	Resuspension factor
grass	$7 \cdot 10^{-9} m^{-1}$
grass with bare rocks	$4 \cdot 10^{-9} m^{-1}$
town	$3 \cdot 10^{-8} m^{-1}$
gravel	$3 \cdot 10^{-9} m^{-1}$

5. SHIELDING

In calculating the gamma radiation dose from the deposited radioactivity the doserate 1 m above an infinite, smooth plane source is normally used as a reference. The actual doserate at a given location is then found by multiplying this reference doserate with a modififying factor, the so-called shielding factor (S), which is defined as

$$S = \frac{D}{D_{ref}}$$

where D is the doserate at a given location and D_{ref} the reference doserate 1 m above an infinite smooth source with a uniform distribution of contamination.

5.1 Outdoor Shielding

In 1968 the U.S. Office of Civil Defence derived shielding factors as shown in Table 5.1 for surfaces of various degrees of roughness.

Table 5.1. Ground roughness reduction factors for gamma-shine resulting from a typical fallout source. (US Office of Civil Defence 1968)

Ground Roughness Condition	
Smooth plane (hypothetical)	1.00
Paved areas	1.00 to 0.85
Lawns	0.85 to 0.75
Gravelled areas	0.75 to 0.65
Ordinary ploughed field	0.65 to 0.55
Deeply ploughed field	0.55 to 0.47

Jensen (1982) calculated an outdoor shielding factor of 0.6 for the Copenhagen suburban area and 0.06 for the city area.

5.2. Vehicle Shielding

Vehicles can provide some shielding for individuals travelling in a contaminated area. Burson (1974) found dose reduction factors of the order of 0.5 to 0.7 for cars, pickups and buses and 0.3 - 0.5 for heavy trucks and trains. Experimental messurements of vehicle shielding have also been made by Lauridsen and Jensen (1982). Their recommendations for automobile shielding factors are shown in Table 5.2.

Table 5.2. Recommendations for automobile shielding factors (Lauridsen and Jensen 1982)

Shielding Status		Type of Vehicle	
		cars	buses
open areas	passengers	0.40	0.40
	no passengers	0.60	0.35
single houses	passengers	0.30	0.25
	no passengers	0.35	0.30
urban areas	passengers	0.25	0.20
	no passengers	0.30	0.25

5.3. Indoor Shielding

The gamma doserate for a person inside a building will depend on the amount of radioactivity on the outside walls, on roofs and on surounding ground surfaces. The degree of protection afforded by the building will depend on factors such as thickness and composition of walls. Shilding effects were reviewed by Burson and Profio (1975 and 1977). The recommendations of Burson and Profio, which were implemented in Wash 1400 (NRC 1975), are summarized in Table 5.3.

Jensen (1984) calculated shielding factors based on building data from France, United Kingdom and Denmark. For single-storey family houses it was assumed that deposition on the surrounding ground surfaces was ten times higher than those on outside surfaces of houses. For multistorey buildings the activity concentrations were assumed to be equal for all surfaces. The results are shown in Tables 5.4 - 5.9. inc.

Table 5.3. Representative shielding factors for surface deposition from WASH 1400 (NRC 1975)

Structure or Location	Representative Shielding Factor[a]	Representative Range
One- and two-story wood-frame house (no basement)	0.4[b]	0.2 - 0.5
One- and two-story block and brick house (no basement)	0.2[b]	0.04 - 0.40
House basement, one or two walls fully exposed:	0.1[b]	0.03 - 0.15
One story, less than 2 ft of basement walls exposed	0.05[b]	0.03 - 0.07
Two stories, less than 2 ft of basement, walls exposed	0.03[b]	0.02 - 0.05
Three or four stony structures, 5000 to 10000 ft^2 per floor:		
First and second floors	0.05[b]	0.01 - 0.08
Basement	0.01[b]	0.001 - 0.07
Multistory structures, >10.000 ft^2 per floor:		
Upper floors	0.01[b]	0.001 - 0.02
Basement	0.005[b]	0.001 - 0.015

(a) The ratio of the interior dose to the exterior dose

(b) Away from doors and windows

Table 5.4. Calculated shielding factors for French single-family houses. (Jensen 1984).

Building data	Built before 1920	Built 1940 - 1950	Traditional	Prefabricated
Outer wall, kgm^{-2}	748	357	493	357
Inner walls, kgm^{-2}	595	340	340	289
Roof, kgm^{-2}	408	357	357	323
Floor to ceiling height, cm	310	250	250	237
Length, m	13.5	13.5	13.5	13.5
Width, m	8.8	8.8	8.8	8.8
Window percentage, %	16	16	16	16
Ground width, m	40	40	40	40
Shielding factor, S	0.025	0.086	0.052	0.090

Table 5.5. Calculated shielding factors for French modern multistorey buildings. (Jensen 1984).

Building data	Size of appartment		
	74 m^2	88 m^2	101 m^2
Outer wall, kgm^{-2}	418	418	418
Inner walls, kgm^{-2}	272	272	272
Partition walls, kgm^{-2}	374	374	374
Floors, kgm^{-2}	418	418	418
Roof, kgm^{-2}	418	418	418
Floor to ceiling height, cm	250	250	250
Length, m	10.38	12.18	13.98
Width, m	8.58	8.58	8.58
Window percentage, %	16	16	16
Road width, m	40	40	40
Shielding factor, ground floor	0.041	0.035	0.030
middle floor	0.010	0.0075	0.0055
top floor	0.017	0.015	0.014

Table 5.6. Calculated shielding factors for British semi-detached and multistorey buildings. (Jensen 1984).

Building data	Semi-detached	Multistorey
Outer wall, kgm^{-2}	320	690
Inner walls, kgm^{-2}	230 - 396	230 - 396
Floors, kgm^{-2}	22	690
Roof, kgm^{-2}	68	460
Floor to ceiling height, cm	300	320
Length, m	16	20
Width, m	14	20
Window percentage, %	10	13
Road and ground width, m	30 - 100	>300
Shielding factor, ground floor	0.073	0.031
middle floor	-	0.008
top floor	0.055	0.012

Table 5.7. Calculated shielding factors for the most common British houses. (Jensen 1984).

Building data	Building type		
	Lightweight	Modern	Traditional
Outer wall, kgm^{-2}	60	366	513-767
Inner walls, kgm^{-2}	30	70-122	213-254
Floors, kgm^{-2}	59	59	59
Roof, kgm^{-2}	49-112	112	112
Shielding factors:			
One - storey,	0.40	0.17	0.11
Two - storey, ground floor	0.33	0.11-0.17	0.05-0.08
first floor	0.33	0.11-0.14	0.06-0.09
Three - storey, ground floor	0.33	0.05	0.05-0.07
first floor	0.25	0.05	0.04-0.06
sec. floor	0.25	0.14	0.07-0.10
> Four storey, ground floor	-	-	0.04
middle floor	-	-	0.04
top floor	-	-	0.07

Table 5.8. Data and calculated shielding factors for Danish single-family houses. (Jensen 1984).

Building data	Building type		
	Lightweight	Modern	Traditional
Outer wall, kgm^{-2}	150	300	400
Inner walls, kgm^{-2}	100	100	200
Roof, kgm^{-2}	100	100	120
Floor to ceiling height, cm	250	250	250
Length, m	15	15	15
Width, m	8	8	8
Window percentage %	25	25	25
Ground width, m	40	40	40
Shielding factor, S	0.28	0.17	0.10

Table 5.9. Data and calculated shielding factors for Danish multistorey buildings. (Jensen 1984).

Building data	Building type		
	Lightweight	Modern	Traditional
Outer wall, kgm^{-2}	250	400	600
Inner walls, kgm^{-2}	100	150	200
Partition walls, kgm^{-2}	250	250	250
Floors, kgm^{-2}	150	400	200
Roof, kgm^{-2}	100	350	250
Floor to ceiling height, cm	280	280	280
Length, m	9.0	9.0	9.0
Width, m	7.8	7.8	7.8
Window percentage, %	25	25	25
Road width, m	20	20	20
Shielding factor, ground floor	0.10	0.06	0.03
middle floor	0.05	0.02	0.01
top floor	0.10	0.03	0.04

Some calculations made by Catsaros and Vassiliou (1987) with the same assumptions as Jensen are shown in Table 5.10.

Table 5.10. Calculated shielding factors for Greek houses and buildings. (Catsaros and Vassiliou 1987).

	Number of storeys					
Story Number	Single	1	2	3	4	5
0	0.139	0.097	0.096	0.096	0.096	0.096
1		0.052	0.039	0.038	0.038	0.038
2			0.052	0.039	0.038	0.038
3				0.052	0.039	0.038
4					0.052	0.039
5						0.052

Le Grand et al. (1987) calculated shielding factors for selected French houses and the results are shown in Table 5.11.

Table 5.11. Shielding factors for common houses and buildings in France. (Le Grand et al. 1987).

Sample	LOIRE		LOIRET		NORD	
	0.5 MeV	5 MeV	0.5 MeV	5 MeV	0.5 MeV	5 MeV
Old single family	0.07	0.21	0.07	0.21	0.08	0.23
Recent single family	0.12	0.38	0.13	0.43	0.08	0.26
Old multifamily	0.04	0.14	0.06	0.15	0.06	0.17
Recent multifamily	0.08	0.17	0.08	0.18	0.09	0.17
Mean	0.08	0.25	0.10	0.32	0.08	0.24
Best shielded apartment	0.03	0.05	0.05	0.07	0.05	0.08
Worst shielded apartment	0.17	0.46	0.17	0.51	0.14	0.37

Jacob and Meckback (1987) took a novel approach to shielding calculations in an urban area. They used the dose rate 1 m above a lawn surface (dry deposited) for two reasons

- doses and dose-rates over grass are used in many dose evaluation codes for the purpose of expressing exposures at unshielded locations,

- the time dependence of the relative source strength of the various deposition areas is different. In many locations lawns contribute a large fraction to the total kerma and only the use of lawns as a reference will lead to relatively time-independent dose reduction factors.

They made assumptions for relative amounts of deposition as given in Table 5.12. Many of these values have since been confirmed by measurement on deposition of Chernobyl fallout. (Roed 1987, Karlberg 1987, Jacob et al. 1987).

Table 5.12. Estimated assumed strengths one week after deposition relative to fresh dry deposition on lawns. (Jacob and Meckback 1987).

Deposition area	Deposition mode	
	Dry	Heavy rain
walls	0.07	0.015
windows	0.0	0.0
roofs	0.15	0.3
light-shafts	0.07	0.015
streets	0.03	0.1
lawns	0.5	0.6
trees	2.0	0.02

The individual shielding factors are shown in Table 5.13. and 5.14.

Table 5.13. Shielding factors for external exposure after a dry deposition of ^{137}Cs From (Jacob and Meckback 1987).

Location	Important sources	Shielding factors present study
Outside in town with vegetation	Trees, Lawns	0.5 - 2.0
Outside in town, without vegetation	Paved areas, walls	0.1
Living rooms in prefabricated house	Trees, Lawns	0.5 - 1.0
Living rooms in semi-detached house	Trees, Lawns	0.1 - 0.2
Living rooms in house-row	Trees, Lawns	0.05 - 0.15
Attic in prefabricated house	Trees, Lawns	0.5 - 0.7
Attic in semidet house and house row	Roof, Trees	0.15 - 0.2
Living rooms in house block, with park	Trees, Lawns, Roof	0.01 - 0.07
Living rooms in house block	Trees, Roof, Walls	0.002-0.015
Basements with windows above ground	Trees, Lawns	0.02 - 0.05
Basements without windows above ground	Light-shaft, Trees	$2 \cdot 10^{-4}$-$7 \cdot 10^{-4}$

Table 5.14. Shielding factors for external exposure after a wet deposition of ^{137}Cs From (Jacob and Meckback 1987).

Location	Important sources	Shielding factors
Outside on streets	Paved Areas, Lawns	0.15 - 0.5
Living rooms and attic in pref. house	Roofs, Lawns	0.3 - 0.50
Living rooms in semi-detached house	Lawns, Roof	0.07 - 0.15
Living rooms in house row	Lawns, Roof	0.02 - 0.05
Attic in semidet, house and house row	Roof, Lawns	0.15 - 0.3
Living rooms in house block, with park	Lawns, Roofs, Pav.A	0.005-0.015
Living rooms in house block	Pav. Areas, Roof	$7 \cdot 10^{-4}$-$5 \cdot 010^{-3}$
Basements with windows above ground	Lawns	0.01 - 0.015
Basements without windows above gr.:		
- semidetached house	Roof	$5 \cdot 10^{-4}$-$1 \cdot 10^{-3}$
- house row and house block	Light-shaft, Lawns	$5 \cdot 10^{-5}$-$1 \cdot 10^{-4}$

6. INFILTRATION OF RADIOACTIVE MATERIAL INTO BUILDINGS

People inside buildings will normally obtain some protection against airborne pollution originating from the outside. This is the result of both physical and chemical processes. Firstly, some of the pollution will be removed from air entering the building by filtration in cracks, crevices and pores through which air diffuses and secondly, some of the pollution that enters the interior of the building may be deposited on floors, walls, ceilings and furniture, so reducing the levels of airborne pollution. In the case of short-lived radionuclides, the time delay before diffusion into the building would also be ameliorating factor, Roed (1986) investigated 17 Danish dwelling houses using ^{7}Be and SF_6 as tracers. He used a multicompartment model in order to separate the dose reduction in the living room from that in other rooms. For a simple single compartment model the following equation was derived.

$$\frac{dC_i}{dt} = \lambda_r \, fC_o - (\lambda_r + \lambda_d) \cdot C_i$$

where C_i and C_o, are the indoor and outdoor air concentration (Bq per unit volume), λ_r is the rate coefficient of ventilation (unit time)$^{-1}$, and f the filter factor (i.e. the fraction of aerosol not retained in cracks and pores on the way in) λ_d is the rate coefficient of deposition (unit time)$^{-1}$.

In order to evaluate and compare the integrated indoor and outdoor radionuclde concentrations C_i must first be solved in terms of C_o. If we assume an outdoor concentration that is constant over a time interval from t=o to t=t_o and that the air concentration outside this time interval is equal to O, equation 1 can then be solved as

$$C_i = \lambda_r \, f \, C_o \, \frac{1-e^{-(\lambda_r + \lambda_d)}}{\lambda_r + \lambda_d}$$

for $0<t<$ to and

$$C_i = \lambda_r \, f \, C_o \, \frac{e^{-(\lambda_r + \lambda_d)(t-t_o)} - e^{-(\lambda_r + \lambda_d)t}}{\lambda_r + \lambda_d}$$

for $t>t_o$

The integrated indoor air concentration is then obtained by integrating these equations over time; the result is

$$\int_0^\infty C_i \cdot dt = \frac{\lambda_r \cdot f \cdot C_o \cdot t_o}{\lambda_r + \lambda_d}$$

the transfer factor from outside to inside is then found as

$$D_{oi} = \frac{\int_0^\infty C_i \cdot dt}{C_o \cdot t_o} = \frac{\lambda_r \cdot f}{\lambda_r + \lambda_d}$$

The equilibium ratio of the indoor and outdoor air concentrations is the same as the ratio of integrated concentration for a finite plume. Thus, data for chronic releases of pollutants can be used as a basis for predicting the ratio of integrated indoor and outdoor concentration for acute releases.

The total 'time-integrated' concentration resulting from the passage of a cloud of non-depositing pollution will be the same, indoors and outdoors, provided that the air exchange rate remains constant regardless of its magnitude. Even though the concentration builds up indoors more slowly than outdoors, it will, on the other hand, decrease more slowly too. As a result total indoor air pollution will simply persist longer than outdoors pollution.

This means, however, that the integrated concentration indoors can be reduced by increasing the air exchange rate (e.g., by opening the windows) after the pollution cloud has passed.

Most studies of indoor-outdoor air pollution relationships have been carried out in buildings with normal heating and ventilation.

The reactor accident at Windscale in October 1957 offered a unique opportunity for studying the protection afforded by buildings as it involved the release of a radioactive aerosol with the characteristics the same as that expected in any reactor accident, and which was sufficiently radioactive to make it possible to make direct measurements. (Megaw 1961). One week after the accident the Windscale deposition of ^{131}I was measured indoors and outdoors in an office building about about 6 km from Windscale, and in a two-storey stone-built house about 9 km from Windscale. Both buildings lay in the path of the smoke plume and neither was in use at the time of the accident. The house had sash windows and six chimneys. Some of the windows in the office building were open during the accident. Unfortunately because of the extra workload on the technical staff, measurements did not begin in the buildings until one week after the accident and it was then too late to measure the indoor and outdoor exposure integrals. For this reason the measurements were later supplemented by an experiment at Harwell. Here there was a newly erected wooden hut, with tight-fitting windows and reasonably tight-fitting doors. The experiment involved the release of radioactive iodine at a distance of approximately 20 m from the hut. Measurements were then made of the deposition and of the indoor and outdoor exposure integrals. In addition, the air exchange in the hut was measured before each series of experiments. Finally, the exposure integrals for an inactive aerosol composed of Aitken nuclei were also measured.

From the results, it was concluded that the iodine inhaled by people inside the building may be 20-80 per cent of that outside, depending on wind velocity and direction. This study was critically reviewed by Gjørup and Roed (1980) who concluded that the house near Windscale would hve reduced the inhalation dose by a factor of about 6 compared with that outside the house. Biersteker et al. (1965) measured SO_2 levels inside and outside 60 houses in Rotterdam. The sampling was conducted in the winter and although people continued to heat and ventilate the houses e.g. the ventilation was not continuous. The results of the study are summarized in the Tables 6.1 and 6.2.

Table 6.1. SO_2 concentration in homes of different ages. (Biersteker et al 1965).

Year of construction	SO_2 indoors (as % of outdoors)
- 1919	30
1920 - 1939	18
1940 - 1959	17
1960+	6

Table 6.2. SO_2 concentrations in homes with different heating systems. (Biersteker et al 1965).

Heating system	SO_2 indoors (as % of outdoors)
central heating	12
oil heaters	17
coal heaters	20
gas heaters	33

As shown in Table 6.1, the SO_2 concentration is lowest in new houses. This is probably due to the improved sealing provided by new houses.

Table 6.2. shows that houses with central heating offer the best protection against indoor SO_2-pollution. This is probably because these houses have a limited internal source of SO_2 in comparison with houses with the other types of heating.

In houses with central heating, SO_2 can be considered as a pollutant of mainly outdoor origin. The measurements in these houses are therefore of relevance to our study, and we can conclude that the Rotterdam homes offered a mean protection factor of at least 8 for reactive air pollution of outside origin, when windows and doors were closed.

In a study performed by Ycom et al (1971), suspended particulate samples were collected for a 12-hour day and night period. Four sampling points were selected for each structure, two outside and two inside. Three basic types of structures were used for the study: public buildings, office buildings, and private homes. Two buildings of each type were sampled simultaneously. Structures in each pair were essentially similar, except for one design feature which might affect the exposure to and the penetration of certain pollutants.

The pair of public buildings used in the study were the Hartford Public Library, built as an air-tight structure over a four-lane highway, the Hartford City Hall, which is located to the north of the library and is separated from it by a busy street. Neither building is air-conditioned, and both are of masonry construction.

The results for suspended matter are shown in the Table 6.3.

Table 6.3 Summary of suspended particulate matter results. (Concentration, $\mu g/m^3$, indoor/outdoor ratio, dimensionless). From (Biersteker et al 1965).

Location	Sampling point	Summer		Fall		Winter	
		Day	Night	Day	Night	Day	Night
Library	Far Outdoor	132	82	150	100	425	189
	Near Outdoor	98	66	115	77	293	130
	Near Indoor	70	45	61	46	74	51
	Far Indoor	66	43	57	44	67	45
	Indoor/Outdoor	0.50	0.52	0.38	0.44	0.16	0.26
City Hall	Far Outdoor	153	78	133	94	327	168
	Near Outdoor	145	76	128	83	285	147
	Near Indoor	78	52	87	50	107	53
	Far Indoor	78	49	82	50	87	51
	Indoor/Outdoor	0.51	0.63	0.62	0.53	0.27	0.30
100 CP	Far Outdoor	104	93	48	38	124	81
	Near Outdoor	118	98	47	40	137	89
	Near Indoor	49	49	34	24	39	41
	Far Indoor	50	46	36	27	38	39
	Indoor/Outdoor	0.48	0.49	0.75	0.71	0.31	0.48
250 CP	Far Outdoor	124	109	66	46	183	97
	Near Outdoor	115	102	58	44	163	98
	Near Indoor	57	67	38	24	57	31
	Far Indoor	56	60	38	23	60	32
	Indoor/Outdoor	0.45	0.55	0.58	0.50	0.33	0.33
Blinn Street	Far Outdoor	79	65	96	74	114	86
	Near Outdoor	87	65	93	70	109	79
	Near Indoor	67	51	52	42	45	32
	Far Indoor	70	56	54	45	49	35
	Indoor/Outdoor	0.87	0.86	0.56	0.61	0.43	0.41
Carroll Road	Far Outdoor	66	56	78	61	103	85
	Near Outdoor	60	49	81	60	116	81
	Near Indoor	73	47	73	37	44	32
	Far Indoor	76	47	76	38	53	33
	Indoor/Outdoor	1.15	0.84	0.97	0.62	0.51	0.39

The values for indoor/outdoor concentrations in the winter season, where the ventilation rate is limited, are of particular interest. They show that the structure afforded a protection factor for inhalation of suspended particulate matter of outdoor origin of at least a factor of 2.5 for the private homes, 2-3 for the offices, 3-5 for the public buildings.

The objective of a study performed by Alzona et al. (1976) in Pittsburgh was to evaluate the protection factor against inhalation of dust of outdoor origin for people indoors. The method consisted of taking filter samples of air and determining the calcium, iron, zinc, lead and bromine in the particles. These elements were known to be associated with particles in specific size ranges. The lead and bromine associated with particles originate chiefly from automobile exhaust gas, and the concentration varies widely during the course of a day. The iron and zinc levels in Pittsburgh are much higher than in other areas, which shows that they are principally of industrial origin. Calcium is an important component of fly ash, and it is also released when using limestone in the production of steel.

The ratio of outdoor to indoor concentrations were measured at equilibrium in ten indoor locations of widely differing character, two of which were cars. The largest value for this ratio was found in a 10 m^2 room without windows in a new university building in Pittsburgh, and it is stated to have been greater than 10 for all types of pollution measured (iron, lead and bromine). For the other nine rooms, the average for bromine was 2.79; for lead 2.38; for iron 4.17; for zinc (only five rooms) 2.44; and for calcium (only four rooms) 10. The lowest value, 1.43, was found in a 30 m^2 large bedroom with eight windows.

One of the rooms was an office (12 m^2) with a large six-paned window which could be opened in sections. This room was investigated in some detail. The outdoor/indoor ratio was measured firstly without any modification (case J), secondly with plastic film over the window (case K), thirdly with plastic film covering all surfaces (case N), and finally with plastic film covering all surfaces except the window (case P). These experiments showed virtually all the pollution in the room entered through the window. In case J the relation between the outdoor and the indoor exposure was 10 for calcium, 5.88 for iron, 1.92 for zinc, 2.04 for lead, and 2.78 for bromine. The experimenters corrected these values for background concentrations indoors on the basis of case N, and thereby found a protection factor of between 3 and 20, but they also point out that the protection factor becomes much greater for calcium, iron and zinc if corrections were made on the basis of case K. Relating to case K, the dose does not significantly alter the factor for lead and bromine, which was 2.94 and 7.69, respectively.

Additional measurements have been reported by Cohen and Cohen (1980 and 1979). The purpose of the measurement was to extend the data base reported by Alzona el al. The results are shown in Table 6.4. Cohen and Cohen recommend a value of 0.45 for submicron particles and values of 0.22 for larger particles.

Table 6.4.

Summary of indoor-outdoor contaminant concentration ratios (from Cohen 1980)

Bldg. type	Age*	Condition**	Windows	Ca	$Fe^{I/O}$***	Pb	Br
University office	8	A	2	0.043(0.015)	0.077(0.021)	0.17(0.06)	0.41(0.18)
Public High school	60(1)	A	1	0.26(0.23)	0.095(0.036)	0.070(0.01)	0.18(0.06)
Public High school	60(1)	A	0	0.17(0.09)	0.13(0.06)	0.15(0.09)	0.25(--)
University office	25	C	4	0.15(0.08)	0.050(0.021)	0.19(0.07)	0.43(0.20)
					0.12		
Store	80(3)	B	20	0.064(0.016)	0.19(0.06)	0.31(0.10)	
Public elem. sch.	50	C	5	0.23(0.16)	0.44(0.18)	0.62(0.16)	0.57(0.28)
Commercial office	80(2)	C	4	0.14(0.09)	0.16(0.05)	0.27(0.05)	
Home-basement	50	B	5	0.38(030)	0.30(0.13)	0.44(0.10)	0.44(0.16)
					0.54	1.05	1.8
Same, 3 wks later	50	B	5		0.32(0.10)	0.47(0.07	
Home, attic	50	B	2	0.16(0.10)	0.31(0.17)	0.50(0.37)	
Home, attic	70	B	1	0.42(0.29)	0.15(0.03)	0.42(0.21)	
					0.38	0.71	
Home,kitchen	90	D	2	0.15(0.03)	0.26(0.04)	0.85(0.28)	
University office	10	B	0		0.10	0.10	0.10
University lab	60	D	2		0.33	0.25	0.43
Home, bedroom	50	B	8		0.27	0.70	0.58
Home, attic	50	B	2		0.10	0.40	0.29
Home, bedroom	70	D	2	0.05	0.33	0.47	0.22
University, lab	60	D	3	0.08	0.13	0.31	0.25
University office	25	C	3	0.15	0.54	0.47	0.58
University, lab	60	C	3	0.10	0.17	0.49	0.33
Average				0.17(0.08)	0.22(0.11)	0.38(0.16)	0.53(0.17)
Homes average				0.23(0.13)	0.25(0.07)	0.53(0.12)	0.38(0.13)

* Parentheses indicate years since last remodeling.
** Condition: A - excellent, B - good, C - fair, D - poor.
*** Parentheses indicate average deviation. Square brackets are values if no data were rejected; conditions for rejecting data are discussed in text.

Roed (1986) measured the indoor/outdoor concentration in all rooms in 17 selected dwellings using ^{7}Be-tagged particles. The air exchange rate in the building was also measured and related to the indoor/outdoor concentration by the following equation (for living rooms):

$$\frac{C_i}{C_0} = 0.24 \cdot \lambda_r + 0.21$$

The mean value of

$$\frac{C_i}{C_0} = 0.32 \pm 0.17$$

Similar experiments on a smaller scale were conducted in Finland and Norway (Chistensen and Mustonen 1987). The results are shown in Tables 6.5. and 6.6.

Table 6.5. Measured concentrations of ^{7}Be in air inside and outside four different Finnish flats. (Christensen and Mustonen 1987).

Collection period	No.of rooms	Vol.of flat(m^3)	^{7}Be conc.(mBq/m^3) inside(I)	outside(O)	I/O ratio
12-16.7.82	4	188	1.418	4.319	0.33 ± 0.05
19-23.7.82	3	150	1.209	2.672	0.45 ± 0.07
26-30.7.82	3	146	0.651	2.809	0.23 ± 0.05
06-10.9.82	3	140	0.571	1.553	0.37 ± 0.09

Table 6.6. Measured concentrations of ^{7}Be in air inside and outside a Norwegian wooden villa. (Christensen and Mustonen 1987).

Collection period	^{7}Be activity (10^{3} counts/m^{3}) inside(I)	outside(O)	I/O ratio
First series			
04-07.10.1982	0.98	1.56	0.63 ± 0.07
12-15.10.1982	0.47	0.68	0.69 ± 0.10
09-12.11.1982	0.25	0.29	0.86 ± 0.23
29.11-02.12.1982	0.23	0.47	0.49 ± 0.09
Second series			
04-08.06.1984	1.49	3.77	0.40 ± 0.03
08-12.06.1984	0.85	1.87	0.46 ± 0.04
12-19.06.1984	0.43	0.92	0.47 ± 0.06
19-21.06.1984	0.63	1.45	0.44 ± 0.08

Measurements made by Risø in Denmark during the Chernobyl accident yielded values for C_i/C_o for caesium and particulate iodine as shown in Table 6.7.

Table 6.7. Indoor/outdoor concentration in single family houses. (Roed 1987c).

Isotope	$D = \frac{C_i}{C_o}$
^{131}I (particulate)	0.39
^{137}Cs	0.27
^{7}Be	0.49

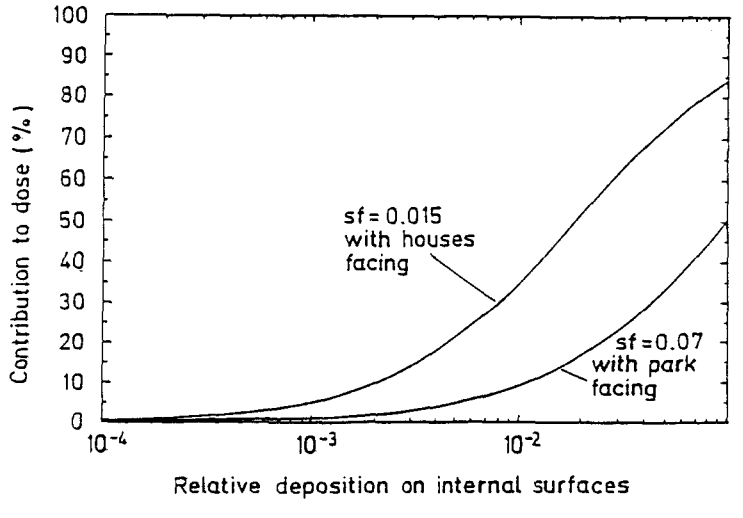
Air-exchange rate $\lambda_E = 0.4\ h^{-1}$.

7. INDOOR DEPOSITION

Indoor deposits of radionuclides could effectively offset the benefits of shielding provided by buildings, particularly in the case of large buildings (e.g. office building, hospitals and schools) where the volume/surface ratio is high. This was stressed by the calculations made by Jacob and Meckbach (1987). Figure 7.1. shows results for the ground floor of a multistorey housing block in different urban surroundings. If the housing block faces a similar building the shielding factor is 0.015 after dry deposition if internal surfaces are not taken into account. The indoor exposure is enhanced by more than a factor of 2 if the internal deposition relative to deposition on a lawn is greater than 2%. In the case of a park facing the house the shielding factors when not considering deposition on internal surface is 0.07 and the relative internal deposition has to be greater than 10% to change the indoor exposure by more than a factor of 2.

Figure 7.1.

Contributions of depositions on internal surfaces to the exposure in the ground floor of the multistorey house block after a dry deposition of ^{137}Cs. If the internal contaminations are neglected, the shielding factor is 0.015 for the case of houses facing the house block and 0.07 for a park facing it.



Indoor deposition was measured by Roed and Cannell (1987) in a relatively modern Danish, single storey, 4-bedroomed house.

Outdoor air was ducted directly into the house using a centrifugal blower. An overpressure was maintained during the measurements in order to ensure that air did not leak into the house by other routes. In this case the following equation is appropriate

$$\lambda_r C_o - \lambda_r C_1 = \lambda_d C_i$$

where the first term on the left-hand side determines the activity in the air entering the building. (λ_r is the rate coefficient of ventilation and C_o the concentration of activity in the inlet air). The second term denotes the radioactivity leaving the building (C_i is the concentration in indoor air). Thus the difference term represents the activity deposited on internal surfaces (λ_d is the rate coefficient of deposition).

λ_d can then be found by measuring λ_r and C_i/C_o.

The average local deposition velocity U_d may now be determinal from the relationsship

$$U_d = \frac{\lambda_d V}{A}$$

where V is the total volume of the house and A its total internal surface.

The results are presented in Table 7.2.

Table 7.2.

Isotope	v_d, mean deposition velocity: 10^{-4} ms^{-1}	Rate coefficient of deposition λ_d : h^{-1}
^{137}Cs	0.6	0.39
^{134}Cs	0.6	0.38
^{131}J (particulate)	1.1	0.65
^{7}Be	0.7	0.44
^{103}Ru	2.0	1.26
^{106}Ru	1.7	1.02
^{141}Ce	3.1	1.89
^{144}Ce	3.9	2.44
^{95}Zr	5.8	3.56

The experiments were repeated after the Chernobyl accident in order to investigate the difference in indoor deposition between furnished and unfurnished rooms (Roed and Cannell 1988), and the results are presented in Table 7.3.

Table 7.3. Deposition parameters in houses (Roed and Cannell 1988).

	Isotope	Deposition velocity - 10^{-4}m/s	Rate coefficient of deposition h^{-1}
partly furnished house	^{7}Be	0.18±0.05	0.11±0.03
	^{137}Cs	1.5±0.3	0.91±0.16
unfurnished room	^{7}Be	0.10±0.06	0.06±0.03
furnished room	^{7}Be	0.33±0.08	0.21±0.05

Here the deposition velocity refers to the indoor air concentration, and in Table 7.4. to the outdoor. The differences in caesium deposition in the first and the second experiments may be due to the different particle sizes in the direct Chernobyl cloud used in the first experiment compared with those in the second where some of the caesium in the air was from resuspended material.

The local deposition velocities were also measured on very smooth surfaces (Table 7.5.), but they are much lower than the mean velocity, which includes very large surface areas, such as carpets.

Table 7.4. Deposition velocities in houses (ref. outdoor air). From (Roed and Cannell 1988).

	Isotope	Deposition velocity · 10^{-4}m/s (reference outdoor air)
partly furnished house	^{7}Be	0.1
	^{137}Cs	0.5
unfurnished room	^{7}Be	0.1
furnished room	^{7}Be	0.2

Table 7.5. Deposition velocity on internal surfaces · 10^{-4} m s^{-1}. (Roed and Cannell 1988).

		Particles		Particles and gas
	Sample No.	^{134}Cs	^{103}Ru	^{131}I
Wallpaper	148	0.01	0.01	0.09
(vertical)	150	0.02	0.01	0.08
Vinyl floor	149	0.09	0,10	0.23
(horizontal)	154	0.13	0.10	0.20
Wooden floor	207	0.08	0.08	
(horizontal)	208	0.05	0.07	

8. RECLAMATION

Reclamation is a general term used to describe the reduction of radiation levels in a contamianted environment. Reclamation may be achieved by decontamination, in which case the radioactive material is physically removed from the area, or by on-the-spot dose-reducing measures such as inverting flagstones, ploughing land or applying a new layer of asphalt to a contaminated road surface.

The dose reduction factor (DRF) in defined as the ratio of the dose 1 m above the surface before the reclamation procedure, to that after.

The decontamination factor (DF) is the ratio of the contamination before and after decontamination. Much of the information available on decontamination has been obtained from nuclear weapons testing and aplies to particles much larger than those expected to be released in a nuclear accident (Linsley 1984). Fallout particles from nuclear explosions are normally >10μm in the vicinity of the detonation whereas those released to the atmosphere in a nuclear accident would be<3·5μm (Bunz 1980). The decontamination efficiency for impervious surfaces can be strongly influenced by particle size whereas for many reclamation procedures not involving displacement of the contaminant, particle size is of no consequence.

8.1. Decontamination

Sweeping an vacuum-sweeping

The efficiency of these methods are strongly dependent on the amount and nature of surface dust present at the time of deposition.

Paved surfaces

DF: 2-10 cost: $ 0.004/m^2.

Sartor et al. (1974) found that the sweeper efficiency was 15% for particles less than 43 μm and the overall efficiency for all particle sizes was 50%. The effort required was 2.56 min/1000 m^2. The overall efficiency of 50% could be raised to 70% by doubling the effort and to 95% by increasing it sixfold. They derived a methematical relationshsip for calculating the effectiveness of dust and dirt removed within each particle size range.

$$M = M^{O} + (M_{O}-M^{O})e^{-kE}$$

where M is the amount of street surface contaminant remaining after sweeping, M_o the initial amount of contaminants, E the amount of sweeping effort involved in using the equipment (min/1000 m^2) and m^o and k are dimensionless constants dependent on sweeper characteristics, particle size of contaminant and street surface.

The findings of Sartor et al. agree well with those of Clark and Cobbins (1963). Their results as reproduced in WASH 1400 (NRC 1975) indicate that the efficiency of the method is sensitive to particle size and initial mass loading in such a way that the method would be inefficient for particles smaller than 20 µm and initial mass loadings below 1.0 g/feet^2.

Calvert et al. (1984) dealt with the efficiency of removing particles of less than 5 µm by use of an improved vacuum sweeper. The overall efficiency for the main pick-up head was about 90% for particles smaller than 2 µm.

The general conclusion is that sweeping and vacuum sweeping on artificial surfaces would have only a marginal effect on small particles at low mass loadings unless an improved vacuum sweeper as described in Calvert et al. (1984) were used. In that case a decontamination factor of 2-10 can be attained. The cost of modifying a regenerative air vacuum sweeper to do this will be only about 5% of the sweeper cost.

Pervious surface

DF: 3-10 cost $ 0.004-0.008/m^2

A small vacuum street sweeper was used for removing contamination from a clipped meadow (Menzel 1962). About half the contamination could be removed by sweeping the meadow twice.

A rotation-broom sweeper with steel bristles removed about 70 percent of the contamination from moist soil with a thin cover

of fescue. A second sweeping gave almost 90 percent removal.

Firehosing and flushing

Paved surfaces

DF: 2-10 cost $ 0.1/m^2

Wiltshire et al. (1965 and 1966) performed experiments with a standard firehose. For one pass they found decontamination factors of 10 for smoothly textured and 2 for roughly textured surfaces for loadings of 43 and 270 g/m^2 when contaminated with particles of diameter 44 µm. Because of the particle size used in this study, the DF values obtained should be viewed as upper limits when considering decontamination of surfaces contaminated with small particles. Dick and Baker (1961) conducted experiments where the contamination consisted of plutonium oxide particles with an average diameter of 0.8 µm. The surfaces were, amongst others, asphalt and concrete pads measuring 3.5 by 3.5 m. Decontamination factors of 10-12 for asphalt and of 4-40 for concrete surfaces were obtained after hosing with water at a pressure of 400-700 psi 2 days after contamination.

Warming (1982 and 1984a) sprayed Rb-86, Ru-103, and Ba-La-140 dissolved in water onto dry asphalt and concrete surfaces. A single firehosing two days later gave decontamination factors of about 2. Hosing after 40-50 days gave nearly no decontamination.

Clark and Cobbin (1964) performed decontamination experiments using mechanized street flushers. The contaminants were 44-100 µm particles. They found decontamination factors of 10 for mass loadings of 54 g/m^2 and of 50 for a mass loading of 130 g/m^2 for roughly textured and of about 50 for smoothly textured asphalt for mass loadings of 54 g/m^2 and 130 g/m^2. These decontamination factors must be considered as upper

limits because of the large size of the contaminated particles.

Run-off studies can give some valuable information about the efficiency of firehosing and flushing of streets. From such experiments, Sartor et al. (1974) found an equation for the rate at which rainfall washes away loose particulate matter from street surfaces. This rate is dependent on rainfall intensity, street surface characteristics and particle size.

The equation given is

$$N_C = N_O(i-e^{-krt})$$

where N_O = initial weight of material of a given particle size, t = time of rainfall, r = rinfall intensity, N_C = weight of material of a given particle size removed after time t, and k is a constant. The constant k depends on street characteristics, but was found to be almost independent of particle size (at least within the size range 10-1000 μ).

Roofs

DF: 2-10 cost 1 l/m^2

By firehosing, Owen et al. (1960) found decontamination factors of more than 10 on flat tar and gravel roofs, and Miller (1960) obtained a decontamination factor of 3 on a concrete roof and a single roofs decontaminated 2 days following the deposition.

However, the contamination in the two experiments consisted of large particles and the decontamination factor obtained must be considered as an upper limit for small particle contamination.

Gjørup et al. (1985) decontaminated aged ^{137}Cs nuclear fallout contamination on roof material by flushing and scrubbing with water. The decontamination had no effect on red tile. On

corrugated eternit the decontamination factor was around 2. The problem with hosing with water is that the displaced contamination is merely translocated to some other surface where it may adhere even more strongly.

Vacuuming

DF: 2-5 cost $ 0.1/m^2

Dick and Baker's (1961) decontamination experiments on hard surfaces also involved vacuum cleaning. Industrial vacuum cleaners were used for decontaminating surfaces embedded with plutonium particles of average diameter 0.8 µm, 99% of them below 2.5 µm. The decontamination procedure was carried out two days after deposition; Dick and Baker found an average decontamination factor of 3.

Similar results may be obtained using domestic machines which are as effective today as the industrial ones used by Dick and Baker.

Removal of surfaces

Removal of surfaces is a much more expensive procedure than sweeping, vacuum cleaning or hosing.

Road planer (paved surfaces)

DF: >100 Cost $ 3/m^2

The cost of the cutting operation depends on the depth of cut and type of machine. A cold planer of 190 cm in diameter grinding off an asphalt surfce 3 cm thick can cover about 700 m surface per hour at a cost of $ 3/m^2 including rubble removal. Barbie et al. (1980) found that by grinding off 1.2 cm, contamination of concrete surfaces can be carried ut for about $ 0.3/m^2 with a speed of abut 2000 m_2/h. The cost of rubble removal must be added to this figure. Decontamination factors will be greater than 100.

Earth-moving equipment (soil surface)

DF: 4-10 Cost $ 0.2/m^2

Many common types of earth-moving equipment have been used in decontamination tests. These include graders, bulldozers, and pan-type scrapers. (Menzel et al. 1960, 1961, Menzel 1962, and Owen 1965). Between 80 and 90 percent of the radioactive surface contaminant was usually taken away when 5 cm of the soil was removed.

Cutting back trees and busches and defoliation

DFR: 10 Cost $ 7/tree

The cost of cutting back trees and bushes can vary widely depending on the size and position of the tree.

Defoliation is possible, but decontamination method has the disadvantage that the tree may then die.

Decontamination of indoor surfaces

DFR: 2-10 Cost $ 1/m^2

Vacuum cleaning and washing are normal procedures for most indoor surfaces. These could be supplemented by more extreme procedures such as removing paint and wall paper.

8.2. Other reclamation procedures

Besides decontamination, a number of dose-reduction measures could be applied to a contaminated area: ploughing, digging gardens, and turning flagstones. These procedures do not remove the contamination but relocate it in such a way that the dose is reduced.

Digging gardens and turning flagstone

DFR: 6 Cost $ 1/m^2

Gjørup et al. (1982) showed that when radioactive material was buried at a depth of one spit, and the flagstones were turned over, the doserate from the deposited activity would be reduced by af factor of 6. This could be achieved in an average garden by an effort of 1 person-day per 50 m^2 surface.

Ploughing

DFR: 15-50 Cost $ 0.1/m^2

In parks and similar open spaces in an urban area, ploughing could be an effective means of reducing doserate. A reduction of a factor of 15-18 can be achieved by normal ploughing. Deep ploughing in which the uppermost layer of the soil is turned into the bottom of the furrow, can reduce the dose rate from the activity deposited by a factor of more than 50 (Hedemann 1979, Roed 1982). Further, ploughing would negate this gain, unless the initial ploughing buried the surface contamination beyond the reach of subsequent ploughing.

9. RECLAMATION AND DECONTAMINATION STRATEGY

9.1. Introduction

The ultimate goal of a reclamation/decontamination study is the provision of a nuclear contingency plan for reclamation and decontamination; in this case of the urban environment.

In developing such a strategy, a host of factors need to be considered in order to provide the most cost-effective strategy for any given scenario.
Some of the important factors to be considered in formulating strategic countermeasures are:

1) Distribution of the deposited material with respect to the different outdoor surfaces.

2) The contribution of the different surfaces to dose rate.

3) The decontamination or dose reduction achievable on the individual surfaces using appropriate methods.

4) The practicability of the various reclamation/decontamination procedures.

9.2. An outline strategy for dose reduction by decontamination

The central part of a town normally consists of tall buildings, extensive paved areas and a limited amount of green areas. In contrast, residential suburbs have smaller buildings, gardens with trees and bushes and a limited amount of paved areas.

Within an urban area, various components (e.g. walls, paved areas, roofs, grassed areas, etc.) can be recognized and the individual contributions to dose of each of these components will depend on their surface area, the amount of radioactive material retained, the energy of the radiations and the degree

of shielding.

As part of the procedure for reducing radiation dose to the populace of a given urban area, it is necessary to first determine the physical characteristics of the area in some detail; that is to determine the size of the buildings, the thickness of the walls, the type of roofs, the extent of grassed and paved areas, the amount of trees, etc.

From a knowledge of the prevailing weather during deposition, and of the content of the radioactive plume, the relative distribution of deposited radioactive matter on the different surfaces can be estimated. An effective source strength can then be defined. Wet deposition and dry deposition will, for example, give different effective source strengths for the various surfaces in an urban area.

Having defined the source strength, the next step is to calculate the relative dose rate at different locations (indoors and outdoors) due to deposition on the different urban surfaces (roofs, walls, paved areas, trees, bushes, etc.).The mean relative dose rate to a member of the local populace can then be found taking into consideration the time that he will spend in the different locations.

The next step is to estimate the decontamination factors achievable for the various surfaces and from this to find the relative source strength after decontamination.

We can then recalculate dose rate at the various locations and show the reduction in dose rate achievable through decontamination.
From the costs of the various decontamination methods and the corresponding achievable reduction in dose rate, the cost of a relative reduction (e.g.,1%) in dose rate for the various types of surfaces can be calculated. These data will then indicate the most cost effective means of dose reduction.

9.3. Strategy plan for dose rate reduction in a given urban complex : worked example.

The following is an example of how to develop a strategy plan for dose reduction in four urban complexes. Examples are given for

1) detached houses in a suburban area.
2) two storey semi-detached houses.
3) rows of terrace houses (2 stories).
4) multi-storey blocks of flats.

The cost and efficiency of different decontamination procedures are given in Table 9.1. (See section 8).
The procedures are relevant for most urban environments.

Table 9.1. Cost and effectiveness of countermeasures

Surface	Reclamation procedure for radiocaesium	DF-DRF factor	Cost ($/ECU per m^2)
Windows	Cleaning	10	2
Roads Asphalt,concrete	Sweeping vacuum sweeping	1 - 5	0.04
Grass	Cutting	2 - 10	0.016
Roads Asphalt,concrete	Firehosing and water-jets	1 - 10	0.1
Roofs	-	1 - 5	3
Walls	-		1
Roads (asphalt)	Planing	> 100	3

Walls	Sandblasting	> 100	10
Roofs	-	3 - 100	20
Grass and Soil	Removal of surface	4 - 10	0.2
Trees	Cutting/defoliating	10-100	7
Garden	Digging	6	1
Fields and parks	Ploughing	15 - 50	0.1

It can be seen that the decontamination factors (DF) or the dose reduction factors (DRF) vary considerably from method to method. The wide ranges quoted for DF's for many of the methods reflect the fact that the decontamination achievable is often dependent on the circumstances in which the contamination occurred, i.e. wet or dry deposition, the amount and intensity of rainfall at the time of wet deposition, how much rain has fallen since deposition occurred, etc. We therefore need to be able to estimate what can be achieved in terms of dose reduction for the individual surfaces after taking into account these different factors. We have done this for the cases wet and dry deposition.

Table 9.2. shows estimates of typical source strengths before and after decontamination, the relative source strengths given are relative to deposition on very short grass where it is assumed that all the deposited material is retained on the grass and none on the soil.

Table 9.2. Relative source strengths before and after decontamination.

	Dry deposition before/after decontamination	Wet deposition before/after decontamination
Walls	0.100/0.050	0.010/0.010
Roofs	1.000/0.500	0.400/0.300
Garden	1.000/0.100	0.800/0.100
Street	0.400/0.200	0.500/0.300
Trees	3.000/0.100	0.100/0.010
Indoors	0.020/0.010	-

Given these relative source strengths before and after decontamination Dr. Meckback from G.S.F. in Munich has calculated the relative contribution to dose in four different urban areas of varying population density, as described by Jacob and Meckback. An example of the type of results obtained hereby is given in table 9.3.

In this table is given the contribution to dose rate at different locations (indoors at ground floor, first floor, attic, and outdoors for streets and gardens) from the different sources such as walls, gardens, trees, etc.
In the last row of table 9.3. the location factor is given before decontamination.

The location factor is defined as the dose rate at the specific location relative to that at the reference situation : the dose rate at an infinite grass surface where all the deposited matter is on the grass cover.

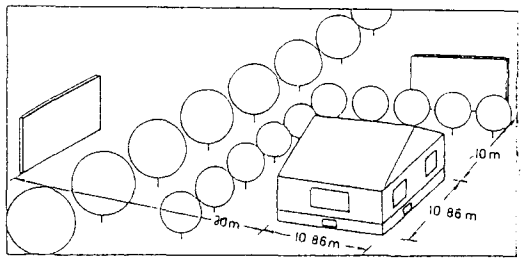
Table 9.3. Percentage contribution to dose rate at different locations.

Row of terrace houses.
Dry deposition, before decontamination
Source energy: 300 keV.

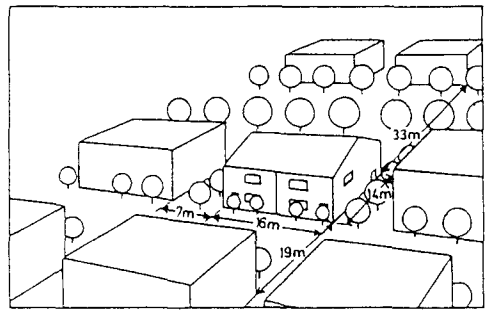
Deposition surface	Relative source strength	Ground floor	First floor	Attic	Street	Garden
Windows	0.050	1.2%	2.4%	0.0%	0.1%	0.1%
Walls	0.100	1.6%	2.0%	0.1%	1.5%	1.7%
Roof	1.000	0.3%	6.8%	74.2%	1.3%	0.4%
Basement windows	0.050	0.0%	0.0%	0.0%	0.0%	0.0%
Light shafts	0.400	0.1%	0.0%	0.0%	0.0%	0.1%
Neighbour building walls	0.100	0.9%	1.3%	0.3%	1.2%	1.2%
Neigbour building roofs	1.000	1.2%	4.0%	4.6%	1.6%	0.9%
Garden	1.000	36.3%	24.1%	3.5%	31.6%	60.1%
Street	0.400	4.0%	2.7%	0.4%	26.8%	0.6%
Ground beyond neighbour building	1.000	5.8%	12.9%	11.0%	9.0%	5.6%
Trees	3.000	48.7%	43.8%	5.9%	26.8%	29.3%
Location factor		0.07	0.04	0.32	0.63	0.94

The four simulated environments are shown in Fig.9.1.

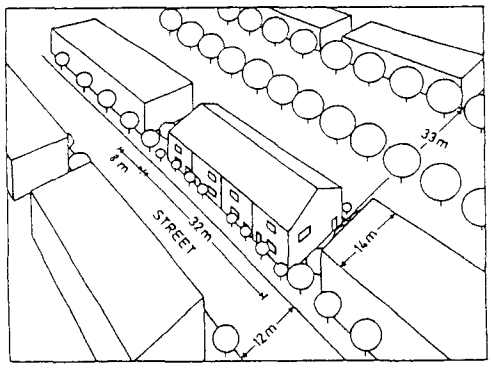
Fig.9.1. The four urban environments as simulated by Jacob and Meckback.



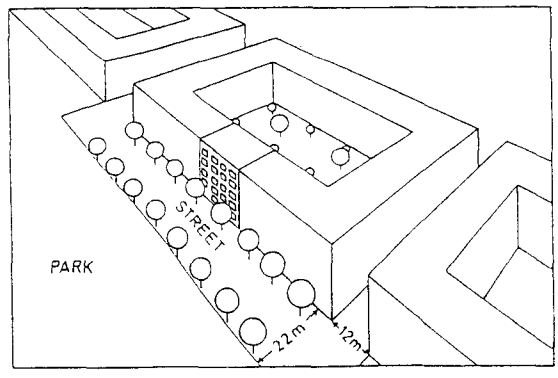
1.1 Detached houses in suburban area.



1.2 Two-storey semi-detached houses.



1.3 Rows of terrace houses (2 stories).



1.4 Multi-storey blocks of flats.

In table 9.4., we show what we have estimated to be achievable decontamination factors together with costs per unit area.

Table 9.4. Achievable Urban Decontamination Factors and Costs.

Surface type:	Roofs	Walls	Streets	Trees	Garden	Internal
Efficiency (DRF):						
DRY:	2	2	2	50	10	2
WET:	1.3	1	1.7	10	8	-
Costs:						
$(ECU)/m^2	3	1	0.04	7	1	1

If we consider the average person living in one of the four urban areas to spend 85 % of his time indoors distributed equally between the different floors, 10 % of his time in the garden, and 5 % on the street, the location averaged dose rates from the different contaminated surfaces can be calculated. This has been done in table 9.5., which shows the contributions to dose in the 4 environments, from a wet and a dry deposition of 1 MBq/m^2 on grass, using the relative source strengths given in table 9.2.

If, for instance, walls and roofs are firehosed, streets are vacuum-swept, trees are cut, gardens are dug, and internal surfaces cleaned by normal domestic cleaning methods, the achievable reclamation efficiencies and costs could be as shown in table 9.4.

From the data in table 9.4., the costs and efficiency of the chosen decontamination/reclamation procedures have been calculated (table 9.5.).

Table 9.5. Dose rates and countermeasures example.
Location averaged dose rates from a deposition corresponding to 1 MBq/m^2 dry deposition on grass. Source energy 662 keV.

1) Single storey detached houses.

Dose rate contribution from different surfaces [uGy/d]:

	Roofs	Walls	Streets	Trees	Garden	Internal
DRY:	6.13	4.95	-	17.91	26.47	0.96
WET:	2.45	0.12	-	0.60	21.18	0

% dose reduction by decontamination/reclamation of the surfaces:

DRY:	5.43%	4.40%	-	30.83%	42.33%	0.85%
WET:	2.52%	0	-	2.21%	76.11%	0

Costs per person per % dose reduction [$/ECU]:

DRY:	16.30	7.43	-	12.05	3.73	76.05
WET:	35.17	-	-	168.13	2.07	-

2) Two storey semi-detached houses (2 stories).

Dose rate contribution from different surfaces [uGy/d]:

	Roofs	Walls	Streets	Trees	Garden	Internal
DRY:	3.32	0.37	-	4.65	9.45	0.96
WET:	1.33	0.04	-	0.16	7.56	0

% dose reduction by decontamination/reclamation of the surfaces:

DRY:	8.94%	0.96%	-	24.20%	45.50%	2.59%
WET:	3.66%	0	-	1.54%	72.85%	0

Costs per person per % dose reduction [$/ECU]:

DRY:	10.74	45.38	-	8.14	2.09	34.67
WET:	26.25	-	-	128.69	1.30	-

3) Rows of terrace-houses (2 stories).

Dose rate contribution from different surfaces [uGy/d]:

	Roofs	Walls	Streets	Trees	Garden	Internal
DRY:	1.21	0.29	1.31	2.94	5.66	0.96
WET:	0.35	0.03	1.25	0.13	4.53	0

% dose reduction by decontamination/reclamation of the surfaces:

DRY:	5.05%	1.19%	5.47%	23.93%	42.76%	4.06%
WET:	1.40%	0	7.96%	1.90%	62.94%	0

Costs per person per % dose reduction [$/ECU]:

DRY:	18.82	32.79	0.24	4.24	2.35	27.85
WET:	68.48	-	0.15	52.00	1.56	-

4) Multi-storey blocks of flats (5 stories).

Dose rate contribution from different surfaces [uGy/d]:

	Roofs	Walls	Streets	Trees	Garden	Internal
DRY:	0.05	0.27	1.77	1.95	4.00	0.96
WET:	0.02	0.03	2.22	0.07	3.21	0

% dose reduction by decontamination/reclamation of the surfaces:

DRY:	0.23%	1.60%	10.32%	22.31%	43.06%	5.69%
WET:	0.09%	0	16.02%	1.06%	50.71%	0

Costs per person per % dose reduction [$/ECU]:

DRY:	172.31	48.44	0.07	0.27	0.47	20.79
WET:	552.06	-	0.05	5.60	0.39	-

The data in table 9.5. shows the percentage contribution to dose rate from the different surfaces where deposition has occurred in different types of urban areas.
For dry deposition before decontamination/reclamation it can be seen that generally, the garden and the trees are the main contributors to the dose. Roofs are also important, especially in environments dominated by smaller houses, and the streets become relatively more important in the city areas.
It can be seen that employment of the proposed countermeasures for gardens and trees, can reduce the total dose rate to about 25% . Decontamination of roofs, streets, walls, and internal surfaces is much less effective.
Defoliation and cutting-back of trees and digging gardens are inexpensive and practicable means of reducing dose rate and would be especially cost-effective in terms of dose reduction per unit cost in suburban areas. These methods would rate highly in a list of priorities. Second priority should be given to street cleaning, also a relatively inexpensive procedure. In spite of roofs being important surfaces they can only be given third priority, as decontamination of a roof is an expensive procedure. Walls and internal surfaces are unimportant surfaces, and they would be given the lowest priority.

For the case of wet deposition, the contribution to dose from ground deposition (gardens and streets) are dominant. In fact, in the sub-urban areas, the reclamation of garden areas alone yields a dose reduction factor (DRF) of about 4.
The dose reduction achieved by decontamination/reclamation of other surfaces will be small. From the view point of cost effectiveness we conclude that decontamination/reclamation of gardens should be given first priority as the countermeasure procedure is efficient, practicable and inexpensive. Street cleaning would have second priority since it is very cheap to perform. Treatment of other surfaces is not cost effective and unless it is simply a case of removing loose surface debris, they should not be considered.

9.4. Discussion and conclusion on strategy for decontamination.

The ultimate goal of a decontamination study is to provide a contingency plan for reclamation of nuclear contaminated urban areas. In developing such a strategy several factors need to be considered in order to provide the most cost effective strategy for any given scenario, some of the important factors are

1. Distribution of the deposited material with respect to the different surfaces.

2. The contribution of the different surfaces to dose rate.

3. The decontamination or dose reduction achievable on the individual surfaces using appropriate methods and the cost of these, and

4. The practicability of the various reclamation, decontamination procedures.

A method for obtaining a strategy is given. This implies the calculation of dose rate at different positions inside and outside houses both before and after the reclamation procedures.

To demonstrate the practicability of the proposed strategy, four examples are given for typical urban/suburban environments with different population concentrations.

It has been shown that in the case of dry deposition, decontamination of trees and gardens is the most cost effective procedure and should be given first priority. Second priority would be given to street-cleaning, and third priority would be given to the decontamination of roofs. The lowest priority should be given to decontamination of walls.

In the case of wet deposition, decontamination of gardens would be given first priority and street cleaning second priority. Treatment of other surfaces would not be worthwhile.

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