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A risk assessment method for accidental releases from nuclear power plants in Europe

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SUMMARY

The 1986 accident at the nuclear power plant in Chernobyl has shown that a severe accident within a nuclear power plant can lead to a large-scale contamination of Europe, where at present over 200 nuclear power reactors are operational. The question raised is to what extent possible accidents with nuclear power reactors pose a risk for the European population. In this report a method is described for evaluating the probability of death due to stochastic effects, combining the probability of accidental releases with the consequences in terms of the excess doses received over a lifetime (70 years). The study is limited to stochastic deaths: victims of death due to very high short term radiation doses possibly occurring in the close proximity of the nuclear power reactor (< 5—10 km) are not included in the risk estimates.

The nuclear power reactors have been categorized according to major (safety) characteristics and operational features. Accident and release probabilities have been estimated for each of these reactor categories. Dispersion and deposition calculations are based on statistical estimates in order to account for different weather conditions and their frequency of occurrence. The various exposure pathways, including ingestion of contaminated food products, inhalation and external exposure, are modelled. Countermeasures to reduce exposure are not considered. The radiation exposure following an accidental release is modelled over a period of 70 years. The results are given as the excess death risk per year forthcoming from the combined operation of all nuclear power reactors involved.

An estimated excess death risk provided for Europe shows a large variation. This risk is less than 10^{-8} per year in Iceland and southwestern parts of Spain and Portugal. It increases from west to east: 2×10^{-8} per year in Ireland, 3×10^{-8}—10×10^{-8} per year in England and large parts of France, Italy and Norway, around 10×10^{-8}—30×10^{-8} per year in the Netherlands, Belgium, Germany and large parts of central Europe. A risk over 100×10^{-8} per year is found in large areas of the former Soviet Union, including the Baltic states, White Russia, Russia and Ukraine. In these countries a risk of 1000×10^{-8} per year is exceeded in the smaller regions around the nuclear reactors. Further towards the eastern border of Europe the risk declines to around 10×10^{-8} per year.

Nuclear power reactors in the eastern European countries dominate the estimated risk pattern and contribute at least 40—50% to the average risk in the western European countries. Improving the reactor safety in eastern European countries leads to considerable reductions in the estimated risk.

Because of large uncertainties the calculated risk should be considered as an indication. A preliminary estimate amounts to an overall uncertainty of a factor of about 15 in western Europe up to a factor of 20 in eastern Europe. Nevertheless, the risk map provides an indication of the probabilistic risk involved (outside the 5—10 km zone around the reactors) due to accidental releases of nuclear power reactors, and the possible reduction of risk due to the improvement of reactor safety in eastern European reactors.

SAMENVATTING

Het ongeval met de kernreactor in Tsjernobyl in 1986 heeft aangetoond dat ernstige ongevallen met kernreactoren kunnen leiden tot een grootschalige besmetting van Europa. Momenteel zijn er meer dan 200 reactoren operationeel in Europa. Dit rapport richt zich op de vraag welk risico mogelijke ongevallen met kerncentrales met zich brengen voor de Europese bevolking. Daartoe is een methode ontwikkeld om een schatting te maken van het sterfterisico dat samenhangt met mogelijke kernreactorongevallen. De nu ontwikkelde methode is toegepast voor het sterfterisico ten gevolge van stochastische effecten, en combineert de kleine kans op ongevallen met de mogelijke gevolgen. Mogelijke acute stralingsslachtoffers in de directe omgeving van de centrale (<5 tot 10 km) zijn niet beschouwd.

Een belangrijk probleem bij de evaluatie van het risico is het ontbreken van voldoende gedetailleerde veiligheidsstudies voor een groot aantal Europese centrales. In deze studie is daarom uitgegaan van een globale categorisatie van kernreactoren op basis van de belangrijkste veiligheidsvoorzieningen.

Wanneer een ongevalsemissie naar de lucht plaatsvindt, wordt de blootstelling aan radioactieve stoffen en ioniserende straling in belangrijke mate bepaald door luchtverspreiding en depositie. Probabilistisch gemiddelde waarden voor verspreiding en depositie zijn geschat voor alle Europese kernreactoren. Vervolgens is een schatting gemaakt van de levenslange blootstelling aan straling die het gevolg is van de besmetting van bodem en lucht, en de daaruit voortvloeiende besmetting van voedsel. Maatregelen om de blootstelling te reduceren zijn niet in de beschouwing opgenomen. De blootstelling is geschat over een periode van 70 jaar na het ongeval. Het sterfterisico wordt geschat voor het gelijktijdig gebruik van alle kernreactoren in Europa.

Het sterfterisico wordt weergegeven in een risicokaart van Europa. Er blijken grote verschillen binnen Europa te bestaan: een risico kleiner dan 1×10^{-8} per jaar wordt geschat voor IJsland en het uiterste zuidwesten van Spanje en Portugal. Het geschatte risico loopt op naar het oosten en bedraagt: 2×10^{-8} per jaar in Ierland, 3×10^{-8}—10×10^{-8} per jaar in Engeland en grote gedeelten van Frankrijk, Italië en Noorwegen, circa 10×10^{-8}—30×10^{-8} per jaar in Nederland, België, Duitsland en grote delen van centraal Europa. Een risico boven 100×10^{-8} per jaar wordt geschat voor grote gebieden in Rusland, Wit-Rusland en de Oekraïne. Naar de oostelijke grens van Europa neemt het risico af tot circa 10×10^{-8} per jaar.

Oosteuropese centrales domineren het risico in Europa, en dragen ook in West-Europa nog ten minste 40—50% bij aan het gemiddelde risico. Het beveiligen van Oosteuropese centrales naar westerse maatstaven zou het risico sterk kunnen reduceren.

Het berekende risico moet worden gezien als een eerste indicatieve risicoschatting, vanwege het feit dat er grote onzekerheden zijn in dergelijke schattingen. Een eerste schatting van de mogelijke onzekerheidsmarge loopt van een factor 15 in West Europa tot een factor 20 in Oost-Europa.

1 INTRODUCTION

1.1 Scope of study

Over 200 nuclear power reactors for commercial electricity production are currently operational in Europe. The safety of nuclear power reactors is a major issue in the discussion on future scenarios for power generation. The discussion focuses on the risk from the of radioactive waste disposal and the risk of severe accidents with the reactor core. The accident with the nuclear power plant in Chernobyl in 1986 has shown that severe accidents with nuclear power plants can lead to contamination of an entire continent.

This study aims at a quantitative risk assessment of the probabilistic excess death risk related to possible accidental releases to the atmosphere from European nuclear power reactors. The risk estimates must be seen as a first attempt to achieving an integrated risk assessment, since many uncertainties are involved in the calculating the source-to-risk chain. The risk assessment method presented in this report combines accident probabilities for all European reactors using air dispersion and deposition calculations with an exposure assessment model. Risk assessment methods can be valuable tools in supporting policy-makers in the decision-making process, providing a common ground for balancing costs and benefits of countermeasures. The method applied in this study could, with adjustments, also be used in the assessment of risks due to other environmental pollutants.

The excess death risk calculated in this study refers to radiation-induced cancer death, attributable to possible accidental releases. Acute deaths due to radiation exposure that may occur in the direct vicinity of the nuclear power reactors (within a range of 5—10 km) are not included in the estimations. The results thus provide individual excess death probabilities outside the 5—10 km zone around a particular reactor. Following an accident, exposure can last for a long period of time. The risk is defined as the probability of death as a consequence of the lifetime doses (70-year follow-up period) received as a result of a possible accidental release. The death risk is calculated by combining the probability of an accidental release (for all European nuclear power reactors) with dispersion and deposition calculations under average weather conditions and using a lifetime follow-up exposure-assessment model. Accident probabilities are expressed per year of operation and multiplied by the consequences in terms of attributable extra deaths.

The excess death risk is estimated for a large number of (receptor) locations in Europe (a grid of 8000 locations). For each receptor location the risk is calculated by summing the contributions to the risk from each of the individual reactors. In this way a risk map of Europe is obtained. The map provides an estimated location dependent probabilistic death risk due to accidents with nuclear power reactors. It should be noted that the risk map does not reflect the situation following a specific accident. It provides a probabilistic view of the risk involved in using nuclear power and the major areas at risk.

1.2 Approach

The risk assessment method applied in this study evaluates the environmental source-to-risk-chain for accidental releases from nuclear power plants. Such an evaluation requires an identification of sources and releases, dispersion and deposition calculations, an exposure assessment method and an integrated risk assessment. The various aspects of the risk evaluation and the approach applied in this study is shown in figure 1.1. The starting point of the risk evaluation is the estimation of accident probabilities and source terms for each of the 217 nuclear power reactors in Europe, which were included in the present evaluation. Since detailed safety studies are not available for each of the currently operational nuclear reactors, the estimates were based upon a categorization of the reactors. Chapter 2 provides the categorization, together with the estimates of accident probabilities, and accidental releases in terms of amounts of released nuclides and the nuclide composition (source terms). Release rates are estimated for 54 of the primary dose-contributing nuclides in the source term.
Following an accidental release to the air dispersion and deposition occurs. Since it is not known beforehand what the weather conditions will be when an accident occurs, we used probabilistically time-averaged dispersion and deposition calculations on the basis of a previously developed model (Van Jaarsveld, 1990). The method used in estimating atmospheric dispersion and deposition is described in chapter 3. Following an accidental release, exposure to radiation can occur through external radiation from nuclides in the radioactive cloud, from nuclides deposited on the ground, and from inhalation and intake of contaminated food products. In chapter 4 the exposure assessment model is described and applied to estimate the major dose-contributing nuclides. In this report the dose refers to effictive dose equivalent, unless specified otherwise.
In order to limit time consuming calculations the risk evaluations are based on dispersion calculations for the primary dose contributing nuclides in the various source terms. The contribution of other nuclides is accounted for by means of correction factors (see chapter 4). The integration of results from the previous chapters occurs in chapter 5, where risk maps for Europe are presented, and overall uncertainties are discussed.

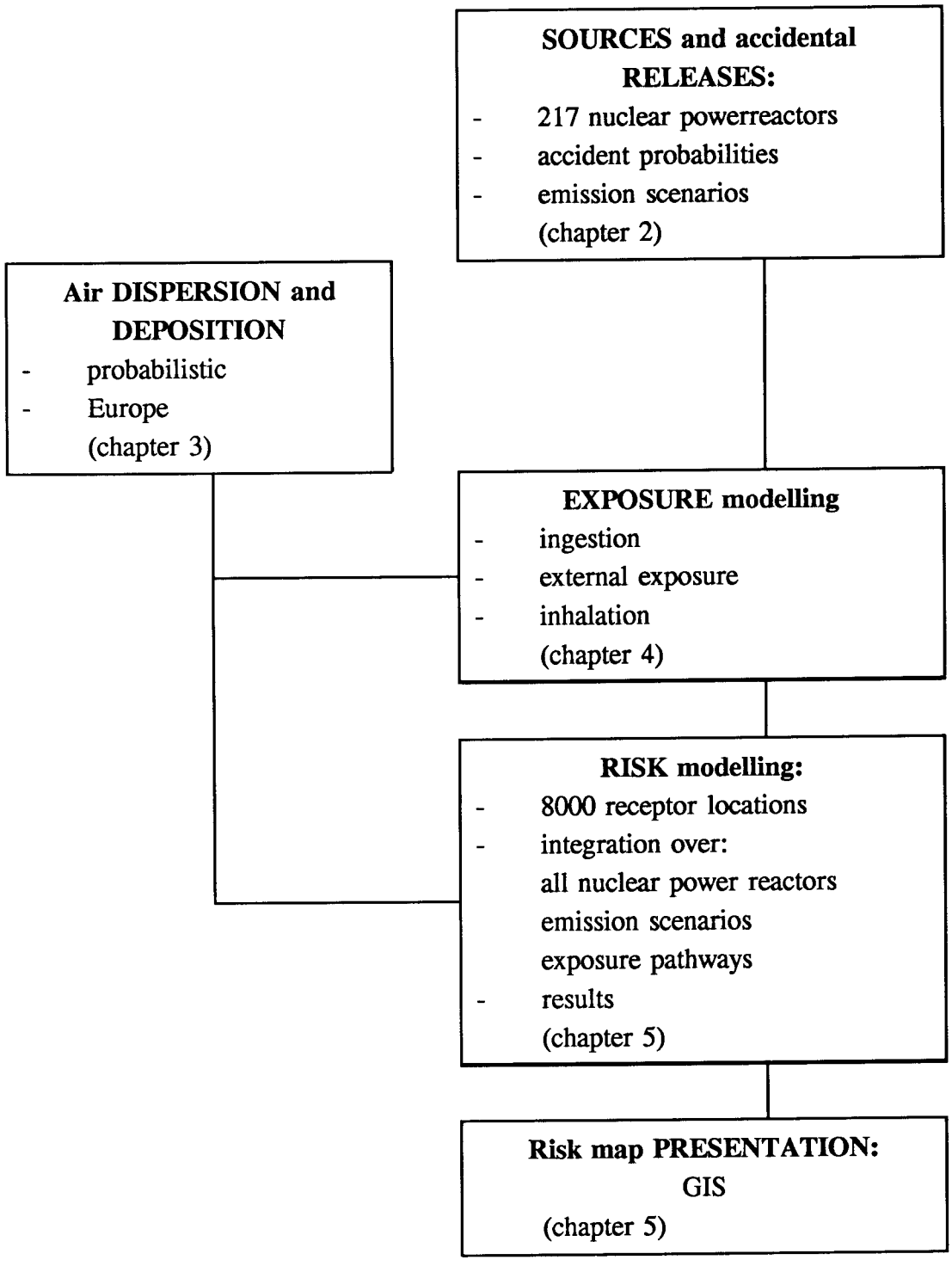


Figure 1.1 Brief outline of the risk-assessment modelling

2 SOURCE TERMS

2.1 Introduction

This chapter provides:
- an overview of nuclear power reactors in Europe (2.2)
- an estimate of accident probabilities, leading to severe damage of the reactor core (2.3.1)
- an estimate of release probabilities and corresponding source terms (releases), in case of severe core damage (2.3.2)
- an estimate of the probabilistic release, when severe core damage occurs (2.3.3)

2.2 Nuclear power reactors in Europe

Nuclear power reactors are selected on the basis of the following criteria:
- operational in July 1992 in Europe;
- their electrical power is more than 50 MW_e.

The selection is based on a list provided by Atomwirtschaft (August/September 1992), entitled "Kernkraftwerke in Europa 1992". Four reactors indicated in that list are excluded, because of the limited electrical power of 10 MW. The selected 217 nuclear power reactors in Europe are categorized in 9 different reactor types (according to the categorization provided by Eendebak *et al.* (1992)):

LWR's Light Water Reactors:

PWR (Pressurized Water Reactor). This is a light water moderated and water cooled reactor (LWR), designed according to "western" ideas about safety systems and containment. The safety systems are redundant and the containment is able to withstand breakage of the largest primary junction pipe of the cooling system.

PWR-V230 (Pressurized Water Reactor, type V230). This is the oldest PWR of Russian design. The redundancy of the safety system and the containment are very limited compared to the above mentioned PWR's. The enclosure systems are designed to only withstand a breakage of cooling pipes with a maximum diameter of 38 mm.

PWR-V213 (Pressurized Water Reactor, type V213). This is an improved version of the PWR-V230. Safety systems are redundant, but the containment is limited.

PWR-V1000 (Pressurized Water Reactor, type V1000). The safety system of this PWR is comparable to that of the 'western' PWR's.

BWR (Boiling Water Reactor). This type is, like the PWR's, designed according to "western" ideas about safety systems and enclosure systems.

LWGR (Light Water Graphite moderated Reactor). This reactor of Russian design (also

called RBMK) is a graphite moderated, 'light water' cooled reactor, in which the nuclear fuel is stored in numerous vertical pressure pipes. These pipes are imbedded in graphite (in the RBMK-1000 (the 'Chernobyl type') this graphite amounts to about 1800 tons). A containment is lacking.

GCR's *Gas Cooled Reactors:*

GCR (Gas Cooled Reactor). These graphite moderated reactors are cooled with carbon-dioxide and enclosed by a steel or concrete pressure vessel. This pressure vessel can be regarded as the enclosure system.

AGR (Advanced Gas Cooled Reactor). This is an improved version of the GCR. The nuclear material can be brought into the reactor core during operation when the maximum overreactivity is limited.

FBR *Fast Breeder Reactor.*

These reactors are all sodium cooled. The nuclear material is mainly enriched with plutonium and covered with a stainless steel coating. The loss of coolant is almost impossible due to the low pressure and a double construction of the reactorvessel.

Table 2.1 provides the number of power reactors of each of the above types for the various European countries. Table A1.1 in appendix 1 provides a detailed list of the nuclear power reactors considered in this study, together with information on the reactortype, location and accident probability (see next section). Figure 2.1 shows the location of these power reactors.

2.3 Accident probabilities and source terms

2.3.1 *Probability of a severe damage to the reactor core*

Eendebak *et al.* (1992) categorized the operational nuclear power reactors in four 'risk-classes', based on reactor design, calculated accident probabilities (through PSA's: Probabilistic Safety Assessments), and the experience of nuclear accidents in the past. Each risk class is characterized by a probability rate of severe damage to the reactor core or melt down (further indicated as core damage): 10^{-3} per year for reactors in class 1, 10^{-4} per year for reactors in class 2, 10^{-5} per year for reactors in class 3, and 10^{-6} per year for reactors in class 4.

In general power reactors are classified as class 2. Classification in classes 3 and 4 occurs when the (quality of) design, the redundancy of safety systems, and/or the industrial organization are considerably better, and a probability safety analysis supports an estimated probability of core damage substantially lower than 10^{-4} per year. Reactors with a poor quality of design and/or severe doubts of the safety procedure/culture were classified in class 1 (see table 2.2).

Table 2.1 Number and type of nuclear power reactors per country (situation in July 1992).

Type / Country		PWR	PWR-V230	PWR-V213	PWR-V1000	BWR	LWGR	GCR	AGR	FBR
Belgium		7								
Germany		14				7				
Bulgaria			4		2					
France		53						2		2
Finland				2		2				
Hungary				4						
Netherlands		1				1				
Spain		7				2				
former Czechoslovakia			2	6						
Great Britain								22	14	1
former Soviet Union			4	4	16		19			1
former Yugoslavia		1								
Sweden		3				9				
Switzerland		3				2				
Total	217	89	10	16	18	23	19	24	14	4

Table 2.2 The classification of the probability of severe damage to the core for the different reactor types (according to Eendebak *et al.* (1992)).

Type / Risk class		PWR	PWR-V230	PWR-V213	PWR-V1000	BWR	LWGR	GCR	AGR	FBR
10^{-3} per year			10				19			
10^{-4} per year		51		16	18	19		24	14	4
10^{-5} per year		35				4				
10^{-6} per year		3								
Total	217	89	10	16	18	23	19	24	14	4

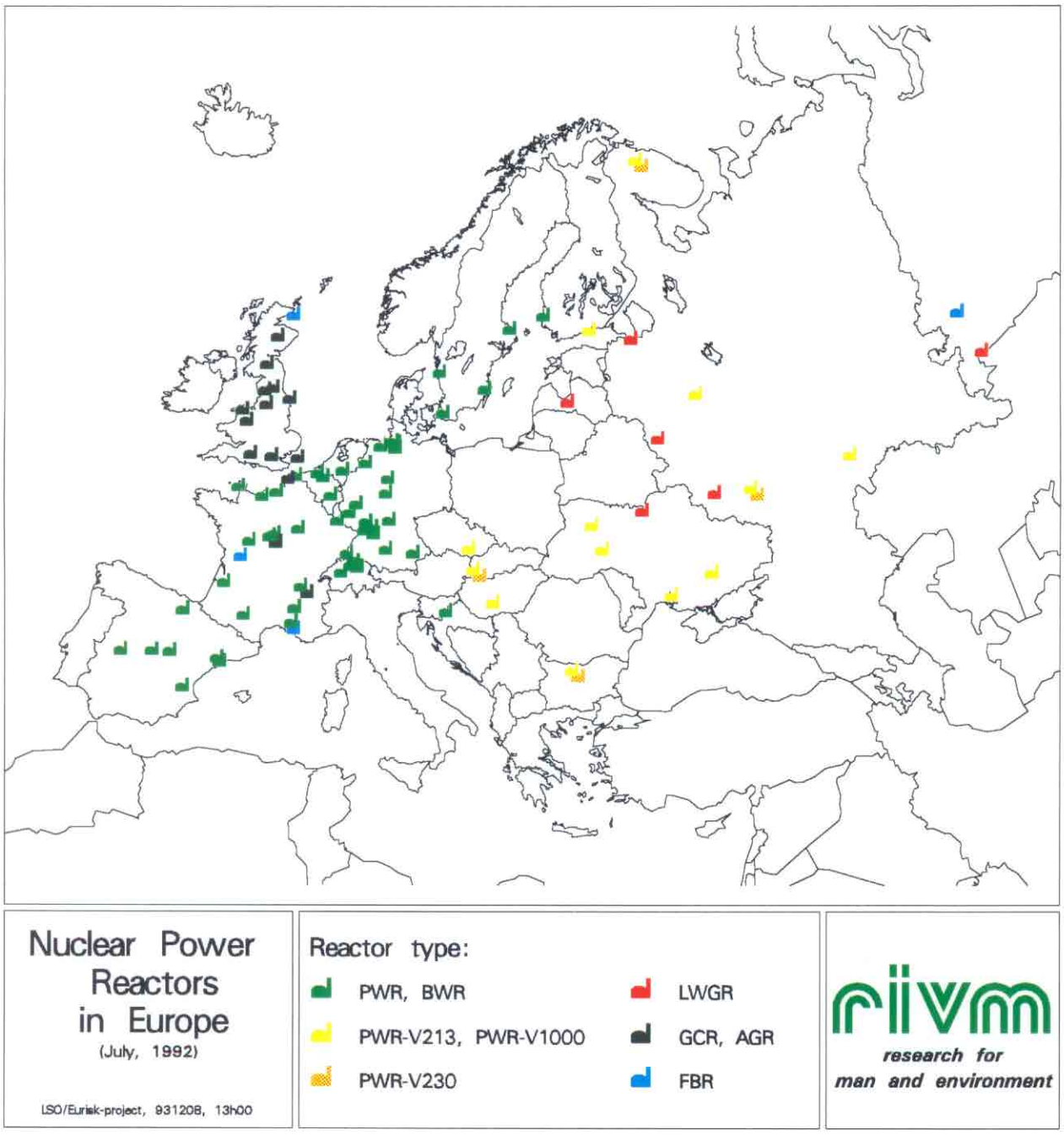


Figure 2.1 Locations and reactor types of the 217 European nuclear power plants operational on July 1st 1992, producing an electrical power in excess of 50 MW

2.3.2 *Release scenarios*

When a core damage occurs various accident scenarios are possible. The different accident scenarios can lead to different atmospheric emissions of nuclides. This section deals with the probabilities that releases occur, under the condition that severe core damage (or a meltdown) has occurred (conditional probabilities and releases). Estimates are needed for the conditional probability that an accident scenario will occur, and for the extent of the emissions associated with each scenario.
Accident scenarios and release probabilities for the various reactor types were estimated by Eendebak *et al.* (1992). We applied their estimates and considered four accident scenarios resulting from a core damage accident:

"Early release":	a large part of the nuclear inventory is released into the environment in a short period of time, due to a failure of the enclosure system (Chernobyl type of accident).
"Bypass of containment":	the enclosure system is malfunctioning due to an improper closure of the ducts through the containment. In this case the nuclides are "bypassing" the enclosure system.
"Late release":	the enclosure system fails after a certain time due to high pressure or due to the core melting through the foundation.
"No containment Failure":	the enclosure system works properly, so that (almost) no radionuclides are released (Three Miles Island type of accident).

The conditional probabilities for the various scenarios and subsequent releases are not evaluated in detail for European nuclear power reactors. However, such scenario studies are available for some of the LWR-reactors in the USA (USNRC, 1987; USNRC 1989). Despite large differences in reactor design Eendebak *et al.* (1992) concluded from those studies that the conditional probabilities for the various scenarios are similar for various reactortypes resulting in a general indication of accident probabilities, as provided in table 2.3. The probabilities provided in column 3 are the conditional probabilities, in case of a severe damage to the reactor core. It is clear that in some reactortypes even an early release of radionuclides can not be excluded. The estimates depend strongly on the expert judgement in the PSA-methodology.

Based on the chemical and physical characteristics the emitted nuclides can be divided into nine different groups. For each of the accident scenarios and for each nuclide group the expected fraction of the reactor inventory that is released to the atmosphere is estimated (see table 2.3; derived from Eendebak *et al.* (1992), based upon the results obtained by NUREG-1150 (USNRC, 1987 and 1989). It should be realized that the generalization provided in the study by Eendebak *et al.* (1992) does not account for significant detailed differences among the various reactors of the same type. This implies that for a specific reactor the analysis can

differ significantly.

The total amount released also depends on the reactor inventory, which is defined mainly by the thermal power of the reactor, but also varies with the 'aging' of the fuel (fuel cycle) in the reactor. Appendix 1 provides a list of inventories for the main dose contributing nuclides, based on a reactor with 3000 MW_{th} thermal power in the middle of the fuel cycle. A reactor with 3000 MW_{th} thermal power is considered representative for a reactor with an electrical power of 1000 MW_e. For reactors with different electrical powers, the reactor inventories are scaled directly proportional to their electrical power, and calculated from the inventories provided in appendix 1, table A1.2.

Table 2.3 Conditional probability of accident scenarios, and accompanying release fractions for nine nuclidegroups[1] (estimates according to Eendebak *et al.* (1992))

Reactor-type	accident scenario	Conditional probability	Xe	I	Cs	Te	Sr	Ru	La	Ba	Ce
LWR[2]	early	0.02	1.0	0.1	0.05	0.05	0.01	0.002	0.001	0.01	0.002
	bypass	0.02	0.8	0.2	0.2	0.1	0.02	0.005	0.002	0.03	0.006
	late	0.2	0.9	0.05	0.01	0.001	5×10^{-4}	1×10^{-5}	1×10^{-4}	5×10^{-5}	1×10^{-4}
	enclosure	0.76	0.005	5×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}
PWR-V230	early	0.40	1.0	0.1	0.05	0.05	0.01	0.002	0.001	0.01	0.002
	bypass	0.02	0.8	0.2	0.2	0.1	0.02	0.005	0.002	0.03	0.006
	late	0.2	0.9	0.05	0.01	0.001	5×10^{-4}	1×10^{-5}	1×10^{-4}	5×10^{-5}	1×10^{-4}
	enclosure	0.38	0.005	5×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}
LWGR	early	0.1	1	0.6	0.4	0.1	0.01	0.007	0.002	0.01	0.002
	late	0.9	1	0.6	0.4	0.1	0.01	0.007	0.002	0.01	0.002
GCR	late	0.1	1	0.6	0.4	0.1	0.01	0.007	0.002	0.01	0.002
	enclosure	0.9	0.01	1×10^{-5}	-	-	-	-	-	-	-
AGR	late	0.01	1	0.6	0.4	0.1	0.01	0.007	0.002	0.01	0.002
	enclosure	0.99	0.01	1×10^{-5}	-	-	-	-	-	-	-
FBR[3]	early	0.01	1	0.6	0.4	0.1	0.01	0.007	0.002	0.01	0.002

1) Xe-group: Xe, Kr; I-group: I, Br; Cs-group: Cs, Rb; Te-group: Te, Sb, Se; Ru-group: Co, Ru, Rh, Pd, Mo, Tc; La-group: La, Zr, Nd, Eu, Nb, Pm, Pr, Sm, Am, Cm, Y; Ce-group: Ce, Pu, Np.

2) LWR : PWR, PWR-V1000, PWR-V213, BWR

3) the early accident scenario is considered the only scenario with relevant emissions

2.3.3 *Calculation of the probabilistic release*

The probabilistic conditional release is defined as the expected release, under the condition that the reactor core is heavily damaged. The probabilistic source term is calculated by means of:

$$L_n = \sum_i p_i \cdot f_{i,n} \cdot A_n \qquad [2.1]$$

where,

- L_n - probabilistic conditional release, when reactor core is heavily damaged (nuclide specific) (Bq)
- Σ - summation over all accident scenarios considered (one to four per reactor type)
- p_i - conditional probability of accident scenario i
- $f_{i,n}$ - fraction of nuclide inventory released to the atmosphere (nuclide specific)
- A_n - reactor inventory (nuclide specific) (Bq).

The probabilistic conditional releases for all reactor types considered in this study are provided in table A1.3 of appendix 1.

2.4 Discussion

Detailed safety analyses are not available for many of the European nuclear power reactors. This provides a problem in evaluating the risks of accidental releases. Nevertheless it is possible to provide some general characteristics of possible source terms on the basis of:

- results obtained from safety studies for various nuclear reactors
- general considerations regarding the probability of accidental releases, based on the experience of nuclear power generation obtained in the previous decades.
- general expert judgement, regarding safety culture and safety status of nuclear power reactors, for which other knowledge is lacking.

Results regarding the above items were obtained from a study performed by KEMA (Eendebak *et al.*, 1992). It is realized that the estimates of accident probabilities are highly uncertain. Eendebak *et al.* (1992) estimate the uncertainty per reactor category to be a factor of three for western European reactors, and a factor of 10 for the eastern European countries and reactor types.

General considerations regarding accident occurrences arrive at similar overall uncertainties: thusfar over 6000 reactor years worldwide have led to the occurrence of two reactor core accidents (Three Miles Island, and Chernobyl). This implies an overall estimated accident probability of about 3.3×10^{-4} per reactor year. The average accident probability density for all European reactors based upon the reactor categorization used in this study (derived from table 2.2) is 2×10^{-4} per operational year per reactor. Based upon the previous experience the

average accident probability lies in the range of 1×10^{-3} — 6×10^{-5} (see appendix 2) per reactor year (95% confidence interval).
The previous experience regarding accidental releases can also be used to give an indication of the average accidental release fraction, and the possible upper limit. In appendix 2 we estimate an upper limit which is six times higher than the average estimated by Eendebak *et al.* (1992). Thus a factor of 6 up as well as down is used as a rough indication of the 95% confidence interval for estimates in western Europe.
Such general considerations however cannot provide insight into possible large differences between the safety aspects of the various individual reactors. The general idea is that western reactors are safer, due to a better safety culture and a better reactor design.

Apart from the lack of knowledge on specific reactors and reactortypes, a problem with the issue of prognostic risk estimates for accidents, is the fact that the reactor and its safety procedures are under constant improvement.

3 ATMOSPHERIC DISPERSION

3.1 Introduction

The risk of a possible accidental release is related to probabilistic estimates of ground deposition and (time-integrated) air concentration. Therefore a method is needed to calculate dispersion and deposition over the entire European continent, for all plant locations and averaged for all weather conditions. The purpose of this chapter is to provide such a method.
A (theoretically) straightforward approach would be to run a long range atmospheric transport model for all possible accidents that might occur with the nuclear power station under consideration and for all meteorological conditions that might be possible. Since over 200 powerreactors are involved in the study, and a large number of nuclides, such an approach would lead to long calculation times and extensive datahandling. Therefore another approach was chosen, applying an atmospheric transport model which provides average dispersion and deposition estimates, based upon a statistical evaluation of meteorological conditions.
The method applied in this study is briefly described in section 3.2. In order to limit calculation times and data handling in the integrated risk calculations some simplifications are provided (section 3.2.2). A comparison of model results with deposition data following the Chernobyl accident is provided in section 3.3. Indications of uncertainties in dispersion modelling are discussed (3.4).

3.2 Methodology

3.2.1 *Atmospheric transport*

Estimates for air concentrations and deposition were obtained applying the Operational atmospheric transport model for Priority Substances (OPS-model), which was developed at RIVM (Van Jaarsveld, 1990). This model is based upon a statistical evaluation of the meteorological conditions, and provides time-averaged air concentrations and yearly depositions for continuous constant releases in a representative meteorological year. The OPS-model provides data in a user defined grid area as will be illustrated later.
We are interested in estimates for time-integrated air concentrations and total deposition following an accidental release. The question is how calculated results obtained for continuous releases relate to probabilistic evaluations of accidental releases over short periods in time.
We will briefly address this question here; further substantiation of the argumentation is provided in appendix 3. Three assumptions are made:

- accident probabilities are constant over time and thus not related to weather (dispersion and deposition) conditions,
- the probability that an accident occurs during the operational period of a nuclear reactor

is small (in line with the accident probability rates provided in chapter 2)
- risk is linearly related to time-integrated concentration in air and deposition.

A probabilistic approach of risk assessment implies that the probability of an accidental release is considered over a certain time-period. A release can occur at any time with a constant probability rate over time. The probabilistic estimate of the accidental release per unit time equals the multiplication of the accident probability rate p (per unit time) and the accidental release L. Due to our assumption that p and L are independent of the weather conditions, pL can be considered as the expected release rate at any moment in time. This situation is fully equivalent to a constant continuous release rate of size pL. It can be shown that the statistical average of the time-integrated air concentration per accidental release L equals the average air concentration found for a continuous release rate (pL) divided by the accident probability density p. Also the probabilistic average total deposition following an accidental release L equals the yearly deposition divided by the accident probability density p.
The OPS model handles one substance per model run. This implies that multiple modelruns are necessary if physical or chemical properties of the released nuclides vary.

3.2.2 *Application and data reduction*

Application of the OPS model provides data on air concentration and deposition within a user specified grid. With a resolution of 100 x 100 km², it is possible to model an area of 3500 x 3500 km² in one run, thus almost completely encompassing Europe. However a major disavantage is the poor resolution in the vicinity of plants. In order to overcome the problem of low resolution for large areas, two separate runs were performed, with different resolutions, and covering a wide range of distances from the source: from less than 10 km up to more than 2500 km. For both those runs one source was situated in the centre of the matrix. The evaluation of releases from over 200 nuclear power reactors would still imply considerable data-handling and/or calculation times. In order to limit datahandling and calculation times we fitted user-functions describing the relationship between distance from the source and the air-concentration and deposition results obtained with the OPS-model (see figure 3.1). The OPS-results were fitted with a user defined function, given by:

$$\frac{A \cdot e^{(-B \cdot r)}}{r^{C}} \qquad [3.1]$$

where,

A - in Bq·a·m^{-3} per Bq released if air concentrations are modelled and
in Bq·m^{-2} per Bq released if depositions are modelled
B - dimensionless parameter
C - dimensionless parameter and

r - = r'/r_0 with r_0 = 1 km and r' the source to receptor point distance in km

A, B and C are the parameters that were fitted to the OPS-results and r reflects the distance to the point of release in kilometers. The exponential factor in this function is a decrease factor (radiological decay, removal by deposition etc.) and the power factor r^C represents the dilution factor that is the consequence of transport distance, and horizontal and vertical dispersion.

In order to account for differences in relation to wind-directions, the OPS matrix was split in four geographical quadrants, representing major wind directions: North-East (south-western winddirection), South-East, South-West and North-West. For each quadrant a separate fit of the user function was obtained (see figure 3.1). The OPS-model handles one substance (nuclide) per evaluation. Modelling results were obtained for two nuclides: Cs-137 and I-131. The OPS model was run for a regular, constant release of a slow-decaying material (Cs-137) without internal energy of the plume, with a parameter set as given in appendix 4. The produced model results for the mean concentration in air and deposition on ground were graphed as a function of distance for the four geographical quadrants. Results for the North-East quadrant are provided in figure 3.2.

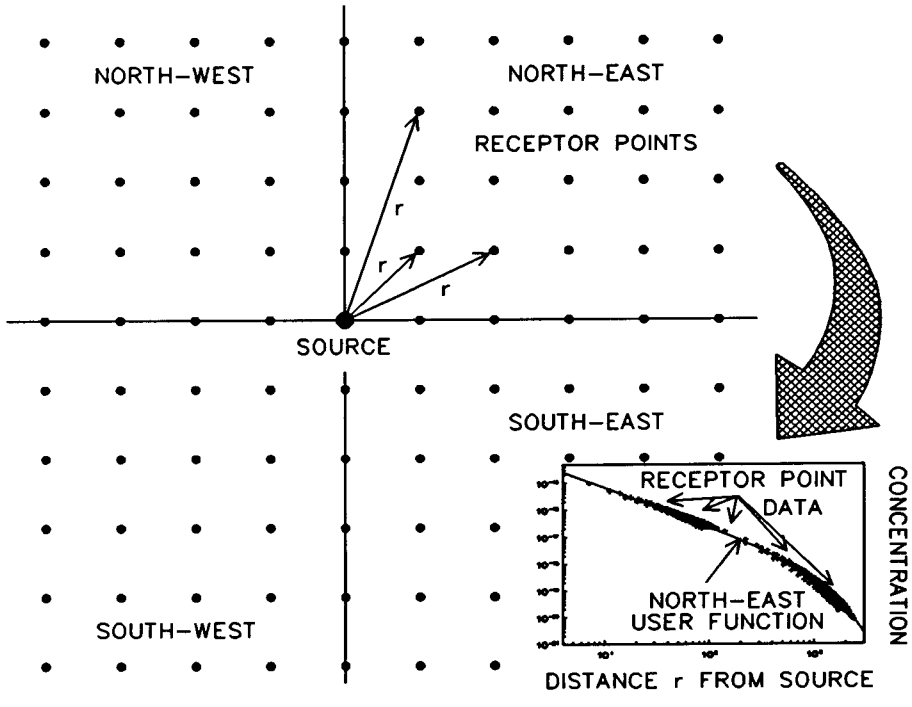
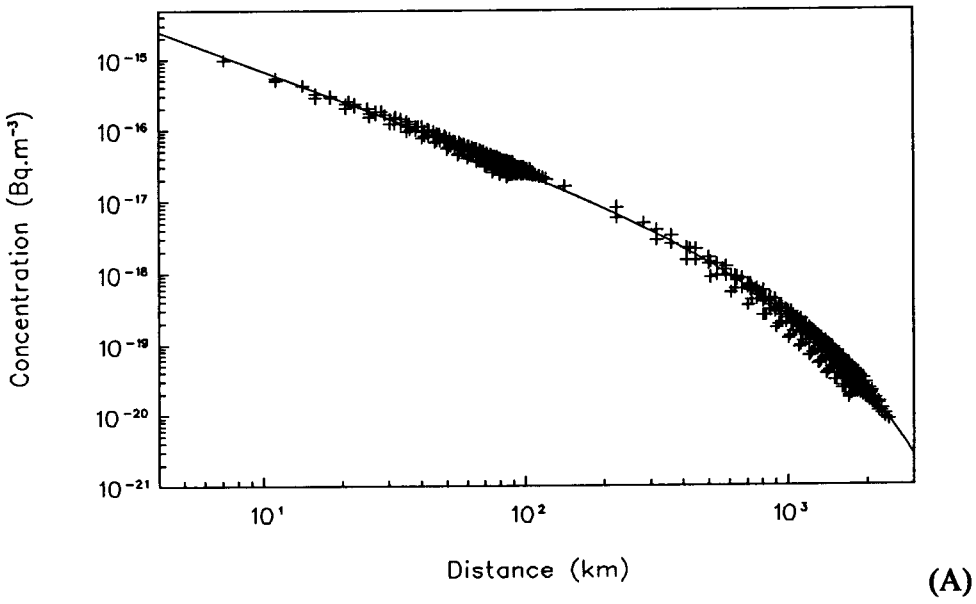
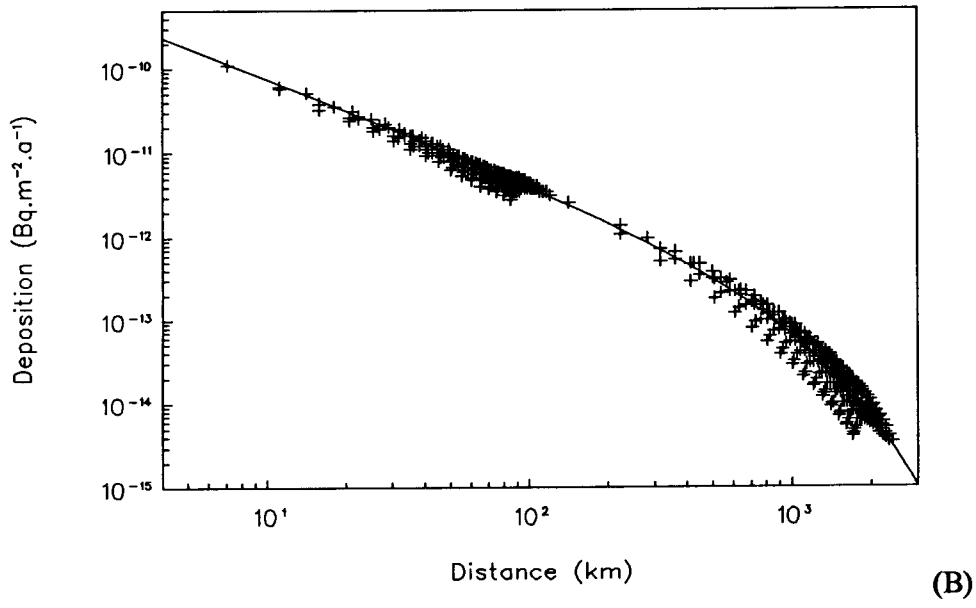


Figure 3.1 Schematic representation of the calculational method, using the concentration or deposition values in a user defined grid and fitting those per quadrant to a user defined function (see equation [3.1] and figure 3.2)



(A)



(B)

Figure 3.2 Concentration (A) and deposition data (B) for Cs-137 in the north-east quadrant as calculated with the OPS model. Activity concentrations are given as a function of distance to the source. Data on Y-axis are for a continuous and constant discharge of 1 Bq.a^{-1}. (Stack)sourceheight is 100m and the plume has no thermal energy. The user function (—), fitted to the data, is also given.

Applying this method sets of values for A, B and C can be found for concentration in air and deposition. The values that were calculated with the user defined function given in equation [3.1] do not deviate more than a factor of two to three from the OPS model results, if data for separate quadrants are fitted, for the range of 10—2500 km.

Different half lifes of radionuclides

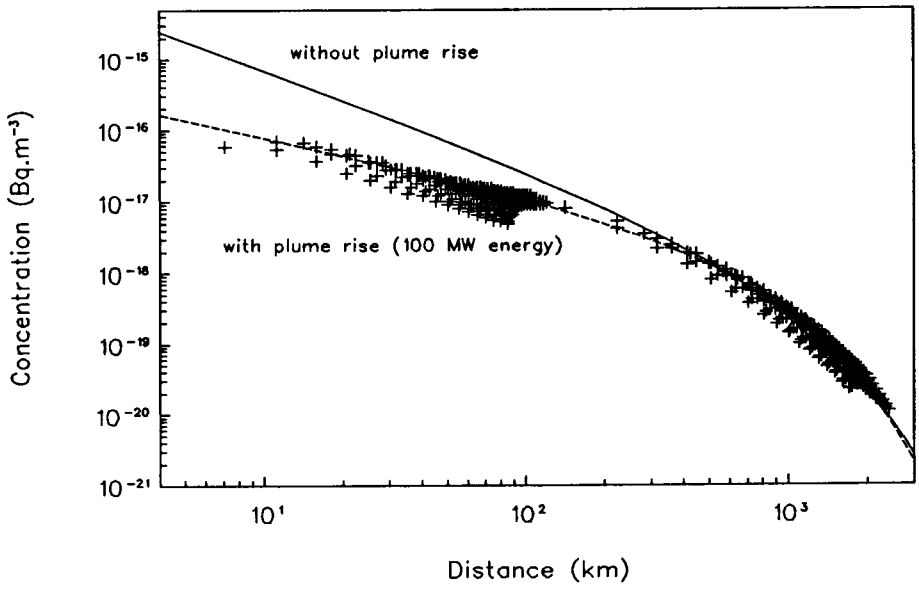
Fitted values for the parameters in the userfunction for Cs-137 are given in appendix 4 (table A4.2). The nuclide Cs-137 can be used as representing the long-lived radionuclides. For a short-lived radionuclide like I-131 ($t_{½} \approx 8$ days) especially for longer distances the concentration in air will fall off more rapidly than that of Cs-137.

Different plume rise

Another parameter that affects the dispersion is the internal energy of the plume that is discharged from the source. If a high internal energy of 100 MW is assumed, an effective plume rise will occur. Results for air concentration and deposition are shown in figure 3.3. The supplemental plume rise will result in a lower air concentration near the source compared to the situation without plume rise: a nearly tenfold difference occurs at 10 km from the source. For longer distances from the source this effect will vanish (at a distance of around 300 km) or even change to an elevation (at distances over 300 km), because at higher altitudes (above the mixing layer) material can be transported over longer distances.
A similar pattern occurs for the deposition, although the effects near the source are less pronounced because the radioactive material could still be washed out by wet deposition from higher altitudes. At 10 km from the source deposition is no more than 3—4 fold lower in the case of plume rise. No difference occurs at around 150—200 km and for larger distances the deposition is slightly elevated in the situation with plume rise. Both results are illustrated in figure 3.3.

3.3 Comparison of OPS-modelling with measured data

The OPS-model in its present form was defined to describe average air concentration and deposition in the Netherlands, thus the weather characteristics are specific for the Netherlands. Just in order to have a rough idea about the applicability for accidents with nuclear power reactors, we have compared model calculations for the Chernobyl location and sourceterm, with actual deposition measurements performed following the reactor accident (appendix 5). It should be noted that the OPS-model provides statistical averages and thus a comparison with Chernobyl data, representing a specific ('real time') accidental situation, can only be seen as an indication of possible uncertainties.
The OPS-calculated values, represented by the user defined fitting functions, are within the range of values found from the measured dataset. Deviations found in this comparison could amount to nearly a factor of 10 for specific locations. Such deviations could be expected due to specific local weather conditions, like rainfall, during passage of the radioactive cloud.



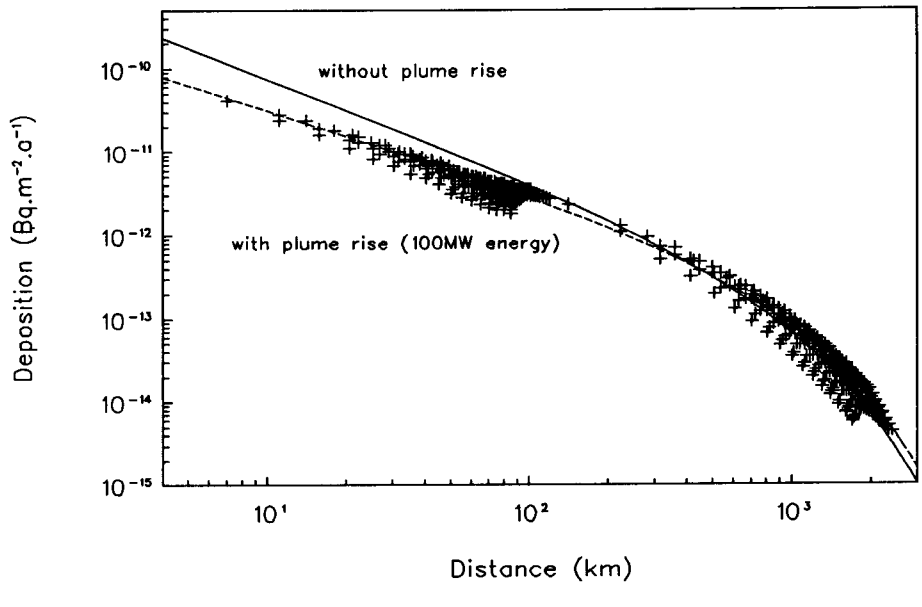


Figure 3.3 Concentration (A) and deposition data (B) for Cs-137 in the north-east quadrant as calculated with the OPS model. Activity concentrations are given as a function of distance to the source. Data on Y-axis are for a continuous and constant discharge of 1 Bq a^{-1}.
(Stack)source height is 100 m and the plume has a thermal energy of 100 MW. The user functions, fitted to the data with (--) and without (—) plume rise, are also given.

Better agreement should be expected if deposition data are averaged over larger areas. UNSCEAR (1988) provided deposition data averaged over larger areas, and a comparison with the model estimates shows deviations are within a factor of 3 (see appendix 5, figure A5.1). The OPS estimates are in most cases lower than the measured data. A result which could be expected, when it is realized that most measured data were obtained in directions north and west from the plant location, and that prevailing winddirections at the moment of the accident were in those directions.
The above gives some confidence that on average the error using OPS is within a factor of 2—3.

3.4 Discussion

3.4.1 *Uncertainties in the atmospheric dispersion modelling*

When atmospheric dispersion is modelled as described above the uncertainties in the end results of the modelling process are due to:

- limitations of the model (e.g., restrictions with respect to the area modelled, the meteorology used)
- uncertainties in the modelling process (e.g., the way deposition, plume rise and dispersion are modelled)
- the simplifications involved in making simple functions fitting the concentration and deposition in different wind directions

Limitations of the model
One of the biggest restrictions is that the weather is limited to a dataset of Dutch weather compiled over a period of ten years. Examples of application of this Dutch probabilistic dataset to, e.g., acidification modelling in Europe (Asman en Van Jaarsveld, 1990; Van Jaarsveld, 1989), indicate however that deviations from more realistic 'European weather' are not very significant (within a factor of 2). The comparison with Chernobyl data provides quite satisfactory results as well (average difference no more than a factor of 3), especially when realizing that the OPS model was not designed for the purpose of real-time evaluations (see appendix 5). Thus, some confidence is gained that the extrapolation to Europe is not a source for large errors if weather data are used in a probabilistic way. In some parts of Europe there may be large deviations from the Dutch 'average' weather although this is not the case for Europe at large. Local topography (e.g., valleys, mountainranges, lakes) may result in varying weather conditions leading to either higher or lower deposition of nuclides. Local effects therefore may deviate strongly from model results.
Because of this phenomenon of local topography of the landscape it is recommended that at least further away from the Netherlands but especially in mountainous areas the model results are interpreted as general only.

Uncertainties in the modelling process

In the modelling process a specific set of parameter values has been chosen (e.g., deposition velocity, washout coefficient, roughness length). All these assumed values give an inevitable uncertainty in the model results that is taken to be about 50% in total.

Because of model requirements the radionuclides were modelled as gasses, with a deposition velocity, while an aerosol approach should have been used, at least for cesium. The uncertainty introduced in this way is not known exactly but is thought to lie within a factor of two. This is not contradicted by the comparison made with the deposition observed from the Chernobyl accident (appendix 5).

Simplifications of the model

An additional uncertainty is due to the approximation of the model results with an exponential equation. The uncertainty that is introduced in this way falls within a factor two to three as can be seen in figures 3.2 and 3.3. Larger uncertainties can occur near the source (less than 10 km) and far away from the source (more than about 2500 km). This study does not focus on evaluations in the close proximity of the sources, and the grid cells used in the risk evaluations are at least 50×50 km. The deviations at very large distances are less relevant since the risk contributions from plants at such distances become small.

The uncertainty due to the approximations of the OPS model will partly cancel out, when dsipersion results for many different reactor sites and different wind directions are added.

Altogether the uncertainty in the model results will fall within a factor of four as far as general concentration and deposition values are concerned. For specific locations in Europe a significant deviation might occur due to local topography and/or weather conditions.

4 EXPOSURE OF THE POPULATION

4.1 Introduction

This chapter describes the methods used for the dose-calculations, and the selection of the major contributing nuclides in the various sourceterms. Purpose of the exposure assessment is to estimate the accumulated lifetime dose-commitment due to the passage of a radioactive cloud and the deposition of radioactive material from that cloud on vegetation and soil. The dose received is calculated assuming that people remain at the same location for a period of seventy years. The exposure pathways considered are inhalation, ingestion, and external exposure. Figure 4.1 provides a schematic representation of the exposure-model used in this study. Section 4.2 gives a detailed description of the various modules in the exposure assessment model. The exposure assessment model described here is implemented in the NucRed computer program, which was developed at RIVM (Slaper, 1993).

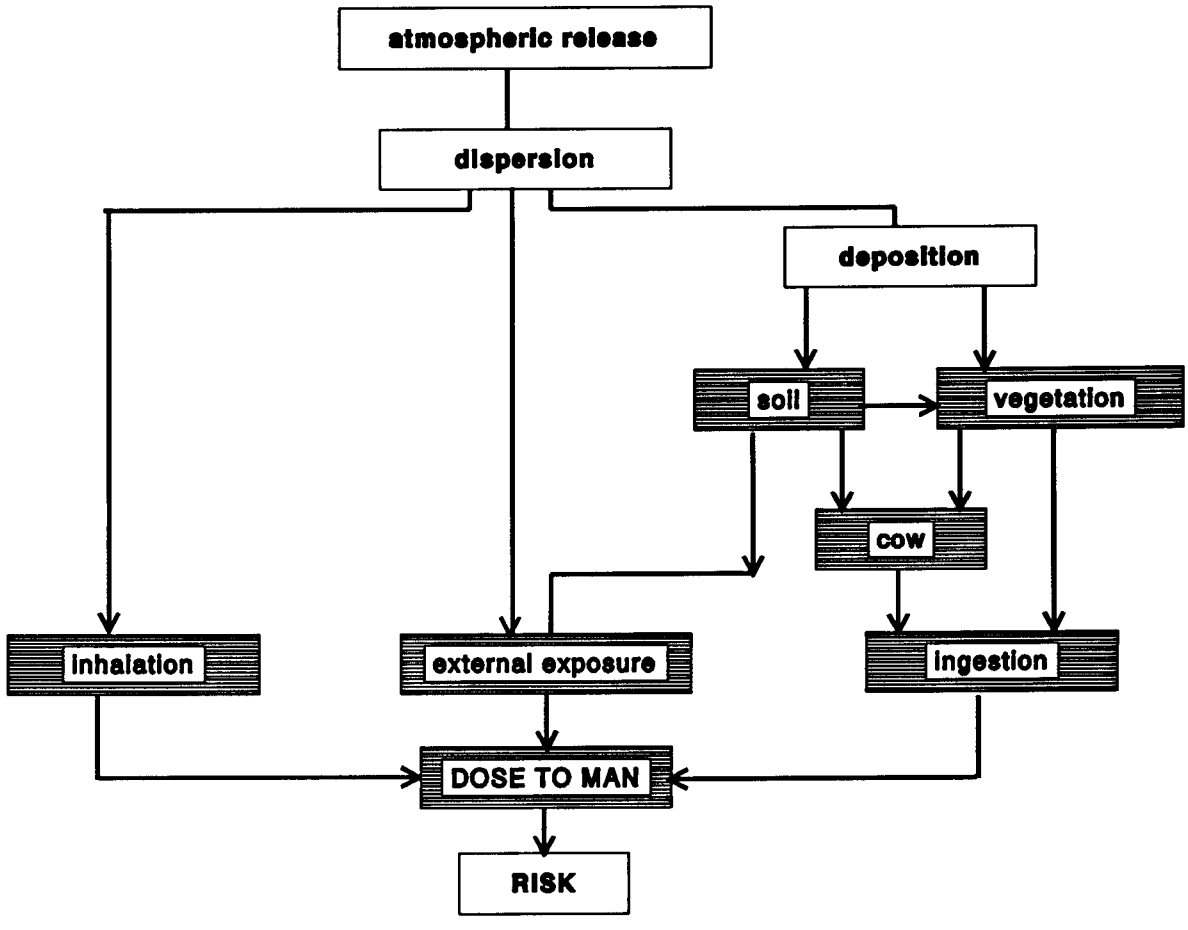


Figure 4.1 Pathways considered in exposure modelling; gray boxes indicate modules in exposure assessment calculations

For each of the source terms provided in chapter 2 the relative contribution of the various nuclides to the overall dose was estimated. Taking into account the relative contribution of

the various nuclides the primary important nuclides were selected for the risk map calculations in chapter 5 (selection results are shown in section 4.3). The dose-contribution of the non-selected nuclides were attributed to the selected nuclides (section 4.3). Results of the exposure model (obtained with NucRed) are compared with other models and discussed in section 4.4.

4.2 Exposure assessment methodology (NucRed-model)

4.2.1 *Inhalation*

During the passage of the radioactive cloud nuclides are inhaled. Resuspension of deposited uclides is not considered. The inhalation dose is calculated for each nuclide by multiplying the total amount of inhaled activity with the dosec nversion factor for inhalation for the particular nuclide. The total inhaled activity is directly proportional to the breathing volume and the time-integrated air concentration. The modelling allows for a possible difference between outdoor and indoor time-integrated air concentration. In the calculations used for the risk maps (see chapter 5) indoor air concentration is assumed to be equal to outdoor air concentration.
The following expression was implemented in NucRed (UNSCEAR, 1988):

$$D_{inh} = C_A \cdot V_B \cdot DC_{inh} \cdot (1 - F_{ind} + F_{ind} \cdot r_{ind}) \qquad [4.1]$$

where:

- D_{inh} - total inhalation dose (Sv)
- C_A - time-integrated air concentration (Bq m^{-3} day)
- V_B - breathing rate (23 m^3 day^{-1} (ICRP-23))
- DC_{inh} - nuclide specific dose conversion factor for inhalation (Sv·Bq^{-1}; see appendix 6, Table A6.1 (based upon: Nosske *et al.*, 1985)
- F_{ind} - fraction of time spent indoors (0.7)
- r_{ind} - reduction fraction for indoor air concentration (no reduction is used in the calculation (value 1.0))

4.2.2 *Ingestion*

Deposition takes place during cloud passage, leading to a contamination of crops and soil. The contamination of cro s occurs through at least two pathways: direct interception during cloud passage and uptake from the soil. After cloud passage the soil remains contaminated, however removal from the upper layers in combination with nuclear decay will decrease the soil concentration. The intake of radionuclides through food can be either by direct consumption of contaminated crops, or indirect through consumption of milk and meat from animals, which consumed contaminated pasture (grass) and soil. Five major food categories are distinguished in the ingestion modelling: vegetables, cereals, roots/tubers, and milk and

meat from cows.
The methodology regarding the modelling of the ingestion pathway is in line with MORIS (Blaauboer *et al.*, 1992), with an exception for the water pathway, which is not included because preliminary estimations showed a relatively small contribution of this pathway. The relatively insignificant contribution of the waterpathway is also reported by Simmonds *et al.*, 1987. A year round consumption of fresh food is assumed, in line with assumptions in the MORIS-project (Blaauboer *et al.*, 1992). Since the concentration in plants due to uptake from the soil is considered to be directly proportional to the soil concentration, we calculated the time-integrated soil concentration over the total evaluation period of 70 years.

soil concentration
The time integrated soil concentration due to a total deposition of O_A per unit area is calculated by means of (IAEA, 1982):

$$C_s = \frac{O_A}{S \cdot (\lambda + \lambda_1)} (1 - e^{-(\lambda+\lambda_1) \cdot T_{end}}) \qquad [4.2]$$

where:

- C_s - time integrated (dry) soil concentration (Bq kg^{-1} day)
- O_A - total deposition per unit area (Bq m^{-2})
- S - mass of soil in plough layer per unit area (kg m^{-2}), calculated multiplying depth of plough layer (m) with soil density (kg m^{-3}; appendix 6; table A6.4)
- λ - physical decay constant for nuclide considered (day^{-1}; appendix 6; table A6.1)
- λ_1 - constant describing removal from plough layer (day^{-1}; specified in equation 4.3)
- T_{end} - end of evaluation period (day; evaluation period of 70 years is taken)

The constant describing the removal of nuclides from soil is calculated applying (Baes *et al.*, 1984):

$$\lambda_1 = \frac{W_R + W_I - W_E}{\theta \cdot h \cdot (1 + \frac{K_d \cdot \rho}{\theta})} \qquad [4.3]$$

where:

- $W_R+W_I-W_E$ - water balance: rainfall (R) plus irrigation (I) minus loss due to evaporation (E) ($m \cdot day^{-1}$; see appendix 6, table A6.4)
- θ - volumetric water content of soil (dimensionless; 0.25 is used)
- h - thickness of plough layer (m; appendix 6; table A6.4)
- K_d - soil affinity of nuclide ($m^3 \cdot kg^{-1}$; appendix 6; table A6.3)
- ρ - soil density ($kg \cdot m^{-3}$; appendix 6; table A6.4)

contamination of crops

Crops can be contaminated via two major routes: direct interception of depositing material and uptake through transfer from the soil. Contamination through irrigation with contaminated water is not included in this evaluation. It is unlikely that this exposure path contributes significantly (Simmonds *et al.*, 1987). In case of an accidental release the interception period is short compared to the growth period of crops. We are interested in the average contamination of crops. The concentration is directly proportional to the total deposition. The probabilistic expectation for the plant concentration is a sum of the contributions due to direct interception and due to the soil-plant transfer (IAEA,1982):

$$C_p = \frac{O_A \cdot F_{ip} \cdot (1 - e^{-(\lambda + \lambda_w) t_{ap}})}{Y_p \cdot (\lambda + \lambda_w)} + C_s \cdot B_v \qquad [4.4]$$

where,

- C_p - nuclide concentration in plants; time integrated ($Bq.kg^{-1}.day$)
- O_A - total deposition per unit area ($Bq\ m^{-2}$)
- F_{ip} - direct interception fraction for crop type p (appendix 6, table A6.4)
- λ - physical decay constant (day^{-1}; appendix 6; table A6.1)
- λ_w - rate constant for reduction of the concentration of material deposited on the surface of vegetation due to processes other than radiological decay (day^{-1}; appendix 6; table A6.4);
- t_{ap} - the time-period during the growing season that crops can be contaminated through direct interception of deposition (days; appendix 6; table A6.4)
- Y_p - the agricultural productivity (yield) or standing crop biomass of the edible portion of vegetation ($kg \cdot m^{-2}$; appendix 6; table A6.4)
- C_s - time integrated concentration of radionuclides in (dry) soil ($Bq \cdot kg^{-1} \cdot day$)
- B_v - concentration factor for uptake of the radionuclide from soil by edible parts of crops, in $Bq \cdot kg^{-1}$ plant tissue per $Bq.kg^{-1}$ dry soil (appendix 6; table A6.3)

Three groups of crops were considered as contributing directly to human ingestion: vegetables, root crops and tubers, and grain. Indirectly grass contributes to human ingestion through ingestion by cows and successive contamination of cow-milk and meat.
The contamination of cowmilk is calculated by means of:

$$C_{milk} = F_{milk} \cdot (I_{cow,grass} \cdot C_{grass} + I_{cow,soil} \cdot C_s) \qquad [4.5]$$

where:

- C_{milk} - time integrated concentration in cowmilk ($Bq \cdot kg^{-1} \cdot day$)
- F_{milk} - transfer to milk ($day \cdot kg^{-1}$; appendix 6; table A6.3)
- $I_{cow,grass}$ - grass intake for cow ($kg \cdot day^{-1}$; appendix 6; table A6.5)
- C_{grass} - time integrated nuclide concentration in grass, per kg fresh weight

(Bq·kg^{-1}·day)

$I_{cow,soil}$ - soil intake for cow (kg·day^{-1}; appendix 6; table A6.5)

Replacing the milk index by meat the same equation is also applied to calculate the concentration in cow meat.

The overall human ingestion is calculated by means of:

$$A_I = \sum_{p=1}^{5} (I_p \cdot F_{b,p} \cdot C_p \cdot e^{-\lambda \cdot t_d}) \qquad [4.6]$$

where,

A_I - total intake of nuclide under consideration (Bq)

I_p - human intake of foodproduct p (kg·day^{-1}; appendix 6; table A6.5)

F_{bp} - reduction factor for removal of radionuclides due to foodpreparation processes (appendix 6; table A6.5)

C_p - time integrated nuclide concentration in foodproduct p, just after harvesting/milking /slaughter prior to foodpreparation (Bq·kg^{-1}·day)

λ - decay constant of radionuclide (day^{-1})

t_d - time between harvesting/milking/slaughtering and consumption (day; appendix 6; table A6.5)

The total dose contribution through ingestion is calculated applying:

$$D_{ing} = DC_{ing} \cdot A_I \qquad [4.7]$$

where,

D_{ing} - total ingestion dose for radionuclide under consideration (Sv)

DC_{ing} - nuclide specific dose conversion factor for ingestion (Sv·Bq^{-1}; appendix 6; table A6.1)

A_I - total intake of nuclide under consideration (Bq)

4.2.3 *External exposure*

Two situations give rise to external exposure:
- exposure from the radioactive cloud, during cloud passage
- exposure due to deposited radioactivity

External exposure from radioactive cloud

The effective dose due to external exposure from the cloud is calculated applying:

$$D_{ext\ cl} = C_A \cdot DC_{ext\ cl} \cdot (1-F_{ind}+F_{ind} \cdot F_{build\ cl}) \qquad [4.8]$$

where,

- $D_{ext\ cl}$ - dose due to external radiation exposure from cloud for specific nuclide (Sv)
- C_A - time-integrated air concentration for specific nuclide ($Bq \cdot s \cdot m^{-3}$)
- $DC_{ext\ cl}$ - dose-conversion factor for unit air concentration with specific nuclide for external radiation dose from infinite cloud ($Sv \cdot s^{-1} \cdot Bq^{-1} \cdot m^3$; appendix 6; table A6.1)
- F_{ind} - time-averaged fraction of time spent indoors (0.7 is used)
- $F_{build\ cl}$ - average indoor reduction factor of external radiation from clouds (0.7 is used in baseline calculations)

The method used in evaluating the dose from external cloud shine is identical to the method applied by UNSCEAR (1988) for the evaluation of the Chernobyl-accident.

External exposure from deposited radioactivity

External exposure due to deposition of a particular nuclide is dependent upon the total deposition, the surface roughness, the removal of nuclides from the surface, the penetration of nuclides in the soil and shielding by buildings etc.. The modelling approach used in this study is equivalent to the method used by UNSCEAR (1988). The external radiation dose is calculated over three separate time-intervals:

- the first month following deposition
- the period between the first month and one year following the deposition
- the period after one year until the end of the evaluation period (in our case 70 years)

During the first month following deposition the external dose is calculated assuming a surface contamination, and shielding due to buildings is considered for the time spent indoors. The modelling for the periods after one month is similar, however two additional multiplication factors are introduced, describing shielding of radiation due to penetration in the ground, and the reduction of the surface contamination in urban areas due to runoff. UNSCEAR 1988 concluded that approximately 50% of the deposited radionuclides were lost with a half life of 7 days. The approach followed here allows for a 50% reduction after one month. The contribution to the external dose is calculated applying:

$$D_{ext\ so\ i} = DC_{ext\ so} \cdot O_A \cdot (1-F_{ind} \cdot (1-F_{build})) \cdot \frac{(e^{-\lambda \cdot t_{i-1}} - e^{-\lambda \cdot t_i})}{\lambda} \cdot F_{runoff\ i} \cdot F_{pene\ i} \qquad [4.9]$$

where,

- $D_{ext\ so\ i}$ - dose (Sv) due to external exposure from a specific nuclide deposited on the ground in period i; three periods are considered: period upto 31 days after deposition, period between 31 days and 365 days, and the

period after 365 days upto the end of the evaluation period (in present calculations 70 years)

$DC_{ext\ so}$ - dose conversion factor for external exposure from surface contamination for a specific nuclide, when no shielding occurs ($Sv \cdot s^{-1}$ $Bq^{-1} \cdot m^2$; appendix 6; table A6.1)

O_A - total deposition for specific nuclide per unit area ($Bq \cdot m^{-2}$)

F_{ind} - fraction of time spent indoors (0.7)

F_{build} - reduction factor for shielding inside buildings (0.3)

$F_{pene\ i}$ - shielding factor due to penetration of nuclides in the ground, this factor is 1 during the first month, 0.5 in the period between one month and one year, and 0.37 after one year

$F_{runoff\ i}$ - provides correction for runoff in urban areas, specification provided below; the runoff correction factor equals 1 in first month and also 1 when considering rural areas.

t_{i-1}, t_i - time at start respectively end of period i (day, see above)

In calculations where a mixed population of 50% urban and 50% rural inhabitants is assumed, the correction for runoff was calculated applying (UNSCEAR,1988):

$$F_{runoff\ i} = 1 - F_{popurban} \cdot (1 - F_{urban\ i}) \qquad [4.10]$$

where,

$F_{popurban}$ - fraction of the population living in urban areas

$F_{urban\ i}$ - fraction of contamination runoff in urban areas (0.5 after first month)

The basic calculations in this report are however performed for a rural population, and no runoff effects are considered ($F_{runoff} = 1$). The total dose due to external exposure from deposited nuclides is provided adding the contribution for each of the three periods considered for all deposited nuclides.

Correction factors applied in exposure modelling

The above modelling equations do not include effects due to daughter nuclides. For 16 of the major nuclides a correction was included to allow for the doses due to ingestion of daughter nuclides. The corrections were introduced as multiplicative ingestion correction factors: for each of the 16 mother nuclides the ingestion dose was multiplied with this correction factor. The correction factors were obtained from Kirchner (1990), and included a 50 years follow up dose due to the direct ingestion of daughter nuclides. Table A6.2 in appendix 6 provides the correction factors for the nuclides as used in this study.

Daughter nuclides can also contribute to other pathways, especially those nuclides with daughters that are strong gamma-emitters. For 11 nuclides with sufficiently short living daughters the external exposure dose conversion factor was increased with the dose-conversion factor of the daughter. For external radiation from the cloud a similar correction

was applied for Cs-137 only, to allow for the contribution of the very short living daughter Ba-137m. In order to estimate an upper limit for the error caused by the neglectance of other mother-daughter relationships in the external exposure pathway we also performed a calculation adding the daughter dose conversion factors to that of the mother for all daughters with halflifes shorter than 1000 years. The results of this calculation showed an overall increase in dose of less than 2% (considering all pathways), thus leading to the conclusion that further detailed analysis in the scope of this study was not necessary.
No correction was necessary with regard to inhalation, since nuclide selection (see next section) was calculated in the close vicinity of the plant, and decay during transport was neglected. Thus the dose contribution of daughters was accounted for in the inhalation dose conversion for the mother nuclides.
In order to take into account that no deposition occurs for noble gasses, we introduced a multiplication correction factor for krypton and xenon isotopes to provide zero deposition.

Method to select major dose-contributing nuclides
The modelling provided above can be applied to estimate the effective doses due to the passage of a contaminated cloud, and deposition of nuclides, provided that time-integrated air concentration and total deposition are available. For that purpose source terms and dispersion characteristics are needed. For the purpose of the overall probabilistic risk assessment dispersion evaluations are needed for over 200 nuclear power reactors in Europe. It is not very practicable to perform the full calculations for all 57 nuclides. Under the assumption that deposition and dispersion characteristics are similar for the various nuclides we can group the various nuclides.
We calculated the relative contribution to the overall dose for all nuclides and exposure pathways for the various probabilistic source terms given in chapter 2. The calculations were based upon a time-integrated air concentration and deposition in the close vicinity of the plant (at 10 km). We applied the functional relationships as provided in chapter 3 (equation 3.1), using the dispersion parameters for the north-eastern winddirection for Cs-137 for a release height of 100 m (as provided in appendix 4 table A4.2). As a result of this procedure deposition and time-integrated air concentrations for all nuclides in the source term are obtained. Applying the exposure model described in this chapter we calculated the effective dose for all nuclides for the various pathways and selected the primary important nuclides.

4.3 Nuclide selection for various source terms

The exposure models described above were used to estimate 'overall' dose conversion factors for all 57 nuclides considered as potential risk contributing in case of a nuclear accident. The 'overall' dose conversion factors are defined for the four major pathways, and provide the doses received per unit deposition and time-integrated air concentration. Results obtained for all nuclides and for the four pathways are given in appendix 6, table A6.6. Summing the 'overall' dose conversion factors for inhalation and external exposure from the cloud, a total

air concentration related dose conversion can be obtained. Summing the ingestion and external exposure from deposited material, provides a total dose conversion due to deposited nuclides. Table 4.1 provides the obtained total dose conversion factors for time-integrated air concentration and deposition for the major dose-contributing nuclides.

Table 4.1 Dose received per unit deposition and time integrated air concentration

nuclide	Deposition related dose conversion Sv/(Bq.m^{-2})	Air concentration related dose conversion Sv/(Bq.s.m^{-3})
I-131	4.4×10^{-9}	2.2×10^{-12}
Cs-137	1.6×10^{-7}	2.3×10^{-12}
Cs-134	5.5×10^{-8}	3.5×10^{-12}

Combining the exposure model with the sourceterms from the various reactortypes and the dispersion calculation as described above, we were able to select the major dose-contributing nuclides for all source terms. Figure 4.2 and table 4.2 provide the results for the major sourceterms, and show that I-131 and Cs-137 are the major contributors to the overall dose at a site 10 km from the source. At longer distances this will be even more pronounced because the concentration of short lived radionuclides will fall off with distance even faster. I-131 and Cs-137 contribute 60—75% of the total dose by all 57 nuclides. Cs-134 contributes approximately 15%, and all other nuclides less than 5% each, and no more than maximally 25% in total. The conclusion from these results is, that calculations for all source terms can be based upon dispersion estimates for I-131 and Cs-137, and the contribution from other nuclides can be attributed to either of these nuclides. All iodine nuclides are grouped with I-131, and all other nuclides considered are grouped with Cs-137. The contribution of the other nuclides is accounted for by means of multiplicative scaling factors, which are defined as the total dose in a particular group (summing the contribution from all nuclides in a particular group) divided by the dose due to either I-131 or Cs-137. Since doses can be either directly related to the deposition (ingestion and external exposure from deposited material), or directly to the time-integrated air concentration (inhalation and external exposure from the cloud), we have calculated the scaling factors for deposition related doses and air-concentration related doses. This allows for maximum flexibility regarding the relation between time-integrated air concentration and deposition.

Ingestion and external exposure were found to be the major exposure pathways: ingestion covers approximately 50-54% of the total dose and external exposure contributes 36-44% to the overall dose for all reactor types except for the FBR-reactortype (see table 4.3).

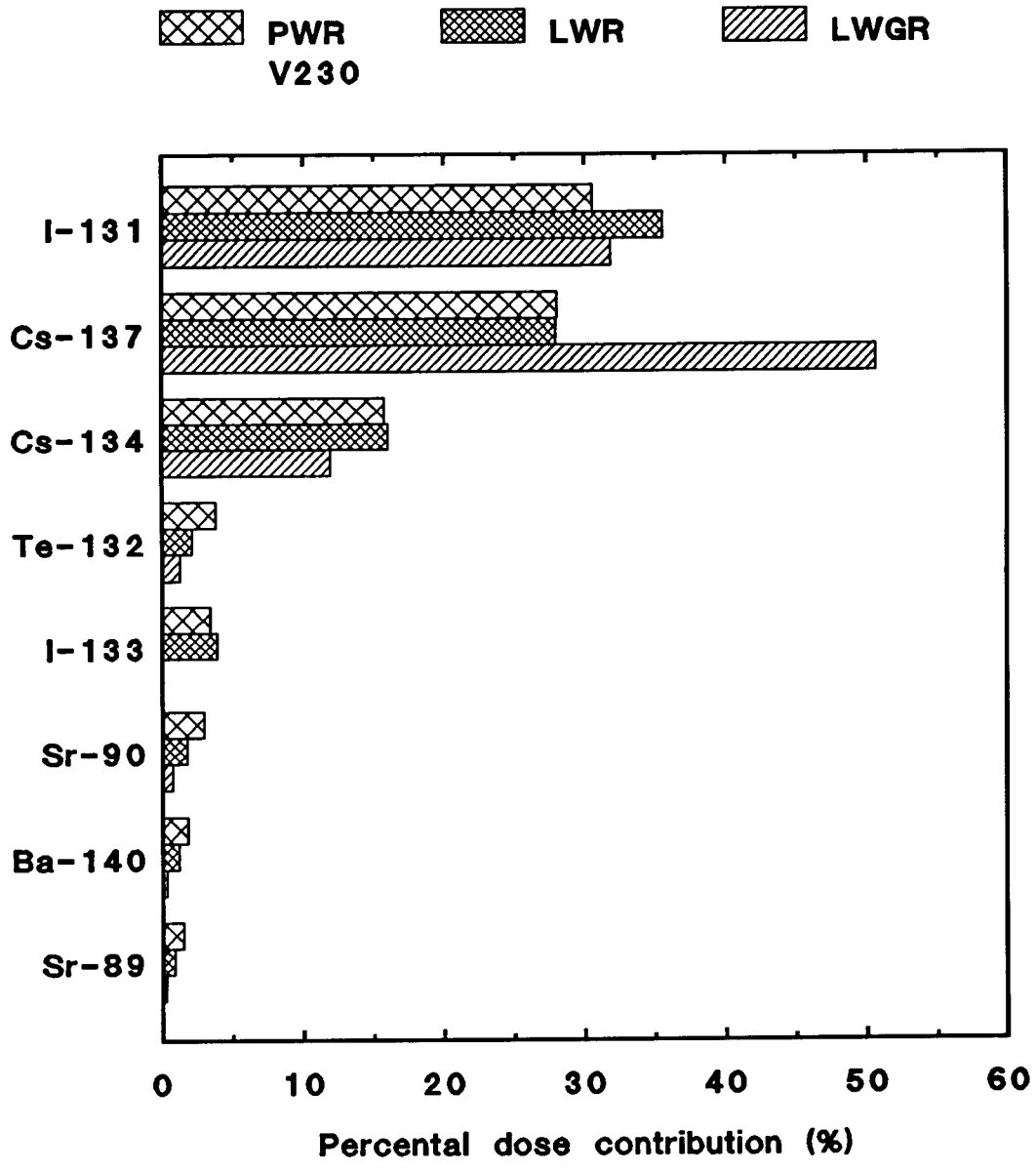


Figure 4.2 Major dose-contributing nuclides at 10 km from an accidental release

The results are provided for an integration period of 70 years: intake and external exposure have been integrated over a period of 70 years. It should be noted that nearly 70% of the total dose is received in the first year following the accident. Looking at the ingestion pathway it is found that direct interception is the major contributor to the dose: 86% of the ingestion dose is due to direct interception. Only approximately 4% is due to soil-plant transfer to crops, and 10% is due to soil-grass-milk/meat transfer (see table 4.4).

Table 4.2 Major five dose contributing nuclides for various probabilistic source terms.

nuclide	LWR-m source term % contribution	PWR-V230 source term % contribution	LWGR/AGR /GCR % contribution	FBR % contribution
I-131	35.6	30.4	31.9	13.9
Cs-137	28.0	28.1	50.7	50.7
Cs-134	16.1	15.8	12.0	24.9
I-133	3.9	3.4		1.4
Te-132	2.2	3.9	1.3	
Sr-90			0.8	
Pu-238				1.2
total contribution for major five	85.8	81.6	96.4	91.2

Table 4.3 Contribution of pathways to overall dose for various probabilistic source terms

exposure pathway	LWR-m source term % contribution	PWR-V230 source term % contribution	LWGR/AGR /GCR % contribution	FBR % contribution
ingestion	53.3	51.7	50.5	41.6
external ground	32.4	34.0	43.7	49.6
inhalation	10.9	12.1	5.8	8.5
external cloud	3.4	2.2	0.1	0.5

Table 4.4 Foodproducts contributing to ingestion dose (VROM, 1992), for LWR-m source term; 70 years dose, adults.

foodproduct	intake kg/day	interception %	soil-plant	total
vegetables	0.641	47%	3%	50%
cereals	0.299	2%	0.3%	2%
roots, tubers	0.342	21%	1%	22%
milk	0.803	12%	5%	17%
meat	0.140	4%	5%	9%

The above calculations were performed for adults, and give a good representation for the overall population, however since accidental releases occur over a short period in time, and doses are primarily received in the first year following the incident, it is relevant to consider the risks for a risk-group composed of small children at the time of the accident. Dose-conversion factors, intake of foods and breathing rate are different for children. We considered 1-year old children and calculated the first year dose for a LWR-source term.

Table 4.5 Major dose contributing nuclides and pathways for one year old children, first year total, for LWR-m source term

nuclide	average diet	extreme diet	adults, extreme diet
I-131	72%	74%	
I-133	7%	6%	
Te-132	4%	5%	
Cs-134	4%	3%	
Ba-140	2%	3%	
rest	11%	10%	
relative first year dose	1.5	2.2	1
ingestion	78%	85%	69%
external soil	7%	5%	11%
inhalation	11%	8%	16%
external cloud	3%	2%	5%

Calculations were made for two different scenarios for food-intake: a set of extremely high intakes and a set of normal average intakes (see appendix 6, table A6.7). Results for one-year-old children are shown in table 4.5. Children receive higher doses, compared with the first year dose of adults: 50% higher for average food-intake, and 120% higher for an extremely high food intake. The difference between the two intake scenarios reflects the importance of the ingestion pathway in small children: around 80% of the total first year dose.

4.4 Discussion

4.4.1 *Comparison of results with other modelling and dose-estimating efforts*

In this section we compare the results obtained with the presented model with various other modelling evaluations.
UNSCEAR (1988) reported on the estimated first year doses following the Chernobyl accident for a large number of locations in Europe, and made model prognostic estimates of time-integrated dosages. UNSCEAR also provided deposition and time integrated air-concentrations for Cs-137 for those locations.
Figure 4.3 provides the results of the comparison of the total dose obtained by UNSCEAR (on the horizontal axis), and the total dose calculated with the NucRed model (on the vertical axis; doses derived from relationship given in appendix 7). The NucRed model was applied for a mixed rural and urban population and various parameters regarding behaviour, shielding by buildings, and run off were chosen in line with choices made by UNSCEAR 1988. Furthermore the food intake for the reference man were used in the comparison (ICRP, 1974). The full straight line indicates exact resemblance of the two estimates, and the dotted line indicates the best fit through all data points. The conclusion can be that the estimates provided with the NucRed model are in close agreement with the UNSCEAR estimates.

The comparison with UNSCEAR (1988), as shown in figure 4.3, was based upon parameter choices regarding behaviour, shielding by buildings and food intake, which differ from the parameters for the risk group considered in this study: rural adults, eating fresh products, and a food consumption which is regarded at the high end of the consumption range. The calculations for the risk group result in approximately twofold higher doses.

Since ingestion was found to be the major exposure pathway, we have compared results obtained with the NucRed model, with results obtained by various other investigators. Results of this comparison are provided in table 4.6 in terms of nuclide intake per unit deposition for various nuclides. Focussing on the major nuclides: I-131 and Cs-137, we can see that uptake estimates obtained by Boone *et al.* (1981) are slightly less (15-20%) for Cs-137, and almost fourfold less for I-131. Boone *et al.* (1981) applied a more detailed ingestion model incorporating various delaytimes between harvesting, food production and consumption for the USA. Their model was presented as an extension of the NRC Regulatory guide 1.109 formulations, which are also provided in table 4.6. As can be seen, except for the Cs-isotopes the NRC-model derives much higher uptakes, and for I-131 the NRC predicts approximately fourfold higher uptakes as compared to the method used in this study. Comparing our calculations with the estimates provided in a report of the Commission of the European Communities (Sinnaeve and Gerber, 1991), we find that the uptakes are five-fold higher in the EC-report (EC-results shown are averaged values of January and

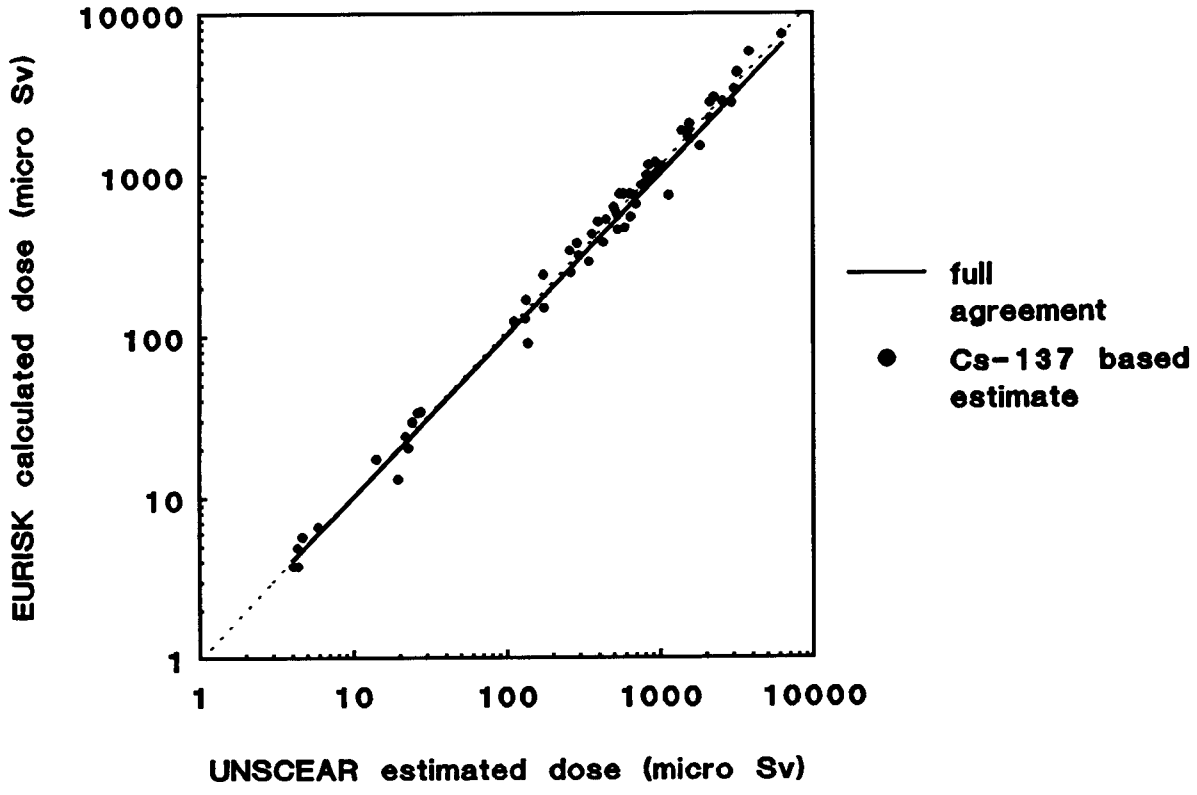


Figure 4.3 Comparison of the UNSCEAR (1988) dose estimate and the EURISK estimate based on Cs-137

July).

Table 4.6 Comparing intake of various nuclides per unit deposition according to various authors

nuclide	lifetime follow up			first year dose	
	this report Bq/(Bq · m^{-2})	Boone *et al.*, 1981 Bq · Bq^{-1} · m^2	NRC, 1977 reg.guide Bq · Bq^{-1} · m^2	this report Bq · Bq^{-1} · m^2	Sinnaeve and Gerber, 1991 Bq/(Bq · m^{-2})
I-131	0.32	0.088	1.23	0.32	1.6
Cs-137	3.43	2.70	2.44	1.28	7.5
Sr-90	2.86	0.47	1.27	1.08	1.5
Cs-134	1.35	2.16	2.23		
Ru-106	0.88	0.12	1.09		

The latter calculations do not include delays between harvesting and consumption, and thus can be regarded as overestimates. The variations due to differences in food-consumption patterns, and food processing in various regions within the EC are small compared to the other model uncertainties as shown in this comparison.

The conclusion can be that the modelling presented in this report is well within the range of results obtained in other studies. The effect of the variations found in the foodchain modelling are maximally a factor 4 in both directions. These differences can be attributed at least partly on the fact that some models allowed for full implementation of delay times, whereas other models assume immediate ingestion of harvested products. Furthermore differences in foodconsumption parameters attribute to variations in model outcomes. As an indicative estimate for the uncertainty range a factor of 2 up and down is in line with the above findings.

Regarding risk groups it is found that small children receive 1.5—2.2 fold higher doses in the first year than adults. The importance of children as a risk-group is further stressed by the fact that the estimated risk factors for children are higher than for adults (see next chapter).
It should be noted that child doses calculated are sensitive to delay times in the foodproduction and consumption chain, since ingestion of I-131 is the major exposure path, and the half-life time of I-131 is 8 days.
Results obtained are insensitive to soil-plant transfer factors, since overall contribution of this exposurepath is low. With the exception of a 10% contribution to the overall ingestion dose for the path grass-cow-meat/milk.

5 RISK ANALYSIS

5.1 Introduction

The previous chapters have covered the various aspects of the source-effect-chain relevant to the evaluation of risks from accidents with nuclear powerreactors. This chapter applies and integrates the results obtained in those chapters in order to derive a risk map of Europe.

5.2 Method for risk map calculation (RISKA-model)

The risk map calculation is performed in an independent separate model: the RISKA-model (Slaper, 1993b), which will be briefly described below. The results from the previous dispersion and exposure models are combined with the estimates of accidental releases, and a database for nuclear power reactors in Europe. The database contains data on plant locations, reactortypes, reactor status (in design, operational, closed etc.). The reactor database also includes an estimate of the probability of severe core damage per year of operation (appendix 1 provides the data for the operational reactors in the database). Data on 295 reactors are included in the database, but only 217 reactors which are indicated as operational are included in the presented calculations (Kernkraftwerke, 1992).
The risk map module calculates the risks for a matrix of receptor points in Europe. The coordinates of the receptor locations are given in degrees latitude and longitude, steps are also specified in a fraction of degrees. The area covered and the number of datapoints for which risks are calculated can be chosen freely, by specifying the starting latitude and longitude, the step sizes and number of steps in the longitude and latitude directions. The overall computation time for the default grid of 8000 receptor locations used in this study is about 30—40 minutes on a 50 MHz PC with a 80486 processor.
The computational sequence per receptor location is as follows:

- for each operational reactor:
  - the distance to the receptor location is calculated (applying the method described in appendix 8).
  - the geographical quadrant (north-east, north-west, south-west and south-east) is determined, looking from reactor to receptor location
  - the time-integrated air concentration and deposition are calculated, applying the descriptive functions provided in chapter 3 (see appendix 4 table A4.2 for parameter values) and multiplying the results with the accident probability and the probabilistic release for the particular reactor
  - (70 year follow-up) doses are calculated multiplying the air concentration and deposition, with the overall dose conversion factors for Cs-137 and I-131 and with the multiplicative scaling factors that account for contribution of other nuclides
  - dose estimates are summed for each pathway and both nuclide groups

- dose contributions for all nuclear reactors are summed, to obtain an overall dose for the specific receptor location
- multiplying the final result with a dose to death risk conversion factor provides the probabilistic death risk due to doses received in the seventy years following an accidental release, for the considered group of the population

Each receptor location can be seen as the centre of a grid cell, and the result for each of the receptor locations is considered to represent the average value within the grid cell. If power plants are close to the grid cell, or inside the grid cell, the calculation could lead to a value which is not representing the average risk within the whole cell. Therefore the model allows for averaging over 9 receptor points evenly distributed within the grid cell area. The criterium for averaging within one grid cell can be chosen freely, and is related to the diagonal distance between grid cells. For the calculations presented here, averaging is performed if the nearest power plant is less than one diagonal distance from the centre of the grid cell. Standard calculations were performed in steps of 0.5 degrees latitude and one degree longitude, and around the Netherlands (52 degrees N) the averaging occurs if the closest plant is within 88 km from the receptor point.
In mathematical terms the following equation covers the risk calculation:

$$R_j = \sum_{i=1}^{N_{cen}} \sum_{k}^{\text{air depo}} \sum_{n}^{\text{I Cs}} (p_i \cdot L_{in} \cdot (\frac{A_{kn} \cdot e^{-B_{kn} \cdot r_{ij}}}{(r_{ij})^{c_{kn}}}) \cdot (DC)_{kn} \cdot F_{kni} \cdot C_{risk} \qquad [5.1]$$

where,

R_j - overall probabilistic death risk for location j (death probability per year)

$\sum_i$ - summation over all nuclear power reactors i (N_{cen} number of reactors)

$\sum_k$ - summation over exposure contribution for two compartments: air concentration and ground deposition

$\sum_n$ - summation over exposure contribution from two groups of nuclides (iodine and caesium)

p_i - probability rate of severe core-damage for reactor i (per year)

L_{in} - conditional probabilistic release for nuclide n and nuclear reactor i

A_{kn} - dispersion parameter for compartment k and nuclide group n

B_{kn} - dispersion parameter for compartment k and nuclide group n

C_{kn} - dispersion parameter for compartment k and nuclide group n (dispersion parameters are obtained from table A4.2 (appendix 4))

r_{ij} - relative distance (over the globe) between reactor location i and location j ($r_{ij} = r'/r_0$ where the value of r' reflects the distance in km and r_0 is 1 km); the distance r' is calculated by means of the expression provided in appendix 8

$(DC)_{kn}$ - overall dose conversion factor for a unit contamination of compartment k with the dose dominating nuclide in group n (I-131 or Cs-137); ($Sv/(Bq \cdot m^{-2})$ for deposition, and $Sv/(Bq \cdot s \cdot m^{-3})$ for time integrated air concentration

F_{kni} - scaling factor for compartment k to account for the contribution of other nuclides in nuclide group n for the reactor type of reactor i.

C_{risk} - conversion factor from dose to death risk; a value of 2.5% per Sievert is applied in the present calculations.

The RISKA program calculates the risk in a user specified grid. For the risk map calculations the grid covered longitudes from -29.5 degrees Wester Length upto 70.5 degrees Easter Length, and latitudes from 33.25 degrees North upto 73.25 degrees North. Results for the 8181 grid cells are saved in a file, which was transferred to a Geographical Information System (GIS) to provide risk maps of Europe.

Risk groups.

Risk is estimated for the population with an average distribution over ages, and the calculated follow-up time over which exposure was estimated was 70 years. Intake of food products was assumed to be according to a high intake for the Netherlands, individuals are assumed to spent 30% of their time outdoors, and since runoff is neglected the calculation is primarily suited for rural populations. In the results section the relative effects for other risk groups are provided.

5.3 Risk map results

Applying the above described method risk maps of the European continent were produced for two situations:

- the situation in July 1992
- situation where eastern European reactors would be operational according to western safety standards (see appendix 1 for changes in input for RISKA-model)

The risk maps are presented in the figures 5.1 and 5.2. It should be noted that overall uncertainties in the calculations can amount to a factor of about 15 in western Europe (up as well as down; see also discussion in section 5.4 and appendix 2).

As can be seen in figure 5.1, the present situation leads to a considerable variation in risk over Europe: an excess death risk lower than 1 per 100 million per year is estimated for Iceland and southwestern parts of Portugal and Spain. Risk is higher towards eastern Europe to over 100 per 100 million per year in large areas of the former Soviet Union. Highest risk is found in the areas where LWGR reactors are found: the excess death risk is well over 1000 per 100 million per year in these areas. For the Netherlands an overall risk of 10 per 100 million per year is expected. The highest risk is calculated in the southwestern part of

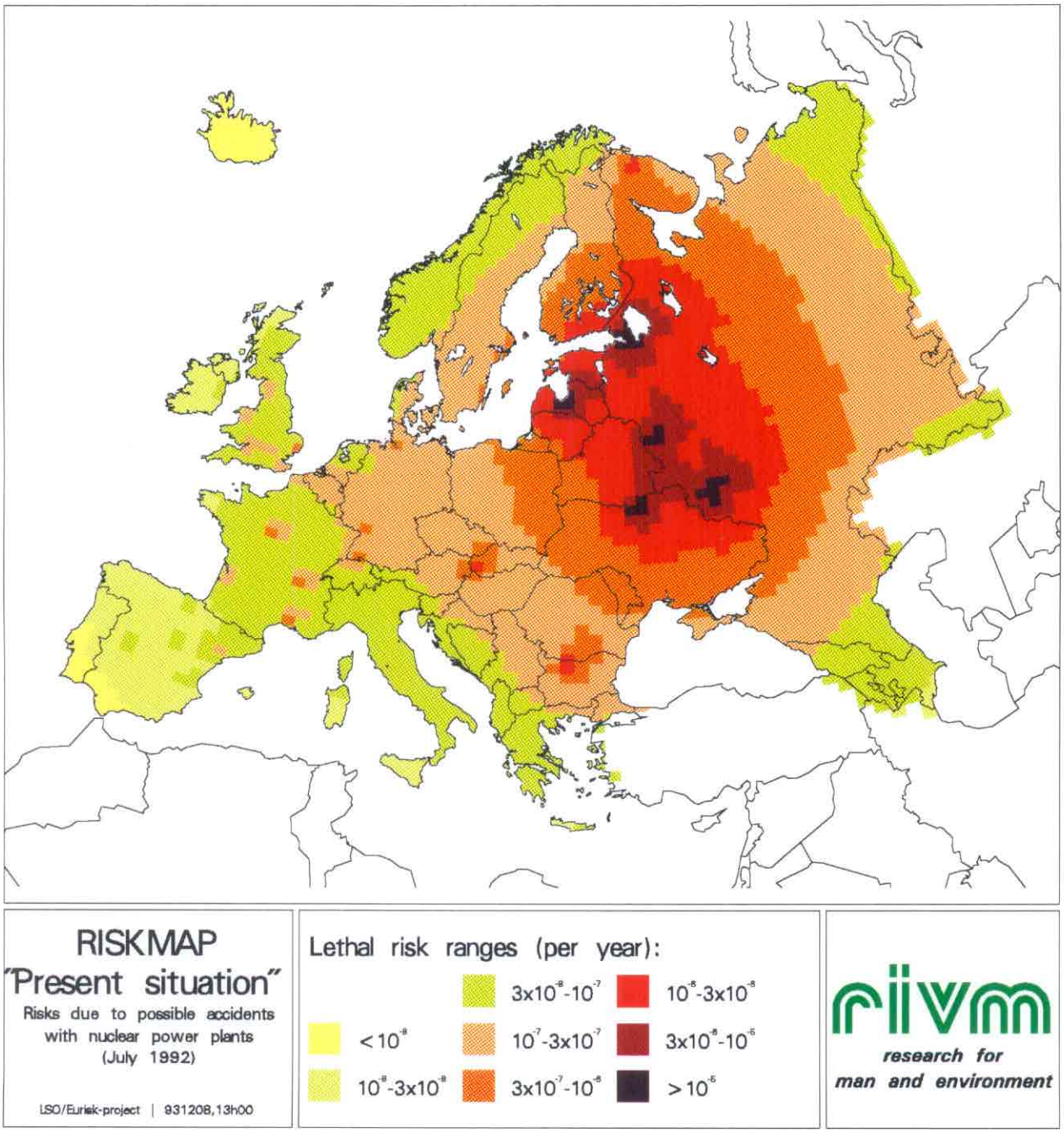


Figure 5.1 Estimated cancer mortality risk due to possible accidentally released nuclides from nuclear power plants in Europe

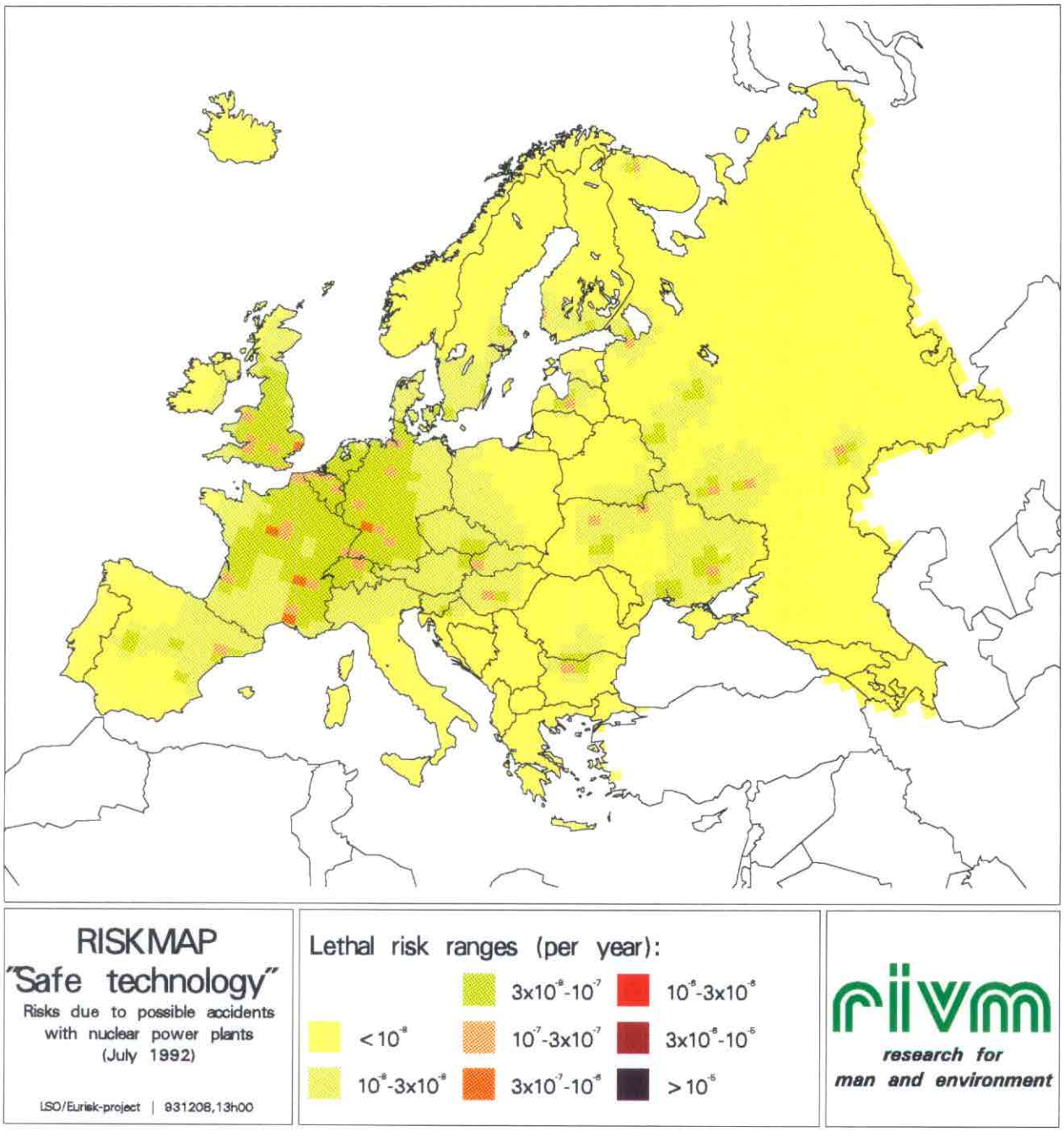


Figure 5.2 Estimated cancer mortality risk due to possible accidentally released nuclides from nuclear power plants in Europe, assuming that all Eastern European reactors involved have a safety level comparable to that of Western European reactors

the Netherlands: 30 per 100 million per year. This risk is primarily due to the power reactors at Doel in Belgium. It should be realized however, that lack of specific information from safety analyses for those reactors has led to the relatively high estimate of 10^{-4} per year for the core damage probability. Lack of knowledge could therefore be the cause of the elevated risk estimates.
Risk all over the European continent is considerably reduced if western safety standards are assumed for the eastern European nuclear reactors. This is shown in figure 5.2. Larger parts of Spain, the whole of Portugal and even large parts of the former Soviet Union then fall in the region where risk is below 1 per 100 million per year. In that case risk in the gridcells does not exceed 100 per 100 million per year over Europe. The risk level in the Netherlands is reduced by approximately 50% to 5 per 100 million per year. Highest risk areas in that case reflect areas with highest density of reactors: France, Belgium, the Netherlands and the western part of Germany.
Averaging the risk for large parts of middle European latitudes (48-56 degrees N) figure 5.3 shows the longitude dependence of risk for the two situations described above. It is clearly seen from these results that largest risk reductions occur in Eastern Europe: more than 100-fold, whereas lowest reductions of approximately 50% are found in western Europe, where the density of nuclear reactors is highest.

The risk calculations are influenced by the assumptions regarding the risk group, and the choices regarding the modelling approaches and parameter values. Table 5.1 provides a preliminary overview of possible changes in the overall calculated risks, when implementing other default settings or modelling approaches. Results provided in this table are based upon recalculations for the LWR-source term.
Implementing the new ICRP-(1991) risk factor of 5% per Sievert would increase all risk estimates by a factor of 2. Implementing the new ICRP (1991) dose conversion factors primarily influences the I-131 contribution, increasing inhalation and ingestion doses for I-131 by 70% (see appendix 9). The overall dose and risk are then increased by 20—30%. Thus completely accounting for the ICRP-60 (ICRP, 1991) increases overall risk with a factor of 2.5. It should be noted however that the increased dose conversion factor for I-131 is primarily caused by the new ICRP approach to give a certain weight to incidence, as compared to the previous approach, where only mortality is included in the weighting factors.

Table 5.1 Several parameters used and their influence on overall risk

risk group	diet	behaviour	risk multiplication factor
adults, rural	extreme NL adults	30% outdoor	1
adults, rural	Reference man (ICRP, 1974)	30% outdoor	0.8
adults, mixed	extreme NL adults	20% outdoor (UN-88)	0.8
adults, mixed	Reference man	30% outdoor	0.6
adults, rural	EC-report 12553	30% outdoor	3
adults, rural	Boone, 1981	30% outdoor	0.6-0.8
adults, rural	NRC-reg. guide 1.109	30% outdoor	1.3-2.5
adults, mixed	UNSCEAR, 1988	20% outdoor (UN-88)	0.5
children (1 year), rural	average child extreme adult	30% outdoor	3
children (1 year), rural	extreme child extreme adult	30% outdoor	4
Applying Risk factor 5% per Sievert			2
Applying new ICRP dose conversion factor for I			1.2-1.3
overall uncertainty			0.07-14

Apart from uncertainties in various aspects of the calculations, which will be discussed later (section 5.4), the outcome is also influenced by the definition of the risk groups under consideration. This includes: behaviour and food intake, as well as sensitivity of the population group (children). Table 5.1 provides some estimates for a number of assumptions regarding the risk group under consideration. Applying intake as specified by ICRP (1974) for the reference man reduces risk by 20%; applying behaviour, and runoff in urban areas as specified by UNSCEAR (1988) also reduces risk with 20%. Combining both UNSCEAR-behaviour and ICRP-intake reduces overall estimated risk for the population with 40% (risk multiplication factor 0.6; see table 5.1).

Since most of the dose is received in the first year following an accident (see chapter 4), small children can be considered a specific risk group. Apart from the fact that doses

received are higher due to higher dose conversion factors, also the risk factor is higher for children than for adults. Results from BEIR-V indicate a twofold higher risk per unit dose, and the ICRP (1991) provides estimates upto a nearly 3-fold higher risk. Combining these figures with the higher doses received by children leads to a preliminary estimate of a 3—4 fold higher risk for children, who are one year old at the time of the accident (see table 5.1).

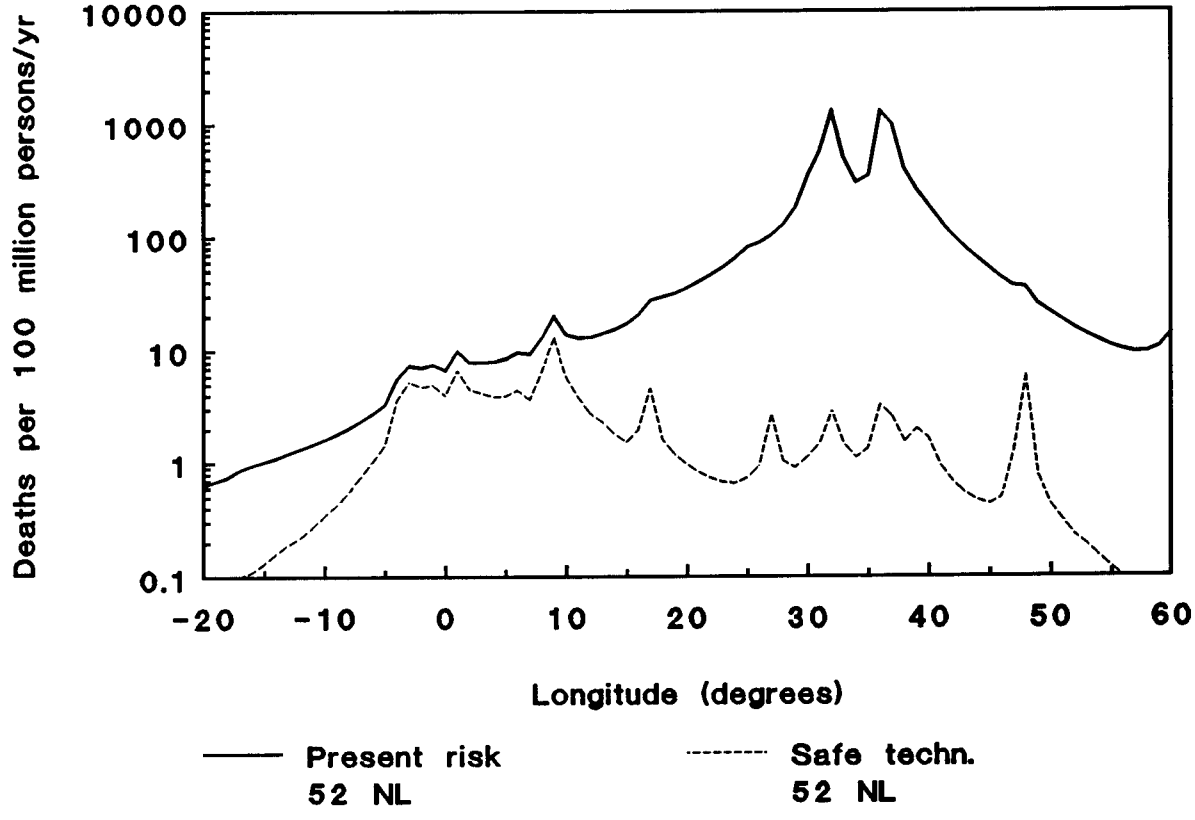


Figure 5.3 Longitude dependence of the risk for the two situations described in figures 5.1. and 5.2

5.4 Discussion

It is realized that the modelling effort presented in this study carries a lot of uncertainties. Dealing with these uncertainties is one of the major issues for further study. This not only holds for the attempts provided in this study, but also for other scientifically based evaluations and risk assessments. In order to aim at a comparison of risks among various environmental problems, the need for comparative handling of uncertainties is compelling. Uncertainties are found all along the chain from source to effect and risk. Basically one could distinguish between uncertainties due to the limitations of the modelling (model expressions, determination of parameters), and uncertainties due to natural variations, like for instance the weather conditions and variability of behaviour and sensitivity (in- and outdoor activity, ingestion pattern etc.). The first type of uncertainty needs to be estimated on the basis of scientific evaluations. The evaluation of the second type of uncertainty depends on the choices regarding risk groups and risk conditions: estimates could be based on average conditions, or on unfavourable conditions. The Dutch policy approach regarding limiting of environmental risks aims at risk limits for individuals: all people in the population need to be protected at a certain risk level (OMR, 1989). Therefore it does not suffice to estimate the overall average risk to the population. This implies the calculation of risks for specific risk groups. In order to keep the risk evaluations comparable, the definition of risk groups should be highly comparable. We have calculated risks for several groups in the general population. The baseline estimates are for rural population with a high intake of foodproducts, spending 30% of the day outdoors. Table 5.1 provides the multiplication factors to be applied if other groups need to be considered. Children are the highest risk group within the general population (3-4 fold higher risk). Risks are calculated assuming no countermeasures. It should be noted that in this evaluation we have not calculated risk at specific locations in the close vicinity of a particular power plant: only stochastic death risk is estimated.

We have provided best estimates for risk under average weather conditions. The overall estimate has a large uncertainty range. The main contributor to the overall uncertainty is the lack of knowledge on accident probabilities and sourceterms. A preliminary upper estimate for the overall uncertainty (95% level) amounts to a factor of about 15 in western Europe, and a factor of 20 in Eastern Europe (see appendix 2 for methods used). The upper estimate can be partially based upon the history regarding accidents. It is more difficult to provide a lower estimate for the risk interval, due to the fact that reactor safety is not static, but highly dynamic: ongoing improvements to security systems and operating procedures can contribute to lower risklevels than estimated in the present study. On the other hand the economic recession, shortage of supplies and regional conflicts in the eastern European countries could provide a further stress on safety features. The presented estimates do not account for drastic changes in the safety culture within the eastern European countries. At present over 33 reactors are under construction in Europe (Kernkraftwerke 1992). It should be noted that risks cannot be easily extrapolated to new reactors, due to the fact that improvements in the

design can influence accident probabilities considerably.

A large uncertainty range implies that the 'real' probabilistic risk could be considerably higher or lower than the best estimate. The relevant question is: what is the probability that a risklimit is exceeded. Assuming a lognormal distribution with an uncertainty range as specified above, we estimated the probability of exceeding two risk limits in relation to the 'best' estimate provided by the modelling (see table 5.2).

Table 5.2 Probability of exceeding risklimits, in relation to best estimates for risk, assuming a factor of 14 uncertainty (95% confidence limit) and a lognormal distribution

best estimate for deathrisk (per year)	probability of exceeding risklimit 10^{-6} per year in %	probability of exceeding risklimit 10^{-5} per year in %
1×10^{-7}	4.3	<0.1
2×10^{-7}	11.5	0.2
3×10^{-7}	18.4	0.4
4×10^{-7}	24.7	0.8
5×10^{-7}	30.2	1.3
6×10^{-7}	35.2	1.8
7×10^{-7}	39.5	2.3
8×10^{-7}	43.4	2.9
9×10^{-7}	46.8	3.6
1×10^{-6}	50.0	4.3

In the Netherlands the risk was estimated at approximately 1×10^{-7} per year. From the results in table 5.2 we can conclude that the probability that in the Netherlands the limit of 1×10^{-6} per year is exceeded, is around 4%. For small children the 3—4 fold higher risk leads to a probability of around 20% that the 1×10^{-6} per year is exceeded. Further analysis of the uncertainties is needed.

6 DISCUSSION AND CONCLUSIONS

One of the aims of environmental policies is the reduction of risks of environmental pollution. Human activities can lead to regular and accidental releases of pollutants. Risk oriented policy approaches require risk assessment methodologies. This study provides a method for an integrated source-risk evaluation of accidental releases to the atmosphere from nuclear power reactors in Europe. The method is applied to obtain a probabilistic estimate of the location dependent mortality risks over the European continent, related to the combined use of all presently operational nuclear power plants for commercial electricity production. Mortality risks are restricted to radiation induced cancer deaths, related to possible accidental releases.
The provided method is based upon an evaluation of the source-risk chain, taking into account: accident probabilities and release scenarios for the European nuclear power reactors, atmospheric dispersion under statistically averaged weather conditions, radiation exposure caused by air and ground contamination, and death risks due to the exposures received in a 70 year follow up period.

Detailed safety studies for many of the European nuclear power reactors are lacking, thus we have to base our estimates on a generalised categorisation of reactors and accident probabilities (taken from Eendebak *et al.* 1992). The estimated probability of severe accidents to the reactor core, averaged over all reactors, agrees well with the probability as calculated from the worldwide history of accidents with nuclear power reactors: 2×10^{-4} per year in the present evaluation, as compared to the 3.3×10^{-4} per year obtained from two severe accidents in 6000 reactor years (Three Miles Island and Chernobyl). For each reactor category 2-4 release scenarios were considered, and releases were estimated for 54 radio nuclides (see chapter 2).
Air dispersion and deposition, under statistically averaged weather conditions are calculated for the European continent. Dispersion and deposition calculations are based upon an atmospheric dispersion model for continuous releases (see chapter 3). This approach is valid when risks are linearly related to the contamination, which is generally assumed to be the case in radiation induced risk evaluations. This implies that the method is not valid for the estimation of victims of death due to very high radiation doses that can be received in the direct vicinity of a reactor.

The exposure model applied in this study includes the major exposure pathways: ingestion, inhalation and external exposure. The dominant exposure pathways for all (probabilistic) source terms used in this study are ingestion (42-53% of total dose) and external exposure from deposited radio-nuclides (32-50%). Primary dose contributing nuclides in the various source terms are I-131, Cs-137 and Cs-134, together contributing 74-95% of the total dose. Dispersion of Cs-137 and Cs-134 is fully comparable. Thus, it is concluded that the dispersion calculations can be restricted to I-131 en Cs-137, and that the contribution of all

the other nuclides is accounted for by means of multiplication factors (see chapter 4 for details).
A comparison of the exposure model presented in this report with other dose estimates shows that results obtained are within the range of estimates obtained for other modelling attempts.

We calculate risks for a reference group consisting of adult individuals in a rural environment with a high intake of fresh food products. A death risk factor of 2.5% per Sievert is assumed. For the above given risk group we obtain a wide range of risks, related to the location. The mortality risk due to accidents of European nuclear power plants is estimated to be around 10×10^{-8} per year in western Europe. In central Europe a large increase in the estimated risk of about 30×10^{-8} in Poland to over 1000×10^{-8} per year in Russia is observed. Approximately 50% of the mortality risk in western Europe has to be attributed to possible accidents of eastern European reactors. Location dependent risks are presented as risk maps of Europe in chapter 5.
Additional calculations were made for the situation where all eastern European reactors are assumed to have the quality of safety measures presently found in Western European reactors. In this case the average risk in western Europe is reduced by nearly a factor of 2 and in eastern Europe by more than a factor of 100. Thus, reactor safety improvement can reduce risks considerably.

The calculations of overall risk show a large degree of uncertainty. The main contributor to the overall uncertainty is the lack of knowledge on accident probabilities and source terms, especially for the eastern European power reactors. Estimates of accident probabilities based upon the operational and accidental history of nuclear power reactors provide general agreement with the release probabilities applied in this study, and could support an indicative uncertainty range. Including the uncertainties of dispersion and exposure estimates, a preliminary estimate of the overall uncertainty of the risks is obtained: a factor of 15 in Western Europe, and a factor of 20-25 in Eastern Europe.

It should be noted that reactor safety is not a static situation, because ongoing improvements to security systems and operating procedures can contribute to lower risk levels. On the other hand, the economic recession, shortage of supplies and regional conflicts in the Eastern European countries could provide a further stress on safety features.

Other factors that can influence the calculated results are: changes in the concepts of exposure-risk relationship, and changes in the population at risk. We provided indications for the relative change in the above risk estimates associated with an implementation of the ICRP (1991) concepts, and for other risk groups.
Implementing recent tissue weighting factors and a dose-risk factor of 5%, in line with ICRP (1991), the risk estimates are increased by a factor of 2.5. An indication of risks for other risk groups shows, that for a combined rural and urban population with average food intake

risks are lowered by a factor of 0.5. For small children at the age of an accidental release the risks are a factor of 3-4 higher.

One of the aims in risk oriented policy approaches is to achieve a common risk oriented basis to weight the relative importance of various environmental issues. Such a common basis can only be achieved if the methodology of risk analysis and the risk groups considered are comparable for various issues. The presented analysis is in close agreement with the modelling and parameter choices proposed in the MORIS-project (Blaauboer *et al.*, 1992). It was found that many of the models developed for continuous releases, could also be applied and translated to the situation of large accidental releases with a low probability of occurrence. The approach is only valid if the contamination is linearly related to the effects considered. Since this is generally assumed to be the case for cancer induction following exposure to ionizing radiation, but not for the occurrence of short term deaths, the method is restricted to the prior case.

The results of this study could be improved in several ways. A reduction of uncertainties could be most effectively achieved by an improved safety analysis of various reactor types, and especially of the eastern European reactors. Furthermore regional estimates could be improved incorporating average regional weather conditions, topography and soil characteristics and land use. A combination with demographic data could provide insight in the population averaged risks involved.
The method developed in this study can be used for risk estimations with other accidental or regular emissions to the atmosphere for a large number of sources. A limitation to the applicability is the fact that only linear relationships between deposition and effect can be considered. The limitation is due to the fact that the air dispersion model used in this study does not allow for an estimation of variations in time-averaged air concentrations and deposition rates.

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Appendix 1 Nuclear power reactors and probabilistic source terms

Table A1.1 Nuclear power reactors (types, locations, accident probability; see at end of table for explanations)

Country	Name	Type	Power	Year	N	E	Probabil.
Belgium	Doel-1	pwr	410	74-08	51,19	4,16	0.0001
	Doel-2	pwr	410	75-08	51,19	4,16	0.0001
	Doel-3	pwr	936	82-06	51,19	4,16	0.0001
	Doel-4	pwr	1065	85-04	51,19	4,16	0.0001
	Tihange-1	pwr	920	75-03	50,32	5,15	0.0001
	Tihange-2	pwr	941	82-10	50,32	5,15	0.0001
	Tihange-3	pwr	1070	85-06	50,32	5,15	0.0001
Germany	Biblis-A	pwr	1204	74-08	49,41	8,27	0.00001
	Biblis-B	pwr	1300	76-04	49,41	8,27	0.00001
	Brokdorf	pwr	1383	86-10	53,52	9,20	0.0001
	Brunsbuettel	bwr	806	76-07	53,54	9,08	0.0001
	Emsland	pwr	1341	88-05	52,32	7,19	0.000001
	Grafenrheinfeld	pwr	1300	81-12	50	10,12	0.0001
	Grohnde	pwr	1395	84-09	52,01	9,25	0.0001
	Gundremmingen-B	bwr	1300	84-03	48,33	10,23	0.0001
	Gundremmingen-C	bwr	1308	84-11	48,33	10,23	0.0001
	Isar-I	bwr	907	77-12	48,37	12,28	0.0001
	Isar-II	pwr	1369	88-01	48,37	12,28	0.000001
	Kruemmel	bwr	1316	83-09	49,55	7	0.0001
	Muelheim-Kaerlich	pwr	1308	86-03	50,26	7,24	0.0001
	Neckarwestheim-1	pwr	840	76-07	49,11	9,14	0.0001
	Neckarwestheim-2	pwr	1316	89-01	49,11	9,14	0.000001
	Obrigheim	pwr	357	68-10	49,28	9	0.0001
	Philippsburg-1	bwr	900	79-05	49,15	8,28	0.0001
	Philippsburg-2	pwr	1349	84-12	49,15	8,28	0.0001
	Stade-1	pwr	672	72-01	53,36	9,28	0.0001
	Unterweser	pwr	1300	78-09	53,32	8,14	0.0001
	Wuergassen	bwr	670	71-12	51,18	9,30	0.0001
Bulgaria	Kozloduy-1	pwr-v230	440	74-07	43,25	24,40	0.001
	Kozloduy-2	pwr-v230	440	75-10	43,25	24,40	0.001
	Kozloduy-3	pwr-v230	440	80-12	43,25	24,40	0.001
	Kozloduy-4	pwr-v230	440	82-05	43,25	24,40	0.001
	Kozloduy-5	pwr-v1000	1000	87-11	43,25	24,40	0.0001
	Kozloduy-6	pwr-v1000	1000	89-03	43,25	24,40	0.0001
France	Belleville-1	pwr	1363	87-10	47,54	1,54	0.00001
	Belleville-2	pwr	1363	88-04	47,54	1,54	0.00001
	Blayais-1	pwr	951	81-06	44,50	-0,34	0.0001
	Blayais-2	pwr	951	82-07	44,50	-0,34	0.0001
	Blayais-3	pwr	951	83-08	44,50	-0,34	0.0001
	Blayais-4	pwr	951	83-05	44,50	-0,34	0.0001
	Bugey-1	gcr	555	72-04	45,46	4,50	0.0001
	Bugey-2	pwr	955	78-05	45,46	4,50	0.0001
	Bugey-3	pwr	955	78-09	45,46	4,50	0.0001

Country	Name	Type	Power	Year	N	E	Probabil.
France	Bugey-4	pwr	937	79-03	45,46	4,50	0.0001
	Bugey-5	pwr	937	79-07	45,46	4,50	0.0001
	Cattenom-1	pwr	1362	86-11	49,24	6,15	0.00001
	Cattenom-2	pwr	1362	87-09	49,24	6,15	0.00001
	Cattenom-3	pwr	1362	89-04	49,24	6,15	0.00001
	Cattenom-4	pwr	1362	91-02	49,24	6,15	0.00001
	Chinon-B1	pwr	919	82-11	47,10	0,15	0.00001
	Chinon-B2	pwr	919	83-11	47,10	0,15	0.00001
	Chinon-B3	pwr	954	86-10	47,10	0,15	0.00001
	Chinon-B4	pwr	954	87-11	47,10	0,15	0.00001
	Cruas-1	pwr	921	83-04	44,40	4,46	0.00001
	Cruas-2	pwr	956	84-09	44,40	4,46	0.00001
	Cruas-3	pwr	955	84-05	44,40	4,46	0.00001
	Cruas-4	pwr	921	84-10	44,40	4,46	0.00001
	Dampierre-1	pwr	937	80-03	47,54	1,54	0.0001
	Dampierre-2	pwr	937	80-12	47,54	1,54	0.0001
	Dampierre-3	pwr	937	81-01	47,54	1,54	0.0001
	Dampierre-4	pwr	937	81-08	47,54	1,54	0.0001
	Fessenheim-1	pwr	920	77-04	47,56	7,33	0.0001
	Fessenheim-2	pwr	920	77-10	47,56	7,33	0.0001
	Flamanville-1	pwr	1382	85-12	49,32	-1,53	0.00001
	Flamanville-2	pwr	1382	86-07	49,32	-1,53	0.00001
	Golfech-1	pwr	1365	90-05	43,37	1,26	0.00001
	Gravelines-B1	pwr	951	80-03	50,59	2,08	0.0001
	Gravelines-B2	pwr	951	80-08	50,59	2,08	0.0001
	Gravelines-B3	pwr	951	80-12	50,59	2,08	0.0001
	Gravelines-B4	pwr	951	81-06	50,59	2,08	0.0001
	Gravelines-C5	pwr	951	84-08	50,59	2,08	0.0001
	Gravelines-C6	pwr	951	85-08	50,59	2,08	0.0001
	Nogent-1	pwr	1365	87-10	48,30	3,31	0.00001
	Nogent-2	pwr	1365	88-11	48,30	3,31	0.00001
	Paluel-1	pwr	1382	84-06	49,30	0,06	0.00001
	Paluel-2	pwr	1382	84-09	49,30	0,06	0.00001
	Paluel-3	pwr	1382	85-09	49,30	0,06	0.00001
	Paluel-4	pwr	1382	86-04	49,30	0,06	0.00001
	Phenix	fbr	250	73-12	43,56	4,48	0.0001
	Penly-1	pwr	1382	90-05	49,55	1,05	0.00001
	Penly-2	pwr	1382	92	49,55	1,05	0.00001
	St. Alban-1	pwr	1381	85-08	45,46	4,50	0.00001
	St. Alban-2	pwr	1381	86-07	45,46	4,50	0.00001
	St. Laurent-A2	gcr	465	71-08	47,43	1,36	0.0001
	St. Laurent-B1	pwr	956	81-01	47,43	1,36	0.00001
	St. Laurent-B2	pwr	956	81-06	47,43	1,36	0.00001
	Superphenix	fbr	1242	86-01	46,35	-0,20	0.0001
	Tricastin-1	pwr	955	80-05	43,56	4,48	0.0001
	Tricastin-2	pwr	955	80-08	43,56	4,48	0.0001

Country	Name	Type	Power	Year	N	E	Probabil.
France	Tricastin-3	pwr	955	81-02	43,56	4,48	0.0001
	Tricastin-4	pwr	955	81-06	43,56	4,48	0.0001
Finland	Loviisa-1	pwr-v213	465	77-02	60,27	26,15	0.0001
	Loviisa-2	pwr-v213	465	80-11	60,27	26,15	0.0001
	Tvo-1	bwr	735	78-09	61,09	21,30	0.0001
	Tvo-2	bwr	735	80-02	61,09	21,30	0.0001
Hungary	Paks-1	pwr-v213	440	82-12	46,38	18,51	0.0001
	Paks-2	pwr-v213	440	84-09	46,38	18,51	0.0001
	Paks-3	pwr-v213	440	86-09	46,38	18,51	0.0001
	Paks-4	pwr-v213	440	87-08	46,38	18,51	0.0001
Netherlands	Borssele	pwr	480	73-07	51,25	3,45	0.00001
	Dodewaard	bwr	59	68-10	51,54	5,39	0.00001
Spain	Almaraz-1	pwr	930	81-05	39,50	-5,40	0.0001
	Almaraz-2	pwr	930	83-10	39,50	-5,40	0.0001
	Asco-1	pwr	930	83-08	41,10	0,34	0.0001
	Asco-2	pwr	930	85-10	41,10	0,34	0.0001
	Cofrentes	bwr	990	84-10	39,14	-1,04	0.0001
	Zorita	pwr	160	68-07	40,25	-3,43	0.0001
	S.M. de Garona	bwr	460	71-03	42,51	-2,40	0.0001
	Vandellos-2	pwr	982	87-12	41,01	0,49	0.0001
	Trillo-1	pwr	1041	88	40,42	-2,35	0.0001
Chechosl.	Bohunice-1	pwr-230	440	78-12	48,10	17,10	0.001
former	Bohunice-2	pwr-230	440	80-03	48,10	17,10	0.001
	Bohunice-3	pwr-v213	440	84-08	48,10	17,10	0.0001
	Bohunice-4	pwr-v213	440	85-08	48,10	17,10	0.0001
	Dukovany-1	pwr-v213	440	85-02	49,12	16,40	0.0001
	Dukovany-2	pwr-v213	440	86-01	49,12	16,40	0.0001
	Dukovany-3	pwr-v213	440	86-11	49,12	16,40	0.0001
	Dukovany-4	pwr-v213	440	87-06	49,12	16,40	0.0001
U. K.	Bradwell-1	gcr	166	62-07	51,44	-0,54	0.0001
	Bradwell-2	gcr	166	62-11	51,44	-0,54	0.0001
	Calder Hall-1	gcr	55	56-08	54,38	-3,30	0.0001
	Calder Hall-2	gcr	55	57-02	54,38	-3,30	0.0001
	Calder Hall-3	gcr	55	59-03	54,38	-3,30	0.0001
	Calder Hall-4	gcr	55	59-04	54,38	-3,30	0.0001
	Chapelcross-1	gcr	55	59-02	54,54	-2,55	0.0001
	Chapelcross-2	gcr	55	59-05	54,54	-2,55	0.0001
	Chapelcross-3	gcr	55	59-11	54,54	-2,55	0.0001
	Chapelcross-4	gcr	55	60-01	54,54	-2,55	0.0001
	Dounreay	fbr	270	75-02	58,40	-3,28	0.0001
	Dungeness A-1	gcr	285	65-09	50,55	0,58	0.0001
	Dungeness A-2	gcr	285	65-11	50,55	0,58	0.0001
	Dungeness B-1	agr	660	83-04	50,55	0,58	0.0001
	Dungeness B-2	agr	660	85-12	50,55	0,58	0.0001
	Hartlepool-1	agr	660	83-08	54,42	-1,11	0.0001
	Hartlepool-2	agr	660	84-10	54,42	-1,11	0.0001

Country	Name	Type	Power	Year	N	E	Probabil.
U.K.	Heysham I-1	agr	660	83-07	54,02	-2,54	0.0001
	Heysham I-2	agr	660	84-10	54,02	-2,54	0.0001
	Heysham II-1	agr	660	88-07	54,02	-2,54	0.0001
	Heysham II-2	agr	660	88-11	54,02	-2,54	0.0001
	Hinkley Point A-1	gcr	270	65-02	51,27	-2,35	0.0001
	Hinkley Point A-2	gcr	270	65-03	51,27	-2,35	0.0001
	Hinkley Point B-1	agr	660	74-04	51,27	-2,35	0.0001
	Hinkley Point B-2	agr	660	76-04	51,27	-2,35	0.0001
	Hunterston B-1	agr	660	76-02	55,53	-4,15	0.0001
	Hunterston B-2	agr	660	77-03	55,53	-4,15	0.0001
	Oldbury-1	gcr	313	67-11	51,27	-2,35	0.0001
	Oldbury-2	gcr	313	68-04	51,27	-2,35	0.0001
	Sizewell A-1	gcr	325	65-06	52,04	1,10	0.0001
	Sizewell A-2	gcr	325	65-12	52,04	1,10	0.0001
	Trawsfynydd-A	gcr	270	65-01	52,54	-3,55	0.0001
	Trawsfynydd-B	gcr	270	65-02	52,54	-3,55	0.0001
	Torness Point-1	agr	701	88-05	57,19	-4,21	0.0001
	Torness Point-2	agr	701	88-12	57,19	-4,21	0.0001
	Wylfa-1	gcr	670	71-01	53,19	-4,38	0.0001
	Wylfa-2	gcr	670	71-07	53,19	-4,38	0.0001
U.S.S.R.	Balakovo-1	pwr-v1000	1000	85-12	52,04	47,46	0.0001
former	Balakovo-2	pwr-v1000	1000	87-10	52,04	47,46	0.0001
	Balakovo-3	pwr-v1000	1000	88-12	52,04	47,46	0.0001
	BN-600	fbr	600	80-04	56,47	61,28	0.0001
	Chernobyl-1	lwgr	1000	77-09	51,16	30,15	0.001
	Chernobyl-2	lwgr	1000	78-12	51,16	30,15	0.001
	Chernobyl-3	lwgr	1000	81-11	51,16	30,15	0.001
	Ignalino-1	lwgr	1500	83-12	56,53	24,08	0.001
	Ignalino-2	lwgr	1500	87-08	56,53	24,08	0.001
	Kalinin-1	pwr-v1000	1000	84-05	56,49	35,57	0.0001
	Kalinin-2	pwr-v1000	1000	86-12	56,49	35,57	0.0001
	Khmelnitski-1	pwr-v1000	1000	87-12	49,25	26,59	0.0001
	Kola-1	pwr-v230	440	73-06	68,53	33,01	0.001
	Kola-2	pwr-v230	440	74-12	68,53	33,01	0.001
	Kola-3	pwr-v213	440	81-03	68,53	33,01	0.0001
	Kola-4	pwr-v213	440	84-10	68,53	33,01	0.0001
	Kursk-1	lwgr	1000	76-12	51,45	36,14	0.001
	Kursk-2	lwgr	1000	79-01	51,45	36,14	0.001
	Kursk-3	lwgr	1000	83-10	51,45	36,14	0.001
	Kursk-4	lwgr	1000	85-12	51,45	36,14	0.001
	Leningrad-1	lwgr	1000	73-12	59,55	30,25	0.001
	Leningrad-2	lwgr	1000	75-07	59,55	30,25	0.001
	Leningrad-3	lwgr	1000	79-12	59,55	30,25	0.001
	Leningrad-4	lwgr	1000	81-02	59,55	30,25	0.001
	Nikolaev-1	pwr-v1000	1000	82-12	46,57	32	0.0001
	Nikolaev-2	pwr-v1000	1000	85-01	46,57	32	0.0001

Country	Name	Type	Power	Year	N	E	Probabil.
U.S.S.R.	Nikolaev-3	pwr-v1000	1000	89-09	46,57	32	0.0001
former	Novo Voronezh-3	pwr-v230	417	71-12	51,40	39,13	0.001
	Novo Voronezh-4	pwr-v230	417	72-12	51,40	39,13	0.001
	Novo Voronezh-5	pwr-v1000	1000	80-05	51,40	39,13	0.0001
	Rovno-1	pwr-v213	402	80-12	50,39	26,10	0.0001
	Rovno-2	pwr-v213	416	81-12	50,39	26,10	0.0001
	Rovno-3	pwr-v1000	1000	86-12	50,39	26,10	0.0001
	Smolensk-1	lwgr	1000	82-12	54,49	32,04	0.001
	Smolensk-2	lwgr	1000	85-05	54,49	32,04	0.001
	Smolensk-3	lwgr	1000	90-01	54,49	32,04	0.001
	Troitsk-4	lwgr	100	60	54,10	61,35	0.001
	Troitsk-5	lwgr	100	60	54,10	61,35	0.001
	Troitsk-6	lwgr	100	60	54,10	61,35	0.001
	Zaporozhe-1	pwr-v1000	1000	84-12	47,50	35,10	0.0001
	Zaporozhe-2	pwr-v1000	1000	85-07	47,50	35,10	0.0001
	Zaporozhe-3	pwr-v1000	1000	87-12	47,50	35,10	0.0001
	Zaporozhe-4	pwr-v1000	1000	88-12	47,50	35,10	0.0001
	Zaporozhe-5	pwr-v1000	1000	89-08	47,50	35,10	0.0001
Yugoslavia former	Krsko	pwr	664	81-09	45,58	15,30	0.0001
Sweden	Barsebaeck-1	bwr	615	75-05	55,47	12,58	0.0001
	Barsebaeck-2	bwr	590	77-03	55,47	12,58	0.0001
	Forsmark-1	bwr	1005	80-06	60,22	18,10	0.0001
	Forsmark-2	bwr	1006	81-01	60,22	18,10	0.0001
	Forsmark-3	bwr	1192	85-03	60,22	18,10	0.00001
	Oskarshamn-1	bwr	462	71-08	57,16	16,25	0.0001
	Oskarshamn-2	bwr	630	74-10	57,16	16,25	0.0001
	Oskarshamn-3	bwr	1205	85-03	57,16	16,25	0.00001
	Ringhals-1	bwr	780	74-10	57,45	12	0.00001
	Ringhals-2	pwr	840	74-08	57,45	12	0.00001
	Ringhals-3	pwr	960	82-06	57,45	12	0.00001
	Ringhals-4	pwr	960	82-06	57,45	12	0.00001
Switzerland	Beznau-1	pwr	364	69-07	47,23	8,33	0.0001
	Beznau-2	pwr	364	71-10	47,23	8,33	0.0001
	Goesgen	pwr	970	79-02	47,23	8	0.0001
	Leibstadt	bwr	1045	84-05	47,36	8,11	0.0001
	Muehleberg	bwr	336	71-06	46,57	7,26	0.0001

Power: power in MW_e

Year: Date of criticality (year and months)

N: Location, latitude north in degrees, minutes

E: Location, longitude east (of Greenwich), in degrees, minutes. ("-" means west of Greenwich)

Probabil. Accident probability rate for severe damage to the reactor core (per year of operation)

Table A1.2 Reactor inventories, based on thermal power of 3000 MW_{th}

Nuclide	Group	LWR-m [Bq]	Graphite reactor [Bq]	FBR [Bq]
Am-241	La	7.22e+13		9.32e+15
Ba-140	Ba	5.56e+18	5.00e+18	5.28e+18
Ce-141	Ce	5.39e+18	4.93e+18	5.46e+18
Ce-143	Ce	4.77e+18		4.45e+18
Ce-144	Ce	3.05e+18	3.09e+18	3.53e+18
Cm-242	La	2.26e+16	2.43e+16	1.09e+18
Cm-244	La	1.09e+15		7.86e+16
Co-58	Ru	3.67e+16		
Co-60	Ru	2.46e+16		
Cs-134	Cs	2.89e+17	1.35e+17	9.14e+17
Cs-136	Cs	1.12e+17	8.67e+16	4.15e+17
Cs-137	Cs	1.75e+17	1.98e+17	6.44e+17
I-131	I	3.05e+18	2.66e+18	3.86e+18
I-132	I	4.41e+18		5.19e+18
I-133	I	6.01e+18		6.59e+18
I-134	I	6.90e+18		7.03e+18
I-135	I	5.39e+18		6.35e+18
Kr-85	Xe	2.06e+16	2.46e+16	3.77e+16
Kr-85m	Xe	8.32e+17		6.02e+17
Kr-87	Xe	1.63e+18		1.01e+18
Kr-88	Xe	2.36e+18		1.39e+18
La-140	La	5.74e+18	5.24e+18	5.49e+18
Mo-99	Ru	5.65e+18	5.03e+18	6.20e+18
Nb-95	La	5.15e+18	4.75e+18	4.72e+18
Nd-147	La	2.16e+18		2.34e+18
Np-239	Ce	6.13e+19	3.44e+19	9.08e+19
Pr-143	La	4.65e+18		4.33e+18
Pu-238	Ce	2.27e+15	1.01e+15	5.01e+16
Pu-239	Ce	8.23e+14	8.33e+14	8.40e+15
Pu-240	Ce	8.23e+14	1.18e+15	1.49e+16
Pu-241	Ce	1.38e+17	1.32e+17	1.14e+18
Rb-86	Cs	9.00e+14		6.82e+16
Rh-105	Ru	1.84e+18		5.46e+18
Ru-103	Ru	3.91e+18	3.99e+18	7.33e+18
Ru-105	Ru	2.70e+18		5.49e+18
Ru-106	Ru	9.65e+17	1.26e+18	4.01e+18
Sb-127	Te	2.24e+17		5.64e+17
Sb-129	Te	1.20e+18		1.38e+18
Sr-89	Sr	3.23e+18	3.12e+18	1.89e+18
Sr-90	Sr	1.37e+17	1.46e+17	2.35e+17
Sr-91	Sr	3.97e+18		2.58e+18
Tc-99m	Ru	4.91e+18		5.43e+18
Te-127	Te	2.15e+17		5.61e+17
Te-127m	Te	4.11e+16		7.54e+16
Te-129	Te	1.13e+18		1.41e+18
Te-129m	Te	1.89e+17		2.04e+17
Te-131m	Te	4.56e+17		5.79e+17
Te-132	Te	4.26e+18	3.14e+18	5.10e+18
Xe-133	Xe	5.86e+18	5.97e+18	6.88e+18
Xe-135	Xe	1.23e+18		7.27e+18
Y-90	La	1.47e+17		2.64e+17
Y-91	La	4.06e+18		2.59e+18
Zr-95	La	5.30e+18	5.00e+18	4.78e+18
Zr-97	La	5.24e+18		5.22e+18

Table A1.3 Probabilistic conditional release L_n (Bq) based upon equation 2.1, deduced from Eendebak *et al.* (1992) and used in this assessment. (LWR-m: all other light water reactors with "middle of cycle" inventory)

	LWR	pwr-v230	lwgr	gcr	agr	fbr
Am-241	5.8e+09	3.3e+10	0	0	0	1.9e+11
Ba-137m	1.2e+15	4.6e+15	7.9e+16	7.9e+15	7.9e+14	2.6e+15
Ba-140	4.5e+15	2.6e+16	5.0e+16	5.0e+15	1.0e+15	1.1e+15
Ce-141	9.7e+14	5.1e+15	9.9e+15	9.9e+14	9.9e+13	1.1e+14
Ce-143	8.6e+14	4.5e+15	0	0	0	8.9e+13
Ce-144	5.5e+14	2.9e+15	6.2e+15	6.2e+14	6.2e+13	7.1e+13
Cm-242	1.8e+12	1.0e+13	4.9e+13	4.9e+12	4.9e+11	2.2e+13
Cm-244	8.7e+10	5.0e+11	0	0	0	1.6e+12
Co-58	5.1e+12	3.3e+13	0	0	0	0
Co-60	3.4e+12	2.2e+13	0	0	0	0
Cs-134	2.0e+15	7.5e+15	5.4e+16	5.4e+15	5.4e+14	3.7e+15
Cs-136	7.8e+14	2.9e+15	3.5e+16	3.5e+15	3.5e+14	1.7e+15
Cs-137	1.2e+15	4.6e+15	7.9e+16	7.9e+15	7.9e+14	2.6e+15
I-129	0	0	0	0	0	0
I-131	4.9e+16	1.6e+17	1.6e+18	1.6e+17	1.6e+16	2.3e+16
I-132	7.1e+16	2.4e+17	0	0	0	3.1e+16
I-133	9.6e+16	3.2e+17	0	0	0	4.0e+16
I-134	1.1e+17	3.7e+17	0	0	0	4.2e+16
I-135	8.6e+16	2.9e+17	0	0	0	3.8e+16
Kr-85	4.5e+15	1.2e+16	2.5e+16	4.7e+15	4.9e+14	7.5e+14
Kr-85m	1.8e+17	5.0e+17	0	0	0	1.2e+16
Kr-87	3.6e+17	9.8e+17	0	0	0	2.0e+16
Kr-88	5.2e+17	1.4e+18	0	0	0	2.8e+16
La-140	4.6e+14	2.6e+15	1.0e+16	1.0e+15	1.0e+14	1.1e+14
Mo-99	7.9e+14	5.1e+15	3.5e+16	3.5e+15	3.5e+14	4.3e+14
Nb-95	4.1e+14	2.4e+15	9.5e+15	9.5e+14	9.5e+13	9.4e+13
Nd-147	1.7e+14	1.0e+15	0	0	0	4.7e+13
Np-237	0	0	0	0	0	0
Np-239	1.1e+16	5.8e+16	6.9e+16	6.9e+15	6.9e+14	1.8e+15
Pr-143	3.7e+14	2.1e+15	0	0	0	8.7e+13
Pu-238	4.1e+11	2.1e+12	2.0e+12	2.0e+11	2.0e+10	1.0e+12
Pu-239	1.5e+11	7.7e+11	1.7e+12	1.7e+11	1.7e+10	1.7e+11
Pu-240	1.5e+11	7.7e+11	2.4e+12	2.4e+11	2.4e+10	3.0e+11
Pu-241	2.5e+13	1.3e+14	2.6e+14	2.6e+13	2.6e+12	2.3e+13
Rb-86	6.3e+12	2.3e+13	0	0	0	2.7e+14
Rh-105	2.6e+14	1.7e+15	0	0	0	3.8e+14
Ru-103	5.5e+14	3.5e+15	2.8e+16	2.8e+15	2.8e+14	5.1e+14
Ru-105	3.8e+14	2.4e+15	0	0	0	3.8e+14
Ru-106	1.4e+14	8.7e+14	8.8e+15	8.8e+14	8.8e+13	2.8e+14
Sb-127	7.2e+14	5.0e+15	0	0	0	5.6e+14
Sb-129	3.8e+15	2.7e+16	0	0	0	1.4e+15
Sr-89	2.3e+15	1.5e+16	3.1e+16	3.1e+15	3.1e+14	1.9e+14
Sr-90	9.6e+13	6.2e+14	1.5e+15	1.5e+14	1.5e+13	2.4e+13
Sr-91	2.8e+15	1.8e+16	0	0	0	2.6e+14
Tc-99m	6.9e+14	4.4e+15	0	0	0	3.8e+14
Te-127	6.9e+14	4.8e+15	0	0	0	5.6e+14
Te-127m	1.3e+14	9.1e+14	0	0	0	7.5e+13
Te-129	3.6e+15	2.5e+16	0	0	0	1.4e+15
Te-129m	6.1e+14	4.2e+15	0	0	0	2.0e+14
Te-131m	1.5e+15	1.0e+16	0	0	0	5.8e+14
Te-132	1.4e+16	9.5e+16	3.1e+17	3.1e+16	3.1e+15	5.1e+15
Xe-133	1.3e+18	3.5e+18	6.0e+18	1.1e+18	1.2e+17	1.4e+17
Xe-135	2.7e+17	7.4e+17	0	0	0	1.5e+17
Y-90	1.2e+13	6.8e+13	0	0	0	5.3e+12
Y-91	3.2e+14	1.9e+15	0	0	0	5.2e+13
Zr-95	4.2e+14	2.4e+15	1.0e+16	1.0e+15	1.0e+14	9.6e+13
Zr-97	4.2e+14	2.4e+15	0	0	0	1.0e+14

Conditional releases, assuming safety improvements for eastern European reactors

Calculations for a situation where eastern European reactors are assumed to have a safety regime, which is comparable to that of western European reactors, were based on changes in the estimated probabilistic releases. The probability rate for severe core damage (p) was estimated at 10^{-3} per year for LWGR and PWR-V230 types. By taking conditional releases (L) of 1.2×10^{14} Bq for Cs-137 and 4.9×10^{15} Bq for I-131 the overall estimate of the product pL for these two nuclides equals that of western European reactors (with a p of 10^{-4} per year and a probabilistic release of 1.2×10^{15} Bq for Cs-137 and 4.9×10^{16} Bq for I-131).

Appendix 2 Indicative uncertainty analyses for risk estimates

Uncertainties regarding the estimation of overall risks occur along the various aspects of the risk estimation. Major sources of error are:

- uncertainty in accident probabilities and released amounts of the various nuclides
- uncertainty in the average dispersion and deposition estimates
- uncertainty in the exposure assessment estimates

This appendix provides a rough and preliminary evaluation of the combined uncertainties in the overall risk estimations reported here.

Uncertainty in accidental releases

Thusfar two severe accidents with a reactor core have occurred in approximately 6000 operational reactor years worldwide. An estimate for the overall probability rate of a reactor core accident (p) leads to: 3.3×10^{-4} per reactor year. This compares well with the overall estimate from Eendebak *et al.* (1992), providing an overall average risk for European nuclear reactors of approximately 2×10^{-4} per reactor year. On the basis of the historical observations we can estimate the uncertainty range for the overall meltdown probability. An upper limit for this probability rate is obtained estimating the value of p, for which the probability of less than three accidents in 6000 reactor years is 5%. If p represents the probability per reactor year that coremelt occurs, then the occurrence of less than three accidents in T reactor years equals (binomial statistics):

$$P_{<3} = (1-p)^T + T\cdot p\cdot(1-p)^{T-1} + \frac{T\cdot(T-1)}{2}\,p^2\cdot(1-p)^{T-2} \qquad [A2.1]$$

Applying the above equation we obtained an upper estimate of 1×10^{-3} per reactor year, for the overall probability of 5% for less than three accidents in 6000 reactor years. A lower estimate for the accident probability is obtained estimating a 5% probability that more than 1 accident occurs in 6000 reactor years. This leads to an estimated accident probability rate of 6×10^{-5} per year.

We can also try to estimate the overall average amount of released material in accidents per reactor year. Only the Chernobyl accident contributes significantly to the overall release in 6000 reactor years. Since Cs-137 was the major dose-contributing nuclide in the Chernobyl accident, we base our estimates upon the released fraction of this nuclide: in the Chernobyl accident 40% of the Cs-137 in the reactorcore was released to the atmosphere (Gudiksen *et al.*, 1989). Thus the overall average released Cs-137 fraction of the reactor core per reactor year is estimated at: $0.4/6000 = 6.7\times10^{-5}$ per reactor year. A standard error of 7×10^{-5} per year is calculated based upon these historical figures. A preliminary upper estimate regarding the average released fraction, based on historical figures is obtained summing the overall average and two times the standard error, thus arriving at: 21×10^{-5} per reactor year. This estimate for the upper limit is 5—6 times higher than the average estimate obtained from Eendebak *et al.* (1992), which equals 3.7×10^{-5} per year. We conclude that an uncertainty of a factor of 6 up can not be excluded for a 95% confidence interval.

It should be noted that this is a rough estimate, which does not take into full account the large differences between various reactors, and the improvements to reactors and safety procedures that occurred over the years. Also the uncertainty due to political and economic instabilities in the eastern European countries is not accounted for.

In trying to assess an overall indication of the uncertainty range we used:

a factor of 6 uncertainty for regions where western reactors contribute significantly, and a factor of 10 in the vicinity of eastern European reactors. The latter factor is in line with the expert judgement on uncertainties as provided by Eendebak *et al.* (1992).

Uncertainty in the dispersion and deposition estimates

Chapter 3 indicates an overall uncertainty of a factor of 4 up as well as down for the uncertainties in the air dispersion and deposition estimates. These findings are in line with the comparison of the OPS-model with SO_2 estimates, and also compares well with the observations from Chernobyl as presented in appendix 5. The factor 4 should also be regarded as a rough indication, and could be larger in regions where average weather conditions are very different from the Netherlands.

Uncertainty in the exposure assessment

Chapter 4 provides an indication of the uncertainties in the exposure assessment model, comparing results from our model with various other modelling attempts. An overall uncertainty range of a factor of 4 (up as well as

down) is estimated for the exposure assessment.

Overall uncertainty estimate

Combining the above three major sources of error, we can obtain an overall uncertainty range, assuming that:

- uncertainties are lognormally distributed, and
- the uncertainty intervals provided above represent the 95% intervals up as well as down
- overall risk is related to a multiplication of the amount of accidental releases, the integrated air concentration and deposition, and the exposure

The standard deviation for the three lognormal distributions for the above provided indications for the 95% uncertainty intervals are: ln(6)/2 for the accidental releases in western Europe, ln(10)/2 for accidental releases in eastern Europe, and ln(4)/2 for both dispersion and exposure calculations. The overall standard deviation of the lognormal distribution in the overall risk is calculated by means of:

$$\sigma_R = \sqrt{(\sum_{i=1}^{3} \sigma_i^2)} \qquad [A2.2]$$

where,

- σ_R standard deviation in ln(RISK)
- σ_1 standard deviation in logarithm of accident probabilities (ln(6)/2 for western European countries, ln(10)/2 in eastern Europe)
- σ_2 standard deviation in logarithm of air concentration and deposition (ln(4)/2)
- σ_3 standard deviation in logarithm of the exposure assessment (ln(4)/2)

The 95% confidence interval was estimated by means of a multiplication factor calculated as:

$$e^{(2 \cdot \sigma_R)} \qquad [A2.3]$$

A factor of 14 uncertainty (up as well as down) in western Europe, and a factor of 22 in eastern Europe is thus found. These estimates must be regarded as preliminary estimates for an overall confidence interval.

Appendix 3 Applying models for continuous releases to probabilistic assessments of accidental releases

Many models have been developed to describe dispersion, deposition and exposure under situations where continued releases occur. In this study we are interested in the time-averaged expectation of time-integrated air concentration and deposition for an accidental release occurring at a random moment in time. It will be shown that as long as the interest is focussed on probabilistic averages the methods for continuous releases can be applied, under the assumptions that:

- accident probability rates are constant over time and thus not related to weather (dispersion and deposition) conditions,
- accident probabilities are small over the operational period of a nuclear reactor
- consequences in terms of risk are linearly related to air and deposition characteristics.

A probabilistic approach of the risk-assessment implies that the probability of an accidental release is considered over a certain time-period. A release can occur at any time with a constant probability rate over time. This implies that the release could occur today, tomorrow or any other day with the same probability. The probabilistic estimate of the accidental release per unit time equals the multiplication of the accident probability rate p (per unit time) and the accidental release L. Due to the fact that we have assumed that p and L are independent of the weather conditions, pL can be considered as the expected release rate at any moment in time. This situation is fully equivalent to a constant continuous release rate of size pL. In the situation of a continuous release the time-averaged air concentration ($<C>$)(or for that matter time-averaged deposition rate) is often calculated. In case of an accidental release the probabilistic mean of the time integrated air concentration (or total deposition) is the relevant parameter. It can be shown by a simple change of integrations that $<C>/p$ equals the probabilistic mean time-integrated air concentration per accidental release L. The number of accidental releases per year is given by $p \cdot 1$ year, and thus for the probalistic mean of the time-integrated air concentration we find: $<C> \cdot (1 \text{ year})$. The conclusion can be that per operational year the time integrated air concentration equals $<C>$. A more technical description follows.
Consider a nuclear power reactor at location x with a probability density for severe accidents that is constant over time, implying that the hazard function is constant $h(t)=h$. The cumulative hazard, representing the expectation for the number of accidents, increases linearly over time: $H(t)= ht$. Although ht can be larger than 1, it is not very realistic that more than one large scale accident occurs at a specific plant. Assuming that a reactor is closed after a large accident, one is interested in estimating the probability density function of first accidents, in stead of the hazard function provided above. It will be shown that hazard function and probability density function for first accident are equivalent in case of low accident probabilities ($ht << 1$).

Determination of probability density function for first large accidents in a specific reactor
Consider an infinitesimal time interval Δ. In that case $h \cdot \Delta$ represents the probability that an accident occurs within this time interval, provided that the reactor is operational. The probability that no accident has occurred at time t following the start of operationality of the reactor can be calculated when $N=t/\Delta$:

$$\lim_{\substack{N\to\infty \\ N\Delta=t}} \binom{N}{0} (1-h\cdot\Delta)^N = \lim_{N\to\infty} (1-\frac{h\cdot t}{N})^N = e^{(-h\cdot t)} \qquad [A3.1]$$

The cumulative probability that a first large scale accident has occurred at time t equals one minus the probability that no accident has occurred:

$$P(t) = 1 - e^{(-h\cdot t)} \qquad [A3.2]$$

For the probability density function for the occurrence of first accidents we find, after taking the first derivative from equation (A3.2):

$$p(t) = \frac{dP(t)}{dt} = h\cdot e^{(-h\cdot t)} \qquad [A3.3]$$

If $ht << 1$, in other words if the accident probability is low over the considered time-period, then p(t) is

constant and approximated by: p(t)=h=p. Hence the expectation for the accidental release is constant over time and equals pL.

Derivation of time integrated air concentration for accidental situation

Let $c_{t0}(t)$ provide the relationship between an accidental release of unit amount at location x and the air concentration at time t after the release at location y. Since weather conditions influence this conversion function in shape and size, this relationship depends upon the time of the release and the weather conditions afterwards. Although it will not be possible to determine this relationship at the moment of the release due to uncertainties in the weather forecast, one could uniquely determine the relationship by means of measurements of air concentrations, or reconstruct the relationship provided that sufficient information on the weather situation is available. Thus if the weather pattern is known $c_{t0}(t)$ is fully determined by the release time t_0. Radiation exposure following an accidental release is at a certain location directly proportional to the time-integrated air concentration (and total deposition).

Given a release time t_0 the time integrated air concentration $(Ct)_{t0}$ is given by:

$$(Ct)_{t_0} = \int_0^T L \cdot c_{t_0}(t)\, dt \qquad [A3.4]$$

Equation A3.4 holds for a release at t_0. We are interested in a probabilistic expectation of the integrated air concentration, taking into account the probability that an accident occurs at t_0 and the probability of a certain dispersion pattern. Multiplying Ct with the probability rate of an accidental release (p) and integrating over a sufficiently long period T to cover a full variety of weather conditions we can estimate the probabilistic expectation of the time integrated air concentration for the period T:

$$\langle Ct \rangle = \int_0^\infty p \cdot (Ct)_{t_0}\, dt_0 \qquad [A3.5]$$

it should be noted that A3.5 holds under the assumption that $pT << 1$. If we are interested in the expectation for the time integrated air concentration over a unit time period of operation the expression in equation A3.5 should be divided by T (dimension becomes time-integrated air concentration per year of reactor operation, or: air concentration). Thus per year of operation we find, combining equations A3.4 and A3.5:

$$\langle Ct \rangle_{year} = \frac{pL}{T} \int_0^T \int_0^\infty c_{t_0}(t)\, dt\, dt_0 \qquad [A3.6]$$

The question rises how the time integrated concentration could be estimated applying dispersion modelling. The straightforward method would be to calculate time-integrated air concentration on the basis of a real time dispersion model for a large number of evaluations covering a wide variety of weather conditions. However such an evaluation is very time-consuming and thus is regarded impracticable. Since we are mainly interested in probabilistic expectations, and not in real time assessments an other approach could be chosen. The approach uses a probabilistic air dispersion model, which provides time-averaged air concentrations and deposition-rates. In order to be able to use such a model one should relate the time-integrated air concentration with the average air concentration. It will be shown that in fact a direct link can be made between those two parameters.

A probabilistic air concentration model provides the average air concentration. One could think of a continuous release with release rate L', as a large number of successive short term releases. Thus in order to calculate the time averaged concentration we can use the above defined relationships $c_{t0}(t)$, to describe the contribution for each of the small releases to the overall concentration. As mentioned before $c_{t0}(t)$ varies strongly with t_0. Since we are determining the average air concentration, we can calculate the average release-to-concentration conversion relation (c(t)):

$$c(t) = \frac{1}{T} \int_0^T c_{t_0}(t)\, dt_0 \qquad [A3.7]$$

This time and thus weather averaged conversion relationship can be used to calculate the average air concentration C_m:

$$C_m = \int_0^\infty L' \cdot c(t)\, dt \qquad [A3.8]$$

Combining equations A3.7 and A3.8 and changing the integration we find:

$$C_m = \frac{L'}{T} \int_0^\infty \int_0^T c_{t_0}(t)\, dt_0 dt = \frac{L'}{T} \int_0^T \int_0^\infty c_{t_0}(t)\, dt dt_0 \qquad [A3.9]$$

Combining equations A3.6 and A3.9 we find a relation between the probabilistic expectation of the time-integrated air concentration per year of operation, and the time-averaged air concentration for a continuous release:

$$\langle Ct \rangle_{year} = \frac{pL}{L'} C_m \qquad [A3.10]$$

As can be seen in equation A3.10 we can use the average air concentration directly for the accident case provided that L' equals pL.

Appendix 4 Applied parameter values in the air dispersion modelling with the OPS-model

The OPS-model was used with several sets of parametervalues. The model was run for Cs-137 and I-131, for 0 internal energy of the plume and 100 MW and for 5 and 100 km grids. All other parameter values were kept the same.

Table A4.1 Parametervalues used in operating the OPS dispersion model (Van Jaarsveld, 1990)

Modelparameter	Parametervalues	Remarks
Material to be modelled	resp. Cs-137 and $(I\text{-}131)_2$	Molecular weights 137 and 262, to be modelled as gasses
Emission height	100 m, heat content: resp. 0 MW and 100 MW	Stack height and internal energy of plume
Diameter of source	0 m	Point source
Type of emission variation	Continuous	Emission is constant during year
Location of source	x: 141.9 km, y: 459.1 km	Location in RD-coordinates (not of specific importance)
Number of grid elements	x en y: 35	17 grid elements west, east, north and south of source location
Grid resolution	5 resp. 100 km	Values refer to two different model runs
Climate	The Netherlands, average over 10 years	Most general data set available
Roughness length	0.15 m	Large-scale roughness length for the Netherlands
Dry deposition	velocity: 0.002 $m.s^{-1}$ (Cs-137), 0.01 $m.s^{-1}$ (I-131), deposition in $mol.(ha.year)^{-1}$	Modelling dry deposition through a dry deposition velocity method
Wet deposition	scavenging: 36 $\%.h^{-1}$ deposition in $mol.(ha.year)^{-1}$	An average scavenging coefficient during precipitation events (± 6% of the time)
Conversion	2.65×10^{-4} $\%.hr^{-1}$ (Cs-137) 0.359 $\%.hr^{-1}$ (I-131)	To be translated as: there is a radioactive decay in $\%.hr^{-1}$

Table A4.2 Parameter values for userfunction (see formula 3.1).

	Concentration *)			Deposition *)		
	A	B	C	A	B	C
Cs-137, stackheight 100m, no thermal energy in plume						
- North-east:	1.74×10^{-14}	0.0015	1.4	1.3×10^{-9}	0.0014	1.23
- South-east:	0.95×10^{-14}	0.0015	1.37	0.7×10^{-9}	0.0012	1.28
- South-west:	2.0×10^{-14}	0.0007	1.58	1.25×10^{-9}	0.00057	1.5
- North-west:	2.2×10^{-14}	0.00125	1.6	1.2×10^{-9}	0.0012	1.45
Cs-137, stackheight 100m, 100 MW thermal energy in plume						
- North-east:	5.0×10^{-16}	0.0020	0.8	3.2×10^{-10}	0.0014	1.0
- South-east:	2.0×10^{-16}	0.0015	0.8	1.0×10^{-10}	0.0013	0.95
- South-west:	2.0×10^{-16}	0.0012	0.8	7.0×10^{-11}	0.0009	1.0
- North-west:	1.5×10^{-16}	0.0016	0.8	9.0×10^{-11}	0.00135	1.0
I-131, stackheight 100m, no thermal energy in plume						
- North-east:	2.3×10^{-14}	0.0022	1.6	6.5×10^{-9}	0.0019	1.51
- South-east:	1.57×10^{-14}	0.0020	1.63	3.5×10^{-9}	0.0019	1.53
- South-west:	4.7×10^{-14}	0.0011	1.9	5.5×10^{-9}	0.0013	1.7
- North-west:	7.3×10^{-14}	0.0016	2.0	1.5×10^{-8}	0.0016	1.9
I-131, stackheight 100m, 100 MW thermal energy in plume						
- North-east:	6.0×10^{-16}	0.0026	0.9	6.0×10^{-10}	0.0023	1.0
- South-east:	2.5×10^{-16}	0.0021	0.9	2.3×10^{-10}	0.0020	1.0
- South-west:	2.2×10^{-16}	0.0018	0.9	1.5×10^{-10}	0.0017	1.0
- North-west:	2.0×10^{-16}	0.0021	0.95	1.5×10^{-10}	0.0021	1.0

*) In case of air concentration the dimension of A is $Bq \cdot a \cdot m^{-3}$ per Bq released; and in case of deposition A is $Bq \cdot m^{-2}$ per Bq released. In applying equation 3.1 $r = r'/r_0$, where $r_0 = 1$ km and r' is the distance in km. B and C are dimensionless.

Appendix 5 Comparison of OPS-modelling with deposition measured following Chernobyl

As described before the OPS-model in its present form was defined to describe average air concentration and deposition in the Netherlands, thus the weather characteristics are specific for the Netherlands. Just in order to have a rough idea about the applicability for accidents with nuclear power reactors, we have compared calculations for the Chernobyl accident, based upon the previously described user functions extracted from OPS, with actual deposition measurements performed following the reactor accident. Calculated results were obtained assuming a Cs-137 release of 8.9×10^{16} Bq (taken from Gudiksen *et al.*, 1990). Figure A5.1 shows the OPS-calculated deposition results for the various quadrants (curves) versus the distance from Chernobyl. Data, as obtained from UNSCEAR 1988, were also plotted in that figure. The dots show results UNSCEAR presented as regional estimates for various areas in Europe, Asia, Africa and America. The triangles show the raw data obtained for various countries, and parts of countries. In the range below 5000 km most data are for European countries westward from the Chernobyl plant. Limited information on deposition close to the reactor site, and in the eastern part of the Soviet Union was obtained. Nevertheless it can be seen that the OPS-calculated values are within the range of values found from this dataset. Because an average deposition is calculated using OPS, the actual deposition values for heavy contaminated areas (e.g. directly below the plume in Sweden and Finland) are estimated to be above those OPS-calculated values, whereas only slightly contaminated areas are estimated to be below the calculated values. Comparing the country based deposition data to the calculated estimates for the relevant quadrant, we find that measured deposition is on average 3-fold higher than the calculated value (based upon the exponent of the average value of ln(observed/calculated) for all data points closer than 5000 km). For the six points closer than 1000 km the underestimation was no more than 20%. Since the observed radioactive plumes were primarily directed to the northwest at first and southwestwards (UNSCEAR, 1988) lateron it is not surprising that the calculated values for west and especially northern Europe are lower than the measured ones. Although we realise that this on itself does not proof the justification of the use of OPS, it provides a satisfying resemblance. Further substantiation of these findings could be obtained by comparing the average results obtained with real-time models with OPS-based estimates.
The above calculations give some confidence that the error using OPS is on average within a factor of 2—3.

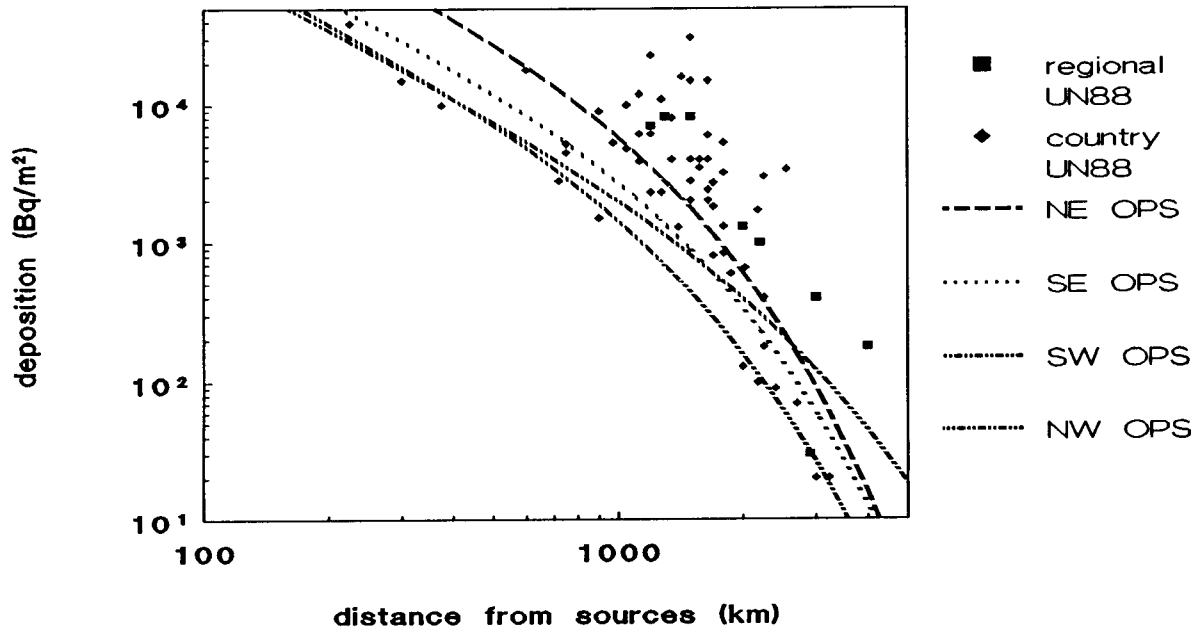

Figure A5.1 Cs-137 deposition. Chernobyl data [UNSCEAR,1988] versus OPS

Appendix 6 Input and results for exposure modelling (NucRed-model)

Table A6.1 Decay constants and dose conversion factors (Nosske *et al.*,1985; Kocher, 1983)

nuclide	decay (λ) s^{-1}	external cloud $Sv \cdot s^{-1}/Bq \cdot m^{-3}$	external ground $Sv \cdot s^{-1}/Bq \cdot m^{-2}$	inhalation $Sv \cdot Bq^{-1}$	ingestion $Sv \cdot Bq^{-1}$
Am-241	5.09e-11	8.28e-16	2.62e-17	1.40e-04	5.90e-07
Ba-137m	4.53e-03	2.66e-14	5.39e-16	1.80e-13	6.90e-13
Ba-140	6.27e-07	8.31e-15	1.89e-16	1.00e-09	2.50e-09
Ce-141	2.47e-07	3.39e-15	7.67e-17	2.40e-09	7.80e-10
Ce-143	5.83e-06	1.17e-14	2.85e-16	9.20e-10	1.20e-09
Ce-144	2.82e-08	8.09e-16	1.88e-17	1.00e-07	5.70e-09
Cm-242	4.92e-08	4.50e-18	8.63e-19	4.80e-06	1.90e-08
Cm-244	1.21e-09	3.84e-18	7.67e-19	7.60e-05	3.10e-07
Co-58	1.13e-07	4.38e-14	8.56e-16	2.90e-09	9.70e-10
Co-60	4.20e-09	1.13e-13	1.98e-15	5.90e-08	7.30e-09
Cs-134	1.07e-08	6.98e-14	1.37e-15	1.30e-08	2.00e-08
Cs-136	6.10e-07	9.80e-14	1.86e-15	2.00e-09	3.00e-09
Cs-137	7.35e-10	7.17e-17	2.35e-18	8.60e-09	1.40e-08
I-129	1.40e-15	3.68e-16	1.93e-17	4.20e-08	6.70e-08
I-131	9.98e-07	1.67e-14	3.52e-16	8.10e-09	1.30e-08
I-132	8.37e-05	1.04e-13	2.04e-15	9.70e-11	1.70e-10
I-133	9.26e-06	2.69e-14	5.74e-16	1.50e-09	2.60e-09
I-134	2.20e-04	1.20e-13	2.31e-15	3.50e-11	6.50e-11
I-135	2.91e-05	7.26e-14	1.29e-15	3.10e-10	5.60e-10
Kr-85	2.05e-09	2.28e-16	1.01e-17	0	0
Kr-85m	4.30e-05	7.23e-15	1.68e-16	0	0
Kr-87	1.51e-04	3.96e-14	7.71e-16	0	0
Kr-88	6.78e-05	9.86e-14	1.56e-15	0	0
La-140	4.79e-06	1.07e-13	1.91e-15	1.30e-09	2.30e-09
Mo-99	2.92e-06	7.17e-15	1.74e-16	1.10e-09	1.40e-09
Nb-95	2.29e-07	3.46e-14	6.72e-16	1.60e-09	6.90e-10
Nd-147	7.31e-07	5.90e-15	1.38e-16	1.80e-09	1.20e-09
Np-237	1.03e-14	9.99e-16	2.84e-17	1.30e-04	1.10e-06
Np-239	3.41e-06	7.36e-15	1.67e-16	6.60e-10	8.80e-10
Pr-143	5.92e-07	1.73e-16	1.99e-17	2.20e-09	1.30e-09
Pu-238	2.50e-10	4.03e-18	7.96e-19	1.30e-04	1.10e-07
Pu-239	9.14e-13	3.65e-18	3.49e-19	1.40e-04	1.20e-07
Pu-240	3.36e-12	3.96e-18	7.64e-19	1.40e-04	1.20e-07
Pu-241	1.53e-09	0	0	2.80e-06	2.40e-09
Rb-86	4.30e-07	4.72e-15	1.55e-16	1.80e-09	2.50e-09
Rh-105	5.45e-06	3.46e-15	7.36e-17	2.60e-10	4.00e-10
Ru-103	2.04e-07	2.11e-14	4.34e-16	2.40e-09	8.20e-10
Ru-105	4.34e-05	3.52e-14	7.36e-16	1.20e-10	2.90e-10
Ru-106	2.18e-08	0	0	1.30e-07	7.40e-09
Sb-127	2.08e-06	2.96e-14	6.09e-16	1.60e-09	1.90e-09
Sb-129	4.38e-05	6.53e-14	1.25e-15	1.70e-10	4.80e-10
Sr-89	1.59e-07	3.77e-16	6.75e-17	1.10e-08	2.50e-09
Sr-90	7.69e-10	9.16e-17	1.38e-18	3.50e-07	3.50e-08
Sr-91	2.03e-05	3.16e-14	6.56e-16	4.50e-10	8.40e-10
Tc-99m	3.20e-05	5.77e-15	1.28e-16	8.80e-12	1.70e-11
Te-127	2.06e-05	3.17e-16	9.61e-18	8.60e-11	1.90e-10
Te-127m	7.36e-08	1.40e-16	6.09e-18	5.80e-09	2.20e-09
Te-129	1.66e-04	2.69e-15	1.06e-16	2.40e-11	5.40e-11
Te-129m	2.39e-07	1.64e-15	5.61e-17	6.50e-09	2.90e-09
Te-131m	6.42e-06	6.47e-14	1.23e-15	1.60e-09	2.40e-09
Te-132	2.46e-06	9.51e-15	2.15e-16	2.40e-09	2.40e-09
Xe-133	1.53e-06	1.55e-15	4.44e-17	0	0
Xe-135	2.11e-05	1.10e-14	2.53e-16	0	0
Y-90	3.00e-06	6.28e-16	1.07e-16	2.30e-09	2.90e-09
Y-91	1.37e-07	5.49e-16	7.32e-17	1.30e-08	2.60e-09
Zr-95	1.25e-07	3.33e-14	6.50e-16	6.50e-09	1.00e-09
Zr-97	1.14e-05	8.63e-15	2.31e-16	1.20e-09	2.30e-09

Table A6.2 Correction factors used in the assessment. For air no correction was needed, for deposition noble gasses do not deposit. The ingestion factors denote the effect of daughter nuclides. The values for external radiation give the terms to be added to the values for their mother nuclides (see A6.1)

Correction	Deposition	External from clouds	External from soil	Ingestion
Am-241	1.00e+00	0	0	1.00e+00
Ba-137m	1.00e+00	0	0	1.00e+00
Ba-140	1.00e+00	0	0	1.82e+00
Ce-141	1.00e+00	0	0	1.00e+00
Ce-143	1.00e+00	0	0	1.96e+00
Ce-144	1.00e+00	0	1.53e-16	1.01e+00
Cm-242	1.00e+00	0	0	1.05e+00
Cm-244	1.00e+00	0	0	1.00e+00
Co-58	1.00e+00	0	0	1.00e+00
Co-60	1.00e+00	0	0	1.00e+00
Cs-134	1.00e+00	0	0	1.00e+00
Cs-136	1.00e+00	0	0	1.00e+00
Cs-137	1.00e+00	2.66e-14	5.39e-16	1.00e+00
I-129	1.00e+00	0	0	1.00e+00
I-131	1.00e+00	0	0	1.00e+00
I-132	1.00e+00	0	0	1.00e+00
I-133	1.00e+00	0	0	1.00e+00
I-134	1.00e+00	0	0	1.00e+00
I-135	1.00e+00	0	0	1.00e+00
Kr-85	0	0	0	1.00e+00
Kr-85m	0	0	0	1.00e+00
Kr-87	0	0	0	1.00e+00
Kr-88	0	0	0	1.00e+00
La-140	1.00e+00	0	0	1.00e+00
Mo-99	1.00e+00	0	1.28e-16	1.01e+00
Nb-95	1.00e+00	0	0	1.00e+00
Nd-147	1.00e+00	0	0	1.04e+00
Np-237	1.00e+00	0	0	1.00e+00
Np-239	1.00e+00	0	0	1.02e+00
Pr-143	1.00e+00	0	0	1.00e+00
Pu-238	1.00e+00	0	0	1.00e+00
Pu-239	1.00e+00	0	0	1.00e+00
Pu-240	1.00e+00	0	0	1.00e+00
Pu-241	1.00e+00	0	0	1.03e+00
Rb-86	1.00e+00	0	0	1.00e+00
Rh-105	1.00e+00	0	0	1.00e+00
Ru-103	1.00e+00	0	1.03e-18	1.01e+00
Ru-105	1.00e+00	0	0	9.09e+00
Ru-106	1.00e+00	0	3.20e-16	1.00e+00
Sb-127	1.00e+00	0	9.61e-18	1.64e+00
Sb-129	1.00e+00	0	1.06e-16	1.00e+00
Sr-89	1.00e+00	0	0	1.00e+00
Sr-90	1.00e+00	0	0	1.07e+00
Sr-91	1.00e+00	0	0	2.86e+00
Tc-99m	1.00e+00	0	0	1.00e+00
Te-127	1.00e+00	0	0	1.00e+00
Te-127m	1.00e+00	0	9.61e-18	1.09e+00
Te-129	1.00e+00	0	0	1.00e+00
Te-129m	1.00e+00	0	0	1.01e+00
Te-131m	1.00e+00	0	4.57e-16	1.31e+01
Te-132	1.00e+00	0	2.04e-15	1.23e+00
Xe-133	0	0	0	1.00e+00
Xe-135	0	0	0	1.00e+00
Y-90	1.00e+00	0	0	1.00e+00
Y-91	1.00e+00	0	0	1.00e+00
Zr-95	1.00e+00	0	0	1.00e+00
Zr-97	1.00e+00	0	1.28e-15	2.94e+00

Table A6.3 Transfer factors used in the assessment. Water to soil distribution transfer factors pasture-milk/meat and soil-plant for several crops.

	K_d	F_{milk}	F_{meat}	B_v (n.g)	B_v (n.g)	B_v (n.g)	B_v (n.g)
	$(m^3 \cdot t^{-1})$	$(d \cdot kg^{-1})$	$(d \cdot kg^{-1})$	vegetables	cereals	roots/tubers	nuclide
Am	700	2.0e-05	5.0e-04	1.0e-03	1.0e-03	1.0e-05	1.0e-03
Ba	60	4.0e-04	1.0e-04	1.5e-01	1.5e-01	1.5e-01	1.5e-01
Ce	850	2.0e-05	2.0e-03	1.0e-02	1.0e-02	1.0e-02	1.0e-02
Cm	2000	2.0e-05	2.0e-04	1.0e-03	1.0e-03	1.0e-05	1.0e-03
Co	45	2.0e-04	1.0e-02	1.0e-02	1.0e-03	1.0e-02	2.0e-03
Cs	1000	5.0e-03	3.0e-02	2.0e-02	2.0e-02	1.0e-02	1.0e-02
I	60	3.0e-03	1.0e-02	2.0e-02	2.0e-02	2.0e-02	2.0e-02
Kr	0	0	0	0	0	0	0
La	650	2.0e-05	2.0e-03	1.0e-02	1.0e-02	1.0e-02	1.0e-02
Mo	20	2.0e-03	7.0e-03	8.7e-02	6.0e-02	4.5e-01	6.0e-02
Nb	350	3.0e-03	3.0e-01	1.0e-02	1.0e-02	1.0e-02	1.0e-02
Nd	650	2.0e-05	4.0e-03	1.0e-02	1.0e-02	1.0e-02	1.0e-02
Np	300	5.0e-06	2.0e-04	1.0e-04	1.0e-04	1.0e-06	1.0e-03
Pr	650	2.0e-05	5.0e-03	1.0e-02	1.0e-02	1.0e-02	1.0e-02
Pu	4500	1.0e-07	3.0e-04	1.0e-04	1.0e-04	1.0e-06	1.0e-03
Rb	600	6.0e-03	1.0e-02	5.2e-03	3.6e-02	2.7e-01	3.6e-02
Rh	60	2.0e-03	0	1.5e-01	1.5e-01	1.5e-01	1.5e-01
Ru	350	1.0e-06	2.0e-03	7.5e-02	7.5e-02	7.5e-02	7.5e-02
Sb	450	2.0e-03	1.0e-03	2.0e-01	2.0e-01	2.0e-01	2.0e-01
Sr	35	2.0e-03	6.0e-04	5.0e-02	2.0e-01	8.0e-02	6.0e-02
Tc	15	1.0e-05	4.0e-02	1.0e+01	1.0e+01	1.0e+01	1.0e+01
Te	300	2.0e-04	8.0e-02	2.5e-02	2.5e-02	2.5e-02	2.5e-02
Xe	0	0	0	0	0	0	0
Y	500	1.0e-05	1.0e-03	1.5e-02	1.5e-02	1.5e-02	1.5e-02
Zr	3000	5.0e-06	2.0e-02	2.0e-04	2.0e-04	2.0e-04	2.0e-04

K_d : soil affinity of nuclide $(m^3 \cdot kg^{-1})$
F_{milk} : transfer to milk $(day \cdot kg^{-1})$
F_{meat} : transfer to meat $(day \cdot kg^{-1})$
B_v (n.g) : concentration factor for uptake of the radionuclide from soil by edible parts of crops, $(Bq \cdot kg^{-1})$ plant tissue per $(Bq \cdot kg^{-1})$ dry soil

Table A6.4 Plant characteristics used in this assessment

	unit	grass	vegetables	cereals	roots/tubers
S	$kg \cdot m^{-2}$	120	280	280	280
h	m	0.1	0.2	0.2	0.2
rho (ρ)	$kg \cdot m^{-3}$	1200	1400	1400	1400
theta (θ)		0.25	0.25	0.25	0.25
$W_R+W_I-W_E$	$m \cdot day^{-1}$	1.1e-3	1.1e-3	1.1e-3	1.1e-3
Y_p	$kg \cdot m^{-2}$	7.4	3.8	0.72	4.6
λ_w	day^{-1}	0.0496	0.0496	0.0496	0.0496
F_{ip}		0.25	0.3	0.05	0.3
t_{ap}	day	30	60	60	60

S : mass of soil in plough layer ($kg \cdot m^{-2}$) (= depth of plough layer × soil density)
h : thickness of plough layer (m)
rho (ρ) : soil density ($kg \cdot m^{-3}$)
theta (θ) : volumetric water content in soil
$W_R+W_I-W_E$: water balance: rainfall plus irrigation minus loss due to evaporation ($m \cdot day^{-1}$)
Y_p : agricultural productivity (yield) or standing crop biomass of the dible portion of vegetation ($kg \cdot m^{-2}$)
λ_w : rate constant for reduction of the concentration of material deposited on the surface of vegetation due to processes other than radiological decay (day^{-1})
F_{ip} : direct interception fraction for crop type p
t_{ap} : time period during growing season that crops can be contaminated through direct interception of deposition (day)

Table A6.5 Diets as used in this assessment (see also table A6.7)

	delay-time t_d days	consumption cow (fresh weight) $kg \cdot day^{-1}$	consumption human $kg \cdot day^{-1}$	preparation reduction F_{bp}
grass	0	85.5	0	1
vegetables	0	0	0.641	0.5
cereals	60	0	0.299	0.3
roots	0	0	0.342	0.5
milk	0	0	0.803	1
meat	20	0	0.14	1
soil		0.6	0	1

Table A6.6 Overall dose conversion factors per unit deposition and time integrated air concentration as calculated in chapter 4.3

Nuclide	Ingestion $Sv/(Bq \cdot m^{-2})$	External soil $Sv/(Bq \cdot m^{-2})$	Inhalation $Sv/(Bq \cdot s \cdot m^{-3})$	External air $Sv/(Bq \cdot s \cdot m^{-3})$
Am-241	5.1e-07	1.0e-08	3.7e-08	6.5e-16
Ba-137m	6.6e-17	6.1e-14	4.8e-17	2.1e-14
Ba-140	1.7e-09	1.4e-10	2.7e-13	6.6e-15
Ce-141	4.2e-10	1.2e-10	6.4e-13	2.7e-15
Ce-143	1.6e-10	2.5e-11	2.4e-13	9.2e-15
Ce-144	4.6e-09	1.5e-09	2.7e-11	6.4e-16
Cm-242	1.5e-08	4.8e-12	1.3e-09	3.6e-18
Cm-244	2.6e-07	1.1e-10	2.0e-08	3.0e-18
Co-58	6.9e-10	2.4e-09	7.7e-13	3.5e-14
Co-60	6.7e-09	9.4e-08	1.6e-11	8.9e-14
Cs-134	2.7e-08	2.8e-08	3.5e-12	5.5e-14
Cs-136	1.5e-09	1.4e-09	5.3e-13	7.7e-14
Cs-137	4.8e-08	1.1e-07	2.3e-12	2.1e-14
I-129	1.3e-07	8.1e-09	1.1e-11	2.9e-16
I-131	4.2e-09	1.7e-10	2.2e-12	1.3e-14
I-132	1.0e-12	1.2e-11	2.6e-14	8.2e-14
I-133	1.3e-10	3.2e-11	4.0e-13	2.1e-14
I-134	1.5e-13	5.4e-12	9.3e-15	9.5e-14
I-135	9.5e-12	2.3e-11	8.3e-14	5.7e-14
Kr-85	0	0	0	1.8e-16
Kr-85m	0	0	0	5.7e-15
Kr-87	0	0	0	3.1e-14
Kr-88	0	0	0	7.8e-14
La-140	1.8e-10	2.0e-10	3.5e-13	8.5e-14
Mo-99	1.9e-10	5.3e-11	2.9e-13	5.7e-15
Nb-95	1.2e-09	1.1e-09	4.3e-13	2.7e-14
Nd-147	4.1e-10	8.9e-11	4.8e-13	4.7e-15
Np-237	9.2e-07	1.2e-08	3.5e-08	7.9e-16
Np-239	9.4e-11	2.5e-11	1.8e-13	5.8e-15
Pr-143	4.8e-10	1.5e-11	5.9e-13	1.4e-16
Pu-238	9.2e-08	2.6e-10	3.5e-08	3.2e-18
Pu-239	1.0e-07	1.5e-10	3.7e-08	2.9e-18
Pu-240	1.0e-07	3.2e-10	3.7e-08	3.1e-18
Pu-241	2.0e-09	0	7.5e-10	0
Rb-86	1.5e-09	1.5e-10	4.8e-13	3.7e-15
Rh-105	3.2e-11	6.9e-12	6.9e-14	2.7e-15
Ru-103	4.8e-10	7.7e-10	6.4e-13	1.7e-14
Ru-105	2.5e-11	8.6e-12	3.2e-14	2.8e-14
Ru-106	6.5e-09	3.5e-09	3.5e-11	0
Sb-127	5.7e-10	1.5e-10	4.3e-13	2.3e-14
Sb-129	5.2e-12	1.6e-11	4.5e-14	5.2e-14
Sr-89	1.7e-09	1.5e-10	2.9e-12	3.0e-16
Sr-90	1.0e-07	2.8e-10	9.3e-11	7.2e-17
Sr-91	5.5e-11	1.6e-11	1.2e-13	2.5e-14
Tc-99m	3.3e-13	2.0e-12	2.3e-15	4.6e-15
Te-127	3.8e-12	2.4e-13	2.3e-14	2.5e-16
Te-127m	2.8e-09	6.3e-11	1.5e-12	1.1e-16
Te-129	1.4e-13	3.3e-13	6.4e-15	2.1e-15
Te-129m	2.4e-09	8.8e-11	1.7e-12	1.3e-15
Te-131m	1.9e-09	1.3e-10	4.3e-13	5.1e-14
Te-132	4.2e-10	4.7e-10	6.4e-13	7.5e-15
Xe-133	0	0	0	1.2e-15
Xe-135	0	0	0	8.7e-15

Nuclide	Ingestion $Sv/(Bq \cdot m^{-2})$	External soil $Sv/(Bq \cdot m^{-2})$	Inhalation $Sv/(Bq \cdot s \cdot m^{-3})$	External air $Sv/(Bq \cdot s \cdot m^{-3})$
Y-90	3.4e-10	1.8e-11	6.1e-13	5.0e-16
Y-91	1.6e-09	1.8e-10	3.5e-12	4.3e-16
Zr-95	7.3e-10	1.7e-09	1.7e-12	2.6e-14
Zr-97	2.4e-10	6.8e-11	3.2e-13	6.8e-15
Total	2.4e-06	2.8e-07	2.0e-07	1.1e-12

Parameters applied in calculations for small children:

Dose-conversion factors were obtained for one year old children. Conversion factors for children were taken for the major 16 nuclides, using the highest values from Henrichs *et al.* (1985): I-131; Cs-137; Cs-134; Te-132; I-133; Sr-90; Ba-140; Sr-89; Ce-144; Cs-136; Ru-106; Pu-238; Pu-241; Am-241; Cm-242 and Pu-240. For the other nuclides dose-conversion factors were as provided in table A6.1.

Other changes regarding input parameters are: breathing rate of 3.8 m^3 per day (ICRP, 1974), and intakes according to table A6.7.

Table A6.7 Intake for small children used in calculations

food-product	'average' intake $kg \cdot day^{-1}$	'extreme' intake $kg \cdot day^{-1}$
vegetables	0.121	0.318
roots/tubers	0.068	0.151
cereals	0.112	0.186
meat	0.025	0.063
milk	0.515	1.236

Appendix 7 Relationship between Cs-137 contamination and total dose for Chernobyl source term, as derived with NucRed model.

We evaluated the Chernobyl source term with the NucRed exposure model and determined a relationship between the Cs-137 contamination and the total dose. Thus, in this approach all nuclides including the various iodine isotopes are grouped with Cs-137, and dose contributions for the other nuclides are accounted for by means of a multiplication factor. A calculation was made with average ingestion parameters (taken from ICRP (1974)), and population behaviour according to the assumptions made by UNSCEAR (1988). The following relationship was derived from the results with the NucRed model:

$$D_1 = 1.84 \cdot 10^{-7} \cdot O_A + 2.93 \cdot 10^{-11} \cdot C_A \qquad [A7.1]$$

where,

D_1 - the dose due to all nuclides and pathways for nuclide-composition according to LWGR-sourceterm (Sv)

O_A - Cs-137 total deposition ($Bq \cdot m^{-2}$)

C_A - time-integrated air-concentration for Cs-137 ($Bq \cdot s \cdot m^{-3}$)

This relationship was applied to obtain dose-estimates (70 years follow up period) on the basis of the deposition and air concentration data provided by UNSCEAR. The result was plotted versus the dose estimated by UNSCEAR (see figure 4.3 for results).

Appendix 8 Calculation of distances over the globe, applied in RISKA-model

Distances are calculated assuming the earth is perfectly spherical. The distance between two locations on the globe is calculated by means of:

$$r_{ij} = r_e \cdot \alpha_{ij} \qquad [A8.1]$$

where,

r_{ij}	-	distance over the globe between locations i and j (km)
r_e	-	radius of the earth (6378 km)
α_{ij}	-	angle between location vectors for locations i and j; location vectors start in the centre of the earth (angle in radians; see expression below for calculation of α)

The angle between the location vectors is calculated by means of:

$$\alpha_{ij} = \arccos(\cos(\theta_i)\cdot\sin(\phi_i)\cdot\cos(\theta_j)\cdot\sin(\phi_j) + \cos(\theta_i)\cdot\cos(\phi_i)\cdot\cos(\theta_j)\cdot\cos(\phi_j) + \sin(\theta_i)\cdot\sin(\theta_j)) \qquad [A8.2]$$

where,

θ_i, θ_j	-	latitude for location i, respectively j
ϕ_i, ϕ_j	-	longitude for location i, respectively j

Appendix 9 Impact of ICRP-48 and ICRP-60 on dose conversion factors

In the evaluation of risk that was carried out, dose conversion factors based on those of the ICRP were used (Nosske et al, 1985; Henrichs et al., 1985). In her publications 48 (ICRP, 1986) and 60 (ICRP, 1991) the ICRP presented new recommendations on respectively the metabolism of plutonium and related elements and on new tissue weighting factors. The expression for the effective dose equivalent is replaced by effective dose, when the new tissue weighting factors are applied. These changes were not yet included in the evaluation of the European risk map. In this appendix the doses refer to effective doses, and effective doses are compared relative to the effective dose equivalents used in the main calculations of the report.

The new ICRP recommendations result in new dose conversion factors that deviate from the earlier ones as far as radionuclides were taken up by a specific organ (Jain et al., 1992). Thus for ingestion, dose conversion factors for the isotopes of iodine have increased with 60 to 70 % partly due to the change of weighting factor of thyroid from 0.03 to 0.05, where the thyroid is the main contributor to the effective dose. Isotopes of cerium have values that are 30 to 45 % higher than the old ones because of the higher weighting factor from 0.06 to 0.12 for colon, which contributes more than 80 % to the effective dose. For the same reason the Np-239 dose conversion factor increased with about 30 %. For Np-237 a decrease of about 50 % occurred due to the lowering of the weighting factor for bone surfaces from 0.03 to 0.01 (factor of 3!), where this tissue is still responsible for about 30 % of the dose in the new situation. The plutonium and americium isotopes have their dose conversion factors decreased by 40 to 50 % for the same reason as was the case with Np-237. For neptunium, americium and plutonium a relatively minor part of the change mentioned is due to the shorter retention half lives in bone that were adopted by the ICRP (1986). Most of the remaining radionuclides have new dose conversion factors that are within ± 20 % of their former values. For inhalation of radionuclides similar changes have occurred in those dose conversion factors. One of the more important radionuclides in this study, Cs-137, has its dose conversion factors changed with only -3.5 % for inhalation (class D); no change occurred for ingestion.

Because of the changes that occurred for all radionuclides, the technique that was used to reduce the number of important radionuclides to be modelled might have resulted in other contributions, of the radionuclides considered, to the effective dose. However, because of the relative changes that occurred, only small changes are expected for the overall dose calculations, except for the relative contribution of I-131. In chapters 4 and 5 the influence of changes in I-131 dose conversion factors was estimated (see table 5.1)