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RADIOECOLOGICAL ASSESSMENT OF THE WYHL NUCLEAR POWER PLANT

May 1978
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On the basis of investigations carried out since the preparation of this assessment, the following substantial corrections to the data employed for the computation of dosage seem appropriate:

The transport factor of 15 (15 pCi/kg plant, fresh/pCi soil, dry) for potatoes specified in Table 7.1.3.-2 was taken from the data of (Herbst, W., 1976). A check showed that although this value was indeed given in the data of (Herbst, W., 1976) for potatoes, the cited source (Schaeffer, R., IAEA-AEA-SM-184/22, 1974) was incorrectly reproduced there.

In accordance with the present state of our knowledge the transfer factor is to be assumed to be in the range from 0.023 to 0.16 (IFEU, 1978). The results of computations for exposure by consumption of potatoes are reduced by this fact.

In addition, new aspects have come out in the evaluation of cobalt emissions. As explained in (Bruland, W. et al., 1979), in radioecological investigations the incorporation of cobalt isotopes in vitamin B_{12} must be taken into consideration, which leads to radiation loading several orders of magnitude higher in the consumption of contaminated animal foodstuffs.

Similarly, the Co transport factor of 0.0094 (pCi/kg plant, fresh/pCi/kg soil, dry) no longer corresponds with the state of our knowledge. Without site-specific studies one must conservatively assume a value higher by about two orders of magnitude. Thus, for example, Kirchmann et al. have determined transport factors of up to 1.9 for various Belgian soils (Kirchmann, R. et al., 1969).

After an initial assessment the result is that the corrections to be made overall would change the overall result of the dosage calculations only unessentially.

1. Mathematical Basis of the Assessment

To calculate the radiation exposure levels that are to be expected, the Department of Environmental Protection used a mathematical model whose principles are described in USNRC Guide 1.109, 1976 and Baker, D.A. et al., 1976. Most assessments of radiation levels in normal operation that have been made in the last few years for nuclear reactors in West Germany have been based on this mathematical model, e.g., the exhaust gas evaluation of the Institute for Reactor Safety (GRS, October 1976) and the waste water evaluation of the Bavarian Biological Testing Institute (BBV, 1976) for the planned nuclear power plant Kernkraftwerk Süd. In the ecological model the radiation dose by a nuclide that passes into food via the exhaust gas of a nuclear power plant is determined by simple multiplication of five quantities, namely, the strength of the emission source, the average long-term dispersion factor (meteorological dilution), the transfer factor (passage from air into food), the food consumption rate, and the dose commitment factor (biological activity of the radionuclides in the body). The organ dose obtained in this way represents the radiation load by a nuclide through the consumption of a food. If we wish to determine the total organ dose for all nuclides and all pathways of radiation exposure (foods), we must add the individual organ doses of all nuclides for all pathways of exposure. The equation for determining the radiation load via the waste water is the same, except that the dilution by the river water must be taken into account instead of meteorological dilution. The formulas on which the ecological mathematical model are based and the assumptions that were made are described in detail in the sections which follow. The various routes of exposure are shown in Fig. 1-1.

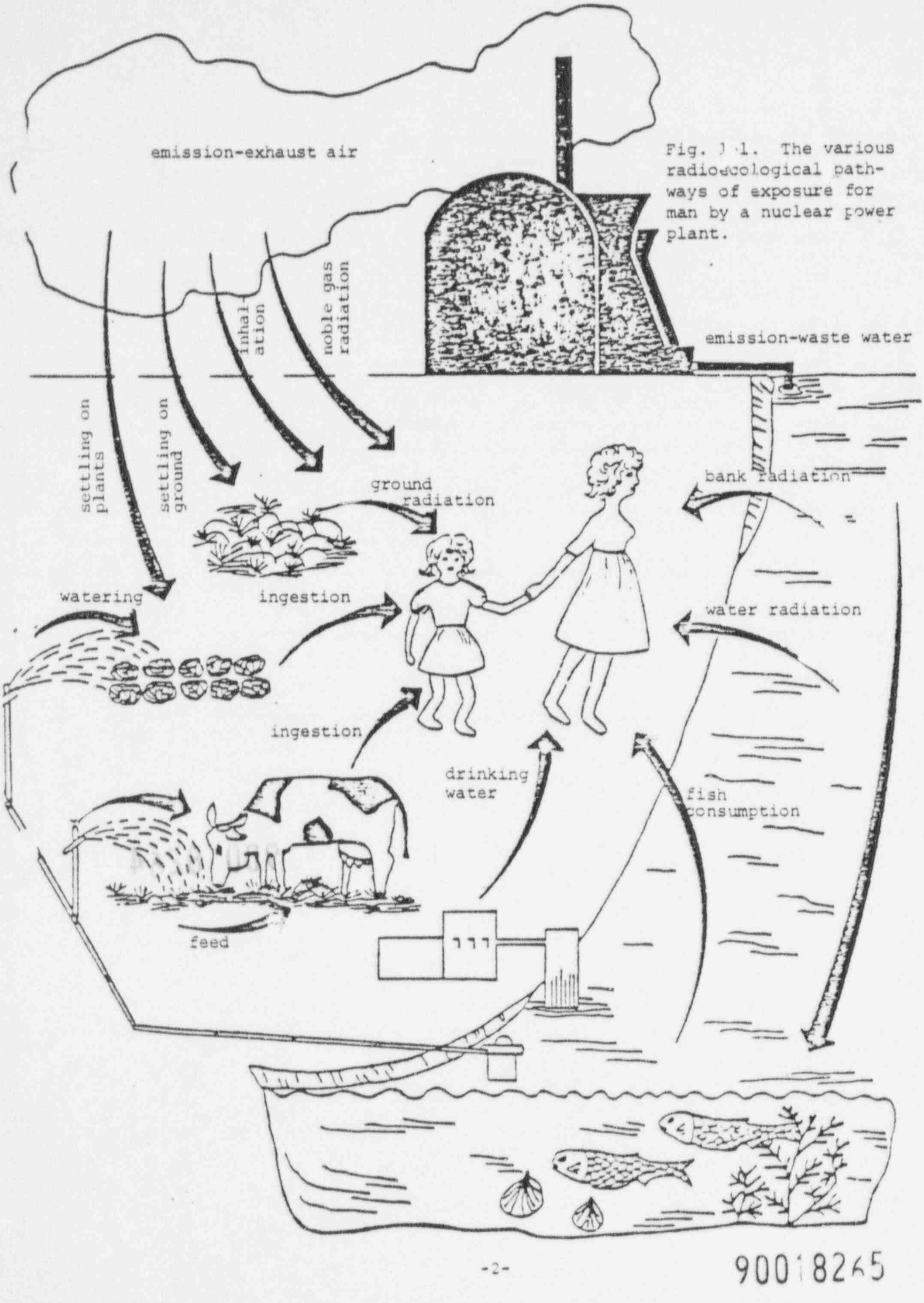

Fig. 1-1. The various radioecological pathways of exposure for man by a nuclear power plant.

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2. Emissions

2.1. Planned Radioactivity Emission of the Kernkraftwerk Süd GmbH Nuclear Power Plant (Wyhl) with the Exhaust Air

In the safety report of Kernkraftwerk Süd (Kernkraftwerk Süd GmbH, 1973) and in correspondence of Kernkraftwerk Süd GmbH to the Institute for Reactor Safety (KKS, 9/13/1976), Kernkraftwerk Süd GmbH proposed the following values per block for leakage of radiotoxic substances with the exhaust air of the planned Wyhl nuclear power plant:

noble gases	80,000 Ci/yr
aerosols (half-lives greater than 8 days)	1 Ci/yr
iodine-131	0.3 Ci/yr

Additional limiting conditions were that the daily radioactivity emissions must be less than 1% of the yearly emission values and the quarterly emissions must be less than 25% of these values. Double emission values would be used for two blocks.

2.2. Planned Radioactivity Emission of the Wyhl Nuclear Power Plant With the Waste Water

The following maximum emission values per block were proposed for the planned Kernkraftwerk Süd nuclear power plant (according to the BBV assessment, 1976):

1,600 Ci tritium per year

10 Ci other radioactive decay products

Our calculations were based on these values.

3. Preexisting Emission Loads at the Site of the Plant

3.1. Preexisting Loads from Radioactive Emissions in the Air

The preexisting load from radioactive emissions in the air

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(mostly from the Fessenheim nuclear power plant) could not be taken into account due to lack of adequate meteorological data and would have to be added to the values determined in this study.

As measurements in the upper Rhine region have shown, these emissions are not negligible (see the discussion in section 5.1.4).

3.2. Preexisting Loads from Radioactive Emissions in the Water of the Rhine

In order to calculate the radiation load, it is also necessary to know the preexisting load of the Rhine with radioactive substances. Several different emission sources must be distinguished:

1. nuclear plants
2. hospitals (I-131, Tc-99 etc.)
3. scientific institutes and industry (P-32, S-35 etc.)

Regarding 1: The preexisting loads due to plants located on the Rhine upstream from the site of the Wyhl plant are given in Table 3.1.

Table 3.1.: Preexisting loads from nuclear power plants located on the Rhine upstream from the site of the Wyhl plant (BBV assessment, 1976)

Plant	Type	Nuclides without tritium (Ci/yr)	Tritium (Ci/a)
Fessenheim I+II	PWR	80	4,000
Kaiseraugst KWK	BWR	5	500
Leibstadt KKL	BWR	5	500
Beznau KKB I+II	PWR	30	5,000
Würenlingen EIR	PWR	30	500
Gösgen KKG	PWR	5	5,000
Mühleberg KKM	BWR	10	500
Graben KKW	BWR	5	500

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Regarding 2: The values from the individual hospitals, most of which are on the French and Swiss side, are not available. According to information obtained from the National Bureau of Health in Berlin, the quantities of I-131 used in nuclear medicine under the conditions in that area average about 2.25 μCi per inhabitant per year. The Department of Environmental Protection therefore assumed a preexisting load of I-131 of 6.75 Ci/yr at Wyhl as a conservative estimate on the basis of the population of about 3 million in the Rhine drainage area above Wyhl). (According to model study Radioecology Biblis 3rd Colloquium Water Pathway 2).

Regarding 3: Additional preexisting loads by radioactive waste water from scientific institutes and industry could not be determined and would have to be added to the emissions specified above.

4. Nuclide Spectra

4.1. Composition of the Aerosol Emissions with the Exhaust Air

Previous experience with nuclear power plants shows that a large number of nuclides can escape as aerosols with the exhaust gas, namely, C-14, Cr-51, Mn-54, Fe-59, Co-57, Co-58, Co-60, Sr-89, Sr-90, Zr-95, Nb-95, Ru-103, Ru-106, Ag-110 m, Sb-124, Sb-125, I-131, Cs-134, Cs-137, Ba-140, La-140, Ce-141, Ce-144, and traces of alpha emitters such as Pu-239, Pu-240, Am-241 and Cm-242 (GRS, October 1976).

Isotopes that are particularly dangerous to human beings are those which pass readily through the food chain and at the same time exhibit a high level of radiotoxicity. This is especially the case with strontium-89 and 90, iodine-131, cesium-134, cesium-137, and to some extent with plutonium, so that these isotopes are generally very important.

I-131 is emitted mainly in gaseous form; only a small percentage, normally less than 10%, is emitted as a solid aerosol. The composition of the emitted aerosols, which, according to what has just been said, are relatively less dangerous for human beings, varies from reactor to reactor and can show strong variation with respect to time for an individual reactor. In West Germany measurements of the aerosol composition in the exhaust air of nuclear power plants have been available for only a few years (see, for example, M. I. Endrulat, I. Winkelmann, STH Reports).

For the calculations of radiation exposure via exhaust air, we used the nuclide compositions indicated in Table 4.1-1.

Table 4.1-1. Composition of radioactive aerosol mixtures in the exhaust air of nuclear power plants with light-water reactors.

Nuclide	Proportion	Half-life
Co 58	0.10	70.78 d
Co 60	0.15	5.27 a
Zn 65	0.10	244 d
Sr 89	0.01	50.5 d
Sr 90	0.01	28.5 a
Cs 134	0.15	2.06 a
Cs 137	0.40	30.1 a
Ce 144	0.08	284.8 d
Pu 239	$2\cdot10^{-4}$	24,400 a

The assumed nuclide composition differs from the composition assumed by GRS (October 1976) and GRS (1977) by a higher proportion of Cs-137. The following tables give the measured proportions of the Cs-137 nuclide in the nuclide composition of various nuclear power plants for various periods of time. It is apparent from the values compiled in these tables that the proportion of Cs-137 in the nuclide composition varies tremendously, so that the nuclide composition assumed in the calculations is necessarily somewhat arbitrary.

An important limitation that should be mentioned is that West Germany has had operational experience with large pressurized water reactors of the 1300-MW class only since 1974, while operating lives of 40 years are planned. It remains to be seen what effect the increasing wear of reactor components will have on the composition of the radioactive emissions.

Table 4.1-2. Proportions of the Cs-137 nuclide in the exhaust gas of various nuclear power plants (after Winkelmann, I. et al., 1975)

Stade nuclear power plant	1974	
First quarter	52.1 %	
Second quarter	20.0 %	
Third quarter	0.4 %	
Fourth quarter	0.0 %	
Annual mean	38.6 %	+
Lingen nuclear power plant	**1974**	
Third quarter	78.3 %	
Fourth quarter	78.3 %	
Annual mean	78.3 %	+
Gundremmingen nuclear power plant	**1974**	
First quarter	6.95 %	
Second quarter	30.39 %	
Third quarter	2.9 %	
Fourth quarter	23.42 %	
Annual mean	23.2 %	+ (relative to annual emissions)

Table 4.1-3. Proportions of the Cs-137 nuclide in 1974 at various nuclear power plants (after Winkelmann, I. et al., 1975)

Gundremmingen nuclear power plant	23.2 %
Stade nuclear power plant	38.63 %
Obrigheim nuclear power plant	0.17 %
Würgassen nuclear power plant	0.035 %
Biblis A nuclear power plant	2.4 %
Lingen nuclear power plant	78.3 %

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4.2. Spectrum of Nuclides in the Waste Water

The relative proportions of the various nuclides is critical in calculations of the radiation dose that is to be expected. Previous experience with nuclear power plants indicates that a large number of nuclides can escape with the waste water, namely, H-3, C-14, P-32, S-35, Ca-45, Cr-51, Mn-54, Fe-55, Co-57, Co-58, Fe-59, Co-60, Ni-63, Zn-65, Sr-89, Sr-90, Y-90, Y-91, Zr-95, Nb-95, Ru-103, Ru-106, Rh-106, Ag-110 m, Sb-124, Sb-125, Te-125 m, I-131, Cs-134, Cs-137, Ba-140, La-140, Ce-141, Ce-144, Pr-144, Pm-147, Eu-154 (after Commission of the European Communities, 1975).

The proportions of the individual nuclides are subject to tremendous variation in some cases. For example, in 1974 the proportion of strontium isotopes in the waste water of the KRB (Gundremmingen, BWR) was more than 57% (Commission of the European Communities, 1975). It is impossible to predict what changes will occur in the composition of the waste water emissions in the course of the expected operating lives of 40 years.

The isotopes which pass readily through the food chain and at the time time exhibit a high level of radiotoxicity are particularly dangerous to man. This is expecially the case for Co-58, Co-60, Zn-65, Sr-89, Sr-90, I-131, Cs-134 and Cs-137 in waste water, so that these radionuclides may be regarded as very important in the determination of radiation exposure.

We therefore selected a nuclide mixture that could be considered conservative on the basis of past experience. No distinction was made between the emissions from pressurized water reactors and boiling water reactors since more than 85% of the preexisting load of nuclides without tritium comes from or will come from pressurized water reactors.

Table 4.2-1. Nuclide Composition in Waste Water

Nuclide	Proportion
Co 58	0,05
Co 60	0.15
Zn 65	0.05
Sr 89	0,10
Sr 90	0.05
I 131	0.10
Cs 134	0.15
Cs 137	0.35

Sources:- Commission of the European Communities, 1975
- BBV Assessment (1976)
- Radiation Protection Commission (1977)
- Model Study Radioecology Biblis 3rd Colloquium Water Pathway 2

Of course, it is impossible to know whether the above assumptions are safe for the entire operating time of the reactor since it is possible that there will be changes in the compositions in the future.

4.3. Nuclide Composition of the Noble Gases

Even in trouble-free operation of a nuclear power plant, it is impossible to retain all radioactive substances in the reactor. Quantitatively, the radioactive noble gases are the most important atmospheric emissions because they are produced in unavoidably large amounts in the fission process, are very volatile, and cannot be bound by any chemical reaction.

Since the radioactive noble gases escaping from a nuclear power plant endanger the human organism to varying degrees (e.g., krypton-88 is about 1000 times more dangerous than krypton-85), the radiation load depends critically on the composition of the emitted noble gas mixture. The noble gas nuclide spectrum has never been

measured for high emission rates, e.g., those which occurred in 1973 in Gundremmingen or in 1975 in Lingen (model study). We must therefore rely on a theoretical calculation.

Noble gases are formed in the reactor core during the fission process. Since the processes involved in the production of noble gases during fission are not known exactly enough, an exact quantitative calculation of their production in the reactor core cannot be made. Consequently, it is necessary for us to rely on the liberation rates given in the KWS Safety Report (1973), which are based on measurements performed on the Gundremmingen boiling-water reactor (Holm, 1978; Schrüfer, 1974). According to these measurements, noble gases are produced in highly variable quantities. For example, Kr-85 is produced at a rate of 750 Ci/yr, while Kr-89 is produced at a rate of 2500 Ci/hr (in each case it is assumed that the reactor is operated at full load). Since noble gases decay after a certain amount of time, which depends on the half-life, the environmental hazard depends to a critical extent on how long the noble gases remain in the reactor plants.

If the noble gases are conveyed to the chimney through extraction systems (paths 2 - 5), the dwell times in the reactor plants depend on the particular space volume and on the exhaust rate. It was assumed that the noble gases become uniformly distributed in the space immediately after entrance. Of course, if the point of leakage is near the exhaust system, the dwell time is considerably reduced. Since the exact construction plans were not available to us, this case, which might result in elevation of the radiation load, could not be considered. It must be expected, therefore, that a large proportion of the gases escapes by leakage rather than by the prescribed path, namely, via the degassing system. It can be assumed that with increasing operating time the leakage or the activity emissions on the liberation paths become greater.

In accordance with Holm (GRS-8, 1978), the Department of Environmental Protection considered the following liberation paths in particular in its calculations:

1. Primary water purification plant -- exhaust system -- chimney

2. Steam generator heating pipe leakage -- condenser -- chimney

3. Leakage from the primary circulation -- plant buildings -- exhaust air -- chimney

4. Leakage from the primary water purification plants -- annular space air -- chimney

5. Leakage from the primary water purification plants -- auxiliary plant buildings -- chimney

Kausz and Spang (1973) show that about 50% of the noble gas emissions reach the outside through leakages while avoiding the exhaust gas system. Measurements of the National Bureau of Health (National Bureau of Health, STH-2/76) have found rates of even 45% to 98.5%.

In the calculation of the nuclide spectra performed here, the experts worked on the basis of the proposed activity emission of 80,000 Ci/yr, an annual power availability of the nuclear power plant of 80%, time delays of 40 days for xenon and 40 hours for krypton in the exhaust gas system, and estimated, free volumes and exhaust gas rates of the system parts of the Biblis type pressurized water reactor (from D. Holm, GRS-8, 1978). In regard to the percent variation of the proportions of noble gas that are released to the environment via the leakage, it is mainly the proportions that do not pass through the exhaust gas system that determine the radiation load because of the short time delays. Four leakage combinations calculated by us are given in Table 4.3-1, whereby, in accordance with experience (STH-2/76), the amount of noble gas released through the delay system was varied between 1.5% and 50% of the total amount of noble gas activity emitted. The nuclide spectra resulting from this distribution are given in Table 4.3-2. It is seen that there are practically no differences among spectra A to D despite the large variation in the percentage released through the exhaust gas system. There is no great change in the spectrum until all of the noble gas is released through the exhaust gas system (case E). However, a physically unrealistically high source strength would be necessary for this if 80,000 Ci/yr were to be released through the chimney.

The four noble gas nuclide spectra given in Table 4.3-2 are based on 0.8% (A), 2.7% (B), 0.9% (C) and 0.45% (D) of the noble gas activity produced in the fuel rods (source strengths). On the other hand, these values could also be regarded as leakage rates of the fuel element cladding, which, with complete (100%) continuous degassing of the reactor coolant in the degassing system and with the amounts of leakage in the other systems (leakage combinations A to D), result in a release rate of 80,000 Ci/yr. When it is considered that the empirical value for fuel element failure is 1% and is conservatively estimated at 10%, and that the primary coolant is actually degassed to as much as 99% (Kausz; Spang, 1973), then a release rate of 80,000 Ci/yr is perfectly realistic and probable and may even be exceeded in very unfavorable cases, especially with increasing fatigue of the material.

The hazard to man by noble gases is usually given by the mean dose factor, which depends essentially on the nuclide spectrum that is used. The bottom row of Table 4.3-2 shows the mean dose factor for gamma submersion for the calculated spectra. In cases A to D it is always about 13×10^{-5}.

The last spectrum (labeled SSK) was recommended by the Radiation Protection Commission (SSK, 1977) for ecological calculations.

It most closely resembles spectrum E, in which it was assumed that all noble gases escape via the exhaust gas system. A quick conversion shows that for a yearly release of 80,000 Ci both spectra are not merely optimistic or unrealistic, but rather patently absurd. Kr-89, which represents a serious radioecological danger and which has a high liberation rate, is not even represented in the SSK spectrum, while Kr-85, which is relatively undangerous due to its low dose factor, is present in a quantity of 2% or 1600 Ci/yr. But this is impossible, for a total of only about 750 Ci/yr Kr-85 is produced in the plant. The SSK is able to arrive at the indicated low mean dose factor of 2.6×10^{-5} only by making absurd assumptions such as this. The origin of the even lower value of 1.6×10^{-5} that is used in the study by the GRS could not be determined because no nuclide spectrum was given. In our further calculations we used the calculated mean dose factor of 13×10^{-5} rem x m^2/Ci x sec.

Table 4.3-1. Percent distribution of the leakage (various assumptions)

		A	B	C	D	E
1	Exhaust gas system	27.000000	78.000000	50.000000	1.500000	100.000000
2	Steam generator leakage	73.000000	21.000000	12.500000	24.600006	0.0
3	Primary circulation	0.0	0.100000	12.500000	24.600006	0.0
4	Annular space air	0.0	0.600000	12.500000	24.600006	0.0
5	Auxiliary plant buildings	0.0	0.300000	12.500000	24.600006	0.0

Table 4.3-2. Composition of the Noble Gases (in %) under various assumptions.

	A	B	C	D	E	SSR
KRYPTON 83 M	1.700000	1.600000	1.100000	1.100000	0.0	0.0
KRYPTON 85 M	1.500000	1.500000	2.300000	2.100000	0.0	2.000000
KRYPTON 85	0.0	0.0	0.0	0.0	17.600006	2.000000
KRYPTON 87	4.220000	9.300000	6.200000	0.620000	0.0	1.000000
KRYPTON 88	11.200000	11.920000	7.100000	7.100000	0.0	2.500000
KRYPTON 89	8.190000	9.000000	12.400000	12.100000	0.0	0.0
KRYPTON 90	3.600000	3.800000	6.500000	6.500000	0.0	0.0
KRYPTON 91	0.900000	0.850000	1.400000	1.500000	0.0	0.0
XENON 131 M	0.0	0.0	0.0	0.0	2.000000	2.000000
XENON 133 M	0.300000	0.100000	0.100000	0.100000	0.0	0.0
XENON 133	3.400000	3.900000	2.700000	0.270000	83.399994	80.000000
XENON 135	12.900000	12.500000	9.690000	8.600000	0.0	10.000000
XENON 135 M	6.100000	6.000000	5.500000	5.500000	0.0	0.0
XENON 137	12.500000	12.800000	16.800001	16.800001	0.0	0.0
XENON 138	20.000000	20.000000	18.500000	18.500000	0.0	0.050000
XENON 139	4.460000	4.600000	7.500000	7.900000	0.0	0.0
XENON 140	1.400000	1.500000	2.700000	2.700000	0.0	0.0
XENON 141	0.0	0.0	0.0	0.0	0.0	0.0
gamma dose factor	0.000129	0.000129	0.000127	0.000127	0.000011	0.000023

5. Abiotic Dispersion

5.1. Meteorological Dispersion and Settling via the Exhaust Air

5.1.1. Validity of Calculations on Meteorological Dispersion

The purpose of an assessment of this type is not to debate controversial scientific questions, but rather to give guarantees. In regard to atmospheric processes, the number of necessary parameters is large, and their reliability is indifferent in some cases. We therefore consider ourselves obliged, not only to give the values for the meteorological attenuation that are obtained from our computations, but also to estimate the corresponding ranges of error. This is always good practice in science and technology.

We are interested in calculating the radiation load that may be expected from the nuclear power plant during normal operation. In regard to the atmospheric dispersion of noxious substances, this means that we are interested only in long-term data, i.e., the planned nuclear power plant is regarded as a continuous emitter. However, since the output of emissions is subject to great fluctuation (from year to year and in the course of each year) (National Bureau of Health and other sources, e.g., Wiechen states that in the weeks following a reactor shutdown the release of I-131 into the atmosphere increases sharply), the weather data must be known for short intervals of time in order to allow accurate attenuation calculations, i.e., it is not sufficient to have mean values for long periods of time. However, this is impossible with the data presently available. It must be noted, therefore, that the elevation of the doses which is obtained by calculating with correctly superimposed values, compared to dispersion calculations with long-term averages, cannot be even approximately estimated.

Our experts give a realistic account of the meteorological attenuation rather than a cautious one. The conservatism that is necessary in judgments of this sort is guaranteed by furnishing all results with an error estimate.

5.1.2. Dispersion Class Statistics

The required four-dimensional dispersion class statistics (wind speed, wind direction, dispersion category, precipitation) for the planned site of the nuclear power plant (Wyhl) have not been prepared. In the opinion of the German Weather Service, statistics of this sort cannot be prepared. Our experts therefore have no choice but to rely on less suitable data. The applicability of this data to the proposed site of Wyhl is checked in each individual case.

The most accurate two-dimensional statistics from the southern upper Rhine region are those recorded by Weather Station No. 803 in Freiburg (Manier). These statistics were prepared in 1962 to 1966 and contain the wind speed in knots classified in 31 steps and the dispersion classes according to Klug's six categories. Statistics from other nearby stations had to be discarded because the data that was of interest to us was classified on too coarse a scale, so that the corresponding matrices are smaller in order than (31 x 6). Karlsruhe was used only for purposes of comparison because it is too distant. The distributions of the dispersion types are compiled in Table 5.1.2-1 for the three weather observation stations Freiburg Weather Bureau, Karlsruhe Nuclear Research Center and Breisach. The mean variation of the diffusion categories is determined from these values to be 25%. There is no reason to suppose that the proposed site of Wyhl deviates from the wind stability conditions in the Rhine Valley by more than this amount. The error inherent in the use of two-dimensional statistics that include the wind direction can be estimated at 1[illegible]%. This determination was the result of a comparison of the three-dimensional sectoral calculation for Jülich (Vogt, Geiss) and the corresponding two-dimensional, integral values.

Table 5.1.2-1. Distribution of the diffusion categories in the upper Rhine Valley (relative proportions at each station)

Diffusion Type	Very unstable	Slightly unstable	Indiff. Unstable	Indiff. stable	Stable	Very stable	Total
After Pasquill	A	B	C	D	E	F	
After Klug	V	IV	III 2	III 1	II	I	
Freiburg	0.021	0.150	0.166	0.290	0.191	0.182	1.000
Karlsruhe	1.022	0.074	0.139	0.397	0.218	0.150	1.000
Breisach	0.075		0.588		0.337		1.000

The wind direction analysis was undertaken at the airport in Bremgarten (southwest of Freiburg), whose location in the Rhine Valley is similar to that of Wyhl. Breisach is less representative for Wyhl because the Kaiserstuhl (highlands), which is nearby, lies in the principal wind direction (SSW) and thus deflects the wind. Bremgarten, on the other hand, like Wyhl, is located on the open plain. Table 5.1.2-2 compares the wind rose data of the two stations.

Due to the Breisgau, the Kaiserstuhl and the indentation of the Black Forest (mountainous region) at Freiburg, the wind direction conditions in Freiburg, unlike the general weather situation, are not applicable to Wyhl.

The distribution of the two wind roses in Table 5.1.2-2 shows a margin of error of 25%, which also covers the site of the planned nuclear power plant.

16.29% calms and 0.64% variable winds were measured in Bremgarten. They were distributed isotropically on the wind rose.

In the fourth dimension the precipitation distribution shows how the washout (or wet deposition) is distributed. Wet deposition causes values four times greater than dry fallout (von Rudloff). Table 5.1.2-3 shows the directions of dispersion for fallout and washout.

We disregarded the direction-correlated washout in our computation of the long-term dispersion factor, but we did draw qualitative conclusions from Table 5.1.2-3 in our treatment of wet settling (section 5.1.7).

5.1.3. Wind Speed, Weak Wind

The modern method of wind speed measurement with the revolving-cup anemometer permits measurement of wind speeds of less than 0.5 m/s. Due to the especially high levels of radiation in the immediate vicinity of a radioactive emitter under weather conditions with light winds, we consider it absolutely essential that light winds be accurately determined. Gradation of wind speed in knots (≈0.5 m/s) is absolutely necessary. In a computer study we demonstrated that a gradual increase in the size of the gradations of the wind speed scale results in underestimation of the radiation load from exhaust air by as much as 200% (!) (Department of Environmental Protection, 1978). After examining the meteorological data from the Kaiserstuhl region, we felt that this was another reason that the Breisach weather data should be discarded.

Table 5.1.2-2. Comparison of the wind roses of Breisach and Bremgarten in %

Wind Direction			East			South			West			North
sector	30	60	90	120	150	180	210	240	270	300	330	360
Bremgarten	9,35	2,97	2,84	1,09	3,94	12,19	25,98	6,22	2,85	1,52	5,50	8,62
Breisach	20,1		4,4		6,3	30,6		14,2	4,5		5,7	14,0

Table 5.1.2-3. Bremgarten wind rose as a function of the type of weather

	direction	30	60	90	120	150	180	210	240	270	300	330	360	var	calm
	average	9,4	3,0	2,8	1,1	3,9	12,2	25,0	6,2	2,9	1,5	5,5	6,6	,6	16,3
FALL OUT	fair weather	8,0	2,4	2,2	0,9	3,5	14,8	33,4	9,2	3,6	1,6	5,1	8,5	,7	6,4
WASHOUT	fog	10,9	3,3	3,0	1,3	1,9	3,5	4,6	1,9	1,8	1,3	9,0	8,9	0	48,6
	rain	5,6	1,6	2,0	0,7	3,4	17,1	36,9	7,2	2,9	1,7	7,2	6,6	0	7,0
	snow	14,2	2,7	1,3	1,2	1,9	6,5	16,8	3,1	1,7	1,4	14,7	11,5	0	23,0

In order to develop an idea of the variance of the wind speed for the Wyhl site, we compared the frequency distributions of the wind speed at the Freiburg and Bremgarten stations (Table 5.1.3-1).

Table 5.1.3-1. Comparison of the wind speeds according to the Beaufort wind scale. Frequency of occurrence per Beaufort force in %.

wind speed	in Beaufort	0, 1	2	3	4	5	6
	in m/sec	up to 1,5	1,6-3,3	3,4-5,4	5,5-7,9	8,0-10,7	>10,7
Bremgarten		41,7	21,2	17,9	12,8	5,1	1,3
Freiburg		41,2	30,3	14,7	9,5	3,3	1,0

A mean error of 12% between these stations can be calculated from these values.

The mean annual wind speeds at Bremgarten (3.0 m/s) and Freiburg Weather Station (2.3 m/s) approximately agree with this range of error. The percentage of weak-wind weather conditions for our dispersion calculations was by no means overestimated. On the contrary, it may be assumed that it is still too low since the data was recorded in the form of hourly means. In other words, light winds that last for less than one hour are averaged out. However, since there is visible drift (e.g., of a cloud of smoke) starting at Beaufort 1, and the maximum receiving point is reached in about 20 minutes at a wind speed of 0.5 m/s and in only 10 minutes at a wind speed of 1 m/s, recording wind data at one hour intervals is inadequate for calculation of the radiation load in the vicinity of a nuclear power plant. The wind speed measured at ground level (the measurements are generally made at a height of 10 m, although at Freiburg they are made at a height of 22 m above the ground) must be calculated high to about twice the emission height. The entire wind profile obtained in this way must be averaged in order to obtain an appropriate dispersion rate. Various formulas are possible for this integration over the wind profile. They differ in result by as much as 20%. According to Vogt (1977), previous experimental studies provide no information about the suitability of the integration models. Our experts proceed on the basis of a physical argument and regard the lower half-cone as relevant for the reflection of the exhaust gas plume on the ground and for the concentration near the ground according to the principle of superposition. They therefore integrate the wind speed from the ground ($\geq$ 10 m) to the height of emission.

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5.1.4. Vertical Dispersion

The vertical dispersion of the cloud of exhaust gas is limited not only below by the ground, but also above when temperature inversions are present. Surface inversions were investigated in the Rhine Valley at Karlsruhe and Strasbourg. The frequency of occurrence of surface inversions was obtained from Kleiss (Table 5.1.4-1.

Table 5.1.4-1. Inversion frequency in the Rhine Valley in %.

inversion frequency total per year	surface inversions ≦ 200 m	surface inversions ≦ 400 m
Karlsruhe	31,8 %	42,5 %
Straßburg	14,6 %	42,9 %

Allowance for inversion conditions in determining the dispersion of noxious substances is handled in various ways. While a general factor of 2 was originally used (BMBW, 1972), this factor is now rejected as too conservative. Therefore, our calculations need to be more realistic. Stable or very stable weather conditions occur in 37.3% of all cases at the site (Table 5.1.2-1). Surface inversions under a 400-m upper limit occur with similar frequency in the upper Rhine Valley, and a large percentage of these inversions have an upper limit of 200 m (Table 5.1.4-1). This suggests that the occurrence of the stable diffusion categories I and II (after Klug) and the occurrence of surface inversions should be equated. Therefore, in accordance with Lindackers et al. (1965), we assume a barrier layer correction factor of 2 only for weather classes I and II. This assumption is very optimistic and probably results in a serious underestimate of the radiation load via the air. In reality, simulation tests (Dunst, 1977) have shown that during an inversion the entire exhaust gas output can settle on the ground a few kilometers from the emission source within only a few hours, i.e., before changes in the stability conditions of the atmosphere can effect further attenuation. In this case no meteorological attenuation occurs for short periods of time, and when the inversion frequency is high (as is the case in the upper Rhine region), this can cause considerable change in the long-term dispersion factor.

Experimental studies conducted during the prolonged Rhine Valley inversion that occurred in the last week of October in 1975 (monthly weather report) revealed large quantities of Kr-85 emissions in Freiburg (Sittkus, Stockburger, 1976). The closest nuclear plant from which this Kr-85 can originate is the reprocessing plant in Karlsruhe 140 km away. A meteorological investigation of this incident is now being conducted by our department.

There is an urgent need for further studies on the long-term effects of the barrier layer intensification. In addition, in the wine-growing region around Wyhl the short-term radiation loads due to the frequent inversions in the fall, during the ripening of the grapes, must be investigated. These radiation loads can be considerable and may amount to as much as 100% of the emission source strength (see above); and this is true not only at the maximum receiving point, but also anywhere within a radius of 20 km, depending on the wind conditions.

5.1.5. Effects on the Meteorological Attenuation That Are not Considered

The following additional effects on the atmospheric diffusion cannot be reliably quantified at the present time. They are merely estimated as sources of error.

When light winds prevail, the gas is discharged with a greater effective stack height. For relatively cool sources, such as those in question here (about 10°C warmer than the ambient air), however, decreases in the effective stack height have also been observed. At greater wind speeds the exhaust gas plume may be drawn downward on the lee side of the chimney due to the sheltering effect of the chimney or of other buildings (downdraft effect). If we roughly estimate the uncertainty in the effective stack height as only about ± 10 m, the error in the dispersion calculation would be about 20%.

The initial attenuation of the exhaust air that occurs while the exhaust air is still inside the chimney is quite negligible at emission heights greater than 50 m.

The roughness of the topography is universally included in the empirically determined dispersion parameters and thus, strictly speaking, is applicable only at the location at which the parameters were measured (i.e., Jülich). Nester (1975) classifies the topography in four roughness categories and rearranges the diffusion classes according to roughness. At the Wyhl site we must consider a mixed topography with highly variable roughness, as is shown in Fig. 5.1.5-1.

Table 5.1.5-1. Topographical roughness in the region around Wyhl (10-km radius, inside West Germany)

roughness category	0	I	II	III	IV
roughness length in cm	0	0 - 3	3 - 30	30 - 150	> 150
vegetation height	0	up to 10 cm	up to 1 m	up to 8 m	greater than 8 m
topography	water, asphalt	grass, pasture land	bushes, fields, vineyards	orchards, woods	timberland, cities
approximate percentage in area around Wyhl	3 %	10 %	50 %	30 %	5 %

The sectoral distribution of the topography around the site according to roughness is not isotropic. As far as the distribution of the radioactivity is concerned, this also results in weightings per sector, depending on the roughness, due to the variable wind profiles. This factor cannot be taken into account at the present time. Therefore, without more extensive studies in this area we are unable to assess the error resulting from disregarding this factor. The height above ground level at which wind shear occurs also depends on the surface roughness. At low wind speed, even at heights less than 100 m (emission height!), there is a shift in the mean wind direction, which is determined from wind rose measurements near the ground, toward the right in the direction of the geostrophic wind. This effect causes a shift in the sectors of maximum radiation load since the wind shear is not taken into consideration in the calculation of the wind profile. Present knowledge does not permit quantitative evaluation of this problem either. Furthermore, we have not considered any emissions except emissions from the exhaust air chimney. The ever-present leakage from the coolant circulation causes radioactive emissions through the machine house roof (low emission height) and through the cooling tower (large emission height, maximum receiving point more distant from the power plant).

The preexisting radiation load of the evaporated Rhine River water contributes to the emission of radioactivity into the air from the cooling tower. Aside from the radioactive emissions, the cooling towers have an adverse effect on the dispersion of the exhaust air from the chimney due to their great height (1 1/2 times the height of the chimneys) and large surface. The deviations caused by this are totally unknown. The clouds of steam disturb the diffusion of the plume of exhaust gas and alter air flow in general since thermodynamic processes are now at work as well. The entire spectrum of effects of a cooling tower on the dispersion due to

a) the design features themselves (wind-shielding, downdraft etc.)

b) the thermodynamic effects caused by mixing of the steam clouds with the exhaust air

c) cooling tower emissions from leakage and preexisting radiation load of the water

cannot be estimated closely enough at this time despite careful analysis of the work that has been started in this area.

Long-range changes in climate and their effects on meteorological factors should also be mentioned, although this is not one of our concerns in this assessment. The nuclear power plant would generate as much waste heat as a large city. At the same time, the amount of water that would be vaporized would be equal to about two thirds of the amount of water vapor released to the atmosphere by Lake Constance. The cooling towers would release this heat and water to the atmosphere at a point compared to the large surface area of a city or a large lake such as Lake Constance. This would certainly result in long-term changes in the local climate of the Wyhl - Weisweil area, which in turn would affect the continuous meteorological data. Therefore, the validity of long-range dispersion calculations for the planned operation of the nuclear power plant over a period of many years on the basis of data gathered before the plant is placed in operation is rather doubtful (see also section 5.1.1).

5.1.6. Determination and Error Analysis of the Long-Term Dispersion Factors

The data used in the dispersion calculation was subjected to critical examination in regard to its reliability in several places. It is now necessary to evaluate the reliability of the mathematical model itself.

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The calculations were performed in strict accordance with the SSK's principles of calculation (BMI, 1977). The mathematical model described there is based on studies performed at the Jülich nuclear power plant. Since we have adopted these principles in their entirety, we see no need to describe the procedure of the calculations again. We have rejected the recommendations of the SSK only in regard to our treatment of inversions; our reasons for this are discussed above (section 5.1.4).

The mathematical model makes several fundamental assumptions that are incorrect. The question of whether or not the model even gives meaningful results will not be known until many years of experimental observation. In particular, the calculations are based on the following erroneous assumptions:

1. The model assumes a Gaussian distribution without justification.

2. The parameters are treated as mathematically independent, which is not true. Wind speed, wind profile, wind direction and weather class are interdependent parameters which also depend on other parameters, namely, pressure, temperature and humidity. However, the latter parameters are not even included in the calculations.

3. The mathematical model must be able to deal with turbulent atmospheric conditions. However, a mathematical description of turbulence (or at least a physical-phenomenological description) has not yet been found (Institute for Applied Mathematics, Heidelberg, 1978)

Consequently, the mathematical model of the SSK and other comparable models can only be regarded as rather unsatisfactory expedients that make it possible to obtain rough estimates.

For the purposes of our dispersion calculations we divided the horizon into 12 sectors of 30°C each. This is an appropriate division since the lateral width of the plume of exhaust air at the maximum receiving point is 15° to 38°, depending on the stability category.

The computation was performed by electronic data processing and had the following results:

A) <u>Maximum Receiving Point</u>

The maximum receiving point is located 500 m from the source in the 210° sector. The exact result is

(500 ± 50) m in sector (210 ± 15)°.

This gives a surface area of the receiving "point" of 2.6 ha in the direction NNE from the plant. The values for the critical receiving "point" are applicable in this area.

B) Concentration of Noxious Substances in the Air

The concentration of noxious substances at this point of maximum radiation load is

$$1.41 \times 10^{-6} \ s/m^3$$

in units of the source strength dose. This is the long-term dispersion factor. The behavior of the long-term dispersion factor in the principal wind direction is shown in Fig. 5-1. as an example of the results of our calculations.

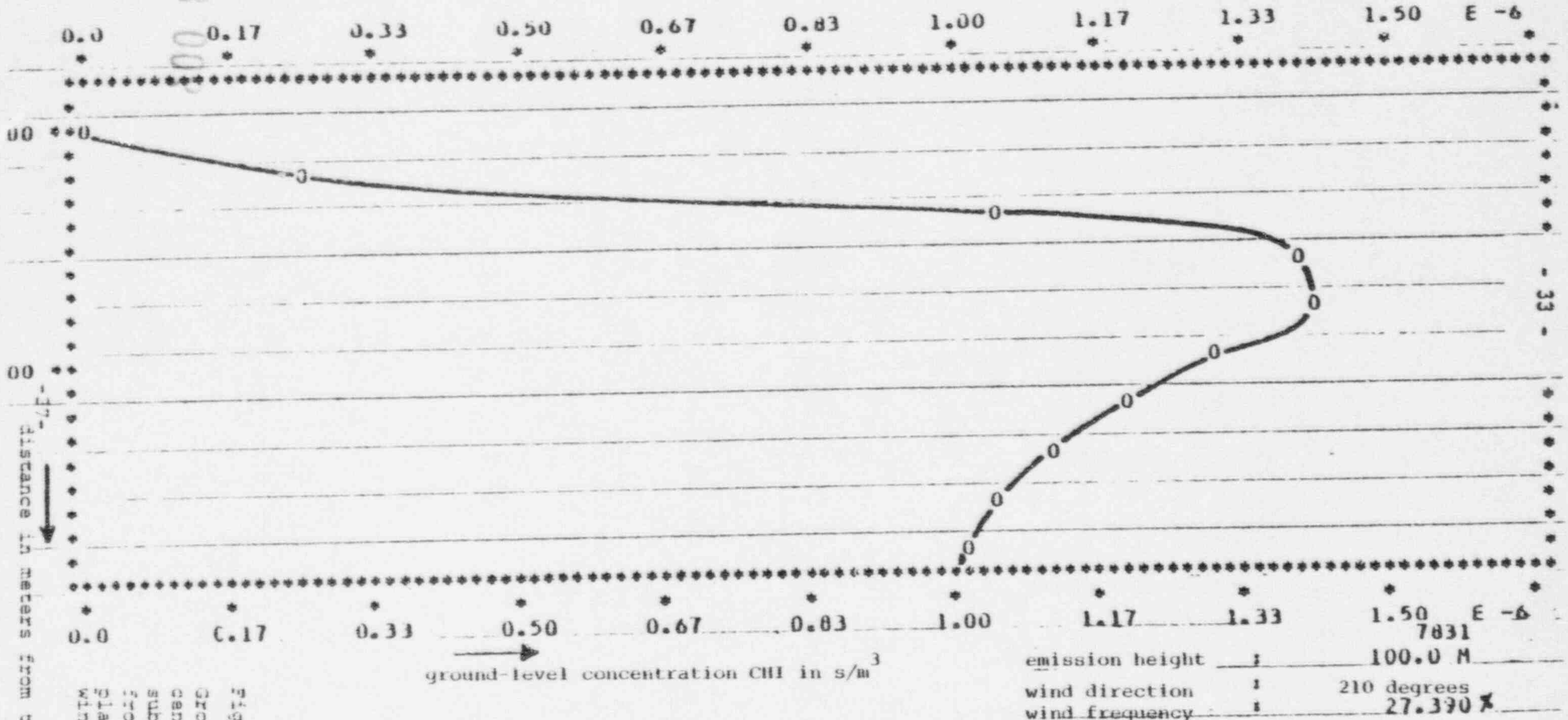

Fig. 5-1. Ground-level concentration of noxious substances emitted from the nuclear power plant in the principal wind direction.

C) Radiation Load in the Rest of the Surrounding Area

The subsequent calculations in this assessment of the radiation load via the various routes of exposure were performed with this value. In order to facilitate conversions to other areas of interest in the region around the emission source, we prepared a map (Fig. 5-2) showing the distribution of the radioactive exhaust air by relative isodose curves around the maximum receiving point (= 100%). Since the long-term dispersion factor enters linearly into the final dose, each individual result can be converted in this way to locations of special interest.

For flat terrain, in which the vertical distance to the emission source (mouth of the chimney) is the same at each point, there is only one maximum in the distribution of the radioactivity. For uneven terrain, such as the terrain in question, secondary maxima may occur. In the present case there is a second critical receiving point in Limberg, where the radioactivity is 50 to 100% of the maximum receiving point (see Fig. 5-2).

D) Gamma Submersion

We did not perform any calculations of our own for gamma submersion because our preliminary studies on allowance for the long-range gamma radiation from the neighboring sectors of the sector under consideration are still in progress. For the time being, therefore, we will adopt the data in the GRS assessment, in which the maximum gamma submersion 100 m from the source in a 30° sector in the principal wind direction NNE is

7.5×10^{-3} s/m^2.

For the maximum receiving point defined in A) at a distance of 500 m from the chimney, the following value is given:

3×10^{-3} s/m^2.

The GRS assessment includes no information at all about the mathematical models and assumptions that led to these results. Therefore, they must be considered very vague.

Until we are able to take into account the effect of the neighboring sectors on the gamma submersion, the error arising from failure to include it in our calculations can only be estimated. According to Hübschmann, Papadopoulos (1975), failure to allow for the neighboring sectors results in underestimation of the gamma dose by a factor of five at a distance of 100 m and by a factor of two at a distance of 500 m.

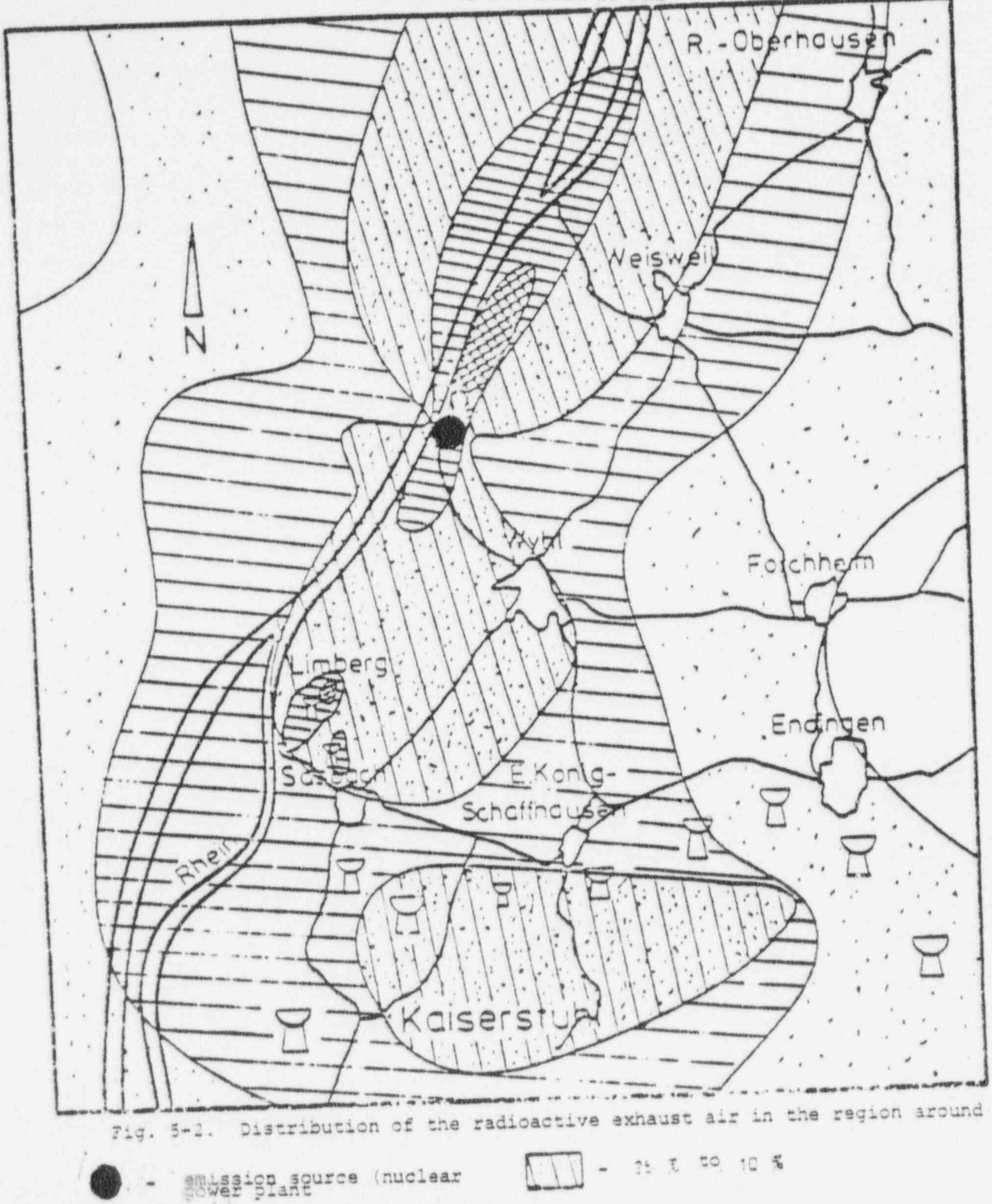

Fig. 5-2. Distribution of the radioactive exhaust air in the region around Wyhl.

- ● - emission source (nuclear power plant
- (symbol) - principal winegrowing areas
- O - maximum receiving point = 100%
- (cross-hatched) - 100 % to 50%
- (dense hatching) - 50 % to 25%
- (hatching) - 25 % to 10 %
- (horizontal lines) - 10 % to 5 %
- (dotted) - 5 % to 2,5 %
- (blank) - 2,5 % to 1 %

E) Calculation of Error Adjustment

The quantifiable errors contained in the calculations are compiled in Table 5.1.6-1.

Table 5.1.6-1. List of the quantitatively determinable errors in the dispersion calculation

Source of error	Discussed in:	Error range (relative to the final result, relative error)
Annual variations of the meteorological data, long-term means	5.1.1	1.00
Variations of the weather stability classes in the upper Rhine Valley, uncertainty in their determination	5.1.2	0.25
Variations of the wind rose in the region in question	5.1.2	0.25
Wind speed at ground level, anemometer measuring error	5.1.3	0.12
Type of wind profile to be assumed (power law), averaging of the wind profile	5.1.3	0.20
Error due to use of integral instead of sectoral diffusion class statistics	5.1.2	0.15
Error due to uncertain effective stack height	5.1.5	0.20
Only for gamma submersion: error due to failure to account for the radiation from neighboring sectors at the maximum receiving point	5.1.6	1.00

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We assume these errors to be mathematically independent. Since it is statistically improbable that all parameters subject to error will assume the extreme values within the error range, we obtain a relative error of 1.2 for the long-term dispersion factor and of 1.6 for the gamma submersion at the maximum receiving point (500 m) in accordance with the rules for error adjustment (both values are relative to the final result).

F. Conservative Values

No conservative assumptions were made in our meteorological dispersion computations; we tried, rather, to come as close as possible to reality. We shall now discuss the conservative approach in order to compare our realistically calculated results with the corresponding conservative values. However, only the realistic values are used in the remainder of our calculations in this assessment.

Conservative values may also be regarded as realistic values in the sense that they are the values at the error limit determined by the error adjustment computation. This quantitative definition of "conservative" follows from our interpretation of the purpose of this type of assessment (which was discussed in section 5.1.1), for when we give an error range we are merely stating that the actual values vary around the theoretically computed values and sometimes reach the unfavorable error limit.

Table 5.1.6-2 is an overview of the final results of the meteorological dispersion computations.

Table 5.1.6-2. Overview of the results of the meteorological dispersion computations.

	Realistic value	Relative error	Conservative Value
long-term dispersion factor (ground-level concentration of noxious substances at the maximum receiving point in units of the source strength dose)	$1.4 \cdot 10^{-6}$ s/m^3	1.2	$3.1 \cdot 10^{-6}$ s/m^3
maximum gamma submersion (100 m)	$7.5 \cdot 10^{-3}$ s/m^2	4.2	$3.9 \cdot 10^{-2}$ s/m^2
gamma submersion at the maximum receiving point	$3 \cdot 10^{-3}$ s/m^2	1.6	$7.8 \cdot 10^{-3}$ s/m^2

G. Explanation of the Discrepancy with the GRS Assessment

The relatively low values that are given for the long-term dispersion factor in the GRS assessment cannot be considered useful because the meteorological data used in the calculations was too coarsely graduated. We were able to reproduce the results of the GRS by dividing the wind speed scale in larger intervals. However, a coarse classification of the wind speed could be avoided with our data material. The values given in the GRS assessment may therefore be considered outdated. Since the calculation of the gamma submersion (which we have tentatively accepted) was based on the same inadequate meteorological statistics, it is probable that the gamma submersion was also underestimated.

5.1.7. Deposition of Radioactive Substances

The noxious substances contained in the air descend to the earth by gravitation and adsorption during dry weather (fallout); during wet weather (rain, snow, fog) they are washed out. The rate of settling multiplied by the emission source strength and the long-term dispersion factor gives the amount of activity that is deposited per unit time for a smooth ground surface (in $Ci/s\ m^2$)

We prefer to use the term "wet deposition" instead of "washout" because it indicates that settling processes that cannot be characterized by the dry settling rate occur not only during rain, but also during snow and fog.

Since the number of water droplets per unit volume is about 16 times greater in fog than in rain, the probability that aerosols will be picked up and subsequently deposited is greater in fog. The fog droplets also act as nuclei and cause rapid growth of the size of the aerosol particles, thereby increasing the falling speed (Liljequist). The two effects together increase the rate at which the radioactive particles settle out to such a great extent that the rates of deposition are about the same for fog and rain, despite the fact that the sinking velocity of the fog droplets is roughly 10 times slower than that of rain droplets. More precise information cannot be given without further investigation in this area.

The same is true for gases (I_2) because the larger total surface of all fog droplets results in greater solubility in water. The situation with snow is about the same as that with fog; however, even less is known about snow washout. Consequently, rain, fog and snow are treated together as "wet deposition".

Table 5.1.2-3 shows that wet deposition of the radioactive emissions of the nuclear power plant will be heavy in the principal wind

direction (from the SSW), i.e., the area to the NNE will have a high radiation load from wet deposition. However, other areas will also receive large amounts of radiation in this way, notably the areas to the S and SSW of the plant.

Due to the increased rate of deposition in fog, rain and snow, and due to the fact that fog and snow are very closely correlated with light winds, the area immediately around the nuclear power plant would be affected by greater radioactive contamination under these conditions (1 - 3 km from the plant; for a light wind of 0.5 m/s measured at a height of 10 m, any wind profile, and duration of the fog, rainfall or snowfall of 1 hour). In this case all directions would be affected to about the same extent.

Published values for deposition rates still vary by as much as 50% (see, for example, Vogt, 1970, or Voilleque, Pelletier). In regard to German conditions, the most appropriate value is that given by Ludwieg, who determined a mean annual deposition rate of about 1.3 cm/s. This value represents an overall value that takes into account both dry fallout and rain washout. We allow for fog and snow, not by correcting the above value, but rather by including an additional, independent error of 20% (73 days of fog per year in the Weisweil- Breisach area). Therefore, for aerosols (solid contaminants) we use the following value:

$$(1.3 \pm 0.7) \text{ cm/s}.$$

The direct deposition of I-131, which is emitted mainly in gaseous form, is described by GRS (1976) and Vogt, K.J. et al (1973) by a deposition factor f_A in $m^3/kg \cdot s$. This reflects the rate of deposition with respect to a vegetation density of 1 kg dry weight/m^2. According to Vogt, K.J. et al. (1974), almost twice as much iodine settles on clover as on grass. In our calculations we therefore used the mean value for grass and clover, which is 0.12 $m^3/kg \cdot s$ according to Heinemann, K. and Vogt, K.J. (1975). For leafy vegetables (lettuce) we adopted the value $f_A = 0.04$ $m^3/kg \cdot s$.

Everything we have discussed so far in regard to deposition is applicable for a ground surface with a low level of roughness, namely, a roughness parameter of $z_o = 0.03$ m (after Baumgärtner et al.). This is equivalent to a vegetation height of 10 cm and thus applies to grass, clover, certain vegetables etc. (See table 5.1.5-1.).

In regard to the deposition of radioactive material in vineyards, which is an important consideration in the area around Wyhl, the deposition constants given above can only be used for the ground of the vineyards. The roughness parameter for the foliage and grapes (vegetation height 1 m) is $z_o = 0.3$ m (Baumgärtner et al.) According to Gudiksen et al., this causes an increase in the rate of dry de-

position by a factor of 3 to 4. The same is true for wet deposition. Therefore, the following deposition rate applies to grapevine foliage and fruit and to other shrublike plants:

$$(4.0 \pm 3.3) \text{ cm/s}.$$

Table 5.1.7 is an overview of the results. The realistic values are used in our further computations; the conservative values are given only for comparison. Values greater than the conservative values can be ruled out on the basis of present knowledge.

Table 5.1.7. Deposition rates used in this study

Disposition of	Deposition on	mean value (realistic) in cm/s	mean error in cm/s	maximum value (conservative) in cm/s
Aerosol	grass, soil	1.3	0.7	2.0
Aerosol	grapevine foliage, grapes	4.0	3.3	7.3

5.2. Dispersion via the Waste Water

In regard to the attenuation of the radioactive waste substances, we assumed that the emissions mix immediately with 25% of the river water, and that it is this water of which further use is made. According to information of Mundschenk (BfG) (Model Study Radioecology Biblis, 3rd Colloquium, p. 37), this is a perfectly realistic assumption.

Measurements at the Biblis nuclear power plant showed mixture with about 20% of the water 1 km below the point of entry with fresh-water cooling and mixture with about 10% of the water with recycle cooling. 6 km below the point of entry the mixture was about 50% for fresh-water cooling and 33% for recycle cooling. The emissions introduced into the river thus remain in a small part of the river for a long time. As far as the Wyhl site is concerned, a more precise experimental study would be required before anything definite could be said about possible effects of the barrage downstream from Wyhl on the mixing processes.

The calculations consider only the emissions of one block. For two blocks (as calculated in the BBV assessment, for example)

the values would have to be increased accordingly.

In regard to the exposure pathway of fish consumption, we also considered the contamination of fish which swim chiefly at the cooling water outlet of the nuclear power plant. Experience at nuc-clear plants already in existence shows that increased numbers of fish are attracted to this area (and caught there) because of the oxygen enrichment of the cooling water. Therefore, this pathway represents a realistic source of contamination.

Exposure via consumption of fish was calculated for both possibilities (fish from the Rhine and fish from the cooling water outlet).

As in the BBV assessment (1976), our calculations were based on a cooling water discharge rate of 60 m^3/s (per block). We did not analyze the use of the cooling water for irrigation purposes or compute the higher radiation loads that one would expect from such use. The BBV assessment includes a conservative computation; if we wished to include this point in our assessment, we would expect a higher radiation load.

Volume Flow of the Rhine

According to the German Hydrology Yearbook (1971), the average volume flow of the Rhine at Rheinfelden is 1030 m^3/s (mean value for 1931 - 1970). The additional volume contributed by the Wiese and the Birse (small tributaries below Rheinfelden) justifies the assumption of a mean volume flow of 1050 m^3/s at Wyhl (the BBV assessment, 1976, uses the same assumption).

On the basis of these assumptions, the following values are obtained for the radionuclide concentration in the river water:

6.35 pCi/l	emissions excluding H-3
692.4 pCi/l	tritium
0.2 pCi/l	I-131 (medicine)

These values include the preexisting concentrations listed below:

5.15 pCi/l	emissions excluding H-3
499 pCi/l	tritium
0.2 pCi/l	I-131 (medicine)

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For our calculations of the contamination of fish that are caught in the cooling water outlet, the radionuclide concentrations are:

10.4	pCi/l	emissions excluding H-3
1,344	pCi/l	tritium
0.2	pCi/l	I-131 (medicine)

This nuclide concentration was used as the basis of calculation for the various exposure pathways. Naturally, the concentration can change significantly at low water level. According to the German Hydrology Yearbook, the average low-water volume flow of the Rhine at Rheinfelden is 464 m^3/s in the winter and 610 m^3/s in the summer; in the dry summer of 1976 the lowest monthly mean volume flow rate was 385 m^3/s.

Furthermore, waste water discharge is not constant, but rather reaches a maximum at the time of the fuel element change, which is usually done in the summer months. According to GANS (BGA) (Model Study Radioecology Biblis, 3rd Colloquium, Water Pathway, p. 27), 40% of the year's activity was emitted during the 3 to 4 weeks of the fuel element change at the nuclear power plants in Stade and Obrigheim, and in 1976 in Stade 60% of the year's activity was discharged during the 4-week changing period. If the river water is used for irrigation or other purposes during this period, the radiation load is many times greater than the load determined in this assessment.

Low-water levels can also result in higher nuclide concentrations in the river water.

6. Passage of Radionuclides into Agricultural Products

6.1. Transport Factors for the Passage of Radionuclides from Soil into Plants

The transport factors for the passage of individual nuclides from the soil into plants are among the most important factors in the determination of radiation exposure by ingestion of foods. The following exposure pathways must be considered:

- ingestion of vegetable foodstuffs-exhaust air pathway
- intake of other vegetable matter (e.g., tobacco, tea, coffee) - exhaust air pathway
- forage plants - ingestion of animal products - exhaust air pathway
- irrigation - consumption of vegetable products
- irrigation - forage plants - ingestion of milk and meat

The unit of the transport factors is pCi/kg fresh plant per pCi/kg soil. In order to obtain a realistic estimate of the transport factors that should be used, we performed an extensive literature search. Pertinent information from the following journals (from about 1950 to the present) was analyzed: Nature, Science, Health Physics, Plant and Soil, Soil Science, Radiation Botany, Journal of Agricultural and Food Chemistry, various USAEC reports and conference reports of the International Atomic Energy Agency.

The analysis of this information revealed that the transport factors used and recommended by the TÜV, GRS, BBV and SSK are far too small in the most important cases, and that many of the experiments in which low transport factors were determined were performed under unrealistic conditions.

There are many natural factors that affect the magnitude of the transport factor, e.g., plant species, soil type, mineral content and water content of the soil, pH of the soil, biological activity of the soil, and the level of radioactive contamination of the soil.

The values that are recommended by the Radiation Protection Commission (SSK) are based for the most p .t not on experimental investigation, but rather are derived from a chain of references to the literature, at the end of which soil and plant concentrations are employed that have no relationship to each other (Boikat, U., 1977).

By division of these concentrations for stable isotopes from (Univ. of Calif., 1968) the USNRC obtained "transport factors" (USNRC, 1976) that were accepted by the SSK, the BBV, and in part by the GRS. The GRS supports its values with the incorrect and biased interpretation of the literature in (Fletcher, I.F. et al., 1971), in which recourse is sometimes had again to the same sources as in (USNRC, 1976). A more exact representation is to be found in (IFEU, 1978).

6.2. Comparison of the Transport Factors Used by the GRS and SSK with Published Values

In this section we shall compare the transport factors for the individual elements of the soil in plants as used by the GRS and recommended by the SSK with experimentally determined transport factors found in the literature. The only elements for which there is an extensive literature are strontium and cesium; very little has been published about the other elements. The elements will be discussed in alphabetical order.

6.2.1 Cerium

The fission products Ce-141 and Ce-144 are the principal representatives of the element cerium in the emissions of nuclear power plants (GRS, 1976). Their half-lives are 32.5 and 285 days, respectively (Seelmann-Eggebert, W. et al., 1974).

The transport factor recommended by the SSK is 0.0025. The GRS uses values between 0.0005 and 0.0068 in its assessments.

The literature contained two papers on the determination of transport factors for cerium (Romney, E.M. et al., 1954; Romney, E.M., et al., 1957). These experiments yielded the following ranges of values, depending on the soils that were used:

for bean leaves	between 0.012 and 0.144
for bean seeds	between 0.003 and 0.028
for oats (grains)	around 0.015
for radishes (root)	between 0.014 and 0.032

(Romney, E.M. et al., 1954;
Romney, E.M. et al., 1957)

The value recommended by the SSK and the largest value used by the GRS are at the lower end of the range of enrichment factors determined in these experiments. The even lower transport factors of 0.0005 and 0.00075 that were used in some cases by the GRS for pasture vegetation and leafy vegetables are many times smaller than the experimentally determined values.

6.2.2. Cobalt

The fis.ion products Co-57, Co-58 and Co-60 are the principal representatives of the element cobalt in the emissions of nuclear power plants (GRS, 1976). They have half-lives of 270 days, 71 days and 5.3 years, respectively. (Seelmann-Eggebert, 1974).

Again there were only two published papers that gave transport factors for the absorption of cobalt from soil. The GRS and the SSK use a value of 0.00094.

Grummit, W.E., 1975, gives the following values as cobalt transport factors for the edible parts of plants:

oats	0.015
rye	0.003
radishes	0.032
carrots	0.011
turnips	0.043
beans	0.010
potatoes	0.020
tomatoes	0.011

(Grummit, W.E., 1975)

Menzel, R.G., 1967, gives values between 0.02 and 2 as the range of variation of cobalt transport factors from the soil into plants. The transport factor for cobalt that is used by the SSK and GRS is thus at the lower end of the range of variation.

6.2.3 Cesium

In the case of the element cesium it is primarily the fission products Cs-134 and Cs-137 that appear in the emissions of nuclear power plants (GRS, 1976). They have half-lives of 2.1 and 30.1 years, respectively (Sellmann-Eggebert, W. et al., 1974).

For cesium the literature contains a fairly large number of works in which experimentally-determined transport factors are given for this element. The results of these investigations are given below.

Barber found transport factors of between 0.015 and 0.6 in pot experiments with ryegrass and various soils.

He observed a strong dependence of the Cs-transfer on the proportion of the organic material in the soil and its cation-exchange capacity (Barber, D.H., 1964).

Bergamini found a large temporal variation of the transfer factor of clover according to the stage of vegetation (Bergamini, P.G., 1970). In addition, the concentration factor was highest in the plants growing in soils with the lowest concentrations of Cs-137. The author specifies the range of variation of the concentration as 0.004 to 33 in the case of contamination of the soil before sowing. The extremely high value was found with low concentration of the Cs-137 solution added to the soil. If Cs-137 was added to the soil in later stages in the growth of the plants, lower transfer factors resulted, namely

- in the case of contamination of the soil
  in the young growth stage: 0.008-5.2
- in the case of contamination of the soil
  at the beginning of efflorescence: 0.014-2.8.

This is to be expected, for the cesium content of the plant increases with the time that is available to the plant for absorption.

Fredriksson and Eriksson investigated the dependence of Cs-137 transfer in clover on the addition of stable cesium and on potassium fertilization (Fredriksson, L., 1958). Without fertilization and without carrier addition, transfer factors of 0.018 to 0.2 were determined (according to the type of soil). The addition of stable Cs-133 caused an enormous increase in plant absorption.

. The variation of the transfer factors were, for the addition of ca. 0.8 mg stable Cs/kg soil: 0.032 - 0.34

ca. 4" " " " : 0.088 - 1.2

ca. 20" " " " : 0.019 - 3.4

ca. 100" " " " : 0.84 - 6.8

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Thus there seems to be only a very small absorptive capacity of the soil for cesium.

This finding of these two authors, however, is contradictory to the observations of other authors who established an increase in transfer with decreasing soil concentrations (Bergamini, P.G., 1970; Dahlman, R.C., 1976; Shartiz, R.R., 1975).

On the other hand, cesium absorption is severely limited by fertilization with potassium. With ca. 4 mg stable Cs/kg soil the following transfer factors were found:

without K-fertilization:	0.088-1.22
with ca. 20 mg K/kg soil:	0.042-1.0
" " 100 mg K/kg soil:	0.016-0.7
" " 500 mg K/kg soil:	0.008-0.096

500 mg K/kg soil corresponds to a potassium fertilization of about 1100 kg/ha. This quantity thus reduces the Cs-absorption by a factor of about 10. This finding is interesting because several other authors carried out Cs-transfer tests at comparably high fertilization levels. The values thus determined are of course considerably below possible outdoor values.

Garrett et al. determined the cause of the surprisingly high Cs-137 concentrations in milk from various regions of Florida (Garrett, A.R., 1971). In the control experiment with the soil from this region they found very high transfer factors:

various grasses:	4.8 - 7.4
white clover:	8.6

Because of these high concentration effects the Cs-137 concentration in the milk of the region was far above the national average. Computation with average soil/plant transfer factors would have given a much too low Cs-137 content in milk due to bomb fall-out and would not have been on the safe side.

Grueter reports on transfer factors of 16.5 in mushrooms (based on fresh weight) (Grueter, H. 1971).

Guliakin et al. determined transfer factors for various plants in pot experiments. The soil concentration of Cs-137 was very high, viz. 30 microcuries/kg (Guliakin, I.W., 1974).

The following transfer factors are given for three different soils:

wheat straw	0.018 - 0.38
wheat grains	0.0085- 0.65

potatoes	0.023 - 0.16
carrots	0.0025- 0.03
legumes(vegetative portions)	0.012 - 0.3

The highest values in each case apply to a sandy-loamy podsol with a relatively small humus component, and the low values apply to a leached, high-humus chernozem soil. The transfer factors given are relatively low. This is presumably due to the extraordinarily high soil concentration of Cs-137 and the high fertilization.

In Marckwordt and Lehr we read: "In a study under realistic conditions we found that transfer factors of Cs-137 from the soil into plants varied considerably, either because of the various soil types or because of environmental conditions (climatic conditions, seasonal effects, type of soil treatment), which it was not possible to test in most cases. If we make allowance for the fact that effects of this last type can only be taken into consideration by making estimates under favorable and unfavorable conditions, then this calls for an adequate margin of safety in the establishment of critical limits." (Marckwordt, U., 1971).

Starting with measured concentrations of Cs-137 in toto and as available to the plants in various soils, the authors therefore estimate upper and lower limits for the contamination of agricultural plants with Cs-137. So-called soil factors are specified; these are the concentrations in pCi/kg dry weight of the agricultural plant referred to the total deposit in nCi/m^2 of ground surface. If one also takes into consideration the mass of the soil at plowshare depth as 240 kg dry weight of soil per m^2 (general basis of computation of the BMI), then we obtain the following transfer factors:

Plant		Cs-137 Transfer Factor lower limit	upper limit
Cultivated land:	wheat	0.0026	0.61
	lucerne	0.038	0.45
	lettuce	0.0062	0.75
Pasture land:	grass	0.0034	1.1

In the case of grass, if one bases the transfer factors on the soil mass of 80 kg/m^2 that is realistic for pasture land, then one obtains values between 0.0032 and 0.35. These values reproduce the actual ratio of the concentrations of Cs-137 in the grass and in pasture soils (in pasture soils the deposited radionuclides are distributed to a soil depth of only about 5 cm because there is no plowing; for the same deposition, then, the soil concentration is three times as high as for cultivated land).

The transfer factors given in the above table can of course be used directly in model computations, for the General Bases for Computation of the BMI determine the soil concentrations of deposited radionuclides in principle on a soil mass of 240 kg/m^2, without consideration of the actual circumstances in the case of pasture lands.

Marckwordt and Lehr compare the estimated maximal concentrations of Cs-137 in plants with the greatest concentrations they measured. In accordance therewith, the estimated values for the upper limits are higher than the maximal measured values by a factor of 2. The authors therefore conclude that their upper estimates are suitable for conservative considerations because they are on the safe side.

Nishita et al. investigated the absorption of Cs-137 from a loam soil in clover (Nishita, H. 1958). In their pot experiments they found a time-dependent increase in the absorption of Cs-137.

	Transfer factors
1st harvest	0.014
6th harvest	0.03
9th harvest	0.062
average	0.03

The ninth harvest was not made until 516 days. The Cs-137 was of course dried-up into a fine fraction of the soil. Due to this a smaller proportion of the Cs-137 employed was available to the plant than could be expected under realistic conditions. The evidence of the Nishita experiments hence cannot be taken as representative of Cs-137 transfer under field conditions.

Sharitz et al. investigated cesium concentrations in plants from the environs of the American Savannah River Nuclear Power Plant in soils with low and with high cesium contamination (Sharitz, R.R., 1975). They found that the transfer factors depended on the concentration of cesium in the soil.

Plant Type	Sagittaria latifolia	Polygonum punctatum
Transfer factor with low soil concentration of Cs-137 (23.8-27.2 pCi/g)	4	2.2
Transfer factor with high soil concentration of Cs-137 (541-548 pCi/g)	0.14	0.12

At low soil concentrations there was an 18- to 30-fold increase in the transfer factor, as can be seen from the table.

D'Souza et al. determined the transfer of Cs-134 in pasture grass in a field experiment extending over 18 months (D'Souza, T.J., 1972). After the grass had been cut quite close to the ground a Cs-134 solution was sprayed on the pasture. The Cs-134 concentration in the pasture grass was then determined for four harvests, with the first harvest taking place after five months and the last after 18 months. Since in the original work only the original contamination of the ground surface is given

in microcuries per m^2, for the computation of transfer factors one must take into consideration the 240 kg/m^2 mass covering of the ground; see "General Bases of Computation" of the BMI of 1977. The transfer factors thus calculated can then be used in computations of radioecological models (after conversion to fresh weight of plants).

Soil type	Transfer Factors Harvest No.			
	1	2	3	4
sandy	1.4	1.4	0.66	0.68
acid brown earth alluvial	2.3	1.6	0.062	0.04
slatey	2.5	3.0	0.094	0.04
acid brown earth	1.4	1.4	0.66	0.7

Under these conditions the absorption of radiocesium in the grass decreases with time. The effect was particularly pronounced in soils with a medium clay content and low humus content, but was not pronounced in soils with low clay content and high humus content: here the cesium contents of the grass actually increased again after the third harvest. In each case, however, the transfer factors were higher than the value in the SSK Bases of Computation for the exhaust pathway by a factor of at least 20. This is all the more remarkable inasmuch as the experiments were carried out under very realistic conditions.

Wiechen investigated why milk produced on marshy soil always contains substantially more Cs-137 than milk from normal soils (Wiechen, A., 1972). He found that in the low-potassium marshy soils Cs-137 is absorbed by the plants to a much greater extent than on other lands used for agriculture.

Soil	Transfer factor
marshy	0.17
dry sandy coastal region of N. Germany	0.12

The above data take into consideration that the water content in marshy soils is ca. 75%, whereas in soils of the dry, sandy coastal region of Northern Germany it is ca. 20% (Wiechen, A., personal communication, 1978). The author points out the significance of his findings in case of emergency: "If for any reason fall-out nuclides repeatedly drop on these areas [marshy lands, author's note], then there will certainly be a drastic increase in the cesium concentration in the milk so that it might become unusable for human sustenance. These marshy regions would then be ruled out for decades for milk production, which could have severe economic consequences for the rural population."

Polikarpov (1971) gives the following transport factors for grass in dependence on the various soils:

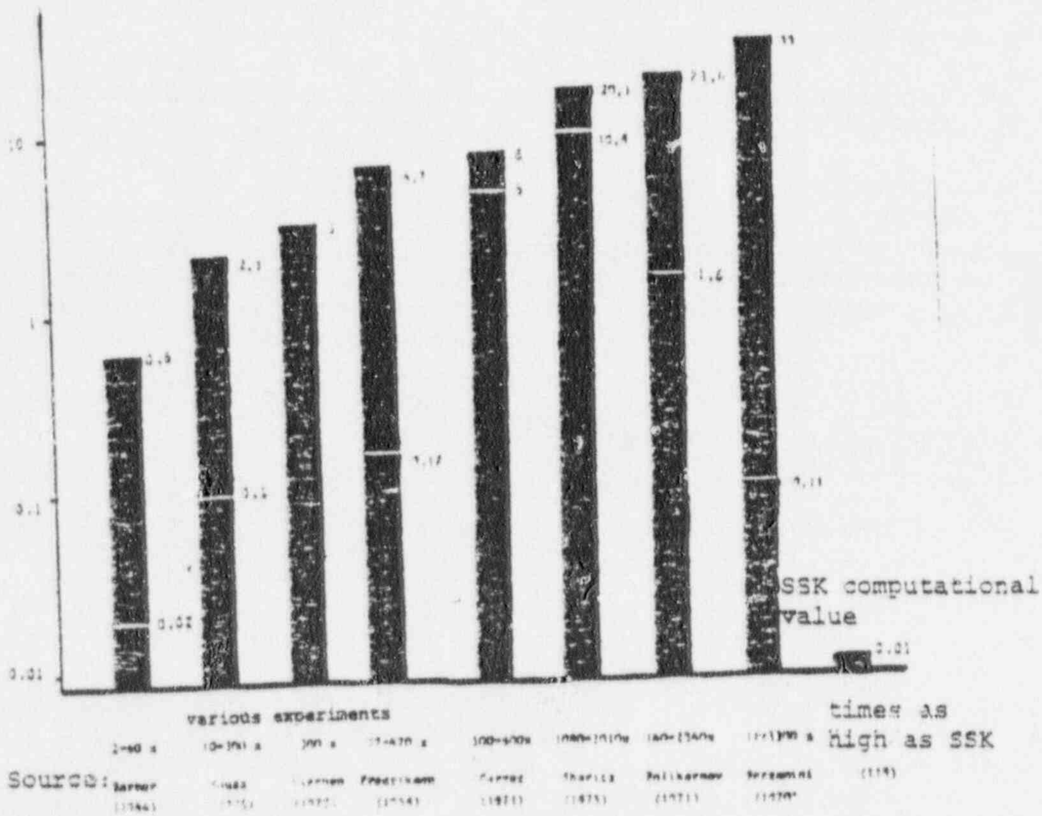
loamy sod-podzolic soils; chernozems	1.62
sandy-loamy sod; podzolic soils; sod-peat soils; silty-boggy soils	7.95
sandy sod podzolic soils	23.6

(Polikarpov, 1971)

In contrast, the GRS uses cesium-transport factors of between 0.00064 and 0.019. The SSK recommends a value of 0.01.

The following figure contrasts the experimentally determined transport factors for cesium in grass, clover, and leafy vegetables with the values recommended by the SSK.

Transport Factors for Cesium $\frac{\text{concentration (plant, fresh)}}{\text{concentration (soil, dry)}}$



COMPARISON OF TRANSPORT FACTORS FOR CESIUM IN PASTURE GROWTH DATA FROM VARIOUS PUBLICATIONS AND SSK COMPUTATIONAL VALUE

(Logarithmic scale) (* lowest soil concentration considered)

6.2.4. Iodine

The GRS and SSK use an iodine transport factor of 0.02 for all plants.

Cline, J. and Klepper, B. (1974) give values between 0.01 (barley) and 0.2 (radishes) for the iodine transport factor.

Menzel (1967) gives a range of variation of 0.02 to 2 for the transport factor of iodine.

Newton, H. and Toth, S. (1952) investigated enrichment factors for tomatoes and buckwheat as a function of the potassium chloride and potassium nitrate fertilization of the soil. They report the astonishing effect that the transport factor for iodine increases by about one power of ten when the concentration of potassium chloride and potassium nitrate in the soil increases (from 0.5 to 5 ppm). The values determined in the whole study ranged from 0.02 to 3.5.

Experiments performed by Cline, J. and Klepper, B. (1975) showed a similar effect of increase in the iodine transport factor, in this case with increasing soil concentration of stable iodine. These authors found transport factors between 0.01 and 120 (the higher values were obtained when the iodine concentration in the soil was increased to 10 - 15 ppm).

The iodine transport factor is especially important where the long-lived isotope I-129 is concerned (half-life 17 million years). This isotope is emitted mainly by fuel reprocessing plants. I-129 becomes enriched, in the soil in the course of the time and can result in considerable radiation exposure by ingestion following uptake by the roots of the plants. The short-lived isotope I-131 (half-life 8 days), on the other hand, which is emitted mainly by nuclear power plants, does not become enriched in the soil because its half-life is too short.

Comparison of the experimentally determined transport factors with the value used by the GRS and SSK shows that the latter falls at the lower end of the range of variation of possible iodine transport factors, which covers four powers of ten.

6.2.5. Manganese

The radioactive corrosion product Mn-54, which has a half-life of 312 days, is a relevant emission of nuclear power plants (GRS, 1976; Seelmann-Eggebert, W., 1974).

Jones, C.H.P. (1957) gives values between 0.125 and 0.675 for the manganese transport factor in oat grain. The higher values are obtained with low manganese concentrations in the soil.

Menzel (1967) gives values between 0.2 and 20 for the manganese transport factor in plants.

Prabhakaran Nair, K. and Prabhat, G. (1977) determined transport factors between 0.13 and 0.19 for corn.

The GRS and SSK use transport factors of 0.03 and 0.029, respectively, for manganese (GRS, 1976, and SSK, 1977).

6.2.6. Plutonium

Pu-238 and Pu-239 are the most relevant plutonium emissions of nuclear power plants and reprocessing plants (GRS, 1976). They have half-lives of 88 and 24,000 years, respectively (Seelmann-Eggebert, W., 1974).

The range of plutonium transport factors given in the literature is very large. The reason for this is that the plutonium emitted from nuclear power plants is usually present in a poorly soluble chemical form at first, so that it is not readily available to plants. With the passage of time, however, this poorly soluble form of plutonium can be converted to more easily dissolved forms that can be absorbed by plants; this conversion is probalby due to the activity of soil life. Lipton and Goldin (1976), for example, found that chelating agents, whether added to the soil with fertilizer or naturally occurring, cause a drastic increase in the uptake of plutonium by pea plants, namely, by more than three powers of ten.

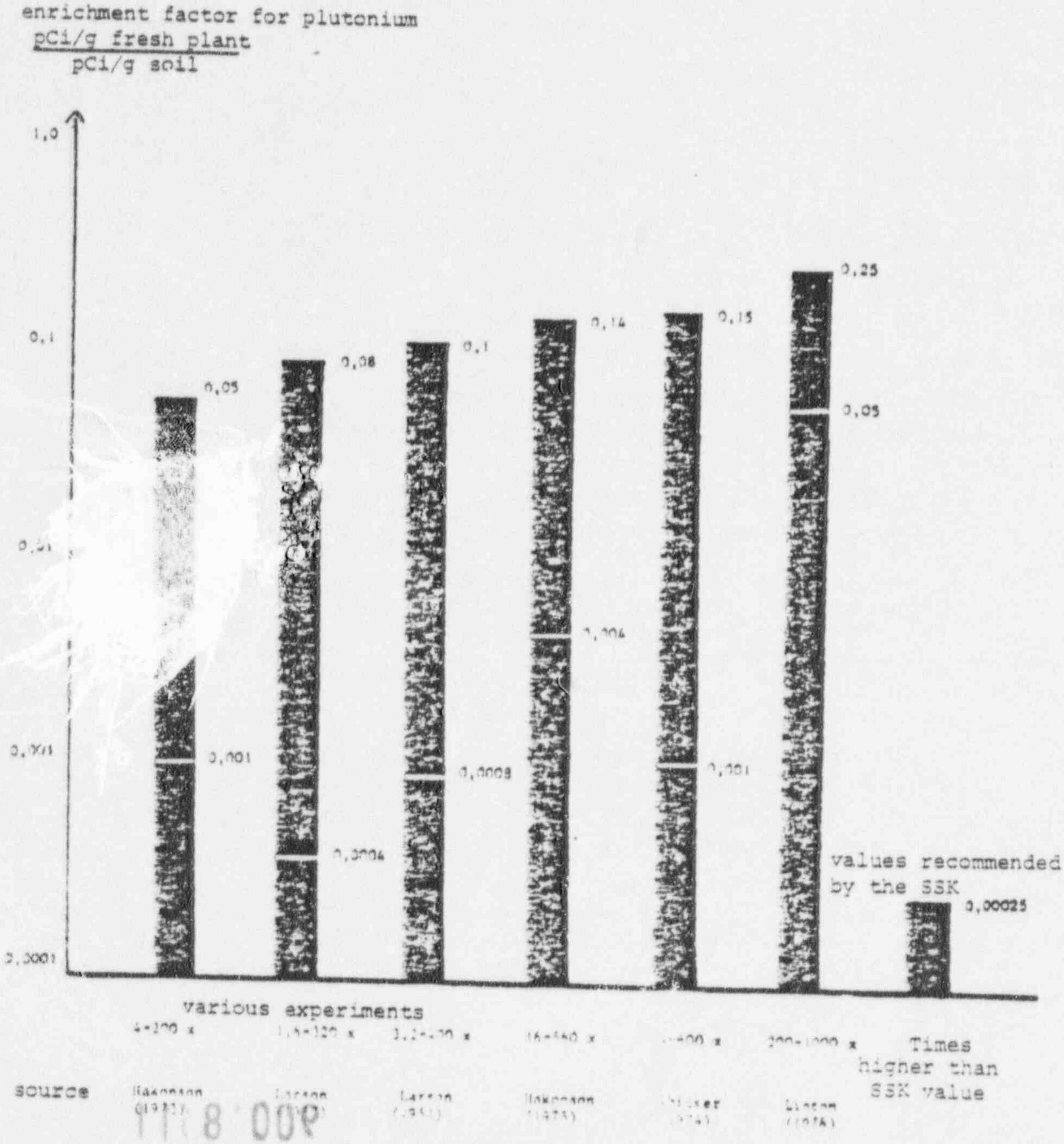
The GRS and SSK use a universal value of 0.00025 as the transport factor of plutonium for all plants. As the following published values show, this value is unrealistically low.

Larsson, K.H. (1951) found values between 0.0084 and 0.101 for plutonium transport factors. The mean value of various measurements was 0.025.

Larsson, K.H. et al. (1957) determined values between 0.00043 and 0.086.

Hakonson, T. E. and Johnson, L.M. (1973 I) studied various types of grass in the open country around a nuclear testing site 20 years after the last nuclear bombs had been detonated. They found transport factors between 0.03 and 4.7.

In an area of open country near Los Alamos where plutonium-containing waste had been dumped, Hakonson, T.E. et al. (1973 II) found values between 0.01 and 0.05 for grass and various meadow plants.



Comparison of the enrichment factors for plutonium in grass, clover and leafy vegetables: Result of various experiments and value recommended for purposes of calculation by the SSK (logarithmic scale).

Whicker (1974) measured plutonium transport factors between 0.001 and 0.15 in the open country around the Rocky Flats Nuclear Research Center. These measurements also revealed that the transport factors depend on the concentration: the lower the plutonium concentration in the ground, the higher the relative transport factor for plutonium from the soil to plants and small animals. It follows from this that the higher value is probably more realistic for the region surrounding nuclear power plants and reprocessing plants than the lower value.

Hakonson, T.E. (1975) determined transport factors between 0.004 and 0.14 in field experiments. The author states that higher transport factors are obtained under field conditions than in laboratory experiments. This may be related to the biological activity of the soil.

Lipton, W.V. and Goldin, A.S. (1976) determined transport factors between 0.05 and 0.25 in laboratory experiments on peas with addition of chelating agents as described above. Values around 0.00035 were determined when chelating agents were not added. Since the transport factors determined under field conditions are closer to the values determined in the laboratory experiment involving addition of chelating agents, it may be concluded that in nature plutonium is affected by processes (such as chelation) that facilitate uptake by plants in the course of time.

The preceding graph compares the measured transport factors for plutonium and the value used for computation by the GRS and SSK.

6.2.7. Strontium

The radioactive isotopes strontium-89 and strontium-90 are discharged into the atmosphere in the emissions of nuclear power plants (GRS, 1976). Their half-lives are 50.5 days and 28.5 years, respectively (Sellmann-Eggebert W., 1974).

A large number of papers have been published that deal with transport factors of strontium from soil to plants.

There are several works on strontium transfer in plants by Andersen.

In a work published in 1963 the absorption of strontium by ryegrass (Lolium perenne), clover (Trifolium pratense), and oats (Avena sativa) from various soils is studied (Andersen, A.J., 1963). The concentration experiments were carried out in pots using 20 different soils. There the following transfer factors were found:

ryegrass	0.16 - 0.17
clover	0.36 - 2.9
oat grains	0.026- 0.1

The author also investigated the influence of various fertilizers on the transfer of strontium into the plant. The addition of calcium increases or lowers the transfer factor according to the chemical form of the fertilizer. Nitrogen fertilization increases the transfer factor somewhat, and the addition of potassium reduces it slightly. Another important finding of this work was that the Sr-content of the plants is negatively correlated with the exchangeable Ca-content of the soil. Andersen thus found clearly higher transfer factors in sandy soils with a low pH and Ca-content than in loamy soils with a high pH and Ca-content.

Evans and Diller determined transfer factors for a series of plants in a loamy soil (Evans, E.J., 1962). The soil concentration in pot experiments was ca. 20 microcuries Sr-90 per kg.

leafy vegetables	0.24 - 1.1
grasses	0.15 - 0.36
wheat grains	0.022 - 0.6
potatoes	0.057
clover and legumes	0.68 - 1.1
leguminous seeds	0.18 - 0.7
other vegetables	0.12 - 0.38
tobacco	0.78
carrot	0.28 - 0.38

The intensive NPK fertilization in the experiments might have led to a reduction in the transfer factors (v. Collins, H.A., 1969). Also it cannot be ruled out that the very high Sr-90 concentrations in the soil contributed to the low transfer factors.

Garrett et al. determined the following transfer factors in hothouse experiments on sandy soils of Florida:

grass	0.78 - 2.2	
clover	3.0	(Garrett, A.R., 1971)

How we are dealing with soils with a very low cation-exchange capacity and low pH-values. The soil concentration in this experiment was very low, at 0.4 microcuries Sr/kg.

For leafy vegetables McIntyre gives transfer factors of 0.8 - 3.6 for stable strontium (McIntyre, D.R., 1977).

Menzel, R.G. (1967) gives the range of variation of strontium transport factors as 0.2 to 20.

Polikarpov (1971) gives values of between 0.7 and 2.3 for the transport factor for strontium in grass.

A work by Nishita et al. is concerned with the transfer of Sr-90 from a loamy soil into clover (Nishita, H., 1958). In the pot-experiments the transfer factors were between 0.96 and 1.2. The authors indicate that earlier experiments with soil concentrations 15 times as high with the same soil had shown a much lower Sr-90 absorption in clover.

Romney et al. investigated the concentration of Sr-90 in plants in pot experiments in three different soils (Romney, E.M., 1954). At a soil concentration of 2.7 microcuries/kg the transfer factors were as follows:

beans (leaves)	1.6 - 6.6
beans (fruit)	0.35- 3.5
barley (fodder)	0.54- 4.6
barley (grain)	1.6
radish (tip)	1.8 - 14
radish (root)	0.38- 21

The Sr-absorption in the plants was highest in the soil with the lowest pH and the lowest content of exchangeable calcium and vice versa. The Sr-90 content in barley was lower in the grains than in the fodder.

In 1959 Romney et al. concerned themselves with the question of how the addition of stable strontium affects the transfer of Sr-90. Three soils were used at a concentration of 2.7 microcuries/kg. The transfer factors in clover were between 3.9 (first harvest after 250 days) and 2.6 (fifth harvest after 550 days). In beans values between 0.1 and 4 were found. These data are with reference to the above-ground parts of the plants (Romney, E.M., 1959). The Sr-90 content of the plants was correlated with the inverse of the content of exchangeable calcium in the soils. The addition of stable strontium at up to 5 meq/100g increased the Sr-90 transfer only slightly.

In a very realistic field experiment D'Souza et al. determined the behavior of Sr-89 in four different pasture soils. After they had cut the grass very close to the ground they sprayed 10 mCi Sr-89 per m^2 on the

ground (D'Souza, T.J., 1972). Two different series of tests were run in parallel: all pasture soils were covered with a transparent sheet of polyethylene to prevent the penetration of rain water into the ground. Alongside of this, tests were run without this covering, under completely natural conditions. The grass was harvested twice and the Sr-89 activity was measured. If one standardizes the specified contamination of the soils to a soil-mass cover of 240 kg/m^2, one obtains the following transfer factors:

	without cover	covered by sheeting
1st harvest	2.2 - 10	2.2 - 18
2nd harvest	1.1 - 4.2	2.2 - 10

The tests showed that especially in sandy soils with a low clay content one must take into account considerable elutriation of the strontium by rain.

In the case of soils with clay contents of 6 to 16% the plant concentrations were reduced little by elutriation of Sr-89 from the ground, on an average by some 20 to 30%. The transfer of Sr-89 was higher in soils with a high humus content and a low clay content than in soils with average contents of humus and clay.

The transfer factors in the above table are relatively high presumably because the running time of the tests was only about three months, which is not specified exactly in the text, so that the major part of the Sr-89 was still in the matting of grass. There, however, it is available to the plant to a greater extent than in the ground.

In pot-experiments Teufel investigated the transfer of Sr in various agricultural plants (Teufel, D., 1977). Because of the short vegetation period only the values for grass and clover are usable for a comparative study. Transfer factors were determined in various soils:

for grass	1.2 - 1.5
for clover	2.6 - 7.4

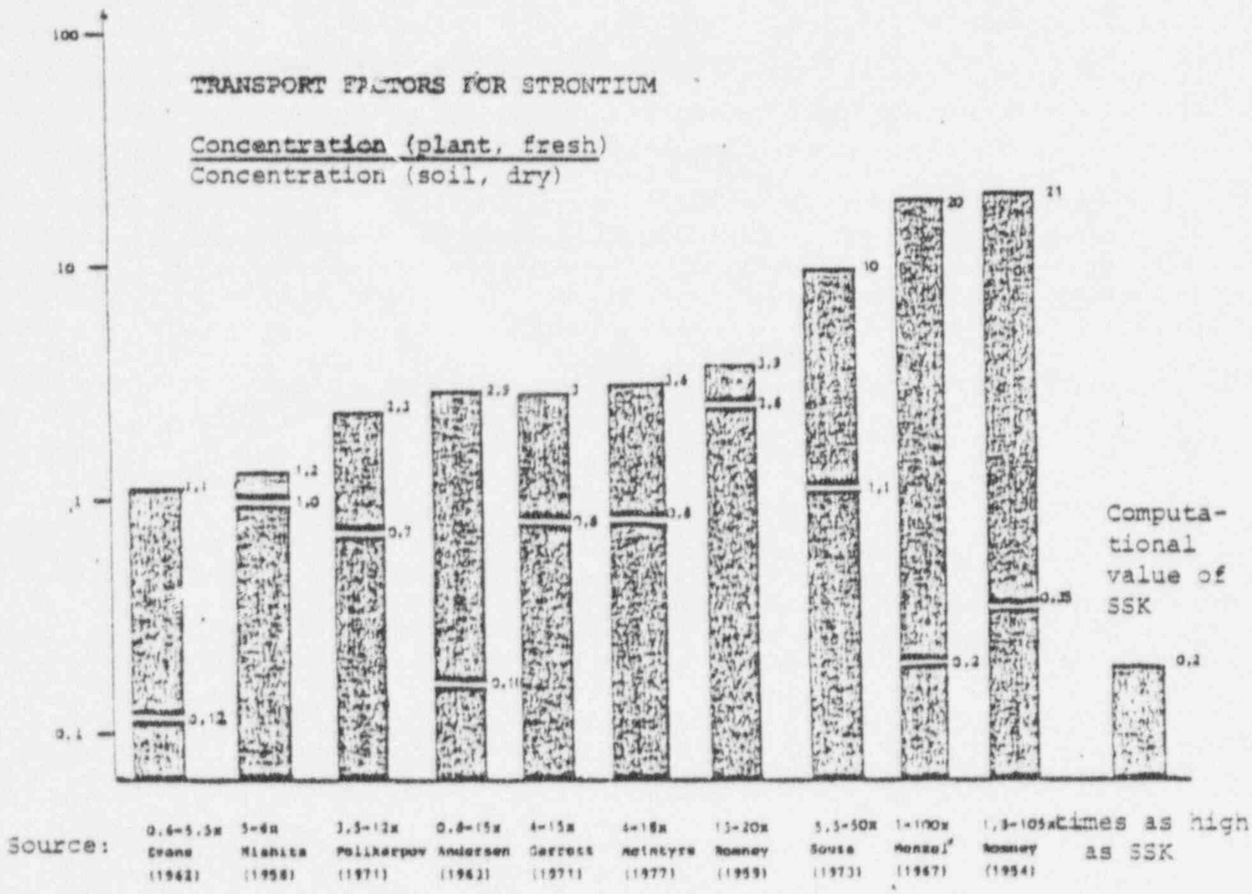
In contrast to this, the GRS uses the following transport factors for strontium:

maize	0.064	
pasture growth	0.2	
leafy vegetables and fruit	0.32	(GRS, 1976)

9001 8514

The SSK recommends a transport value of 0.017 (in SSK, 1977 I) and of 0.2 (in SSK, 1977 II) for strontium for all plants.

The figure on this page gives a graphic summary of the results of concentration experiments on strontium.



COMPARISON OF TRANSPORT FACTORS FOR STRONTUM IN PASTURE GROWTH AND VEGETABLES. DATA FROM VARIOUS PUBLICATIONS AND COMPUTATIONAL VALUE OF THE SSK.

(logarithmic scale) (*= specified range of variation)

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The graph shows that the values used by the GRS and recommended by the SSK are also too small for this important radionuclide.

6.2.8. Technetium

The emissions of Tc-99 have never been considered in calculations of the radiation load in the environs of nuclear plants, even though the fission yield of Tc-99 is 6.1% (i.e., Tc-99 is produced in about as much quantity as Sr-90 and Cs-137, it behaves much like iodine in the human body (and thus becomes concentrated in the thyroid gland), has a high transport factor from soil to plants, and has a half-life of 215,000 years (Wildung, R.E. et al., 1977).

Wildung, R.E. et al. (1977) determined transport factors of 22 to 130 for the uptake of Tc-99 by soybean plants and of 85 to 170 for the uptake of Tc-99 by wheat. The experiments revealed that the enrichment factor is clearly dependent on the concentration. The highest enrichment factors were found at the lowest radionuclide concentrations in the soil.

The SSK recommends a transport factor of 0.25 for Tc-99 from soil to all plants (SSK, 1977).

It can be said in summary that the transport factors for the passage of radionuclides from soil to crop plants that are recommended by the GRS, TÜV and SSK and that have been used in assessments (on which the licensing of the Wyhl nuclear power plant (Kernkraftwerk Süd) is based) by the Institute for Reactor Safety (October 1976, Exhaust Air) and by the Bavarian Biological Testing Institute (October 1976, Waste Water) (GRS, 1976; Bayr., 1976) in almost all investigated cases are either at the very lower end of the range of values given in the literature or are far below the values that may be regarded as realistic (in some cases they are too low by several powers of ten). It follows that the results of these assessments are unrealistically low and that the claim that the calculations are conservative (i.e., pessimistic) is not true. It also follows that any licensing granted on the basis of these assessments violates the Radiation Protection Law, especially section 45.

6.3. Transport Factors for Meat and Milk

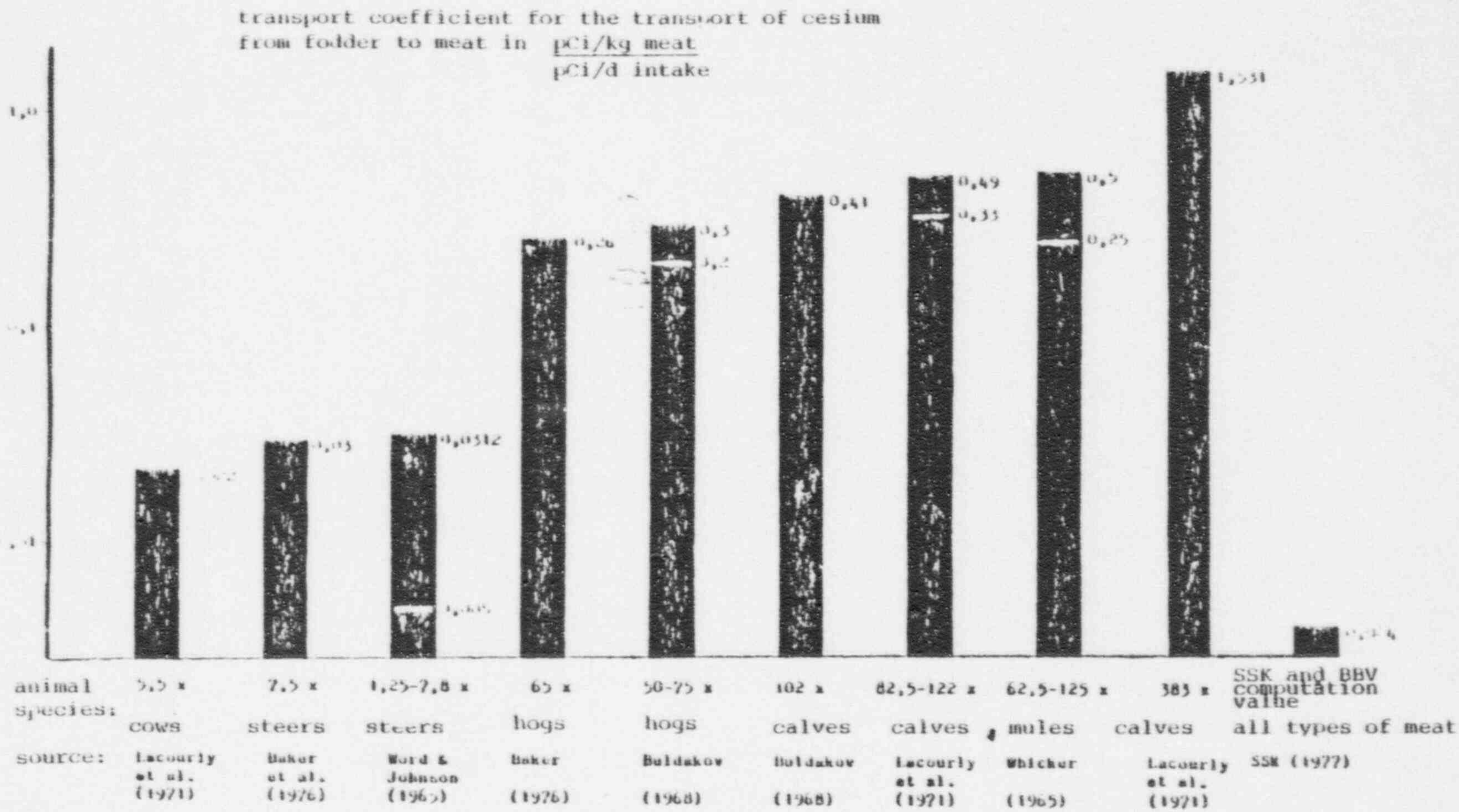
The transfer factor is also important in determining expected levels of radiation exposure by ingestion of meat or milk. The transfer factor indicates the percentage of the daily intake of a given radionuclide that is found in 1 kg of meat or in 1 liter of milk. The magnitude of this value depends to a great extent on whether or not the radionuclide is in a state of equilibrium in the body; the type of forage and the age and species of the animals

also affect this value.

The transfer coefficients that are recommended by the Radiation Protection Commission (SSK, 1977) for calculating radionuclide transport from the fodder to the meat and milk and that are used, for example, in assessments by the Bavarian Biological Testing Institute for the Wyhl plant (BBV, 1976) were compared by the Department of Environmental Protection with published values. The following graph shows the results of this study for the example of the radionuclide cesium and for the transport to meat.

The graph shows that the value recommended by the SSK is smaller than realistic values by a few powers of ten for this extremely important route of exposure as well.

For strontium Baker et al. (1976) gave a value of 0.003 for cattle and a value of 0.0073 for hogs. The SSK recommends a value of 0.0006, which is lower than Baker's values by about one power of ten.



Comparison of transport coefficients for the transport of cesium from forage to meat:
Results of various experiments and the value recommended by the SSK for purposes of calculation
(logarithmic scale.

7. Analyzed Routes of Exposure

7.1. Exposure via Exhaust Air

7.1.1. Gamma Submersion by Noble Gases

Since noble gases are unable to enter the food chain to any significant extent due to their chemical nature, most of the radioecological load takes the form of direct gamma radiation from the radioactive cloud issuing from the chimney. The effect of this radiation on human tissue depends on the distance from the radioactive cloud, on the absorption of the radioactivity by the air, and on the dose factor of the nuclide in question, which gives the effect (in rem/s) of one curie on a unit area of human tissue.

We have the following:

$$sd = qu \cdot ch \cdot dc$$

where sd = radiation load

qu = source strength

ch = meteorological dispersion factor

dc = dose factor

If we accept the dose factors given in SSK (1977) for the individual noble gases and use the dispersion factors for gamma submersion calculated in section 5.1, we obtain the following doses (in mrem/yr) for one block:

Distance	100 m	500 m (maximum receiving point)
realistic value	77	31
conservative value	403	80

These values must be doubled for two power plant blocks.

It is well known that calculations of this sort are partly a matter of knowledge and fact and partly a matter of assumptions and estimates. As was shown above and discussed in greater detail elsewhere (5.1.6), merely the uncertainty in the dispersion factors re-

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sults in a considerable difference between the realistic and the conservation value of dosage. Other uncertainties that can lead to an increase in dosage and that we were unable to examine further and make allowance for are therefore discussed briefly below:

a) the rates of liberation in the core of the reactor are not scientifically certain values, but rather were derived from a limited number of measurements and inferferences. The diffusion processes that occur in the reactor core are largely unknown. A change in the liberation rates can cause a considerable increase in dosage.

b) The possible effects of small disturbances, such as a problem in the exhaust gas system, were not considered.

c) Other routes of liberation, e.g., through the machine house roof, were not considered.

d) The dose factors taken from the literature (SSK, 1977), which are used as multiplicative factors in the calculation, were not checked.

e) The value of the dispersion factor, which is computed on the basis of simplified models and presents only an unsatisfactory picture of the real conditions, is also uncertain and questionable. For example, in the calculation of the gamma submersion the absorption coefficient is considered independent of the wavelength of the radiation, which is certainly incorrect.

f) The emitted noble gases disintegrate according to the following radioactive series:

$$\underset{1{,}79\,s}{\text{Xe-141}} \xrightarrow{\beta^-} \underset{24{,}7\,s}{\text{Cs-141}} \xrightarrow{\beta^-} \underset{18{,}3\,m}{\text{Ba-141}} \xrightarrow{\beta^-} \underset{3{,}93\,h}{\text{La-141}} \xrightarrow{\beta^-} \underset{32{,}51\,d}{\text{Ce-141}} \xrightarrow{\beta^-} \underset{\text{stable}}{\text{Pr-141}}$$

$$\underset{13{,}5\,s}{\text{Xe-140}} \xrightarrow{\beta^-} \underset{64\,s}{\text{Cs-140}} \xrightarrow{\beta^-} \underset{12{,}79\,d}{\text{Ba-140}} \xrightarrow{\beta^-} \underset{40{,}2\,h}{\text{La-140}} \xrightarrow{\beta^-} \underset{\text{stable}}{\text{Ce-140}}$$

Xe-139 (39,7 s) $\xrightarrow{\beta^-}$ Cs-139 (9,3 m) $\xrightarrow{\beta^-}$ Ba-139 (82,7 m) $\rightarrow$ La-139 stable

Xe-138 (14,1 m) $\xrightarrow{\beta^-}$ Cs-138 (32,2 m) $\xrightarrow{\beta^-}$ Ba-138 stable

Xe-137 (3,83 m) $\xrightarrow{\beta^-}$ Cs-137 (30,1 a) $\xrightarrow{\beta^-}$ Ba-137 stable

Xe-135 (9,17 h) $\rightarrow$ Cs-135 ($2\cdot10^6$ a) $\xrightarrow{\beta^-}$ Ba-135 stable

Xe-135m (15,3 m) $\rightarrow$ Cs-135m (53 m) $\rightarrow$ Ba-135m (28,7 h)

Kr-91 (8,6 s) $\xrightarrow{\beta^-}$ Rb-91 (58 s) $\xrightarrow{\beta^-}$ Sr-91 (9,5 h) $\xrightarrow{\beta^-}$ Y-91 (58,5 d) $\xrightarrow{\beta^-}$ Zr-91 stable

Kr-90 (32,3 s) $\xrightarrow{\beta^-}$ Rb-90 (2,6 m) $\rightarrow$ Sr-90 (28,5 a) $\rightarrow$ Y-90 (64,1 h) $\rightarrow$ Zr-90

Kr-89 (3,18 m) $\xrightarrow{\beta^-}$ Rb-89 (15,2 m) $\rightarrow$ Sr-89 (50,5 d) $\rightarrow$ Y-89 stable

Kr-88 (2,80 h) $\xrightarrow{\beta^-}$ Rb-88 (17,8 m) $\rightarrow$ Sr-88 stable

Kr-87 (76,3 m) $\xrightarrow{\beta^-}$ Rb-87 ($4,7\times10^{10}$ a) $\rightarrow$ Sr-87 stable

In the calculation of the radiation load, only the noble gases were considered. As the radioactive series show, however, most of the noble gases disintegrate into some highly radioactive nuclides of the elements cesium, strontium, rubidium, barium and cerium. None of these radioactive decay products were considered even though they cause an increase in the calculated radiation load.

7.1.2. <u>Ground Radiation by Deposited Radionuclides</u>

A portion of the radionuclides discharged into the atmosphere by the nuclear power plant is deposited on the ground in the immediate and distant area around the plant by the processes of fallout and washout (dry and wet deposition). This produces an external radiation

source for human beings in this area by the gamma emission of the nuclides. Since the contamination of the ground and thus the radiation load increase in the course of time, the SSK calculated the radiation load for the 50th year of reactor operation (SSK, 1977).

The gamma submersion dose from radionuclides that have settled onto the ground is determined by the following formula:

$$D_{io} = Q \cdot f \cdot \chi \cdot v_G \cdot S \cdot \frac{(1 - e^{-\lambda i \cdot t})}{\lambda i}$$

where D_{io} = gamma submersion dose in mrem/yr by nuclide i

Q = emission source strength in Ci/yr

f = nuclide fraction of the nuclide in questio[n]

χ = meteorological long-term dispersion factor

v_G = rate of deposition by dry and wet deposition

S = dose factor for gamma submersion (ground) in $\frac{rem \cdot m^2}{Ci \cdot s}$

λ_i = decay constant of the nuclide i

Table 7.1.2-1 : Gamma submersion dose factors (ground) in $\frac{rem \cdot m^2}{Ci \cdot s}$

Nuclide	Dose factor (ground)
Co-90 [sic]	2.20 E - 03
CO-60	6.00 E - 03
Zn-65	1.90 E - 03
Sr-90	0.00 E + 00
Cs-134	3.90 E - 03
Cs-137	1.40 E - 03
Ce-144	7.30 E - 05
Pu-239	6.10 E - 06
I-131	1.10 E - 03

(after SSK, 1977)

For an individual staying in this area for an entire year, we obtain a radiation load by this route of exposure of 14.6 mrem/yr as the sum of the investigated radionuclides. This radiation dose is applicable for a height of one meter above the ground according to the SSK (1977).

The exhaust gas assessment of the Association for Reactor Safety for the Wyhl plant (Kernkraftwerk Süd) (GRS, 1976) did not consider the radiation dose to be expected by this route of exposure, although the statement "external γ-radiation = 12.6 mrem/yr" in Table 7.5 (p. 13-22) gave the impression that this route of exposure had been taken into account. An examination will show that the 12.6 mrem/yr refers only to the gamma radiation dose from the cloud of noble gas.

7.1.3. Ingestion of Vegetable Foodstuffs

The land of the planned nuclear power plant is located within a wooded zone that is about 1000 to 1500 meters wide at the proposed site. Outside of this wooded zone, both sides of the Rhine Valley are used mainly for farming. Wine, fruit and tobacco are the main agricultural products in the Rhine plain, in the Kaiserstuhl (mountainous region south of Wyhl) and in the foothills, while grain crops, root crops, forage crops and timber are important in the remaining areas.

In the area of 10-km radius around the site there were farming and forestry operations on the German side of the river with a total area of 18,222 ha in 1975. The land-use data is given below:

- pasture land (meadows, fields etc.)	10%
- tilled land	34%
- forest	29%
- vineyards	9%
- orchards	2%

The following table shows the exact distribution of land use in the Emmendingen district.

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	total commercially exploited area	area used for agriculture	tilled land	pasture land	forest area	vineyards	orchards
Emmendingen district							
total	65 496	28 710	13 025	12 156	23 240	2 425	679
Endingen	2 735	1 907	839	283	360	681	172
Forchheim	1 437	1 124	948	121	331	46	7
Herbolzheim	3 377	1 760	893	446	805	323	66
Kenzingen	4 050	1 150	672	165	2 336	264	22
Riegel	870	652	471	124	113	51	1
Leiselheim	261	197	86	12	40	78	19
Jechtingen	803	437	177	32	190	179	44
Weisweil	1 040	677	531	135	122	2	3
Wyhl	1 545	851	622	207	322	3	4
Rheinhausen	2 054	1 209	875	317	512	0,2	3

Table 7.1.3-1. Land use in 1975 in the Emmendingen district (areas in ha) (GRS, 1976)

Even in normal operation the planned nuclear power plant would release radioactive substances to the environment which would be dispersed by meteorological processes and deposited on agriculturally exploited land in the immediate and more distant areas around the plant. In this case radioactive substances would reach human crop plants by two routes:

- by settling directly onto crop plants
- by settling onto the ground, physical enrichment of the deposited nuclides as deposition continues over a period of years, and transport of the radionuclides from the soil to the edible parts of the plants via the root system.

The assessment therefore calculates the radiation exposures to be expected from ingestion of the following vegetable foods:

- leafy vegetables
- root vegetables
- potatoes
- grains
- grass and
- clover as forage plants for animals
- wine (section 7.1.5.)

The transfer factor describing the transfer of a radionuclide from the air at ground level to the edible part of a crop plant is calculated with the following formula:

$$f_i = V \cdot 86\,400 \cdot \left[\frac{r \cdot T_v \cdot F \cdot (1-e^{-\lambda_{Ei} \cdot te})}{\lambda_{Ei} \cdot Y_v} + \frac{B_{iv} \cdot (1-e^{-\lambda_i \cdot te})}{\lambda_i \cdot P} \right] \cdot e^{-\lambda_i \cdot th}$$

The transfer factor for the deposition of I-131 from ground-level air onto plants is calculated with the following equation:

$$f_{I-131} = \frac{f_A \cdot 86400 \cdot F}{\lambda_{Ei\,(I-131)}}$$

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where f_i = transfer factor for nuclide i in pCi/kg fresh plant : pCi/m^3 air

V = total rate of deposition of the radionuclides by fallout and washout

r = retention factor, dimensionless

The retention factor indicates the fraction of the deposited radioactivity which is initially retained on the part of the vegetation that is above ground.

T_v = translocation factor, dimensionless, indicates the fraction of activity deposited on the leaves which enters the edible part of the plant in the course of time.

F = factor describing the loss of externally deposited activity during preparation of the plants in the kitchen, dimensionless.

λ_{Ei} = effective decay constant for isotope i in 1/d. λ_{Ei} is a measure of the rate at which superficially adsorbed radionuclides disappear from the surface of the plant by weathering processes and physical decay.

Y_v = vegetation density of the plants in kg fresh weight/m^2

t_e = exposure time in d
t_e is the time from sprouting of the plants until harvest.

B_{iv} = transport factor for a nuclide i in pCi/kg fresh plant : pCi/kg soil by root uptake from the soil.

P = 224 kg/m^2 mass of the earth in the plowed layer under one square meter of ground surface

λ_i = physical decay constant of a nuclide i in 1/d

$\lambda_i = \frac{\ln 2}{t_{1/2\ i}}$, where $t_{1/2\ i}$ is the half-life of the nuclide

t_B = reactor operating time

t_B is also the period of time in which radionuclides can accumulate in the soil

t_h = storage time of the products

f_A = see section 5.1.7.

(after: USNRC, 1976, and Baker, D.B., 1976)

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The first term inside the brackets of the equations for computation of the transfer factor gives the direct deposition of radioactivity on the surface of the plant, and the second term inside the brackets gives the enrichment of radionuclides from the soil into the plants. The expression $e^{-\lambda_i \cdot t_h}$ takes into account the decay of the radionuclides during storage.

The following values are used in the calculation of the transfer factors for vegetable foods:

Retention factor for deposition on plants

The value of r = 0.33 was adopted from the GRS assessment for use in this calculation.

Loss factor during kitchen preparation

The loss factor F gives the loss of externally deposited radioactivity during harvesting and kitchen preparation. A loss factor of 0.28 is used in the GRS assessment.

The literature contained only one paper in which this question was explored in detail. Rohleder employed various decontamination measures in an effort to reduce the amount of artificial fission products contained in kale from normal atomic bomb fallout. After washing kale in cold water for 1 hour, this author still found 91% of the cesium-137 and 87% of the zirconium-95 and niobium-95 that were originally contained in the kale. After treating the kale for three hours in warm water (35°C), he still found 62% of the cesium-135 and 74% of the zirconium/niobium-95 (Rohleder).

A loss factor of 0.4 is used in the calculation for leafy vegetables and grains, and a factor of 1 is used for potatoes and root vegetables. The factor of 1 takes into account the fact that for these vegetables the radioactivity is not the result of external contamination, but rather of root and leaf uptake and subsequent transport to the edible part of the plants (potatoes and root vegetables).

Vegetation Density

The following values (in kg fresh weight per square meter) are taken from Fletcher, I.F. et al. (1971) as the vegetation density of the plants:

leafy vegetables	1.5 kg/m^2
root vegetables	4.0 kg/m^2

potatoes	1.8 kg/m^2
grains	0.34 kg/m^2

Exposure time of the plants

An exposure time of three months is used for leafy vegetables, root vegetables, potatoes and grains (after Fletcher, I.F., 1971). An exposure time of 30 days is used for pasture vegetation. According to Baker, D. A. et al. (1976), this is the typical amount of time that passes before cows return to graze in the same part of the pasture. In the assessment by the GRS and the recommendations of the Radiation Protection Commission, the value for pasture plants is also 30 days. However, an exposure time of only 2 months is applied to all other crop plants.

Translocation factor

This factor indicates the fraction of activity deposited on the leaves which enters the edible part of the plant in the course of time.

T_v = 0.1 for potatoes and root vegetables

T_v = 1 for pasture vegetation and leafy vegetables

(after Baker, D.A. et al., 1976)

According to the GRS (1976), the product $r \cdot T_v$ is 0.05 for cereal grains. This means that 5% of the deposited radionuclides are directly adsorbed on the grains of the cereal plants.

We used the same values as in the GRS assessment for the mass of the earth in the plowed layer under one square meter of ground surface (224 kg/m^2), for the wash-off half-life, i.e., the time required for 1/2 of deposited radioactive substances to be removed from the surface of the plant by meteorological processes (14 days), and for the storage time of potatoes, grains and stall fodder (1/2 year).

We used the reactor operating time recommended by the SSK (1977) for this type of calculation (50 years = 18,250 days).

Table 7.1.3-2. Transport factors for plants in pCi/kg fresh plant over pCi/kg soil.

nuclide	grass	clover	leafy veg.	potatoes
Co	0,0094	0,02	0,02	0,02
Zn	0,4	0,4	0,4	0,4
Sr	3,2	7,2	2,5	0,75
I	0,2	0,2	0,2	0,2
Cs	5,9	8,5	0,75	15
Ce	0,0005	0,0005	0,00075	-
Pu	0,1	0,1	0,1	0,1

nuclide	root vegetables	cereal grains
Co	0,032	0,015
Zn	0,4	0,4
Sr	15	1,67
I	0,2	0,2
Cs	0,07	0,48
Ce	0,032	0,015
Pu	0,1	0,1

(Baker, D.A. et al., Food, 1976; Bergamini et al., 1970; Fletcher, I.F., 1971; Garett, A.R. et al., 1971; GRS, 1976; Grummit, 1975; Guljakin, J.W. et al., 1974; Hakonson Th. E. et al., 1973, 1974; Herbst, W., 1976; Lipton, W.V., Goldin, A. S., 1976; Marckwordt, U., 1971; Romney, E.M. et al., 1954, 1957 and 1959; Souza, T.J. et al., 1972; Teufel, D., 1977; Vose, P.B., Koontz, H.V., 1959; Whicker, 1974; UCRL, 1968; USNRC, Guide 1.109, 1976).

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The transfer factors (f_i) determined by the method described above are compiled in Table 7.1.3-3.

Table 7.1.3-3. Transfer factors in pCi/kg fresh plant : pCi/m^3 air

nuclide	leafy vegetable	potatoes	root vegetables	cereal grains
Co 58	1 669	61	30	472
Co 60	2 237	643	588	3 233
Zn 65	2 577	657	529	2 280
Sr 89	2 471	51	476	269
Sr 90	134 325	39 638	784 836	90 580
I 131	2 037	0	0	0
Cs 134	6 019	69 468	477	4 929
Cs 137	42 755	806 883	3 946	29 031
Ce 144	1 887	254	157	2 033
Pu 239	11 118	9 556	9 329	12 409

The following rates of consumption were used:

Table 7.1.3-4. Annual consumption of vegetable foodstuffs.

	adults
leafy vegetables	50 kg
root vegetables	50 kg
potatoes	90 kg
grains	90 kg

The radiation doses to be expected from the ingestion of vegetable foodstuffs are given in section 9.

7.1.4. Ingestion of Animal Foodstuffs

Table 7.1.4-1 lists the numbers of different types of animals raised in the various farming communities in the Emmendingen district. Quantitatively, the most important animals are cattle and hogs, which we shall discuss below in regard to their contribution to the possible radiation dose received by human beings from the planned nuclear power plant.

	horses	cattle	includ-ing milk cows	hogs	chick-ens	ducks	geese	turkeys	bee swarms	sheep	goats
total for Emmendingen district	764	26 144	9 247	33 368	146 400	947	974	21	2 047	2 268	230
Endingen	52	776	106	1 472	4 625	20	-	2	23	-	2
Forchheim	13	737	231	2 307		-	-	-	2	-	-
Herbolzheim	62	1 030	290	1 650	5 443	-	16	-	80	21	14
Kenzingen	47	306	103	1 054	4 258	145	34	6	-	43	5
Riegel	9	541	229	747	1 772	17	4	-	20	807	2
Leiselheim	2	26	11	195	497	-	-	-	-	1	2
Jechtingen	-	208	21	122	825	8	4	-	-	-	1
Weisweil	22	346	109	2 705	3 830	30	3	6	-	12	-
Wyhl	16	437	129	1 203	1 233	26	-	-	-	-	-
Rheinhausen	44	674	184	1 548	2 473	67	4	9	-	63	-

Table 7.1.4-1. Animal raising in the Emmendingen district (number of head) (after GRS, 10/1976)

It has been known for many years that radioactive substances reach the human body by the pathway pasture - cow - milk. Since babies drink relatively large quantities of milk and are thus an especially endangered group, this exposure pathway is extremely important.

The quantity of short-lived radionuclides, such as I-131, which is absorbed by the body from milk in one year depends on the length of the grazing season. The grazing season in Germany is variable, but according to Hoffman, F.O. (1973), it is up to 200 days for pasture land located at low altitudes. According to measurements on American reactors, 30 to 70% of I-131 is emitted in gaseous form. However, since no measurements of this sort have been made for German pressurized water reactors, the Department of Environmental Protection, in agreement with IRS-W-13 (IV, 1975), based its calculations on the conservative assumption that 100% of the iodine emissions consists of elemental gaseous iodine. The transfer of radioisotopes from the air to grass and clover was discussed in the preceding section. Clover generally absorbs larger percentages of radionuclides than does grass. The enrichment factors for clover and grass are compiled in Table 7.1.3-2. The forage consumption of the cows must also be known before we can compute the percentages of the radioisotopes that find their way into 1 liter of milk. For this parameter we assumed (after Baker, D.A. et al., 1976) a daily consumption of 75 kg of fresh pasture vegetation during the grazing season and a daily consumption of 10 kg of silo fodder (consisting of 5 kg of hay and 5 kg of clover) during the stall-feeding period. Five kg of stall fodder is equivalent to an initial fresh weight of 18 kg. Our calculations were based on an average storage time of 1/2 year for the stall fodder. Another parameter that enters into the calculation is the transfer coefficient, which indicates what fraction of the radioactivity ingested by the cows each day with their fodder reappears in one liter of milk. The values used in our computations are compiled in Table 7.1.4-2. It should be mentioned that the especially critical isotope I-131 is contained in sheep's milk and goat's milk in significantly higher concentrations than in cow's milk. Therefore, babies who drink fresh sheep's milk or goat's milk are exposed to significantly higher radiation doses than babies who drink cow's milk. We therefore calculated the expected radiation dose for babies from ingestion of sheep's milk and goat's milk on the basis of the assumption that daily milk consumption is 0.5 liters during the grazing season of these animals (0.75 yr). We further assumed that the daily consumption of fresh forage is 8 kg for sheep and 6 kg for goats.

The Department of Environmental Protection also considered the radiation doses that could be expected from ingestion of meat. We performed complete computations for the radiation exposure to be expected from ingestion of beef and pork.

The transfer factor for the transfer of radionuclides from the fodder to beef and milk is given by the following formula:

$$f_i = f_{iv} \cdot V \cdot S_d$$

where f_{iv} = transfer factor for grass or stall forage (see section 7.1.3).

V = forage consumption in kg fresh weight/day (see above).

S_d = transfer coefficient in $\frac{\text{pCi/l milk}}{\text{pCi/d intake}}$ and $\frac{\text{pCi/kg meat}}{\text{pCi/d intake}}$

S_d gives the percentage of activity absorbed per day which is receovered in one liter of milk or in one kilogram of beef (Table 7.1.4-2.)

In regard to the hog forage, the possible contamination was calculated as follows:

It was assumed that young pigs weighing 20-25 kg are obtained from a breeding farm with no radionuclide exposure. An average quantity of feed per day of

10 l whey

3 kg potatoes and

1 kg cereal grain

was assumed for the five-month fattening period until slaughter maturity was reached (after Kirchgaessner, 1973).

According to Karavaer et al. (1973), about 85% of the radionuclides contained in whole milk remain in the whey. The radionuclide load of the fodder was calculated as described above.

Therefore, the transfer factor for pork is given by the following formula:

$$f_i = (f_{iv\ whey} \cdot V_{whey} + f_{iv\ potatoes} \cdot V_{potatoes} + f_{iv\ grain} \cdot V_{grain}) \cdot S_d$$

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where f_i = transport factors of the fodder ingredients

V = fodder consumption per day

S_d = transfer coefficient for pork

The transfer coefficients for the transport of radionuclides into beef, pork and milk are compiled in Table 7.1.4-2.

Table 7.1.4-2. Transfer coefficients for the transport of radionuclides into beef, pork and milk

nuclide	beef $\frac{\text{pCi/kg meat}}{\text{pCi/d intake}}$	pork $\frac{\text{pCi/kg meat}}{\text{pCi/d intake}}$	milk $\frac{\text{pCi/l milk}}{\text{pCi/d intake}}$
Co	0,013	0,013	0,001
Zn	0,05	0,14	0,039
Sr	0,003	0,0073	0,0029
I	0,02	0,09	0,01
Cs	0,1	0,26	0,012
Ce	0,0012	0,005	0,0006
Mn	0,005	0,02	0,00025
Pu	0,005	0,01	0,000002

Transfer factor for I in sheep's milk = 0.40 and in goat's milk = 0.47

After: Buldakov, L.A. et al., 1968; Ward, G.M., Johnson, J.E., 1965; Baker, D.A. et al., 1976; Strahlenschutzkommission, 1977; Annekov, B.M., 1971; Hoffman, F.O., 1975)

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The transfer factors determined in this way are compiled in Table 7.1.4-3.

Table 7.1.4-3. Transfer factors in pCi/kg meat (or pCi/l milk) : pCi/m^3 air

nuclide	Beef	Milk	Pork
Co 58	3 479	268	38
Co 60	5 049	388	110
Zn 65	18 477	14 412	17 745
Sr 89	896	866	57
Sr 90	35 055	33 886	3 632
I 131	12 600	6 300	4 820
Cs 134	221 462	26 575	114 197
Cs 137	1 973 473	236 817	1 160 282
Ce 144	408	204	23
Pu 239	4 548	2	411

The following values were used as consumption rates:

Table 7.1.4-4. Annual food consumption.

	Adults	Babies
Milk	360 liters	320 liters
Beef	100 kg	
Pork	40 kg	

The radiation doses to be expected from the ingestion of animal foodstuffs are given in section 9.

7.1.5. Exposure Pathway Grapevines - Wine

Wine was taken into account as a relevant foodstuff in the 1976 assessment of the Association for Reactor Safety (GRS); however, recalculation with the individual parameters that were given showed the following disparity between the result of the multiplication and the final values in Table 11.2 of the GRS assessment:

	whole body	bone	liver	kidney
GRS (Tab. 11.2.)	0,08 mrem	0,4 mrem	0,1 mrem	0,04 mrem
results of the recalculation	1,93 mrem	5,85 mrem	0,9 mrem	0,27 mrem
the radiation doses given in the GRS assessment (Table 11.2) were too small by the following factors:	24 x	14,7 x	9 x	6,7 x

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We therefore felt that it was necessary to reexamine the individual parameters, especially the transfer factors of the grapevine - wine exposure pathway.

In this case three transfer factors must be determined:

(f_1) : air - leaf surface - grapes - wine

(f_2) : air - grape surface - grapes - wine

(f_3) : air - soil - plant - wine

The determination was based on the following mathematical model:

$$(f_1) : V_1 \times 86\,400 \times T_{V1} \times \frac{r_1'(1-e^{-\lambda_{ei} \cdot t_{e1}'}) + r_1''(1-e^{-\lambda_{ei} \cdot t_{e1}''})}{\lambda_{ei} \cdot Y_{V1}} \times$$

$$\times \frac{\text{kg Leaf}}{\text{l wine}} \times e^{-\lambda_i \cdot t_h}$$

$$(f_2) : V_2 \times 86\,400 \times T_{V2} \times \frac{r_2 (1-e^{-\lambda_{ei} \cdot te_2})}{ei \cdot Y_{V2}} \times e^{-\lambda_i \cdot t_h}$$

$$(f_3) : V_3 \times 86\,400 \times \frac{B_{iv} \cdot (1-e^{-\lambda_i \cdot t_3})}{\lambda_i \cdot P} \times 0{,}72 \times e^{-\lambda_i \cdot t_h}$$

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A value of 0.04 m/s is used for the rates of deposition V_1 and V_2; this value takes into account the greater surface roughness and the filter effect caused by the shrublike nature of the grapevines (see 5.1.7).

The retention factors r_1 and r_2 take into account both the distance separating the individual grapevines and the proportion of nuclides deposited on the surface of the leaves and grapes. Two retention factors r_1' and r_1'' were determined for (f_1); this is necessary because the vines only have leaves at the beginning of the growing season, and later in the season they have both leaves and grapes on which nuclides are deposited. The separation of the grapevines is taken into account with a factor of 0.4 since about 44 vines grow on 100 m^2 of vineyard area. (National Viniculture Institute in Freiburg, 1977).

The grapevines begin to sprout at about the beginning of May; within three months the leaf surface area reaches 320 m^2/100 m^2. The first pruning is performed at the end of July; this reduces the leaf surface area to 250 m^2/100 m^2. Before the second pruning in the second half of August, the leaf surface area increases to 280 m^2/100 m^2; the second pruning reduces it to 250 m^2/100 m^2 and it remains at this value for the rest of the season (National Viniculture Institute, Freiburg, 1978). Since the leaves are very small at the beginning of the season, the first month of exposure time was disregarded.

During the following months (t_{e1}' = 60 days) the total surface area of the grapes is considered negligible; the nuclides are distributed over the available leaf surface. This is taken into account in r_1'.

Once the grapes have become so large that it is necessary to include their surface area in the calculation, this is expressed in terms of its ratio to the leaf surface area r_2 to r_1''.

The grape surface area is 60 m^2/100 m^2 (GRS, 1976). Accordingly, in the months of August and September (t_{e1} = 60 days) the nuclides are distributed on the grapes and leaves in a ratio of 0.2 to 0.8.

In regard to the exposure time of the grapes (te_2), we disregard the first six weeks of the 100-day period from flowering to harvesting during which the grapes are hanging on the vine (National Viniculture Institute, Freiburg, 1978).

The meteorological half-life of the nuclides on the leaves and grapes is assumed to be 30 days because the surfaces are covered with adhering spray agents.

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The translocation factor Tv_1 indicates the percentage of the nuclides which passes from the surface of the leaves into the plant and then into the grapes. This factor is nuclide-specific and was obtained from the following sources: Aarkrog, A., 1975; Rohleder, U., 1972; Middleton, L.J., Squire, H.M., 1963.

The translocation factor Tv_2 indicates the percentage of the nuclides remaining on the grapes despite wind and rain which eventually passes into the wine. This factor is 0.5 (GRS, 1976).

The leaf mass Yv_1 is subject to the same variation as the leaf surface area as a result of the prunings and has a mean value of 0.5 kg/m^2. In this case the low weight of the younger leaves is taken into consideration (National Viniculture Institute, Freiburg, 1977).

With an average grape yield of 1.1 kg/m^2, the wine yield Yv_2 is 0.825 l/m^2 (assuming that 0.71 l of wine is produced from 1 kg of grapes) (GRS, 1976). It is assumed that 100% of the nuclides present inside the grapes passes into the wine.

Since the literature contains no values for enrichment factors (Biv) of wine, we adopted the values that were available for the most closely related plants.

P gives the mass of the earth in the plowed layer under one square meter of ground surface area. Its value is 224 kg/m^2.

The reactor operating time t_b is 50 years (Basis of Calculation of the Radiation Protection Commission, 1977).

The expression $e^{-\lambda i \cdot th}$ gives the decay of the radionuclides during storage of the wine.

The decay time t_h gives the minimum storage time of the wine, which is 120 days (GRS, 1976); however, this value does not take into consideration the fact that a small portion is consumed as grape must before the end of the 120-day storage period.

The rate of consumption was assumed to be 400 l of wine per year (after GRS, 1976).

In our determination of the radiation dose to be expected by this pathway, we considered the fact that the nearest wine-growing area is not located at the maximum radiation receiving point, but rather at Limberg. Consequently, the long-term dispersion factor for the maximum receiving point was not used. We adopted a more realistic value for Limberg (according to section 5.1.6).

The transfer factors for wine are compiled in Table 7.1.5-1. Due to a lack of utilizable translocation factors for Zn, Ce and Pu, exposure via the leaves (f_1) was calculated only for Co, Sr and Cs.

Table 7.1.5-1. Transfer factors in pCi/l wine : pCi/m^3 air.

Nuclide	Transfer factor
Co 58	3 015
Co 60	11 678
Zn 65	3 986
Sr 89	1 027
Sr 90	44 559
I 131	0
Cs 134	71 242
Cs 137	150 344
Ce 144	3 875
Pu 239	12 023

The radiation doses to be expected from consumption of wine are given in section 9.

7.2 Exposure via Waste Water

As in the SSK Recommendations (SSK, 1977), it is necessary to make a basic distinction between internal and external exposure in our discussion of the radiation doses to be expected by the waste water pathway.

The pathways of internal exposure include the following:

- drinking water
- ingestion of fish
- irrigation - ingestion of vegetable foodstuffs
- irrigation - forage plants - ingestion of milk (ingestion of meat)
- livestock watering - ingestion of milk/ingestion of meat

The above pathways of exposure were computed with respect to their significance for the radiation dose of the most unfavorable location. They are discussed in detail below. We did not consider external exposure from direct radiation while swimming and boating and from sediment radiation (recreational outings on the banks of the Rhine etc.). The radiation doses resulting from these activities would have to be added to the dose from internal exposure.

7.2.1. Exposure by Ingestion of Fish

The radiation dose from ingestion of contaminated fish was determined with the following formula:

$$D_{io} = Q_i \cdot \frac{1}{W} \cdot f_i \cdot d_{io} \cdot V$$

where D_{io} = radiation dose for an organ o caused by isotope i (in mrem/yr)

V = rate of consumption (in kg/yr)

d_{io} = dose commitment factor for organ o and nuclide i (in mrem/pCi)

Q_i = source strength of the nuclide (in pCi/s)

W = volume flow rate of the river (in l/s)

f_i = transfer factor for nuclide i (in $\frac{\text{pCi/kg fish meat}}{\text{pCi/l water}}$)

The transfer factors in Table 7.2.1-1 were obtained from the recent technical literature. There were large variations in the factors given in the literature for the critical nuclides.

In the case of strontium values of up to 370 were given (Hübel, Ruf and Herrmann, 1973). Kohlemainen et al. (1966) and Hakonson and Whicker (1975) determined maximum transfer factors for cesium of $6840/(K)_w$ and $13{,}000/(K)_w$, respectively. ($(K)_w$ gives the content of stable potassium in mg/l water.)

In the case of cesium it is necessary to consider the dependence of the enrichment on the potassium concentration of the water because cesium and potassium behave similarly in the organism of the fish and relatively more cesium is absorbed when the potassium concentration of the water is low.

Table 7.2.1-1. Transfer factors for fish in $\frac{\text{pCi/kg fish meat}}{\text{pCi/l water}}$

Nuclide	Factor
H 3	1
Co 58	400
Co 60	400
Zn 65	1 000
Sr 89	40
Sr 90	40
I 131	520
Cs 134	$4\,000/K_{(w)}$
Cs 137	$4\,000/K_{(w)}$

after BBV, 1976; Blanchard, R.L. & Kalm, B. jun., 1971; Hermann, H. et al., 1975; Hakonson & Whicker, 1975; Kohlemainen et al., 1966; Vanderploequ et al., 1975

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The assessment of the Bavarian Biological Testing Institute (1976) uses a potassium concentration of 6 mg/l as the annual average of the Rhine at Weisweil. In the opinion of our experts, this is an unacceptable value for a conservative estimate of the radiation dose that is to be expected. Since the estimation of the radiation dose to be expected is performed on the assumption of an operating time of 50 years, and since we can assume that the level of potassium pollution of the Rhine (especially from the Alsatian potash mines) will drop off sharply in the years to come as a result of increasing environmental protection measures, a value of 6 mg/l is clearly too high.

Measurements by Egger (1887) at Mainz showed a concentration of 1.9 mg K/l, and measurements by Pagenstecher (1837) at Basel showed a concentration of 1.2 mg K + Na/l (cited by Livingstone, 1963). The Department of Environmental Protection therefore assumed a potassium concentration in the Rhine of 1 mg/l for the purpose of making a conservative estimate of radiation doses that can be expected.

However, due to the uncertainties in regard to the transfer factors, it is debatable whether our assumptions put us on the safe side.

The chosen consumption rate of 50 kg/yr can be regarded as realistic for a critical population group (see the survey by Schaefer, BGA, of sport fishermen and professional fishermen, Model Study Radioecology Biblis, 3/17/1977), especially since we must expect increasingly large catches as a result of the possible improvement of the water quality of the Rhine.

7.2.2. <u>Irrigation - Ingestion of Vegetable Foodstuffs</u>

Due to the steady lowering of the water table in the upper Rhine region as a result of the canalization of the Rhine, and due to the relatively dry climate, the irrigation of farmland in the Rhine plain below the planned nuclear power plant is a realistic source of contamination.

Since it is very likely that the water quality of the Rhine will be improved by intensification of environmental protection measures, it must be assumed that Rhine water will be used to an increasing extent for irrigation of agricultural land.

Irrigation is performed mostly in the summer months. As was mentioned in section 5.2, a large percentage of yearly nuclear power plant emissions is also discharged in the summer months (during the changing of the fuel elements). Consequently, calculations of expected radiation dose that are based on uniform discharge of emissions would result in underestimation of the dose. A more exact study is necessary.

The following formula is used to compute the radiation exposure due to ingestion of crop plants grown in irrigated soil:

$$D_{io} = Q_i \cdot \frac{1}{W} \cdot f_i \cdot V \cdot d_{io}$$

where V = consumption rate in kg/yr

f_i = transfer factor for the nuclide i

The transfer factor f_i is determined by the following formula; this formula is similar to the equation in section 7.1.3 for the transfer of radionuclides from ground-level air to plants, the difference being that the rate and duration of irrigation must be considered instead of the rate of deposition of aerosols.

See section 7.1.3 for explanation of the meanings of the symbols.

$$f_i = R \cdot \left[\frac{r \cdot T_v \cdot F \cdot (1 - e^{-\lambda_{Ei} \cdot t_e})}{\lambda_{Ei} \cdot Y_v} + \frac{t_r \cdot B_{iv} \cdot (1 - e^{-\lambda_i \cdot t_e})}{12 \cdot \lambda_i \cdot P} \right] e^{-\lambda_i \cdot t_h}$$

where R = irrigation rate for garden vegetables 3 $l/m^2 \cdot d$

pasture land 1 $l/m^2 \cdot d$

potatoes 1.25 $l/m^2 \cdot d$

(after: Ruhr-Stickstoff-AG, 1978; Perrot, 1949)

r = retention factor = 0.33

T_v = translocation factor for pasture vegetation and leafy vegetables = 1

for potatoes and root vegetables = 0.1

F = loss factor for leafy vegetables = 0.4

for all other plants = 1

λ_{Ei} = effective decay constant for isotope i in d = $\lambda_i + \lambda_w$

t_e = exposure time in d for pasture vegetation = 30 days

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for vegetables and potatoes = 90 days

Y_v = vegetation density in kg fresh weight/m^2

for pasture plants = 0.85

leafy vegetables = 1.5

root vegetables = 4.0

potatoes = 1.8

B_{ir} = enrichment factor from the soil to the plant for nuclide i; (Table 7.1.3-2)

P = mass of the earth in the plowed layer (224 kg/m^2)

t_r = duration of irrigation = 4 months for vegetables and potatoes

= 5 months for pasture land

t_B = 18,250 days

t_h = 180 days (for potatoes and stall forage)

λ_i = physical decay constant

Leafy vegetables, root vegetables and potatoes were regarded as possible exposure pathways. The assumed rates of consumption are given in Table 7.1.3-3.

In our determination of the transfer factors, we departed from the recommendations of the Radiation Protection Commission (SSK) in several points.

For example, the SSK's determination of the amount of additional water that must be supplied by irrigation, which is based on mean precipitation values (SSK, 1977 II, p. 41), is scientifically unsatisfactory.

It must be realized that during irrigation part of the water immediately evaporates. Also, according to Ruhr-Stickstoff-AG (1978), significantly higher irrigation rates than 1 l/m^2·d are advisable for vegetable cultivation. 600 l/m^2 per year is given for intensive outdoor vegetable cultivation, and 150 l/m^2 per year is given for field vegetable cultivation (Ruhr-Stickstoff-AG p. 506).

On the basis of the vegetation period of 4 months for vegetable growing and the recommendations for agricultural practice (Ruhr-Stickstoff), we determined mean values of 1.25 to 5 l/m^2 per day. For the purpose of obtaining realistic estimates, we based our

calculations on a mean value of 3 l/m^2 per day.

Our calculation of the nuclide concentration that accumulates in the soil in the course of the operating life of the reactor was based on the irrigation time of 4 months per year. This fact is not taken into consideration in the SSK recommendations; the SSK assumes a continuous irrigation rate of 1 l/m^2 per day over the entire year.

Consequently, the calculations of the SSK and Department of Environmental Protection give equivalent results in regard to the radionuclides accumulated in the soil at the end of the reactor operating time from irrigation with river water. The difference is that in the first part of the formula, for the portion of radioactivity that reaches the plants directly with the irrigation, we obtain higher values as a result of our more realistic approach (irrigation only in the summer).

Naturally, we cannot rule out the possibility that in especially dry years, when irrigation rates are higher, or during fuel element changes, when emission rates are higher, irrigated crop plants may cause higher radiation doses than are calculated in this study.

The transfer factors determined in this way are compiled in Table 7.2.2-1.

Table 7.2.2-1. Transfer factors in pCi/kg fresh plants : pCi/l irrigation water.

nuclide	leafy vegetables	root vegetables	potatoes
H 3	1	1	1
Co 58	4,4	0,1	0,1
Co 60	5,5	0,8	0,5
Zn 65	5,6	0,7	0,4
Sr 89	5	0,4	0
Sr 90	121,8	691,4	14,8
I 131	2	0	0
Cs 134	8,8	0,7	25,7
Cs 137	41,2	3,8	296,3

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7.2.3. Irrigation - Forage Plants - Ingestion of Milk and Meat

This exposure pathway concerns the possibility of contamination by ingestion of milk and beef from cattle which have grazed in pastures irrigated with Rhine water or which have been fed in the winter with stall fodder from irrigated pastures.

The following formula was used in our calculations:

$$D_{io} = Q_i \cdot \frac{1}{W} \cdot f_{iv} \cdot s_d \cdot d_{io}$$

where f_{iv} = transfer factor for grass or stall fodder for the passage of

radioactivity from the irrigation water to the plant in

$\frac{\text{pCi/kg plant}}{\text{pCi/l water}}$

(calculation analogous to sections 7.2.2 and 7.1.4)

s_d = transfer coefficient for the passage of daily ingested radioactivity from the forage into the milk or meat

(Table 7.1.4-2)

The transfer factors determined in this way are compiled in Table 7.2.3-1.

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Table 7.2.3-1. Transfer factors in pCi/kg beef or pCi/l milk : pCi/l irrigation water

nuclide	beef	milk
H 3	1	1
Co 58	3,1	0,2
Co 60	4,4	0,3
Zn 65	15,4	12,0
Sr 89	0,7	0,7
Sr 90	11,0	10,6
I 131	2,3	1,1
Cs 134	86,9	10,4
Cs 137	602,2	72,3

7.2.4. Livestock Watering - Milk (Meat)

This pathway concerns the possibility of contamination by ingestion of milk and beef from cattle which have been watered with radioactively contaminated water. The calculations were performed in accordance with SSK recommendations.

The following formula was used:

$$D_{io} = Q_i \cdot \frac{1}{W} \cdot L \cdot S_d \cdot d_{io}$$

where L = daily water consumption of the cow (75 l/d)

S_d = transfer coefficient for the passage of daily ingested radioactivity into the milk or meat

(see Table 7.1.4-2)

The transfer factors determined in this way are compiled in Table 7.2.4-1.

Table 7.2.4-1. Transfer factors in pCi/kg beef or pCi/l milk : pCi/l water.

nuclide	beef	milk
H 3	1	1
Co 58	1	0,1
Co 60	1	0,1
Zn 65	2,9	3,8
Sr 89	0,2	0,2
Sr 90	0,2	0,2
I 131	1,5	0,8
Cs 134	7,5	0,9
Cs 137	7,5	0,9

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7.2.5. Annual Consumption Rates (Waste Water Pathway)

The following consumption rates were assumed for the waste water pathway.

The annual consumption rates were used for the exposure pathways irrigation - forage plants - milk/meat and livestock watering - milk/meat.

Table 7.2.5-1. Annual consumption of foodstuffs used in the waste water pathway calculations

	Adults
cow's milk	360 l
leafy vegetables	50 kg
root vegetables	50 kg
potatoes	90 kg
beef	100 kg
fish	50 kg

The radiations doses to be expected from exposure by the waste water pathway are given in section 9.

8. Examination of the Dose Factors

8.1. Dose Factors - Mathematical Basis

The dose factor or dose commitment factor gives the radiation dose in mrem/yr which is produced per quantity of radioactivity in pCi/a absorbed by man.

The radiation exposure in an organ depends on the following factors:

- quantity of nuclide absorbed
- type of radiation emitted
- energy of the radiation
- relative biological activity of the radiation
- effective half-life of the nuclide in the organ
- selection or enrichment factor of the organ
- mass and form of the organ
- microdistribution of the activity in the organ

These factors depend in turn on the following factors (among others):

- chemical composition of the incorporated substance
- quantity of the stable element in the diet
- age and state of healty of the individual
- genetic constitution

The dose commitment factor is calculated by the following formula:

$$\frac{dD}{dt} = 0{,}05115 \, \frac{E_{eff} \cdot f_a}{m \cdot \lambda_{eff}} \left(1 - e^{-\lambda_{eff} \cdot 50\ a}\right) \frac{mrem}{a}$$

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where E_{eff} = effective energy in MeV

f_A = fraction of the absorbed activity which reaches the organ

m = mass of the organ in g

λ_{eff} = effective decay constant, takes into account the physical and biological half-life

For very long-lived nuclides the dose commitment factor gives the dose which is active in the fiftieth year of continuous radionuclide absorption (dose commitment). For nuclides with short and medium effective half-life the yearly dose and the dose commitment are the same.

The dose commitment factors used by the Association for Reactor Safety (GRS) and recommended by the Radiation Protection Commission (SSK) have a number of deficiencies. These deficiencies are set forth below, and, when possible, more realistic dose commitment factors are calculated.

8.2. State of Health and Genetic Constitution

The dose commitment factor applies to the so-called average human being, who weighs 70 kg, is healthy, and has a "normal" genetic constitution. Diseases or environmental influences which might lead to an increase in the rate of absorption of radionuclides or to a higher biological half-life of radionuclides are not taken into consideration. Lack of time prevented us from investigating this area in greater detail. However, it may be expected that these factors would have a considerable effect on the dose commitment factor. The content of the stable element in the diet is an example of such a factor; the dose commitment factor would be strongly affected by iodine deficiency, for example.

A paper on the measurement of plutonium in cows in the vicinity of an American nuclear research center seems very important in this connection. In this study plutonium values of 0.5 to 10 pCi/kg were found in beef. However, a concentration of 100 - 200 pCi/kg Pu-239 was found in one of the cows, although the cow was subject to the same environmental conditions as the other cows. The author of this report states that this cow had a genetic abnormality, and although this abnormality did not adversely affect the cow's viability, it did represent a possible explanation for the high plutonium concentration. (Smith, D.D., 1973)

Since man does not have a uniform genotype, and since not all individuals are healthy, some individuals may be exposed to significantly higher radiation doses than those calculated here for a normal, healthy person; this will depend on an individual's genetic constitution and state of health.

8.3. Chemical Form of the Radionuclide

Depending on the chemical form in which the radionuclide is present in the diet, the dose commitment factor (and thus the radiation dose resulting from absorption of the radionuclide) may vary by several powers of ten.

This will be discussed with the example of cobalt-60 (half-life 5.2 years), which is emitted as a corrosion product by nuclear power plants.

The dose factor used by the GRS and recommended by the SSK for Co-60, e.g., in the liver, is 2.15×10^{-6} mrem/pCi. Although this dose factor is valid only when the Co-60 is present in inorganic form (as a salt), it is used for all exposure routes.

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the Co-60 is present as vitamin B_{12}, for example, the dose factor would be between 3000×10^{-6} and 8100×10^{-6} mrem/pCi (ICRP, 1969). The dose factor for this organic form of Co-60 is thus 1400 to 3800 times greater than the dose factor used by the GRS and SSK. This aspect was not investigated either. However, we are afraid that the exclusive use of dose factors that are valid only for the inorganic form of the radionuclide will result in significant underestimation of the radiation dose since animal and vegetable foodstuffs are the source of the radionuclide.

8.4. Transferability of Animal Experiments to Man

In the great majority of cases the values used to determine the dose factors are results of animal experiments that are applied to man. The most commonly used experimental animals are rats, mice and guinea pigs, that is to say, animal species whose organisms are much smaller than the human organism. As a rule, the smaller an organism is, the faster is the rate at which physiological processes occur in it. A readily apparent example of this is the higher respiratory rate and heart rate in the smaller organism. Furthermore, due to the smallness of the organs, the metabolic exchange processes occur more rapidly, so that the biological half-lives of elements in a given organ are shorter than they are in larger organisms. In regard to the application to human beings of radionuclide biological half-lives determined in animal experiments, as has been done in many cases, e.g., by the International Commission on Radiological Protection, there is a definite danger that the dose factors, and thus the radiation exposure, will be systematically and significantly underestimated.

8.5. Transfer Factors Gastrointestinal Tract - Organ

The investigation of the transfer factors for the passage of radionuclides from the gastrointestinal tract to the various organs resulted in an incorrect calculation of the dose factors for the most important radionuclides by the International Commission on Radiological Protection and the Association for Reactor Safety.

The transfer factor f_A consists of two parts: f_1 describes the passage of radionuclides from the gastrointestinal tract into the blood, and f_2 describes the passage of the radionuclides from the blood into the individual organs ($f_A = f_1 \cdot f_2$).

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Example: Strontium

The International Commission on Radiological Protection gives the values necessary for calculation of the dose factors in "Recommendations - Permissible Dose in Incorporation of Radionuclides" (ICRP, 1966). On p. 96 a value of 0.21 is given as the transfer factor for the transfer of the radionuclides Sr-85 m, Sr-85, Sr-89, Sr-91 and Sr-92 from the food into the bones. These strontium isotopes are short-lived and thus relatively undangerous nuclides. Although the individual isotopes of an element show exactly the same chemical behavior, a significantly smaller value (0.09) is assigned to Sr-90, which has a half-life of 28.5 years and is thus by far the most dangerous isotope of strontium.

The value of 0.21 is based on a scientific paper (Jowsey, J. et al., 1953) and can therefore be regarded as scientifically acceptable. The low value of 0.09, on the other hand, is taken from a personal communication (Durbin to Morgan, 8/7/1958) which cannot be checked and which apparently is unpublishable. Consequently, there is no reason to pay further attention to this value.

Example: Plutonium

Depending on the chemical form in which it is present, plutonium is absorbed in highly variable quantities in the gastrointestinal tract. While only about one millionth of the plutonium present in the food in the form of the insoluble, inorganic compound plutonium dioxide is absorbed in the gastrointestinal tract, the absorption rate increases by a factor of about 10,000 for soluble or organic plutonium compounds. As the operating time of a reactor increases, an increasingly large percentage of the plutonium deposited on the ground is made available to plants by the action of microorganisms and is able to reach man via the metabolism of food plants. Therefore, it is totally unrealistic to base calculations of plutonium dose factors on insoluble plutonium dioxide (as has been done by the GRS and SSK). In a basic study in which plutonium dose factors are calculated, the GRS writes the following on p. 18: "For poorly soluble material like PuO_2 we assume a blood-absorbed fraction of $f_w = 10^{-4}\%$ in man. For soluble plutonium compounds, particularly plutonium citrates and nitrates, transfer factors of up to about 2% have been measured. The present model computation assumes poorly soluble material for all actinides and uses an absorption value of $f_w = 10^{-4}\%$." (GRS-I, p. 18, 1977). Although the GRS is acquainted with the fact that the absorption of organic and soluble plutonium compounds is significantly greater than that of inorganic and insoluble plutonium compounds, it bases its calculation of plutonium dose factors on a transfer factor that is too low by a factor of 20,000. This transfer factor enters linearly into the calculation of the dose factor and thus into the calculation of the plutonium radiation exposure.

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8.6. Microdistribution of the Radionuclides

A uniform distribution of the radionuclide in the organ is normally assumed in calculations of dose factors. However, this can result in considerable underestimation of the radiation exposure.

An example is plutonium:

It was formerly assumed that plutonium distributes itself uniformly in bone. However, recent studies show that plutonium becomes concentrated entirely in the organic parts of bone, especially in the osteogenetic cells, and is not incorporated in the mineral phase of the bone (Priest, 1977). Consequently, the radiosensitive part of the bone is exposed to significantly higher radiation doses than would be expected by assuming homogeneous distribution.

8.7. Organs Investigated

Only a few organs are considered in the assessments of the GRS and in the mathematical models of the SSK (whole body, thyroid, liver, kidney, gastrointestinal tract and bone). However, it is precisely these organs which are either only moderately radiosensitive (bone and thyroid) or actually relatively radioresistant (liver, kidney, lung, muscle) (Pabst et al., 1976; Barth, G. et al., 1968). The radiosensitive organs (Pabst, 1976; Barth, 1968) are systematically disregarded:

- embryonal tissue

- lymphatic tissue

- thymic tissue

- bone marrow

- testis

- ovary

8.8. Radiation Exposure of Embryos

One of the most serious radiobiological problems is the accumulation of radioactive substances during pregnancy and the irradiation of the human embryo, which is by far the most radiosensitive stage of human life due to the rapid rate of cell division and the determinant development of all organs. Various studies

show that human embryos are 100 to 1000 times more radiosensitive than human adults (e.g., Stewart, A., 1973).

The formation and destruction of cells in the adult human body are in a state of transient equilibrium. Therefore, the biological half-life of absorbed radionuclides is shorter in the adult than in the embryo. Embryonal and fetal tissue are formed at a relatively fast rate, while the occurrence of destructive processes is very limited. This high rate of formation results in a high rate of assimilation. Radionuclides are also assimilated at a fast rate; they are incorporated with a high biological half-life in the body of the embryo and later the child. It follows, therefore, that the embryo receives the highest radiation dose at a given concentration of radionuclides in the environment.

Although this has been known for a long time, the embryonal radiation exposure has never been considered in any assessment for nuclear power plants in West Germany. Even the mathematical models of the Radiation Protection Commission (SSK), which will be included in the legal regulations in the future, do not consider the embryonal stage in any way. As in many other cases, therefore, we are confronted with an intolerable discrepancy between the actual licensing procedure for nuclear power plants and the explicit requirement stated in section 45 of the Radiation Protection Law that radiation exposure be considered at the "most unfavorable points of exposure" and that "all relevant exposure pathways" be taken into account.

The following table gives the dose factors used for children and adults.

Table 8-1. Dose commitment factors, ingestion (in mrem/pCi)

	H 3	Co 58	Co 60	Zn 65	Sr 89	Sr 90	J 131	Cs 134	Cs 137	Ce 144	Pu 239
bone	0	$1,12\cdot10^{-5}$	$3,64\cdot10^{-5}$	$1,71\cdot10^{-5}$	$4,40\cdot10^{-4}$	$9,6\cdot10^{-2}$	$4,16\cdot10^{-6}$	$1,25\cdot10^{-4}$	$2,03\cdot10^{-4}$	$5,3\cdot10^{-7}$	5.78
whole body	$1,3\cdot10^{-7}$	$1,12\cdot10^{-5}$	$3,64\cdot10^{-5}$	$3,34\cdot10^{-5}$	$1,11\cdot10^{-5}$	$2,26\cdot10^{-3}$	$3,52\cdot10^{-6}$	$1,47\cdot10^{-4}$	$9,21\cdot10^{-5}$	$2,62\cdot10^{-8}$	$1,28\cdot10^{-2}$
liver	$1,3\cdot10^{-7}$	$8,79\cdot10^{-6}$	$2,88\cdot10^{-5}$	$7,67\cdot10^{-5}$	-	-	$5,96\cdot10^{-6}$	$2,2\cdot10^{-4}$	$1,36\cdot10^{-4}$	$2,06\cdot10^{-7}$	$7,12\cdot10^{-2}$
kidney	$1,3\cdot10^{-7}$	$8,73\cdot10^{-7}$	$2,04\cdot10^{-6}$	$5,13\cdot10^{-5}$	-	-	$1,02\cdot10^{-5}$	$1,59\cdot10^{-3}$	$1,44\cdot10^{-3}$	$1,22\cdot10^{-7}$	$5,45\cdot10^{-2}$
lung	$1,3\cdot10^{-7}$	$1,12\cdot10^{-5}$	$3,64\cdot10^{-5}$	$3,34\cdot10^{-5}$	-	-	-	$1,9\cdot10^{-5}$	$1,25\cdot10^{-5}$	$2,62\cdot10^{-8}$	-
thyroid	$1,3\cdot10^{-7}$	$1,12\cdot10^{-5}$	$3,64\cdot10^{-5}$	$3,34\cdot10^{-5}$	-	-	$1,95\cdot10^{-3}$	$1,47\cdot10^{-4}$	$9,21\cdot10^{-5}$	$2,62\cdot10^{-8}$	-
bone marrow	$1,9\cdot10^{-7}$	$1,12\cdot10^{-5}$	$3,64\cdot10^{-5}$	$3,34\cdot10^{-5}$	-	$8,86\cdot10^{-4}$	-	$1,47\cdot10^{-4}$	$9,21\cdot10^{-5}$	$2,62\cdot10^{-8}$	-

The relevant dose factor for babies for the thyroid gland (I-131) was assigned a value of 2.8×10^{-2};

after (Bacher D., Müller W., 1977; Bonka H., Brüssermann K., 1973; Comper W., 1972; Eisenbud M., 1973; ICRP, 1966; ICRP, 1974; ICRP, 1969; Priest N.D., 1977; SSK, 1977; Thorne U.C., 1976; UCRL, 1968;)

9. Results of the Radioecological Computer Program

The radiation doses to be expected from consumption of foodstuffs from the area around the planned nuclear power plant in Wyhl are compiled in the following tables. All values are given in the unit millirems per year.

Table 9-1. Expected radiation exposure for adults by the exhaust air pathway at the maximum radiation receiving point (the values given for wine refer to Limberg)

	bone	whole body	liver	kidney	lung	thyroid	bone marrow
leafy vegetables	323.1570	10.66030	6.338	58.23085	0.5636070	6.493669	6.490270
root vegetables	1699.026	39.82692	0.9219438	5.536096	0.05800158	0.3575549	15.81009
potatoes	463.7017	128.7335	185.3232	1925.761	15.95267	125.3524	126.4571
grains	415.0981	13.13156	7.802197	72.18167	0.7400115	4.815944	8.026098
pork	178.4991	80.73059	117.1302	1236.743	10.99872	85.58655	80.63095
beef	903.9614	348.8726	510.7510	5287.656	47.07425	378.0032	346.6206
milk	836.7500	162.1351	222.1370	2285.408	20.96605	208.7419	154.5775
wine	937.2998	109.7058	143.7951	1383.262	13.30292	95.96688	101.1307
grand total	5757.414	893.7932	1195.847	12254.75	111.1560	904.9138	839.7419

	bone	whole body	liver	kidney	lung	thyroid	bone marrow
leafy vegetables							
CO 58	0.004153952	0.004153952	0.001260112	0.0003237857	0.004153952	0.004153952	0.004153952
CO 60	0.02714225	0.02714225	0.02147518	0.001521159	0.02714225	0.02714225	0.02714225
ZN 65	0.0067902596	0.01912705	0.04392351	0.02937779	0.01912705	0.01912705	0.01912705
SR 89	0.02416037	0.0006095131	0	0	0	0	0
SR 90	256.5998	6.746099	0	0	0	0	2.644709
J 131	3.035649276	0.004780155	0.03309377	0.01185159	0	2.643099	0
CS134	0.2507516	0.2949309	0.4413932	3.150065	0.03812633	0.2949309	0.2949309
CS137	7.714931	3.500209	5.168603	54.72638	0.4750555	3.500208	3.500208
CE144	0.0031777972	8.789224'-06	6.510611'-05	4.072651'-05	8.789224'-06	6.789224'-06	8.789224'-06
PU239	23.56690	0.06324857	0.3518227	0.2693026	0	0	0
total	323.1570	10.66033	6.038639	59.23085	0.5636078	6.493659	6.490278
root vegetables							
CO 58	7.466663'-05	7.466663'-05	5.859997'-05	5.819998'-05	7.466663'-05	7.466661'-05	7.466663'-05
CO 60	0.007134393	0.007134393	0.005644795	0.0003995396	0.007134393	0.007134393	0.007134393
ZN 65	0.002010195	0.003926352	0.009016506	0.006030597	0.003926352	0.003926352	0.003926352
SR 89	0.004554218	0.0001174133	0	0	0	0	0
SR 90	1674.315	39.41620	0	0	0	0	15.45254
J 131	0	0	0	0	0	0	0
CS134	0.3197499	0.02337303	0.03497993	0.2528093	0.003020939	0.02337330	0.02337300
CS137	0.7120337	0.3230458	0.4770275	5.050879	0.04384444	0.3230459	0.3230458
CE144	1.479288'-05	7.312709'-07	5.743697'-06	3.405154'-06	7.312709'-07	7.312709'-07	7.312709'-07
PU239	23.96515	0.05307163	0.2952108	0.2259651	0	0	0
total	1699.026	39.82692	0.8219438	5.536096	0.05800158	0.3575549	15.81009
potatoes							
CO 58	0.0002732798	0.0002732798	0.0002144759	2.130116'-05	0.0002732798	0.0002732793	0.0002732798
CO 60	0.01404311	0.01404311	0.01111103	0.0007870316	0.01404311	0.01404311	0.01404311
ZN 65	0.004493977	0.008777514	0.02015675	0.01348163	0.008777514	0.008777514	0.008777514
SR 89	0.0008975994	2.264359'-05	0	0	0	0	0
SR 90	152.2098	3.583274	0	0	0	0	1.404770
J 131	0	0	0	0	0	0	0
CS134	5.210558	6.127076	9.159774	66.27246	0.7919359	6.127076	6.127076
CS137	262.0754	118.9022	175.5777	1859.057	16.13755	118.9022	118.9022
CE144	4.307338'-05	2.129535'-06	1.674367'-05	9.916154'-06	2.129535'-06	2.129535'-06	2.129535'-06
PU239	44.13694	0.09785336	0.5441094	0.4166415	0	0	0
total	463.7017	128.7335	185.3232	1925.761	16.95267	125.0524	126.4571

	bone	whole body	liver	kidney	lung	thyroid	bone marrow
grains							
CO 58	0.002114559	0.002114559	0.001655551	0.0001569223	0.002114559	0.002114559	0.002114559
CO 60	0.07060868	0.07060868	0.05546620	0.003957190	0.07060868	0.07060868	0.07060868
ZN 65	0.01559519	0.03046078	0.06995035	0.04678558	0.03046078	0.03046078	0.03046078
SR 89	0.01473397	0.0001194359	0	0	0	0	0
SR 90	347.8269	8.188430	0	0	0	0	3.210155
J 131	0	0	0	0	0	0	0
CS134	0.3695749	0.4347377	0.5536279	4.702265	0.05619053	0.4347377	0.4347377
CS137	9.529269	4.278007	6.317144	66.98741	0.5836199	4.278007	4.278007
CE144	0.0033447966	1.704467*-05	0.0001343151	7.936827*-05	1.704465*-05	1.704466*-05	1.704466*-05
PU239	57.17921	0.1270681	0.7068161	0.5410324	0	0	0
total	415.0981	13.13156	7.802197	72.18167	0.7400115	4.815544	8.026098
pork							
CO 58	7.566219*-05	7.566219*-05	5.936139*-05	5.827597*-06	7.566219*-05	7.566219*-05	7.566219*-05
CO 60	0.001067733	0.001067733	0.0003447954	5.583996*-05	0.001067733	0.001067733	0.001067733
ZN 65	0.05334479	0.1053658	0.2419623	0.1518343	0.1053658	0.1053658	0.1053658
SR 89	0.0003445386E3	1.124799*-05	0	0	0	0	0
SR 90	6.198609	0.1439256	0	0	0	0	0.05720802
J 131	0.01069397	0.009640741	0.01532117	0.02622379	0	5.312798	0
CS134	3.806555	4.476521	6.599555	48.41951	0.5785980	4.476521	4.476521
CS137	167.4931	75.99072	112.2121	1198.128	10.31351	75.99072	75.99072
CE144	1.733688*-06	8.570305*-08	6.738487*-07	3.959751*-07	8.570305*-08	8.570305*-08	8.570305*-08
PU239	0.8446506	0.001870506	0.01043468	0.007564265	0	0	0
total	178.4091	80.73059	119.1902	1236.743	10.99872	85.58655	80.63095
beef							
CO 58	0.01731768	0.01731769	0.01359129	0.001345951	0.01731768	0.01731769	0.01731768
CO 60	0.1225223	0.1225223	0.09694070	0.006366634	0.1225223	0.1225223	0.1225223
ZN 65	0.1404251	0.2742806	0.6259690	0.4212754	0.2742805	0.2742806	0.2742806
SR 89	0.01752177	0.0004420264	0	0	0	0	0
SR 90	149.5675	3.521079	0	0	0	0	1.380387
J 131	0.06588774	0.05913593	0.1001279	0.1713600	0	32.75999	0
CS134	19.45515	21.70326	32.49108	234.7496	2.805194	21.70325	21.70326
CS137	712.2043	323.1230	477.1418	5952.090	43.85495	323.1233	323.1230
CE144	7.683534*-05	3.800746*-06	2.988371*-05	1.769811*-05	3.800746*-06	3.800745*-06	3.800746*-06
PU239	23.36661	0.05174612	0.2878376	0.2203253	0	0	0
total	903.9614	348.8726	510.7510	5287.656	47.07425	379.0032	346.6206

	bone	whole body	liver	kidney	lung	thyroid	bone marrow
milk							
CO 58	0.004802555	0.004802555	0.003769150	0.0003743421	0.004802555	0.004802555	0.004802555
CO 60	0.03389566	0.03389566	0.02681954	0.001899647	0.03389566	0.03389565	0.03389566
ZN 65	9.3543121	0.7701768	1.769643	1.182936	0.7701768	0.7701768	0.7701768
SR 89	0.06096637	0.001538016	0	0	0	0	0
SR 90	520.4685	12.25317	0	0	0	0	4.803678
J 131	0.1257093	0.1054447	0.1902303	0.3094479	0	58.36797	0
CS134	7.972498	9.375658	14.03159	101.4101	1.211820	9.375658	9.375658
CS137	307.6726	139.5893	296.1254	2182.504	18.94536	139.5893	139.5893
CE144	0.0001383935	6.841342*-06	5.379070*-05	3.135652*-05	6.841342*-06	6.841342*-06	6.841342*-06
PU239	0.03655199	8.191598*-05	0.0004556756	0.0003487999	0	0	0
total	836.7930	162.1351	222.1370	2285.409	20.96605	208.7419	154.5775
wine							
CO 58	0.04502398	0.04502398	0.03533579	0.003509459	0.04502398	0.04502399	0.04502398
CO 60	0.8501579	0.6501579	0.6726524	0.04764622	0.8501579	0.8501579	0.8501579
ZN 65	0.99080375	0.1775097	0.4076347	0.2726423	0.1775097	0.1775097	0.1775097
SR 89	0.06075063	0.001519960	0	0	0	0	0
SR 90	570.3547	13.42710	0	0	0	0	5.263902
J 131	0	0	0	0	0	0	0
CS134	17.81049	20.94514	31.34647	226.5495	2.707135	20.94514	20.94514
CS137	162.7724	73.84695	109.0495	1154.641	10.02291	73.84695	73.84895
CE144	0.002190666	0.0001082933	0.0008514563	0.0005042662	0.0001082933	0.0001082933	0.0001082933
PU239	165.3144	0.4103850	2.282765	1.747342	0	0	0
total	937.2998	109.7058	143.7951	1383.262	13.80292	95.86688	101.1307

Table 9-2. Expected radiation exposure for adults by the waste water pathway from utilization of Rhine water.

values given in mrem/yr

	bone	whole body	liver	kidney	lung	thyroid	bone marrow
fish consumption	176.2276	71.57874	104.0446	941.5823	10.44731	112.4420	70.62004
leafy vegetables	186.1969	4.860202	0.7329037	7.250619	0.08275950	0.6627634	2.210464
root vegetables	1051.023	24.79065	0.07103920	0.6644819	0.01215461	0.04993546	9.751193
potatoes	52.76889	6.730520	8.532829	88.62352	0.7906929	5.777252	6.154695
irrigation - milk	128.2620	8.652502	8.771429	98.84274	0.8707602	6.564141	7.004134
irrigation - meat	61.61485	14.33774	20.03094	205.3853	1.867026	13.92429	13.86211
livestock watering - milk	2.402912	0.2124233	0.2334452	1.580675	0.06297459	0.6290392	0.1946537
livestock watering - meat	1.046027	0.2886032	0.4026296	3.542059	0.05017297	0.5182378	0.2834612
grand total	1659.538	131.4510	142.8195	1337.467	14.18403	140.5673	110.0803

The following values were obtained for the consumption of fish from the cooling water outlet.

fish consumption	[illegible]	117.3215	173.5316	1543.407	17.12688	177.9997	115.7621

	bone	whole body	liver	kidney	lung	thyroid	bone marrow
fish consumption							
H 3	0	0.004497164	0.004497164	0.004487164	0.004487164	0.004497164	0.006558165
CO 58	0.07093328	0.07093328	0.05567003	0.005528991	0.07093328	0.07093328	0.07093328
CO 60	0.6915937	0.5915957	0.5471797	0.03875993	0.6915937	0.6915997	0.6915997
ZN 65	0.2707500	0.5298332	1.214417	0.8122500	0.5288332	0.5238332	0.5288332
SR 89	0.5573330	0.01435990	0	0	0	0	0
SR 90	63.79997	1.431333	0	0	0	0	0.5611333
J 131	0.09042174	0.07651067	0.1275465	0.2217071	0	42.38519	0
CS134	23.74998	27.92999	41.79999	302.0999	3.609999	27.92999	27.92999
CS137	65.99666	40.83099	60.29332	638.3999	5.541666	40.93099	40.83099
total	176.2276	71.57874	104.0446	941.5823	10.44751	112.4420	70.62004
leafy vegetables							
H 3	0	0.004487164	0.004497164	0.004487164	0.004487164	0.004487164	0.006558165
CO 58	0.0007802665	0.0007802665	0.0006123697	6.081399*-05	0.0007802665	0.0007802665	0.0007802665
CO 60	0.009509493	0.009509493	0.007523995	0.0005329456	0.009509493	0.009509493	0.009509493
ZN 65	0.001516200	0.002961466	0.006800730	0.004548598	0.002961456	0.002961466	0.002961466
SR 89	0.06966662	0.001757503	0	0	0	0	0
SR 90	185.1358	4.358409	0	0	0	0	1.708651
J 131	0.0003477759	0.0002942719	0.0004932557	0.0008527156	0	0.1630203	0
CS134	0.05224999	0.06144599	0.09195995	0.6646199	0.007941995	0.06144599	0.06144599
CS137	0.9269655	0.4205592	0.6210212	6.575519	0.05707916	0.4205592	0.4205592
total	186.1969	4.860202	0.7327037	7.250619	0.08275950	0.6627634	2.210464
root vegetables							
H 3	0	0.004467164	0.004497164	0.004487164	0.004497164	0.004437164	0.006558165
CO 58	1.773331*-05	1.773331*-05	1.391749*-05	1.392249*-06	1.773331*-05	1.773331*-05	1.773331*-05
CO 60	0.001383199	0.001383199	0.001094359	7.751936*-05	0.001383199	0.001333199	0.001383199
ZN 65	0.001895249	0.0003701839	0.0008500915	0.0035685748	0.0003701830	0.0003701830	0.0003701830
SR 89	0.005573329	0.0001405999	0	0	0	0	0
SR 90	1050.527	24.74059	0	0	0	0	9.699188
J 131	0	0	0	0	0	0	0
CS134	0.004156247	0.004387749	0.007314995	0.05296749	0.0006317499	0.004887749	0.004887749
CS137	0.08549670	0.03878944	0.05727865	0.6064798	0.005264580	0.03878944	0.03878944
total	1051.023	24.79065	0.07103920	0.6644819	0.01215461	0.04993546	9.751193

9001 9006

	bone	whole body	liver	kidney	lung	thyroid	bone marrow
potatoes							
H 3	0	0.008076899	0.008076899	0.008076899	0.008076839	0.008076899	0.01180470
CO 58	3.191998E-05	3.191998E-05	2.505149E-05	2.468048E-06	3.191598E-05	3.191998E-05	3.191998E-05
CO 60	0.001556099	0.001556099	0.001231199	8.720996E-05	0.001556099	0.001556099	0.001556099
ZN 65	0.0001949390	0.0003807598	0.0003743796	0.0005848198	0.0003807593	0.0003837591	0.0003807598
SR 89	0	0	0	0	0	0	0
SR 90	40.49277	0.9532679	0	0	0	0	0.3737147
J 131	0	0	0	0	0	0	0
CS134	0.2746686	0.3230104	0.4834167	3.493786	0.04174564	0.3230104	0.3230104
CS137	11.59969	5.444197	8.039206	85.12099	0.7388976	5.444197	5.444197
total	52.76889	6.730520	8.532829	88.6[illegible]352	0.7906929	5.777252	6.154695
irrigation-milk							
H 3	0	0.03230760	0.03230760	0.03230760	0.03230760	0.03230760	0.04721880
CO 58	0.0002553598	0.0002553598	0.0002004119	1.970439E-05	0.0002553593	0.0002553598	0.0002553598
CO 60	0.003734618	0.003734639	0.002954878	0.0002993039	0.003734639	0.003734630	0.003734638
ZN 65	0.02339290	0.04569119	0.1049256	0.07017839	0.04569119	0.04569119	0.04569119
SR 89	0.07022353	0.001771559	0	0	0	0	0
SR 90	116.0063	2.730984	0	0	0	0	1.070642
J 131	0.001377192	0.001165316	0.001973092	0.003376769	0	0.6455587	0
CS134	0.4445999	0.5228495	0.7824959	5.555311	0.06757915	0.5228495	0.5228495
CS137	11.71216	5.313745	7.846572	83.08134	0.7211923	5.313745	5.313745
total	128.2620	8.652502	8.771429	88.84274	0.8707602	6.564141	7.004134
irrigation-meat							
H 3	0	0.008974332	0.008974332	0.008974332	0.008974332	0.008974332	0.01311633
CO 58	0.001099466	0.001099465	0.0008628348	8.569947E-05	0.001099465	0.001099466	0.001099466
CO 60	0.01521519	0.01521519	0.01263339	0.00108527154	0.01521519	0.01521519	0.01521519
ZN 65	0.009339100	0.01628806	0.03740403	0.02501733	0.01628806	0.01628806	0.01628806
SR 89	0.01950665	0.0004520097	0	0	0	0	0
SR 90	33.43997	0.7872334	0	0	0	0	0.3086233
J 131	0.0007998843	0.0006758249	0.001145983	0.001961255	0	0.3749458	0
CS134	1.031937	1.213558	1.816209	13.12624	0.1558545	1.213558	1.213558
CS137	27.09799	12.29421	18.15431	192.2221	1.668595	12.29421	12.29421
total	61.61485	14.33774	20.33094	205.3853	1.867026	13.92429	13.86211

	bone	whole body	liver	kidney	lung	thyroid	bone marrow
livestock watering-milk							
H 3	0	0.03230760	0.03230760	0.03230760	0.03230750	0.03230760	0.04721880
CO 58	0.0001276799	0.0001276799	0.0001002059	9.952196'-06	0.0001276799	0.0001276799	0.0001276799
CO 60	0.001244879	0.001244879	0.0009843591	6.976794'-05	0.001244879	0.001244879	0.001244879
ZN 65	0.03740771	0.01446887	0.03322643	0.02222315	0.01446887	0.01446887	0.01446887
SR 89	0.02006399	0.0005061596	0	0	0	0	0
SR 90	2.129790	0.05152800	0	0	0	0	0.02020080
J 131	0.001001595	0.0008475028	0.001434977	0.002455833	0	0.4694974	0
CS134	0.01847499	0.04524659	0.06771554	0.4394019	0.005943133	0.04524659	0.04524659
CS137	0.1457946	0.06614619	0.09767514	1.034207	0.008577499	0.06614619	0.06614619
total	2.402912	0.2124233	0.2334452	1.580675	0.06297469	0.6290392	0.1946537
livestock watering meat							
H 3	0	0.008974332	0.0[illegible]974332	0.009974332	0.008974332	0.008974332	3.01311633
CO 58	0.0003546665	0.0003546665	0.0002733500	2.764499'-05	0.0003546665	0.0003546665	0.0003546665
CO 60	0.003457958	0.003457998	0.002735999	0.0001937999	0.003457998	0.003457998	0.003457998
ZN 65	0.001570350	0.003067232	0.007043611	0.004711047	0.003067232	0.003067232	0.003067232
SR 89	0.005573329	0.0001405999	0	0	0	0	0
SR 90	0.6079997	0.01431333	0	0	0	0	0.005611330
J 131	0.0005216618	0.0004414078	0.0007473839	0.001279080	0	0.2445299	0
CS134	0.08906245	0.1047375	0.1567500	1.132874	0.01353750	0.1047375	0.1047375
CS137	0.3374875	0.1531162	0.2261000	2.393999	0.02078125	0.1531162	0.1531162
total	1.046027	0.2886032	0.4026296	3.542059	0.05017297	0.5182378	0.2834612

The following values were obtained for the consumption of fish from the cooling water outlet.

fish consumption							
H 3	0	0.008704953	0.008704953	0.008704953	0.008704953	0.008704953	0.01272262
CO 58	0.1162736	0.1162736	0.09125406	0.009063117	0.1162736	0.1162736	0.1162736
CO 60	1.133668	1.133658	0.8969679	0.06353521	1.133658	1.133668	1.133668
ZN 65	0.4430125	0.8668617	1.990667	1.331437	0.8668617	0.8668617	0.8668617
SR 89	0.9135764	0.02304710	0	0	0	0	0
SR 90	99.66312	2.346236	0	0	0	0	0.9198079
J 131	0.1357445	0.1140146	0.1939475	0.3303837	0	63.16150	0
CS134	38.93091	45.78276	63.51963	455.2312	5.917533	45.78276	45.78276
CS137	147.5227	66.93002	98.83261	1046.462	9.083881	66.93002	66.93002
total	298.8582	117.3215	170.5316	1543.407	17.12583	177.9997	115.7621

10. Emission Sources, Exposure Pathways and Factors Not Considered

10.1. Emissions from the Secondary Circulation

Complete tightness of heat exchangers cannot be guaranteed. Therefore, there is the danger that radioactive substances from the primary circulation, which is under high pressure and which is radioactively contaminated af_er even short operating times, will leak into the secondary circulation. Heating pipe damage in the steam generators was observed in 22 of the water-cooled power reactors inspected in 1975 (GRS, no. 19). "It is usually the result of uniform corrosion" (GRS, no. 21). It can be assumed, therefore, that this type of damage will occur more and more frequently with increasing length of operation. According to figures given in the GRS assessment (GRS, 1976, pp. 3-6), about 12,000 tons of steam per year are discharged to the environment from the secondary circulation without first passing through exhaust gas systems or filtration systems. Since the secondary circulation contains about 600 tons of water, this means that a quantity of water or steam equivalent to the capacity of the secondary circulation is discharged from the secondary circulation to the environment once every 18 days on the average. This means that a considerable quantity of radioactive emissions may leave the nuclear power plant in this way without being filtered or controlled. Moreover, these emissions occur at a low emission height, which means that there is about ten times less meteorological attenuation of these emissions than of the smokestack emissions.

Although the GRS is aware of this problem, the GRS assessment (1976) does not include a determination of the radiation exposure to be expected from these emissions.

The Department of Environmental Protection is preparing a study of this problem.

10.2. Fruit and Other Foodstuffs Not Considered

Considerable quantities of fruit of all types are cultivated in the immediate vicinity of Wyhl (see Table 7.1.3-1).

Satisfactory determinations of the radiation exposure that might be expected from consumption of this fruit is not possible because no studies have been performed for determination of transfer factors.

In this case it does not seem advisable to use any one variety of fruit as a representative for all other varieties, as was

done in the 1976 assessment of the Association for Reactor Safety. Neither does it seem appropriate to transfer the required data from other plants since it must be assumed that fruit, in view of its peculiar morphology, also exhibits physiologically specific behavior. There is evidence in the literature that certain varieties of fruit show characteristic enrichment with certain elements.

Raspberries, black currants and gooseberries contain relatively large concentrations of Zn, Mn and Co; these concentrations are correlated with the supply of these elements in the soil (Krupyshev, P.V., 1967). It was also shown that in these types of fruit the Mn content increases during the period of flowering and fruit growth, so that aside from the leaves, the highest concentration is found in the carpels (Krupyshev, P.V., 1969). Apples contain high concentrations of Fe, Mn, Co and Sr (Vigorov, L.I., Sumenkova, T.N., Pashilov, V.A., 1972, and Shkvaruk, N.M., Moiseichenko, V.F., Shkvaruk, R.N., Khanchak, N.E., Shinyan, O.I., 1972). Furthermore, a correlation was found between the surface texture of fruits and their capacity for absorption of radiostrontium (Merten, D., Buchheim, W., 1967).

These references strengthen our suspicion that for individual varieties of fruit there may be both a high level of enrichment of certain nuclides and high nuclide-specific absorption through the surface. It therefore seems critically important that further studies be performed and that separate calculations be performed for the transfer factors of the following fruits:

Apples

pears

quinces

plums

mirabelle plums

peaches, apricots

red and black currants

gooseberries

strawberries

raspberries

blackberries

bilberries

whortleberries

sea buckthorn

sloes

hips

elderberries

rhubarb

In addition, it is necessary to expand the list of foods considered in the GRS assessment since most people have a somewhat more varied diet. The expanded list should include the following:

beans

peas

tomatoes

cucumbers

paprika

eggplant

zucchini

gourd

mangel-wurzel

onions

garlic

horseradish

celery

red beets

comfrey

asparagus

mushrooms (some mushrooms, especially boletus, are known for their high Cs-137 content (BMI, 1974)

nuts (hazelnuts, walnuts, chestnuts, almonds)

spices (parsley, chives, savory, balm; seed spices: fennel, caraway, anise, mustard)

teas: peppermint, sage, chamomile

lamb meat

poultry

snails

game meat

Important radioecological data is presently lacking for almost all of these previously unconsidered foods. The universal factors, e.g., for all plants, that are recommended by the SSK (1977) are scientifically unacceptable.

Consequently, nothing can be said at this time about radiation exposure via possibly important exposure pathways due to lack of basic knowledge.

10.3. Other Factors Not Considered

The following factors were also disregarded:

sediment radiation at the banks of the Rhine from radionuclides that have settled out of the waste water

radiation exposure by iodine-131, tritium in the exhaust air, carbon-14 and all radioactive decay products of the noble gases emitted from the nuclear power plant

radiation exposure for babies (other than by I-131) and embryos.

11. Summary

1. In its calculation of expected radiation exposure, the Department of Environmental Protection used an ecological model of computation on which most such assessments for nuclear power plants are based (e.g., exhaust air assessment of the Institute for Reactor Safety, waste water assessment of the Bavarian Biological Testing Institute for the planned nuclear power plant in Wyhl).

2. The Department of Environmental Protection made an effort to proceed on the basis of realistic assumptions that would make it possible to obtain a realistic estimate of the radiation exposure that might be expected from the planned nuclear power plant (Kernkraftwerk Süd, Wyhl).

Overly conservative assumptions were avoided. For example, our expert analysts thought it unrealistic to assume an annual consumption of 880 l of Rhine water below the nuclear power plant. Such an assumption, the conservativeness of which is emphasized by the SSK and the BBV, would, in the estimation of the Department of Environmental Protection, result in a whole body dose of only about 1.8 mrem/year anyway.

Furthermore, the Department of Environmental Protection considers it unrealistic to assume that many totally unclothed persons would loiter near the fence surrounding the nuclear power plant. This assumption would be relevant only for the beta submersion dose, which was calculated to be 0.8 mrem/year in this unrealistic case. As in other cases, examination of this case showed that in the recommendations of the Radiation Protection Commission and in the assessments of the GRS and BBV, overly conservative assumptions were made mainly in regard to exposure pathways whose contribution to the total radiation exposure is rather insignificant.

Our experts at the Department of Environmental Protection therefore proceeded on the basis of realistic assumptions.

3. It was found that the uniform bases of calculation used by the SSK, BGA, BMI, TÜV, GRS, BBV and the Jülich and Karlsruhe nuclear power plants, which produce the well-known result that the maximum radiation dose in the vicinity of a nuclear power plant is below the level specified in section 45 of the Radiation Protection Law, are incomplete and incorrect and do not meet the requirements set forth in section 45.

In particular, we found the following errors:

The meteorological long-term dispersion factor assumed in the GRS assessment was about 2.5 times too low, so that the meteorological attenuation was about 2.5 times too high. This factor enters linearly into the calculation of the radiation doses. The principal cause of this error is an overly coarse gradation of the wind speed classes in the GRS assessment, with the result that weather situations involving light winds, in which radioactive emissions are only slightly attenuated, were averaged out.

The assumed nuclide spectrum for radioactive aerosols was not conservative. In particular, the percentage of cesium-137 that was used was too small.

The enrichment factors for the passage of radionuclides from the soil into crop plants were between 10 and 1000 times too low in the most critical cases (see, for example, the figures on pp. 41, 44 and 48).

The transfer coefficients for the passage of radionuclides from forage into beef, pork and milk were between 10 and 100 times too low in the most critical cases (see, for example, the figure on p. 51). These transfer coefficients enter linearly into the calculation of the radiation exposure by these exposure pathways.

The transfer factors for the passage of radionuclides from foodstuffs into the bloodstream via the gastrointestinal tract were between 10 and 20,000 times too low (see, for example, plutonium on p. 91). These transfer factors enter linearly into the calculation of the radiation exposure.

The value assigned for the biological half-lives of radionuclides in the human organism were too low for some radionuclides.

The nuclide composition of the radioactive noble gases was totally unrealistic. Consequently, the calculated radiation exposure from radioactive noble gases was about 5 times too low.

The errors listed above apply to an equal or similar extent to the bases of computation of radiation exposure that are recommended by the Radiation Protection Commission (SSK, 1977).

It was also found

that important exposure pathways, such as the submersion radiation exposure from radionuclides from the exhaust air emission deposited on the ground, were not calculated in the GRS assessment.

4. In the present assessment the Department of Environmental Protection calculated the radiation exposure to be expected from the following exposure pathways:

radiation exposure from noble gases

radiation exposure from ground radiation

radiation exposure from consumption of the following foodstuffs: milk, beef, pork, cereal grains, potatoes, leafy vegetables, root vegetables, wine and fish.

5. These calculations show that the permissible maximum values specified in section 45 of the Radiation Protection Law would be clearly and sometimes greatly exceeded.

The most important results are summarized below. The complete results of the computer programs are given in section 9 above.

Radiation Doses from Radioactive Exhaust Air

ADULTS (All values in mrem/yr)

	whole body	thyroid	kidney	bone
noble gas radiation	31	31	31	31
ground radiation	15	15	15	15
total of various foodstuffs	784	809	10,872	4,820
wine	110	96	1,383	937
Total	940	951	12,300	5,803
maximum value in accordance with section 45 of the Radiation Protection Law	30	90	90	180

Babies (only iodine-131

	thyroid
cow's milk	753
goat's milk	2,204
sheep's milk	2,501
maximum value in accordance with section 45 of the Radiation Protection Law	90

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Radiation Dose by Radioactive Waste Water with Utilization of Rhine water

Adults

	whole body	thyroid	kidney	bone
fish+	72 (117)	112 (178)	942 (1,543)	176 (289)
other foodstuffs	60	28	396	1,483
Total	131	141	1,338	1,660
Maximum value in accordance with section 45 of the Radiation Protection Law	30	90	90	180

+The values in parentheses give the radiation doses for consumption of fish from the cooling water outlet.

The radiation doses given above are based on only one of the two nuclear power plant blocks that are planned for Wyhl.

6. The following factors, influences and exposure pathways, all of which cause an increase in the radiation exposure, have not been taken into consideration in the above results:

discharge of unfiltered radioactivity through safety valves and steam valves and from the machine house

discharge of radioactivity from the fuel element storage tanks, in which the consumed fuel elements presumably would have to be stored for very long periods of time due to difficulties with the planned reprocessing

emissions of the radionuclides iodine-131, tritium (exhaust air), carbon-14 and all radioactive decay products of the emitted noble gases

exposure pathways fruit, nuts, mushrooms, spices, game meat etc.

sediment radiation at the banks of the Rhine from radionuclides that have settled out of the waste water

radiation exposure by inhalation (inhalation of radionuclides)

effects of the structure and water release of the planned cooling towers (reduction of the meteorological attenuation and increase in the wet deposition)

increase in the nuclide concentration of the Rhine during the irrigation period in the summer due to above-average quantities of emissions while the fuel elements are being changed

the calculation of the radiation exposure was performed only for adults. For babies it was performed only for radiation exposure by iodine-131. It was not performed for the most sensitive stage of human life, i.e., the embryonal stage, because important radiological bases of calculation are unavailable.

preexisting radiation loads from radioactive emissions of other nuclear power plants and nuclear facilities with the exhaust air.

Although none of these problems has yet been taken into consideration (mainly because of a lack of scientific data), it can definitely be said on the basis of the effects calculated in this assessment, that the legally stipulated maximum permissible values will be significantly exceeded with the design of the planned pressurized water reactor in Wyhl, even if the plant is properly operated as authorized.

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Appendix

Biological Effect of Ionizing Radiation

1. Basic introduction
2. Excursus: Radiation damage on the molecular level
3. Statistical studies on radiation damage in man
4. Discussion of the tolerance limit
5. Discussion of the dose factors
    a) calculation, average person
    b) radiosensitivity of the tissues
    c) inhomogeneous microdistribution
6. Radiation damage in the embryo and fetus
    a) elevated organ concentration
    b) dependence on the stage of development
    c) threshold value, late damage
    d) statistical studies on the human embryo

1. Basic Introduction on the Effects of Radiation on Man

Due to its ionizing effect, the radiation emitted in radioactive disintegration causes damage to or destruction of human cells, organelles and biological molecules. The extent of the damage depends on the penetrating power of the radiation, the duration of the irradiation, whether the whole body is irradiated or only parts of it, and on other factors. We can distinguish between latent and manifest damage, i.e., early and late damage.

We shall be concerned to discuss the somatic effects of low doses of radiation to which people are exposed from the "normal" operation of nuclear power plants, rather than the effects of a nuclear accident (which are death, acute radiation sickness and early damage).

Even small radiation doses (mrem range) are able to cause impairment of vital organs and manifest and fatal disorders of the organism. Many results of radiobiological experiments and extensive statistics prove this. In particular, many irradiation experiments have been performed on animals in order to determine the risk to man of radiological testing and handling methods.

2. Excursus: Radiation Damage on the Molecular Level

Radioactive radiation has a variety of biological consequences. Independent of the location of the radiation source (i.e., inside or outside a cell), the following types of damage may occur:

1. Damage to the cell membrane (e.g., Petkau, A., 1972; Scott, K.G., et al., 1973)

2. Changes in the enzyme pattern (e.g., Mintzel-Landbeck, L. Hagen, U., 1976; Cerutti, P.A., 1974)

3. Effects on the genetic substance of DNA (deoxyribonucleic acid) (e.g., Weish, P., Gruber, E., 1975; Várterész, V., 1966).

Radiobiology is concerned especially with effects on the genetic substance because changes in this molecular area and all of the consequences of these changes are passed on to the daughter cells during cell division (e.g., mutations, transformation of a healthy cell into a cancer cell). Although the probability that damage will be caused is greater when large doses are involved, the disintegration of a single radionuclide can be enough to cause the mutation of one or more cells. For example, the disintegration of one tritium particle incorporated in the DNA can cause an average of 2.1 strand breaks in the DNA (Cleaver, J.E., et al., 1972). As the discussion of the

molecular events shows, any determination of a dose threshold value must be considered arbitrary.

Unfortunately, only a few papers have been published on the biological effects of natural radiation and on synergistic environmental effects. It has been experimentally demonstrated that a decrease in the natural background radiation induces a decrease in the cell growth rate (IAEA, 1976), which is concrete evidence that even natural radiation affects cellular mechanisms. This conclusion is supported by the results of epidemiological studies in Kerala (India), where a high level of background radiation caused by thorium-containing rock is associated with a high incidence of mongolism (Down's syndrome) and other mental deficiencies of genetic origin (Kochupillai, N., 1976).

The question of the reversibility of radiation damage has been under discussion for several years. Enzymatic mechanisms are known which are able to repair the DNA strand breaks mentioned above. In in vitro experiments the best results that have been achieved so far have been repair of 80% of artificially induced strand breaks (Mitzel-Landbeck, L. et al., 1976).

The efficiency of these mechanisms in vivo depends on the type of radiation source. The strand breaks produced by UV radiation are not repaired by the same enzymes as strand breaks caused by gamma rays (Hariharan, P.V., Cerutti, P.A., 1976). I-125 incorporated in DNA produces strand breaks, a maximum of 50% of which can be repaired (Painter, R.B. et al., 1974). Little is known about the consequences of unrepaired strand breaks. In most cases they probably result in the death of the cell. An error in the repair can have disastrous results (mutations).

Ionizing radiation can produce not only strand breaks, but also chemical changes in the subunits of the DNA (Cerutti, P.A., 1974). This damage can be identical to damage produced by alkylating agents (including such environmental poisons as benzpyrene and nitrosamines) (Trosko, J.E., Chu, E.H.Y., 1975). The carcinogenic effect of ionizing radiation is demonstrable by the identity of damage produced by ionizing radiation and alkylating agents (so-called radiomimetic substances).

3. Statistical Studies on Radiation Damage in Man

The most important result of these molecular biochemical experiments and of many animal experiments is the significant increase in cancer as a late consequence of low-level radiation (Little, J.B. et al., 1975). There is a linear relationship between the amount of time by which life is shortened and radiation dose (Bacq, Z.M., Alexander, P., 1955); this linear relationship can also be observed

in the mrem dose range. The shortening of life is a result of many kinds of late damage, e.g., loss of vitality with diminished resistance to disease, early onset of aging processes, and diminished power of regeneration.

Statistics from the United States and Japan confirm the results of these experiments. Some of the problems in this area which have been statistically analyzed are the following: Disease status of workers in nuclear power plants (Mancuso, T.F. et al., 1976; Wagoner, J.K. et al., 1963), effects of increased natural radioactivity (Gentry, J.T. et al., 1959; Kochupillai, N., 1976; Pincet, J., Marsé, L., 1975), atomic bomb victims of Hiroshima and Nagasaki (Jablon, S. et al., 1965; Brill, A.B. et al., 1962), radiation injury in pregnant women (Stewart, A., Kneale, G.W., 1970). A common finding in these studies was a significant increase in morbidity and mortality of the individuals involved, primarily in regard to cancer.

A few examples:

Among workers in nuclear power plants the incidence of cancer is clearly a function of radiation exposure (Mancuso, T.F. et al., 1976). The incidence of congenital deformities is directly dependent on the radiation dose from natural radiation (study on 1.24 million infants in New York State) (Gentry, J.T. et al., 1959). In a study on 19 million children in Great Britain, Stewart at Oxford University determined that a doubling of the number of radiographs of pregnant women is associated with a doubling of the risk of leukemia and other forms of cancer in their offspring, and that irradiation with 80 mram in the first trimester is enough to double the incidence of leukemia and cancer in children up to 10 years old (Stewart, A., Kneale, G.W., 1970). In West Germany a dose of 60 mrem/year by nuclear power plants is allowed; the thyroid dose for children may be 90 mrem/year. The risk of irradiation of unborn children and newborns is discussed in greater detail in section 6.

A comprehensive study by the National Academy of Science in America, which was commissioned by HEW (BEIR report, 1976), contains estimates of the risk for cancer, leukemia and hereditary diseases from chronic low doses of radiation. On the basis of the data in this report, we have determined the following results of an additional radiation dose of 60 mrem/year in West Germany (the legally permitted dose):

a) an additional 40 to 700 cases per year of serious dominant hereditary disease in the first generation,

b) a fivefold increase in the number of cases of hereditary disease after a few more generations, and

c) an additional 500 - 1400 cancer deaths per year.

In a risk assessment performed for the AEC, Gofman, J.W. et al. (1971) estimate even higher values (2000 to 12,000 cancer cases per year at 60 mrem additional radiation exposure for West Germany).

The BEIR report contains a critical discussion of the validity of this type of risk assessment:

"It is clear that these estimates are subject to great uncertainty. The range of credible values is large, and there is no guarantee that the actual values fall within this range. We are well aware that future information will necessitate revisions. The estimates that have been given are not exact scientific values (as scientists we would prefer to reserve judgment until reliable information becomes available), but they are reasonable values that are based on present scientific knowledge, and however rough and uncertain these estimates may be, they are better than no estimates at all since some degree of orientation is advisable when we are dealing with radioactivity."

The population that will be affected and the politicians have to have some idea of the risk; the known qualitative effects of radioactive radiation must be quantified. The presently available estimates are the only quantifications that are possible at this time. In the course of this discussion we shall elaborate on the difficulties that prevent absolutely certain knowledge of these relationships (no radioactive irradiation experiments on man, observation time too short, sometimes difficult diagnosis of the diseases caused by radiation, cause-and-effect relationships difficult to survey, etc.).

The quantifications of the radiation risk from nuclear power plants that are given in the large studies (e.g., BEIR report, UNSCEAR report, papers by Gofman or Tamplin) are not overestimates since only the mean values were taken from the possible risk range. This fact is illustrated by the BEIR report, in which various mathematical models and various parameters were used to estimate the cancer death risk in the United States if the maximum permissible radiation levels were reached; values of 2000 to 9000 deaths per year were obtained from these calculations, but the final estimate that was given was 3000 to 4000 deaths per year. If conservative assumptions are desired, i.e., if we wish to consider the greatest possible damage that could result from the use of nuclear technology, then we would have to work with figures from the upper range of these estimates.

4. Discussion of the Tolerance Limit

In light of present knowledge, it is no longer possible for any radiologist to determine a so-called threshold value, below which

genetic and somatic damage will not occur. Stokke et al. found damage of bone marrow cells at a dose of only 8 mrem (Stokke, T. et al., 1968).

Although radiation doses in the mrem range had at one time been demonstrated to have biological effects only on fungi, damage to rat bone marrow has now been observed after doses of only a few mrem Sr-90 per week. Lung cancer has been induced in hamsters with Po-210 doses of only a few mrem (I ctle, J.B. et al., 1970). The eminent radiation geneticist H.J. Muller stated on the basis of his experiments with fruit flies (Drosophila), that there is no threshold value below which ionizing radiation is ineffectual. To a certain degree of probability, every single ionization must be regarded as effectual (Beck et al., 1959). This is especially true where the development of cancer is concerned. It could well be that it is precisely low radiation doses that are capable of giving rise to cancer since high doses damage the cells so severely that they die and are then usually dissimilated by the organism, so that cancer cannot develop. Low doses also damage the cells, but usually not so severely that the cells are unable to continue living and dividing. Multiplication of the cells results in multiplication of the cell damage, which in turn may result in the development of a malignant tumor. There is no minimum dose below which radioactivity can be regarded as definitely not cancer-inducing (Oberling, C., 1944).

In regard to carcinogenesis, it must be considered that the disease is usually triggered by several factors (BEIR report, 1972), and that the induction of a tumor must be regarded as the result of radiation damage to <u>one</u> or only a few cells. The BEIR report therefore assumes a linear dose-effect relation with no threshold value for carcinogenesis and leukemogenesis. The threshold value is a hypothetical quantity that is not supported by any theory of tumor induction or empirical evidence.

There is still another difficulty. Cancer often does not appear until many years after the causative damage has occurred. There may be a latent period of 5 to 30 years (Cleaver, J.E. et al., 1972). The long dormancy of the cancer makes it much more difficult to establish causal connections between irradiation (cause) and cancer (effect), i.e., many cancer patients will never know that their disease was caused by the action of radiation (including, for example, x-ray examinations).

5. Discussion of the Dose Factors

a) By definition, the dose factors establish a connection between organ dose and the time integral of the concentration, i.e., they express the biological radiation load by a given quantity of radioactivity, specifically in the organs of the human body, in the

form of a single numerical value. The following list of the parameters that have to be considered shows how many imponderables and individual differences cannot be taken into account.

The radiation dose in an organ depends on the following factors:

1. Quantity of the absorbed nuclide
2. Type of radiation emitted
3. Energy of the radiation
4. Half-life of the nuclide in the organ, dwell time
5. Selection factor of the organ
6. Mass and form of the organ
7. Distribution of the activity in the organ
8. Chemical compound of the incorporated substance

These parameters depend in turn on the following:

age

sex

state of health

genetic constitution

food composition

In estimating the radiation exposure from a nuclear facility, the estimate is based on the "critical" points, e.g., on the "critical" population group, i.e., the group which will be affected most strongly by the radiation exposure.

These risk calculations are based on the so-called "average" person, i.e., on a model of the adult human being in which rigid physical and physiological parameters are used (e.g., 70 kg body weight, 1.2 l fluid intake per day etc.). However, this does not represent the critical population group, for there are population groups whose physical and physiological characteristics lead to greater radiation exposure.

For example, liquid intake is higher in certain metabolic diseases (e.g., diabetes mellitus). All metabolic diseases result in changes in the normal parameters due to changes in metabolic processes, so that, for example, the dwell time of incorporated radionuclides in the organism may be increased. The radiation dose is thus increased. (Belcher, E.H. et al., 1971; Barth, G. et al., 1968).

Furthermore, population groups in which previous damage has been caused by other noxae must also be regarded as critical groups. This means that the effect of the additional radiation exposure will be greater in these groups than in groups in which there is no preexisting damage. The following examples illustrate this:

Previous damage to an organ (the lungs of smokers, the skin of photosensitive individuals) enhances the effect of ionizing radiation and thus increases the danger of radiation damage.

A number of chemical compounds, e.g., certain antibiotics, sex hormones and the like, increase the radiosensitivity of tissues, which are then more susceptible to damage.

Metabolic changes, such as diabetes mellitus, nephrosis, hyperthyroidism and the like, result in greater radiosensitivity of the organism due to previous injury of the tissues, increased dwell time of the radionuclide, or restriction of possible repair mechanisms. (It is precisely the metabolic changes characteristic of a metabolic disease and the associated changes in the radionuclide distribution in the organism and tissues which are utilized in nuclear-medical diagnostics.). (Belcher, E.H. et al., 1971; Emrich, D., 1971; Barth, G. et al., 1968)

These effects show that our calculations should not be based on the group of average persons, but rather on the population group in which deviations from the normal parameters of the average person result in an increase in radiation dose. The customary procedure of basing these calculations on the normal population does not conform to present knowledge in medical science, which is now concerned mainly with cybernetic control systems and multifactorial cause-effect analyses, and which for the most part now rejects schematic-mechanical approaches to the human body.

b) Radiosensitivity of the Tissues

The dose factors that have been calculated in the past have generally been for the bone, lung, liver, kidney, and gastro-

intestinal tract and in some cases for the spleen, thyroid and muscle. The testbooks on nuclear medicine and radiation protection judge most of these tissues to be moderately radiosensitive (bone, thyroid) or even relatively radioresistant (liver, kidney, muscle) (Pabst et al., 1976; Sauter, 1971; Barth, G. et al., 1968). Only the spleen and gastro-intestinal tract exhibit a high degree of radiosensitivity. The other highly radiosensitive tissues are the following:

embryonal tissue

lymphatic tissue

thymic tissue

bone marrow

testicular tissue

ovarian tissue

(Pabst et al., 1976)

Specific dose factors are not used for these highly radiosensitive tissues, i.e., the moderately radiosensitive tissues and relatively radioresistant tissues are considered in most calculations of the relative biological activity of nuclear power plant emissions, while the highly radiosensitive tissues are not included in the calculations. The radiosensitivity of cells increases with their reproductive activity and decreases with their degree of differentiation (Barth, G. et al., 1968). The more frequently and faster a cell multiplies and divides (the process of cell division is the most radiosensitive), the less specialized it is and the more radiosensitive it is. This is true of embryonal tissue, hematopoietic tissue (lymph nodes, spleen, thymus, bone marrow), white and red blood cells, and gonadal tissues (testis, spermatozoa, ovary, and follicle). This provides the pathophysiological explanation of the fact that primarily embryonal damage, carcinoma of the hematopoietic tissues, especially leukemia (i.e., cancer of the white blood cells), and genetic defects occur in individuals who are exposed to radioactivity.

c) Inhomogeneous Microdistribution

Furthermore, it is necessary to distinguish not only among the various radiosensitive tissues, but also to consider the specific exposure <u>within</u> each organ. Bone is an example. It consists of marrow, compact substance (shaft), spongy substance (epiphysis), a layer of cartilage (covering the articular surface of the bone), and

the endosteum (tissue lining the medullary cavity) and periosteum (tissue covering the outside of the bone). Radionuclides are not uniformly distributed in this bone system (Sugahara and Hug, 1971):

> Alkaline earths are deposited principally in the epiphysis and the calcification zones during the growth period; in adults they are deposited mainly in the mineral zone under the periosteum, in the endostium, and in newly formed compact substance and spongy substance. This applies to the elements calcium, radium, strontium, barium and phosphorus.
>
> Depending on their manner of incorporation and chemical structure, plutonium and thorium are deposited in bone, especially in the periosteum, endosteum and bone marrow.
>
> Americium, rare earths and transuranium elements become concentrated in the bone marrow and at the surface of the bone.

Our criticism of the dose factors can be expanded. Other problems include the following: important radionuclides are disregarded, there is no specific evaluation of the organic damage with respect to effects on the total organism, undiscoverable damage, e.g., of the immune system, synergism, potentiating effect of damage to more than one organ. However, the questions that have been raised should be reason enough to regard the use of the customary dose factors for calculations of the radiation exposure as completely unsatisfactory.

6. Radiation Damage in the Embryo and Fetus

The greatest radiological problem is without doubt the accumulation of radioactive substances during pregnancy and subsequent irradiation of the human embryo.

a) Elevated Organ Concentration

The formation and destruction of cells are in transient equilibrium in the adult human body. This means that radionuclides absorbed by the adult human body have a shorter biological half-life than radionuclides absorbed by the embryo. Embryonal and fetal tissue is formed at a relatively fast rate; destructive processes occur only to a limited extent. This high rate of formation results in a high rate of substance absorption, including radioactive substances; the absorbed radionuclides are then incorporated in the body of the embryo and later the child with a high biological half-life. It follows that for a given concentration of radionuclides in the environment, the embryo has by far the greatest radiation exposure.

Experiments on rats and dogs have shown that 38% of the strontium and 66% of the cesium intake of the mother animal reaches the fetus transplacentally and is deposited in it (Sikov, M.R., Mahlun, D.D., 1969). In the pig (the experimental animal whose physiology is closest to human physiology) the deposition of radiostrontium in the fetuses is ten times greater than in the uterus and placenta. Thus, radiostrontium is able to pass through the placental barrier (Werner, H., 1971). Long-term studies on large and small mammals with extremely low doses of radioactive iodine (I-131), e.g., absorbed from atmospheric radioactive fallout, showed absorption rates of mother to fetus of 1 : 3 (Book S.A., Goldman, M., 1975; Eisenbud, M., 1968). A fetal thyroid gland contains 4 to 5 times the dose of I-131 as the thyroid of an adult.

The artificial plutonium radioisotope becomes distributed through the entire body in the course of fetal development, but especially large amounts are deposited in the bones and liver. Pu-239 is present in the newborn's liver in amounts 20 times greater than in the adult liver. Calculation of the Pu-239 radiation dose required to induce cancer showed that the value for adults is 11 times greater (45 times greater in the case of leukemia) than the dose that is sufficient for tumor development in the newborn (Sikov, M.R. and Mahlun, D.D., 1972).

When these high absorption rates are converted for the body weight of a fetus (e.g., a fetus weighing 1 g compared to a woman weighing 60 kg), we find that the child in utero is exposed to a radiation dose that is several powers of ten greater.

b) Dependence on the Stage of Development

Comprehensive experiments performed by Wilson and Russell on rats and mice show that the radiosensitivity is much greater in the early stages of embryonal development than in the later stages (Braun et al., 1973; Hug, Zuppinger, 1972; Wilson, J.G., 1973). The zygote (the fertilized ovum before implantation in the uterus) is extremely radiosensitive; it is destroyed by relatively low doses of radiation and then resorbed; if it survives, exencephaly (brain located outside the skull) and cataract are likely to occur. Irradiation during the organ-forming stage (in the human embryo in the second and third weeks) causes (according to the authors cited above) a lower resorption rate but a high incidence of deformities, e.g., hydrocephalus, microphthalmia (abnormal smallness of the eyes), anencephaly (absence of a brain), micrencephaly (abnormal smallness of the brain), deformities of the teeth, nose, retina and herniation. In the fetal phase (in man, from the fifth week after conception until birth) the CNS (especially the cerebellum) and the eyes are highly radiosensitive; total absence and defective development of these organs have been observed.

c) Threshold Value, Late Damage

In experiments with relatively low radiation doses (5 R) (e.g., wholebody dose allowed by the West German Radiation Protection Law for occupationally exposed persons, women in nuclear power plants), investigators have found significant genetic damage, growth retardation, accelerated aging processes and skeletal changes (Wilson, J.G., 1973; Jacobsen, L., 1968; Medic[illegible]morandum on the Industrial Use of Nuclear Energy, 1976). In va[illegible]xperimental animals (e.g., rat, cow) radionuclide concentratio[illegible]i/g body weight increase the death rate of newborns and redu[illegible]eight and fertility.

Another important problem must be considered in this connection. Many types of embryological radiation damage are undiscoverable, especially after irradiation with small doses over long periods of time. (Quote from Braun et al., 1973: "Our knowledge about radiation damage in the fetal stage is unsatisfactory. Developmental anomalies induced by ionizing radiation in the late fetal period show definite morphological/anatomical manifestations only in a very small number of cases and can be discovered only by histological and biochemical methods. Radiation damage of this sort is no less important than "drastic" deformities; the damage simply may not manifest itself until the postnatal stage, by which time it has become difficult to recognize the qualitative connections between cause and effect.")

The types of damage involved here are growth retardation, losses of activity, nervous disorders and biochemical defects (Wilson, J.G., 1973; Medical Memorandum on the Industrial Use of Nuclear Energy, 1976).

The animal experiments prove that there is also no threshold value or tolerance limit for embryonal radiation damage (Jacobsen, L., 1968). Genetic defects, chronic changes and cancer can be caused by even the smallest doses of radiation (Wilson, J.G., 1973).

d) Statistical Studies on the Human Embryo

It is becoming increasingly apparent that the results of animal experiments very probably are valid in man as well. The evaluation of data from the initial phase of radiotherapy, consequences of radiography, studies on the atomic bomb victims of Hiroshima and Nagasaki, and extensive statistics from recent years show significant effects of radiation on embryos and infants.

Stewart found an almost 100% increase in leukemia and other cancers in children whose mothers had had x-rays of the abdomen during the first trimester of pregnancy compared to children whose mothers had not had obstetric x-rays (Stewart, A., 1973; Stewart, A., Hewitt, D., 1965; Stewart, A. and Kneale, G.W., 1970; Stewart, A. et al, 1958). Studies by MacMahon and Kneale confirm this (MacMahon,

B., 1962; Kneale, G. W., 1971). Even children whose parents had been x-rayed 5 to 15 years <u>before</u> the mother's pregnancy had a significantly greater chance of developing leukemia (Gibson, R.W. et al., 1968; Kessler, I.I. and Lilienfeld, A.M., 1969).

Many other studies could be cited. However, the results that have been given are sufficient to show that the child in utero represents by far the most radiosensitive stage of human life and the stage that is most threatened by radiation. This fact has not been considered in any way in the licensing procedure for nuclear power plants.

According to the presently lawful West German Radiation Protection Law, the radiation exposure resulting from leakage of radioactive substances into the air or water from nuclear power plants must be kept "as low as possible" and may not exceed 30 mrem/yr for the whole body or 90 mrem/yr for the thyroid. In addition, section 45 of the Radiation Protection Law stipulates: "This radiation exposure must be computed for the most unfavorable cases and points of exposure, with due consideration being given to all relevant exposure pathways, including the food chains."

Therefore, calculations of the radiation exposure expected for man must be submitted as part of the licensing procedure for nuclear power plants. However, these calculations are based on the physiological conditions in a "healthy, average person". Unfavorable cases, such as sick and old persons, children, and especially children in utero, are not taken into consideration. This procedure does not comply with the explicit demands of the law.

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ABBREVIATIONS USED IN THE TEXT

BBV = Bavarian Biological Testing Institute.

Ci = Curie. Unit for the quantity of a radioactive substance in which 37 billion disintegrations per second occur.

GRS = Association for Reactor Safety, a private company which prepares assessments for nuclear power plants. Financed by the federal government and by the TÜV.

KWS = Kernkraftwerk Süd GmbH, Karlsruhe.

mrem = Unit of Radiation Dose.

pCi = picocurie = 10^{-12} Curie.

RSK = Reactor Safety Commission. Advises the Secretary of the Interior. 13 members, exclusively advocates of nuclear energy.

SSK = Radiation Protection Commission. Advises the Secretary of the Interior. 13 members, exclusively advocates of nuclear energy.

SSVO = Radiation Protection Law.

TÜV = Industrial Supervisory Association. Industrial association whose members include, for example, all nuclear power plants and manufacturers of nuclear power plants.
Prepares assessments on nuclear power plants by commission of the licensing authorities.

BMI = Federal Department of the Interior.

BMBW = Federal Department of Education and Science.

BGA = Federal Bureau of Health.

IRS = Institute for Reactor Safety.

DWD = German Weather Service.