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ABSTRACT

A revision of the most necessary input data for the BIOPATH code has been performed. Both nuclide independent and nuclide dependent parameters have been treated.

Especially for the uptake in feed chains statistical properties are given to enable sensitivity analyses of the results.

In appendix the present diet and food consumption habits are reviewed where average intake values are given both for Sweden and for different parts of the world.

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1 INTRODUCTION

An application of the radioecological model BIOPATH (82) was used to calculate the radiological contributions to man from leakage of groundwaterborne nuclides from a repository. BIOPATH is a computer code based upon compartment theory. The turnover of the nuclides in the biosphere is modelled by transfer coefficients (turnover/year) between selected reservoirs. This work presents a revision of some necessary input data needed for such calculations. The parameters can be divided into the following two groups:

- nuclide independent parameters
- nuclide specific parameters.

The nuclide independent parameters, with the exception of some common transfer coefficients, constitute site or area specific values for

- consumption of food-stuff for man
- animal diet
- yield of terrestrial and aquatic food-stuffs
- retention on the surface of vegetation.

The group of the nuclide-specific parameters has in its turn two main groupings:

- turnover in the biosphere
- up-take in food-chains.

This work has been performed to provide a better basis for the calculations, and when possible intervals are quoted for the parameters. The work is also aimed to give statistical properties of the parameters to enable sensitivity analyses of the results to be performed.

The work has been concentrated upon the following nuclides:

- technetium
- iodine
- cesium
- radium
- thorium
- protactinium
- uranium
- neptunium.

Some modifications of the BIOPATH-model have been carried out since the first version, and the parameters studied in this report refer to the latest version (82).

2 NUCLIDE INDEPENDENT TRANSFER COEFFICIENTS

2.1 Deposition

The most commonly used deposition velocity is 0.01 m s^{-1} (49). There are variation intervals of up to about four orders of magnitude in that value due to differences in the chemical or physical form of deposit, type of deposition surface and meteorological conditions. Dry deposition is a function of particle size, ground roughness, meteorological and vegetation conditions. In (60) estimates for the long term average tropospheric turnover times for submicron particles are about one week. For iodine the corresponding value is from one to ten years. This transfer is of minor importance for the doses when the spreading of the nuclides is governed by contaminated groundwater, and the doses are mostly determined by the concentration in the water and the subsequent concentration in soil. The transfer factor chosen from atmosphere to ground is 52 $year^{-1}$. The transfer to a reservoir, which is directly connected to the atmosphere, is then obtained by the proportions of the reservoir to total area.

2.2 Resuspension

The resuspension process is usually described in radioecological models by the use of a resuspension factor. This is taken as the annual average ratio of the concentration of a radionuclide in the air due to resuspension to the quantity of the radionuclide per unit area of ground surface, expressed in m^{-1}.

Resuspension factors are very variable depending upon a number of factors such as meteorological conditions and vegetation cover, as well as sources and magnitudes of disturbances. Nor is the process quite well understood.

However, another way of deriving the transfer coefficients has been used, as it seems more convenient to get the transfer rate expressed in $years^{-1}$: by use of an erosion formula developed by agricultural scientists such as Chepil (64). The erosion rate is among other things a function of the soil erodibility index, the soil roughness factor, the climatic factor and the equivalent vegetative cover.

The erosion rate for a wheat-field in Texas, USA, is calculated in the report to vary from 0 to 18 tons per acre and year. However, that does not give the whole picture as most of the erosion occurs by creep and saltation. It is little known about the resuspended fraction, but supposed values are in the range of 3 to 40 %.

Following assumptions are used:
- 1 ton per acre and year is used as the erosion rate,
- 10 percent of that is the resuspended fraction
- the depth of the soil is 30 cm and the bulk density is 1.4 ton/m^3

This will give a transfer rate of $6 \cdot 10^{-5}$ $year^{-1}$. Naturally, as pointed out above this is a particularly site specific factor and varies a great deal depending upon the prevailing conditions. The exposure by inhalation is derived from the concentration in soil instead from the atmosphere the importance of the

resuspension process is, however, of minor importance for the turnover of the nuclides. We think the value derived above might be rather high, although for example the value for denutation from the Fyris river is about $3 \cdot 10^{-3}$ mm year^{-1} (79).

2.3 Transport from the upper ocean to the continents

The transport of salt from seawater to land is dominated by seaspray. The transfer of nuclides is described by the same value as for the transport of salt. Evidently, seaspray is not the only transfer mechansim: other processes are also probable. Six stations sampling airborne particulate and deposited material in the vicinity of Windscale, England, have quoted radionuclide concentrations in excess of those which are to be expected from nuclear weapon material (62).

However, the transfer coefficient which will be used in the calculations is the same as has been used previously, $1.7 \cdot 10^{-7}$ year^{-1}, since it has not been possible to obtain other useful information. This transfer is of limited interest since for most nuclides the exposure on the global area is dominated by the consumption of aquatic food-stuffs.

2.4 Turnover of water in soil

The migration of nuclides in soil depends upon a large number of factors such as the physiochemical form of the element, the composition of the soil, the pH, the organic content of the soil, and the competing anion or cation species.

For some nuclides specific information is available about the migration rates based upon experience from releases, fallout or experiments. For all nuclides there are however retardation or distribution factors which describe their mobility compared to that of water. In the absence of other data we have used these to describe the migration rates from the soil reservoirs.

The water velocity in the ground can be described by Darcy´s law (76)

$$V = k \cdot I$$

where V = water velocity (m/time unit)
k = permeability (m/time unit)
I = hydraulic gradient (dimensionless)

For vertical flow I = 1 which gives V = k.

The permeability in unsorted soils can vary between 10^{-11} - 10^{-5} ms^{-1}. We have chosen a value of 10^{-7} ms^{-1} for the upper soil as the permeability decreases with depth.

That results in an assumed water turnover in the upper 30 centimeters of about 10 times a year. For the deeper soil reservoir, with a supposed permeability of $5 \cdot 10^{-8}$ ms^{-1}, the corresponding turnover of water takes about 0.5 year, giving a transfer coefficient for the water of about 2 $year^{-1}$.

About the same value is obtained if the so-called sponge model for the water balance is used. The sponge model implies that the pore volume of the soil must be saturated before any addition can be expected to the ground water nor

is there any surface water run-off. If a water balance is made for the system the following relationship is valid

$$Q = P - E$$

where Q = run-off
P = precipitation
E = evapotranspiration

In this case the potential ground water formation in a year is about the same as the run-off during the year. "Potential ground water formation" is defined as the maximum amount of the annual precipitation that can possibly form groundwater.

In the following table the mean values (in mm) of the parameters mentioned above (77) are summarized for different areas in Sweden.

Table 1 Mean values of runoff, evapotranspiration and precipitation in mm per year for different areas in Sweden

Area	Q	E	P
Halland	440	545	985
Gotland	175	485	660
Södermanland	205	475	680
Västmanland	305	485	790
Uppland	215	470	690
Northern Jämtland	750	170	920
Northern Lappland, excluding the mountain region	350	245	595
Northern Lappland including the mountain region	685	190	825

Different types of soils have different field capacities, that is the capacity to retain water after drainage. Typical values for different soils are shown in the table below.

Table 2 Field capacity for different species of soil

Type of soil	Field capacity mm
Moraine	65 - 140
Sand	40
Sand with gravel	15 - 25
Fine sand	120 - 130
Sandy fine gravel	10 - 20
Fine gravel	2 - 4
Gravel	2
Peat	200

The variations are considerable. In clay soils for example a transfer rate of 2-3 $year^{-1}$ is obtained.

2.5 Turnover of water in the global zone

The global water balance used for the turnover of nuclides in the global zone has been studied more closely (78).

Another report which also discusses the sizes and flows of global water is reference (61).

There are some discrepancies between the two reports concerning the magnitude of the soil moisture and the ground water, but they are of little importance. The values quoted in the reports give turnover times between 200 days and one year for the soil moisture, and about 330 years for the ground water.

We therefore suppose that in the future the values used for the global water cycle will be the following:

surface soil - deep groundwater
$0.38\ year^{-1}$

surface soil - well mixed layer
$0.44\ year^{-1}$

deep groundwater - surface soil
$2.9 \cdot 10^{-3}\ year^{-1}$

deep groundwater - well mixed sea
$1.7 \cdot 10^{-4}\ year^{-1}$.

It can also be mentioned that the previous groundwater volume was taken to be $6 \cdot 10^4$ km^3, which is not really adequate in this study. A better value is a factor of ten lower, which describes the amount of water involved in more active water turnover better than the previous values which also included the deeper situated water.

3 NUCLIDE DEPENDENT TRANSFER COEFFICIENTS

3.1 Sedimentation - freshwater system

For most nuclides the only available information about sedimentation is reported as distribution factors k_d: the quotient between the amount of nuclides in the sediment and the amount of nuclides in the water expressed in (m^3 kg^{-1}).

By using the k_d-factors and the following expression (58) the transfer to the sediment is received.

$$k_{ws} = k_d \cdot s / h_m (1 + k_d \cdot ss) /^{-1} \qquad (1)$$

where

k_d = the sediment concentration factor (m^3 kg^{-1})
s = sedimentation rate (kg m^{-2} $year^{-1}$)
h_m = mean water depth (m)
ss = the amount of suspended material in the water (kg m^{-3}).

The sedimentation growth in lakes is about 4-6 mm per year (66), naturally depending upon type of lake. For these calculations a value of 5 mm per year has been chosen. The upper sediment is very rich in water, probably up to 60-70 %. Therefore this sedimentation growth will give about 1.5 kg m^{-2} for the transport of material. The amount of suspended material in the lakes depends upon a large number of factors, but most values are between (0.1-10) 10^{-3} kg m^{-3} for the phytoplankton content (67).

The contents of zooplankton and detritus are of the same magnitude.

There can also be contributions from the nearby vegetation. However, it is to be expected that the major constituent in lake consists of those already mentioned. This gives $3 \cdot 10^{-3}$ kg m^{-3} as an approximate average value.

In flowing water there is more suspended material, and inorganic material from denutation can be of the same order of magnitude as the organic.

There are considerable variations in the material transport in Swedish rivers in different parts of the country. Specific values are quoted in reference (79).

In this work the transfer from the sediment to the water has been determined for an imaginary lake with the following properties

- mean depth: 10 m
- amount of suspended material: $3 \cdot 10^{-3}$ kg m^{-3}
- sedimentation growth: 1.5 kg m^{-2}.

This implies that the transfer coefficients presented below ought to be adjusted to site specific conditions.

The importance of the transfer from the water to the sediment for the dose calculations naturally depends upon the nuclide. In the short term low sedimentation rates can increase the doses because of the assumptions made. On the other hand, if the lake can be expected to dry-up in the future with contempory use of the

sediment as farming-land, low sedimentation rates can underestimate the exposure under those conditions.

Technetium

There appears to be little known about the behaviour of technetium in fresh-water systems. In reference (31) technetium has been released to an experimental fresh-water pond and most of it remained in solution in the water. After the 36 days the experiment lasted a few percent of the released technetium were found in the sediments. This is in good agreement with the value obtained from the expression (2) which is $0.03\ year^{-1}$. Thus a distribution factor of 200 (68) was used.

Iodine

There are also difficulties to find pertinent information about the behaviour of iodine in fresh-water systems. However, the literature gives about the same distribution factor for that nuclide as for technetium, namely 200 (68), which in turn gives a transfer rate of about $0.03\ year^{-1}$.

Cesium

Cesium has been studied for a number of years, as considerable amounts were attributed to global fallout in the sixties.

The adsorption of cesium in clays is well-known and has also been used in connection with dating sediments. The sedimentation varies with the type of lake. In (69) the fraction of cesium

added to a lake that is found in the sediment is 0.38.

With a concentration factor of 30 000 (68) the transfer rate will be 2.3 $year^{-1}$. As high sedimentation rates can decrease the concentration of the nuclide in the water, a lower value has been chosen. For many releases of cesium to freshwater, the high bioaccumulation to fish will be the dominant contribution by that exposure pathway. In order not to underestimate the exposure a lower value has been chosen. A transfer rate of 0.38 $year^{-1}$ has therefore been chosen to be representative for cesium.

Radium

Radium has been studied mostly in connection with mining and milling.

Distribution factors are available for some Swedish lakes. Lakes situated in northern Sweden in the vicinity of an uranium mineralization had distribution coefficients about $1.2 \cdot 10^5$ (70). For other Swedish lakes the corresponding values are about 10^4 to $2 \cdot 10^5$ (51). If the value $1 \cdot 10^5$ is chosen as a base for the calculations, the transfer rate will be 3.8 $year^{-1}$ according to expression 2. However this value might overestimate the transfer rate to the sediments since the measurable content of radium can depend on the generation of radium from its mother nuclide thorium.

Thorium

In several north American lakes the geochemical behaviour of thorium, uranium and plutonium has

been studied (80). Distribution coefficients were obtained for the nuclides. The coefficients were regressed against limnological parameters such as mean depth, flushing time, pH, suspended solids, conductivity, total alkanity and dissolved organic carbon. No significant correlation was found with any of these limnological parameters other than the dissolved organic carbon (d.o.c.) content of the water.

The distribution coefficients for thorium were found to be linear functions of the dissolved organic contents of the water. The coefficients varied from about $1 \cdot 10^4$ ($1\ kg^{-1}$) up to $8 \cdot 10^6$ ($1\ kg^{-1}$) for which the corresponding organic carbon contents were about $1\ mg\ l^{-1}$ and upwards.

For a distribution factor of 10^5 the transfer rate will be $12\ year^{-1}$ according to expression 2.

If the sinking rate of particles is used, where thorium is bound to the material the transfer rate will be $14\ year^{-1}$. Then a mean sinking rate of 140 m per year, as an average of the span 70-210 m (71) per year has been used.

The two different derivations give about the same value. However, to be consistent with the other nuclides the value derived by the first alternative has been chosen.

Protactinium

Protactinium appears to behave like thorium in the oceans (65) and therefore the same transfer coefficient has been chosen, namely $5\ year^{-1}$.

Uranium

There seems to be a similar dependence between the distribution coefficient and the amount of dissolved organic carbon in the water for uranium as there is for thorium. The values are lower, and for lakes with relatively high alkanity > 0.5 mM and pH > 7, rather than dissolved organic carbon, appears to control the adsorption behaviour of uranium (80). In Sweden the amount of uranium in water and sediment has been measured for some lakes. For the lakes in northern Sweden the distribution coefficient reported is about $4.3 \cdot 10^4$ (mean value) (70). For other lakes in southern Sweden the factor varied from about 200 up to $2.5 \cdot 10^5$ (51). If the value $1 \cdot 10^4$ is used for the calculations of the transfer rate it will be 1.2 $year^{-1}$.

Neptunium

It also seems to be very difficult to find data about the behaviour of neptunium in fresh-water systems. However, as the nuclide seems to behave most like uranium in the marine environment, we have chosen the same value namely 1.2 $year^{-1}$ to describe its removal from the water to the sediment.

In Table 3 the derived transfer coefficients for an imaginary lake are summarized.

Table 3 Transfer to sediment in a fresh water reservoir with a mean depth 10 m, sedimentation growth 1.5 kg per m^2 and an amount of suspended material $3 \cdot 10^{-3}$ kg per m^3

Element	Transfer coefficient (yr^{-1})
Technetium	$3 \cdot 10^{-2}$
Iodine	$3 \cdot 10^{-2}$
Cesium	$3.8 \cdot 10^{-1}$
Radium	3.8
Thorium	5
Protactinium	5
Uranium	1.2
Neptunium	1.2

3.2 Sedimentation - brackish water

For deriving the transfer coefficients which describe the transport from the water to the sediment reservoir in the intermediate area, i.e. the Baltic, the same expression has been used for most of the nuclides as was applied for lakes. The basic data used for the Baltic has been taken from reference (73).

Sedimentation growth 0.5 kg m^{-2} $year^{-1}$
Suspended material $1 \cdot 10^{-3}$ kg m^{-3}

The mean depth of the Baltic is 60 meters.

The distribution coefficients available are either for fresh or marine water while the Baltic constitutes of brackish water with a mean salinity of about 7 $^{o}/oo$.

Thus the distribution coefficients chosen have mostly been those for fresh-water.

In table 4 the transfer coefficients calculated under the conditions above are shown as well as the k_d-values used.

Table 4. Distribution factors used. Transfer to the sediments in the Baltic.

Element	Distribution factor ($l\ kg^{-1}$)	Transfer coefficient
Technetium	200	$1.6 \cdot 10^{-4}$
Iodine	200	$1.6 \cdot 10^{-4}$
Cesium	-	$2.7 \cdot 10^{-2}$
Radium	100 000	$9.2 \cdot 10^{-1}$
Thorium	10 000 000	$5.8 \cdot 10^{0}$
Protactinium	1 000 000	$5.8 \cdot 10^{0}$
Uranium	1 000	$8.3 \cdot 10^{-3}$
Neptunium	1 000	$8.3 \cdot 10^{-3}$

For cesium the transfer coefficient has been obtained from reference (74) using data from Salo (75).

3.3 Sedimentation - well-mixed layer

The same expression as formula No 1 has also been used for the transport to the sediment in the surface sea reservoir. The nuclides are not treated explicitly with the exception of neptunium since the knowledge about its behaviour in marine system has been increased. The nuclide seems also to be rather important in the calculations of the dose contribution to man. The assumptions made are as follows:

- suspended sediment level 10^{-4} kg m^{-3} (54, 57)

- sedimentation rate 10^{-2} kg m^{-2} $year^{-1}$

- mean depth 100 m.

Naturally these values vary, but in order not to overestimate the transfer to the sediment, the values for the oceans have been used. In the reference (57) higher values are quoted for the near land situated water masses. The k_d-values used are shown in table 5.

The derived transfer coefficients are also shown in table 5.

Table 5. Distribution coefficients used and transfer coefficients derived to the sediments in the well-mixed sea.

Element	k_d-value (l kg^{-1})	Transfer coefficients ($year^{-1}$)
Technetium	$1 \cdot 10^4$	10^{-3}
Iodine	$1 \cdot 10^2$	10^{-5}
Cesium	$5 \cdot 10^2$	$5 \cdot 10^{-5}$
Radium	$3 \cdot 10^4$	$3 \cdot 10^{-3}$
Thorium	$5 \cdot 10^6$	$3.3 \cdot 10^{-1}$
Protactinium	$5 \cdot 10^6$	$3.3 \cdot 10^{-1}$
Uranium	$5 \cdot 10^2$	$5 \cdot 10^{-5}$
Neptunium	$5 \cdot 10^3$	$5 \cdot 10^{-4}$

Neptunium

Reports about the levels and behaviour of neptunium in the marine environment are rather sparse. However, in one report (63) neptunium has been measured in different areas contaminated

by different sources. In contrast to what has previously been assumed the behaviour of neptunium is quite different to that of plutonium (72). The behaviour was more similar to that of uranium. Neptunium seemed to be present more in the water-phase than plutonium, and to have a relatively high mobility (64).

Neptunium is generally considered to be present in well oxygenated sea water as the highly soluble and mobile pentavalent species NpO_2^+. A simulation of the physico-chemical conditions in the Irish sea sediment taking into account some factors such as Eh, pH, Mn Fe^{2+}, Fe^{3+} has been performed to investigate if neptunium will be reduced to the much less soluble tetravalent state after burial in the bottom sediments. That is an important factor for deciding the long-term behaviour of neptunium in the marine environment. The laboratory experiments supported that the reduction to Np IV will occur below the top 10-20 cm of the sediments (64).

The distribution factor between water and sediment for the pentavalent state of neptunium is given in the same report to be about 5 000, while the corresponding value for plutonium is several orders higher in agreement with the report earlier quoted.

Another value for the water-sediment factor found in the literature is 50 000 (57).

As a basis for our calculations we have chosen 5 000 as it will give higher activity levels in the water and higher contemporary doses to man.

3.4 Sedimentation - deep sea

For most nuclides the removal from the water reservoir to the sediment has been derived from the information concerning residence times in the oceans.

In (65) the following residence times are given:

Iodine		$4 \cdot 10^5$ years
Cesium	about	$6 \cdot 10^5$ years
Thorium	"	100 years
Protactinium	"	100 years
Uranium	"	$4 \cdot 10^5 - 4 \cdot 10^6$ years

The investigations concerning the behaviour of naturally occuring radium in the oceans have shown, that in most oceans there is a marked vertical gradient in concentration, with higher values occurring in deep water. Radium is chemically comparable to barium so therefore the residence time for barium has been used. It is about $4 \cdot 10^4$ years. For neptunium the same value has been chosen as for uranium since it is reported that these nuclides seem to behave in a similar manner (64).

In reference (58) a global compartment model of the turnover of iodine is described. The corresponding transfer coefficient in that report is $2.2 \cdot 10^{-6}$ $year^{-1}$ which is in good agreement with the value derived from the residence time.

Concerning technetium the information is sparse, therefore the same value has been chosen as for radium. The transfer to the sediment is probably higher than for iodine but less than for thorium or protactinium.

In Table 6 the derived transfer coefficients are shown.

Table 6 Transfer coefficients to the sediment from the deep sea

Element	Transfer coefficient (yr^{-1})
Technetium	$2.5 \cdot 10^{-4}$
Iodine	$2.2 \cdot 10^{-6}$
Cesium	$1.7 \cdot 10^{-6}$
Radium	$2.5 \cdot 10^{-4}$
Thorium	$1 \cdot 10^{-2}$
Protactinium	$1 \cdot 10^{-2}$
Uranium	$2.5 \cdot 10^{-7}$

3.5 Turnover of the nuclides in the sediments

For the feedback of the nuclides from the sediments the assumptions used are:

- equilibrium between water and the sediment
- the sediment reservoir, which is supposed to have an exchange with water, is 10 cm in depth.

This means that the feedback can be written as

$$k_{out} = \frac{k_{in}}{K_d} \cdot \frac{M_w}{M_s}$$

where K_d = distribution factor
M_w = amount of nuclide in water
M_s = amount of nuclide in sediment.

3.6 Migration in soil

Based on the values derived for the water turnover the main part of the nuclide specific transfer coefficient for migration in soil is obtained from the following expression (76).

$$\frac{V_v}{V_n} = 1 + \frac{\rho}{n} K_d \qquad (2)$$

where V_n = velocity of the contaminant
V_v = average velocity of water
ρ = bulk mass density
n = porosity
K = distribution coefficient

The value for the quotient ρ/n ranges typically from 4-10.

Since increased retardation increases the doses by higher accumulation, the highest value of 10 has been chosen as a base for the calculations. The general behaviour of the different elements in soil is also treated in chapter 7. At the end of this chapter in Table 7 the different transfer coefficients derived are summarized.

Technetium

The chemistry of technetium in soil is very complex. Field experiments concerning the uptake of technetium in vegetation have given transfer rates from the upper soil of about 1.3 $year^{-1}$ (14). The equilibrium distribution coefficient between soil and water for the nuclide species TcO_4^- has been observed to be between 0.007-2.8 l kg^{-1} (50).

Transfer rates from the upper soil in the wide range of up to 24.7 $year^{-1}$ and down to $1.4 \cdot 10^{-2}$ $year^{-1}$ has been reported.

In summary, the nuclide technetium seems to be very mobile under oxidizing conditions, but the mobility is considerably lower if reducing conditions prevail.

The value of 0.6 $year^{-1}$, as a geometric mean, has been chosen for the downward-migration from the upper soil.

Iodine-129

Previously it was generally considered that the turnover of iodine deposited upon soil was the same as for water. This now seems to be a very questionable assumption. Several reports (42, 58, 59) show a strong adsorption of iodine in the upper soil.

An analysis of the available data for the dynamic global transport of naturally occuring stable iodine is to be found in reference (58). The mean residence time in the upper soil layer was shown to be $4 \cdot 10^3$ years which is much higher, by a factor of about 40, than the value obtained from the corresponding mean residence time for water and equilibrium distribution coefficients.

There is also some experimental verification that the migration of iodine in soil will decrease more with time. In contrast to the surface soil the global transport model supposes that the mobility of iodine-129 circulating in regions under the surface soil in the litosphere

is retarded very little compared to the ground-water flow.

In another report (59) the distribution of iodine-129 in the terrestrial environment surrounding a nuclear fuel reprocessing plant has been determined. The concentration decreased exponentially with increasing depth.

The half-depths, the depth required for the concentration to decrease to half its initial value, were determined. They varied from 4-10 cm, with a mean value of 7 cm.

The mean value of 7 cm during 25 years would give a transfer rate of

$$7 \text{ cm per } 25 \text{ year} = 0.3 \text{ cm per year.}$$

The upper soil was assumed to be 30 cm deep which gives a transfer rate of

$$\frac{0.3}{30} \text{ year}^{-1} = 0.01 \text{ year}^{-1}$$

Cesium

The behaviour of cesium in the soil has been studied since the fall-out from atmospheric nuclear testing in the sixties.

Russian investigations have given penetration rates in the range of 0.1 - 1 cm per year (45). The residence time in the plough-layer for the short-lived isotope cesium-137 usually used in Sweden has been about 100 years, which gives a transfer coefficient of about 10^{-2} year^{-1}.

The sorption of cesium in soils occurs from ion-exchange, so high amounts of other kat-ions present can decrease the distribution coefficient. The pH-dependence is not so strong in alkaline milieu. The distribution coefficient is in general relatively high, but there is a large interval for the values found from different experiments, from 6 up to $2.7 \cdot 10^4$ (83).

In Finland determinations of the K_d-value for some nordic clays in groundwater has been performed. In sand the sorbtion was low, about a factor of 6 but for muddy clay and heavy clay the values were 188 and 723 respectively.

Based upon a distribution factor of 500 the transfer coefficient derived will be $2.1 \cdot 10^{-3}$ $year^{-1}$, which is lower than the general value used, but very comparable to the first one mentioned. This value is chosen also with the object not to underestimate the accumulation in soil.

Radium

Most of the radium in soil is strongly fixed, its acid soluble form predominates among the mobile fractions (81). There is a direct positive correlation between radium and barium in soil profiles. Sandy soils usually have less radium than clay soils, and the highest concentration is usually found in the subsoil.

The migration of radium appears to depend on the Ba^{2+} ion concentration (81). Measurements from Russian soils of varying composition have shown a concentration of radium in the upper 10-40 cm. The increasing content of radium with depth is also reported in the same reference.

With decreasing pH in soil the mobility of radium will increase and thereby the uptake in the food-chains also increases.

The distribution coefficients for soil in the literature are found to vary from 0.15 m^3 kg^{-1} up to 3 m^3 kg^{-1}. The value that will be used is 1.2 m^3 kg^{-1} from reference (87).

Thorium

Natural contents of thorium in average soil are in the range of 2-12 mg kg^{-1} (85). The content increases with clay content due to adsorption and to the Th-contents of minerals in the clay fraction relative to contents in coarser soil constituents.

Soil solutions contain different species of thorium depending upon the level of pH. At high pH-values thorium would be virtually absent due to the low solubility of ThO_2 under such conditions (85).

Above pH 4 the major constituent in solution will be $Th(OH)_2^{2+}$ (85).

Limited information seems to be available with respect to thorium in natural waters. However, about the same range as for uranium contents (0.1-10 µg/l) or slightly lower can be expected. This is based upon some reported levels and from expected chemical behaviour of thorium in aqueous solutions (85).

Studies of the mobility of thorium from a natural deposit situated at Morro do Ferro, Brazil, has given a mobilization rate of about

$4 \cdot 10^{-8}$ per year (86). This value has been received by measuring the total Th-content by a stream that drains the hill and by analyses of dated cores from sedimentary deposits in the drainage basin. This indicates the slow migration rate of thorium.

The distribution factor chosen, 20 m^3 kg^{-1}, has been taken from reference (87).

Protactinium

The nuclide protactinium seems to behave in a similar manner to thorium in an aquatic environment (65), and studies of the geochemical behaviour seem to confirm these results. The same distribution coefficient has been chosen for protactinium as for thorium, namely 20 m^3 kg^{-1}.

Uranium

Measurements of the amounts of radium and uranium in some soils in the vicinity of Ranstad, Sweden, have often shown discrepancies from the expected quotient of one for the nuclides.

In particular there are discrepancies for the upper soil layer while the lower layer more often has the quotient one (47). This can to some extent be explained by the difference in the transport of the elements in soil.

The adsorption of uranium in soil depends, among other things, upon the level of organic substances in the soil. Increasing levels of clay and organic substances in the soil have a strong tendency to decrease the migration of uranium relative to radium, while with increasing pH and

calcium concentrations an increased transport can be expected.

From reference (87) the distribution coefficient of 0.6 m^3 kg^{-1} has been chosen as a basic value for deriving the transfer coefficients.

Neptunium

The adsorptive behaviour of neptunium and other actinides have been compared. The distribution factor was about 320 for the pentavalent ion of Np-237 in a suspension of < 2 µm clay particles from a silt loam soil. This value was about a factor ten lower than for uranium. The adsorption was reduced when organic matter and iron were removed from the clay particles. Other K_d-values measured for two sandy subsoils ranged from $1.6 \cdot 10^{-4}$ m^3 kg^{-1} to $4 \cdot 10^{-3}$ m^3 kg^{-1}. The values were quite insensitive to sodium ion concentration and relatively insensitive to calcium ion (15).

Other investigations have also reported a much lower adsorption of neptunium compared to americium and curium. The observed mobility of neptunium in soils and its ready uptake from soils by plants indicates that the element has been present as the pentavalent NpO_2^+ ion. However, neptunium (V) can be reduced to the much less soluble Np(IV) under anaerobic conditions.

However, the same distribution coefficient as for uranium, 0.6 m^3 kg^{-1}, has been chosen in order not to underestimate the accumulation in soil during such long time spans that are involved in the calculations.

Table 7 Transfer coefficients derived for migration in the soil ($year^{-1}$)

Element	Upper soil - deeper soil	Deeper soil - groundwater
Tc	$6 \cdot 10^{-1}$	$4 \cdot 10^{-2}$
I	$1 \cdot 10^{-2}$	$2.5 \cdot 10^{-4}$
Cs	$2.1 \cdot 10^{-3}$	$1.6 \cdot 10^{-4}$
Ra	$8.8 \cdot 10^{-4}$	$6.7 \cdot 10^{-5}$
Th	$5.3 \cdot 10^{-5}$	$4.0 \cdot 10^{-6}$
Pa	$5.3 \cdot 10^{-5}$	$4.0 \cdot 10^{-6}$
U	$1.6 \cdot 10^{-3}$	$1.2 \cdot 10^{-4}$
Np	$1.6 \cdot 10^{-3}$	$1.2 \cdot 10^{-4}$

4 INTERCEPTION

The interception fraction "r" is a parameter representing the fraction of the deposited activity that is retained directly the surfaces of vegetation. The data base for interception is limited, especially for vegetation other than grasses. The lack of data for leafy vegetables and many other garden crops is particularly apparent.

The value used earlier in the BIOPATH-program for the retention of radionuclides on the vegetation surface has been 0.80 (72). This value seems rather high when studying the literature. The values found vary between 0.02-0.80. The retention is strongly dependent on the surface density of the vegetation. Retention and yield values can be combined with advantage into a single parameter (50), the mass interception fraction (r/Y_v), to take into account the expected strong correlations between the interception and the standing biomass.

Following Table 8 shows the recommended values for the r/Y_v parameter (50).

Table 8 Estimation of the uncertainty associated with r/Y_v values for forage grasses assuming r/Y_v log normal distribution

	Value (m^2/kg dry wt)	Cumulative probability of value (%)
Most probable value	1.5	33
Geometric mean (median)	1.8	50
Arithmetic mean	2.0	59
99th percentile	5.1	99
Geometric standard deviation	1.6	

The geometric mean of the parameter has been chosen as a base for the calculations. It is 1.8 m^2 per kg dry weight. The standard deviation of the logarithms of the observations is 0.44; this value will be used for uncertainty analyses of the results.

5 WEATHERING CONSTANT

The halftime of the radionuclides deposited on the surfaces of vegetation is called "T_W". The value of "T_W" can be expected to vary strongly depending on the vegetation, climate and season. In a report (40) fourteen mean values of T_W have been analysed. A log normal probability curve for aerosols deposited on grass and other vegetation during the growth period has been presented. The mean values are between 10.5 and 27 days, but values up to 71 days have also been observed during winter conditions. The median value is 14.9 days. The most commonly used value of 14 days corresponds to the 41 % and is a factor of two less than the 99 % value of 27.2 days.

6 CONCENTRATION FACTORS - TERRESTRIAL ENVIRONMENT

Radioisotopes of an element in soil pass into the roots of vegetation in the same manner as the inactive isotopes.

The root uptake of an element depends on various soil properties, some of them interrelated, such as texture, clay content, dominant clay mineral, exchangeable calcium and potassium, other exchangeable cations, pH and organic matter content (33). Furthermore crop species, precipitation, early or late irrigation are parameters which must be considered.

Assuming that the concentration of a given nuclide is constant in the ploughed layer during the growing season, and that the plant concentration is at equilibrium with the soil concentration at the time of harvest, the root uptake can be estimated.

As the soil-to-plant concentration factors are extremely variable a single factor to predict the uptake is of small value. For that reason the root uptake for different elements is if possible reported as a range of values and a selected value.The latter is generally calculated as a geometric mean.

The transfer factor via root uptake, often called soil-factor, is usually expressed as:

$$\frac{\text{Bq(kg dry vegetation)}^{-1}}{\text{Bq(kg dry soil)}^{-1}}$$

Based on pot experiments, when a radionuclide is mixed homogeneously in a known amount of soil,

the calculation of the soil-factor is relatively simple.

In field experiments, in which the plough layer or the ground surface is contaminated, the normal technique in Sweden, it is convenient to express the soil uptake by following transfer factor:

$$\frac{\text{Bq(kg dry vegetation)}^{-1}}{\text{Bq m}^{-2}}$$

which is used to compare or combine different factors to describe root uptake.

The average dry weight of the top soil in Sweden can be estimated to 250 kg m^{-2}. The soil-factor can be calculated approximately from the transfer factor above by multiplying it with the average dry weight: 250 kg m^{-2} (2).

The feeding-stuff consumption by cattle is usually expressed in kg dry-matter per day. For that reason the transfer via root uptake in pasture in models is generally expressed directly by the soil-factor.

For some elements one single soil-factor is obtained from the literature. Therefore this value is used for other crops with consideration of their dry matter content as listed in Table 9.

Table 9 Average dry matter content in some crops.

Crop	Average dry matter, percent of fresh weight
Corn (grain)	85 %
Leafy vegetables	10 %
Potatoes	20 %
Pasture grass	20 %
Hay	83.5 %

6.1 Technetium

The most stable chemical form of technetium in aerated solutions and under aerobic conditions in soil is the pertechnetate ion (TcO_4^-) which is very soluble in water. It is sorbed poorly in soil, and a downward movement in soils has been confirmed (14). Pertechnetate is also the most mobile chemical form of technetium (21) and its removal rate from the root zone has been studied. The environmental half-time is estimated to 100-180 days (14, 18, 21) but the mobility may be considerably decreased by reduction.

Data for leaching, as well as for root uptake, also indicate a time dependent decrease in the mobility as well as in the plant availability when the element is deposited in soil as the pertechnetate ion.

It is apparent from several experiments that technetium in solution is easily available to plants and can also be located in the tissues of a variety of crops. In experiments with Pisum sativum for example, the concentration of technetium was 5-10 times higher in the leaves than in the roots (17, 25).

Some of the soil factors reported, calculated on the basis of laboratory experiments, are too high to be relevant for the current application (18, 24, 25).

In Swedish lysimeter tests the uptake during the first season indicated soil-factors of 25-125 (per kg dry weight of vegetation/kg dry soil). However, in reference (14) it is suggested that a steady state concentration of technetium in vegetation might not differ substantially from the prediction of a soil-factor of "one" made in reference (13).

There is not a marked difference in the root uptake of technetium in different types of soil. It seems, however, to be lower from organic than mineral soils and to be lower from soils with low pH-values. A low content of organic matter and a high pH may result in a high transfer of technetium to crops (18).

In the literature two types of transfer values are reported for the soil-plant pathway, a short term (2-3 years) and a long term value (11). In the case of accident studies, the higher short term value ought to be chosen but otherwise the long term value is recommended.

The concentration of technetium in terrestrial food-stuffs can be estimated from the following factors.

Crop	Concentration factor Range	Value selected	Ref
Pasture grass		1	(13, 14)
Green vegetables		$1 \cdot 10^{-1}$ *)	
Grain		$9 \cdot 10^{-1}$ *)	
Potatoes		$2 \cdot 10^{-1}$ *)	

*) Calculated from fresh weights.

6.2 Iodine

Iodine is partly bound to humic and fulvic acids, as well as to other organic compounds in the soil and it becomes available to plants during mineralization of the organic material (23).

The element is taken up by the plant roots in an anionic form. It is sorbed to a large extent in soils rich in clay minerals and organic matter. Iodine is estimated to be retarded in the surface layer of the top soil and thus shortlived isotopes never reach the root zone. In Swedish experiments it has been found that the presence of soluble chlorides may decrease the root uptake of iodine to certain crops (4).

Crop	Concentration factor Range	σ^2	Value selected	Ref
Pasture grass	$2.5 \cdot 10^{-2}$-$2.5 \cdot 10^{-1}$	0.44	$8 \cdot 10^{-2}$ *)	(2, 4, 13, 27, 29, 45)
Green vegetables	$2.3 \cdot 10^{-3}$-$4.3 \cdot 10^{-3}$	0.03	$3 \cdot 10^{-3}$ **)	(4, 45)
Grain			$2 \cdot 10^{-3}$ **)	(4, 45)
Potatoes			$1 \cdot 10^{-3}$ **)	(4, 45)

*) Geometric mean value
**) Calculated from fresh weights.

6.3 Cesium

The clay and organic matter content in soil influences the root uptake of cesium considerably. Increasing clay content depresses the plant absorption whilst increasing organic matter content enhances the uptake.

The uptake of cesium from the soil is also influenced by the amount of exchangeable potassium present, which in turn is balanced between its removal via crops and through leaching, and through break-down of potassium-rich clay minerals and possible fertilization. From soils depleted in potassium plants will absorb more cesium than from soils with a high potassium content. The uptake of cesium in pasture grass can be ten times higher from coarsely textured soils than from finely textured ones (32, 45).

It has been found experimentally that, where the top soil layers were uniformly contaminated with cesium, but the subsoil was not, the shallow rooted crops accumulated more of the nuclide than deeper rooted plants (32).

To sum up: the root uptake is estimated using following concentration factors. The lower limits refer to soils rich in clay minerals and exchangeable potassium, but with a low content of organic matter and low pH.

Crop	Concentration factor Range	σ^2	Value selected	Ref
Pasture grass	$2.5 \cdot 10^{-2}$ - $5 \cdot 10^{-1}$	0.75	$1 \cdot 10^{-1}$ *)	(2, 13, 27, 45)
Green vegetables	$1.9 \cdot 10^{-2}$ - $1.8 \cdot 10^{-1}$	0.42	$2 \cdot 10^{-2}$ **)	(45)
Grain	$5.3 \cdot 10^{-3}$ - $3.9 \cdot 10^{-2}$	0.33	$1 \cdot 10^{-2}$ **)	(45)
Potatoes	$7.5 \cdot 10^{-3}$ - $7.0 \cdot 10^{-2}$	0.42	$2 \cdot 10^{-2}$ **)	(45)

*) Geometric mean value
**) Calculated from fresh weights.

6.4 Radium

Swedish investigations have verified that the concentration of radium is relatively low in soils with a high ion exchange capacity. In calcerous soils a relative accumulation of radium has been established. Furthermore it has been found that the content of soluble radium compared to the total radium content decreases with increasing pH (47), and the reverse (30). As the root uptake of the element also depends on its mobility in soil it is shown that the uptake of radium is low in calcerous soils (47).

The leaching rate of radium is larger at a low pH-level in soil rich in organic matter but otherwise poor in colloids (47). In reference (48) the soil factors for three types of crops are reported:

- vegetables $8 \cdot 10^{-3}$
- grain $1 \cdot 10^{-2}$
- forage $9 \cdot 10^{-2}$

These factors agree with the values chosen in this report.

Crop	Concentration factor Range	σ^2	Value selected	Ref
Pasture grass	$3.3 \cdot 10^{-3}$ - $7.5 \cdot 10^{-2}$	0.81	$2 \cdot 10^{-2}$	(9, 15, 34, 29, 38)
Green vegetables			$7 \cdot 10^{-3}$*)	(34)
Grain	$1 \cdot 10^{-2}$ - $2 \cdot 10^{-2}$		$1 \cdot 10^{-2}$**)	(16, 43)
Potatoes			$3 \cdot 10^{-3}$*)	(34)

*) Calculated from fresh weights.
**) Based on Swedish investigations and calculated from fresh weights (43).

6.5 Thorium

It may be thought that positively charged thorium compounds will be strongly absorbed in soils. The direct correlation between the thorium activity in crops and the total activity in soil is however very weak. There are complexes of thorium with organic acids in the soil water, and hydrolysis products with very low chemical activities and in equilibrium with ThO_2 in the soil (47). There does not appear to be a correlation between the root uptake of thorium and specific soil characteristics (47).

Crop	Concentration factor Range	σ^2	Value selected	Ref
Pasture grass			$5 \cdot 10^{-3}$ *)	(27)
Green vegetables			$5 \cdot 10^{-4}$ **)	
Grain	$8 \cdot 10^{-4}$	0.19	$8 \cdot 10^{-4}$ ***)	(43)
Potatoes			$1 \cdot 10^{-3}$ **)	

*) Geometric mean value (27)
**) Calculated from fresh weights.
***) Based on Swedish investigations and calculated from fresh weights.

6.6 Protactinium

Data concerning the root uptake of protactinium are very sparse.

In this report a general soil factor of

$$3 \cdot 10^{-3} \text{ kg dw/kg}^{-1} \text{ dw}$$

has been chosen in default of a better value. This factor has been selected for pasture grass and calculated for the other crops from their dry matter content.

Crop	Concentration factor Range	σ^2	Value selected	Ref
Pasture grass	$1 \cdot 10^{-3} - 1 \cdot 10^{-2}$	0.44	$3 \cdot 10^{-3}$	(9, 20, 29)
Green vegetables			$3 \cdot 10^{-4}$ *)	
Grain			$3 \cdot 10^{-3}$ *)	
Potatoes			$6 \cdot 10^{-4}$ *)	

*) Calculated from fresh weights.

6.7 Uranium

Positively charged uranium compounds will in low concentrations be strongly absorbed in soils. The retention of uranium in soils is dependent on the organic matter content as well as the presence of clay minerals. In Swedish investigations (47) it was discovered that the mobility of uranium increases with increasing calcium concentration and high pH-values. Under oxidizing conditions uranium can occur in soil solutions as the uranyl ion, UO_2^+, at low pH-values. At higher pH-values the uranyl ion may form complexes with carbonate ions. These complexes may be relatively mobile because of their negative charge (35).

The root uptake of uranium decreases in connection to an increasing mineral uptake while in general a higher pH-level in soil would convey to an increased root uptake.

	Concentration factor		
Crop	Range	Value selected	Ref
Pasture grass		$4 \cdot 10^{-3}$	(47, 53)
Green vegetables		$4 \cdot 10^{-4}$*)	
Grain		$1 \cdot 10^{-3}$*)	(43)
Potatoes		$8 \cdot 10^{-4}$*)	

*) Based on Swedish investigations and calculated from fresh weights.

6.8 Neptunium

Neptunium is fairly soluble in all soils. It is however associated less than other transuranic

elements with organic material, especially in an acid environment (44).

Investigations have revealed that the migration of neptunium in soils, and its transfer from soils to plants is substantially greater than, for example, plutonium and americium (46). The lowest uptake of neptunium is reported for soil with a high clay content.

	Concentration factor			
Crop	Range	σ^2	Value selected	Ref
Pasture grass	$7.5 \cdot 10^{-3}$ - $1.5 \cdot 10^{-1}$	0.75	$3 \cdot 10^{-2}$ *)	(5, 6, 13, 27)
Green vegetables			$3 \cdot 10^{-3}$ **)	
Grain			$4 \cdot 10^{-4}$ **)	
Potatoes			$6 \cdot 10^{-3}$ **)	

*) Geometric mean value
**) Calculated from fresh weights.

7 DISTRIBUTION FACTORS

The transfer of radioactivity from animal feed to animal products is described by distribution factors.

They are defined as the fraction of daily intake of a specific nuclide, which at equilibrium is likely to be recovered per litre of milk or per kilo of beef.

Only cattle, or more precisely dairy cows, are considered in this report. Currently the consumption of pork is certainly at least as important as the consumption of beef. As pigs do not consume any roughage, which is assumed to be the most important pathway for soil activity, their meat has not been taken into consideration.

The distribution factors are expressed in day per unit of mass or volume:

day l^{-1}, intake → milk

day kg^{-1}, intake → meat

Regarding the entire daily intake of radioactivity following pathways are considered:

- grazing from root-uptake and inadvertent consumption of soil
- direct deposition on the vegetation
- drinking water.

Lacking of relevant data the importance of physical and chemical form for the metabolic turnover of the element in animal, is neglected.

Technetium

Distribution factors (day per unit of mass or volume)

	Range	σ^2	Value selected	Ref
Feed-milk	$9.9 \cdot 10^{-3}$-$2.5 \cdot 10^{-2}$	0.07	$1 \cdot 10^{-3}$ *)	(1, 20, 27)
Feed-beef	$9.9 \cdot 10^{-4}$-$4 \cdot 10^{-1}$	3.0	$1 \cdot 10^{-3}$ **)	(3, 20, 27)

*) There are only a few data available on the metabolism of technetium in animals producing food-stuff. When calculating the risks associated with technetium, conservative assumptions from studies of iodine are often used because of similarities in metabolic behaviour. In Swedish experiments with lactating goats the concentration of technetium in milk was about 1/10 of the corresponding values for radioiodine (19). The distribution factor with respect to the transfer of technetium from intake to milk for that reason has been reduced to $1 \cdot 10^{-3}$ day l^{-1}.

**) The Swedish experiments with goats show that the transfer of technetium to beef is not of any importance in ruminants. As most of the distribution factors found in the literature seem to have been estimated on the assumption that technetium behaves like iodine a distribution factor of $1 \cdot 10^{-3}$ has been chosen for beef.

Iodine

Distribution factors (day per unit of mass or volume)

	Range	σ^2	Value selected	Ref
Feed-milk	$5 \cdot 10^{-3}$-$1 \cdot 10^{-2}$	0.04	$7 \cdot 10^{-3}$ *)	(1, 12, 22, 27, 29)
Feed-beef	$2.9 \cdot 10^{-3}$-$2 \cdot 10^{-2}$	0.31	$8 \cdot 10^{-3}$ *)	(3, 12, 27, 29)

*) Geometric mean value

Cesium

Distribution factors (day per unit of mass or volume)

	Range	σ^2	Value selected	Ref
Feed-milk	$2.5 \cdot 10^{-3}$-$1.2 \cdot 10^{-2}$	0.21	$7 \cdot 10^{-3}$*)	(1, 12, 45)
Feed-beef	$7.2 \cdot 10^{-3}$-$9.3 \cdot 10^{-2}$	0.55	$3 \cdot 10^{-2}$**)	(3, 12, 27)

*) Higher values are given for cows fed on a high grain diet, and lower values when the diet contains more crude fibres. This is explained in part by the assumed sorption of cesium to clay mineral colloids in pasture grass, and sorption to fibres in rumen (45).

**) Geometric mean values.

Radium

Distribution factors (day per unit of mass or volume)

	Range	σ^2	Value selected	Ref
Feed-milk	$1.4 \cdot 10^{-4}$-$1.5 \cdot 10^{-2}$	1.82	$3 \cdot 10^{-3}$ *)	(1, 9, 16, 29, 34, 51)
Feed-beef	$5.0 \cdot 10^{-4}$-$9.9 \cdot 10^{-4}$	0.04	$7 \cdot 10^{-4}$ **)	(3, 27, 29)

*) From Swedish investigations (51)

**) Geometric mean value.

Thorium

Distribution factors (day per unit of mass or volume)

	Range	σ^2	Value selected	Ref
Feed-milk			$5 \cdot 10^{-6}$*)	(1, 20)
Feed-beef	$1 \cdot 10^{-4}$**)-$5 \cdot 10^{-3}$	1.28	$7 \cdot 10^{-4}$***)	(3, 20)

*) In the absence of relevant data derived from human data by Ng et al (20).

**) The concentration of thorium in cattle muscle is estimated from the concentration in cattle bone and the contents in bone and muscle of rats (20).

***) Geometric mean value.

Protactinium

Distribution factors (day per unit of mass or volume)

	Range	σ^2	Value selected	Ref
Feed-milk			$5 \cdot 10^{-6}$ *)	(9, 20, 29)
Feed-beef	$1 \cdot 10^{-3}$**)			
	$5 \cdot 10^{-3}$***)	0.22	$3 \cdot 10^{-3}$****)	(3, 20)

*) Estimated by Ng et al (20) in the absence of direct information

**) Based on estimates for plutonium and americium in which the retention functions were assumed to be the same for the internally deposited nuclides, and the transfer was presumed to be proportional to the gastrointestinal absorption factor recommended in (39). Later it was recommend that these uptake factors be modified (41). For that reason the distribution factor could possibly be reduced by a factor of at least 2 but the conservative value has been selected.

***) A single value, $1.6 \cdot 10^{-6}$ day kg^{-1}, is based on the assumption that the transfer of protactinium is comparable with americium and curium (29). It has not been included in the calculation of selected values.

****) Geometric mean value.

Uranium

Distribution factors (day per unit of mass or volume)

	Range	σ^2	Value selected	Ref
Feed-milk	$4 \cdot 10^{-5}$-$6 \cdot 10^{-4}$	0.61	$2 \cdot 10^{-4}$ *)	(1, 9, 29, 53)
Feed-beef	$5 \cdot 10^{-3}$-$3 \cdot 10^{-2}$ **)	0.27	$1 \cdot 10^{-2}$ *)	(3, 20)

*) Geometric mean value

**) A single value, $1 \cdot 10^{-6}$ day kg^{-1}, is based on the assumption that the transfer of uranium is comparable with americium and curium (29). It has not been included in the calculation of selected values.

Neptunium

Distribution factors (day per unit of mass or volume)

	Range	σ^2	Value selected	Ref
Feed-milk			$5 \cdot 10^{-6}$ *)	(1, 20, 27, 29)
Feed-beef	$2 \cdot 10^{-4}$-$5 \cdot 10^{-3}$ **)	0.86	$1 \cdot 10^{-3}$ ***)	(3, 20, 27)

*) Estimated by Ng et al in the absence of direct information.

**) A single value, $1.6 \cdot 10^{-6}$ day kg^{-1}, is based on the assumption that the transfer of neptunium is comparable with americium and curium (29). It has not been included in the calculation of selected values.

***) A distribution factor, $1 \cdot 10^{-3}$, in reference (20) is based on the estimates for plutonium and americium. The retention functions are assumed to be proportional to the gastrointestinal uptake recommended in (39). Later these uptake factors were modified (41). The distribution factor could possibly be reduced by a factor 10 or more, but a conservative value has been selected.

In Table 10 the values selected for the different elements are summarized.

Table 10 Distribution factors selected (day per unit of mass or volume)

	Feed-milk	Feed-beef
Technetium	$1 \cdot 10^{-3}$	$1 \cdot 10^{-3}$
Iodine	$7 \cdot 10^{-3}$	$8 \cdot 10^{-3}$
Cesium	$7 \cdot 10^{-3}$	$3 \cdot 10^{-2}$
Radium	$3 \cdot 10^{-3}$	$7 \cdot 10^{-4}$
Thorium	$5 \cdot 10^{-6}$	$7 \cdot 10^{-4}$
Protactinium	$5 \cdot 10^{-6}$	$3 \cdot 10^{-3}$
Uranium	$2 \cdot 10^{-4}$	$1 \cdot 10^{-2}$
Neptunium	$5 \cdot 10^{-6}$	$1 \cdot 10^{-3}$

8 CONCENTRATION FACTORS - AQUATIC ENVIRONMENT

With respect to the aquatic environment the concentration or bio-accumulation factors for the uptake of elements in fish are included for three different types of water:

- fresh water
- brackish water
- sea water.

The concentration factors (CF) are expressed by:

$$\frac{\text{Bq(kg fresh fish)}^{-1}}{\text{Bq(litre water)}^{-1}} = \text{litre kg}^{-1}$$

When a range of value is obtained by compilation of the reference data, the geometric mean value has been selected as a general rule.

For some elements there are no data regarding brackish water. In such cases the concentration factors have been estimated from the factor for fresh water.

For sea and brackish water the possibility of future exposure from other marine foodstuffs than fish is possible. Therefore the concentration factors to other aquatic products have also been studied, see Table 11.

Table 11 Concentration factors (Bq/kg^{-1} per Bq/kg^{-1}) for different marine species.

Element	Crustacea (57)	Molluscs (54, 57)	Seaweed (54)	Plants (55)	Plankton (54)
Technetium	1 000	1 000	100 000	4 000	1 000
Iodine	100	100	1 000	4 000	1 000
Cesium	30	10 30 (57)	10	20	100
Radium	100	100	100	100	100
Thorium	1 000	1 000	1 000	3 000	10 000
Protactinium	10	10	100	6	1 000
Uranium	10	10	10	67	5
Neptunium	100	1 000	1 000 2 000	6	2 000

Technetium

	Concentration factors - fish		
	Range	Value selected	Ref
fresh water	10-125	35	(7, 13, 31)
brackish "		30*)	
sea "		10	(10, 13, 57)

*) CF estimated from fresh-water factor.

Iodine

	Concentration factors - fish		
	Range	Value selected	Ref
fresh water	15-30	20	(7, 13)
brackish "		20	(8)
"	10-20	20	(10, 13)

Cesium

	Concentration factors - fish		
	Range	Value selected	Ref
fresh water	280-4800	1300*)	(7, 13, 50)
brackish "	200-250	200	(8, 37)
sea "	30-40	40	(10, 13)

*) Geometric mean value (50).

Radium

	Concentration factors - fish		
	Range	Value selected	Ref
fresh water	10-200	25**)	(13, 16, 52)
brackish "	2-10	10	(52)
sea "	50-100	50*)	(13, 57)

*) The concentration factors obtained from the literature for sea water seem unreasonably high since the bio-accumulation factors usually decrease with increasing salt content in water. Therefore the lowest value is selected for sea water.

**) Based upon the geometric mean for the concentration factor, water-fish muscle.

Thorium

	Concentration factors - fish		
	Range	Value selected	Ref
fresh water		30	(13, 20)
brackish "		30*)	
sea "	1000-10000	3000**)	(13, 20, 57)

*) The same CF as for fresh water has been selected due to lack of other data.

**) The values of this concentration factor, which are reported in the literature seem unwarrantedly high, especially as the bio-accumulation factors usually decrease with increasing salt content in water. As the best estimate the geometric mean value has been selected.

Protactinium

Concentration factors - fish

	Range	Value selected	Ref
fresh water		10	(13, 20)
brackish "		10*)	
sea "	10-1000	100	(13, 20)

*) The same CF as for fresh water due to lack of other data .

Uranium

Concentration factors - fish

	Range	Value selected	Ref
fresh water	2-10	5	(13, 52)
brackish"		5*)	
sea "	0.1-20	1**)	(13, 20, 52, 57)

*) In the absence of reported values the same CF has been selected for brackish as for fresh water

**) Congruent with reference (20).

Neptunium

Concentration factors - fish

	Range	Selected value	Ref
fresh water		10	(13, 20)
brackish"		10*)	
sea "		10	(13)

*) The same CF as for fresh water is selected due to lack of other data.

In Table 12 the values selected for the different elements are summarized.

Table 12 Concentration factors water-fish selected for the aquatic environments

Element	Fresh-water	Brackish-water	Sea-water
Technetium	35	30	10
Iodine	20	20	20
Cesium	1 300	200	40
Radium	25	10	50
Thorium	30	30	3 000
Protactinium	10	10	100
Uranium	5	5	1
Neptunium	10	10	10

9 YIELD VALUES

The dose to the regional populations is determined by the yield values for the terrestrial and aquatic environments. For that reason a summary of estimated Swedish yield values has been done.

9.1 Terrestrial environment

The yield values for agricultural products in Sweden have been based upon reference (36).

Table 13 Biological yield (kg m^{-2})

Crops	Fresh weight		Dry matter
	Range	Average	Average
Grain (autumn-sown wheat)	0.3-0.6	0.4	
Pasture, rotational grazing in all (10 x 0.05)			0.5
Green vegetables		2.7	
Root vegetables		3.4	
Potatoes, non-irrigated	2.1-4.3	3.4 *)	
irrigated	2.4-4.5	4.1 *)	

*) weighted average

9.2 Aquatic environment

9.2.1 Fresh water

The annual production of fresh water fish in Sweden naturally depends on the type of lake. As a general rule estimates of economic levels for fish catchment give the following annual yield

values which are probably reasonable for an area of 10 000 m^2 (84).

Lakes: 7 kg fish

Flowing water: 20 kg fish

9.2.2 Baltic Sea

The Baltic Sea is an area of intensive fishing and landings have been increased over the past few years. From reference (73) the amount for human consumption is reported to be about 2×10^8 kg fish per year. No account is taken to the amount of fish landed for industrial purposes as that is a minor part of the total yield.

10 DIET AND CONSUMPTION RATES

10.1 Human beings

The consumption of food-stuffs by human beings has been studied (for closer information see Appendix 1).

The values estimated to be the most appropriate for use are shown in Table 14.

Table 14 Food consumption in Sweden, per head and year. Average intake during 1978-80.

Food products	kg per head
Milk	317.6 (128 kg as cheese)
Meat	52.9
Vegetables	34.6
Cereals	73.7
Potatoes	70.9
Egg	11
Fish	14.7

10.2 Cattle

A Swedish dairy cow currently (1980) produces an average of 5 500 kg milk a year. When the cows are dry they need about 10 kg dry matter a day as roughage. During the lactation period more concentrate is required and for this period a consumption of dry matter can increase up to 19 kg a day, see Table 15.

Table 15. Values for dry matter consumed by a dairy cow in relation to its milk production, as well as the recommended fraction of dry matter intake as roughage and concentrate.

Litre milk day^{-1}	Dry matter intake	Fraction of intake as roughage	as concentrate
10	11.5	0.8	0.2
20	15.5	0.6	0.4
30	19	0.5	0.5

The annual consumption of roughage is divided into:

- 2 000 kg hay
- 6 000 kg pasture grass.

See following table.

Fraction of the year the dairy cows in different parts of Sweden can consume fresh forage.

Area	No of days	Fraction of the year
South of Sweden	150	0.4
North of Sweden	100	0.3

In this report it is assumed that the entire nutritional requirement for a dairy cow is satisfied by grazing. In relation to milk

production the daily consumption of dry matter is assumed to lie between: 11.5-19 kg, with an average value of:

- 16 kg dry matter day^{-1}, see Table 15

The potential risk from ingestion of contaminated soil during the grazing periods is also considered (82). According to reference (28) the pasture grass can be contaminated with soil particles which constitutes between 0.3 and 2.9 % of the dry matter content and implies a soil consumption in the range of 0.05-0.5 kg day^{-1}. In this report a consumption rate of

- 0.3 kg soil day^{-1}

has been chosen. In the absence of relevant data the importance of the chemical form for the turnover in animals is neglected, and the elements in the soil are assumed to behave as in feeding-stuff.

When the cattle are grazing inhalation of contaminated soil particles is a possible pathway, especially in dry weather.

The inhalation capacity is reported to lie in the range of 59-104 l per minute, with an average in rest of 86 l per minute (26).

With a particle concentration estimated to two or three milligrammes per cu.m. air it implies that only 10-15 milligrammes of soil particles are inhaled per hour.

In comparison to the intake of radioactivity by way of food stuffs this is judged to be less important.

10.2.1 Drinking water

The cattle's need for drinking water is influenced by several factors, but may for a lactating cow of average size be estimated to between 50-70 l daily. During pasturing the requirement of water can increase more than 50 % to round:

> 75-110 litre day^{-1} for a dairy cow with an average water consumption of 90 l day^{-1}.

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CONSUMPTION HABITS.
A STATISTIC SUMMARY AND COMPARISON

Anders Appelgren

List of contents

1 INTRODUCTION

The different constitutents of our present diet and food consumption habits are reviewed. Average intake values are given both for Sweden and for different parts of the globe.

2 MILK PRODUCTS

The total consumption of milk products has been divided into the following types which have been investigated

1 Milk sold outside dairies and consumed on farms
2 Market milk, culture milk, skimmed milk etc
3 Thin, thick and culture cream

Cheese products are treated separately later in this chapter.
The annual consumption in Sweden during the last few years has been about 190 l/head, and since 1965 the consumption has increased by more than 25 l/head. The portion of milk consumed on farms and sold outside dairies has decreased radically, from 22% in 1965 to 3 % in 1980. In 1980 4 % of the total milk products consumed consisted of cream products.
During the years 1978-1980 Swedes consumed the following amounts of milk products

1978	189.2 l/head and year
1979	189.1 l/head and year
1980	190.5 l/head and year
Average	189.6 l/head and year

It can be assumed that the intake of milk for children is greater than for adults. This is confirmed by ICRP 23 which recommends the following values for intake of milk for reference man

Adult man	110 l/head and year
Adult woman	73 l/head and year
Child (10 years)	164 l/head and year

In the global view, the variations of milk intake between different parts of the world, are rather large. This is shown in the following table from ICRP 23

Far East	19 l/head and year
Near East	78 l/head and year
Africa	35 l/head and year
Latin America	88 l/head and year
Europe	180 l/head and year
North America	310 l/head and year
Oceania	210 l/head and year

Compared to other countries, Sweden has a rather large consumption of cheese. As can be seen below the consumption has increased slowly during recent years

1965	7.8 kg/head and year
1970	8.9 kg/head and year
1975	11.1 kg/head and year
1978	12.1 kg/head and year
1979	12.7 kg/head and year
1980	13.4 kg/head and year

The average intake per head and year 1978-1980 was 12.8 kg.

It can be noted, that for production of 1 kg cheese approximately 10 l milk is needed.

3 MEAT

The consumption of meat in Sweden for the years 1978-1980 was

1978	53.0 kg/head and year
1979	53.4 kg/head and year
1980	52.3 kg/head and year
Average	52.9 kg/head and year

Meat is here defined as

- fresh and frozen meat
- cured meats and provisions
- canned meat (including meat soups).

Poultry is treated separately at the end of this chapter.

Of the amounts quoted above, 53 % consists of fresh and frozen meat, 40% of cured meats etc,and 7% of canned meat.
After 1960 a small increase in the meat consumption can be noticed.

In the global view, the meat intake is very low in Asia and Africa. This can be seen in the following table from ICRP 23

Far East	9 kg/head and year
Near East	13 kg/head and year
Africa	15 kg/head and year
Latin America	37 kg/head and year
Europe	41 kg/head and year
North America	91 kg/head and year
Oceania	114 kg/head and year

The intake values in the table above include poultry.

The consumption of poultry in Sweden is as follows

1978	4.4 kg/head and year
1979	4.8 kg/head and year
1980	4.6 kg/head and year
Average	4.6 kg/head and year

4 VEGETABLES AND FRUITS

In the determination of the consumption of vegetables (fruits are treated later), carrots and other root fruits are not taken into consideration (see chapter Root fruits). Therefore, the vegetable consumption consists of cucumbers, cabbage, onion, salad, mushrooms, tomatoes, peas, beans etc. The vegetables can be fresh, frozen, dried, canned etc. With these premises the total intake of vegetables in Sweden is

1978	34.4 kg/head and year
1979	35.0 kg/head and year
1980	34.4 kg/head and year
Average	34.6 kg/head and year

About 40% of these consumption rates consists of imported vegetables. 65% of the vegetables eaten are fresh, 10% frozen and 25% treated in some other way.

The intake of vegetables is increasing. In 1960 we consumed 21 kg/head, in 1970 30 kg/head and in 1980 about 35 kg/head. In some statistics it can be seen that the increase is reduced to a nearly constant level for the years 1975-80.

Looking at fruit consumption, it must be remembered that over 65% of the intaken amounts consist of imported fruit. The table below includes the following types of fruits and berries

- fruits and berries, fresh and frozen
- fruits and berries, dried, canned and jams etc
- chestnuts, almonds and nuts

Total consumption of fruit and berries in Sweden

1978	65.0 kg/head and year
1979	70.2 kg/head and year
1980	65.5 kg/head and year
Average	66.9 kg/head and year

ICRP 23 adds the consumption rates of vegetables and fruits, excluding potatoes, cassava and other edible roots. This gives the following consumption of vegetables and fruits in different parts of the world

Far East	47 kg/head and year
Near East	145 kg/head and year
Africa	78 kg/head and year
Latin America	114 kg/head and year
Europe	115 kg/head and year
North America	188 kg/head and year
Oceania	141 kg/head and year

5 CEREALS

In this short study, cereals are divided into the following groups

- flour and its products (excluding bread) and hulled grain
- Soft and hard rye-bread
- Cakes, confectionary, biscuits etc.

In 1978-1980 the total consumption of cereals in Sweden was

1978	73.0 kg/head and year
1979	74.3 kg/head and year
1980	73.8 kg/head and year
Average	73.7 kg/head and year

Before 1965 the cereal consumption was more than 80 kg/head and year, it decreased slowly to the beginning of the 70ies and since then the consumption has been relatively constant.

35% of the total cereal intake consists of fluor, etc - much of this flour is of course converted to bread and cakes in households - 50 % as bread and 15 % as cakes etc.

For the cereal intake in other parts of the world, ICRP 23 gives the following values (in terms of flour and milled rice)

Far East	147 kg/head and year
Near East	163 kg/head and year
Africa	120 kg/head and year
Latin America	103 kg/head and year
Europe	137 kg/head and year
North America	68 kg/head and year
Oceania	89 kg/head and year

6 ROOT-FRUITS

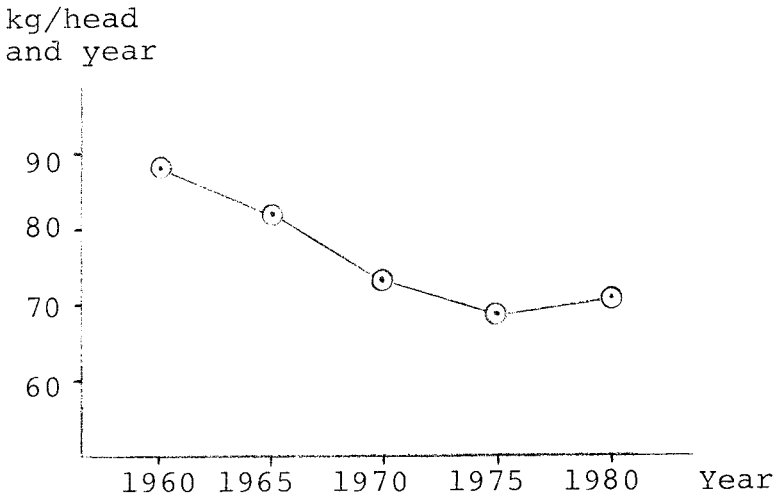
Root fruits include

- potatoes and their products
- other edible roots such as carrots and common beets etc

The potato products can be divided into

- fresh potatoes
- other potatoe products

The total consumption of potatoes and their products in Sweden from 1960 can be seen in the diagram below



For the years 1978-80 the total potatoe consumption in Sweden was

1978	71.1 kg/head and year
1979	70.7 kg/head and year
1980	70.9 kg/head and year
Average	70.9 kg/head and year

The proportion of fresh potatoe consumption in the figures above is 93%.

The intake of other edible roots in Sweden amounted to 6-8 kg/head and year for the last 20 years. The average value for the years 1978-80 is 6.1 kg/head and year.

The intake of root fruits in other parts of the world are given in ICRP 23. These values show the intake of starchy roots, which include potatoes, cassava and other edible roots.

Far East	57 kg/head and year
Near East	16 kg/head and year
Africa	173 kg/head and year
Latin America	90 kg/head and year
Europe	138 kg/head and year
North America	50 kg/head and year
Oceania	53 kg/head and year

7 EGGS

The egg consumption in Sweden has been constant for a relatively long period. Values from 1960 and onward show an annual intake of about 11 kg per head, which is also the value for the years 1978-80.

ICRP 23 gives the estimated intake values for some geographical regions

Far East	1 kg/head and year
Near East	2 kg/head and year
Africa	1.5 kg/head and year
Latin America	4 kg/head and year
Europe	8.5 kg/head and year
North America	20 kg/head and year
Oceania	11 kg/head and year

8 FISH

The fish consumption in Sweden has been approximately 15 kg/head and year for a relatively long time. On the other hand the distribution of different fish products has changed. The following fish products are included in the total fish consumption

- fresh fish
- frozen fish
- canned and prepared fish
- crustaceans and molluscs

To determine the true intake of fresh fish it has been assumed that 40% of the landed weight disappears when the fish are cleaned.

The total fish consumption for the years 1978-1980 was

1978	14.6 kg/head and year
1979	14.5 kg/head and year
1980	15.0 kg/head and year
Average	14.7 kg/head and year

The portion of fresh fish consumption has been reduced from 51% in 1960 to 21% in 1980, frozen fish has increased from 8% to 17% during the same period, canned and prepared fish has increased from 36% to 46%, and consumption of crustaceans/molluscs has increased from 5% to 16% of the total fish consumption.

An investigation from 1967 made by the National Institute for Public Health shows that about 25% of the fishermen and 2% of the total poulation eat more than 36 kg/head and year. 4% of the fishermen eat more than 70 kg/head and year.

A report from Westin (1970) shows that 93% of the total fish consumption consists of fish caught in salt or brackish water and the rest (7%) in lakes and rivers. Applying these percentage values to the fish consumption average 1978-1980 gives

13.7 kg from salt/brackish water
1.0 kg from lakes and rivers

Of the fish caught by Swedish fishermen in salt or brackish water (1980 total: 219378 tons)

58% was landed on the west coast
27% was landed on the south coast
15% was landed on the east coast

Finally the fish consumption in different parts of the world is presented. The values from ICRP 23 show landed weight, which means they are not fully comparable with the Swedish consumption values.

Far East	10 kg/head and year
Near East	4 kg/head and year
Africa	6 kg/head and year
Latin America	7 kg/head and year
Europe	14 kg/head and year
North America	9 kg/head and year
Oceania	8 kg/head and year

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