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**Uranium and radium in Finnsjön
– an experimental approach for
calculation of transfer factors**

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Studsvik Energiteknik AB, 1981-05-07

URANIUM AND RADIUM IN FINNSJÖN - AN EXPERIMENTAL
APPROACH FOR CALCULATION OF TRANSFER FACTORS

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ABSTRACT

The radiological safety studies for underground disposal of HLW show that the future individual and collective doses to an important extent may originate from groundwater borne radium and uranium which reach the biosphere. Indications that the dispersion rates presently used give rise to overestimations of calculated doses justified an investigation for more realistic turnover rates of radium and uranium than those which now are in use.

Within one of the sites selected for testing, the area around lake Finnsjön, a small number of environmental samples were collected and analyzed with respect to radium and uranium and the new transfer coefficients between soil and lake water were derived.

The dose rates obtained with the new transfer factors show a close agreement for radium and a slight increase for uranium compared with earlier calculations.

Godkänd av

Lars Ramberg

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1. INTRODUCTION

The principal mechanism for the dispersion of radioactive nuclides from HWL disposed underground is the dissolution of the contaminants in groundwater. The turnover rates of the nuclides between groundwater and soil or lake or sea water and bottom sediments are, however, hardly known and data related to Swedish conditions in particular are lacking. The on-going radiological safety studies which are part of the Swedish KBS program indicate that the potential future individual and collective radiation doses to a relatively important extent may originate from groundwater borne radium and uranium which reach the biosphere. The paucity of relevant information contributes to the wide spans used in the calculated dispersion rates of radium and uranium in those parts of the biosphere which are of importance when calculating the dose burden to man.

There are indications that the dispersion rates which at present are used in the KBS studies can give considerable overestimations of the calculated concentration in certain parts of the ecosystem with respect to radium and uranium. The apparent radiological importance of these nuclides may therefore constitute an unrealistic basis for site selection, design criteria and safety evaluation.

To justify another choice of turnover rates than those which now are in use, complementary information are needed, preferably from the potential disposal sites. Experimental efforts may improve the calculations of the contributions to the total dose burden from radium and uranium.

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2. DISTRIBUTION OF URANIUM AND RADIUM IN SOIL AND WATER

2.1 Migration of radium and uranium from bedrock deposits

Radium and uranium are present in the same kind of geological material. However, differences in geochemical behaviour with respect to weathering, sedimentation etc create a varied distribution of these elements in the environment. In decay chains with radioactive daughter products, the distribution of the daughter product also depends to a certain extent on the turnover process to which the parent nuclide is subjected. Uncertainties in the turnover of the parent nuclide can in some cases be amplified in the dose calculations for the daughter nuclide. Radium-226 with a half-life of $1.6 \cdot 10^3$ years is an alkaline earth metal with barium as nearest analogous and is a daughter nuclide of thorium-230 in the decay chain of uranium-234 - thorium-230 - radium-226.

The presence of radium and uranium in groundwater is dependent on their transfer rates from the bedrock to water and on their stability in solution. The transfers of radium and uranium are determined by the solubility of the bedrock which in turn is controlled by the chemical status of the water; oxygen and carbon dioxide tension, temperature etc (7). Oxygen and carbon dioxide are the gases which have the greatest influence on the mobility of radium and uranium, the solubility increases at low and high pH-values (4, 6, 18, 19). Diffusion processes between water and bedrock may promote an enrichment of minerals and alkaline and acid waters are usually enriched on radium and uranium.

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When radium and uranium are transferred into solution they are carried away with the groundwater. The transport schedule is dependent of the stability of the elements in the water as well as the sorption capacities of the aquifer rocks. The processes which can be responsible for the precipitation of radium and uranium are (6):

- hydrolysis of uranium hydroxides
- adsorption on surfaces of organic matter
- breakdown of complex ions
- reduction of the oxidation state
- formation of insoluble salts

The interactions between groundwater and bedrock are of special interest as governing mechanism for the release of activity to the biosphere from a hypothetical storage area for spent nuclear fuel. In view of the relatively large dose contributions involved, the decay chain of uranium-234 - thorium-230 - radium-226 is of particular interest. The amount of radium-226 appearing in the biosphere depends on how rapidly uranium and radium are transported through the soil, which influences the radium levels in the groundwater.

Radium follows its analogous elements barium and calcium. Adsorption processes are responsible for the enrichment of radium in e.g. peat and clay and radium is generally characterized by significantly lower biological availability. Table A.1 shows typical concentration values for radium-226 in soil from different geographical areas.

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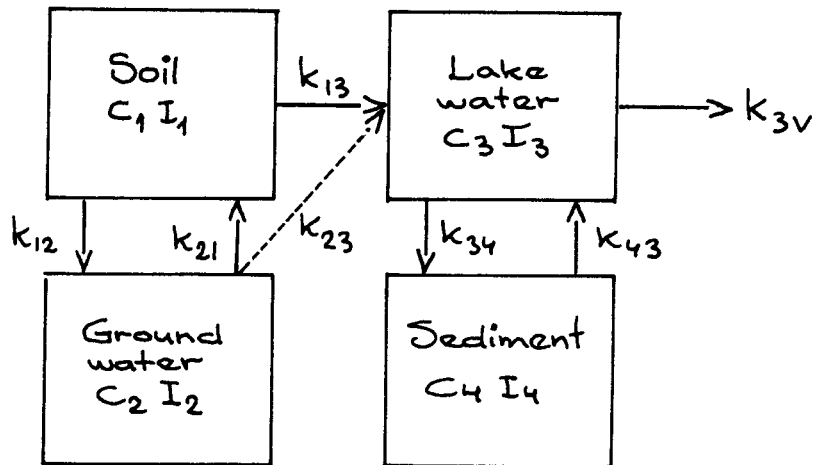
2.2 Distribution of uranium and radium in the hydrosphere

The contents of uranium and radium in natural waters are displayed in Table A.2. The average concentration of uranium and radium in sea water is $3 \mu\text{g}\cdot\text{l}^{-1}$ and $0.1 \text{pg}\cdot\text{l}^{-1}$ respectively. At steady state conditions the ratio of radium to uranium is the same as between their half-lives, e.g. $3.6\cdot 10^{-7}$. All values for radium/uranium higher than $3.6\cdot 10^{-7}$ indicate an accumulation of radium to uranium. A deficit of radium relatively to uranium exists in the oceans and to a lesser degree in lakes. This is dependent on the very low solubility of thorium-230 and its rapid transfer to the bottoms through sedimentation. This fact is supported by the excess of radium to uranium that has been observed in the deep sea sediments (8). Analysis of thorium-230 in water and sediments have given a deficit in water which is more marked than for radium. Inversely, there is an excess of thorium-230 in the sediments compared with Ra-226. This can be explained by the fact that Th-230 accumulate in the sediments where the daughter nuclide Ra-226 is generated and leached out to the overlaying water (9, 10). Uranium and radium concentration values for different fresh water localities in Sweden are displayed in Table A.3, while Table A.4 shows values of uranium and radium for lake water and sediment from lake Hornborgasjön and river Flån.

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3. DERIVATION OF TRANSFER COEFFICIENTS BETWEEN SOIL AND LAKE WATER

Principles for derivation of transfer coefficients from field measurements of concentrations in different parts of a defined ecosystem are displayed below.



C_i = Concentration of reservoir i

I_i = Inventory of reservoir i

V = The discharge volume flow from the lake

$k_{i,j}$ = Fractional transfer rate from reservoir "i" to "j", i.e. the so called transfer coefficient

The transfer of uranium and radium from ground-water to lake water is assumed to occur under equilibrium conditions in the soil and groundwater reservoir

$$\frac{k_{1,2}}{k_{2,1}} = \frac{I_2}{I_1}$$

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The direct transport from groundwater to the lake may be included whereby the above equation is expressed as follows

$$\frac{k_{1,2}}{k_{2,1} + k_{2,3}} = \frac{I_2}{I_1}$$

Similar chemical equilibrium is assumed to occur between lake water and lake sediment as between the soil and groundwater reservoirs

$$\frac{k_{3,4}}{k_{4,3}} = \frac{I_4}{I_3}$$

Making it a condition that the amount of activity which is supported to the lake is balanced by the amount removed through runoff and radioactive decay in water column and bottom sediment, the leaching rate from soil to lake water becomes

$$k_{1,3} \cdot I_1 = k_{3,v} \cdot I_3 + \lambda \cdot I_3 + \lambda \cdot I_4$$

Substitution of

$$k_{3,v} \cdot I_3 = C_3 \cdot V$$

gives

$$k_{1,3} = \frac{C_3 \cdot V + \lambda (I_3 + I_4)}{I_1}$$

The amount of groundwater directly discharging into the lake is assumed to be of minor importance, making $k_{1,3} \gg k_{2,3}$.

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3.1 Area description

At the request of KBS, the Swedish Geological Survey, (SGU) has started a long-term project aimed to clarify the geohydrological situation of a test area close to the lake Finnsjön in northern Uppland (1).

The investigation area is about 25 km² and is limited by a water-shed. The test area is relatively flat, rich in outcrops and often swampy. Big areas of forests and agriculture have been drained by ditches. The soil mainly consists of moraine. The clay content varies between 1 and 4 % (1). In connection with test drillings, chemical analysis of the bedrock have been performed (2).

Lake Finnsjön which is connected to the investigation area is a long and narrow lake. Depth soundings have recorded a maximum depth of about 4 m. The lake can be characterised as oligotrophic - dystrophic, i.e. with low fertility and brown water. It is connected to a lake system which extends in N-S direction and which discharges towards north. The lake system is fed with water from surrounding forests and bogs. The area seemed suitable to test the applicability of the approach described to calculate the transfer of U and Ra within this defined ecosystem.

3.2 Environmental parameters

The following environmental parameters were assumed to be relevant for the investigation area.

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Lake FinnsjönArea 4.4 km².

Average depth 2 m.

Volume 8.8·10⁹ l.

Of the discharge from the drainage area around the lake system 80 % is supplied to the area upstream lake Finnsjön. The average runoff from Finnsjön is calculated to 1.5·10¹⁰ l·yr⁻¹. The nominal residence time for the water of lake Finnsjön then becomes

$$\frac{8.8 \cdot 10^9}{1.5 \cdot 10^{10}} = 0.6 \text{ yr}$$

Lake sediment

The sediment area is assumed to be the same as for the lake surface, i.e. 4.4 km². A 10 cm deep sediment layer is assumed to take part actively in the exchange process with the overlaying water. The water content of the sediment is 55 % and its density is calculated to 1.2, which will give a total active sediment weight (dry weight) of 2.4·10¹¹ g.

Drainage area

The runoff from surrounding drainage area is calculated to 10 l·s⁻¹ km⁻². A runoff from lake Finnsjön of 1.5·10¹⁰ l·yr⁻¹ will give a drainage area of 48 km².

Soil

A drainage area of 48 km² and an assumed depth of 2 m will give a soil volume of 9.6·10⁷ m³. The average density of 2.2 gives a total weight of 2.1·10¹⁴ g actively participating in the exchange processes.

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3.3 Sampling and analysis

During September 1979 a few samples were randomly collected from the Finnsjön area:

- water from the lake
- bottom sediment
- water from a well
- soil samples

The water sample was collected from a boat. The sediment sample was collected with an Ekman grab which penetrated 10 cm down into the substrate. Groundwater was collected from a well situated about 100 m from the shoreline. Soil samples were collected from the uppermost surface layer and at 0.5 m depth.

To compare the concentration values in lake Finnsjön with values obtained in the marine environment, water and sediment samples from the Baltic (Tvären Bay) were analyzed with respect to uranium and radium.

The uranium contents of the different samples were determined with delayed neutron technique while the radium contents were analyzed radio-chemically.

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4. CALCULATION OF TRANSFER RATES OF
URANIUM AND RADIUM BETWEEN SOIL AND
LAKE WATER

4.1 Uranium

The concentration values of uranium in the different reservoirs of the ecosystem are displayed in Figure B.1.

The calculated inventories of uranium in the different reservoirs are displayed in Figure B.2.

Calculation of the transfer coefficient $k_{1,3}$ between soil and lake water using the parameters displayed beneath:

$$\begin{aligned} C_3 &= 1.7 \cdot 10^{-2} \text{ Bq l}^{-1} \\ V &= 1.5 \cdot 10^{10} \text{ l} \cdot \text{yr}^{-1} \\ I_1 &= 8.88 \cdot 10^{13} \text{ Bq} \\ I_3 &= 1.52 \cdot 10^8 \text{ Bq} \\ I_4 &= 1.48 \cdot 10^{10} \text{ Bq} \\ \lambda &= 1.55 \cdot 10^{-10} \text{ yr}^{-1} \end{aligned}$$

$$k_{1,3} = \frac{C_3 V + \lambda (I_3 + I_4)}{I_1} = 2.9 \cdot 10^{-6} \text{ yr}^{-1}$$

When choosing the environmental parameters, the inventory in the soil reservoir may be regarded as the one comprising the largest uncertainty. Dependent whether one uses the concentration value of surface soil or deep soil in the calculation, the transfer coefficient will range between $(3 - 5)10^{-6} \text{ yr}^{-1}$.

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4.2 Radium

Concentration values and calculated inventories of radium for the different reservoirs are displayed in Figure B.3 and B.4.

For the water samples of Finnsjön and Baltic, values below the detection limit were obtained.

Calculation of the transfer coefficient $k_{1,3}$ between soil and lake water using the parameters displayed beneath:

$$\begin{aligned}
 C_3 &= < 6 \cdot 10^{-4} \text{ Bq l}^{-1} \\
 V &= 1.5 \cdot 10^{10} \text{ l yr}^{-1} \\
 I_1 &= 1.07 \cdot 10^{13} \text{ Bq} \\
 I_3 &= < 5.55 \cdot 10^6 \text{ Bq} \\
 I_4 &= 9.25 \cdot 10^9 \text{ Bq} \\
 \lambda &= 4 \cdot 10^{-4} \text{ yr}^{-1}
 \end{aligned}$$

$$k_{1,3} = 1.3 \cdot 10^{-6} \text{ yr}^{-1}$$

Dependent whether one uses the concentration value of surface or deep soil in the calculation, the transfer coefficient will range between $(1 - 2)10^{-6} \text{ yr}^{-1}$.

4.3 Choice of environmental parameters

A sensitivity test will indicate how variations of environmental parameters will have influence on $k_{1,3}$. Using the set of parameters chosen in the calculations of $k_{1,3}$ for uranium and radium, as a starting point, and varying the parameters one by one, will give the percentage deviation of k from the original value (Table A.5 - A.6, Figure B.5).

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- A change of the lake volume does not affect the transfer coefficient.
- A change of the sediment volume affect $k_{1.3}$ (Ra) but not $k_{1.3}$ (U).
- $k_{1.3}$ is inversely proportional to the soil volume.
- $k_{1.3}$ is proportional to the outflow rate from the lake
- The concentration values for Ra in lake water and sediment will cause a change of k in a proportional manner, while the concentration of U in lake sediment does not affect k_U .

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5. CALCULATION OF GROUNDWATER TURNOVER RATES

The transfer of groundwater can be derived from differences in nuclide concentrations in groundwater and lake water. Assume the same amount of the nuclides are brought to the lake as the amount removed from it, i.e. steady state conditions prevail:

$$V_{gw} \cdot a_{gw} \cdot k_{2,3} = V_1 \cdot a_1 \cdot k_{3,V}$$

$$k_{2,3} = \frac{V_1 \cdot a_1}{V_{gw} \cdot a_{gw}} \cdot k_{3,V} \text{ yr}^{-1}$$

V = the volume of the ground water (gw) and lake reservoir (l)

a = the specific activity of the ground water and lake water

With a ground water reservoir that comprises 20 % of the total volume of the soil reservoir we become

$$V_{gw} = 0.2 \cdot 9.6 \cdot 10^{10} = 1.92 \cdot 10^{10} \text{ l}$$

Uranium values

With the same parameters used to calculate the transfer coefficient $k_{1,3}$

$$\begin{aligned} V_1 &= 8.8 \cdot 10^9 \text{ l} \\ a_{gw} &= 2.6 \cdot 10^{-2} \text{ Bq l}^{-1} \\ a_1 &= 1.7 \cdot 10^{-2} \text{ Bq l}^{-1} \end{aligned}$$

$$k_{3,V} = \frac{1.5 \cdot 10^{10}}{8.8 \cdot 10^9} = 1.7 \text{ yr}^{-1}$$

we arrive to

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$$k_{2,3} = \frac{8.8 \cdot 10^9}{1.9 \cdot 10^{10}} \cdot \frac{1.7 \cdot 10^{-2}}{2.6 \cdot 10^{-2}} \cdot 1.7 = 0.51 \text{ yr}^{-1}$$

The turnover of the ground water pool will then be about 2 yr.

Radium values

$$\begin{aligned} a_{\text{gw}} &= 2.0 \cdot 10^{-3} \text{ Bq l}^{-1} \\ a_1 &< 6.29 \cdot 10^{-4} \text{ bq l}^{-1} \end{aligned}$$

The same calculations will give

$$k_{2,3} = 0.3 \text{ yr}^{-1}$$

Since a_1 is an upper limit value the same should hold for the calculated $k_{2,3}$ and the turnover rate of the ground water pool will then be > 4 yr.

Variations of environmental parameters and anomalies in the uranium and radium concentration values within the investigation area will contribute to fluctuations in the calculated turnover rate of the ground water (Table A.7). To create a more solid foundation for the derivations of the k-factors will demand a larger sampling effort with a more detailed cover of the precipitation area.

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6. SIGNIFICANCE OF CALCULATED TRANSFER
RATES FOR THE DOSE BURDEN TO MAN

The obtained transfer rates $(3 - 5) \cdot 10^{-6} \text{ yr}^{-1}$ and $(1 - 2) \cdot 10^{-6} \text{ yr}^{-1}$ for uranium and radium respectively between soil and lake water can be compared with the values $1.2 \cdot 10^{-3} \text{ yr}^{-1}$ and $1.8 \cdot 10^{-4} \text{ yr}^{-1}$ earlier used (3), where the transfer rate of uranium has been derived from leaching experiments in agriculture soils (4). Uptake of uranium and radium in clover and leaching rates in different soils were studied using uranium concentrated soil residues which were mingled with agriculture soils. Comparison was also made with soils in which uranyl-nitrate-solution was added. The experiments from (4) showed an increased transport in the acid leaching-water compared to the uranyl-nitrate-solution with higher pH; the leaching rate amounted to about 1 % per year for a one m thick earth-layer, while the radium content of the perkolated water was almost nil. These experimental studies indicate that uranium is leached considerably more rapidly through Swedish moraine soils than has been assumed in previous studies on vitrified wastes (17). The field and laboratory studies which have been carried out with respect to strontium and radium indicate that radium is dispersed considerably more slowly than strontium through soils under widely varying conditions. In our previous studies, however, it has been assumed that radium is leached through the soil at the same rate as strontium.

The transfer coefficients used in earlier investigations together with the values obtained in this study are displayed beneath

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$k_{1.3}(U)$	$k_{1.3}(Ra)$	Ref
$1.2 \cdot 10^{-3}$	$1.8 \cdot 10^{-4}$	(3)
$(3 - 5) \cdot 10^{-6}$	$(1 - 2) \cdot 10^{-6}$	this study

It is generally agreed that low leach rates reduce the potential hazard. However, the composition of the leaching solution, flow rates, temperatures and other factors may vary significantly with the environment.

To test the radiological consequences for man when using the new transfer factors for the soil-water system in this study compared to (3), an analysis was performed with the BIOPATH program. Simulating a continuous release of 10^{-6} Ci yr⁻¹ ($3.7 \cdot 10^4$ Bq yr⁻¹) of uranium-234 into the ground water pool of the biosphere model (3) and changing the transfer factor between soil and lake water will give the dose rates displayed in Table A.8. The dose rates show a close agreement for radium and a slight increase of the regional dose for uranium, compared with (3). In the case of uranium the lower transfer factor will cause an accumulation in the soil compartment which will give rise to an increased dose rate. For radium the relatively rapid decay rate will counterbalance the increased accumulation in the soil, and the dose rate remains unaltered.

For radium-226 generated by uranium-234, the increased dose rates obtained reflects the slower transfer of uranium which causes an increased amount of uranium and consequently an increased amount of radium in the soil compartment. This means a more intense contact with the

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groundwater and, as the drinking water constitutes the principal part of the dose contribution, an increased dose rate.

It can however be discussed whether the water discharge to the lake from surrounding land areas consists of surface runoff or if it mainly originates from the groundwater pool. Table A.9 displays the obtained dose rates if the transfer factor between groundwater and lake is changed to the new ones for the soil-lake. We arrive to larger differences in the calculated dose rates than in previous case. This reflects the importance of deeper knowledge concerning the transfer of radionuclides in different ecological sub-systems to calculate the dose burden to man.

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Table A.1

Typical concentrations of Ra-226 in soil excluding high background areas. From (6)

Region	Bq g ⁻¹ (dry weight)	Ref
British Guiana	$(1.1-5.9) \cdot 10^{-2}$	13
Czechoslovakia	$3.7 \cdot 10^{-3} - 1.4 \cdot 10^{-1}$	14
Germany	$5.6 \cdot 10^{-3} - 4.8 \cdot 10^{-2}$	15
Ireland	$4.8 \cdot 10^{-2} - 1.1 \cdot 10^{-1}$	14
Italy	$2.7 \cdot 10^{-2} \pm 4.1 \cdot 10^{-3}$	16
United Kingdom	$3.0 \cdot 10^{-3} - 5.6 \cdot 10^{-2}$	14, 15
USA	$3.0 \cdot 10^{-2} - 1.0 \cdot 10^{-1}$	14
USSR	$3.7 \cdot 10^{-3} - 4.8 \cdot 10^{-2}$	14

Table A.2

Content of Uranium and Radium in Natural Waters^a. From (6)

Type of Waters	Natural Conditions	Radium (g/liter)			Uranium (g/liter)			$\frac{Ra}{U}$
		min.	max.	mean	min.	max.	mean	
Surface Waters	Oceans and seas	8.0×10^{-14}	4.5×10^{-11}	1.0×10^{-13}	3.6×10^{-8}	5.0×10^{-6}	3.0×10^{-6}	3×10^{-8}
	Lakes	1.0×10^{-13}	8.0×10^{-12}	1.0×10^{-12}	2.0×10^{-7}	4.0×10^{-2}	8.0×10^{-6}	1×10^{-7}
	Rivers	1.0×10^{-13}	4.0×10^{-12}	2.0×10^{-13}	2.0×10^{-8}	5.0×10^{-5}	6.0×10^{-7}	3×10^{-7}
Waters of Sedimentary Rocks	Zone of intensive water circulation	1.0×10^{-13}	6.0×10^{-12}	2.0×10^{-12}	2.0×10^{-7}	8.0×10^{-6}	5.0×10^{-6}	5×10^{-7}
	Zone of highly impeded water circulation	1.0×10^{-11}	1.0×10^{-8}	3.0×10^{-10}	2.0×10^{-8}	6.0×10^{-6}	2.0×10^{-7}	1×10^{-3}
Waters of Magmatic Acid Rocks	Zone of intensive water circulation (waters of the weathering shell)	1.0×10^{-12}	7.0×10^{-12}	2.0×10^{-12}	2.0×10^{-7}	3.0×10^{-5}	7.0×10^{-6}	1×10^{-6}
	Zone of impeded water circulation (waters of deep tectonic fissures)	2.0×10^{-12}	9.0×10^{-12}	4.0×10^{-12}	2.0×10^{-7}	8.0×10^{-6}	4.0×10^{-6}	2×10^{-6}
Waters of Uranium Deposits	Zone of intensive water circulation (waters of the oxidation zone)	8.0×10^{-12}	2.0×10^{-9}	8.0×10^{-11}	5.0×10^{-5}	9.0×10^{-2}	6.0×10^{-4}	1×10^{-7}
	Zone of impeded water circulation (waters of the reduction zone)	1.0×10^{-11}	8.0×10^{-10}	6.0×10^{-11}	2.0×10^{-6}	3.0×10^{-5}	8.0×10^{-6}	1×10^{-5}

^a Modification of table from A. N. Tokarev and A. V. Shcherbakov, Radiohydrogeology, Moscow (1956) AEC-tr-4100 (1960).

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Table A.3

Uranium and radium concentration values for different fresh water localities in Sweden. From (5)

Uranium

<u>Water type</u>	<u>Locality</u>	<u>Concentration (Bq ℓ^{-1})</u>	<u>Ref</u>
Surface water	Pösan	$2.8 \cdot 10^{-1}$	(5)
	Flian	$(1.6-2.1) \cdot 10^{-1}$	(5)
	Hornborgasjön	$(1.3-3.5) \cdot 10^{-2}$	(5)
River water		$(5-3.5) \cdot 10^{-2}$	(11)
Bog water	Masugnsbyn	$3.8 \cdot 10^{-1}$	(11)
Spring water	Northern Sweden	$(4.3-20) \cdot 10^{-3}$	(11)

Radium

<u>Water type</u>	<u>Locality</u>	<u>Concentration (Bq ℓ^{-1})</u>	<u>Ref</u>
Surface water	Flian	$(8.1-13) \cdot 10^{-3}$	(5)
	Hornborgasjön	$(5.9-8.1) \cdot 10^{-3}$	(5)
Drinking water		$(7.3-37) \cdot 10^{-3}$	(12)

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Table A.4

Values of uranium and radium for lake water and sediment from lake Hornborgasjön and river Flían (Sweden). From (5).

Uranium

<u>Water (Bq l⁻¹)</u>	<u>Sediment (Bq kg⁻¹)</u>
1.0·10 ⁻¹	1.0·10 ²
2.2·10 ⁻¹	6.3·10 ¹
2.3·10 ⁻¹	6.3·1
1.4·10 ⁻²	6.3·10 ¹
3.6·10 ⁻²	4.4·10 ¹
1.6·10 ⁻¹	7.0·10 ¹

Radium

<u>Water (Bq l⁻¹)</u>	<u>Sediment (Bq kg⁻¹)</u>
1.3·10 ⁻²	1.0·10 ²
8.1·10 ⁻³	6.9·10 ¹
8.1·10 ⁻³	4.8·10 ¹
5.9·10 ⁻³	1.9·10 ¹
1.2·10 ⁻²	2.2·10 ¹
1.0·10 ⁻²	5.2·10 ¹

Table A.5

Percentage deviation of k_u when varying one environmental parameter from +50 % to -50 % by volume

	-50	-40	-30	-20	-10	±0	010	+20	+30	+40	+50 % deviation
lake volume	±0	±0	±0	±0	±0	±0	±0	±0	±0	±0	±0
sediment volume	±0	±0	±0	±0	±0	±0	±0	±0	±0	±0	±0
soil volume	+100	+67	+43	+25	+11	±0	-9	-17	-23	-29	-33
runoff	-50	-40	-30	-20	-10	±0	+10	+20	+30	+40	+50
concentration lake	-50	-40	-30	-20	-10	±0	+10	+20	+30	+40	+50
concentration sediment	±0	±0	±0	±0	±0	±0	±0	±0	±0	±0	±0
concentration soil	+100	+67	+43	+25	+11	±0	-9	-17	-23	-29	-33

Table A.6

Percentage deviation of k_{Ra} when varying one environmental parameter from + 50 % to -50 % by volume

	-50	-40	-30	-20	-10	±0	+10	+20	+30	+40	+50 % deviation
lake volume	±0	±0	±0	±0	±0	±0	±0	±0	±0	±0	±0
sediment volume	-15	-12	-9	-6	-3	±0	+3	+6	+9	+12	+15
soil volume	+100	+67	+43	+25	+11	±0	-9	-17	-23	-29	-33
runoff	-35	-28	-20	-14	-7	±0	+7	+14	+21	+28	+35
concentration lake	-37	-29	-20	-13	-8	±0	+8	+13	+20	+29	+37
concentration sediment	-15	-12	-9	-6	-3	±0	+3	+6	+19	+12	+5
concentration soil	+100	+67	+43	+25	+11	±0	-9	-17	-23	-29	-33

Table A.7

Percentage deviation of groundwater turnover rate when varying one environmental parameter from +50 % to -50 % by volume

	-50	-40	-30	-20	-10	±0	+10	+20	+30	+40	+50 % deviation
lake volume	-50	-40	-30	-20	-10	±0	+10	+20	+30	+40	+50
ground- water volume	+100	+67	+43	+25	+11	±0	-9	-17	-23	-29	-33
concentration lake (U)	-50	-40	-30	-20	-10	±0	+10	+20	+30	+40	+50

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Table A.8

Dose rates obtained with the new set of transfer coefficients between soil and lake water compared with (3)

Ra 226

$k_{1,3}$ (yr^{-1})	maximum local indi- vidual dose rate (Sv yr^{-1})	maximum regional individual dose rate (Sv yr^{-1})	coll dose rate (manSv yr^{-1})	ref
$1.8 \cdot 10^{-4}$	$2.1 \cdot 10^{-8}$	$1.7 \cdot 10^{-9}$	$4.5 \cdot 10^{-6}$	(3)
$1.0 \cdot 10^{-6}$	$2.1 \cdot 10^{-8}$	$1.8 \cdot 10^{-9}$	$4.7 \cdot 10^{-6}$	(this study)

U 234

$k_{1,3}$ (yr^{-1})	maximum local indi- vidual dose rate (Sv yr^{-1})	maximum regional individual dose rate (Sv yr^{-1})	coll dose rate (manSv yr^{-1})	reg
$1.2 \cdot 10^{-3}$	$6.0 \cdot 10^{-10}$	$3.4 \cdot 10^{-11}$	$2.8 \cdot 10^{-7}$	(3)
$3.0 \cdot 10^{-6}$	$6.0 \cdot 10^{-10}$	$5.8 \cdot 10^{-11}$	$3.4 \cdot 10^{-7}$	(this study)

Ra-226 generated by U-234

$k_{1,3}$ (yr^{-1})	maximum local indi- vidual dose rate (Sv yr^{-1})	maximum regional individual dose rate (Sv yr^{-1})	coll dose rate (manSv yr^{-1})	ref
$1.8 \cdot 10^{-4}$	$7.6 \cdot 10^{-12}$	$4.1 \cdot 10^{-10}$	$5.3 \cdot 10^{-6}$	(3)
$1.0 \cdot 10^{-6}$	$8.8 \cdot 10^{-12}$	$1.0 \cdot 10^{-9}$	$6.7 \cdot 10^{-6}$	(this study)

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Table A.9

Dose rates obtained with the new set of transfer coefficients between ground water and lake water compared with (3)

Ra 226

$k_{2,3}$ (yr^{-1})	maximum local indi- vidual dose rate (Sv yr^{-1})	maximum regional individual dose rate (Sv yr^{-1})	coll dose rate (manSv yr^{-1})	ref
$4.0 \cdot 10^{-4}$	$2.1 \cdot 10^{-8}$	$1.7 \cdot 10^{-9}$	$4.5 \cdot 10^{-6}$	(3)
$1.0 \cdot 10^{-6}$	$2.1 \cdot 10^{-8}$	$2.4 \cdot 10^{-9}$	$6.1 \cdot 10^{-6}$	(this study)

U 234

$k_{2,3}$ (yr^{-1})	maximum local indi- vidual dose rate (Sv yr^{-1})	maximum regional individual dose rate (Sv yr^{-1})	coll dose rate (manSv yr^{-1})	ref
$1.0 \cdot 10^{-3}$	$6.0 \cdot 10^{-10}$	$3.4 \cdot 10^{-11}$	$2.8 \cdot 10^{-7}$	(3)
$3.0 \cdot 10^{-6}$	$6.0 \cdot 10^{-10}$	$3.4 \cdot 10^{-10}$	$1.0 \cdot 10^{-6}$	(this study)

Ra-226 generated by U-234

$k_{2,3}$ (yr^{-1})	maximum local indi- vidual dose rate (Sv yr^{-1})	maximum regional individual dose rate (Sv yr^{-1})	coll dose rate (manSv yr^{-1})	ref
$4.0 \cdot 10^{-4}$	$7.6 \cdot 10^{-12}$	$4.1 \cdot 10^{-10}$	$5.3 \cdot 10^{-6}$	(3)
$1.0 \cdot 10^{-6}$	$7.5 \cdot 10^{-12}$	$7.7 \cdot 10^{-9}$	$2.3 \cdot 10^{-5}$	(this study)

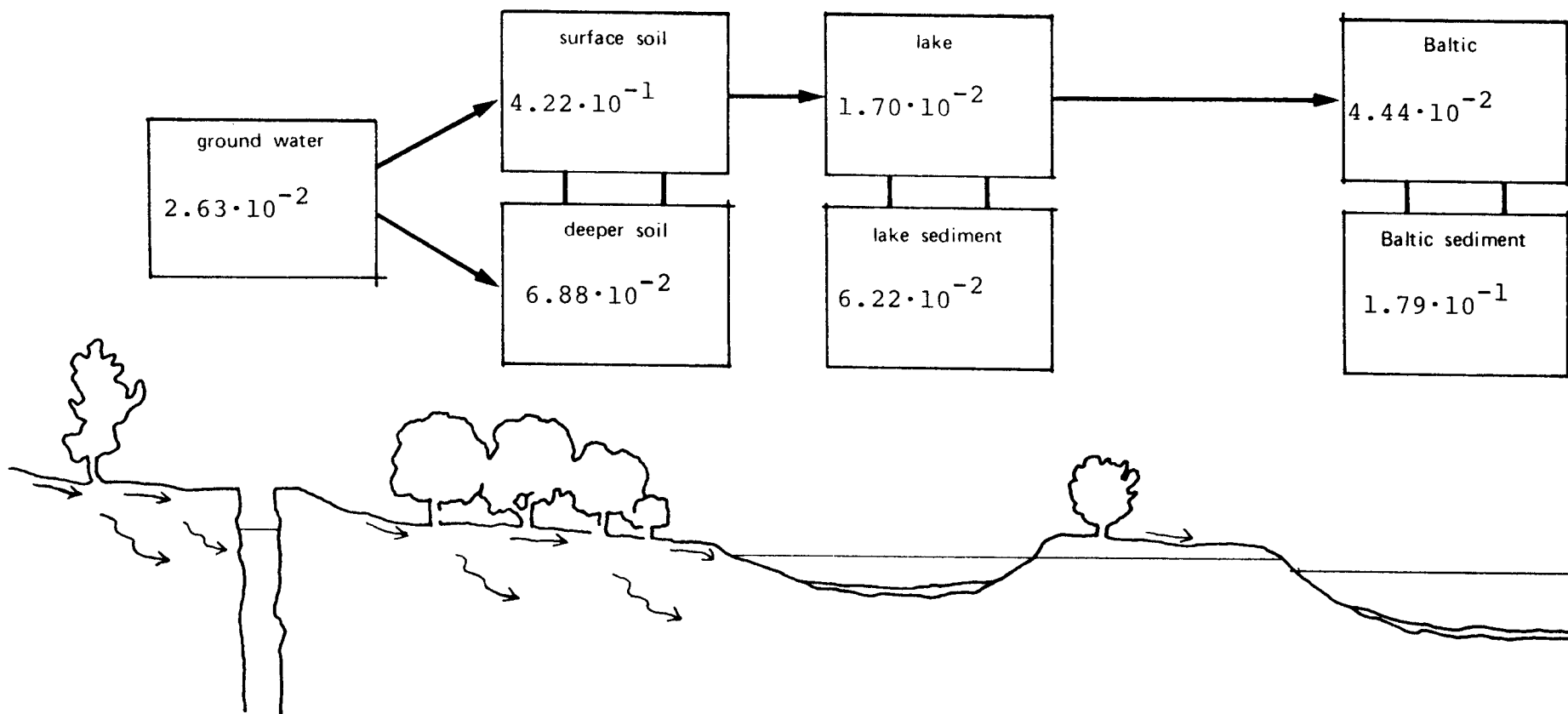


Figure B.1

Specific activity concentrations of U in the different reservoirs, expressed in Bq l⁻¹ and Bq g⁻¹ (dry weight) respectively.

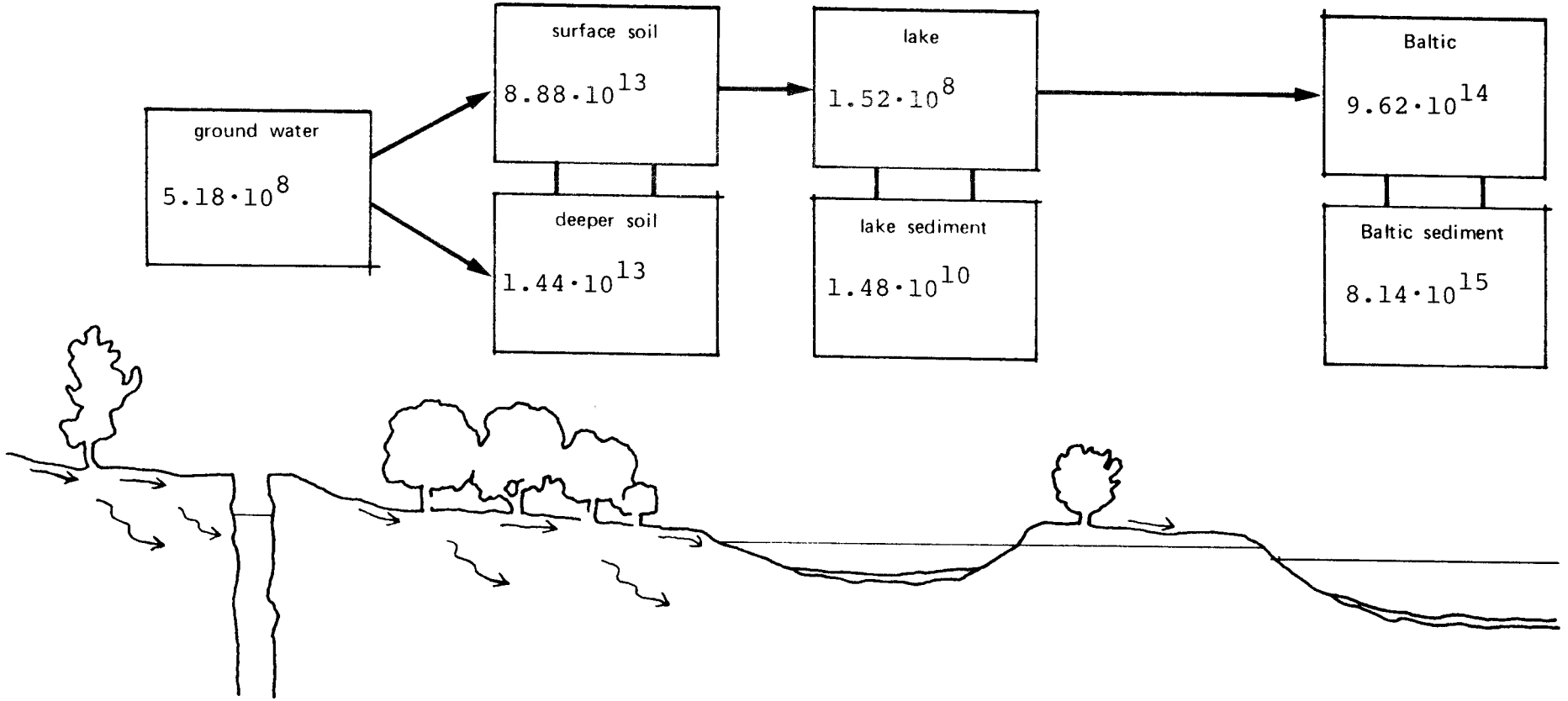


Figure B.2
Inventories of U in the different reservoirs expressed in Bq.

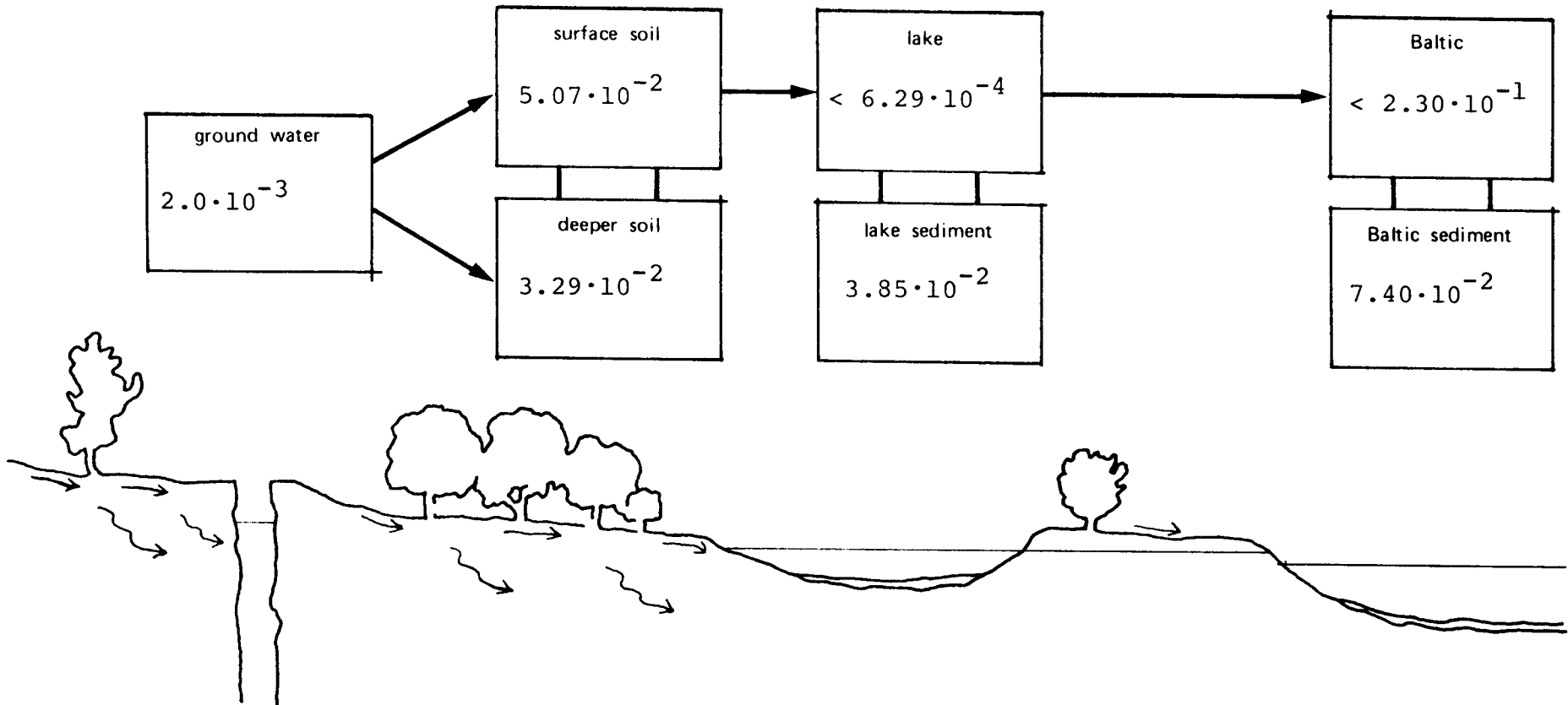


Figure B.3

Specific activity concentrations of Ra in the different reservoirs, expressed in Bq l^{-1} and Bq g^{-1} (dry weight) respectively.

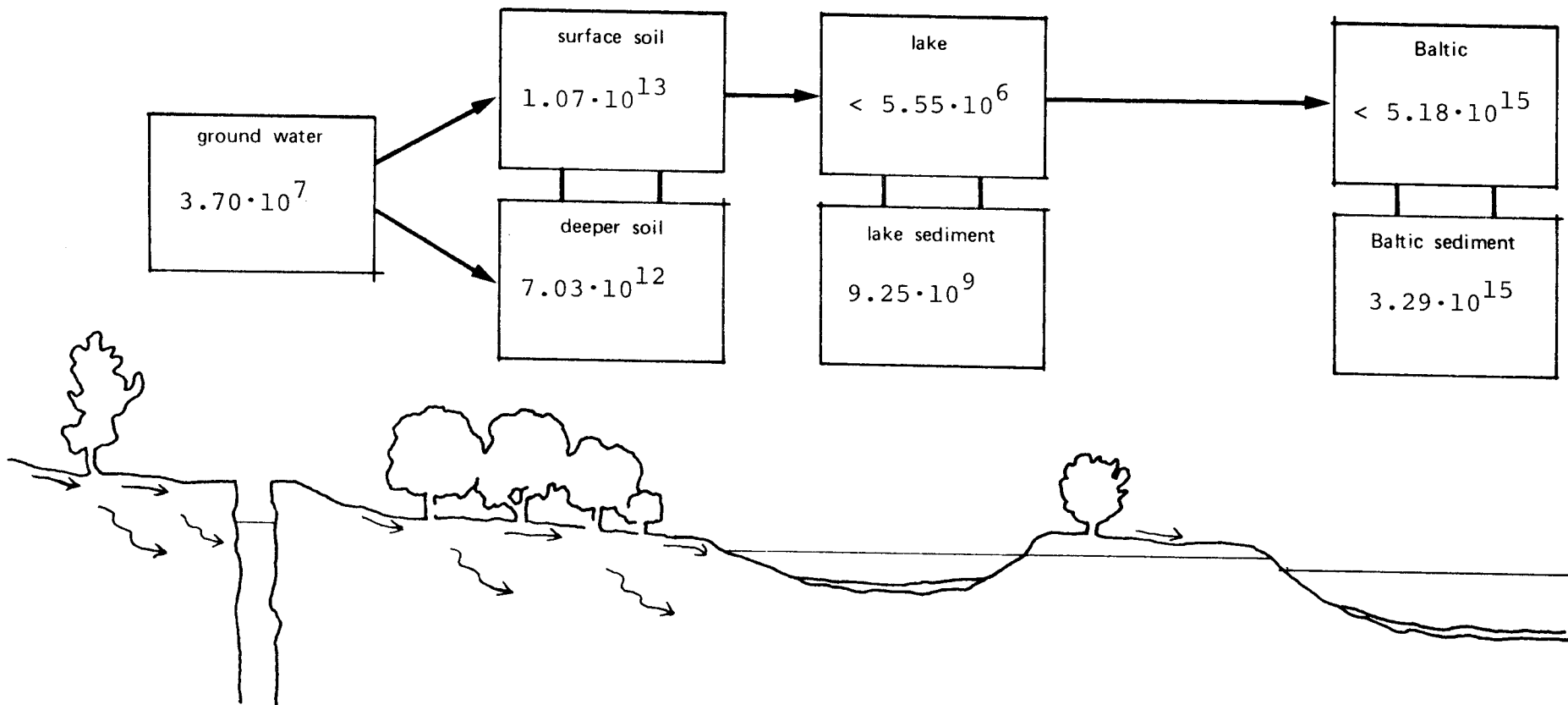


Figure B.4

Inventories of Ra in the different reservoirs expressed in Bq.

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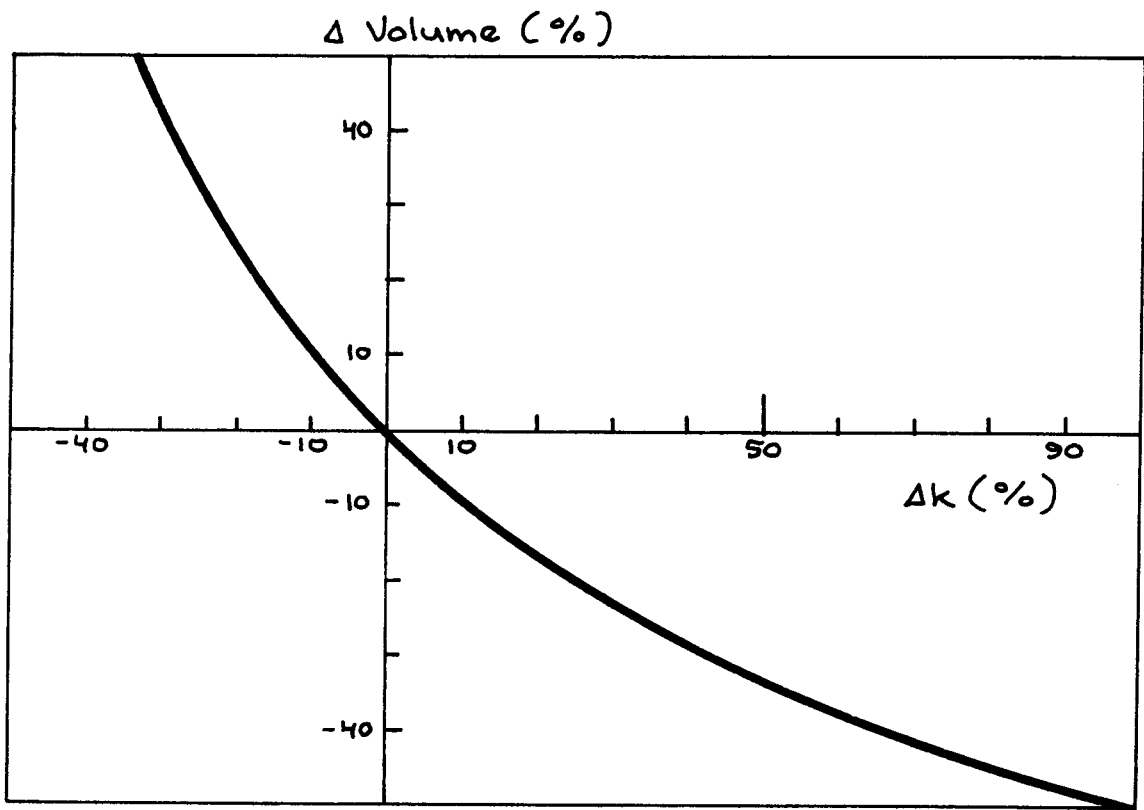


Figure B.5

Percentage deviation of k_U and k_{Ra} with variation of the soil volume.

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