

**Lake Tranebärssjön:
Basic Characteristics and Evaluation of Restoration Options**

Restoration Strategies for Radioactively Contaminated
Sites and their Close Surroundings
RESTRAT

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1 Terms of reference

This report is submitted as Technical Deliverable ‘TD13’ against the requirements of Work Package 1 for the Restoration Strategies for Radioactive Sites and their Close Surroundings, RESTRAT, project.

The RESTRAT project, which is funded by the European Commission under the Nuclear Fission Safety Programme, has the overall objective of developing generic methodologies for ranking restoration techniques as a function of contamination and site characteristics. The development of this generic methodology is based on an analysis of existing remediation methodologies and contaminated sites, and is structured in the following steps:

- 1 characterise relevant contaminated sites;
- 2 identify and characterise relevant restoration techniques;
- 3 assess the radiological impact;
- 4 develop and apply a selection methodology for restoration options;
- 5 formulate generic conclusions and develop a manual.

This work package, which is jointly funded by the Swedish Radiation Protection Institute, is concerned with the dose assessments for Lake Tranebärssjön and impact of restoration techniques. The aims of this Work Package are to:

- 1 describe the main geographic, hydrographic and geological features of the site;
- 2 describe the local human activities;
- 3 describe the major sources of radioactive contamination;
- 4 identify possible restoration options;
- 5 assess the impacts of the radioactive contamination and the effect of each restoration option;
- 6 evaluate and rank each remediation option, in terms of the suitability for the Lake Tranebärssjön site.

2 Introduction

The Ranstad Tailing Site is situated in the southern part of Sweden, in the Billingen-Häggum district, Figure 1. The tailing stems from a former uranium processing plant of AB Atomenergi, which operated on the uranium mined in a nearby open pit mine. Nowadays these plant processes filter resin from fuel fabrication at Västerås and waste from incineration handling of low-level radioactive waste at Studsvik.

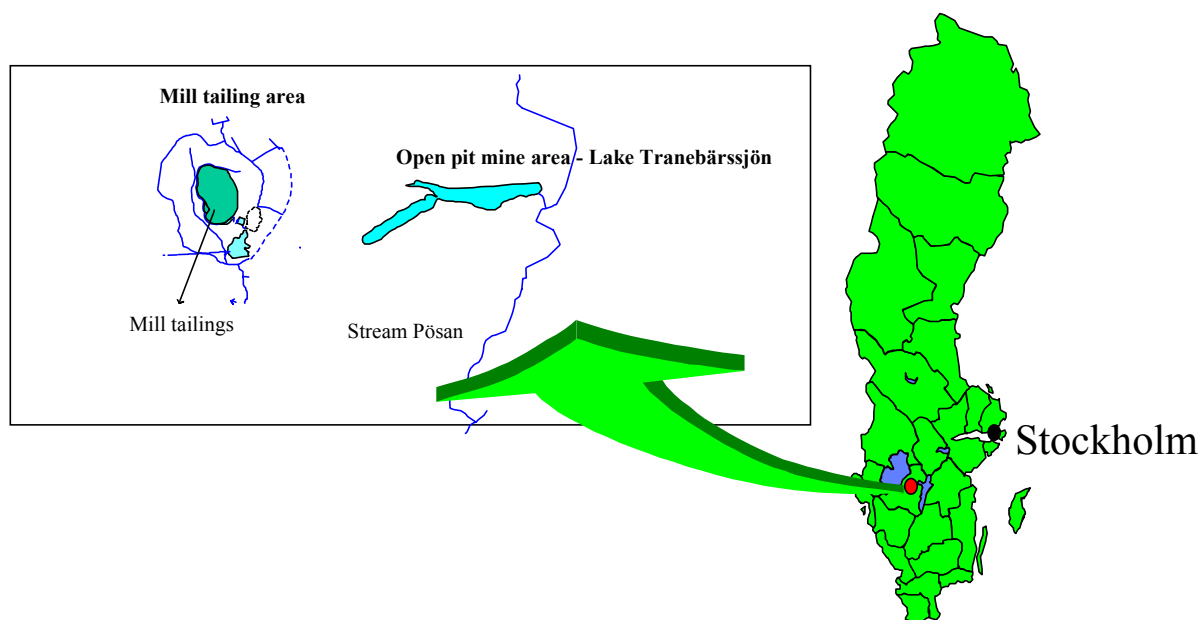


Figure 1 Geographical location of the Ranstad tailing site and the former open pit mine Lake Tranebärssjön.

The industrial facilities for the uranium mining and milling in Ranstad, Sweden were built between 1960 and 1965. Test operations (mining and ore processing) were carried out between 1965 and 1969, during which period 1.5 million tonnes of alum shale were mined. About 215 tonnes of uranium oxide were extracted from the ore. Studsvik AB was responsible for the planning and performance of the remediation until 1991. Another company, AB SVAFO, a sister company to the Swedish Nuclear Fuel and Waste Management Company, was founded in 1992 and has since then been responsible for the completion of the remediation.

In 1984 the mining permit ceased and the planning for the remediation began at that time. The planning resulted in a remediation plan which was presented for the responsible county authority, Länsstyrelsen Skaraborgs län, in October 1988 (nowadays Länsstyrelsen Västra Götaland). After a comprehensive review the county administration presented the conditions for the remediation in January 1990. The content of the plan can involve covering the mill tailing area with a multi-layer system (the dry depository) and as filling the open pit mine with weathered shale and water (the wet depository). The objective of the restoration work is that no maintenance work would be necessary in the future. An environmental control program was initiated before the restoration period. Environmental measurements have since then been carried out.

The mill tailing area (see Figure 1) is described in RESTRAT-TD10 (Stiglund, 1999). The Lake Tranebärssjön originates from the trench, 2000 m long, 100-200 m wide and 15 m deep, left by the alum shale mining operations (see Figure 2). The lake has existed since 1990, when groundwater pumping in the open pit mine stopped. The general aim of the remediation was to re-shape the open pit into a lake. An overflow into a nearby Pösan stream keeps the water level constant.

3 General site characterisation

3.1 Description of the area

3.1.1 Geography and topography

The Ranstad area and the former open pit mine, Lake Tranebärssjön, are situated in the southern part of Sweden in the county of Västra Götaland, at the borderline between the communities of Skövde and Falköping. The location of the area is about 20 km south of the city of Skövde, in the Billingen–Häggum district next to the Billingen mountain.

3.1.2 Geology and hydrogeology

The surroundings of Lake Tranebärssjön vary. One lakeside mainly consists of the undisturbed geological formation, with moraine on the top, then a limestone layer, then an alum shale layer, and the lowest layer of interest is sandstone. The other side of the lake, including most of the lake bottom, consists of backfilled limestone and alum shale, covered by a thinner layer of backfilled moraine on the top, see Figure 2, and Table 1 for mineralogical description.

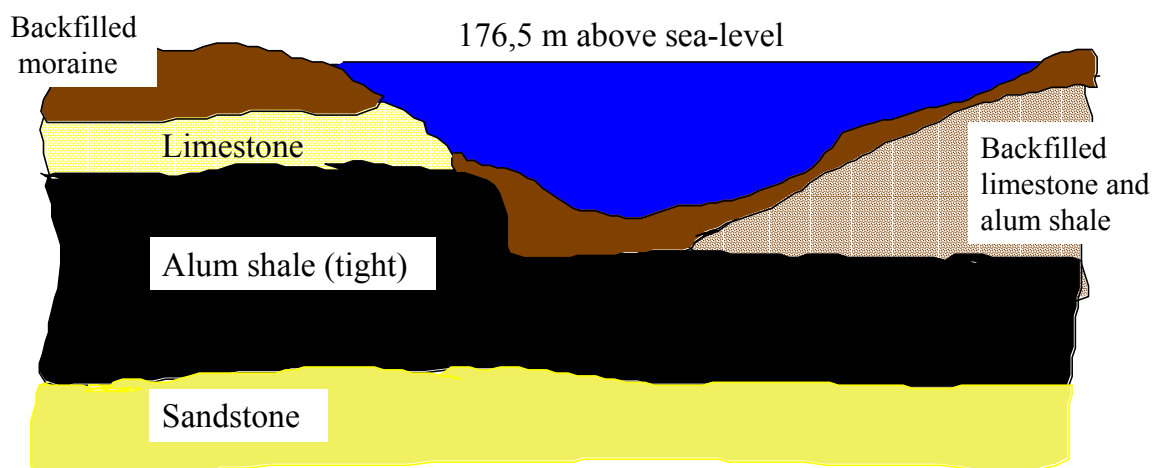


Figure 2 Profile of Lake Tranebärssjön and its surroundings.

Table 1 Mineralogical description of Lake Tranebärssjön.

Thickness (m)	Name	Remark	Composition
6	Moraine	Original & Backfill	Quartz, Kaolinite etc.
5	Limestone	Original	Calcite
~15	Limestone & alum shale	Backfill	Calcite (50%) and Alum shale (50%)
25	Alum shale	Original	Quartz (20%), illite (30%), K-feldspar (10%), chlorite (5%), goethite (13%), organics (22%)
45	Sandstone	Original	Quartz

3.1.3 Pedology

The soil layer consists of moraine with high content of clay minerals, the depth of the layer is about 5–8 metres (Hörnsten *et al* 1977).

3.1.4 Meteorology

The meteorological data were supplied by the SMHI weather station of Skövde (see Table 2). The evapotranspiration is about 545 mm year⁻¹.

Table 2 Meteorological data for Skövde.

Month	Normal temperature 1961-1990 °C	Rainfall 1961-1990 mm/month
January	-2.8	48
February	-2.9	33
March	0.0	36
April	4.6	39
May	10.6	49
June	15.0	49
July	16.2	63
August	15.2	68
September	12.1	67
October	7.1	70
November	2.2	70
December	-2.3	54
Mean	6.3	54

3.1.5 Hydrology

The inflow of water to the lake is dominated by groundwater arising from the moraine and limestone aquifers. The total volume of the water mass today is about 1 million cubic metres. In the western part of the lake the water depth is about 4 metres while it is about 15 metres (the maximum depth) in the eastern part. The western part consists to a large extent of wetland with plane slopes. The eastern part, where the outlet is situated, has more abruptly sloping borders. The average flow at the outlet was 28 litre per second during 1997 and 48 litre per second during 1998. It is relatively constant during the year except for the snow melting period. Lake Tranebärssjön discharges to the Pösan stream (see Figure 1).

Based on the analytical values, the water body in the lake is stratified and can be divided into two layers, divided by a thermocline at the depth of 8-10 metres. In the upper layer, oxidising conditions prevail and sulphates and carbonates of calcium and sodium, with a slightly basic pH dominate the layer. The lower layer, showing reducing conditions and almost oxygen-free conditions at the bottom, has a similar chemical composition, with an increased amount of Ca²⁺, SO₄²⁻ and CO₃²⁻ (and also manganese and nickel) and the pH value is almost neutral.

Twice a year, during spring and autumn the difference in temperature between the two layers are equalized. The two water masses are then mixed fast and drastically, which give rise to an increased colloid content in the surface water. The colloids probably consist of iron oxides hydroxides. This process stains the lake red. The situation is stable only for some days, then the original state of the lake system recovers. This is further discussed in Section 3.2.

3.1.6 Population

Demography

The area is sparsely populated, with some detached farms in the near vicinity. Two small villages, Stenstorp (southeast of Ranstad) and Skultorp (northeast of the tailing site), are situated within five kilometres of Ranstad. Stenstorp is inhabited by about 2000 people and Skultorp by about 3500. Forest and pastureland mainly cover the area around the Ranstad site.

Habits

In the Ranstad area the critical group concerned consists of farmers living in the neighbourhood. They are assumed to consume locally produced meat and drink locally produced milk. The fish is captured in the Pösan stream. The drinking water is taken from the limestone aquifer. Critical group characteristics are summarised in table 3. The data considered are typical for the Swedish population.

Table 3 Critical group consumption.

Foodstuff	Consumption rate
Water m ³ /year	0.73
Fish kg/year	70
Milk m ³ /year	0.33
Meat kg/year	95

Consumption of locally produced crops are not considered for the critical group, since the arable land around the Ranstad site is mainly used for grazing of cattle or for growing crops used as fodder for animals.

Activities

One part of the lake is a wetland and is inhabited by many species of birds. Ornithologists frequently visit the area. Canoeing and other recreational activities take place on the shores of the lake. Swimming is not very common, but there is a beach located on the south shore of the lake. Fish species (salmonoid species) have been introduced in the lake, so fishing is possible, although not in large amounts since the fish population is quite small.

3.2 Physico-chemical characteristics

3.2.1 The water column

In order to make a dose assessment model for Lake Tranebärssjön, the lake is divided into a compartment system, see Section 5.1. The compartment structure includes several sections with an aquatic phase. There have been extensive analytical campaigns since 1965, with enforced activities after 1992. Most of their results are reported in Sundblad *et al.* (1996), which also contains the data

sets related to the Lake Tranebärssjön (see also Stiglund (1999)). These data are based on measurements of water samples taken from the lake at depths of 0.5 and 14 metres. River water samples from four points along the River Pösan, before and after the confluence with the outlet of the lake, were also collected and analysed. In addition, ten groundwater-monitoring pipes are in operation. The analyses were performed independently in parallel by two laboratories. The monitoring campaign is continued until present, results from 1996 were available as spreadsheet files. Available primary (measured) data sets include:

- time series and annual averages for pH, temperature, conductivity and a number of anion and cation concentrations for the two lake water sampling points and the groundwater pipes;
- metal, sulphur and silica concentrations for pore water from the sediment of the lake;
- precipitation and temperature trends (1995 and long-time annual mean);
- annual trends in water discharge at various stations (surface, lake and groundwater);
- annual trends in sulphate, iron, nickel, cadmium, aluminium, manganese, magnesium, calcium, and uranium contents at various stations (surface, lake and groundwater).

Based on these data sets, other values were computed, such as the amount of transported metals per year. The analytical values relevant for the previously described aquatic compartments were taken from this data pool, supplemented by some additional material from Studsvik. Furthermore, in August 1996, new samples were taken by Studsvik as part of the RESTRAT project to close gaps in the previous investigations, and to verify the accuracy of those measurements (see Tables 4 and 5). The pH was determined *in-situ*, then the new samples were shipped to and analysed at the FZ Rossendorf. All results, together with the results from the previous sampling campaigns, were checked for consistency. These values exhibit several general features:

- lack of data for several important anions in case of the pore water samples;
- unusually high values for iron and manganese, exceeding the saturation limit for many minerals of these elements;
- missing details about the sampling, conditioning, filtering and analytical procedures applied in the analysis from the Swedish contractors of Studsvik;
- ionic strengths higher than usually found in fresh water lakes, in a range between 0.03 and 0.05;
- well-buffered pH values in the neutral range between 6.75 and 7.87.

To overcome at least some of the limitations and inconsistencies of the primary analytical data set, chemical speciation modelling with EQ3/6 (Wolery, 1995) was applied to each of the four aquatic system compartments.

In Tables 4 and 5, the columns labelled **Ua** and **La** refer to unfiltered samples collected in August 1996 in Ranstad and sent to FZ Rossendorf for analyses. The columns **Ub** and **Lb** are samples from the same campaign, but they were analysed after filtering, done in three steps for 450, 100, and 15 nm particle size. The columns **Uc**, **Ud**, **Lc**, and **Ld** are values from a sampling campaign in February 1996, analysed by Swedish contractors of Studsvik Eco & Safety AB. Columns labelled with **MW** give the molecular weight of the atoms or ionic units specified in the first column, “n.d.” stands for “not determined”. Upper limit values denote, that the analytical value was below the respective

detection limit. The methods given in the last column are: IC = ion chromatography, ICP-MS = inductively coupled plasma - mass spectrometry, F-AAS = flame - atom adsorption spectrometry, they all apply to the samples analysed by FZ Rossendorf.

Table 4 Analysis of water from the upper layer of Lake Tranebärssjön, at a depth of 0.5 meters.

Sample:	Ua	Ub	Uc	Ud	MW	Ua	Ub	Uc	Ud	Method
Unit:	µg / L	µg / L	µg / L	µg / L	g / mol	mol / L	mol / L	mol / L	mol / L	
F-	n.d.	n.d.	546		18.9984			2.874E-05		
PO ₄ ³⁻	< 2000	< 2000	36		94.971	< 2.1E-05	< 2.1E-05	3.791E-07		IC
NO ₃ ⁻	8000	6500	1900		62.005	1.290E-04	1.048E-04	3.064E-05		IC
NO ₂ ⁻	< 1000	< 1000	10		46.0055	< 2.2E-05	< 2.2E-05	2.174E-07		IC
SO ₄ ²⁻	7.5E5	7.5E5	6.64E5	8.39E5	96.058	7.808E-03	7.808E-03	6.912E-03	8.734E-03	IC
HCO ₃ ⁻	2.0E5	1.65E5			61.017	3.278E-03	2.704E-03			IC
Cl ⁻	12800	13600	11700		35.453	3.610E-04	3.836E-04	3.300E-04		IC
Si	1920	2600		2870	28.086	6.836E-05	9.257E-05		1.022E-04	ICP-MS
K ⁺	5200	6000	4810	5500	39.098	1.330E-04	1.535E-04	1.230E-04	1.407E-04	F-AAS
Na ⁺	7720	7370	7370	8050	22.99	3.358E-04	3.206E-04	3.206E-04	3.502E-04	F-AAS
Ca ²⁺	323000	321000	353000	392000	40.08	8.059E-03	8.009E-03	8.807E-03	9.780E-03	ICP-MS
Mg ²⁺	18300	17700	21000	22200	24.305	7.529E-04	7.282E-04	8.640E-04	9.134E-04	F-AAS
Fe	< 100	< 50		1080	55.847	< 1.8E-06	< 9.0E-07		1.934E-05	F-AAS
Al ³⁺	< 1	n.d.	20	5.1	26.9815	< 3.7E-08		7.412E-07	1.890E-07	ICP-MS
Zn ²⁺	n.d.	52.3		4.6	65.38		7.999E-07		7.036E-08	ICP-MS
U	126	130		141	238.0	5.293E-07	5.461E-07		5.924E-07	ICP-MS
Pb ²⁺	n.d.	n.d.		0.9	207.2				4.344E-09	ICP-MS
Ni ²⁺	14.9	34.4		75	58.7	2.538E-07	5.860E-07		1.278E-06	ICP-MS
Mn ²⁺	56.5	1.6		1481	54.938	1.028E-06	2.912E-08		2.696E-05	ICP-MS
Cd ²⁺	n.d.	n.d.		0.4	112.41				3.558E-09	ICP-MS
As	< 1	n.d.		3.4	74.922	< 1.3E-08			4.538E-08	ICP-MS
Th	0.3	0.7			232.0381	1.293E-09	3.017E-09			ICP-MS
pH	7.82	8.11	7.87							

Suspended material was 2.9 mg L⁻¹.

Table 5 Analysis of water from the lower layer of Lake Tranebärssjön, at a depth of 14 meters.

Sample: Unit:	La µg / L	Lb µg / L	Lc µg / L	Ld µg / L	MW g / mol	La mol / L	Lb mol / L	Lc mol / L	Ld mol / L	Method
F-	n.d.	n.d.	607		18.9984			3.195E-05		
PO ₄ ³⁻	< 2000	< 2000	42		94.971	< 2.1E-05	< 2.1E-05	4.422E-07		IC
NO ₃ ⁻	2800	4400			62.005	4.516E-05	7.096E-05			IC
NO ₂ ⁻	< 1000	< 1000	6		46.0055	< 2.2E-05	< 2.2E-05	1.304E-07		IC
NH ₄ ⁺	n.d.	n.d.	133		18.0383			7.373E-06		
SO ₄ ²⁻	1060000	1070000	1067000	1306000	96.058	1.103E-02	1.114E-02	1.111E-02	1.360E-02	IC
HCO ₃ ⁻	375000	260000			61.017	6.146E-03	4.261E-03			IC
Cl-	12600	13500	12700		35.453	3.554E-04	3.808E-04	3.582E-04		IC
Si	3650	4700		5600	28.086	1.300E-04	1.673E-04		1.994E-04	ICP-MS
K ⁺	4640	6600	6520	5620	39.098	1.187E-04	1.688E-04	1.668E-04	1.437E-04	F-AAS
Na ⁺	7620	8020	10000	8840	22.99	3.314E-04	3.488E-04	4.350E-04	3.845E-04	F-AAS
Ca ²⁺	506000	494000	597000	608000	40.08	1.262E-02	1.233E-02	1.490E-02	1.517E-02	ICP-MS
Mg ²⁺	23400	24400	37000	34600	24.305	9.628E-04	1.004E-07	1.522E-03	1.424E-03	F-AAS
Fe	140	< 50		40200	55.847	2.507E-06	< 9.0E-07		7.198E-04	F-AAS
Al ³⁺	n.d.	n.d.	20	6.7	26.9815			7.412E-07	2.483E-07	ICP-MS
Zn ²⁺	n.d.	3.4		20.6	65.38		5.200E-08		3.151E-07	ICP-MS
U	249	252		241	238.03	1.046E-06	1.059E-06		1.012E-06	ICP-MS
Pb ²⁺	< 1	n.d.		1.4	207.2	< 4.8E-09			6.757E-09	ICP-MS
Ni ²⁺	121	136		242	58.7	2.061E-06	2.317E-06		4.123E-06	ICP-MS
Mn ²⁺	5380	3690		9734	54.938	9.793E-05	6.717E-05		1.772E-04	ICP-MS
Cd ²⁺	n.d.	n.d.		1	112.41				8.896E-09	ICP-MS
As	< 1	n.d.		6.6	74.922	< 1.3E-08			8.809E-08	ICP-MS
Th	0.3	0.8			232.0361	1.293E-09	3.448E-09			ICP-MS
pH	7.23	7.76	6.75							

Suspended material was 59 mg L⁻¹.

In the upper layer of the lake, the content of oxygen is large enough to transfer most of the element into their higher valence states. However, manganese will remain in the divalent state, Mn²⁺. The measured oxygen content corresponds to a redox potential of +780 mV. The charge imbalance is small and well within the analytical uncertainty for the major components sulphate or calcium. The water is slightly oversaturated with respect to the mineral calcite. In general, the available data set for this compartment is the one with the highest reliability and internal consistency. The only problem here is the inherent large fluctuations of the temperature, moving between a winter minimum of 0.3°C and a summer maximum of 22.5°C.

The lower layer of the lake exhibits a more complex chemistry and strongly reducing properties. Reasons may be the strong stratification of the lake water. The stratification severely limits oxygen transport from the lake surface to its lower levels, combined with various oxygen-consuming processes such as the oxidation of sulphide components of the alum shale or organic material sinking down to the bottom of the lake. At irregular intervals the chemistry of the lake shifts dramatically, with enormous amounts of iron (and probably manganese) particles moving upwards to the top layers of the lake, partly settling on its shores. All these phenomena make it complicated to assign a unique set of chemical parameter to this compartment. The momentary best approximations assume a redox potential, Eh, of between -190 mV, the transition zone from sulphide to sulphate, and +320 mV, the

transition zone from Fe^{2+} to Fe^{3+} . Eh will not be lower than -190 mV because most of the sulphides in the alum shale at the ground were already at least partly exposed to oxygen before the lake was filled, so oxidation could start. On the other hand the immense amount of iron dissolved in the waters can only be stable in the divalent state, Fe(III) would precipitate as hydrous ferric oxides with a slow transformation to more stable products such as hematite or goethite. Unfortunately, the transition from U^{4+} to UO_2^{2+} at around -90 mV falls right into this proposed Eh range. Thus, two scenarios must be considered for the contaminant uranium for all modelling, one at a Eh of around -150 mV, and another at a Eh of around $+150$ mV.

Compared to the lake water data sets, the pore water analysis of both the backfill and the alum shale lacks data for all anions other than sulphate (see Table 6). Another gap is the pH, which was not determined. Nevertheless the situation is not impossible. A comparison of the ionic water composition shows close relationships between the upper lake layer and the pore water from the backfill. The same is true for the lower lake layer and the pore water from the alum shale. This leads to the approximation that, besides sulphate, carbonate is the only other important anion. This is supported by the presence of calcite as a major component of both the backfill and the alum shale. Finally, due to the composition of the alum shale, a reducing environment is to be expected for the water in there. The water in the backfill will be considerably more oxidised. Based on these assumptions and the available analytical information, modelling with EQ3/6 gave reasonable values for the total carbonate concentrations in the backfill and alum shale pore waters of $4.6 \cdot 10^{-3}$ mol L⁻¹ (total ionic strength of 0.0325 mol L⁻¹) and $7.4 \cdot 10^{-3}$ mol L⁻¹ (total ionic strength of 0.052 mol L⁻¹), respectively. Moreover, the pH was computed from the charge balance to be 7.04 and 6.83, respectively; again values that fall well into the expected range. Finally, the analytical results for silica gave, in the speciation modelling, saturation indices of quartz very close to one, so this mineral may be in equilibrium with the silica content of the pore waters. The temperature of the pore water should be rather constant throughout the year and similar to the value measured for the lower layer of the lake, thus it was set to 7.0°C.

Table 6 Analysis of pore water from the backfill and the alum shale in Lake Tranebärssjön.

Compartment:	Backfill Mean	STD	Alum shale Mean	MW	Backfill Mean	STD	Alum shale Mean
Component:	µg / L	µg / L	µg / L	g / mol	mol / L	mol / L	mol / L
S	486250	2061.55	279000	32.06	1.517E-02	6.430E-05	8.702E-03
Si	7597.50	1854.73	4700	28.086	2.705E-04	6.604E-05	1.673E-04
K ⁺	10375	206.16	6310	39.098	2.654E-04	5.273E-06	1.614E-04
Na ⁺	12550	525.99	8000	22.99	5.459E-04	2.288E-05	3.480E-04
Ca ²⁺	583000	19849.43	383000	40.08	1.455E-02	4.952E-04	9.556E-03
Mg ²⁺	35700	1726.27	18100	24.305	1.469E-03	7.103E-05	7.447E-04
Fe	78925	29176.40	24	55.847	1.413E-03	5.224E-04	4.297E-07
Al ³⁺	8.75	1.44	6.51	26.9815	3.244E-07	5.344E-08	2.413E-07
Zn ²⁺	32.50	10.26	18.1	65.38	4.971E-07	1.569E-07	2.768E-07
U	275.75	240.91	167	238.03	1.158E-06	1.012E-06	7.016E-07
Pb ²⁺	0.48	0.17		207.2	2.309E-09	8.261E-10	
Ni ²⁺	55.50	19.87	14	58.7	9.455E-07	3.386E-07	2.385E-07
Mn ²⁺	8527.50	426.88	15.4	54.938	1.552E-04	7.770E-06	2.803E-07
Cd ²⁺	0.42	0.22	0.1	112.41	3.747E-09	1.921E-09	8.896E-10
As	16.10	6.78	0.301	74.922	2.148E-07	9.045E-08	4.018E-09
Co ²⁺	25.40	10.85		58.9332	4.310E-07	1.841E-07	
Cu ²⁺	3.93	2.04	1.86	63.546	6.181E-08	3.210E-08	2.927E-08
Sr ²⁺	1297.50	55.60	737	87.62	1.481E-05	6.346E-0	8.411E-0
Ba ²⁺	57.70	6.02	21.8	137.33	4.202E-07	4.387E-08	1.587E-07

The data source is a sampling campaign from February 1996, analysed by Swedish contractors of Studsvik Eco & Safety AB. The data for the backfill being the average of four different profile depths in the middle part of the lake, whereas the alum shale data are single point values from the deepest part of the lake.

The above-discussed modelling results were combined with the experimental data to yield a set of selected best values. They are listed in Table 7. Superscript¹⁾ identifies values set as approximations based on general chemical considerations as explained above, whereas ²⁾ mark values set according to the modelling results when there is no experimental information available at the moment. Average values (MEAN) and standard deviations (STD) were calculated assigning equal weights to all listed values. There are no values given for cadmium, arsenic, thorium, mercury, chromium, copper, and cobalt, because the available data is either too widely scattered, or there are hardly any measurements of those values above their respective detection limits.

Table 7: Selected best set of analytical values for the Lake Tranebärssjön compartments, all concentrations are in mol/L.

Compartment:	Lake: Top Layer		Lake: Bottom Layer		Backfill	Alum shale	
Abbreviation:	T		B		F	A	
Component:	Mean	STD	Mean	STD	Mean	Mean	STD
F-	2.874E-05	3.16E-05	3.195E-05	7.84E-06			
PO ₄ ³⁻	3.791E-07	2.42E-07	4.422E-07	2.00E-07			
NO ₃ ⁻	8.816E-05	5.13E-05	5.806E-05	1.82E-05			
NO ₂ ⁻	2.174E-07	1.74E-07	1.304E-07	2.17E-07			
NH ₄ ⁺			7.373E-06	2.88E-06			
SO ₄ ²⁻	7.816E-03	7.44E-04	1.172E-02	1.25E-03	8.702E-03	1.517E-02	6.43E-05
HCO ₃ ⁻	2.991E-03	4.06E-04	5.203E-03	1.33E-03	4.56E-3	7.35E-3	
Cl ⁻	3.582E-04	2.69E-05	3.648E-04	1.39E-05			
Si	8.771E-05	1.74E-05	1.656E-04	3.48E-05	1.673E-04	2.705E-04	6.60E-05
K ⁺	1.375E-04	1.28E-05	1.495E-04	2.35E-05	1.614E-04	2.654E-04	5.27E-06
Na ⁺	3.318E-04	1.42E-05	3.749E-04	4.57E-05	3.480E-04	5.459E-04	2.29E-05
Ca ²⁺	8.664E-03	8.29E-04	1.375E-02	1.49E-03	9.556E-03	1.455E-02	4.95E-04
Mg ²⁺	8.146E-04	8.84E-05	1.228E-03	2.86E-04	7.447E-04	1.469E-03	7.10E-05
Fe	1.934E-05	7.59E-06	3.612E-04	5.07E-04	4.297E-07	1.413E-03	5.22E-04
Al ³⁺	4.651E-07	3.90E-07	4.948E-07	3.49E-07	2.413E-07	3.244E-07	5.34E-08
Zn ²⁺	4.351E-07	5.16E-07	1.835E-07	1.86E-07	2.768E-07	4.971E-07	1.57E-07
U	5.560E-07	3.26E-08	1.039E-06	2.39E-08	7.016E-07	1.158E-06	1.01E-06
Pb ²⁺	4.344E-09	9.65E-10	6.757E-09	3.38E-09			
Sr ²⁺					8.411E-06	1.481E-05	6.35E-07
Ni ²⁺	7.058E-07	5.22E-07	2.834E-06	1.12E-06	2.385E-07	9.455E-07	3.39E-07
Mn ²⁺	9.338E-06	1.53E-05	1.141E-04	5.68E-05	2.803E-07	1.552E-04	7.77E-06
Cd ²⁺	3.558E-09	4.45E-09	8.896E-09	1.69E-08			
As	4.538E-08	5.20E-08	8.809E-08	1.09E-07			
Th	2.155E-09	1.22E-09	2.370E-09	1.52E-09			
pH	7.87	0.14	6.75	0.06	7.04	6.83	
O ₂ (aq)	6.28E-04	1.29E-04	3.75E-06	5.63E-06			
T / °C	8.00	7.60	7.60	0.50	7.0	7.0	

3.2.2 Solid phase

The characterisation of the relevant solid phases is not as satisfactory as compared to the aqueous phase. There are only rather general mineralogical investigations for this site published, mainly considering the alum shale. From the available data and speciation modelling based on them, the following conclusions can be drawn:

- The backfill consists of an ill-defined and varying mixture of alum shale debris, limestone, and moraine constituents (quartz and weathered aluminosilicates such as illite or kaolinite).

- The alum shale composition was taken from Andersson *et al.* (1985), on average it consists of quartz (20%), illite (30%), K-feldspar (10%), chlorite (5%), pyrite (13%), and organics (22%).
- The sediment at the bottom of the lake was analysed only with regard to the various elements, no mineralogical data could be drawn from that.

Analysis is complicated by a considerable content of organic matter of various origins and in different state of degradation. If possible, further investigations should be performed to specify a more detailed the mineralogical background for the site.

3.3 Radiological characteristics

The lake Tranebärssjön is not considered to be a radiological problem even though the Swedish Radiation Protection Agency has decided that radium-226 should be measured four times a year at the outlet of the lake (station 13). During the last three years the radioactivity has not exceeded 10 mBq/l see Table 5.

Table 5 Radium-226 content in lake Tranebärssjön.

Year and month	Lake Tranebärssjön, stn 13 mBq/l
1995-03	3.8
1995-07	1.4
1995-10	4.5
1995-12	<1.5
1996-03	3.3
1996-07	1.5
1996-12	7.7
1997-01	7.7
1997-05	6.3
1997-10	2.4
1997-11	<5.8

3.3.1 Source term

The contaminants are the same as for the Ranstad tailing site (Sundblad *et al.*, 1996), namely uranium-238, both as a radionuclide as well as a toxic element, and also manganese and nickel. Source terms for the contaminants in Lake Tranebärssjön are given in Table 8.

The contaminants in Lake Tranebärssjön arise from three different sources.

- 1 Bank material surrounding the lake, which is the material which was removed before the mining in the open pit could start. The material consisted of alum shale and moraine.
- 2 Backfill material in the lake, which is the same material as above.
- 3 Alum shale. Weathering processes have occurred within the shale during the 30 years when the open pit was being pumped dry.

Table 8 Source term of Lake Tranebärssjön.

	Bank kg/kg	Backfill kg/kg	Alum shale kg/kg
Uranium, U	6.0E-5	6.0E-5	3.0E-4
Nickel, Ni	4.0E-4	4.0E-5	2.0E-4
Manganese, Mn	5.0E-4	5.0E-5	2.5E-4

3.3.2 Discharge/effluents

The mean values of the flow rates out of the lake, as well as in the recipient River Pösan, are shown in Table 3. The outflow from the lake has been relatively stable during the years after the filling of the lake. Before the lake was filled up, the water was pumped away during 30 years in order to keep the open pit dry. The pumping affected an area of about 1 km². It took about 1.5 years to fill the lake with water. The amount of water being pumped away was about half the amount measured today at the outlet. Discharges from the Lake Tranebärssjön and flow rate in the Pösan stream are given in Table 9.

Table 9 Flowrates in m³/year.

Streams/Rivers	Flowrate m ³ /year (1997)
Outflow from the lake	8.83 E5
River Pösan	9.3 E6

3.3.3 Contamination

Since the lake was formed, and filled with water, the transport of different contaminants has been measured, both within the lake (at the surface and in the bottom water) and at the outlet from the lake.

The contamination that occur will take place in the lake itself, the recipient Pösan stream, into which the water from the lake discharges, and in the watercourses further downstream; the Slafsan stream, River Hornborgaån and Lake Hornborgasjön.

The transport of uranium from the outlet of the lake, as well as downstream of the Pösan stream, has increased during 1998, after decreasing during 1996 and 1997. One explanation for this may be that the bank material mentioned above, which later became backfill material after the mining ceased, has weathered and contributes to the high concentrations of uranium and other heavy metals in the lake. The weathering products are now delivered into the lake by the groundwater.

The substances studied is leaching from the banks around the lake into the lake. The backfill in the lake is of the same type of material as in the banks. Furthermore there is a leakage from surrounding alum shale, which was affected by lowering of the groundwater table during the operation of the pit. Today the groundwater has returned to its original level.

4 Restoration options

Since there is a lack of information in order to restore a lake this study has focused on the restoration of the outgoing water from the lake. Two different alternatives have been chosen as realistic for this particular lake:

- to let the water from the lake pass through a sandfilter, or
- pass through a constructed wetland beneath the outlet before the water enters the recipient Pösan stream

Decontamination factors of the two alternatives are shown in Table 10.

Table 10 Values of characteristics of techniques considered.

Restoration option	Df-factor U	Df-factor Ni	Df-factor Mn
A. No remediation	-	-	-
C2 Sandfilter	10(5-100)	10(5-100)	5(2-10)
D3. Wetland	100(10-1000)	100(10-1000)	5(2-100)

5 Radiological impact assessment

The compartment scheme for the lake Tranebärssjön site is based on site information in Stiglund (1999) and shown in Figure 3.

The contaminants are not only radionuclides, i.e. uranium-238, but there are also significant levels of manganese and nickel. So any risk assessment should also take account of classical chemical-toxically effects.

5.1 Compartment scheme

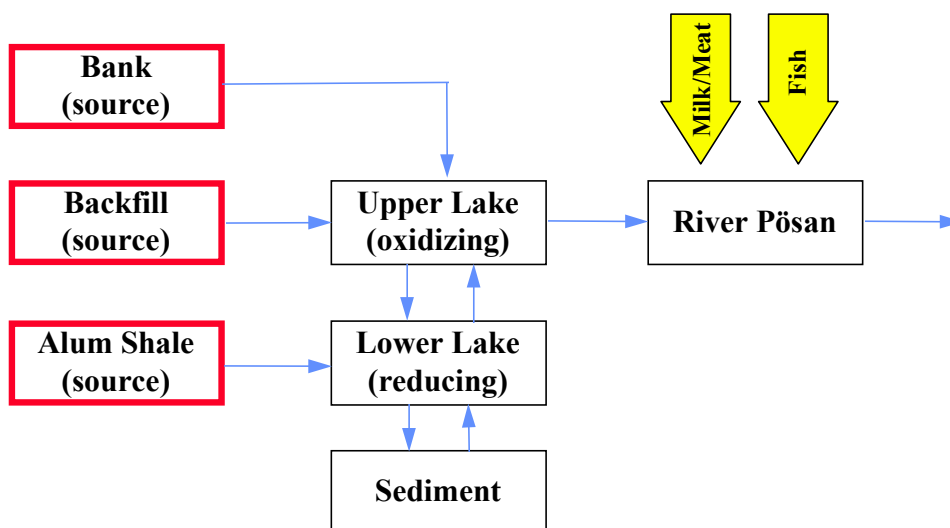


Figure 3 Compartment scheme for Lake Tranebärssjön.

5.2 Critical group and exposure pathways

A critical group, concerned for individual dose, should by definition consist of a real or fictitious group of individuals, which due to their location and living habits obtain the highest exposure.

At the Ranstad tailing site the critical group consists of farmers living in the neighbourhood. They consume locally produced meat and drink milk from cows. The fish is captured in the Pösan stream. The drinking water is taken from the limestone aquifer. Remediation workers are not considered since external exposure are not valid for the actual substances and the inhalation of dust is not valid for the restoration options.

The exposure pathways to the public were calculated, using steady state factors, for the uptake in biota and the further transfer along the food chains. The important exposure pathways for the critical group are indicated in the compartment scheme (Figure 3). The exposure pathways considered are:

- consumption of water;
- consumption of fish;
- consumption of milk and meat from cattle drinking the water.

The committed collective intake/dose is based on production data.

5.3 Doses to public

The doses calculated consist of individual and committed intakes/doses.

5.1.1 Individual doses to critical group

The results of the dose calculations/intakes for the individual doses/intakes during Year 1 are shown in Table 11.

Table 11 Individual dose (Sv/a) and intake (kg/a) at Lake Tranebärssjön during Year 1.

Exposure pathway	Manganese kg/a	Nickel kg/a	Uranium kg/a	U-238 Sv/a
Exposure	Mean	Mean	Mean	
Fish	1.7E-3	5.0E-5	2.2E-5	1.2E-5
Milk	9.1E-6	2.0E-5	2.1E-6	1.2E-6
Meat	1.8E-5	9.0E-7	2.9E-7	1.7E-7
Total	1.7E-3	7.0E-5	2.4E-5	1.4E-5

The contribution to the total dose/intake from different pathways is graphically depicted in Figure 4.

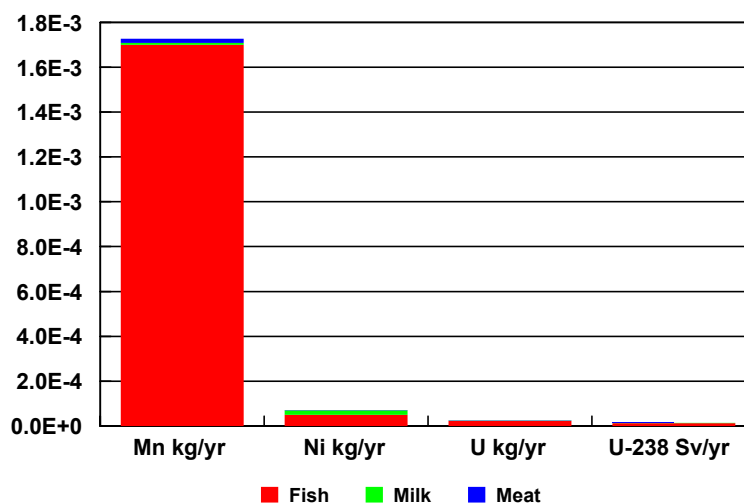


Figure 4 The contribution to the total dose (Sv/a) and intake (kg/a) from different pathways.

The intake of fish from the Pösan river is the dominant exposure pathway for manganese and uranium. For nickel the intakes of milk and fish are in the same order caused by the high value for the distribution factor to milk compared to the other nuclides.

5.1.2 Collective doses to public over 100 years and 500 years.

The effects of different restoration techniques are given by different decontamination factors (Df-factors). The source term is divided by the Df-factor in order to reduce it in proportion to the effect of the restoration technique. The base case, as the situation is without any restoration is called Option A. Option C2 means that the outflowing water from the lake is filtrated through sand and Option D3 means that the water is leaving the lake and flows into a wetland.

Table 10 shows the restoration options considered for the Lake Tranebärssjön.

The results of the dose/intake calculations for the collective doses/intakes at 100 years and 500 years for the different restoration options are shown in Table 12.

Table 12 Collective intake, (mankg), and doses, (manSv), at the Lake Tranebärssjön.

Case	Year	Manganese mankg Mean	Nickel mankg Mean	Uranium mankg Mean	Uranium manSv Mean
A	100	1.6E+00	1.4E+00	1.2E-01	6.9E-02
	500	4.4E+00	5.9E+00	4.8E-01	2.7E-01
C2	100	7.2E-01	2.3E-01	1.4E-02	8.1E-03
	500	2.4E+00	1.0E+00	5.7E-02	3.3E-02
D3	100	5.9E-01	1.1E-01	3.5E-03	2.0E-03
	500	2.2E+00	5.5E-01	1.6E-02	8.9E-03

As can be seen in Table 12 the intake of manganese and nickel is dominating the intake for all restoration options.

In Figure 5 the collective dose commitment from uranium-238 for the public is graphically depicted for the different restoration options at 100 and 500 years.

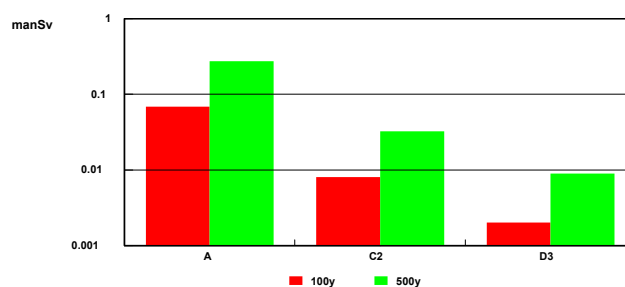


Figure 5 The total collective dose of U-238 for the different restoration options at 100 and 500 years.

As can be seen in Figure 5 the reduction of total committed dose to the public is quite large for the different restoration options. The calculations show that the Option C2, sand filtration, reduces the dose with one order of magnitude and that the Option D3, wetland filtration, reduces the dose with a factor of thirty.

6 Ranking of restoration options

Decisions on clean-up in long-lasting exposure situations may well go far beyond purely radiological protection considerations. Satisfying the justification principle requires that the overall effect of the actions involved should do more good than harm, taking account of relevant radiological and non-radiological factors. Most decisions require multiple criteria to be taken into account. The field of multiple criteria analysis offers a number of approaches which take explicit account of multiple criteria in providing structure and support to the decision-making process. In case of restoration of contaminated sites there are several criteria or attributes that need to be considered when choosing an ‘optimum’ restoration strategy. When the performance and costs of all the protection options have been assessed, a comparison is needed to define the *optimum* protection option. When the optimum is not self evident, the comparison can be carried using a quantitative decision-aiding technique. Of the different techniques available, cost-benefit analysis has been used to evaluate if the remediation options are justified on economical grounds and multi-attribute utility (MAU) analysis for ranking different remediation options. Moreover, annual individual doses to critical groups before implementation of remedial measures have been assessed for comparison with the clean-up criteria recommended by IAEA.

6.1 Method – Evaluation of remediation options (MAU analysis)

6.1.1 Attributes

The attributes that have been considered in this study include:

- *Health attributes*
 - collective doses to population
 - doses to remediation workers
 - non-radiological health factors
- *Economic attributes*
 - costs of remedial actions (incl. costs of labour and monitoring)
 - costs of monitoring of remedial options
 - costs of disposal of generated waste (in broad categories)
 - loss/gain of taxes due to loss/gain of income
- *Social attributes*
 - reassurance of the public
 - discomfort, disturbance and anxiety from the remedial action
 - loss/gain of income

The restoration options included for the Lake Tranebärssjön site have been identified in Chapter 4. The economical and radiological data for quantifying the various attributes for each of those options are shown in Table 13.

6.1.2 Utilities

Utility functions for the attributes *monetary costs* and *radiation doses* have been calculated from the figures in Table 13 on monetary cost components and residual collective doses after remediation. Linear (risk neutral) utility functions have been used.

Table 13. Remediation costs and collective doses to population and workers for different restoration strategies at the Lake Tranebärssjön site.

Restoration strategy	Collective dose to population [man Sv]		Collective intake of metals to population [man·kg]		Monetary costs of restoration [kECU]	Fraction of activity left on-site
	100 y	500 y	100 y	500 y		
			Manganese/Nickel			
A	0.069	0.27	1.6/1.4	4.4/5.9	0	1
C2	0.0081	0.033	0.72/0.23	2.4/1.0	400,000	0.1
D3	0.002	0.0089	0.59/0.11	2.2/0.55	700,000	0.03

Utility functions for monetary costs

Utility functions have been determined for remediation costs (including labour costs) and monitoring costs:

$$u_{\text{remedia}}(x) = 100 \cdot \left(1 - \frac{x}{700,000} \right) \quad \text{for } 0 \leq x \leq 700,000 \text{ kECU}$$

Utility functions for health factors

The following utility functions for the radiological health components have been determined for the exposed population and workers implementing the remedial actions. Both radiological and non-radiological health factors are considered for the Lake Tranebärssjön sites as the heavy metals nickel and manganese would expose the population through contaminated foodstuffs.

$$u_{\text{dose, pop,100}}(x) = 100 \cdot \left(1 + \frac{0.002 - x}{0.069 - 0.002} \right) \quad \text{for } 0.002 \leq x \leq 0.069 \text{ man Sv}$$

$$u_{\text{dose, pop,500}}(x) = 100 \cdot \left(1 + \frac{0.0089 - x}{0.27 - 0.0089} \right) \quad \text{for } 0.0089 \leq x \leq 0.27 \text{ man Sv}$$

$$u_{\text{non-rad,100}}(x) = 100 \cdot \left(1 + \frac{0.70 - x}{3.0 - 0.70} \right) \quad \text{for } 0.70 \leq x \leq 3.0 \text{ man kg nickel + manganese}$$

$$u_{\text{non-rad,500}}(x) = 100 \cdot \left(1 + \frac{2.75 - x}{10.3 - 2.75} \right) \quad \text{for } 2.75 \leq x \leq 10.3 \text{ man kg nickel + manganese}$$

Utility functions for social factors

The utility function u_{reas} for reassurance would be linked to both the residual dose and the fraction of activity remaining on the site after the remedial measure has been implemented. However, the residual dose and remaining activity are not necessarily correlated. A remedial measure that has left all the activity on site in a contained form (capping, surface barriers etc.) might give a substantial dose reduction and thus a low value of the residual doses. Detailed information on how social factors like reassurance are linked with individual doses and activity concentration on site is not available. Therefore, utility functions for 100 years and 500 years integration time have been proposed which gives a low value only when both sub-utilities have low values:

$$u_{reas,100}(x, y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 + \frac{0.002 - x}{0.069 - 0.002} \right)_{dose} + \frac{1}{2} \cdot \left(1 + \frac{0.03 - y}{1.0 - 0.03} \right)_{activity} \right)$$

for $0.002 \leq x \leq 0.069$ man Sv and $0.03 \leq y \leq 1$

$$u_{reas,500}(x, y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 + \frac{0.0089 - x}{0.27 - 0.0089} \right)_{dose} + \frac{1}{2} \cdot \left(1 + \frac{0.03 - y}{1.0 - 0.03} \right)_{activity} \right)$$

for $0.0089 \leq x \leq 0.27$ man Sv and $0.03 \leq y \leq 1$

where y is the fraction of activity remaining on site after the remedial measures has been implemented. The value of the utility function u_{reas} is 100 for a residual dose of 0.002 (0.0089) man Sv and a remaining fraction of the initial activity of 0.03 (best strategy) and 0 for a residual dose of 0.069 (0.27) man Sv and a remaining activity fraction of 1.0 (worst strategy).

The utility function u_{distur} for disturbance has been related to the remediation costs:

$$u_{distur}(x) = 100 \cdot \left(1 - \frac{x}{700,000} \right) \text{ for } 0 \leq x \leq 700,000 \text{ kECU}$$

6.1.3 Weighting factors

Weighting factors are scaling factors that reflect the relative importance of each of the attributes. The weighting factors assigned to the different attributes at the same hierarchy level can be either the ratio of value ranges, R , of the attributes or by assigning values to the ratio of the weighting factors at that level. Both methods have been used here.

Weighting factors for major attributes

The major weighting factors considered in this study include those for monetary costs, health and social factors. The sum of these weighting factors should respect the following conditions:

$$w_{health} + w_{economic} + w_{social} = 1$$

The assessment of the weighting factors is discussed in Hedemann Jensen (1999) where conversion/scaling constants between weighting factors has been expressed as:

$$C_1 = \frac{w_{economic}}{w_{health}} \cong \frac{w_{economic}}{w_{dose, pop}} = \frac{R_{economic}}{\alpha \cdot R_{dose, pop}}$$

$$C_2 = \frac{w_{social}}{w_{health}} \approx \frac{r_{psy}}{r_{rad}}$$

The value C_1 can be determined for a 100 and 500 years integration time for the collective dose from the values given in Table 13. The value of C_2 is more difficult to assess but a value of 0.2 - 0.3 has been argued for in Hedemann Jensen (1999). From these values of C_1 and C_2 the weighting factors for health, economical and social factors have been calculated as shown in Table 14.

Weighting factors for health sub-attributes

The weighting factors for health sub-attributes include those of radiation induced stochastic health effects to the affected population and workers and non-radiation induced stochastic health effects to the affected population. The sum of these weighting factors should respect the following conditions:

$$w_{dose,pop} + w_{non-rad,pop} = 1$$

The conversion/scaling constant, C , for the health sub-attributes can according to Hedemann Jensen (1999) be expressed as:

$$w_{dose,pop} = C \cdot R_{dose,pop} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$

$$w_{non-rad,pop} = C \cdot R_{non-rad,pop} \cdot l \cdot r_{non-rad}$$

The conversion/scaling constant, C , can be determined from the dose ranges, R , given in Table 13. The value of C is given by (see Hedemann Jensen (1999)):

$$C \cong \frac{1}{R_{dose,pop} + R_{non-rad,pop}}$$

From the calculated values of C_{100} and C_{500} (for 100 and 500 years integrating time for the collective dose to the population) the weighting factors for the collective population doses have been calculated as shown in Table 14. The risk factor for ingestion of manganese and nickel, $r_{non-rad}$, is at present unknown and the weighting factor for exposure to manganese and nickel, $w_{non-rad,pop}$, has therefore not been determined.

Weighting factors for economic sub-attributes

The weighting factors for economic sub-attributes include those for cost of remediation and costs of waste disposal. The sum of these weighting factors should respect the following conditions:

$$w_{remedia} + w_{waste} = 1$$

The conversion/scaling constant, C , for the economical sub-attributes can according to Hedemann Jensen (1999) be expressed as:

$$w_{remedia} = C \cdot R_{remedia}$$

$$w_{waste} = C \cdot R_{waste}$$

The conversion/scaling constant, C , for the economic sub-attributes can be determined from the cost ranges in Table 13 and the weighting factors for remediation costs, waste disposal costs and monitoring costs have been calculated as shown in Table 14.

Weighting factors for social sub-attributes

The weighting factors include those for reassurance and disturbance. The sum of these weighting factors should respect the following conditions:

$$w_{distur} + w_{reas} = 1$$

The conversion/scaling constants for the social sub-attributes can according to Hedemann Jensen (1999) be expressed as:

$$C_1 = \frac{w_{reas}}{w_{distur}}$$

In Hedemann Jensen (1999) it is argued that $w_{reas} > w_{distur}$ and that $C_1 \approx 5 - 7$. From these values the weighting factors for disturbance and reassurance have been calculated as shown in Table 14.

Table 14 Weighting factors for attributes and sub-attributes applied in the optimisation of remediation of the Lake Tranebärssjön site. The values in the left of the double columns are for an integration time of 100 years and in the right column for an integration time of 500 years.

Health factors			Economic factors		Social factors	
$9.57 \cdot 10^{-6}$	$3.73 \cdot 10^{\text{SYMBOL-5}}$		1.0	1.0	$2.39 \cdot 10^{\text{SYMBOL-6}}$	$9.32 \cdot 10^{\text{SYMBOL-6}}$
Dose population	1	1	Remediation costs	1	Reassurance	0.86
Dose workers	-	-	Waste disposal costs	-	Disturbance	0.14
Non-radiation	-	-	Monitoring costs	-	Loss/gain of income	-
			Loss/gain of taxes	-		

It should be emphasised that value setting of weighting factors is the crucial issue of any optimisation because subjective judgements inevitably will enter the process.

6.2 Results

IAEA has proposed clean-up criteria in terms of individual doses. The individual doses assessed at the Lake Tranebärssjön site are of the order of $15 \mu\text{Sv} \cdot \text{a}^{\text{SYMBOL-1}}$ at the time of decision to introduce remediation (year 1). According to the IAEA criteria, clean up is sometimes needed for an individual dose range of $10 - 100 \mu\text{Sv} \cdot \text{a}^{\text{SYMBOL-1}}$ if a constraint for controlled practices is applied. Without the application of a constraint, IAEA suggests that for individual doses of $10 - 100 \mu\text{Sv} \cdot \text{a}^{\text{SYMBOL-1}}$ clean up would rarely be needed. Based on these recommendations it can therefore be concluded that remediation would probably not be needed for the Lake Tranebärssjön site.

In a cost-benefit approach, the monetary costs, X , of the remediation strategies are compared with the benefit of the collective dose reduction, ΔS . The net benefit, ΔB , is given as:

$$\Delta B = \alpha \cdot \Delta S - (\alpha \cdot S_{\text{work}} + X_{\text{remedia}} + X_{\text{waste}})$$

None of the remedial options are justified on economical grounds alone when only the central estimates of collective dose are used together with a α -value of $100,000 \text{ ECU} \cdot \text{manSv}^{\text{SYMBOL-1}}$. Not even a higher value of α (e.g. $200,000 \text{ ECU} \cdot \text{manSv}^{\text{SYMBOL-1}}$) and more extreme values from the calculated collective dose distribution (e.g. the 95th percentile) would make any of the options economically justified.

In the multi-attribute approach, overall scores, U_i , of the remediation options i has been determined from the weighted sum of utilities for each of the attributes considered:

$$\begin{aligned} U_i &= \sum_{j=1}^3 w_j \cdot u_{ij} \\ &= w_{\text{health}} \cdot (w_{\text{dose,pop}} \cdot u_{\text{dose,pop}} + w_{\text{non-rad}} \cdot u_{\text{non-rad}}) \\ &\quad + w_{\text{economic}} \cdot (w_{\text{waste}} \cdot u_{\text{waste}} + w_{\text{remedia}} \cdot u_{\text{remedia}}) \\ &\quad + w_{\text{social}} \cdot (w_{\text{distur}} \cdot u_{\text{distur}} + w_{\text{reas}} \cdot u_{\text{reas}}) \end{aligned}$$

It has not been possible to determine the risk factors for ingestion of manganese and nickel, and consequently no value for the weighting factor, $w_{non-rad}$, has been determined.

The weighting factors above have all been sampled in a triangular distribution between 0.67 - 1.5 times the most probable value given in Table 14. Similarly, the values of all the utilities, $u(x)$, are determined from the utility functions in which the values of x are sampled in a triangular distribution between 0.67 - 1.5 times the central values of x given in Table 13. Negative correlation between collective doses and remediation costs has been applied with a correlation coefficient of -0.8 . The evaluation of the different strategies has been made with the forecasting and risk analysis program CRYSTAL BALL. Latin Hypercube Sampling technique was used and the number of trials were 10,000. The results for the scores, U_i , for the options A - D3 are shown in Figure 6. The error bars represent the 5% and 95% percentiles of the distributions of U_i .

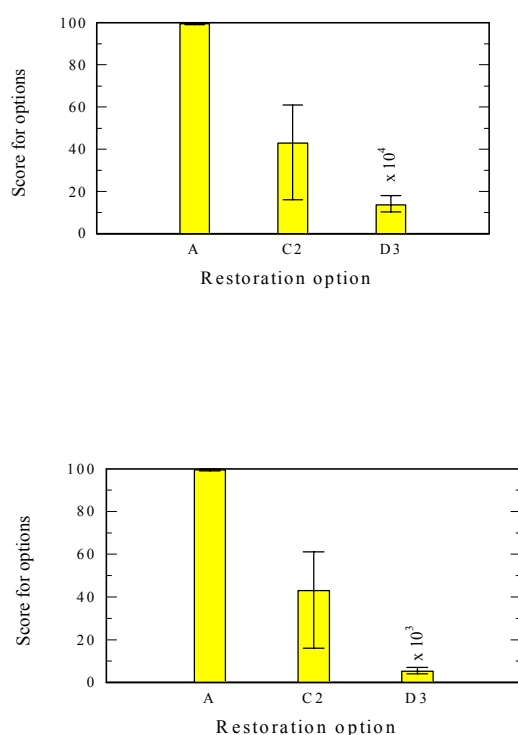


Figure 6 Overall evaluation of scores for different remediation strategies for the Lake Tranebärssjön site. The left picture shows the results for an integration time of 100 years for the collective dose and the right picture for an integration time of 500 years.

As can be seen from Figure 6, option A has the highest score. Options C2 and D3 both have a significantly lower score than that of Option A due to high remediation costs. The 'no remediation' option, Option A, can thus be considered as the optimum solution for the Lake Tranebärssjön site and also the cheapest.

The ranking of the different remedial measures using multi-attribute utility analyses allows the inclusion of factors that are not easy to quantify in monetary terms as is required in cost-benefit analysis. The weighting factors assigned to the different attributes have been determined by use of scaling

factors in terms of weighting factor ratios, and their values were sampled around a most probable value. Notwithstanding this advantage of the multi-attribute method there are difficulties with the determination of weighting factors for the different attributes. Without any terms of reference for the weighting between attributes, value settings by a decision-maker could lead to 'optimised' results that might be useless because of a subjective bias of the decision-maker in the selection of weighting factors. Therefore, the outcome of any multi-attribute analysis, including the present study, should be judged very carefully in the light of the values assigned to the weighting factors before any firm conclusions could be drawn.

7 Conclusions

The objectives of this technical deliverable has been to use the former open pit mine at Ranstad as an example, in the decision-making procedure for remediation, developed by RESTRAT. The procedure involved characterisation of the site with available data, identification of contamination source, identification of critical group and exposure pathways through which man can be exposed to these contaminants and dose assessment for different remediation alternatives. This information was combined with the calculation of economic and social costs in a multi-attribute utility analysis to rank each restoration technique.

The former open pit mine had been transformed into a lake with the aim to make it look as natural as possible. The purpose of the remediation options was to reduce the outflow of contaminants from the artificial lake.

The result shows that the “no remediation option” was the optimum solution for the lake and also the cheapest solution. Options C3 and D3 both have a significantly lower score than Option A. The ranking of the different remedial measures, using multi-attribute utility analyses, allows the inclusion of factors that are not easy to quantify in monetary terms as is required in cost-benefit analysis which give an essential support in the decision procedure. However, the values of the weighting factors used in the present study should be evaluated carefully before any final conclusions should be made.

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