PART 5

FRESHWATER AND FISH

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23. The freshwater environment

23.1 Introduction

Severe radioactive contamination of the freshwater environment could have serious consequences for both drinking water and fish.

Most of the Nordic countries have an abundance of freshwater lakes and rivers. Finland alone has about 56,000 lakes, each with a surface area of 1 hectare or more. Nearly 10% of Finland's surface is covered with lakes and rivers. In Sweden, about 9% of the surface area is freshwater, in Norway about 5%, and in Denmark only about 2%. Freshwater plays a minor role in Iceland, but even there numerous rivers discharge from the volcanic soils to the Ocean.

In Finland and in Sweden, about 50% of the drinking water is purified surface water and the amount of groundwater used is gradually increasing. The percentages of surface water used as drinking water supplies in Norway and Denmark are 87% and 10 % respectively.

23.2 Important Radionuclides

Cs-137 and ^{90}Sr are likely to be the most important radionuclides with respect to long term radioactive contamination of freshwater.

After fallout into natural water systems, radiocaesium becomes quickly attached to suspended particulates which eventually falls to the bottom of the lake or river, whereas radiostrontium tends to remain in the aqueous phase. Therefore, for drinking water ^{90}Sr contamination may well be of greater concern than ^{137}Cs. When these radionuclides have actually entered the human food chain, radiocaesium may well be the prime cause for concern since it resides in the edible parts of the fish whereas radiostrontium becomes incorporated into bone.

23.3 Seasonality

If radioactive deposition occurs in the presence of snow and ice, radionuclides may reach the water during the thaw. By then, nuclides with a short physical half-life may already have disappeared due to radioactive decay. Flood water during snowmelt may, on the one hand dilute the amounts of radionuclides in a watershed or, on the other hand, add radionuclides from the catchment area.

If radioactive deposition occurs in the absence of snow and ice radionuclides will contaminate the surface water directly and may rapidly enter the aquatic food chain. Fish which eat contaminated plankton become contaminated almost immediately. Deposition during summer increases the transfer of radionuclides to fish since fish metabolism is faster during the warm season. During the cold period, fish metabolism is slow and thus uptake and excretion of radiocaesium are also slow.

23.4 Contamination of surface water

Contamination of the freshwater environment occurs either by direct deposition (dry or wet) into lakes and rivers or, long after direct contamination has ceased, by run-off from the surrounding catchment area. Climatic conditions and the nature of the catchment area are the main factors determining the degree of contamination through this indirect route. The nature of the catchment area is determined by soil type, vegetation and topography (Radioecology in Nordic Limnic Systems, 1991). The physico-chemical nature of the fallout will largely govern its behaviour in aquatic systems.

23.5 Contamination of fish

Transfer of radiocaesium into fish depends on many factors, one of the most important being the type of the lake. In oligotrophic lakes (i.e. lakes deficient in nutrients), uptake by fish is much higher than in eutrophic lakes (i.e. lakes abundant in nutrients). Water exchange conditions of a lake also strongly affect the radiocaesium content of fish. Shorter water residence times in lakes will favour a more rapid decline of radiocaesium in fish.

The main factors determining the ^{137}Cs content of fish are (Kolehmainen *et al.*, 1966):

- the limnological type of the lake (this is the main factor accounting for 10- to 100-fold differences in the same fish species in different lakes),
- the quality of food eaten by fish (accounts for 2- to 3- fold difference),
- the biological half-life of ^{137}Cs in fish (varies from 20 to 200 days at 15^{o}C in different species and produces up to 10-fold differences in various species in the same water course),
- the potassium concentration in the water and its conductivity (the ^{137}Cs content of fish is inversely proportional to the potassium content of the water).

Thus, the lake characteristics which favour high radiocaesium activity concentrations in fish, are:

- oligotrophic lake, or a shallow lake
- long water residence time
- a high proportion of fells or bogs in the catchment area increasing run-off to the lake
- low concentrations of potassium and low conductivity in lake water.

23.6 Reduction of contamination by natural processes and environmental half-lives

After fallout is deposited on the water radiocaesium is removed rapidly from the aqueous phase by binding to particulate material in water and sinking to the bottom. Up to 90% of deposited radiocaesium is removed from the lake water in just a few months (Radioecology in Nordic Limnic Systems, 1991; Saxén, 1987) but there will then be a period of slower decline. Radiostrontium activity concentrations in water fall much more slowly than those of radiocaesium.

The residence time of radiocaesium in lake ecosystems depends on a number of factors. During the first few months after the deposition of fallout, the environmental half-life of ^{137}Cs in water will be about 50 days (i.e. a rapid decrease). In lakes with a high sedimentation rate it will be shorter than in lakes with a low sedimentation rate. In the first few years, the environmental half-life of ^{137}Cs in water will be about 1 year, depending on the characteristics of the lake, and 2-3 years after the fallout event it will be about 3-5 years (Saxén, 1994).

The relationship between the input to and output from lake ecosystems will change with time (especially after the initial fallout) and affect the changes in radiocaesium content of the water. In shallow lakes with aerobic conditions, benthic organisms will recirculate radiocaesium from sediment into the food chain.

Radioactive substances deposited on the catchment area will, to some extent, be transferred to the watershed with run-off water (either in solid or in dissolved form), which will largely determine their significance for aquatic systems. The ion exchange capacity of soil in the catchment area affects the behaviour of the radionuclides. The caesium ion is easily fixed in the lattice structure of clay minerals, but in soils rich in organic matter there will be a lower capacity for retaining radiocaesium and contamination in run-off will be higher (Hilton *et al.*, 1993; Saxén, 1994). Strontium, on the other hand, is much more mobile, and is largely removed from the catchment area by run-off to and from the watershed (Radioecology in Nordic Limnic Systems, 1991).

Fish obtain radionuclides mainly through their food chain and only absorb relatively small amounts directly from water (Kolehmainen *et al.*, 1967; Ugedal *et al.*, 1988). They therefore continue to ingest contaminated feed long after the radiocaesium in the aqueous phase has disappeared. Fish species with different feeding habits reach maximum values of radiocaesium body content at different times. Non-predatory fish reach maximum values in the summer following the discharge. Depending on the lake characteristics, predatory fish reach a maximum contamination level 2-3 years after fallout occurs.

After reaching maximum values radiocaesium body burdens of predatory fish then fall with an effective ecological half-life of 1-3 years over the next few years and eventually 4-5 years (Brittain, 1991). The half-lives for ^{137}Cs in certain categories of freshwater fish are given in Table 5.1.

Table 5.1 *Effective ecological half-lives of ^{137}Cs in different types of fish and in perch and pike in large Finnish drainage basins in the 4-5 years after reaching maximum activity concentrations (Saxén, 1994).*

Fish	Effective ecological half-life (year)
Predators	0.72 - 4.8
Non-predators	0.84 - 3.4
Intermediate	0.91 - 3.9
Perch	0.81 - 5.0
Pike	0.75 - 5.2

23.7 Transfer factors

Transfer of radionuclides after deposition to water and fish can be described using aggregated transfer factors (T_{ag}) = radionuclide activity concentration in water or in fish (Bq kg^{-1} fresh weight) / amount of the radionuclide deposited (Bq m^{-2}).

The annual averages and variation in transfer factors for ^{137}Cs from deposition to surface water with variation are given in Table 5.2. Curves fitted to the points are given in Figure 5.1.

Table 5.2 *Annual averages of transfer factors, $TF_W = C_W/D$, from deposition to surface water in large drainage basins, with variation. C_W = annual average ^{137}Cs activity concentration in surface water (Bq m^{-3}) and D = average deposition of ^{137}Cs to the same area (Bq m^{-2}) (Saxén, 1994).*

Year after deposition	TF_W	
	Mean	Variation
1st Year	0.054	0.0035 - 0.064
2nd Year	0.013	0.0066 - 0.034
3rd Year	0.0076	0.0032 - 0.019
4th Year	0.0058	0.0024 - 0.012
5th Year	0.0042	0.0017 - 0.010
6th Year	0.0042	0.0019 - 0.0091

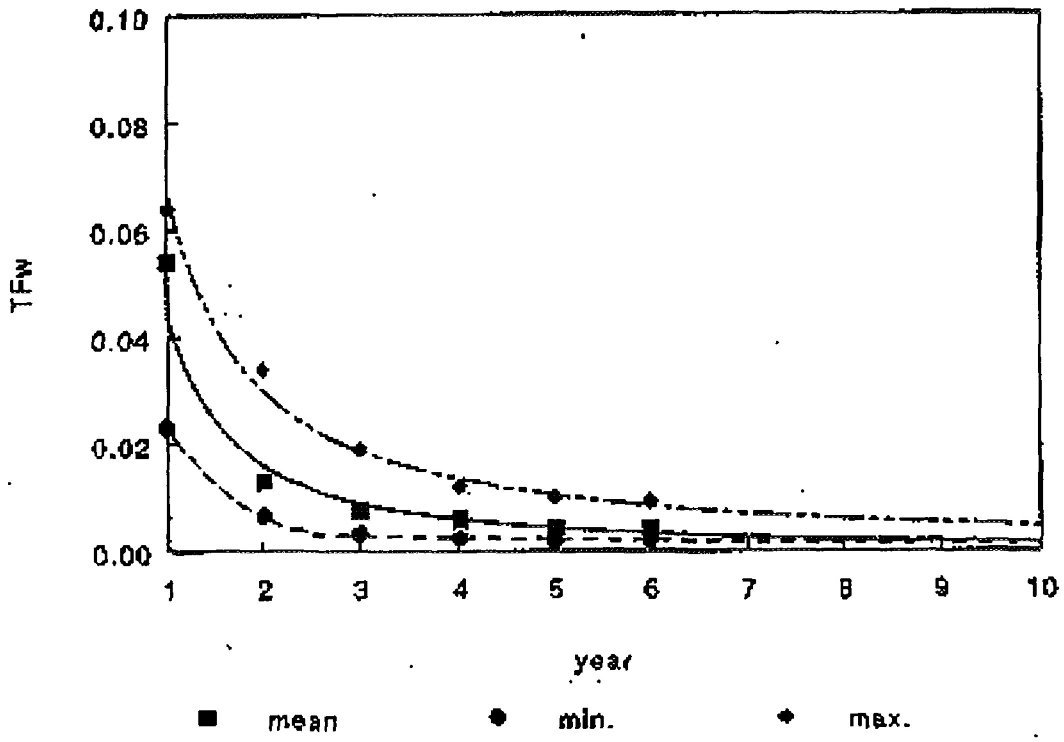

Figure 5.1 *Radiocaesium transfer coefficients with variation from deposition to surface water Bq m^{-3} / Bq m^{-2}.*

Average activity concentrations of ^{137}Cs in surface water during the first half year after deposition can roughly be estimated using the following equation (Saxén, 1994):

$$C_w = 45.5 \times D - 59.2 \ (\mathrm{Bq\ m^{-3}}) \tag{5.1}$$

Where:

C_w = the average concentration of ^{137}Cs in surface water (Bq m^{-3})

D = the average deposition of ^{137}Cs to the catchment area (kBq m^{-2})

Aggregated transfer coefficients for fish in the Nordic countries are summarised in Table 5.3.

Table 5.3 Aggregated transfer coefficients for radiocaesium to fish in Nordic countries, mean ± standard deviation (Aarkrog et al., 1988; Brittain, 1991; VAMP, 1992; Hammar et al., 1991; Saxén, 1994).

Year after deposition	T_{ag}, $m^2\ kg^{-1}$		
Finland	**Predators Mean ± SD**	**Non-predators Mean ± SD**	**Intermediate Mean ± SD**
1st	0.027 ± 0.016	0.031 ± 0.012	0.054 ± 0.025
2nd	0.14 ± 0.07	0.069 ± 0.042	0.14 ± 0.064
3rd	0.13 ± 0.11	0.036 ± 0.028	0.099 ± 0.062
4th	0.088 ± 0.057	0.026 ± 0.016	0.071 ± 0.051
5th	0.065 ± 0.032	0.013 ± 0.008	0.045 ± 0.028
Sweden		**Brown Trout**	**Arctic char**
1st		0.17	0.11
2nd		0.17	0.11
3rd		0.10	0.077
4th		0.064	0.050
5th		0.048	0.040
Norway	**Brown trout**		
1st	0.0036		
2nd	0.031		
3rd	0.024		
4th	0.016		
Denmark	**Fish**		
2nd	0.033 - 0.15		

An example of lake specific transfer coefficients from deposition to fish in two oligotrophic lakes, which greatly differ from each other with respect of lake size, depth, and water residence time, is given in Table 5.4.

Table 5.4 *Aggregated transfer coefficients from deposition to perch and pike (Bq kg^{-1} in fish/Bq m^{-2} in deposition) in two contrasting oligotrophic lakes (1st year = the year when the deposition occurred). Lake 2 represents conditions of maximum accumulation (rather shallow, small lake, with negligible water exchange).*

Year after deposition	**Lake**	
	1	**2**
Perch		
1st year	0.031	-
2nd year	0.045	0.14
3rd year	0.045	0.15
4th year	0.036	0.10
5th year	0.025	0.089
6th year	0.016	0.067
Pike		
1st year	0.016	-
2nd year	0.060	0.30
3rd year	0.069	0.26
4th year	0.030	0.30
5th year	0.022	0.19
6th year	0.020	0.20

In an eutrophic Finnish lake transfer coefficients for ^{137}Cs to different fish species varied from 0.003 to 0.01 m^2 kg^{-1} in the second year after deposition.

The use of average transfer coefficient values for different lakes may be generally more useful than the use of extreme values, because the trophic status of lakes and the other factors affecting the transfer of radionuclides to fish are not always known. The annual averages and ranges in transfer coefficients for fish in Finnish conditions are given in Table 5.5.

Table 5.5 Annual averages (and ranges) in aggregated transfer factors, T_{ag} (m^2 kg^{-1}), for radiocaesium to fish in large Finnish drainage basins 1-5 years after deposition (Saxén, 1994).

Year after deposition	**T_{ag}**	
	Mean	Variation
1st	0.041	0.022 - 0.073
2nd	0.120	0.065 - 0.200
3rd	0.091	0.019 - 0.240
4th	0.046	0.012 - 0.075
5th	0.037	0.0055 - 0.062

Typical changes in aggregated transfer coefficients as a function of time for large eutrophic and oligotropic lakes and for a small oligotrophic lake with negligible water exchange are given in Figs. 5.2, 5.3 and 5.4.

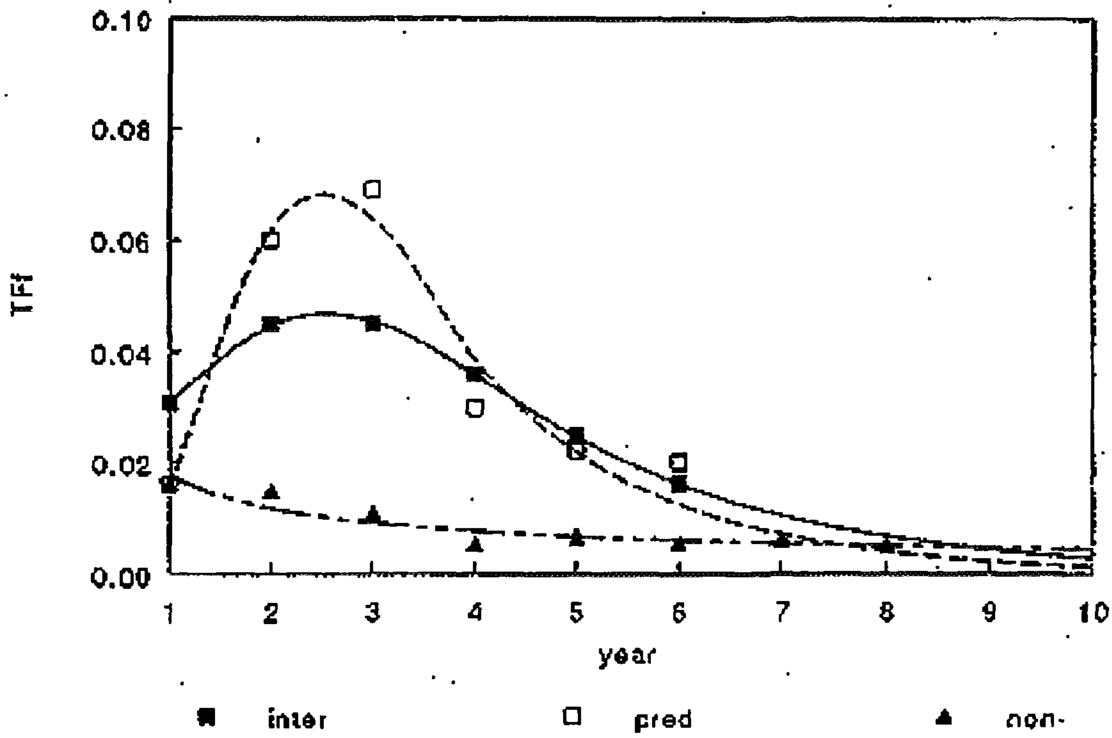

Figure 5.2 Changes with time since deposition in radiocaesium transfer coefficients for three different types of fish (predatory, non-predatory and intermediate) in a large oligotrophic lake.

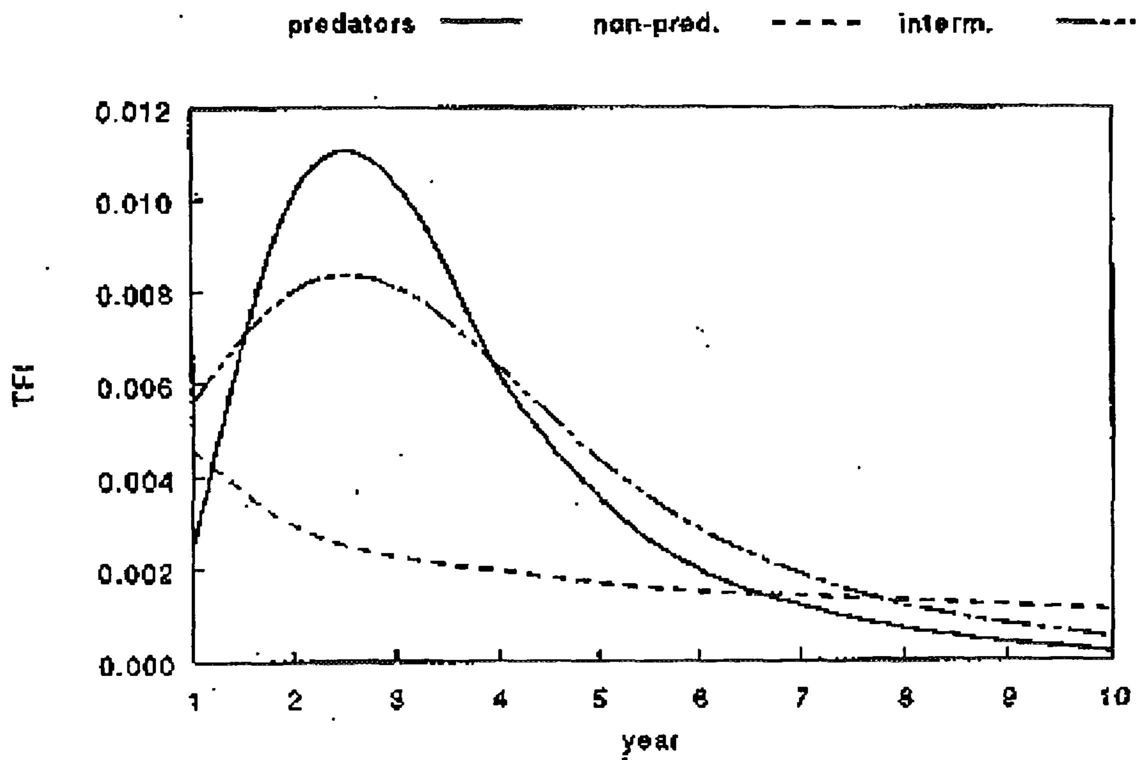

Figure 5.3 *Changes with time since deposition in radiocaesium transfer coefficients for three different types of fish (predatory, non-predatory and intermediate) in a large eutrophic lake.*

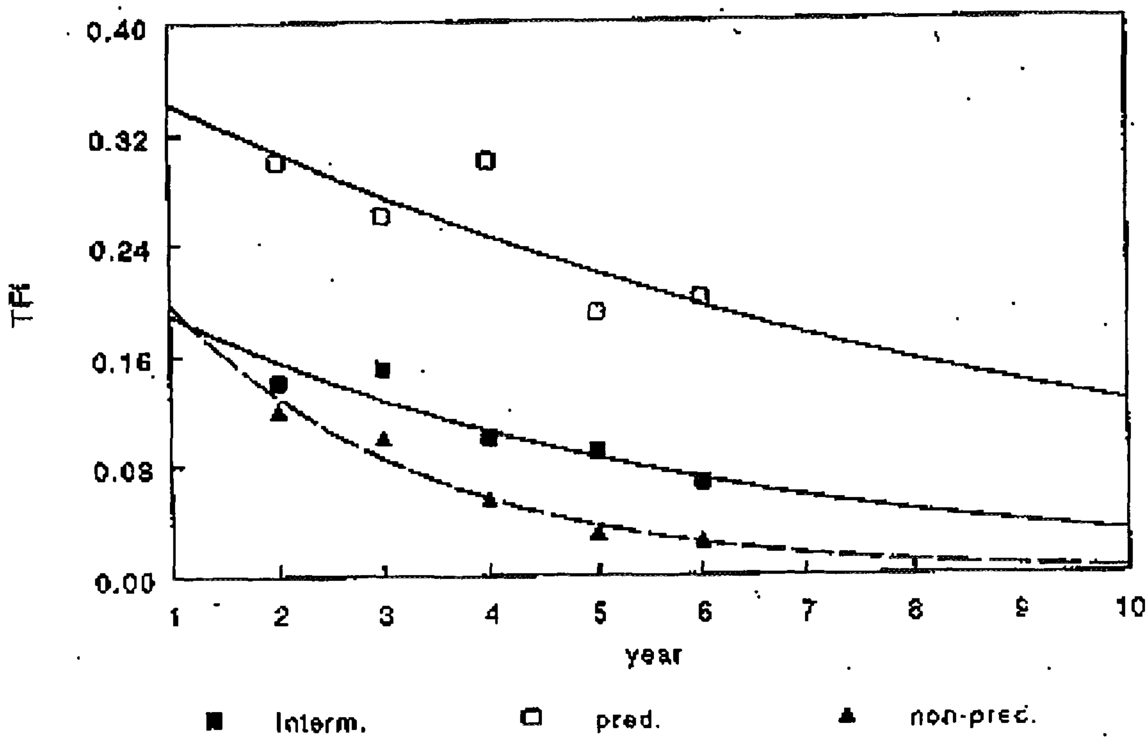

Figure 5.4 *Changes with time since deposition in radiocaesium transfer coefficients for three different types of fish (predatory, non-predatory and intermediate) in a small oligotrophic lake with a long water residence time.*

According to a Swedish study (Andersson *et al.*, 1991) the ^{137}Cs activity concentration (Bq kg^{-1} dry weight) in perch can be estimated using the following equation:

$$Cs_{pe}(t) = Cs_{soil} \times k_1 \times k_{dev} - (k_{1/2} + k_2) \times Cs_{pe}(t-1)$$

where: k_1 = primary dose constant ($month^{-1}$)
k_2 = decrease of ^{137}Cs in the perch ($month^{-1}$)
$k_{1/2}$ = radioactive decay constant for ^{137}Cs ($month^{-1}$)
$Cs_{pe}(t-1)$ = Cs content of the perch one year before year t (i.e. Bq kg^{1} dry weight)

Cs-137 activity concentrations in fish in the first half year after the fallout has been deposited can be roughly estimated using the following equation (Saxén, 1994):

$$C_f = 62.6 \times D - 93 \text{ (Bq kg}^{-1}\text{ wet weight)}$$

where: C_f = the average concentration of ^{137}Cs in fish (Bq kg^{-1}),
D = the average deposition of ^{137}Cs to the catchment area (kBq m^{-2}).

23.8 Countermeasures for fish

The choice of possible countermeasures will be influenced by the overall fallout situation including the season, magnitude of the deposition, radionuclide composition, and the scale and type of land area contaminated. Countermeasures in the aquatic environment may be required for two reasons: (i) to limit contamination of fish and (ii) to limit contamination of drinking water.

Possible countermeasures for reducing radiation dose via fish are:

- limiting consumption of fish
- adding different chemicals (lake liming) to the lakes to decrease biological uptake of radionuclides
- adding chemicals to the catchment area to decrease leakage of radionuclides to watersheds, or
- preventing transport of radionuclides in runoff water
- removal of contaminated soil from the catchment area, or removal of contaminated snow if the fallout occurs in winter
- actions to prevent flooding in contaminated areas (building flood plain dams).
- decreasing contamination by food processing

Some countermeasures, such as removal of contaminated soil or snow from the catchment area and actions to decrease transport of radionuclides with runoff to the watershed, are theoretically possible, but only in limited circumstances. They are considered to be applicable to discharges over only a limited area and are not discussed further.

23.8.1 Limiting consumption of fish

Effectiveness, costs and practicality

Imposing limitations on the extent of fish consumption following a fallout event is a relatively easy and effective countermeasure, but its success and practicability depends on the availability of alternative foodstuffs. Monetary costs are in the form of lost income for the fishermen and the possible difference in the price of fish and the substitute foodstuff.

23.8.2 Treating fish before consumption (brining)

Effectiveness, costs and practicality

The radiocaesium content of fish can be greatly reduced by soaking in brine (3%NaCl). Three or four successive soakings of at least 4 hours each in brine can reduce fish radiocaesium content by more than 60% (Petäjä *et al.*, 1992). The texture of the fish is not markedly altered and the final salt content of the fish is only slightly increased. The costs of soaking in brine are trivial, constituting only the purchase of salt and the labour involved.

23.8.3 Addition of chemicals (liming)

The effect of liming lakes has been studied with various types of lime and with different application methods in a large number of lakes in Sweden (Andersson *et al.*, 1991). The most important liming methods were lake liming, wetland liming and potassium treatment.

Effectiveness

All the liming methods appeared to be rather ineffective as an acute remedial action for the aquatic environment. No rapid decrease in radiocaesium activity concentrations of fish was achieved with any of the liming methods. A maximal enhancement of 5% was achieved in the yearly decrease in ^{137}Cs contents of pike with the liming methods.

Cost

A summary of costs of the different liming methods is given in Table 5.6.

Table 5.6 *A summary of costs/unit volume (kECU m^{-3} * 10^6) of the most important liming methods, adapted from Andersson et al. (1991).*

Countermeasure	Average	Median	Min	Max
Lake liming	6.9	3.6	1.8	38
Wetland liming	35.0	22.6	10.8	112
Potassium treatment	7.0	6.5	3.0	16.2

23.9 Countermeasures for drinking water

In a heavy fallout situation, the priority response should be to minimise the consumption of household water. The need for household water decreases considerably for the initial period of sheltering and an estimated need for 45 L^{-1} d^{-1} per person for someone indoors can be reduced to a minimum need of 5 L^{-1} d^{-1} per person. In water towers or other indoor areas of water treatment plants sufficient purified water would be available for drinking water needs during the most acute period.

The assessment and choice of countermeasures will be governed by the nature of the contamination. If the main contaminant is radiocaesium, the countermeasures should be implemented immediately after the fallout has deposited to get the highest reduction of radiation dose. About 50% of the radiocaesium in raw water is removed during normal operations in the water treatment process, since it coprecipitates with aluminium sulphate in the coagulation stage (Saxén, 1987). Some other radionuclides are also removed during the coagulation process (Lowry and Lowry, 1988). Radionuclides in particulate form are largely removed at the sand, activated carbon or other filtration stage. If the main contaminant is ^{90}Sr, the situation is more complex, since strontium is not removed from water by normal treatment (Pesonen *et al.*, 1972).

The following possible measures for ensuring the provision of safe drinking water over the long term should be considered:

- substituting surface water with uncontaminated ground water. The content of natural radionuclides in the ground water must be checked first,
- substituting the contaminated water source with a source located in an uncontaminated or less contaminated area,
- if neither of the above two actions is possible, countermeasures to remove deposited radionuclides from raw water should be considered.

23.9.1 Use of an alternative water supply

Effectiveness, costs and practicality

Water supply organisations should have contingency plans to substitute a contaminated surface water source with ground water or with a less contaminated water source. The effectiveness in reducing radionuclide intake depends on the contamination level of the substituting water source and on the dilution grade, if the contaminated water is only partly substituted. If it is possible to use ground water, the effectiveness for artificial radionuclides is more than 90%, but ground water may in some cases contain rather high amounts of natural radionuclides.

23.9.2 Ion exchange method

Effectiveness, costs and practicality

The most effective method for removal of radionuclides from water is the use of ion exchange or reverse osmosis. A mixed bed ion exchanger removes virtually all the radionuclides, both in cationic and anionic form, and the efficiency of the method can be nearly 100% (Lowry and Lowry, 1988). The practicability of this countermeasure may be low on a large scale, but it might be possible on a small scale. A disadvantage of the method, besides costs, is the need for treatment and storage of the used radioactive ion exchange resins, which could be regenerated to give liquid waste which can be further concentrated.

Costs arise from construction, operation and maintenance of an ion-exchange plant, purchase of resins, and the treatment and storage of used ion exchange resins. Examples are given in Houck *et al.* (1985).

24. An outline strategy for dose reduction in the contaminated aquatic environment

As a demonstration of strategies for dose reduction following radioactive contamination we consider an event with deposition levels of 1 MBq of both ^{137}Cs and ^{90}Sr m^{-2}.

In the contaminated example area there are three lakes. The areas, depths and types of the lakes are given in Table 5.7. The largest, with a surface area 100 km^2, is used as a raw water source for a water treatment plant. It produces part of the household water for the population in the contaminated area, which has 500,000 inhabitants. The average consumption of household water is about 200 L^{-1} d^{-1} per person in block of flats and 150 L^{-1} d^{-1} per person in single-family households in the countryside. The need for drinking water is 2 L^{-1} d^{-1} per person. This means that the total need for household water in the contaminated area is normally about 90 000 m^{-3} d^{-1} and for drinking water 600 m^{-3} d^{-1}. People living in the countryside have their own ground water wells and for the urban population water treatment plant of the neighbouring town produces the rest of the household water. The lakes in the area are important for recreational fishing by the local population mainly during the summer time.

Table 5.7 *Surface areas, mean depths and limnological types of the three lakes in the fallout area.*

	Area (km^2)	Mean depth (m)	Limnological type	Water residence time
Lake 1	100	6	Oligotrophic	short
Lake 2	0.05	3	Oligotrophic	long
Lake 3	1	4	Eutrophic	short

The deposition of fallout is assumed to occur on the 10th June. Because it is summer time and the lakes are not frozen, lake and river waters are contaminated immediately. Also plankton eating fish are contaminated soon after deposition.

24.1 Drinking water

24.1.1 Estimation of concentrations

Annual average transfer coefficients taken from the curves in Figure 5.1 were used to calculate the activity concentrations for ^{137}Cs in the water of the largest lake, assuming the deposition of 1 MBq m^{-2} of ^{137}Cs(Table 5.8). Estimates of maximum activity concentrations for ^{90}Sr in lake water was based on the assumption that the direct deposition, 1 MBq m^{-2}, was evenly distributed to the whole volume of the lake. Further, the concentration of ^{90}Sr was assumed to decrease with a half life of one year during the first year after fallout, after that with a half life of five years (Table 5.8).

Table 5.8 *Estimated annual mean activity concentrations of ^{137}Cs and ^{90}Sr in lake water during ten years after the deposition of 1 MBq m^{-2} of both ^{137}Cs and ^{90}Sr.*

Year after deposition	**^{137}Cs, kBq m^{-3}**			**^{90}Sr, kBq m^{-3}**		
	mean	**min**	**max**	**mean**	**min**	**max**
1	44	5	67	82	33	130
2	16	4	30	51	21	80
3	9	3	19	44	18	69
4	6	2.5	14	38	16	60
5	4	2.2	10	33	14	52
6	3	2	8	29	12	46
7	3	2	7	25	10	40
8	2	2	6	22	9	35
9	2	2	5	19	8	30
10	2	1	5	17	7	26

Radiation doses from drinking water without any countermeasures were calculated annually for 10 years after the deposition (Table 5.9). The first year, the first few months following the deposition, are the most important, because the activity concentrations are then highest. Cs-137 is quickly removed from the water phase, while the decrease in ^{90}Sr concentrations is slower. In calculation of radiation doses it was also assumed that 50% of the ^{137}Cs is removed in the water treatment process, but that no ^{90}Sr is removed during the process.

Table 5.9 Internal radiation doses from drinking water during ten years after a deposition of ^{137}Cs and ^{90}Sr (1 MBq m^{-2}) without any countermeasures, based on a daily water consumption of 2 L per person.

Year after deposition	Internal radiation dose (mSv y^{-1})	
	^{137}Cs	^{90}Sr
1	0.22	2.0
2	0.083	1.2
3	0.046	1.1
4	0.031	0.94
5	0.027	0.82
6	0.017	0.70
7	0.014	0.63
8	0.011	0.54
9	0.010	0.46
10	0.008	0.40

24.1.2 Strategy

In the water tower and other inside rooms of the water treatment plant there is often sufficient water for drinking water needs for the most critical period, if consumption is limited. After that the water produced by the neighbouring town would be used for drinking water. Care must be taken to ensure that the ground water wells in the countryside are protected and covered well so that transfer of deposited radionuclides into the wells is prevented. Cleaning water with the ion exchange method is expensive compared to the saved radiation dose and suggests that this countermeasure would not be a prime implementation measure in a fallout situation.

24.2 Fish

If the deposition is spatially limited, as in this example, the most effective way to reduce radiation doses via fish, is to stop eating fish from lakes in the contaminated area. If the contaminated area is large, and there is lack of uncontaminated foodstuffs, some countermeasures to reduce radioactivity in fish should be considered.

No acute method for reclamation of the contaminated lakes exists. The cheapest way to reduce radiation doses via consumption of fish is brine treatment of fish in households. Liming of lakes or the catchment areas give only a slight decrease in ^{137}Cs contents of fish over a longer time period.

24.2.1 Estimation of concentrations

Transfer coefficients of ^{137}Cs from deposition to fish in the three lakes, given in Figs. 5.2, 5.3 and 5.4, were used to calculate the ^{137}Cs content of fish in lakes 1, 2 and 3 for ten years after the deposition event. The results are given in Table 5.10 together with the estimated doses to fish consumers due to ingested contaminated fish.

Table 5.10 *Predicted ^{137}Cs activity concentrations in different fish types (C_f) after deposition of 1 MBq m^{-2} in the three lakes described in Table 5.7. Also given is the predicted internal doses from consumption of fish based on a fish consumption rate of 4 kg y^{-1}.*

Year	Predators		Non-predators		Intermediate	
	^{137}Cs, kBq kg^{-1} fresh weight	**Dose, mSv y^{-1}**	**^{137}Cs, kBq kg^{-1} fresh weight**	**Dose, mSv y^{-1}**	**^{137}Cs, kBq kg^{-1} fresh weight**	**Dose, mSv y^{-1}**
			Lake 1			
1	15	0.8	18	1	31	1.7
2	62	3.5	12	0.7	45	2.5
3	64	3.6	9	0.5	45	2.5
4	39	2.2	8	0.4	36	2.0
5	22	1.2	7	0.4	25	1.4
6	13	0.7	6	0.3	16	0.9
7	7	0.4	6	0.3	11	0.6
8	4	0.2	5	0.3	7	0.4
9	2	0.1	5	0.3	4	0.3
10	1	0.07	4	0.3	3	0.2
			Lake 2			
1	340	19	198	11	190	10
2	310	17	130	7	160	9
3	275	15	85	5	130	7
4	245	14	56	3	105	6
5	220	12	37	2	86	5
6	196	11	24	1	70	4
7	175	10	16	0.9	57	3
8	156	9	10	0.6	47	3
9	140	8	7	0.4	39	2
10	125	7	5	0.3	32	2
			Lake 3			
1	2	0.1	5	0.3	6	0.3
2	10	0.5	3	0.2	8	0.4
3	10	0.6	2	0.3	8	0.5
4	6	0.4	2	0.1	6	0.4
5	4	0.2	2	0.1	4	0.2
6	2	0.1	1.5	0.09	3	0.2
7	1	0.07	1	0.08	2	0.1
8	1	0.04	1	0.07	1	0.07
9	0.4	0.02	1	0.07	0.8	0.04
10	0.2	0.01	1	0.06	0.5	0.03

24.2.2 Lake liming

As an example the cost of the saved radiation dose via fish using liming to reduce radioactivity of fish has been estimated. There are several liming methods, but lake liming and wet land liming were chosen as examples in the calculations.

Liming was assumed to have been carried out during the first year after deposition. The effect of liming was assumed to enhance the decrease in the ^{137}Cs activity concentrations in fish 5% from the second year onwards, which is probably too high. The costs for liming were taken from Table 5.6. The cost of the saved radiation dose was calculated in (ECU m^{-3})/(Sv kg^{-1}), because the liming costs depend on the water volume of the lake and the radiation dose via ingestion of fish depends on the amount of fish eaten. The cost of the saved radiation dose achieved by two liming methods as a function of time is given in Figure 5.5 for different types of fish.

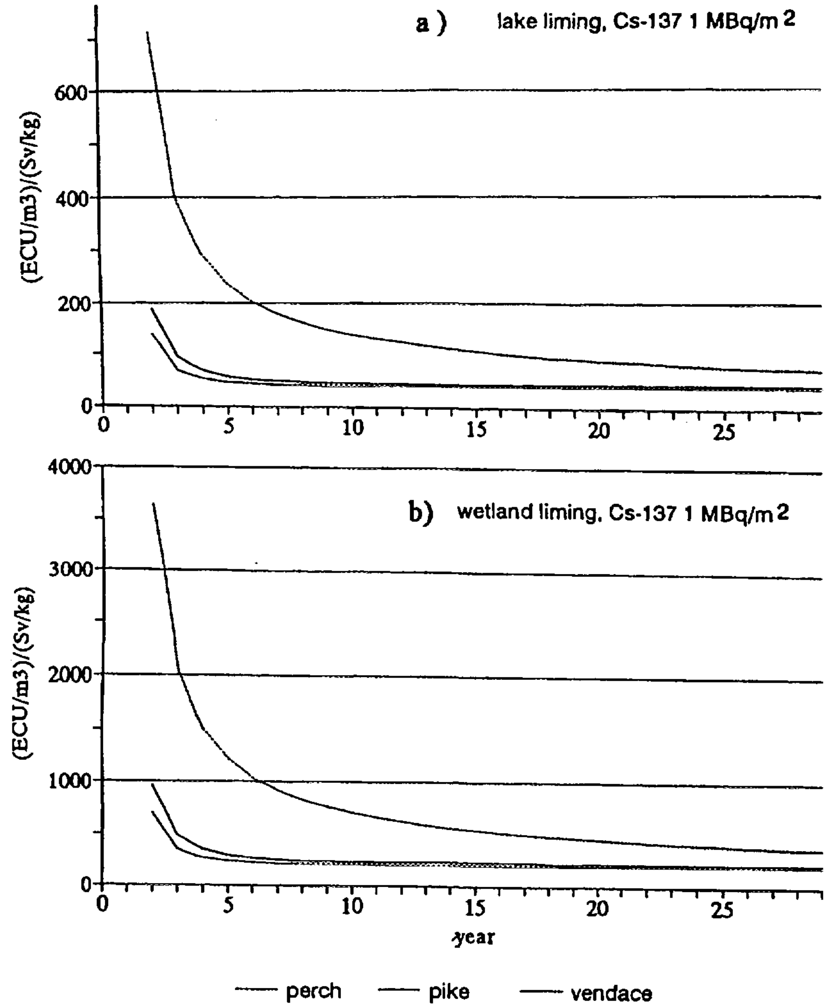

***Figure 5.5** The estimated cost of the saved radiation dose as a function of time in three fish types: Predators (pike), non-predators (vendace) and intermediate (perch), in a case of a) lake liming, and b) wet land liming.*

A conclusion is that liming methods are too expensive with respect to their decontamination effect on radiocaesium in freshwater fish. Brine treatment is a cheap and effective method for decontaminating fish.

***Table 5.13** A summary of effectiveness of different countermeasures for reducing radiation doses via fish and drinking water.*

Countermeasure	Effectiveness, % reduction	Cost
	Fish	
Brine treatment in households	60	cheap
Wet land liming	< 5	$6*10^9$ ECU/Sv[a)]
Lake liming	< 5	10^9 ECU/Sv[a)]
Limitations in consumption	up to 100	not estimated
	Drinking water	
Use of an uncontaminated source or ground water	up to 100	not estimated
Ion exchange	> 90	not estimated

a) for a lake of 100 km^2 and average depth 6 m, and fish consumption of 4 kg y^{-1} for 10 years

25. References

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