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RECLAMATION OF CONTAMINATED URBAN AND RURAL ENVIRONMENTS FOLLOWING A SEVERE NUCLEAR ACCIDENT

Nordic Nuclear Safety Research

BER 6

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Preface

This project was funded by the Nordic Nuclear Safety Research (NKS) programme. NKS is a co-operative body in nuclear safety, radiation protection and emergency preparedness. It's purpose is to support Nordic research projects, organise exercises, provide information and make recommendations to decision makers.

The project was initially co-ordinated by Judith Melin at the Swedish Radiation Protection Institute, but as she change place of work, Per Strand continued. The other contributors are acknowledged at the front page of their respective parts of the report. Reviewers have been John Sandalls (AEA Technology) and Brenda Howard (Institute of Terrestrial Ecology), UK. Their help is gratefully acknowledged.

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Introduction

In the event of a severe nuclear accident releasing radioactive materials to the atmosphere, there is a potential for widespread contamination of both the urban and rural environments. In some instances of environmental contamination, natural processes may eventually reduce or eliminate the problem without man's intervention. The situation with respect to radioactive contamination is no different except that radioactive contamination will also disappear through normal physical radioactive decay. In other cases, man is often able to mitigate potential harmful effects by cleaning, washing, abrading or by the application of chemicals. The actions taken by man to mitigate the potential **harmful** effects of contamination are described as **countermeasures. In the case of radioactive contamination, the objective of countermeasures is to minimise radiation doses to man**.

Following the deposition of radioactive material, the human population may be exposed to external and/or internal irradiation. External irradiation arises from contaminated land and contaminated exposed surfaces generally. Internal irradiation can arise from inhalation or from ingestion of food which has been contaminated directly and/or indirectly.

Whether the contamination occurs in urban or rural environments, a number of effective strategies are available to minimise the resulting dose. In the urban environment, the main objective is to minimise external irradiation from contaminated surfaces (roads, roofs, gardens etc). In urban environments with agricultural production and food gathering the objective is to produce contamination-free products and to minimise external radiation doses to people such as farm workers and foresters.

The decision as to whether countermeasures should be employed must be based on the net potential benefit to man. This means that considerations such as monetary cost and stress through social upheaval must not outweigh the potential benefits to be gained by reduction of radiation dose.

This document is intended as a guide to those groups who may, at very short notice, be called upon to manage and reclaim radioactively contaminated urban and rural environments in the Nordic countries. However, much of the information and recommendations are also equally applicable in other countries.

The document is divided into seven distinct parts, namely:

1 The Urban Environment
2 The Cultivated Agricultural Environment
3 Animals
4 Forests
5 Freshwater and Fish
6 Management and Disposal of Radioactive Waste from Clean-up Operations
7 Radiation Protection and Safety of Clean-up Operators
8 Resources Available in Society

EXECUTIVE SUMMARY

Executive Summary

The ultimate aim of this document is to provide a basis for preparing contingency plans for reducing radiation dose to man through reclamation and decontamination, and to illustrate some of the possibilities and difficulties in this planning. Our aim is to provide an input into the decision-making process, but this report avoids political considerations.

When developing a countermeasures strategy for any given fallout scenario there are several important factors that must be considered, namely the:

1. level of contamination,
2. type of deposit (dry, wet),
3. deposition characteristics,
4. time of the year when the deposition occur,
5. depth profile of the deposited material,
6. cost per unit dose reduction,
7. surface structure,
8. sequence of the different decontamination procedures,
9. contribution of different contaminated surfaces to the external dose rate,
10. contribution of different contaminated surfaces to the internal dose,
11. influence of natural dose reduction processes (e.g. weathering and diffusion) and normal human activity (e.g. street cleaning, ploughing),
12. dose reduction achievable on the individual surfaces using appropriate methods,
13. practicability of the various reclamation/decontamination procedures,
14. extra dose to the personnel performing the decontamination and to radiation protection personnel.

In addition, factors like countermeasure availability, effort and acceptability must be evaluated before implementation.

In the following text we will discuss each of the above aspects using examples. The estimates given in this summary, on the cost effectiveness of different countermeasures, are based on an assumed deposition event of 1 MBq m^{-2} of both ^{137}Cs and ^{90}Sr.

The *level of contamination* is, of course, of the utmost importance as a very low deposition rate may require no remedial action, whilst a very high rate may call for immediate evacuation and therefore no early decontamination may be appropriate except for essential buildings which may still be used. In the intermediate deposition range reclamation/ decontamination procedures may be feasible depending on cost-benefit calculations or other considerations, such as social-psychological or political factors. We learned from the Chernobyl accident that social-psychological factors are very important. Even today, when the dose-rate from the radiation has decreased considerably, reclamation of a contaminated areas is considered to be of great importance by residents and also for people who were relocated and want to return to their original homes. Another consideration for reclamation is the long term contamination levels in food products.

The deposition mode for radiocaesium and radiostrontium can vary; the deposition can come as dry deposition, in foggy conditions or as wet deposition. All these situations will give different deposition patterns.

In the case of dry deposition, trees can intercept a greater proportion of the contamination than during wet deposition. Because some of the contamination is retained above ground in forests, this could give rise to a fairly high contribution to the outdoor external dose, as there is virtually no shielding, and also, to a lesser extent, to the indoor dose, especially in terraced houses, as the irradiation may enter through windows which provide low shielding for people inside the house. For agricultural areas, dry deposition will be most important for the first harvest if the deposition happens in late spring or summer. In this case, agricultural products may be highly contaminated. Compared to other environments, like forests and crop fields, lakes have generally smooth and smaller, definite surfaces, and dry deposition to them will therefore be lower.

Wet deposition can occur with moderate rainfall, heavy rain, or, depending on the time of the year, it may come with hail or snow. The distribution of deposited material may vary, even on different horizontal surfaces. Run-off may play an important role, especially when it rains heavily, as part of the contamination may be carried away with the run-off water. This phenomenon is most important for hilly terrain and for paved areas and roofs, and may lead to high localised activity concentrations in bogs, lakes and sewage sludge treatment plants.

When material is dry deposited it initially stays on the surface. In contrast, in some cases of wet deposition, especially during rain storms, part of the deposited matter is found deeper in the soil.

The *surface structure* is also important. If a deposition occur just after a ploughing then the soil surface is very rough, and scraping, deep ploughing, skim and burial ploughing etc. will be less effective than if the soil surface was more smooth.

To define a strategy for decontamination it is important to quantify the *contribution of the contamination on different surfaces to total dose rates.*

To validate the effect of a decontamination method it also has to be compared to the *natural decontamination processes* such as weathering and *routine man-made processes*. For instance, traffic and normal street cleaning on roads with a high traffic intensity will rapidly decontaminate the roads so that normally no additional decontamination will be necessary.

The *sequence of the decontamination procedures* is also important. If a town has to be decontaminated, and it has been decided that some of the walls and the roofs must be decontaminated and also the gardens, then it is important to start with the roofs, then proceed to the walls and finally the gardens to avoid cross-contamination.

The *practicability of the various reclamation/decontamination procedures* must also be considered. If a special tool, treatment or machinery has to be used, then the availability of equipment has to be considered, and whether they are available in sufficient quantity to do the job. If not, other methods need to be used, or more equipment produced quickly. This will, for example, be the case if special ploughs, e.g. for deep-ploughing or skim and burial ploughing, is needed, as normally only a few of these special ploughs are available.

To reduce the *dose to the personnel performing the decontamination work* it will often not be feasible to decontaminate during the first few weeks after contamination occurs. However, this has to be balanced against the need for early decontamination to avoid relocation of the public in affected areas.

This study has considered various reclamation procedures that may be implemented in the event of radioactive contamination. Although suitable procedures vary in different environments such as forest, lake, agricultural, urban, some important common conclusions can be drawn for all of these areas. Furthermore some specific conclusions (for instance about the priority of different reclamation procedures in different environments) can also be made.

First of all, before a reclamation procedure starts, it is important that decision-makers make it clear which goal they intend to achieve. They might want to (i) reduce the average external dose to the members of the public to a certain level, (ii) reduce the contamination of agricultural produces like meat, crops etc. down to a pre-set level, (iii) obtain a certain decontamination factor or dose reduction factor for a specific surface, (iv) implement reclamation procedures in such a way that the cost per reduced person-Sievert does not exceed a certain amount of money, or (v) spend a fixed sum of money on reclamation of a certain area and get the highest reduction in person-Sievert possible for the specified amount of money.

A combination of some of the described options or other options may also be a possibility.

Having set the goal, a strategy has to be devised before the reclamation can be performed, and the decision-maker must be aware that performing some reclamation procedures, that may reduce the dose to a certain level, may exclude the application of other procedures which could have reduced the dose further. For instance, ploughing a contaminated grassy field and sowing new grass reduces the intake of radioactive material by animals considerably. Having performed this procedure, however, deep ploughing or a skim and burial ploughing will have little effect compared to what it would have had if normal ploughing had not been carried out.

Sometimes, it is important to start reclamation quickly, but it may also be beneficial to wait. For an example of a grassed field it is important to act fairly quickly to provide less contaminated grass for grazing animals. In other cases, such as the forest environment, it is beneficial to wait until short lived isotopes have disappeared. The effect of the different procedures are dependent on the time of the year when the contamination has occurred. If it occurs in the middle of the winter many crops will not have been contaminated and the planning of the countermeasures for agriculture is not so urgent.

If the cost of certain procedures is nearly the same, independent of the contamination level, then the cost of a person-Sievert reduction will be inversely proportional to the contamination level.

Some general priorities are given below for the different environments considered:

Urban environments
If a decision-maker wants to achieve the highest dose reduction to members of the public for the money spent on reclamation some priorities lists can be made using the strategies outlined in Part 1 of this report. These are shown in Table 1.

Decontamination of gardens can be done by a special digging procedure and only needs well instructed man power. The user or owner of the house may be able to do this after receiving adequate information and instructions. *Decontamination of streets* can be performed by the municipal authorities. *Decontamination of walls and roofs* may be performed either by specialist firms or by the user or owner using hobby pressurised water cleaner equipment. *Decontamination of trees* can be performed by foresters or gardeners or by the owner or user by cutting trees and hedges. *Indoor decontamination* can be performed by the owner or user using vacuum-cleaners.

Table 1. *Priority order for decontaminating different urban surfaces.*

Priority	**Deposition mode**	
	Dry deposition	**Wet deposition**
One-storey detached houses		
1st	Streets	Streets
2nd	Gardens	Gardens
3rd	Walls	Roofs
4th	Trees	Trees
5th	Roofs	
6th	Internal surfaces	
Two-storey semi-detached houses		
1st	Streets	Streets
2nd	Gardens	Gardens
3rd	Trees	Roofs
4th	Roofs	Trees
5th	Internal surfaces	
6th	Walls	
Rows of terrace houses		
1st	Streets	Streets
2nd	Gardens	Gardens
3rd	Trees	Trees
4th	Roofs	Roofs
5th	Internal surfaces	
6th	Walls	
Multi-storey houses		
1st	Streets	Streets
2nd	Trees	Gardens
3rd	Gardens	Trees
4th	Internal surfaces	
5th	Walls	
6th	Roofs	

Cultivated agricultural environments
In the agricultural environment the time of the year that radioactive fallout occurs will influence the consequences in the short-mid term. Crops can easily be rendered unsuitable for food and animal feed. However, countermeasures such as the removal of crops and preparation of land for new crops can be considered.

Further steps then have to be decided upon. Crucial issues are then the contamination levels of the different soil types. Priority should be given to peaty and sandy soils, especially those acreages used for temporary or permanent grasslands; our study has indicated that, in an agricultural environment with different soil and crop types, these soil types deliver disproportionally large fractions of radiocaesium to food and feed from acreages with complex soil environments. For radiostrontium priority should be given to natural pastures.

Available soil/or crop treatment countermeasures are:

A. Removal of crops.
B. Removal of a slice of the top soil layer.
C. Replacement of the uppermost soil layer by skim and burial ploughing.
D. Additional fertilisation with potassium.
E. Additional liming.

A summary of the cost effectiveness for one specific deposition level, 1 MBq m^{-2}, in a model of a Nordic agricultural environment is given in Table 2. This study shows that exclusion of land from feed production is advisable. Excluding the natural pastures on peat soils (making up about 3% of the acreage) will reduce radiocaesium transfer by 60%. Excluding all peat soils (about 15% of the acreage) from crop production give about 90% reduction in radiocaesium transfer in the long term.

Table 2. *The cost effectiveness of different soil based countermeasures studied in environments of the Nordic countries assuming* ^{137}Cs *deposition of 1 MBq* m^{-2}.

Soil based countermeasure	Product	% reduction in person-Sv	Cost (ECU/person-Sv)
Removal of soil layer, 0 - 5 cm	Bread grain	95	40000
	Potatoes	95	2100
	Milk	95	4100
Skim and burial ploughing	Bread grain	90	7300
	Potatoes	90	400
	Milk	90	760
Annual additional K-fertilisation	Bread grain	39	94000
	Potatoes	50	4000
	Milk	67	2200

Action is required when pasture is expected to produce grass with unacceptably high radionuclide contents. For this situation, the grass sward has to be incorporated into the soil either by normal ploughing, with about 90% reduction of the nuclide transfer in the next crop, or by skim and burial ploughing with a higher efficiency. If removal of a slice of the top soil or replacement of the uppermost soil layer are thought necessary on some fields, these fields should be removed from production until the required countermeasures can be implemented.

Another action is needed when radionuclide transfer to arable crops is unacceptably high. Then, removal of a slice of the top soil or replacement of the uppermost soil layer is needed. If contamination levels are not so high as to require the reduction obtained through removing a slice of the top soil or replacement of the uppermost soil layer is necessary, complementary application of extra fertilising with potassium and extra liming can reduce the transfer to the crops by 30-60% on soils deficient in potassium available to crops and soils poor in calcium, respectively.

Animals

The transfer of radioactivity to animals and animals products is greatly dependent on management practices. In cultivated areas, the transfer of radiocaesium to animals is low compared to the transfer in unimproved areas or to wild animals. However, the rate of food production is higher in cultivated areas. Thus, both transfer and production rates have to be compared. Nevertheless, it is clear that to avoid the highest doses to critical groups and to achieve the greatest reduction in activity concentrations it is essential to consider animal products from unimproved areas (including game) in countermeasure strategies. A summary of recommended countermeasures and their cost effectiveness (for one specific deposition level) is given in Table 3.

Table 3. *Recommended countermeasures for different animal products, and their cost effectiveness assuming a ^{137}Cs deposition of 1 MBq m^{-2}.*

Product	Countermeasure	% reduction in person-Sv	Cost (ECU/person-Sv)
Cattle milk	1. Bentonite as powder or in concentrate	50	140
	2. AFCF[1)] as powder or in concentrate	90	470
Cattle meat	1. Bentonite as powder or in concentrate	45	170
	2. AFCF as powder or in concentrate	80	553
	3. Special feeding	50	740
	4. AFCF in boli	70	1020
Sheep meat	1. AFCF in salt lick	50	230
	2. AFCF in boli	50	1670
Goat milk	1. Bentonite as powder or in concentrate	50	19
	2. AFCF as powder or in concentrate	90	64
Reindeer meat	1. AFCF in salt licks	50	15
	2. AFCF in boli	50	95
	3. Change slaughter time	70	82

[1)] AFCF is a derivative of hexacyanoferrate.

The use of bentonite (a clay mineral), mixed with concentrate or given directly to animals, requiring daily management by man is the most cost effective countermeasure for radiocaesium, and gives about 50% reduction in radiocaesium activity concentrations in both milk and meat. More expensive, but also more effective, is AFCF (a hexacyanoferrate derivative) administered in the same way. This gives a 80-90% reduction in the contamination levels. Both bentonite and AFCF strongly bind the Cs ion so that it cannot be transferred across the animals gut.

If domesticated animals (e.g. goat, cattle, sheep) or semi-domesticated reindeer are grazing unimproved areas without daily contact with man, AFCF can be administrated either in a salt lick or boli. The salt lick is most cost-effective, but results in larger variability in activity concentrations in animal products because individual animals use the salt lick to different extents. On average, the reduction for radiocaesium would be about 50% for both methods.

Highly effective radiostrontium binders are not yet available. Moderately effective zeolites (clay minerals) may reduce transfer of radiostrontium to milk by up to a factor of two. For radiostrontium, the best currently available additive for animals is to increase the calcium content of the diet. The effectiveness of this approach will depend on the Ca intake and milk yield of the ruminant, but may achieve a reduction of 2-3 fold.

If the above mentioned countermeasures are not possible or do not give the necessary reduction, the use of special feeding should be considered. By giving the animals uncontaminated or lightly contaminated fodder over a period of time activity levels in meat and milk will be reduced. The length of time needed will depend on the original activity concentration levels and the biological half-lives of the isotopes in the animals. If uncontaminated fodder cannot be produced in the area or cannot be imported, then the animals can be transported to an uncontaminated area. These countermeasures are useful for all relevant radionuclides, including both radiocaesium and radiostrontium.

When using contaminated pastures for grazing, one should also bear in mind that the smaller animals such as sheep and goats will have higher activity concentrations in meat and milk than larger animals like cattle if they are grazing on the same pastures. This is also valid for wild animals (e.g. roe deer and moose from the same area).

For wild animals, AFCF might be made available in contaminated areas in salt licks. However, for population dose reduction, another important possible countermeasure is to change the hunting season. This may reduce radiocaesium activity concentrates in meat in species, such as roe deer and moose, which have a seasonal variation in activity levels and for which the hunting season does not coincide with the minimum levels in the animals.

Dietary and food processing advice may be important for special population groups which have a high consumption rate of wild animal meat such as hunters.

After restriction of food production from an area, banning of food products is the most drastic countermeasure available, and this is very expensive and socially disruptive.

Forests

Forests have the capacity to trap and retain radionuclides for a substantial period of time. The rotation period of a forest stand in the Nordic countries is about 100 years. The time of decomposition of organic material in a forest environment can be several hundred years. The retention of radionuclides in the forest environment is thus considerably longer compared to other environments.

Countermeasures in forest areas outside the urban environment are generally not worth implementing due to high costs compared to the reduction in dose. For a populated forested area, clear felling can, however, be considered as a countermeasure in the first year after deposition. To minimise contaminated timber products the surface part of the trees can be removed before processing.

Freshwater and fish

In some Nordic countries more than half of the drinking water is taken from lakes or rivers, which can be readily contaminated with fallout radionuclides. The season and nuclide composition of the fallout, together with environmental factors, determine the duration of and temporal changes in radionuclide activity concentrations in water. Therefore, the possible actions for decreasing radiation doses must be estimated separately in every case. If serious contamination of water with radiocaesium occurs, the highest dose reduction is achieved by actions carried out during the first months after deposition. In case of radiostrontium, long-term countermeasures may be needed. If countermeasures are needed, decreasing intake of household water to a minimum will be the first action. After this short term action, the use of uncontaminated ground water should be considered or drinking water should be imported from an uncontaminated area. Water treatment plants are recommended to make plans for these actions which take into account the situation for their specific area. The most costly measure is to introduce methods for removing deposited radionuclides from raw, untreated water by the use of ion exchange filters.

To reduce radiation doses from fish consumption, the most effective and less expensive method is brine treatment of fish at home. Liming of the lakes gives only a small decrease in ^{137}Cs contents of fish over a longer time period. Transfer of radiostrontium to fish is low and will rarely cause problems.

Management and disposal of radioactive waste from clean-up operations

Clean-up of large contaminated areas will probably create enormous amounts of radioactive waste which have to be safely disposed of. In urban areas, the most important waste categories are removed soil and vegetation from clean-up of gardens as well as effluents and solids from cleaning streets by firehosing and sweeping. Sludge from municipal water treatment plants may also have to be treated and disposed of as radioactive waste. In agricultural areas contaminated soil, if the uppermost soil layer from fields is removed, forms the most voluminous waste. The amounts of contaminated crops is considerably smaller but its activity concentrations may be rather high, especially if the crops is removed shortly after a dry deposition event. Very high amounts of waste will be produced from the decontamination of forests: stems (or only bark if trees are barked), tree crowns, litter, humus and soil.

Some of the waste forms will probably need pretreatment prior to disposal. For example, effluents from firehosing cannot be directly disposed of and removal of the majority of the radionuclides will be necessary.

Transportation of radioactive waste to final disposal sites can be accomplished using ordinary transportation and haulage equipment such as trucks, dump trucks, bulldozers etc. The means of transportation largely depends on the distance to final disposal sites. It is also important to ensure safety during transportation, especially to prevent the spread of contaminated materials while loading and transporting the waste.

The most likely modes for final disposal are burial of radioactive waste into shallow ground trenches and surface mounds, which may need lining and drainage to prevent the leaching of radionuclides from the waste. If no large centralised disposal site can be constructed in a short time, the waste will have to be placed in a large number of smaller sites.

Radiation protection and safety of workers

The external radiation exposure in reclamation work does not seem to lead to unmanageably high doses. With a fallout of 1 MBq m^{-2} of ^{137}Cs the doses can be expected to be no more than 20 μGy d^{-1} as long as the reclamation work does not include handling of fallout in a more concentrated form.

The highest exposure rates in urban and suburban areas are found in gardens and a deposition in dry conditions generally gives higher exposure rates compared to wet deposition. Exposure rates due to 1 MBq m^{-2} ^{137}Cs will be about 5 μGy d^{-1}.

Gardens in suburban areas are generally larger than those in urban areas and this gives higher exposure rates. Reclamation work close to large trees and in the woods may also lead to higher doses, in some cases in excess of 20 μGy d^{-1}. However, limitations in the exposure time to avoid exceeding the equivalent dose limit of 0.5 Sv, will generally not be necessary with this level of fallout.

The time required to do reclamation work in an urban area could roughly be estimated by the following rule of thumb: The person-power-days in reclamation = the number of persons living in the house to be decontaminated. The doses from reclamation of a medium-sized multi-storey building can roughly be estimated to be 300 μGy for the specified deposition.

A rule of thumb to estimate the time required to do reclamation work in a suburban area is: 5 person-power-days in reclamation work for each house. The doses from reclamation of one house would, in the specified case, not be more than 100 μGy.

As an example of agriculture reclamation work, Table 5 summarises key procedures and parameters involved whilst moving a harvested crop from a field to a waste deposit area. The doses to a larger team of workers are compared to the doses to a smaller team doing the same job as the larger team but in a longer period of time. The question considered is thus which team size will be more efficient from a radiation protection point of view.

Table 5. *Summary of example cases where two teams of workers remove contaminated crops from a field deposited with 1 MBq $^{137}Cs\ m^{-2}$, to a waste deposit area.*

	Team of three workers		Team of two workers	
Radiation from ground during work on a tractor (shielding 0.4)	0.8 μGy h^{-1} (to each worker)		0.8 μGy h^{-1} (to each worker)	
Weight of crop	200 kg m^{-3}		200 kg m^{-3}	
Hay cart load	30 m^3, 6000 kg		18 m^3, 3600 kg	
Time per hectare	1 h		2.5 h	
	Wet deposition	Dry deposition	Wet deposition	Dry deposition
Activity in the load on the hay cart	1.8 GBq	3.6 GBq	1.1 GBq	2.2 GBq
Exposure dose rate from load on hay cart to driver	2.5 μGy h^{-1}	5.0 μGy h^{-1}	1.9 μGy h^{-1}	3.8 μGy h^{-1}
External collective dose for 8 h work	30 μGy	40 μGy	20 μGy	30 μGy
External dose per worker for 8 h work	10 μGy	13 μGy	10 μGy	15 μGy
External collective dose per hectare	3.6 μGy	5.0 μGy	6.5 μGy	9.0 μGy
External dose per worker per hectare	1.2 μGy	1.7 μGy	3.3 μGy	4.5 μGy

Note: In table 5 a three-person-team is compared to a two-person-team. There was actually one more person involved in each of the two examples from the agricultural area above. Or, to be more correct, one more task was handled, namely the first step of cutting the grass prior to moving the crop from the field to the waste deposit area. The person doing this would, however, be doing the same job in the same period time in both examples. Thus the doses received will be the same (0.8 μGy h^{-1} or 6.4 μGy d^{-1}) in both cases and this task will not influence the comparison. It was therefore excluded from the comparison table.

A conclusion from these examples, with a deposition of 1 MBq ^{137}Cs m^{-2}, is that the doses from agriculture reclamation work on a 100 hectare field that will take about 12 working-days are estimated to be roughly 500 μGy.

Precautions to avoid inhalation and intake of radioactive material should always be considered to avoid doses from internal radiation. This should not be too difficult to achieve by using respiratory protection and observing precautions such as not smoking or consuming food during the work to avoid intake of radioactive material. Finally people should shower and change to clean clothes after reclamation work.

From a radiation protection point of view it can thus be concluded that it is better to use a larger team in reclamation work if their work is more efficient per person-hour compared to a smaller team. If the working conditions are comparatively the same for a large team as for a small team then the shorter time it takes for the large team to complete the work leads not only to smaller individual doses but also to a smaller collective dose.

Resources available in the society
There are resources available in all Nordic societies to conduct reclamation operations if necessary. In all Nordic countries at least some organisations exist that have staff with relevant knowledge and some equipment which could be a part of a decontamination operation. Public and private agricultural machinery could also be an important resource. Societies planning and preparedness for reclamation should meet realistic demands for early actions and outline a cost-effective strategy that implies reasonable use of personnel and equipment resources. Some short term planning before an accident is necessary, but the most practical and helpful long term planning will have to be completed after the fallout. Authorities with responsibilities for reclamation must ensure that research studies are in progress and that there are sufficient competent persons available to maintain the planning process and emergency preparedness at an adequate level.

An organisation for decontamination operations should be set up with all the competence needed for the necessary labour, welfare and cost-efficient operations. To keep society prepared, decision-makers and the organisations chief staff, experts and other key-personnel need information, education and retraining. Exercises on regional, national and bilateral level should also be considered.

Information to the public and specifically to farmers should be given in advance to make the public prepared to make their own decisions and ensure maximum co-operation with the authorities.

PART I

THE URBAN ENVIRONMENT

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1. Contamination and reclamation in the urban environment

Since the majority of the population of Nordic countries, and indeed most of Western Europe, reside in towns and cities, decontamination and reclamation of urban areas must figure prominently in nuclear accident contingency planning.

If clean-up is to be both efficient and cost-effective a number of factors must be taken into account. They are:

- distribution of the deposited radionuclide(s) on the various urban surfaces (roofs, soil, walls, roads etc.)
- radiation levels on the various surfaces
- attenuation of radiation through shielding by urban structures (e.g. walls)
- radioactive characteristics of the contamination
- habits of the populace with respect to time spent indoors and outdoors and time spent on various floors within buildings typical of particular urban complexes
- decontamination by natural processes, described as weathering (which includes rain, traffic, routine cleaning)
- diminution in radiation levels through radioactive decay
- decontamination achievable by artificial means
- availability of equipment and machinery for cleaning surfaces
- methods of waste disposal
- potential for resuspension and deposition of contaminated particulate elsewhere.

2. Factors affecting distribution and levels of contamination in urban areas

2.1 Important radionuclides

Of all the radioactive materials which might be released in the event of a severe accident, ^{134}Cs and ^{137}Cs would present the greatest radiation hazard to the populace of a contaminated urban complex in both the medium and long-term.

2.2 Deposition, distribution and retention on urban surfaces

Airborne pollution, whether in the form of gases or particulates, arrives at the earth's surface either by dry deposition (i.e. in the absence of precipitation), wet deposition (in the presence of precipitation) or through occult deposition(in foggy conditions).

The distribution of radioactive fallout between the various components of an urban complex (e.g. walls, roofs, roads etc.) will depend largely on two factors, namely:

i) the deposition process (i.e. dry or wet)

Since about 1986, sufficient experimental observations have been made to make rather crude predictions regarding the behaviour of fallout under both dry and wet fallout conditions. Little is known about deposition in or on snow.

ii) the profile of the urban complex

Both the height and distribution of buildings will be important factors determining the deposition pattern.

2.2.1 Dry deposition

Since 1986, studies of the behaviour of fallout from nuclear weapons testing and from the Chernobyl accident in 1986 have provided an insight into where and for how long fallout will persist in the urban environment. In the Nordic countries, many of the studies have been conducted at the Risø National Laboratory in Denmark.

Soon after a radioactive cloud from Chernobyl passed over the city of Roskilde in Denmark in 1986, various radionuclides were identified and measured on a variety of urban surfaces in and around the city. The prevailing weather at the time of deposition was dry, stable and the windspeed about 3 m s^{-1}. The amount of each radionuclide on surfaces was measured (per unit area) and expressed relative to that found for paved areas (Table 1.1).

Table 1.1 *Deposition on various urban surfaces relative to deposition on paved areas*

Radionuclide	I	Cs	Ru	Ba	Ce	Zr
Paved areas	1	1	1	1	1	1
Walls	0.6	0.2	0.1	0.1	0.1	0.2
Windows	0.5	0.1	0.04	0.04		0.2
Grass (clipped)	5	6	1.1	1.2	1.0	1.0
Trees	17	10	7	6	6	13
Roofs	7	4	1	12	13	

Table 1.1 shows that the walls intercepted little fallout (per unit area), whereas trees and roofs intercepted about an order of magnitude more than horizontal paved areas.

2.2.2 Wet deposition

Precipitation scavenging, or washout of particles and gases, from the atmosphere can make a significant contribution to deposition. In the case of Chernobyl fallout, it led to areas of high deposition even at large distances (> 2000 km) from the reactor site and areas with wet deposition were generally much more contaminated than those receiving only dry deposition in the same vicinity.

Run-off is a term used to describe deposited rainwater which is not retained on the area receiving the rainfall. As run-off water can retain and carry away some of the radioactive material falling on impervious surfaces such as roads and roofs it needs to be evaluated in accident consequence assessments. The amount of radioactive material retained is clearly a factor to be considered in the assessment of dose. Equally, the fate of the contamination carried away in the run-off should also be considered.

Total run-off consists of surface run-off and infiltration, where infiltration is the flow of water through the surface. Infiltration of soils is often high, but construction materials in the urban environment are generally sufficiently impervious to prevent infiltration. For these impervious surfaces the following equation is valid:

$$Q = P - I_a$$

where Q is the direct run-off in mm,
P the total rainfall in mm, and
I_a the initial accumulated rainfall in mm prior to run-off.

Roed (1987) showed that the amount of run-off from roofs was highly influenced by the type of surface material. For rainfall (P) of 9.2 mm shortly after the Chernobyl accident he found that for roofs with a pitch of 45^0, the I_a values were 1.8 mm for cement tile, 4.2 mm for red clay tile, 1.4 mm for eternite (an asbestos type of material) and 0 mm on silicone treated surfaces. At the same time, Roed measured the concentration of several radionuclides in run-off water and rain water and comparisons of rainfall and run-off water are given in Table 1.2.

***Table 1.2** Concentration of radionuclides in run-off water relative to that in rainwater for a precipitation event of 9.2 mm.*

Surface type	Radionuclide		
	Cs	**Ru**	**Ba**
Cement tile	0.49	0.56	0.40
Red tile	0.55	0.65	0.58
Eternite	0.14	0.30	0.37
Silicone treated eternite	0.74	0.52	0.67

In some recent experiments at Risø, a similar situation was observed for run-off of radiocaesium falling on road surfaces. For a rainfall of 6 mm, an I_a of 3.8 mm was observed for asphalt and 3.4 mm for concrete. The ratio of the activity concentration of ^{137}Cs in run-off water to that in rainwater was 0.16 for red tile and 0.21 for concrete tile.

The retained wet deposition is defined as the amount of radioactive material retained on a given surface after precipitation has stopped.

The distribution of material deposited on walls by wet deposition is dependent not only on run-off but also on windspeed and wind direction since the amount of rainwater striking the wall is dependent on the angle of incidence.

2.3 Seasonality

Deposition and retention of radioactive fallout on various urban surfaces will vary according to the time of year. This dependence on time of year is known as 'seasonality'. For example, the pattern of contamination on the ground resulting from deposition in snow will be different from that formed by rain. Similarly, deciduous bushes and trees will intercept fallout most effectively when in leaf. Other meteorological conditions, such as wind speed and intensity of rain, which generally follow a seasonal pattern, will also influence the deposition pattern.

3. Dose reduction by natural processes

3.1 Weathering

Following the initial interception of fallout by an urban surface, the surface will continue to be exposed to the weather. Weathering (the action of rain, snow, frost) will tend to displace the adsorbed radioactive material. Sometimes, pedestrian and road traffic and routine street cleaning are also included under the heading of weathering.

Two years after wet deposition of fallout from Chernobyl on the town of Gävle in Sweden, Roed and Sandalls (1989) redetermined the distribution of radiocaesium on walls, roads, paved areas and grassed areas. The measurements were made in the town centre and in a nearby industrial area. The relative amounts on the different urban surfaces immediately after deposition and 2 years later, are shown for radiocaesium in Table 1.3. Table 1.4 shows the relative amounts of radioiodine, wet deposited on different surfaces immediately after deposition.

Table 1.3 *Distribution of wet deposited radiocaesium relative to grassed areas for a precipitation event of 5-10 mm.*

Surface	Immediately after deposition	2 years after deposition
Grassed areas	1	1
Road pavings with		
a) heavy traffic	0.4-0.8	0.01 - 0.05
b) light traffic	0.4-0.8	0.05-0.2
Roofs	0.3-0.9	0.1-0.7
Walls	0.001-0.03	0.01-0.03

Table 1.4 *Distribution of wet deposited radioiodine relative to grassed areas immediately after a precipitation event of 5-10 mm.*

Surface	Radioiodine levels immediately after deposition
Grassed area	1
Paved area	0-0.03
Roofs	0-0.04
Walls	0.01-0.03

4. Forced decontamination of urban surfaces

Various methods have been tested for physically removing radioactive contamination from urban surfaces and some of the more successful will be described here.

4.1 Street cleaning with brushes and vacuum cleaners

The efficiency of street cleaning methods is strongly dependent on the dust loading and the nature of the surface. Urban surfaces generally carry a burden ofparticulates ranging from sub-micron up to 1000 μm in diameter. The dust loading and the nature of the surface are important factors affecting the decontamination efficiency of the different methods.

Sartor et al (1957) found that the efficiency of sweeping with a brush in removing dust from streets was 15% for particles of less than 43 μm diameter, the overall efficiency for all particles was about 50%. The following equation relating particle size to efficiency of removal was derived:

$$M = M_2 + (M_0 + M_2) \cdot e^{-kE}$$

M is the amount of contamination on the surface of the street after sweeping, M_0 the amount of contaminant before sweeping, E the effort in using the equipment (relating to time per unit area) and M_2 and k are dimensionless constants depending on sweeper characteristics, particle size and street surface.

The studies of Clark and Cobbin (1964) indicated that the efficiency of sweeping is sensitive to particle size and initial mass loading in such a way that the methods would be inefficient for particle sizes smaller than 20 μm and for loadings below about 11 g m^{-2}.

Using a rotating broom sweeper Menzel (1962) removed about 70% of the contamination from moist soil carrying a thin cover of a grass species called Fescue. A second sweeping removed about 90% of the contamination.

The use of a stiff street broom on an asphalt and a concrete road with a dust loading of 50 g m^{-2} by Roed & Sandalls (1990) failed to remove very small dry deposited particulates contaminated with radiocaesium. This was consistent with the findings of Clark and Cobbin (1964). With a dust loading of 200 g m^{-2}, 40% was removed from the asphalt road and 57% from the concrete road.

Calvert et al., (1984) attempted to remove road particles by sweeping with an "improved" vacuum sweeper: the overall efficiency for the pick-up head was about 90% for particles smaller than 2 μm.

By fire-hosing roof material, Owen et al (1960) found decontamination factors of more than ten on flat tar and gravel roofs, and Miller (1960) obtained a decontamination factor of three on a concrete roof and a shingle roof treated two days after contamination.

Gjorup et al (1982) used brushes and vacuum cleaners to remove aged ^{137}Cs fallout but the treatment had no effect on red clay tiles. On corrugated asbestos a DF (decontamination factor, defined as the contamination level before relative to that after decontamination) of about two was achieved.

An overall conclusion is that mechanical removal of contaminated street dust may significantly reduce the contribution to dose rate from road surfaces which have a high dust loading, but little or no effect can be expected at dust loadings of less than 50 g m^{-2}.

4.2 High pressure water-hosing

A high-pressure washer (28-48 Bar) has been used to remove plutonium particles (0.8 mm) from asphalt and concrete surfaces (Dick and Baker, 1961). The decontamination factor on asphalt was 10-12 and on concrete 4-40, by practically immediate application.

Using a KEW Powerforce 1002 K high pressure water jet with a maximum nozzle pressure of 100 Bar, a decontamination factor of 2.2 on Chernobyl radiocaesium contamination was found by Roed & Sandalls (1990). In further tests, Roed and Andersson (1993) used the KEW high pressure cleaner to remove radiocaesium from roofs in the CIS countries (former Soviet Union) and recorded decontamination factors of 1.3-2 depending on the amount of moss and algae present. The more moss and algae per unit area the greater the efficiency of decontamination.

As a conclusive remark relating to these references it can be said that high pressure water hosing can lead to a large reduction in contamination level, if applied very rapidly, but as 'natural' weathering will relatively quickly reduce the level of contamination on a road paving, the method is significantly less effective when applied in inhabited areas after a few years.

4.3 Cutting and removing grass

When fallout is deposited onto grass, as in the case of dry deposition, most of it will eventually be transferred to the surface of the soil, unless the grass is cut and the contaminated grass removed. Krieger and Burmann (1969) found that the transport process from grass to soil had a half-life in the order of 7-18 days. However, if the grass is cut and removed immediately following dry deposition, decontamination factors of 2-10 can be achieved (CEC 1991).

4.4 Pruning trees and bushes

This can be a highly effective means of reducing radiation levels in the environment but the reduction per unit mass of biomass will depend on how effectively the plant has intercepted and retained the contamination.

4.5 Digging small gardens

Digging gardens to a depth of one spade can reduce gamma radiation levels at the soil surface by a factor of about six (Gjorup et al. 1982). This is an attractive countermeasure since it can be performed easily and does not require special tools.

4.6 Ploughing large gardens

A dose reduction factor of about 15 can be achieved by ploughing to a depth of 30 cm (Roed 1982). However, subsequent ploughing may return much of the radioactive contamination to the surface. Deep-ploughing to a depth of 45 cm can reduce the dose rate at the soil surface by a factor of 20 (Roed 1982).

4.7 Removal of layer of soil

Various ordinary types of earth-moving equipment have been used to remove contaminated soil. These include graders, bulldozers and pan-type scrapers (Menzel et al. 1961; Menzel 1962; Owen, 1965). The reported dose reductions were typically 80-90% by removal of roughly the top 5 cm. The results are in reasonable agreement with those obtained by Melin et al. (1991) near the Chernobyl power plant. In these experiments, a dose rate reduction of about 85% was achieved by scraping to a depth of 20-50 cm in a radius of 10 m around the point of measurement. A disadvantage of surface soil removal as a means of decontamination is the generation of large amounts of radioactive waste (cf. part 6).

4.8 Washing hard surfaces with ammonium nitrate solution

Under ideal conditions, washing hard surfaces with NH_4NO_3 solution (0.1 M) gave decontamination factors of 1.5-4 (Sandalls, 1987). The treatment, however, must be carried out shortly after the deposition. Roed & Sandalls (1989) showed that 5 years after contamination, only about 15% could be removed from brick and tiles using a solution of NH_4NO_3 (0.1 M).

4.9 Steam-cleaning

Steam-cleaning of radiocaesium contaminated Fletton bricks, concrete paving slabs, concrete tiles and clay tiles has not been found to be effective (Sandalls, 1987). Only hard blue engineering bricks showed a significant loss (40%).

4.10 Sand-blasting

Sandalls (1987) used dry sand-blasting on various urban surfaces and obtained decontamination factors of 1.5 - 100 on brick and stone. Roed (1985) used wet sandblasting on clay roof tiles contaminated with radiocaesium from the Chernobyl fallout and removed two-thirds of the contamination.

Roed (1992) removed about 80% of Chernobyl contamination on asbestos roofs in the former Soviet Union by wet sand blasting 6 years after deposition occurred.

4.11 Road planing

By removing a thin layer of asphalt from a road surface using a road planer, Barbier et al. (1980) obtained a decontamination factor of more than 100.

Road planing is generally a very effective method for removal of contamination, since it removes the surface to which the contaminants are attached. However, the method is rather expensive and not a very fast way to decontaminate.

5. An outline strategy for dose reduction in the contaminated urban environment

The ultimate goal of a reclamation/decontamination study is the provision of a contingency plan for reducing radiation doses to man through reclamation and decontamination. In developing such a strategy, many factors need to be considered in order to provide the most cost-effective strategy for any given scenario.

Some of the important factors to be considered in formulating strategic countermeasures are:

1) Distribution of the deposited material with respect to the different outdoor surfaces.
2) The contribution of the different surfaces to dose rate.
3) The decontamination or dose reduction achievable on the individual surfaces using appropriate methods.
4) The practicability of the various reclamation/decontamination procedures.

The central part of a town normally consists of tall buildings, extensive paved areas and a limited amount of green areas. In contrast, residential suburbs have smaller buildings, gardens with trees and bushes and a limited amount of paved areas.

Within an urban area, various discrete components occur (e.g. walls, paved areas, roofs, grassed areas, etc.) and the individual contributions to overall dose of each of these components will depend on their surface area, the amount of radioactive material retained, the energy of the radionuclides present and the degree of shielding. As part of the procedure for reducing radiation dose to the populace of a given urban area the physical characteristics of the area need to be determined initially in some detail. This should include determination of the size of buildings, thickness of the walls, type of roofs, extent of grassed and paved areas, amount of trees, etc.

From a knowledge of the prevailing weather during deposition, and of the content of the radioactive plume, the relative distribution of radioactive contamination on the different surfaces can be estimated. An effective source strength can then be defined. Wet deposition and dry deposition will, for example, give different effective source strengths for the various surfaces in an urban area.

Having defined the source strength, the next step is to calculate the relative dose rate at different locations (indoors and outdoors) due to deposition on the different urban surfaces (roofs, walls, paved areas, trees, bushes, etc.). The mean relative dose rate to a member of the local populace can then be estimated taking into consideration the time that they will spend in the different locations.

The next step is to estimate the decontamination factors achievable for the various surfaces and, from this, to estimate what the relative source strengths will be after decontamination. The new dose rates at the various locations can then be calculated to show the reduction in dose rate achievable through decontamination.

From the costs of the various decontamination methods, and the corresponding achievable reduction in dose rate, the cost of a fractional reduction in dose rate for the various types of surfaces can be calculated. A comparison of these data will then indicate the most cost-effective means of dose reduction.

Case studies
The following are examples of how to develop a strategy plan for dose reduction in four typical urban complexes. Examples are given for:

1) detached houses in a suburban area
2) two-storey semi-detached houses
3) rows of two-storey terrace houses
4) multi-storey blocks of flats.

The cost and effectiveness of different decontamination procedures are given in Table 1.5. The procedures are suitable for most urban environments.

***Table 1.5** Cost and Effectiveness of Procedures for Removal of Radiocaesium*

Surface	Procedure	DF or DRF	Cost (ECU per m^2)
Windows	Cleaning	10	2
Asphalt, concrete	Sweeping	1-5	0.1
Asphalt, concrete	Vacuum sweeping	1-5	0.04
Grass	Cutting	2-10	0.016
Asphalt, concrete	Water jets	1-10	0.1
Roofs	--- " ---	1-5	3
Walls	--- " ---	1	1
Roads (asphalt)	Planing	>100	3
Walls	Sandblasting	>100	10
Roofs	-	3-100	20
Grass and soil	Removal of surface	4-10	0.2
Trees	Cutting/defoliating	10-100	7
Garden	Digging	6	1
Fields and parks	Ploughing	15-50	0.1

It can be seen that the decontamination factors (DF) or the dose reduction factors (DRF) vary considerably between methods. The wide ranges quoted for DFs for many of the methods arises because the decontamination achievable is often dependent on the circumstances in which the contaminated occurred, i.e. wet or dry deposition, the amount and intensity of rainfall at the time of wet deposition, how much rain has fallen since deposition occurred, etc. It is necessary therefore to be able to estimate what can be achieved in terms of dose reduction for the individual surfaces after taking these different factors into account. This has been done for both wet and dry deposition and the results are included in Table 1.5. The Table also shows the cost of treating each of the surfaces.

Table 1.6 shows estimates of typical source strengths before and after decontamination. The relative source strengths given in the table are relative to deposition on very short grass where it is assumed that all the deposited material is retained on the grass and none on the soil.

Table 1.6 *Relative source strengths before and after decontamination.*

Surface type	**Dry deposition: before/after decontamination**	**Wet deposition: before/after decontamination**
Walls	0.100/0.050	0.010/0.010
Roofs	1.000/0.500	0.400/0.300
Garden	1.000/0.100	0.800/0.100
Street	0.400/0.200	0.500/0.300
Trees	3.000/0.100	0.100/0.010
Indoors	0.020/0.010	-

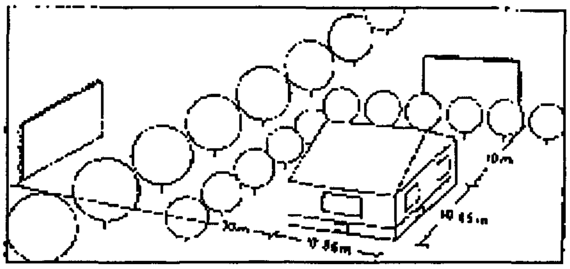
Assuming these relative source strengths before and after decontamination the relative contribution to dose rate in four different urban areas of varying population density has been calculated (Jacob and Meckback, 1987). An example of the type of results obtained by this method is given in Table 1.7. The Table shows the contribution to dose rate at different locations from the different sources such as walls, gardens, trees, etc.

The location factor is defined as the dose rate at the specific location relative to that on an infinite grass surface where all the deposited matter is retained on the grass. The pre-decontamination, location factor is given, in the last row of Table 1.7.

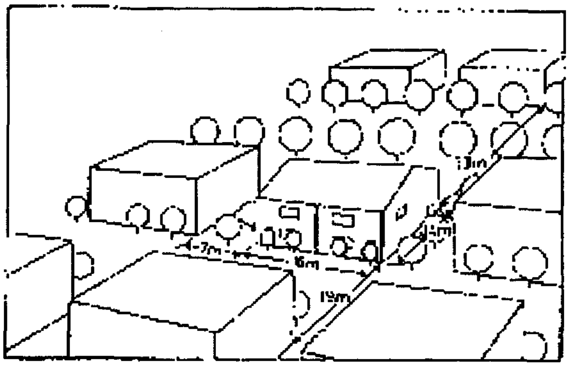
Table 1.7 Fractional contribution (%) to dose rate at different locations in and around terrace houses. Dry deposition, before decontamination. Source energy: 300 keV.

Deposition surface	Relative source strength	Ground floor	First floor	Attic	Street	Garden
Windows	0.05	1.2%	2.4%	0.0%	0.1%	0.1%
Walls	0.10	1.6%	2.0%	0.1%	1.5%	1.7%
Roof	1.00	0.3%	6.8%	74.2%	1.3%	0.4%
Basement windows	0.05	0.0%	0.0%	0.0%	0.0%	0.0%
Light Shafts	0.40	0.1%	0.0%	0.0%	0.0%	0.1%
Neighbouring walls	0.10	0.9%	1.3%	0.3%	1.2%	1.2%
Neighbouring roofs	1.00	1.2%	4.0%	4.6%	1.6%	0.9%
Garden	1.00	36.3%	24.1%	3.5%	31.5%	60.1%
Street	0.40	4.0%	2.7%	0.4%	26.8%	0.6%
Ground beyond building	1.00	5.8%	12.9%	11.0%	9.0%	5.6%
Trees	3.00	48.7%	43.8%	5.9%	26.8%	29.3%
Location Factor		0.07	0.04	0.32	0.63	0.94

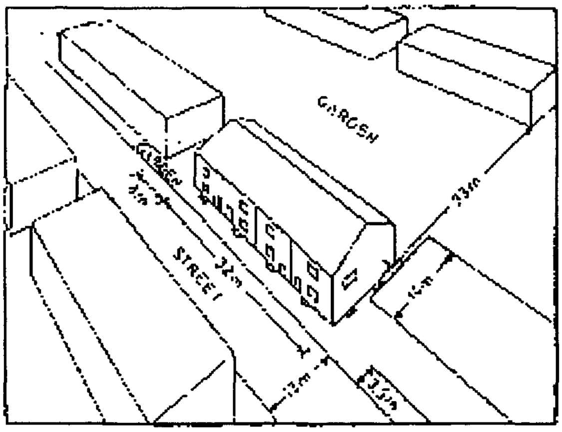
The four simulated environments are shown in Fig 1.1.



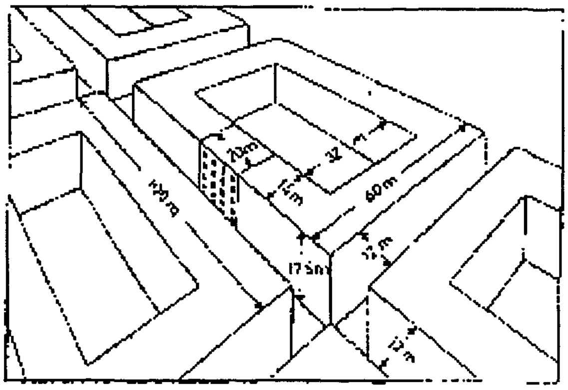
Single-storey detached house



Two-storey semi-detached houses



Rows of terrace houses (2 stories)



Multi-storey blocks of flats

Figure 1.1 *The four urban environments as simulated by Jacob and Meckback*

Table 1.8 shows estimated achievable decontamination factors based on Risø's experimental data, together with estimated costs per unit area, if, for instance, walls and roofs are fire-hosed, streets are vacuum-swept, trees are cut down, gardens are dug and internal surfaces cleaned by normal domestic cleaning methods.

Table 1.8 *Estimated achievable decontamination factors and costs*

Surface type	**Roofs**	**Walls**	**Streets**	**Trees**	**Garden**	**Internal**
Efficiency (DRF):						
Dry deposition	2	2	2	50	10	2
Wet deposition	1.3	1	1.7	10	8	-
Costs ECU m^{-2}	3	1	0.04	7	1	1

If it is considered that the average person living in one of the four urban areas spends 85 % of their time indoors (distributed equally in time between the different floors), 10 % in the garden and 5 % on the street, the location-averaged dose rates from the different contaminated surfaces can be calculated. This has been done in Table 1.9, which shows the contribution to dose rate in the four environments, from wet and a dry deposition of 1 MBq m^{-2} on grass, using the relative source strengths given in Table 1.6. The costs and efficiency of the chosen decontamination/reclamation procedure have been calculated from the data in Table 1.8.

From Table 1.8. it is seen that generally, prior to decontamination, the garden and the trees are the main contributors to the dose from a dry deposition. Roofs also seem important, especially in environments of smaller houses, whereas the streets become more important in areas of high population density. It is seen that the employment of the proposed inexpensive and practicable countermeasures for gardens and trees can reduce the dose rate by about a factor of 4. In the wet deposition case, the dose contributions from ground deposition on gardens and streets are the most important. Reclamation of the garden areas alone generally gives a dose reduction by a factor of about 4.

__Table 1.9__ Dose rates and countermeasures example. Location averaged dose rates from a deposition corresponding to 1 MBq m^{-2} dry deposition on grass. Source energy 662 keV.

	Roofs	**Walls**	**Streets**	**Trees**	**Garden**	**Internal**
1) Single-storey detached houses						
Dose rate contribution from different surfaces [μGy d^{-1}]:						
DRY: WET:	6.13 2.45	4.95 0.12	- -	17.91 0.60	26.47 21.18	0.96 0
% dose reduction by decontamination/reclamation of the surfaces:						
DRY: WET:	5.43% 2.52%	4.40% 0	- -	30.83% 2.21%	42.33% 76.11%	0.85% 0
Costs per person per % dose reduction [ECU]						
DRY: WET:	16.30 35.17	7.43 -	- -	12.05 168.13	3.73 2.07	76.05 -
2) Two-storey semi-detached houses						
Dose rate contribution from different surfaces [μGy d^{-1}]:						
DRY: WET:	3.32 1.33	0.37 0.04	- -	4.65 0.16	9.45 7.56	0.96 0
% dose reduction by decontamination/reclamation of the surfaces:						
DRY: WET:	8.94% 3.66%	0.96% 0	- -	24.30% 1.54%	45.50% 72.85%	2.59% 0
Costs per person per % dose reduction [ECU]						
DRY: WET:	10.74 26.25	45.38 -	- -	8.14 128.69	2.09 1.30	34.67 -
3) Rows of terrace-houses (2 stories)						
Dose rate contribution from different surfaces [μGy d^{-1}]						
DRY: WET:	1.21 0.35	0.29 0.03	1.31 1.25	2.94 0.13	5.66 4.53	0.96 0
% dose reduction by decontamination/reclamation of the surfaces:						
DRY: WET:	5.05% 1.40%	1.19% 0	5.47% 7.96%	23.93% 1.90%	42.76% 62.94%	4.06% 0
Costs per person per % dose reduction [ECU]:						
DRY: WET:	18.82 68.48	32.79 -	0.24 0.15	4.24 52.00	2.35 1.56	27.85 -
4) Multi-storey blocks of flats (5 stories)						
Dose rate contribution from different surfaces [μGy d^{-1}]:						
DRY: WET:	0.05 0.02	0.27 0.03	1.77 2.22	1.95 0.07	4.00 3.21	0.96 0
% dose reduction by decontamination/reclamation of the surfaces:						
DRY: WET:	0.23% 0.09%	1.60% 0	10.32% 16.02%	22.31% 1.06%	43.06% 50.71%	5.69% 0
Costs per person per % dose reduction [ECU]						
DRY: WET:	172.31 552.06	48.44 -	0.07 0.05	0.27 5.60	0.47 0.39	20.79 -

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PART 2

THE CULTIVATED AGRICULTURAL ENVIRONMENT

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7. Contamination and Reclamation in the Cultivated Agricultural Environment

Local agricultural practices in the Nordic countries has evolved over many years. To a large extent, these practices and the crops grown have been chosen for their suitability to local conditions such as climate and soil type. From time-to-time, market demands and political decisions have had a marked influence on the choice of crops. These factors have resulted in a great diversity in agriculture in the Nordic countries. The diversities mean that in the event of contamination of agricultural land by radioactive fallout the consequences may differ greatly from region to region. For crops and soils contaminated directly by radioactive fallout there are five primary causes for concern, namely:

1. short-term internal contamination of man and animals through ingestion of surface-contaminated mature crops
2. internal contamination of crops through foliar intake
3. contamination of mature crops from resuspended soil
4. direct irradiation of agricultural workers, and
5. internal irradiation from inhalation of resuspended soil particulates.

In the short-term, most of the radionuclides likely to be released to the atmosphere in the event of an accident have a potential to cause problems in agriculture and many have the potential for causing long-term problems.

Generally, the magnitude of the problems created will depend on the:
- deposition mechanism (wet or dry)
- radionuclide composition of the fallout
- type of farming system (i.e. arable or dairy)
- type of soil (for instance organic soils are more sensitive than mineral soils with respect to radiocaesium)
- state of development of the crop which in turn is determined by the season of the year.

8. Factors Affecting Magnitude and Duration of the Contamination

8.1 Deposition and retention on soils and crops

Fallout will be deposited either on bare soil, or on surfaces of a growing crop and underlying soil. The amounts of fallout intercepted by the growing crop will depend on factors such as whether the deposition was dry or wet and, where wet deposition occurs, on the amount and intensity of the rainfall or other precipitation.

8.1.1 Seasonality

The consequences of a radioactive release to the atmosphere can be critically dependent on the time of year and the term used to describe this is «seasonality». Seasonality means the varying response to radioactive contamination of environmental samples according to the time of the year when contamination occurs. It has been shown for barley, for example, that the fractional initial intake three months before harvest is about 5% of the deposited activity but if contamination takes place one month before harvest, up to 36% of the contamination may be taken up by the crop. If a field of barley is contaminated with ^{137}Cs at a rate of 1 kBq m^{-2} three months before the harvest the mature grain will contain 2 Bq kg^{-1} (dw) but if the contamination occurs one month before the harvest, the concentration in grain will be 100 Bq kg^{-1} (dw) (Aarkrog, 1983). Similarly, had a Windscale-type accident occurred in a Scandinavian country at a time where dairy cattle were indoors, the ^{131}I contamination in milk would have been of little or no consequence. Seasonality has been discussed in detail by Aarkrog (1975); Eriksson et al. (1976); Eriksson (1991) and Aarkrog (1992).

8.1.2 Natural Field Losses

The 'natural field loss factor' is defined as the reduction of contamination on crops caused by processes other than physical radioactive decay. For contamination on grassland in the growing season, regardless of whether the deposition was wet or dry, the average field loss half-life is about 13 days when the activity is expressed per unit area and about 8 days if expressed per unit mass of vegetation, since growth of the sward dilutes the contamination.

In addition to the natural field losses, the amount of radioactive fallout on crops will also decrease as a consequence of physical radioactive decay. If both factors are taken into account the effective ecological half-life on the crop will be given by:

$$\text{Effective ecological half life} = \frac{\text{Field loss half life} \cdot \text{radioactive half life}}{\text{Field loss half life} + \text{radioactive half life}}$$

The field loss factor has been discussed by Chamberlain (1970); Eriksson (1977, 1991); Miller and Hoffman (1983); Hoffman et al. (1989) and Chamberlain and Garland (1991).

Short- term problems after a nuclear accident are outside the scope of this book. It is sufficient to state that direct contamination of leafy vegetables and fruit by short- and long-lived radionuclides may occur. Even where the radionuclides are long-lived, the problem will be only short term since the product has only a limited life. Similarly, contamination of milk due to ruminants eating surface contaminated vegetation, may lead to initially high levels of radiocaesium, radiostrontium and radioiodine in milk. Furthermore, if the surface becomes contaminated, there may be an external radiation hazard to farm workers, coupled with an internal hazard through inhalation (see Parts 1 and 7).

8.2 Mid-Long-term aspects

8.2.1 Soil-to-Plant Transfer

The type of soil will strongly influence the behaviour of the most mobile and therefore radiologically important radioelements, in particular radiocaesium and radiostrontium. For radiocaesium it has been shown that the soil-to-plant transfer increases with increasing amounts of organic matter (Fredriksson and Eriksson, 1966; Barber, 1964; Mascanzoni, 1988; Eriksson and Rosén, 1991 and Sandalls and Bennett, 1992). A peaty soil containing nearly 90% organic matter had a soil-to-plant transfer factor for radiocaesium which was one order of magnitude greater than that of a brown earth containing 19% organic matter and the differential persisted for at least five years after contamination occurred (Sandalls and Bennett, 1992). Soils with such high proportions of organic matter are not common in the cultivated environment of the Nordic countries but they do occur in natural and semi-natural ecosystems. Soils containing clay minerals have the capacity to "fix" radiocaesium and render it more or less unavailable for plant uptake. This emphasises the importance of taking into account both the levels of radiocaesium contamination on the ground surface and the type of soil contaminated. **It would be wrong to use only levels of deposition as a criterion for implementation of countermeasures.**

Similarly, for radiostrontium contamination, the soil properties can have a profound influence on soil-to-plant transfer but for a very different reason. No naturally occurring soil component appears to have a capacity for "fixing" radiostrontium and rendering it unavailable. The soil-to-plant transfer of radiostrontium is largely governed by the presence of calcium. Plants have a finite capacity for taking up calcium and the transfer factor for radiostrontium will be lower on a calcium-rich soil than on calcium-deficient soil.

In the months following deposition, the severity of the situation and the possible implementation of countermeasures can be carefully assessed. There is generally a wealth of experimental data on the transfer rates for crops and soils typical of Western Europe (IUR, 1982, 1984, 1986, 1987) and the subject of soil-to-plant transfer has been extensively studied in the Nordic countries (Fredriksson et al., 1969a, b; Lönsjö and Haak, 1975, 1986; Eriksson, 1977; Haak, 1983a, b, c, d). More recently, soil-to-plant transfer of the Chernobyl fallout has been studied in Scandinavia (Eriksson and Rosén, 1989, 1991; Rosén, 1991, 1996; Rosén et al., 1996).

The structure and management of agriculture in the Nordic countries is known in detail, and there is a great deal of information on soil-to-plant transfer available. Therefore it should be possible to predict the magnitude of the mid to long-term potential problems arising from land contamination.

9. Natural Migration of Radionuclides in Soils

All the most likely potential long-term radioactive contaminants, including radiocaesium, radiostrontium and plutonium, show very little mobility in soils and movement down the soil profile is very slow. On most undisturbed soils, even the ^{137}Cs fallout from the 1960s nuclear weapons tests is still mostly within the topmost 10 cm of soil. The mobility of radiostrontium is somewhat greater than that of radiocaesium, but even so, downward movement of radiostrontium would not be significantly greater than that of radiocaesium in the first one or two years after an accident. Plutonium, like radiocaesium, hardly moves down soil profiles at all. Generally, any movement down the soil profile is a result of mechanical disturbance from drying and cracking or through the activities of animals, particularly earthworms.

The low mobility of these contaminants means that generally sufficient time for monitoring and mapping of concentrations in soil is available before any countermeasures need to be taken, thereby avoiding precipitate and inappropriate actions. The low mobility in soils also means that many root crops will continue to grow in the undisturbed soil with virtually no contamination of the underground parts of the crop. However, shallow-rooted crops which obtain their nutrients at or near the ground surface may become internally contaminated through root uptake.

10. Disturbance of Deposited Radionuclides Through Cultivation

Normally, a crop rotation system is used on farms in the Nordic countries. The number and sequence of the crops varies from region to region and even from farm to farm. However, at any given time of the year, a given area of land will either:

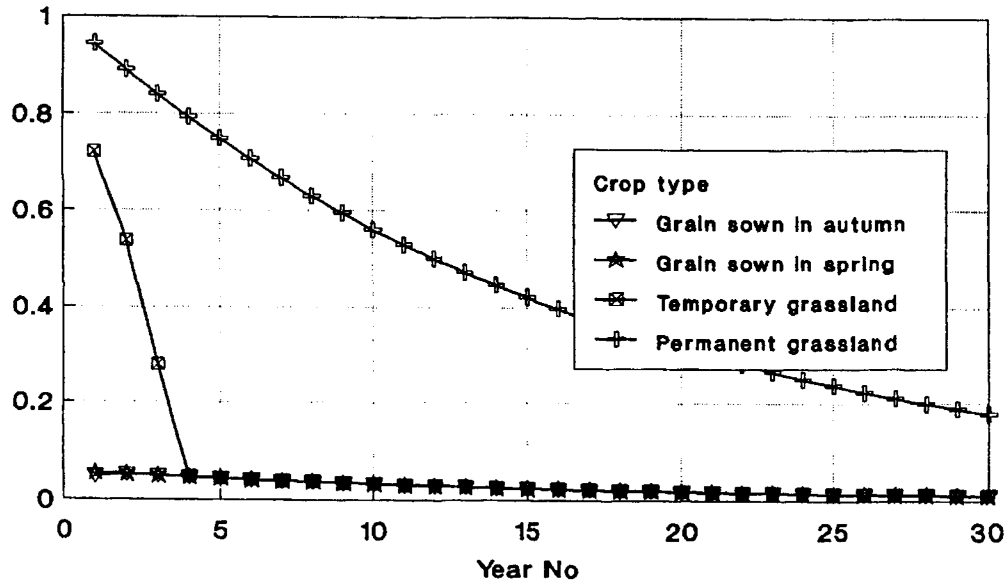
1. be bare, following procedures such as ploughing and harrowing
2. carry a recently sown young crop
3. carry a crop somewhere between the shooting stage and maturity
4. carry the remains of a recently harvested mature crop or
5. be permanent pasture.

When radioactive fallout first occurs, surfaces become superficially contaminated. On permanent pasture the contamination remains on or close to the surface but for arable land the normal cycle of land management will affect distribution. The contamination will either (i) become distributed between the soil surface and the growing crop, or (ii) be largely removed with a harvested crop, or (iii) be mixed into the soil by ploughing. The normal treatment of the land thus gives rise to four different placements of the fallout, namely:

1. on or close to the soil surface on permanent pasture which may persist for many years - penetration to deeper soil layers is slow.
2. in the upper surface layer on temporary grassland, where it is incorporated into the soil by ploughing according to the rotation scheme, one fraction each year. Most of the fallout is incorporated within five years.
3. in the sowing layer (0 - 5 cm layer) after deposition on land prepared for sowing. If grass is sown, this placement lasts until ploughing takes place according to the normal scheme, generally within 3 - 5 years.
4. onto the growing annual crops and underlying soil surface. This placement lasts until harvesting and the autumn ploughing.

The gamma radiation levels from radiocaesium on the surface of the soil is reduced to about one-sixth (Gjörup, 1982) or more (see Part 1) by ploughing. The radiation exposure on fields with the fallout mixed into the sowing layer can be reduced to about one-half (Hedemann Jensen, 1979; Roed, 1982; Lönsjö, 1983a, b).

The dose rate from the fields can be described in models considering the soil types and the crop rotation and the influence of time and normal soil management on the placement and position of the fallout in the soil profile. Relative values obtained from such a model are presented in Figure 2.1 for a sequence of years after deposition.



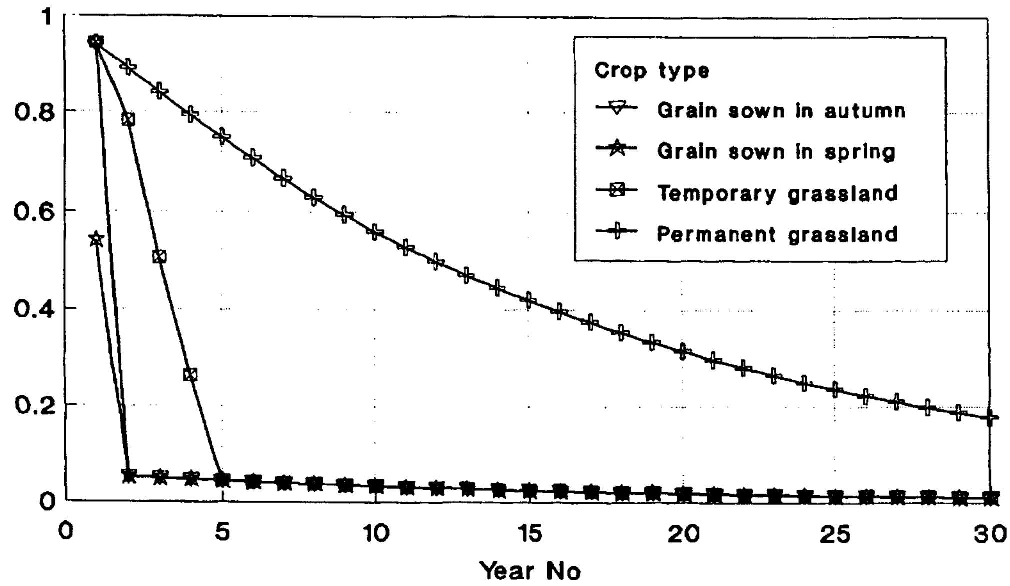
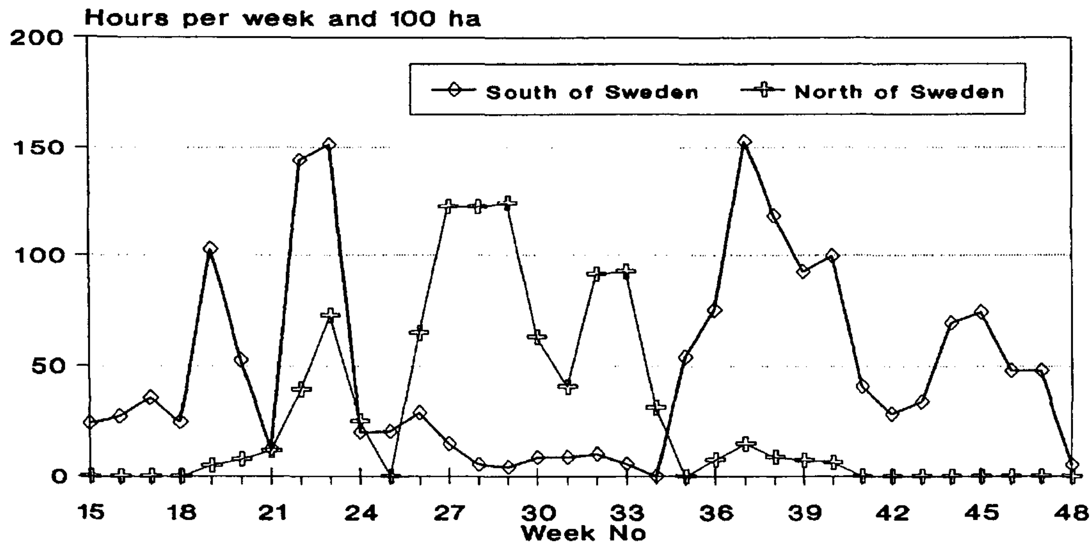


Figure 2.1 *Estimated development for up to 30 years of the dose rate (relative to the level in the year of fallout) on different types of cropland following deposition of ^{137}Cs at different times of the year. Two cases are considered, deposition before and after the normal autumn ploughing (upper and lower figure respectively). The normal ploughing is assumed to reduce the radiation level on the fields by a factor 6. Residence half-time of ^{137}Cs in the soil layers has been set at 20 years. The farmland corresponds to Middle Swedish conditions and the soil is assumed to consists of 60 % loam, 25% sandy and 15% organic soils.*

The extra dose to the workers arising from working in the fields also depends on the time spent in the fields, on the protection given by shielding on tractors and on the difference between «field» doses and «indoor» doses. The time spent in fields varies greatly during the season depending on the geographical situation, on the degree of mechanisation and on the cropping and animal husbandry system of the farms. Figure 2.2 below shows how these factors can influence the range in the weekly working hours per 100 ha of farmland during the season.



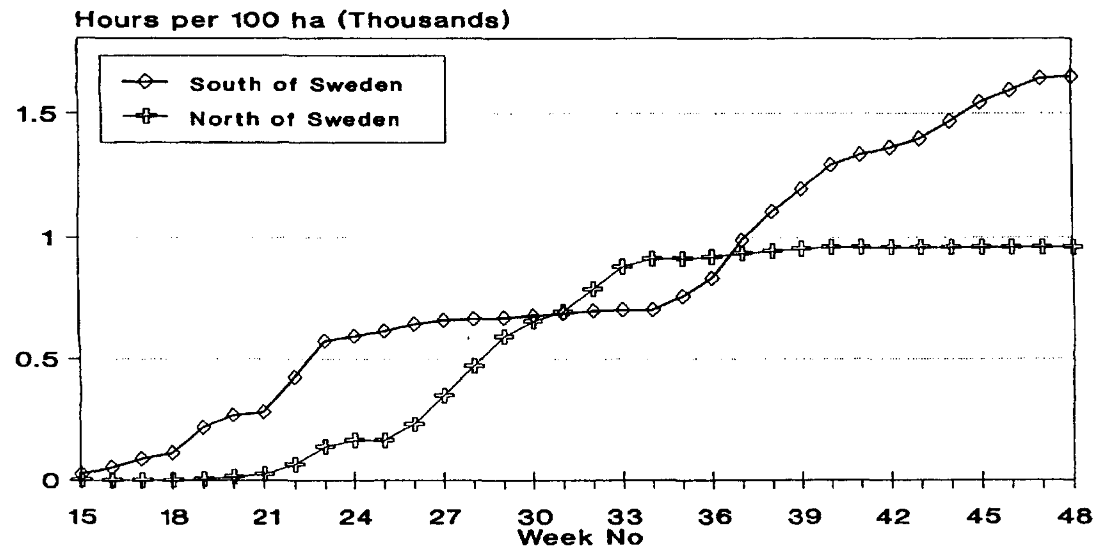


Figure 2.2 *Distribution of working hours in the field per 100 ha of farmland. Example of the difference between regions in weekly working hours (upper figure), and hours accumulated over the season (lower figure), in two different districts of Sweden (based on data from SCB (1990) and SLU (1989)).*

11. Influence of Soil Properties on Soil-to-Plant Transfer

The influence of different soil properties on the plant uptake of radiocaesium and radiostrontium was widely studied in Sweden in the 1960s and 1970s (Fredriksson et al., 1969; Haak and Lönsjö, 1975) and in other countries (IUR, 1982, 1984, 1986, 1987). These studies and more recent investigations showed that the clay content of soils exerted a strong negative influence on the availability of radiocaesium for plant uptake and this explains why uptake on the highly organic and peaty soils is generally much higher than on mineral soils (Sandalls and Bennett, 1992; Rosén et al., 1996a, b).

These studies also confirmed the earlier qualitative findings on the inverse relationship between the calcium content in the soils and the plant uptake of radiostrontium. A low calcium status of the soil gave high radiostrontium uptake and vice versa.

A general conclusion from these studies is that predictions carry large uncertainties because of the complex nature of the soil, the varying placement (with respect to depth) of the contaminant and the variable feeding depth of the plant root systems. The latter is influenced by the content of water in the different soil layers during the season. Another factor which contributes to the uncertainty is the possibility that resuspended fallout on soil particles can influence the total transfer to the crop through foliar intake.

To handle these uncertainties and to obtain simple and acceptable estimates of the expected transfer rate from soil to plant of a radionuclide, it is necessary to develop a soil classification system and then to use average transfer factor values. The classification should depend on easily observable or measurable but dominating characteristics.

Such a system was used by Eriksson (1977) and by Haak (1983 a and b) for estimates of the radiocaesium and radiostrontium migration in agricultural systems at the county level in Sweden. For the radiocaesium transfer, the classification was based on the clay content of the soil with the content of organic matter and sand as known modifying characteristics. Figure 2.3 shows the relationship found between soil clay content and the radiocaesium uptake by plants, for barley grain in a series of field experiments on twelve different soils (Fredriksson et al., 1969).

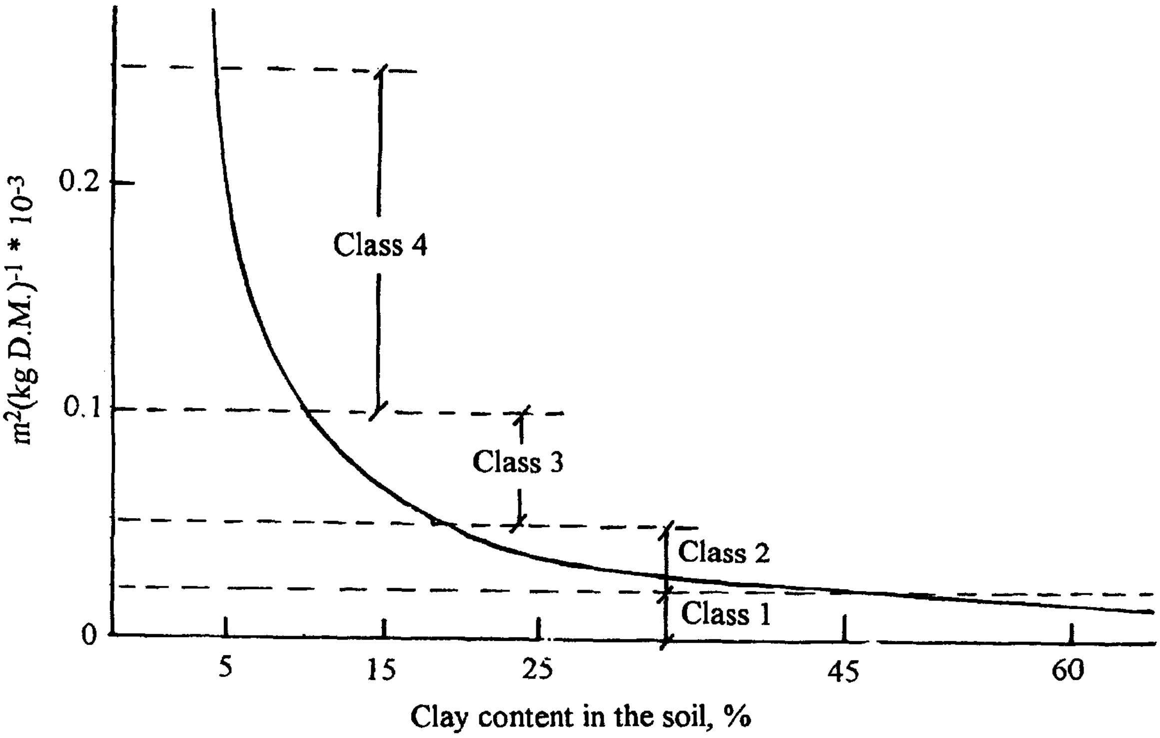


Figure 2.3 *Relationship between clay content of soil and the radiocaesium transfer to barley grain found in a series of field experiments over four years with twelve different soil types (Fredriksson et al., 1969).*

Table 2.1 gives an example of a more rigid soil class system with average values for the expected transfer of radiocaesium to the crop products based mainly on the clay content of the soil (Eriksson, 1977). Thus, the basis for classifying a land district was the fractions of the various soil types, which contributed to the total soil inventory. Then also other information on the surface soils such as content of organic matter, the pH-level, the geological origin of the soils and the fertilisation practice in the different districts influenced the classification. These expert judgements were based on experimental experience.

Table 2.1 *Estimated transfer of ^{137}Cs ($m^2\ kg^{-1}$ (dw)$*10^{-3}$) to various crop products expected by root uptake from mineral soils distributed on the different classes 1a-4a, (see Figure 2.3 above) and compared with the distribution of the soils in the two class system including clay and sandy soils (organic soils >20 % dw of organic matter not included).*

Crop	**Soil Classes**				**D.M.%**	**Relative transfer per kg fresh weight**
	1b		**2b**			
	1a	**2a**	**3a**	**4a**		
Grain	0.02	0.04	0.08	0.18	83.5	1
Peas	0.07	0.15	0.30	0.70	83.5	3.7
Oil seeds	0.05	0.10	0.20	0.45	83.5	2.5
Potatoes	0.15	0.30	0.60	1.40	21.0	1.9
Beets	0.30	0.60	1.20	2.80	20.0	3.6
Leafy veg.	0.75	1.50	3.00	7.00	20.0	3.6
Carrots	0.15	0.30	0.60	1.40	21.0	1.9
Radishes	0.30	0.60	1.20	2.80	20.0	3.6
Hay, silage	0.15	0.30	0.60	1.50	83.5	7.7
Natural pasture	0.45	0.90	1.80	4.50	83.5	23.1

Generally the heavy clays are rich in soluble potassium and belong to class 1a, medium clays to class 2a, light clays to class 3a and the sands and some mineral soils rich in organic matter are in class 4a. The organic soils or peat soils were not included in this mineral soil system 1a - 4a, but were introduced in the three class system, indicated in the table by 1b - 3b.

The peat soils are mostly used for grass production and the acreage in the Nordic countries is small. However, when the peat soils are used in agriculture they are critical with regard to the radiocaesium transfer to the crops in both the short and long term. The reasons for this are that (i) initially a very large fraction of the fallout radiocaesium tends to be sorbed exchangeably onto the voluminous organic part of the soil, and (ii) that the potassium status of these soils is generally rather low. Because of the low content of mineral substances (clay and sand), the move to a stronger average sorption of radiocaesium is a slow process. This results in more persistent and in higher transfer rates to the crops from peat soils than from mineral soils for years after the deposition. Information on the range of transfer factors for radiocaesium in the short and intermediate terms have been determined in the years after the Chernobyl accident (Haugen et al., 1990; Rosén, 1991; Prister, 1992; Rosén, 1996; Rosén et al.1996). The long-term transfer of radiostrontium to crops can be treated in a way similar to that for radiocaesium. Figure 2.4 shows, as an example, the relationship between extractable amounts of calcium in the soil and the resulting ^{90}Sr/Ca-ratio in barley grain found in field experiments (Haak & Lönsjö, 1975).

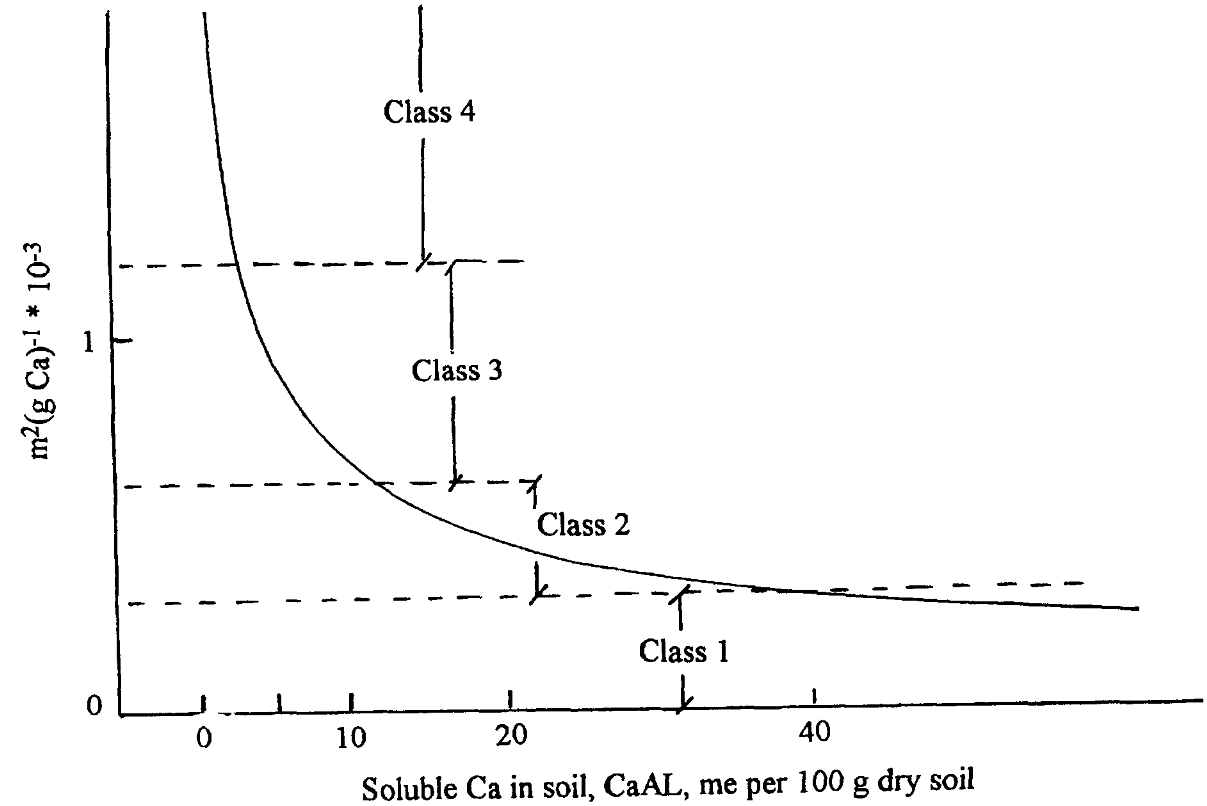


Figure 2.4 *Relationship found between Al-soluble calcium in mineral soils (Egnér et al., 1960) and the resulting ^{90}Sr/Ca-ratio in barley grain found in field experiments during 8 years with 12 different soil types.*

For radiostrontium, a rigid soil class system similar to that for radiocaesium has been developed (Eriksson, 1977). The expected long-term transfer of radio-radiostrontium to the crop products is shown in Table 2.2. Such systems are "two dimensional" only, compared to the real situation. Estimated mean values for soils and crops as shown have been obtained in experiments during a limited time period. The influence of the time factor and of the changes in the dynamic equilibrium in soil between radionuclides and potassium and calcium in the agricultural ecosystem is difficult to include in the systems.

Table 2.2. Estimated average transfer of ^{90}Sr, ($m^2\ kg^{-1}$ (dw)$*10^{-3}$) to various crop products contaminated by root uptake from mineral soils in the different classes 1a-4a and by the complementary system with only three soil types, 1b, loams, 2b, sandy soils and 3b, peaty soils.

Crop	**Soil Types**				**Crop dw (%)**	**Relative transfer to grain per kg fresh weight**
	1b and 3b		**2b**			
	1a	**2a**	**3a**	**4a**		
Grain	0.15	0.30	0.60	1.20	83.5	1
Peas	0.15	0.30	0.60	1.20	83.5	1.0
Oil seeds	1.00	2.00	4.00	8.00	83.5	6.7
Potatoes	0.30	0.60	0.95	1.80	21.0	0.44
Root crops						
Roots	3.80	6.00	9.50	18.00	20.0	4.6
Grass						
Hay, silage	3.70	5.80	8.10	13.7	83.5	17.1
Natural pasture	5.60	10.00	19.00	38.00	83.5	34.2

Good predictions within reasonable limits (Haak, 1983b) can be made on the long-term transfer rates in a given case. However, such predictions depend on appropriate classification of the land and many factors have to be considered. One factor is that detailed knowledge of the soils in a contaminated area may not be readily available when needed and so a simpler classification system might prove to be more practical than that outlined above. Such a simplified system may have loams as class 1, sandy soils as class 2 and organic soils as class 3. With regard to the root uptake of ^{90}Sr in the simplified system the main difference lies between loams and peat soils on the one hand and the sandy soils on the other. The latter generally have lower calcium content and lower pH-values than the former. Table 2.3 below shows the layout of a suggested prediction system utilising available information.

Table 2.3 *Suggested levels for the transfer of ^{137}Cs and ^{90}Sr to crop products by root uptake the year after fallout, (m^2 kg^{-1} (dw) $*10^{-3}$) and for the ecological half-time[a] in soil and in top layers of different types of grassland. Hay grass and cultivated pastures make the temporary grasslands. Natural pastures are permanent. The soil classes are loams, sand (<15 % clay) and peat (> 20 % dw organic matter)*

Crop type	**(m^2 kg^{-1} (dw)$*10^{-3}$)**			**Ecological half-time**		
	Loam	**Sand**	**Peat**	**Loam**	**Sand**	**Peat**
			^{137}Cs			
Bread grain	0.05	0.2	2	20	20	20
Fodder grain	0.05	0.2	2	20	20	20
Potatoes	0.3	1.2	12	20	20	20
Hay grass	5	10	100	1[b]	2[b]	3[b]
Cult. past.	5	10	100	1[b]	2[b]	3[b]
Nat. past.	10	20	200	2[b]	3[b]	4[b]
			^{90}Sr			
Bread grain	0.5	1	0.5	20	20	20
Fodder grain	0.5	1	0.5	20	20	20
Potatoes	1	2	1	20	20	20
Hay grass	10	20	10	2[b]	2[b]	2[b]
Cult. past.	10	20	10	2[b]	2[b]	2[b]
Nat. past.	20	40	20	2[b]	2[b]	2[b]

[a] See Part 3.

[b] Initially, after the deposition year, the reduction in transfer with time is expected to be larger and the half-time shorter than later in the following years.

Table 2.3 is based on recalculation of experimental data on root uptake for 4 years with 24 different Swedish soils from the large agricultural districts. Also the data obtained after the Chernobyl fallout in the northern districts have contributed to the table. The experiments, (Fredriksson et al., 1969; Haak & Lönsjö, 1975; Mascanzoni, 1988), gave the results shown in Table 2.4.

Table 2.4 *Transfer of radiocaesium and radiostrontium to grain, m^2 kg^{-1} (dw)$*10^{-3}$.*

Soil type	**^{137}Cs**	**^{90}Sr**
Loam	0.038 ± 0.026	0.32 ± 0.11
Sand	0.078 ± 0.104	0.47 ± 0.20
Peat (75% organic matter)	0.60 ± 0.04	0.19 ± 0.09

After Chernobyl, data from mineral soils in the northern counties indicated a radiocaesium transfer similar to, or somewhat higher than, that obtained in the earlier experiments (Eriksson & Rosén, 1991, Rosén et al., 1996). On a specific sandy soil in a fertilisation experiment, a range of (0.28-0.82) $*10^{-3}$ was observed in the control for a sequence of years after 1986. On well fertilised mineral soils in other experiments, the transfer to barley showed a range of (0.07-0.19) $*10^{-3}$, while on an a well fertilised organic soil rich in calcium the transfer varied within the range (0.47-1.11) $*10^{-3}$. Thus, as was expected from the discussion above, the variation is also large within the soil classes.

11.1 Critical soils and crops

Soil-to-plant transfer of radiocaesium is highest on organic and sandy soils. Plants on soils with a moderate to high clay content will be far less vulnerable. For ^{90}Sr, sandy soils (often with low pH-values) will be most critical. In general, the transfer of radiostrontium to plants by root uptake is an order of magnitude higher than that of radiocaesium.

In the long-term, permanent pasture will be the major problem. Of secondary, but significant importance, are leafy vegetables and the leaves of root crops rich in minerals which are used as cattle feed. Root crops show a relatively high uptake of radiostrontium but uptake of both radiocaesium and radiostrontium to grain is small.

12. Countermeasures in Agriculture

Having established from the above radiological considerations that land and/or crops are contaminated to unacceptable levels, countermeasures will be needed to reduce the contamination to levels where the likely radiological consequences are greatly reduced, hopefully to insignificant levels. At best, normal agricultural practices may suffice (e.g. ploughing) but at worst abandonment of land may be the only solution. Between these two extremes, there are many actions which can be taken but the most useful and practicable methods are those most closely allied to normal agricultural practices. The composition and physical half-life of the radioactive contamination will be major factors which need to be taken into consideration when determining which countermeasures are required.

The countermeasures cover:

1. actions taken immediately after contamination has taken place in order to combat problems from both short-lived radionuclides (^{131}I in particular) and long-lived radionuclides, and to avert potential long-term problems of, for example, soil contamination by transfer from standing crops to soil. These actions are prompt countermeasures.
2. actions which need not be taken after a detailed assessment of the nature, deposition rate and spatial distribution of contamination have been made. These actions are delayed countermeasures.

The ultimate objectives of the countermeasures are to prevent contaminated agricultural products entering the food chain and to enable land management to continue as normally as possible with minimum disruption. A range of options are available and these are discussed in Sections 13.1 and 13.2.

12.1 Mitigation of short-term consequences

12.1.1 Removal and Disposal of Contaminated Crop

Removing and discarding the contaminated crop is one way to reduce the contamination level of the land. The potential effectiveness of this measure depends on the type and stage of development of the crop because it determines the fraction of the fallout which has been intercepted by the crop and that is still retained by the crop at the time of the decontamination operation. A dense crop can intercept 25 to 50% of wet deposition and sometimes even more of dry deposition. However, if removal is delayed, then the efficiency of removal will be less, and in general, the half-time of loss of contamination on undisturbed vegetation is 2-4 weeks.

This method creates radioactive waste which must be stored or disposed of adequately.

12.2 Mitigation of the long-term consequences

12.2.1 Removal of shallow layer of soil

Complete removal of the contaminated layer of surface soil is without undoubtedly the most effective and most publicly acceptable means of decontamination. On high quality land, where the fertility and drainage would not be impaired by removing a shallow layer (5-10 cm) of soil, this is an effective but expensive countermeasure. The effectiveness of turf cutters in removing surface contamination from grassland was assessed on a 50 m^2 test area using simulated fallout (Cobbin and Owen, 1965). The turf cutters were used to shave off thin uniform layers which were left in place on the ground. The strips were cut manually into suitable lengths, rolled and placed in carriers. Maximum efficiency was achieved when the cut depth was set at 3-4 cm as this minimised the bulk of material to be removed, while still providing sufficient thickness to roll the turf. Removal of surface sods carried away between 97 and 99% of deposited material but stones and roots can cause the cut to be uneven, leaving holes in the turf. The turf would be in a form amenable to collection and transportation. In a similar more recent pilot scale field experiment in, 1991, where ammonium molybdate solution was used to simulate radioactive fallout, some 70-90% of the contamination was removed (Jouve et al., 1991).

The disadvantages of such approaches are the enormous logistic and disposal problems. A 5 cm layer of soil over an area of 1 hectare has a volume of 500 m^3 and a mass of about 1000 tonnes.

An alternative to removing the waste from the site would be to place it in self-shielding piles. A typical mound could be a flat-topped pyramid 3 m high with a base of 15 m x 30 m. If the top 5 cm were removed from a field, the base of such a mound would occupy only about 3% of the surface area of the field. Resuspension from the mound could be prevented by growing grass or even covering with, for example, asphalt (Smith and Lambert, 1978)

Methods designed to remove only the minimum amount of soil while removing virtually all the contamination without creating any problems from resuspension are being investigated at CEN Cadarache, France (Jouve et al., 1991). Two methods are under investigation. In the Decontaminating Vegetal Network (DVN) method, grass seed and a growing medium made of a mixture of peat, polysaccharides and water is sown on undisturbed contaminated soil and grown for a sufficiently long period to produce a cohesive network of roots and support material with much of the contaminated soil entrained in the network. The DVN is also very effective in suppressing resuspension of contaminated soil. The technique is often used to grow grass for fixing soil on sloping terrain and preventing erosion. The mixture is suitable for application by helicopter (at a rate of 0.3 km^2 d^{-1}) or from farm vehicles.

When the root mat is well established the DVN is removed, together with about 1-2 centimetres of the topmost layer of soil along with it most of the contamination. The grass seed can be left for a year or more without causing any problems. Eventually the thin cohesive layer of soil is removed with a mechanical turf cutting machine and disposed of. The standard turf cutting machines can cover about one ha per day.

12.2.2 Ploughing

At least three different types of plough may be considered for countermeasure operations:
- normal or mould board ploughs turning over the topmost 20 or 30 cm
- deep ploughs turning over the topmost 50 or 100 cm of soil
- skim-and-burial ploughs.

12.2.2.1 *Normal or Mould Board Plough*

Normal or mould board ploughing, to a depth of 20-30 cm, where suitable, would often be the method of choice since these ploughs and the appropriate tractors are used by, or are available to, most arable farmers and no reduction in fertility would result. The ready availability of normal ploughs means that any problems of resuspension could be suppressed quickly after contamination if deemed necessary. The disadvantage of ploughing to a depth of only 20-30 cm is that ploughing in subsequent years would tend to return the contamination to the surface. In any case, contamination in the topmost 20-30 cm would still be available to most crops, although diluted. The efficiency of different types of ploughs in placing a contaminated surface layer deeper in the soil profile was studied by Nilsson (1983).

12.2.2.2 *Deep Ploughing*

Large ploughs drawn by powerful tractors can work at depths of up to 1 metre. Such ploughing would render virtually all the contamination unavailable for uptake by arable crops and grasses, and subsequent normal ploughing would leave the contamination largely untouched. Drawbacks of deep ploughing are that the land drains are destroyed and the upper fertile layer of soil may be lost forever, causing significant reductions in fertility.

Deep ploughing also reduces the external dose rate significantly (Roed, 1982, see Part 1).

12.2.2.3 *Skim-and-Burial Ploughing*

Recent research work at the Riso National Laboratory in Denmark has resulted in the production of a full-size working implement known as the 'skim-and-burial plough' Roed, 1992). This plough can be set to skim off the topmost 5-10 cm layer of soil and place it as a discrete layer at a depth of about 50 cm beneath a non-inverted soil layer. Such a plough has the advantage that soil fertility would not be impaired in most circumstances. Of course, land drains could still be damaged but this is seen as a secondary and surmountable problem.

The change with time of the average dose rate has been calculated for a typical farm in Middle Sweden after radioactive depositions occurring early and late in the season, before and after normal and after deep ploughing according to the normal rotation scheme. The results given in Figure 2.5 indicate that 4-5 years after the deposition the radiation level due to the radiocaesium fraction of the fallout has been reduced to 20 and 10 % respectively. The reason for the moderate reduction is that permanent grasslands are considered as parts of the farm and thus influence the average. If the whole acreage is ploughed or deep ploughed as soon as possible, regardless of the rotation scheme, a similar reduction might be achieved in the first year after the fallout is deposited.

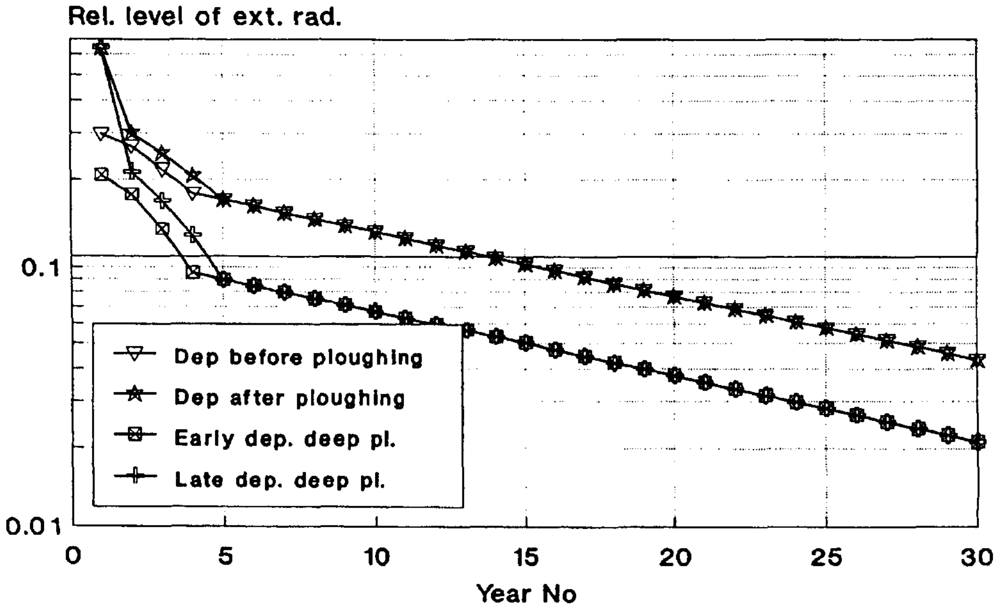


Figure 2.5 Estimated average dose rates (relative units) for the 30 years following deposition of ^{137}Cs at different times of the year. The calculations are for a farm in Middle Sweden assuming a reduction factor of 6 following normal ploughing and a factor of 20 after deep ploughing. Residence half-time of ^{137}Cs in the soil layers has been set at 20 years. The farmland is assumed to consists of 60 % loam, 25% sandy and 15% organic soils.

12.2.3 Application of Chemical Fertilisers

The root uptake of radiocaesium and radiostrontium is strongly influenced by the presence of potassium and calcium respectively in soil solution, or at least in a form accessible to the roots. The influence of stable analogues can often be exploited to reduce root uptake of the respective radioisotopes of these elements. Of course, on soils which are rich in these elements the soil-to-plant transfer of contaminant radiocaesium and radiostrontium would be less than on soils deficient in potassium and calcium and countermeasures may not be necessary. This is another case where determination of soil properties would be necessary to determine the need, or otherwise, for implementation of countermeasures.

Micro-plot experiments have been conducted in the field to study the transfer of ^{137}Cs to different crops when influenced by different levels of potassium applied to different soils (Fredriksson et al., 1966; Lönsjö et al., 1990). Three soils were used, one loam, one clay loam and one silty clay. During the first year, radiocaesium and the potassium applied was mixed into the plough layer at rates of 0, 156 and 625 kg K ha^{-1} as different treatments. During the following years, the annual potassium fertilisers were mixed into the sowing layer at rates of 0, 31 and 156 kg K ha^{-1}, respectively.

The experiment continued for a period of 7 crop rotations with oats, white mustard and peas. The species were selected to represent oil seed crops, grain crops and leguminous seed crops, respectively. (Fredriksson et al., 1966). The plant uptake of radiocaesium was generally lower on soils rich in clay and on soils given ample applications of potassium to cover the loss by cropping. White mustard accumulated the most radiocaesium, next came oats and finally peas.

The effect of lime on the radiostrontium uptake from Ca-deficient soils has been studied since the 1950s in both pot and micro-plot field experiments. The latter showed that in the normal fertilization conditions when lime is added to raise the pH of acid soils the radiostrontium uptake by crops can fall by a factor of about 2 or 3 (Fredriksson et al., 1969).

13. Factors to be Considered Before Implementation of Agricultural Countermeasures

Before implementation of countermeasures, both technical, economic and social aspects must be considered. Whilst the prime motive for employing countermeasures is to minimise radiation dose, the economic consequences of lost production must also be taken into account. Ultimately, it is the farmer who shoulders the burden of responsibility for applying the countermeasures and it is the farmer who risks the losses, unless compensation is paid.

As discussed above, the severity of the contamination may be largely governed by the time of year when contamination takes place and this phenomenon is described as seasonality. The Nordic 'farming year' can be conveniently divided into five periods, namely:

1. start of the growing season (spring)
2. early summer hay harvest (June)
3. late summer hay harvest (August)
4. late summer or early autumn cereal harvest (August - September)
5. outdoor season for grazing dairy animals.

The consequences of surface contamination of young arable crops in the early part of the growing season may only be severe if the soil type is such that soil-to-plant transfer factors are relatively high, as they are for example, for radiocaesium on peaty and highly organic soils.

When fallout occurs on mature grass ready for harvesting as hay it may be best to cut the grass and, if levels are sufficiently elevated, to dispose of it. In any case, the act of cutting and collecting the grass will serve to remove a significant proportion of the contamination from the land (Menzel and James, 1961; James and Menzel, 1973) if it is accomplished quickly enough after fallout is deposited. Similarly, fallout on mature grass in August may be dealt with in the same way.

In September, surface contamination of mature cereal crops may be a cause for concern. Harvesting the contaminated crop followed by disposal may well be the chosen countermeasure if contamination levels are unacceptably high.

Fallout in the autumn, just prior to land tillage, is likely to be dealt with either by scraping off a surface layer of soil or by ploughing.

Fallout on grazing land will have the greatest impact in the summer when the animals are normally grazing outdoors.

14. An outline strategy for dose reduction in the contaminated cultivated agricultural environment

14.1 The agricultural environment and the type of countermeasures

The agricultural environment consists, as mentioned before, of ploughed land with annual crops and temporary grassland in rotation, and permanent pastures. Depending on topography and on climatic and soil conditions, the use of the land varies considerably within and between regions and can range from an extreme of 100% ploughed land to 100% permanent pastures.

In most cases the permanent pastures cannot be treated with countermeasures such as removal of crops and removal of thin soil layers or with deep ploughing etc. In some cases they can be fertilized and limed, if these measures seem applicable, and if they are considered too contaminated they can be excluded from feed production. Ploughed land including temporary grasslands can be treated with a wider spectrum of remedial activities, although these may have to be adjusted to the season of the fallout.

Temporary grassland and croplands after harvest are comparatively flat, and under favourable moisture condition most of the mineral soils can be treated with available methods for deep ploughing and skim burial and with removal of contaminated upper soil layers. Also the soil management in the spring time results in relatively even soil surfaces.

Deposition of fallout on crops which have not been harvested or on ploughed land may offer more difficulties in the first case. The crops have to be taken care of and disposed of in the first case and in the second the soil surface may be uneven if the harrowing of the ploughland in the autumn is not practised.

Depending on the time of the year of the fallout this type of land may consist of;
1. Ploughed land, levelled or not levelled
2. Land carrying a crop
3. Cropped land before ploughing.

In case of a radioactive fallout, there are means available for reduction of the radiation level on the land and of the transfer of the radioactive nuclides to the food chain in the short term as well as in the long term. Before countermeasures are decided on, the cost and resulting dose to workers, has to be compared to the estimated savings in the future. The methods and activities considered here include:

Group A.	(i)	Removal of the crops.
Group B.	(ii)	Removal of a slice of the upper soil layer.
Group C.	(iii)	Placement of the upper contaminated soil layer by a deep ploughing device to a level deeper in the soil profile.
Group D & E.	(iv)	Agrochemical countermeasures such as potassium fertilisation and liming to reduce the uptake of the fallout radionuclides by crops.

These methods have experimentally well founded data, but the practical efficiency depends also on the quality of the work as it is carried out in the field. Group A counter- measures are carried out during the year of the fallout and the costs are those of normal harvests. Group B and Group C measures require a certain time for organising and start up. Group B measures would be carried out by a team of workers. The field is ploughed around 5 cm deep and the soil scraped in heaps, loaded on dump trucks and transported to a near in field disposal site for wastes. Group C is calculated at a cost of three times that of normal ploughing at moderate speed and with average equipment. Groups D and E, namely extra potassium fertilisation and extra liming respectively, can be implemented when lime and fertilisers are available.

14.2 Model area

The real agricultural environment is a complex system and calculations therefore have to be supported by a model area containing the main features of such a system. Middle Sweden was selected as a test site for the case study. The acreage of the study area was limited to 100 ha or 1 km^2 and this area was intended to be representative of the region containing both agricultural and forest land. The agricultural portion was 38 ha, of which 32.2 ha was used for crop production and pasturing.

The cultivated fraction of the area was assumed to be composed of different soil types and crop lands. Loamy soils (>15 % clay) cover 60 %, sandy soils (mineral soils with <15 % clay) 25 %, and peaty soils (>20 % organic matter) 15 % of the acreage. On average, bread grains were assumed to be cultivated on 5.5 ha, coarse or feed grain on 17.4 ha, hay and silage grass on 5.4 ha, cultivated pasture on 1.3 ha, natural pasture on 2.5 ha and potatoes on 0.2 ha. The remaining agricultural land up to 38 ha was assumed to be used for other minor crops not listed, to be in fallow and not used for cropping. On average the number of milking cows in the system are assumed to be four, with a milk production per cow of 6000 kg y^{-1}.

The average assumed crop production is given in Table 2.5 and is the same on the three different soil types.

Table 2.5 Annual crop production, kg dry matter per ha in the case study area.

Crop	Annual production (kg dw ha^{-1})
Bread grain	3000
Hay grass	6000
Feed grain	3000
Cultivated pasture	5500
Natural pasture	3500
Potatoes	4000

Another assumption is that the average distribution of the crops depends on the soil type as the properties of the soil will tend to determine land use. In practice more bread grains are cultivated on loams than on sandy soils and none are grown on peat soils. Also, more of the coarse or feed grains are cultivated on sandy soils than on peat soils. Grasslands cover most of the peat soils. The distribution of the crops on the different soil types of the case study is shown in Table 2.6. The total production of dry matter by the acreage used is shown in Table 2.7. The farmland used support a variety of domestic animals. Among them the 4 milking cows are considered here. Their annual feedstuffs in tons of dry matter is derived from different crops and soil combinations, Table 2.8.

Table 2.6 Percentages of the acreage (32.2 ha) used for the different crops on the three soil types

Crop	Loam	Peat	Sand	Total
Bread grain	13.00	0.00	4.27	17.10
Hay grass	6.44	6.44	3.90	16.80
Feed grain	36.05	4.05	13.50	54.00
Cult. pasture	1.55	1.55	0.94	4.04
Nat. pasture	2.98	2.98	1.81	7.77
Potatoes	0.00	0.00	0.62	0.62
Total	60.00	15.00	25.00	100.00

Table 2.7 Annual crop production, tons of dry matter, on 32.2 ha

Crop	Loam	Peat	Sand	Total
Bread grain	12.38	0.00	4.13	16.50
Hay grass	12.44	12.44	7.53	32.42
Feed grain	35.24	3.92	13.05	52.20
Cult. pasture	2.75	2.75	1.66	7.15
Nat. pasture	3.36	3.36	2.03	8.75
Potatoes	0.00	0.00	0.80	0.80
Total	66.16	22.46	29.20	117.82

Table 2.8 Cow feedstuffs derived from crop and soil combinations, tons of dry matter per cow year.

Crop	Loam	Peat	Sand	Total
Hay grass	0.79	0.79	0.48	2.06
Feed grain	1.07	0.12	0.39	1.58
Cult. pasture	0.24	0.24	0.14	0.62
Nat. pasture	0.28	0.28	0.17	0.74
Total	2.38	1.43	1.19	4.99

14.3 Assumptions on fallout nuclides and their transfer in the case study area

1. Fallout level: The calculations are based on a deposition of 1 MBq m^{-2} of both ^{137}Cs and ^{90}Sr.

2. Transfer factors: The transfer of fallout to crops with start the year after fallout is deposited, continuing up to 29 years following, is considered. The transfer factors used for the first year are shown in Table 2.9.

Table 2.9 Transfer factors for plant uptake of ^{137}Cs and ^{90}Sr, (T_{ag}, $m^2\ kg^{-1}$ (dw)) assumed in the case study during the first year after the fallout year.

Crop	^{137}Cs			^{90}Sr		
	Loam	Peat	Sand	Loam	Peat	Sand
Bread grain	0.00005	0.002	0.0002	0.0005	0.0005	0.001
Hay grass	0.005	0.1	0.01	0.01	0.01	0.02
Feed grain	0.00005	0.002	0.0002	0.0005	0.0005	0.001
Cultivated pasture	0.005	0.1	0.01	0.01	0.01	0.02
Natural pasture	0.01	0.2	0.02	0.02	0.02	0.04
Potatoes	0.0003	0.012	0.0012	0.001	0.001	0.002

3. Ecological half-lives of the nuclides: The ecological half-lives with regard to the plant availability of the nuclides ploughed into arable soil is assumed to be 20 years during the first 30 years after deposition. This is a simplification, since there is experimental evidence that the ecological half life of transfer rates of fallout nuclides on grasslands is not a constant but develop linearly with time from a short half-time, 1-2 years, in the beginning to a longer half-time, 20 and up to 40 years after 30 years for radiocaesium and radiostrontium respectively (Eriksson, 1994).

4. Nuclide transfer from cow feed to milk: The transfer coefficients feed-milk (for definition see Part 3) used in the case study are for ^{137}Cs 0.008 d kg^{-1} and 0.0013 d kg^{-1} for ^{90}Sr.

5. Effects of ploughing temporary grasslands: In practice temporary grassland is gradually converted to ploughed land according to the applied rotation scheme. In the following case study it has been assumed that a quarter of the land is ploughed and replaced by new grassland every year. By the placement at a lower depth of the contaminated surface soil layer by ploughing, the radionuclide transfer is reduced considerably in the next crop. It is also reduced to the same extent in the new grassland products. In year 1 (when the deposition year = 0) the unploughed fraction of the grassland is = 0.75 and the ploughed fraction = 0.25. In year 2 these fractions are equal, = 0.5, and by year 4 the whole acreage of the temporary grassland would normally have been ploughed.

6. Effects of extra potassium fertilisation and liming: Additional potassium fertilisation (above that normally applied) means that in total up to 200 kg K ha^{-1} is assumed to have been added after the fallout is deposited and that this higher potassium status of the land has been retained in the following years by smaller applications. Depending on the previous status of the soils the reduction in radiocaesium transfer differs between the soil types. In the following it is assumed that the improved potassium status reduces the transfer of radiocaesium to plants by 25 % on loams, 50 % on sandy soils and by 70 % on peat soils.

Liming increases the content of calcium ions and the pH-level in the soil; the increased competition from calcium ions reduces crop uptake of radiostrontium. An average reduction to 50 % can be achieved on mineral soils. The expected absolute effect is higher on the sandy soils than on loams or on normal organic soils. It is assumed that the relative liming reduction factor is 50, 50 and 33 % respectively and that it has a half life of 20 years.

14.4 Cost effectiveness of countermeasures

A variety of different countermeasure scenarios are discussed below. In each case the deposition of ^{137}Cs and ^{90}Sr is assumed to be 1 MBq m^{-2}. The transfer of radionuclides in this case study has been calculated for the model described above, initially for the case when no countermeasures are applied (Eriksson, 1994). The relative transfers of ^{137}Cs and ^{90}Sr to crops and to milk, over the total 30 years, for all crops and soil types are shown in Table 2.10. The origin of the activities in the crops is given by the distribution on the soil types alone, whereas the origin in the milk is given by both the distribution on the crops and on the soil types. The total transfer to the crops was calculated to be 11200 MBq of ^{137}Cs and 4880 MBq of ^{90}Sr and to milk was 454 and 30 MBq respectively.

Table 2.10 *Calculated relative transfer of ^{137}Cs and ^{90}Sr, % of total to crops and to milk, for 30 years in the study area. No extra treatments have been carried out on the land*

	Soil type							
Crop	**Loams**	**Peat**	**Sand**	**Total**	**Loams**	**Peat**	**Sand**	**Total**
	(60 %)	(15 %)	(25 %)	(100 %)	(60 %)	(15 %)	(25 %)	(100 %)
	^{137}Cs in crops, % of total				^{137}Cs in milk, % of total			
Bread grain	0.1	0	0.1	0.2	-	-	-	-
Hay grass	1.4	27.8	1.7	30.9	1.1	22.9	1.4	25.4
Feed grain	0.2	1.0	0.3	1.5	0.1	0.4	0.1	0.6
Cultivated pasture	0.3	6.1	0.4	6.8	0.3	6.8	0.4	7.6
Natural pasture	2.7	54.5	3.2	60.5	3.0	59.9	3.5	66.4
Potatoes	0	0	0.1	0.1	-	-	-	-
Total	4.7	89.5	5.8	100.0	4.6	89.9	5.4	100.0
	^{90}Sr in crops, % of total				**^{90}Sr in milk, % of total**			
Bread grain	1.8	0	1.2	3.0	-	-	-	-
Hay grass	6.6	6.6	7.9	21.0	5.8	5.8	7.0	18.5
Feed grain	5.1	0.6	3.8	9.5	2.1	0.2	1.6	3.9
Cultivated pasture	1.4	1.4	1.8	4.6	1.7	1.7	2.1	5.5
Natural pasture	19.1	19.1	23.2	61.4	22.4	22.4	27.2	72.0
Potatoes	0	0	0.5	0.5	-	-	-	-
Total	34.0	27.7	38.3	100.0	32.0	30.1	37.9	100.0

Table 2.10 shows that the transfer of fallout radiocaesium in the agricultural system has one main source, the organic soils. About 90% of radiocaesium in crops and milk are thus transported from 15% of the land, 50-60% of which is from the natural pastures on these soils. Thus, 3% of the acreage is responsible for about 60 per cent of the radiocaesium transfer to feeds and foods. Excluding natural pastures on peat soil from the provision of feed sources might therefore be a first obvious step to take to reduce the transfer of radiocaesium to foods. The next step might be the exclusion of all peat soils, comprising 15% of the acreage, from production of cow feed. Then a reduction by 90 per cent of the potential radiocaesium transfer is possible. This exclusion of land from feed may be temporary, and required for a short or longer period of time depending on the ecological half-lives.

With regard to the radiostrontium transfer Table 2.10 indicates that the main source is the natural pastures. Exclusion of these natural pastures comprising about 8% of the acreage, from production of feed might reduce the radiostrontium transfer to milk by about 70 per cent. For other countermeasures reducing radiocaesium and radiostrontium transfer to cows, see Part 3.

After elimination of sources with high transfers of radionuclides to food and feed, such as those mentioned above, other countermeasures might also be applied. The data obtained from this study and from SLU (1989) have been used for estimates of the effects and costs of a series of countermeasures in the agricultural system, without exclusion of land (Tables 2.11-2.17). Table 2.11 introduces the effects and costs of removal of contaminated crops and 5 cm superficial top soil layers. Such methods are applicable on ploughed land (cf. above). It appears that removal of contaminated crops, Method A, is a comparatively easy and cheap method per unit of removed radioactivity. However, the result is less effective in removing radioactivity compared with that of the other methods, B and C. These methods have to be applied to remove the major parts of the deposition after heavy fallout events.

Table 2.11 *Calculated costs of removal of activity in person hours and ECU per ha of countermeasures A - C, and cost per unit removed from the land. Deposition: 1 MBq m^{-2}*

	Season when	**Removal**		**Removal Cost**		**Cost/unit removed**	
	deposition occurred	%	MBq ha^{-1}	Person hours	ECU ha^{-1}	ECU, %	ECU MBq^{-1}
	A. Removal of crops after wet deposition.						
Cereals	Summer	10	1000	4	94	9	0.09
Cereals	Autumn	25	2500	7	118	5	0.05
Grass	Spring	10	1000	4	94	9	0.09
Grass	Summer	25	2500	7	118	5	0.05
	B. Removal of top soil layer						
About 5 cm		95	9500	35	1800	19	0.19
	C. Skim and burial ploughing						
About 50 cm depth		90	9000	9	318	4	0.04

Table 2.12 Calculated costs in ECU per MBq for 30 years of reduced radionuclide transfer on ploughed land to the crops, and per saved person-Sv by countermeasures B and C. For clarity the deposited nuclides are considered separately. No exclusion of crop land is assumed.

Crops	Nuclide	Reduction ha^{-1}, MBq	ECU MBq^{-1}	Saved person-Sv	ECU person-Sv^{-1}
			Countermeasure B		
Bread grain	^{137}Cs	3.55	510	0.046	39,130
Potatoes	^{137}Cs	65.36	28	0.85	2,120
Cow feed	^{137}Cs	230.45	7.8		
Milk	^{137}Cs	30.46	54	0.40	4,080
Bread grain	^{90}Sr	2.5	720		
Potatoes	^{90}Sr	10.64	170		
Cow feed	^{90}Sr	77.83	21		
Milk	^{90}Sr	1.66	980		
			Countermeasure C		
Bread grain	^{137}Cs	3.36	95	0.044	7,220
Potatoes	^{137}Cs	61.92	5.2	0.8	400
Cow feed	^{137}Cs	218.33	1.3		
Milk	^{137}Cs	28.86	10	0.38	760
Bread grain	^{90}Sr	2.37	130		
Potatoes	^{90}Sr	10.08	28		
Cow feed	^{90}Sr	73.74	3.9		
Milk	^{90}Sr	1.58	180		

To judge the benefit of investments in expensive single countermeasures such as B and C the cumulative effects in the system during a long period needs to be considered. In the following (including Table 2.12) the calculations comprise a period of 30 years. The effect of other countermeasures of annual nature, such as extra potassium fertilisation, is an improved potassium status of the soil. Due to the natural reduction of radiocaesium transfer the effect of a maintained better potassium status of the soil on the radiocaesium transfer to the crops will decrease with time. The improved lime status of the soil is maintained by liming with longer intervals, 10 - 15 years, and is basic to agricultural production especially on acid soils. On these soils liming is a countermeasure to reduce the radiostrontium transfer with the crops, but the absolute effect will decrease with time due to the ecological half-life of the radionuclide.

The costs for a person-Sv saved by a countermeasure depends on the original deposition rate, transfer rates in the food chain and the costs and the effectiveness of the countermeasure. The influences of costs and effects are shown in Table 2.12 for countermeasures B and C. To save a person-Sv in bread grain by reducing the radiocaesium content the costs are 20 times higher than those for saving a person-Sv in potatoes and 10 times higher than those for saving a person-Sv in milk. The reasons for the variation in foods depends on the low radiocaesium transfer to grain from the soils used for grain crops compared to the radiocaesium transfer to potatoes from the sandy soils. The lowest cost per person-Sv should have been observed for the milk, which depends on the grassland crops, if the natural pastures had been available for the countermeasures. The countermeasures B and C could only influence the radiocaesium flow from ploughland crops which delivered proportionally less radiocaesium to the cow feed (Cf. Table 2.10). However, if B and C for some reasons cannot be implemented, the countermeasures D and/or E can be used as complementary to A, B and C.

Table 2.13 provides the basic data needed to calculate the cost efficiency in the study area for the countermeasures D and E, namely extra K-fertilisation and extra liming, respectively, which is given in Table 2.14 and Table 2.15.

Table 2.13 *Reduction by K-fertilisation and liming of the transfer of ^{137}Cs and ^{90}Sr to the whole crops, to dairy cow feed and to cow milk produced in the study area in the first year and for 30 years.*

Crops	Crops, MBq		Dairy feed fraction[a]	Costs, feed[b]		Dairy feed, MBq		Milk, MBq	
	1 y	30 y		1 y	30 y	1 y	30 y	1 y	30 y
Countermeasure D. K-fertilisation for reduction of ^{137}Cs transfer									
Bread grain	0.55	8.12	-	-	-		-	-	-
Potatoes	0.47	6.88	-	-	-		-	-	-
Feed grain	7	104	0.121	155	3,740	0.85	13	0.1	2
Hay grass	700	2320	0.254	101	2,450	178	588	23	77
Cult past.	154	510	0.347	33	800	53	176	7	23
Nat. past.	482	4540	0.338	62	1,510	162	1528	21	202
Total	1344	7480		351	8,490	394	2300	52	304
Countermeasure E. Liming for reduction of ^{90}Sr transfer									
Bread grain	5	50	-	-		-	-	-	-
Potatoes	0.8	8	-	-		-	-	-	-
Feed grain	15	157	0.121	49	1,480	1.8	19	0.04	0.41
Hay grass	128	364	0.254	33	965	32	93	0.69	1.98
Cult. past.	28	80	0.347	11	318	10	28	0.21	0.59
Nat. past.	89	878	0.338	20	600	30	300	0.64	6.30
Total	266	1540		113	3,380	74	440	1.60	9.30

[a] Fraction of the acreage of the crop used for production of feed for the dairy cows.
[b] Costs for extra K-fertilisation and liming of the acreage used for dairy cow feed, in ECU.

Table 2.14 Estimated cost/efficiency for reduction in radionuclide transfer to crops, dairy feed and milk by extra K-fertilisation and extra liming, (ECU MBq^{-1}).

Crop	Crops		Dairy feed		Milk	
Source	**1 y**	**30 y**	**1 y**	**30 y**	**1 y**	**30 y**
Countermeasure D: K-fertilisation for reduction of the radiocaesium transfer						
Bread grain	730	1,210	-	-	-	-
Potatoes	32	52	-	-	-	-
Feed grain	180	299	181	299	1,380	2,270
Hay grass	0.6	4.1	0.6	4.1	4.4	32
Cult pasture	0.6	4.6	0.6	4.6	4.7	34
Nat pasture	0.4	0.9	0.4	0.9	2.9	7.1
Total	1.8	7.6	0.9	3.6	6.8	28
Countermeasure E: Liming for reduction of the radiostrontium transfer						
Bread grain	27	76	-	-	-	-
Potatoes	6.2	18	-	-	-	-
Feed grain	27	79	27	79	1,280	3,660
Hay grass	0.9	10	0.9	10	44	490
Cult pasture	1	11	1	11	49	530
Nat pasture	0.7	2	0.7	2	33	94
Total	2.8	15	1.5	7.8	72	360

Table 2.14 shows that the costs for reduction are reversed to the original transfer rates from soil to the crops of ^{137}Cs and ^{90}Sr. The costs for reduction is lowest on natural pastures and during the early phases of the period after the initial deposition year. High costs for the reduction of radionuclide content in crop products and in foods indicates that the implementation of countermeasures is not so urgent as the transfer rates are low. Low costs for reduction of the radionuclide content depends on high transfer rates from soil to plant and requires that effective countermeasures of different kinds are implemented as soon as possible to be economic. In the system described in Table 2.14 the high transfer rates, as indicated by the low cost for reduction of the nuclide transfer on natural pastures, are not acceptable and therefore this source had better be eliminated completely from food production.

Table 2.15 below shows the cost-efficiency, ECU person-Sv^{-1}, for reduction of the radiocaesium transfer to crops and milk in the study area by fertilisation. The difference between Year 1 and the whole period of 30 years is large, especially for milk. The high costs for the grain crops and the low for milk indicates again the sensitivity of the use of the grass crops for animal husbandry.

Table 2.15 *Estimated cost-efficiency of K-fertilisation for reducing* ^{137}Cs *transfer to foods derived from different crops, ECU person-Sv*$^{-1}$ *(1 MBq = 0.013 person-Sv).*

Crop	**Food crops**		**Cow milk**	
Source	**Year 1**	**30 years**	**Year 1**	**30 years**
Bread grain	56900	93900	-	-
Potatoes	2440	4000	-	-
Feed grain	-	-	107100	177000
Hay grass	-	-	340	2500
Cultivated pasture	-	-	370	2700
Natural pasture	-	-	230	580
Total	-	-	530	2200

In Table 2.16 below the data for countermeasures B, C and D are compared with regard to the cost per person-Sv reduced when using the bread grain, potatoes and milk produced in the study area as foods. It is found that for the crops the costs decreases in the order D>B>C with the relative numbers 100, 42, 8, whilst for milk the order is B>D>C with the relative numbers 100, 53, 18. Consequently Countermeasure C, Skim and Burial Ploughing, seems to be a favourable alternative, if it can be implemented within a reasonable time after deposition has occurred.

Table 2.16 *Countermeasures B - D compared with regard to the costs for saving one person-Sv potentially caused by* ^{137}Cs *in foods produced over 30 years in the study area*

Countermeasure		**Food item**	**ECU person-Sv**$^{-1}$	
			Year 1	**30 years**
B.	Removal of	Bread grain	-	39,100
	soil layer	Potatoes	-	2,100
		Milk	-	4,080
C.	Skim and	Bread grain	-	7,220
	burial	Potatoes	-	400
	ploughing	Milk	-	750
D.	Annual	Bread grain	56,200	92,800
	K-fertilisation	Potatoes	2,400	3,980
		Milk	530	2,160

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PART 3

ANIMALS

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16. Contamination of animals

16.1 Important radionuclides

The radionuclides of most concern with respect to contamination of animals after a nuclear accident are radioiodine, radiocaesium and radiostrontium (ICRP 30, 1979). Of the other significant anthropogenic radionuclides likely to be released in most accidents, only small proportions of that ingested will be absorbed in an animals gut, and the main animal products, milk and meat, will not normally be contaminated to a significant extent. Animal products will mostly be contaminated as a result of ingestion of contaminated feed and possibly, but to a much lesser extent, from inhalation (for radioiodine only). Direct external contamination of animals is of little or no consequence in human food production. Radioiodine and radiostrontium are important with respect to contamination of milk; radiocaesium contaminates both milk and meat. Over the mid to long term perspective of this document, short-lived radioiodine isotopes such as ^{131}I are not relevant, and hence will not be considered.

The physical and chemical form of a radionuclide can influence its absorption in the animal gut. For example, following the Chernobyl accident radiocaesium incorporated into vegetation by root uptake was more readily absorbed than that associated with the original deposit (e.g. Beresford *et al.*, 1989; Ward *et al.*, 1989; Hansen and Hove, 1991; Mayes *et al.*, 1996).

16.2 Transfer of radionuclides to animals

Transfer of radionuclides to animals may be described using different transfer parameters, which assume linearity between fallout deposition rates (Bq m^{-2}) and activity concentrations in plants and animal products (Bq kg^{-1} or L^{-1}). Conventional transfer coefficients, F_m or F_f, are defined as the equilibrium ratio between the activity concentration in milk or meat respectively divided by daily intake. Consequently, F_m will be expressed in units of days per litre (d L^{-1}) and F_f in days per kilogram (d kg^{-1}), i.e.:

$$F_m \text{ or } F_f = \frac{\text{activity concentration in food product (Bq L}^{-1}\text{ or Bq kg}^{-1})}{\text{activity in animal feed (Bq d}^{-1})}$$

Transfer coefficients are generally higher for young animals than for adults (Howard, 1989). Whilst in common usage, it should be remembered that single recommended values for transfer coefficients do not take account of the effect of interactions between radionuclides and stable analogues, or of homeostatic mechanisms or the effects of physiological status, all of which can significantly affect transfer for some radionuclides. This problem, and approaches to deal with such variation, will be discussed later with respect to radiostrontium.

The calculation of transfer coefficients requires information on both the daily intake of feed (in k d^{-1}) and the contamination level of the feed (Bq kg^{-1}). Estimates of these values can easily be obtained for housed farm animals fed under controlled conditions. However, in many semi-natural or natural environments animals graze in areas with highly variable soil and vegetation types and with characteristically highly variable transfer of radionuclides to plants. In addition, it is difficult to estimate the daily herbage intake of animals grazing semi-natural environments. For these reasons *aggregated transfer coefficients*, T_{ag}, have often been used, which integrate all processes from soil deposition, via plant uptake to animal products:

$$T_{ag} = \frac{\text{activity concentration in the food product (Bq kg}^{-1}\text{ or Bq L}^{-1}\text{)}}{\text{activity of deposit per unit area (Bq m}^{-2}\text{)}}$$

Aggregated transfer coefficients refer to the relationship between ground deposit and contamination of plants or animals when direct contamination is negligible, and consequently *they should not be used under conditions of continuous fallout.*

Aggregated transfer coefficients are easy to derive compared to the conventionally used transfer coefficients. However, in common with other transfer parameters the use of aggregated transfer coefficients requires specific knowledge of the environments in which they were obtained. Because of the complexity of semi-natural and natural environments and lack of knowledge, there are currently few alternatives to the use of T_{ag} values for making mid to long-term predictions in many types of semi-natural environments. However, to make mid to long-term predictions T_{ag} values need to be combined with estimates of effective ecological half-lives (i.e. the time required to reduce the activity concentration to one half the original level, including physical decay), to allow for changes in radionuclide uptake with time.

In this chapter the transfer of radiocaesium and radiostrontium to animals will be presented both as transfer coefficients and aggregated transfer coefficients. As mentioned, *aggregated* transfer coefficients should not be used in the initial period after fallout has been deposited when there is direct contamination of animal feed, and therefore transfer is affected by interception and weathering. In such circumstances the transfer coefficients should be used, taking due account of the bioavailability of the deposit. Transfer coefficients are most useful when there is good information on an animals daily intake of radionuclides (i.e. for farm animals where feeding schedules are well defined).

16.2.1 Farm animals

For most animal meat products, only radiocaesium is important as other radionuclides do not significantly contaminate muscle. Farm animal products are the most important foodstuff determining radiocaesium intake by the average consumer in the Nordic countries. The major potential source of radioiodine and radiostrontium to humans is milk and milk products. Of the different species, the smaller animals have the highest transfer of radiocaesium from fodder to meat and milk.

When considering long-term consequences of radioactive fallout special attention should be paid to animals grazing unimproved pasture and woodland, since the transfer of radiocaesium is higher in such environments compared to cultivated areas (Hove and Strand, 1990; Howard *et al.*, 1991). However, in such ecosystems the variation in soil types, vegetation species and transfer of radionuclides to the different species will make prediction of radionuclide intake by the animals difficult. In the Nordic countries, sheep and goats, and to some extent also cattle, commonly graze unimproved pastures during summer.

In addition to the higher transfer on unimproved pastures and woodlands, the production rate and subsequent ingestion of mushrooms will significantly influence radiocaesium activity concentrations in animal products from these areas. Fungal fruiting bodies appear in the autumn, and will cause elevated radiocaesium activity concentrations in animals which consume them because of the higher uptake of radiocaesium in many species of fungi compared to most vegetation species. Even if mushrooms are eaten in only small amounts they often contain high radiocaesium activity concentrations and therefore can influence radiocaesium intake to a considerable extent.

16.2.1.1 Sheep

When fed under controlled conditions, a transfer coefficient of 0.49 d kg^{-1} has been recommended to estimate the radiocaesium activity concentrations in lamb muscle (IAEA, 1994a). Since 1991, a Nordic study has been ongoing with the aim of quantifying transfer to sheep grazing a range of unimproved areas and the aggregated transfer coefficients obtained in this study are summarised in Table 16.1. Hove *et al.* (1994) concluded that the T_{ag}'s in the Nordic countries clearly fall within two groups: low values of 0.0005-0.003 m^2 kg^{-1} were observed in Denmark, Faeroe Islands and Finland, whilst higher values of between 0.015-0.047 m^2 kg^{-1} were observed in Iceland, Norway and Sweden. Because the grass-to-meat concentration ratios were of similar magnitudes in most of the study locations, the higher radiological vulnerability to radiocaesium observed in Finland (forest pasture), Iceland, Norway and Sweden must relate to differences between soil types and uptake of radiocaesium into herbage. In addition, higher aggregated transfer coefficients from organic soils may be accentuated by the ingestion of a range of more highly contaminated vegetation species by sheep grazing in forested or mountainous areas.

Table 16.1 *Aggregated transfer coefficients for radiocaesium from soil to lamb meat at 6 different sites in the Nordic countries in 1990-93 (from Hove et al., 1994).*

T_{ag} (m^2 kg^{-1} fw) Mean ± SD	Country
0.00055 ± 0.00064	Denmark
0.0030 ± 0.0017	Faeroe Island
0.00083 ± 0.00054	Finland
0.0148 ± 0.0006	Iceland
0.0390 ± 0.0037	Norway
0.047 ± 0.012	Sweden

In their study of nuclear weapons fallout data, Hove and Strand (1990) calculated T_{ag} values for transfer of ^{137}Cs from soil to lamb meat on unimproved pastures of 0.013-0.093 m^2 kg^{-1} in the period 1966-1972, and 0.07-0.10 m^2 kg^{-1} in 1986-1988. Similar T_{ag} values of 0.024-0.136 m^2 kg^{-1} were also observed for Chernobyl ^{137}Cs in 1986-1988; the highest value was obtained in 1988 when mushrooms were abundant. This demonstrates that potential intake of radiocaesium via fungal fruiting bodies needs to be taken into account when assessing the consequences of radiocaesium fallout. An assessment was made by Mehli (1996) of the importance of fruiting bodies compared to vegetation as sources of radiocaesium to sheep. The results showed that in years when fruiting bodies were abundant, 70-80% of the radiocaesium in the animals may be due to ingested mushrooms. In certain areas, the selective intake by sheep of highly contaminated ericaceous species will also increase radiocaesium activity concentrations in the meat.

16.2.1.2 Goats

Goats are kept for both milk and meat production. Norway has the largest number of goats of the Nordic countries, and large volumes of goat milk are used in whey cheese production.

i) Meat

For transfer from feed to goat meat, Hansen and Hove (1991) reported the transfer coefficient, F_f, for ionic radiocaesium to be 0.23 d kg^{-1}.

ii) Milk

Radiocaesium

The transfer of radiocaesium from different feeds to goat milk has been investigated by Hansen and Hove (1991). For hay harvested in the period 1986-1989, they found lower F_m values in 1986 of 0.042±0.007 d L^{-1}, compared with those from 1987-89 which ranged from 0.091±0.018 d L^{-1} to 0.124±0.023 d L^{-1}. The increase in the transfer coefficients were interpreted as a consequence of higher bioavailability of radiocaesium in the later years. Lower F_m values were observed for contaminated willow bark, and soil fed to goats.

In a summary of aggregated transfer coefficients to goat milk in Norway, Strand (1994) reported T_{ag} values ranging from 0.008-0.03 m^2 kg^{-1}.

To demonstrate the effect of differences in pasture type on the radiocaesium levels in goat milk, Garmo and Hansen (1993) grazed goats on meadow and willow pastures. The aggregated transfer coefficients were 0.0002 and 0.001 m^2 kg^{-1} respectively. The higher value on willow pasture was due to both higher transfer of radiocaesium from soil to plants, and the presence of plant species with an higher uptake of radiocaesium on this pasture compared to the meadow. Data on milk from goat grazing natural mountain pastures in Northern Norway (Troms county) in 1987-88 gave T_{ag} values of 0.002-0.004 m^2 kg^{-1} for a period when ingestion of mushrooms was negligible (Hove and Strand, 1990).

When mushrooms are abundant the T_{ag} may increase 2-4 fold (Hove and Strand, 1990; Hove *et al.*, 1990). This is one probable reason for the higher T_{ag} values of 0.011 and 0.014 m^2 kg^{-1} found by Strand and Hove (1996) for the years 1993-94 in two mountain areas in Southern Norway.

Radiostrontium

There is much less information available on radiostrontium transfer to goat milk than there is for radiocaesium, with Coughtrey (1990) giving an expected model prediction value of 0.056 d L^{-1} after 100 d, based on the small amount of available information. Recent studies have provided more relevant data (Howard *et al.*, 1995a; Beresford *et al.*, 1997; Crout *et al.*, in press). A model of radiostrontium transfer in dairy goats was developed which was based on calcium metabolism, because radiostrontium behaviour in animals is dependent on calcium. Assuming a bodyweight of 55kg, a calcium intake of 10 g d^{-1} (about twice requirement) and a milk yield of 1.5 L d^{-1}, the model predicts an F_m value of 0.024 d L^{-1}. Individual measurements of F_m for $^{85}SrCl$ administered to lactating goats ranged from 0.0038 to 0.033 d L^{-1}, but the higher values were obtained from goats in negative calcium balance. Later studies with goats receiving 12 g d^{-1} Ca measured F_m values of 0.02 d L^{-1} (Beresford *et al.*, 1997). Further consideration of the relationship between calcium intake and radiostrontium transfer to milk is given in section 16.2.1.3.

16.2.1.3 Cattle

Transfer of radiocaesium to cattle milk and meat is lower than that to sheep and goat products. For instance, Strand and Hove (1996) found that the aggregated transfer coefficient to goat milk was 2-4 fold higher than that for cow milk.

i) Meat

There have been few studies on aggregated transfer coefficients to beef, reflecting the generally low rate of beef production from unimproved grazing. In a study in Valdres, Norway, radiocaesium activity concentrations in both milk and meat were measured in an experimental herd grazing unimproved mountain pastures. The average T_{ag} value for beef from these cattle were estimated to be 0.006 $m^2\ kg^{-1}$.

ii) Milk

Radiocaesium

In the above study in Norway, the range of T_{ag} values for milk was 0.003-0.0045 $m^2\ kg^{-1}$. The difference between the T_{ag} values for milk and beef are smaller than what would be expected from the transfer coefficient values of 0.0079 d kg^{-1} and 0.051 d kg^{-1} for cow milk and meat (beef) respectively, given by the IAEA (1994a).

The vulnerability of cow milk production in the different Nordic countries to ^{137}Cs fallout has been studied by Hansen and Andersson (1994). Their results are summarised below.

The mean transfer coefficients ranged from 0.0045 to 0.0285 d L^{-1}, with an overall mean of 0.0094 ± 0.0132 d L^{-1}. These values are in agreement with those reported in the literature (Hansen and Andersson, 1994). The transfer of ^{137}Cs from vegetation to cow milk did not increase significantly from 1986 to 1992, although data from some individual farms indicated a tendency of increasing values. This is in contrast to other studies showing increased transfer coefficients to lamb meat and goat milk after the first harvest following the deposition of fallout (Howard *et al.,* 1989; Hansen and Hove, 1991).

When estimating mean T_{ag} values for whole countries, Hansen and Andersson (1994) found that T_{ag}'s decreased with time after 1986 for all countries. The T_{ag} was lowest for Denmark, Finland and Sweden and about 2-10 times higher for the Faeroe Islands, Iceland and Norway. In 1987, the T_{ag} values were about 0.0005, 0.0011 and 0.0006 $m^2\ L^{-1}$ for Denmark, Finland and Sweden, respectively, whilst those for the Faeroe Islands, Iceland and Norway were about 0.0043, 0.0016 and 0.0023 $m^2\ L^{-1}$ respectively. By 1989, the values of the aggregated transfer had decrease to about 0.00025 $m^2\ L^{-1}$ for Denmark, Finland and Sweden, whereas the values for the Faeroe Islands, Iceland and Norway were about 0.0014 $m^2\ L^{-1}$. This indicates that cows' milk production in the Faeroe Islands, Iceland and Norway is considerably more sensitive to ^{137}Cs fallout than the other Nordic countries, which is consistent with the findings of Hove *et al.* (1994) for lamb meat discussed earlier. The only discrepancy between the two products, of a low T_{ag} value for cow milk compared to a high T_{ag} for lamb meat in Sweden, is because the particular flock of lambs studied grazed an uncultivated mountain pasture where high radiocaesium transfer values occurred. The reason for the other discrepancy in the Faeroe Islands is unclear, but is possibly due to interactions between soil types and the different production systems for cow milk and lamb meat (Hansen and Andersson, 1994).

Data on milk from some individual farms in Norway using unimproved pastures was also studied by Hansen and Andersson (1994). At these farms the aggregated transfer values were about 0.004 $m^2 L^{-1}$ which is higher than the dairy bulk milk discussed above. Less than 5% of cow milk in Norway is produced on uncultivated pastures; dairy milk is mainly produced on farms with intensive production of high quality roughage and concentrates. Assuming constant F_m values, the results indicated considerably higher transfer from soil to vegetation on uncultivated pastures than on cultivated pastures (Hansen and Andersson, 1994).

Radiostrontium
Few studies on radiostrontium transfer to animal products have been performed in the environment following the Chernobyl accident. This is probably due to two main reasons:
1) the low quantity of ^{90}Sr in the fallout (in Sweden the level of ^{90}Sr was < 1% of that of ^{137}Cs (Suomela and Melin, 1992)), and it's concomitantly low contribution to the total dose, and
2) the laborious and costly analysis needed for ^{90}Sr, which is a beta emitter (compared to the gamma emitting radionuclides such as ^{131}I, ^{134}Cs and ^{137}Cs).

Information is, however, available from studies in the USSR following the Kyshtym accident, the USA in the period of atmospheric nuclear weapons fallout and from recent studies within EC programmes. A range of transfer coefficients for radiostrontium transfer to cow milk have been reported, with 0.0028 predicted after 100 d of continuous feeding, using model calculations, by Coughtrey (1990), the IAEA (1994a) gave a reported range of 0.001 - 0.003 around this value. These values are largely based on pre-Chernobyl studies on radiostrontium transfer to milk.

However, transfer coefficients for particular radionuclides are not necessarily constant, and can vary due to a number of factors. Indeed, for radiostrontium, which has a stable analogue, calcium, which is an essential nutrient under homeostatic control, the recommendation of a single value of transfer coefficient is not valid. For radiostrontium, the absorption of its stable analogue, calcium, varies inversely with dietary calcium intake at a given calcium requirement. It has therefore been proposed that under normal ranges of calcium intake the transfer of radiostrontium to milk is likely to be inversely proportional to dietary calcium intake (Comar *et al.*, 1966). Using experimental data, Sirotkin (1978) reported a double-exponential relationship between dietary calcium intake and the transfer of radiostrontium to cows milk. Therefore the commonly quoted single value transfer coefficients for radiostrontium transfer to milk are not applicable across a wider range of dietary calcium intakes.

Recently, Howard *et al.* (1997) have derived a simple relationship for ruminants between the radiostrontium transfer coefficient (F_mSr) for cow milk and the calcium intake (I_{Ca}) where:

$$F_m Sr = \frac{0.11 x [Ca] milk}{I_{Ca}}$$

Using this relationship the transfer coefficient for radiostrontium can be predicted on the basis of the calcium intake of dairy animals.

To test the validity of the relationship Howard *et al.* (1997) collated available literature on radiostrontium transfer to cow milk and compared it to calcium intake (Fig. 16.1). This showed the expected relationship between F_mSr and calcium intake, assuming a [Ca]milk of 1.15 g kg^{-1}, and was similar to the trend reported independently by Sirotkin (1978).

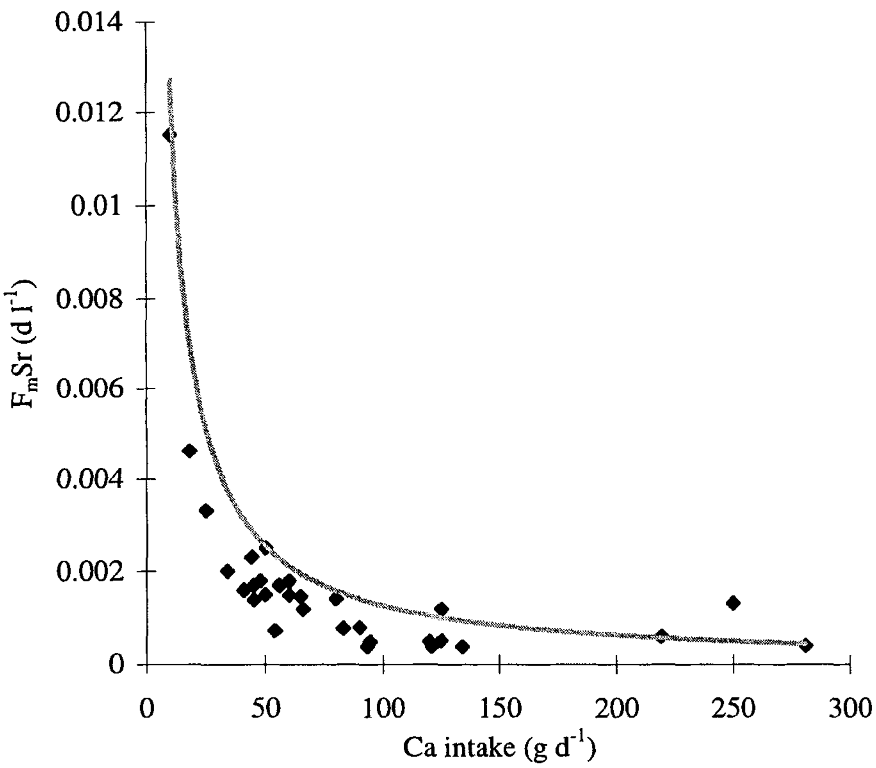


Figure 16.1 *Relationship between the observed (point data) and predicted transfer coefficient for cow milk and the daily calcium intake using the relationship derived by (Howard et al., 1997).*

16.2.1.4 Pigs

Following the Chernobyl accident fewer studies have been performed on the transfer of radionuclides to pigs than to most other farm animals. This is probably partly due to the feeding practice; most pigs are usually intensively managed, and their radionuclide intake is normally low and can be easily controlled..

In a post-Chernobyl, German study, pigs were fed contaminated whey (Voigt*et al.,* 1989). The F_f values derived for pork were in the range 0.35-0.45 d kg^{-1}, which was similar to that as for veal and sheep meat in the same study. No significant influence of different feeding vegetation strategies was found on the transfer of radiocaesium to pigs, with transfer coefficients to pork in the ranges 0.18-0.26 for potato-feeding and 0.17-0.33 d kg^{-1} for grain-feeding, which were both lower than those for whey-feeding (Voigt *et al.,* 1988). These values were in good agreement with earlier data in the literature, and were used as the basis of the F_f value recommended by the IAEA (1994a) of 0.24 d kg^{-1}.

16.2.2 Wild animals

Although products from farm animals are the most important pathway for radiocaesium intake by the average consumer, meat from animals such as moose, roe deer and reindeer may be important sources of radiocaesium for special groups of the population. Other deer species, such as red and white-tailed deer, are not considered in this report since their consumption rate in Nordic countries is low and few studies on transfer to these animals in the Nordic countries have been performed. Since the number of red deer killed has recently been increasing (Statistics Norway, 1996) attention should be paid to people with a high intake of meat from these animals in the case of a future contamination event.

16.2.2.1 Moose

In some areas within the Nordic countries moose constitute an important part of some peoples diet, and in Sweden moose hunting constitutes about 5-10% of the total meat consumed (Bergman *et al.,* 1991). Studies on transfer of radiocaesium to moose meat have mainly been conducted in Sweden and Finland.

The application of an aggregated transfer coefficients assumes a positive correlation between deposition and activity concentrations in the product for a given ecosystem and soil type. In areas where the deposition varies little, the ecosystem and soil type can have a considerable effect, as has been noted for moose (Nelin, 1995). Moose with access to farmland can have lower contamination levels than those without such access (Johanson and Bergström, 1994). In contrast, moose with access to wet bogs may become more highly contaminated (Nelin, 1995).

In summer, moose in Sweden were found to graze mainly on fireweed and birch, both of which are species with low levels of radiocaesium, but in autumn they change their diet to include more contaminated species such as bilberry (Johanson *et al.,* 1994). In the autumn, consumption of fungal fruiting bodies will also increase the radiocaesium intake by moose, as for other free ranging animals. Although Johanson *et al.* (1994) found on average only small amounts of mushrooms in rumen samples (1.3-1.6%), in one animal mushrooms constituted 20% of the rumen contents, and 1% of mushrooms in rumen contents was estimated to contribute about 25% of total radiocaesium intake. The contribution of mushrooms to radiocaesium intake will be higher in years when mushrooms are particularly abundant. Thus, whether hunting is conducted before or after mushrooms appear will have an important influence on the potential contribution of moose meat to radiocaesium transfer to humans.

Aggregated transfer coefficients for moose meat are summarised in Table 16.2. Few sources give individual data. To give an idea about the variation between individual moose from the same area, values from an area of Sweden where the mean radiocaesium activity concentration in moose meat was 750 Bq kg^{-1} (fw) and ranged from 100 to 3,000 Bq kg^{-1}. As demonstrated in the Table, calves have consistently higher mean radiocaesium activity concentrations than adults.

Table 16.2 *Aggregated transfer coefficients for radiocaesium to moose meat.*

T_{ag} (m^2 kg^{-1} fw)	Year	Reference
0.011-0.026 (all ages)	1979	Rantavaara (1982)
0.013	1985	Nylén (1996)
0.015 (calves)	1985	Bergman *et al.* (1991)
0.010 (adults)	1985	Bergman *et al.* (1991)
0.014 (calves)	1986*-91	Rantavaara (priv. comm.)
0.010 (adults)	1986*-91	Rantavaara (priv. comm.)
0.009-0.032 (calves)	1986-90	Bergman *et al.* (1991)
0.006-0.017 (adults)	1986-90	Bergman *et al.* (1991)
0.02	1986-88	Von Bothmer *et al.* (1990)
0.018-0.024	1986-91	Johanson and Bergström (1994)
0.009-0.019	1989	Johanson *et al.* (1994)
0.009-0.087 mean 0.032	1991	Nelin (1995)
0.006-0.02	1985-1996	Nylén (1996)

* 1986 data given in Rantavaara *et al.* (1987).

From the values presented in Table 16.2 a mean value of 0.02 m^2 kg^{-1} for transfer of radiocaesium to moose meat is recommended. Lower values close to 0.01 m^2 kg^{-1} were observed by Nylén (1996) in northerly counties in Sweden, but in these areas values closer to 0.02 m^2 kg^{-1} were also recorded in some years. The difference might be explained by the availability of mushrooms.

Johanson *et al.* (1991; 1994) also estimated transfer coefficients, F_f, for moose and found values ranging from 0.04-0.29 d kg^{-1}. These values may be applied instead of T_{ag} values when surface contamination of the vegetation is dominating.

16.2.2.2 Roe deer

Transfer of radiocaesium to roe deer is very variable both between and within countries, and, as for moose, access to farmland is one of several important factors influencing contamination levels of the animals. Roe deer have a variable diet, and consequent large seasonal variations in radiocaesium intake rates. As for many other free ranging animals, the highest radiocaesium activity concentrations occur in August-September when fungi are most abundant (e.g. Karlén *et al.*, 1991; Strandberg and Knudsen, 1994). Maximum values recorded by Karlén *et al.* (1991) in August-September were 3-4 times higher than those of the average spring and summer values. Dwarf-shrubs (heather, cowberry and bilberry) are also important contributors to roe deer radiocaesium intake in autumn (Karlén *et al.*, 1991).

Table 16.3 presents some aggregated transfer coefficient values from the literature. The values are largely derived from data where roe-deer have some access to farmland, and the results are therefore most relevant for such environments. The variability in activity concentrations is large, and the calculation of meaningful aggregated transfer coefficients is difficult. Compared to the data in Table 16.2, the data in Table 16.3 indicates that there is a higher transfer of radiocaesium to roe deer than to moose.

Table 16.3 Aggregated transfer coefficients for radiocaesium to roe deer.

T_{ag} ($m^2 kg^{-1}$ fw)	Year	Reference
0.15 (autumn)	1988-89	Karlén *et al.*(1991)
0.04 (rest of year)	1988-91	Johanson and Bergström (1994)
0.05 (annual mean)	1988-91	Johanson and Bergström (1994)
0.005-0.1	1987-94	Kiefer *et al.* (1996)

16.2.2.3 Reindeer

Reindeer is an important food source in the Nordic countries, especially for the Sami population. A comprehensive review, combining information on reindeer nutritional ecology and radiocaesium transfer is given by Gaare and Staaland (1994) which also discusses factors such as seasonal migration patterns.

Since the atmospheric nuclear weapons tests mostly in the 1950-60's it has been known that reindeer meat is especially sensitive to radiocaesium fallout, because reindeer ingest lichens as a major food source. During winter lichens may constitute 70-80% of the food intake, even during summer lichens constitute 10-20% of the intake (Gaare and Staaland, 1994). Lichens have a high ability to accumulate radiocaesium directly from precipitation; they do not absorb many nutrients, or radionuclides via root uptake.

Due to the importance of lichens in the diet, radiocaesium concentrations in reindeer will change according to changes in intake of lichens. There is therefore an increase in radiocaesium activity concentrations in reindeer in the autumn from late August/early September to November. During winter (November - early May) radiocaesium activity concentrations remain relatively constant, although there is a slow decrease due to reduced food intake. Then there is a faster decrease as they change to summer grazing (from May to end June), where they reach a stable low plateau which lasts to late August. The start of the increase in radiocaesium activity concentrations will depend on the time when mushrooms, which are also eaten by reindeer, appear.

The difference between maximum winter radiocaesium concentrations and minimum summer concentrations in reindeer varies from 2-4 fold (Strand and Hove, 1996) to 20 fold (Åhman and Åhman, 1994). For instance, Åhman and Åhman (1994) estimated T_{ag} values ranging from 0.6-1.1 $m^2 kg^{-1}$ during the winter of 1986-87, with minimum values of about 0.025 $m^2 kg^{-1}$ during summer. Other transfer values are given in Table 16.4.

Table 16.4 Aggregated transfer coefficients for radiocaesium transfer to reindeer.

T_{ag} (range or mean±SE)	Year, month	Reference
1-2	1987 Winter	Strand (1994)
0.12 ± 0.01	1986 September	Åhman and Åhman (1994)
0.44 ± 0.02	1986 Nov - Dec	Åhman and Åhman (1994)
0.76 ± 0.03	1987 Jan - Apr	Åhman and Åhman (1994)

Transfer of radiocaesium to reindeer meat is not uniformly high in all Nordic countries. Pálsson *et al.* (1994) measured the transfer of radiocaesium to reindeer in Iceland and found low ^{137}Cs activity concentrations in meat compared to that from other Nordic countries. The data presented by Pálsson *et al.* for 1990-91 corresponds to an aggregated transfer coefficient of 5.6 x 10^{-3} m^2 kg^{-1}. The difference from the values in other Nordic countries (e.g. those presented in Table 16.4) was attributed to a diet dominated by a particular lichen species which is much less highly contaminated than the reindeer lichen commonly consumed elsewhere.

In cases where the reindeer diet is known and can be quantified, transfer coefficients, F_f can be used to estimate the radiocaesium activity concentrations in animals. F_f values of 0.20-0.50 d kg^{-1} have been calculated from feeding experiments (Strand and Hove, 1996; Åhman, 1990).

16.3 Long term behaviour of radionuclide contamination

When radioactive contamination of the environment occurs it is important to know for how long foodstuffs will remain contaminated. This is determined by the physical decay rate of each radionuclides, and by various natural physical, chemical and biological processes which influence accumulation and elimination of radionuclides in the environment. One example is the rate of fixation in soil, which influences bioavailability and hence uptake by vegetation.

The effective ecological half-life (T_{eff}) is a parameter used to describe the long term decline in contamination levels in the environment. This is defined as the time over which the activity concentration in the product falls to one half of the initial value when no measures are taken to reduce the contamination levels. It is expressed as:

$$\frac{1}{T_{eff}} = \frac{1}{T_{phys}} + \frac{1}{T_{eco}}$$

where T_{phys} represents the physical half-life and T_{eco} the ecological half-life of the radionuclide being considered.

For short lived radionuclides the effective ecological half-life is essentially the same as the physical half-life. The ecological half-life will be more important for long lived radionuclides.

16.3.1 Farm animals

Long term behaviour of radionuclides and their decline in farm animals is of most concern when the products (e.g. milk and meat) are obtained from animals on unimproved pastures, since most animals are given feed from cultivated areas when they are housed. Fodder from cultivated areas will, in the long term, have generally low contamination levels (see Part 2), and consequently radionuclide activity concentrations in animals given these feedstuffs will also be low.

As earlier mentioned (section 16.2.1) variation in soil and vegetation types across unimproved pastures may be considerable, giving rise to differences in radionuclide intake by individual animals. Diet selection will also influence the long term behaviour of radionuclides in animals, since long term behaviour of radiocaesium differs between vegetation species (e.g. Haugen and Uhlen, 1992). Thus, changes with time might be observed in the relative radiocaesium activity concentrations in animal products from different grazed ecosystems. Additionally, appearance and abundance of mushrooms will influence the long term behaviour of radiocaesium in animal products, since many mushroom species have longer T_{eff} than those of vegetation species. Quantifying these effects is further complicated by the variable rate of production of fruiting bodies from year to year.

16.3.1.1 Sheep

Sheep are mainly slaughtered in the autumn, following summer grazing on mainly unimproved pastures. Due to differences in ecological half-lives between vegetation species due to species differences or underlying soil types, long term behaviour of radiocaesium in sheep may differ between grazing areas. In the autumn, appearance of mushrooms will strongly influence radiocaesium activity concentrations in sheep, and also contribute to differences between areas. The first estimates of effective ecological half-lives for sheep meat were reported by Hove and Strand (1990). On the basis of nuclear weapons tests fallout in Norway they calculated values of between 22-27 years (Table 16.5). In agreement with this long half-life, the Nordic studies during 1990-1993, compiled by Hove *et al.* (1994), only found significant decreases in lamb meat in Denmark from 1990 to 1992 (but there was no significant difference between 1992 and 1993). Within the other Nordic countries there was no significant decrease and the within year variation in most locations was much greater than that from year to year (Hove *et al.*, 1994). The authors also stressed that observations from only four years are of limited use for effective ecological half-life calculations.

***Table 16.5** Summary of effective ecological half-lives for ^{137}Cs in sheep.*

T_{eff}, years (range)	Time period	Reference
22-27	1959-1988	Hove and Strand (1990)
3.4	1990-1993	Rosén *et al.* (1995)
2.2[a] -8.5	1988-1995	Strand and Hove (1996)
2.4-7.8	1987-1995	Mehli (1996)[b]

[a] The lower value indicate estimates without the influence from ingested mushrooms.
[b] Range of half-lives without influence from ingested mushrooms.

Ingestion of contaminated mushrooms by animals will influence long term contamination in animal products, since the estimated T_{eff} will be affected by years with unusually high or low mushroom abundance. At one of their study sites, Strand and Hove (1996) found that the calculated half-life increased from 2.2 to 7 years when using sheep radiocaesium activity concentrations measured before and after mushrooms appeared respectively. Furthermore, Mehli (1996) found that radiocaesium from ingested mushrooms could contribute roughly 70-80% to the radiocaesium body burden of sheep, and that declines in sheep radiocaesium concentrations due to that in grazed vegetation should be about 7.8 years. In addition, some longer term data suggest that the T_{eff} is not constant but increases with time (see for instance Part 2) due to the influence of more than one environmental factor in determining decreases, and half life estimates will therefore be dependent on the number of years being considered.

16.3.1.2 Goats

Goats are milked throughout the grazing season, and the T_{eff} will be affected by the time needed for radiocaesium activity concentration in goat milk to become equilibrated with radiocaesium intake rates once the goats are released outdoors. Garmo (1996) reported that a T_{eff} based on milk radiocaesium activity concentration values measured at the beginning of the grazing season was lower than for values based on data from later in the grazing season. Such changes with time could be due to equilibration, or changes in diet or radiocaesium uptake by vegetation over the grazing period. Furthermore, the T_{eff} increased as additional data for extra years was incorporated into the calculation. Some of Garmo's results are shown in Table 16.6, together with other reported values. The estimates vary considerably, therefore assuming one mean T_{eff} value will a have limited validity.

***Table 16.6** Summary of effective ecological half-lives for ^{137}Cs in goat milk.*

Range of T_{eff}, years	Time period	Reference
15-25	1963-1988	Hove and Strand (1990)
3-14.5[a]	1986-1995	Garmo (1996)
5.7-30[b]	1986-1995	Strand and Hove (1996)

[a] The T_{eff} is calculated from the mean goat milk radiocaesium activity concentrations during the grazing period. It increases with time after the Chernobyl fallout.

[b] Data from some study sites show no decline. The value of 5.7 is based on data which are influenced by ingested mushrooms, thus also even shorter T_{eff} values may be obtained.

16.3.1.3 Cattle

Radiocaesium

The review by Hansen and Andersson (1994) summarised the Nordic studies on effective half-lives in cows milk (Table 16.7). The estimated T_{eff} for the transfer of ^{137}Cs to cows milk estimated, on the basis of whole countries, ranged from 1 to 2 years. The exception was Iceland, where the T_{eff} was estimated for global fallout (not Chernobyl fallout) and was much longer at 18.4 years. The mean T_{eff} values for individual farms in Sweden were similar to that of the whole country, whilst for the individual farms in Norway (where cattle graze unimproved pastures) longer half-lives were experienced (Hansen and Andersson, 1994; NRPA, unpublished).

The data for Chernobyl fallout from Norway indicate that T_{eff} in milk are considerably longer from cattle grazing on unimproved pastures compared to milk from cows grazing on cultivated pastures. The even longer T_{eff} observed for radiocaesium from nuclear weapons tests (Hove and Strand, 1990) suggests that there will also be an increased T_{eff} with time for the Chernobyl fallout (which also corresponds with the longer T_{eff} in Iceland).

Table 16.7 Estimated effective half-lives for ^{137}Cs in cows milk sampled from dairies and dry milk factories, or individual farms.

T_{eff} (±SD, range), years	Country (area)	Years	Reference
Milk from dairies or dry-milk factories			
1.6	Denmark	1987-1991	Hansen and Anderson (1994)
1.6	Faeroe Islands	1987-1991	"
1.4	Finland	1986-1991	"
18.4[a]	Iceland	1986-1992	"
2.0	Norway	1986-1991	"
2.3	Norway	1987-1991	"
1.0	Sweden	1986-1990	"
3[b]	Sweden	1986-1990	Suomela and Melin (1992)
Milk from individual farms			
1.0±1.4 (0.2-3.7)	Sweden	1986-1992	Hansen and Andersson (1994)
6.8, 7.7, 24	Norway[c]	1988-1995	NRPA, unpublished
5±3 (2.6-8.5)	Norway[d]	1988-1995	

[a] Global fallout (all other data are for Chernobyl fallout)
[b] Data from the diaries Tärnaby and Vittangi.
[c] Data from three individual farms in Valdres using unimproved pastures.
[d] Data from three individual farms at Brønnøysund using unimproved pastures.

No studies are known to the authors specifically studying long term decline in radiocaesium levels in cattle meat, but similar general declines as those reported in milk from diaries/dry-milk factories would be expected (see Table 16.7). Also in the limited cases of production on unimproved pastures the decline is expected to follow the T_{eff} in milk.

Radiostrontium
Few data are available on the effective ecological half-life of radiostrontium. Suomela and Melin (1992) observed a T_{eff} of about 3 years in milk from Swedish diaries in the first years after the Chernobyl accident. However, Suomela and Melin stated that the value was rather uncertain due to the low levels of ^{90}Sr.

16.3.2 Wild animals

16.3.2.1 Moose

The data on moose meat from Sweden and Finland show similar aggregated transfer values both before and after the Chernobyl accident, indicating that there is no significant decrease in radiocaesium activity concentrations in moose (Bergman *et al.*, 1991; Rantavaara *et al.*, 1987). One explanation for this is that no significant changes occurred in mean radiocaesium concentrations in fireweed and birch, which were two of the main constituents of moose diet in summer (Bergman *et al.*, 1991). Furthermore, more recent studies have also failed to detect any significant decrease in moose radiocaesium activity concentrations (Johanson and Bergström, 1994; Nylén, 1996). Since moose are hunted in the autumn, the year to year variation in mushroom abundance will also make predictions of the long term behaviour of radiocaesium difficult (see also section 16.3.1.1). Thus, currently available information indicates that the physical half-life of ^{137}Cs should be applied as an estimate of the T_{eff} of radiocaesium in moose meat.

16.3.2.2 Roe-deer

As for the other free ranging animals considered in this report, radiocaesium activity concentrations in roe deer exhibit large seasonal variations (e.g. Kiefer *et al.*, 1996), and occurrence of mushrooms in the autumn will influence the estimates of the T_{eff}. Karlén *et al.* (1991) have concluded that there had been no decrease in ^{137}Cs activity concentrations in roe-deer in central Sweden since 1986. In accordance with this, Johanson and Bergström (1994) found that the decline of the ^{137}Cs level in Swedish roe deer is determined by the physical half-life (see also section 16.3.2.1). In contrast to this, Kiefer *et al.* (1996) found that roe deer radiocaesium activity concentrations in Germany declined during 1986-1991 with a effective ecological half-life of about 3 years, but from 1991 to 1994 there was no further decrease.

16.3.2.3 Reindeer

Lichens dilute radiocaesium activity concentrations by new growth, but since the growth rate is low the biological half-life of radiocaesium in lichens is relatively long compared with other grazed vegetation. Consequently, since lichens are major feeds for reindeer, reindeer meat will have similarly long effective ecological half-lives.

Table 16.8 summarises some of the reported values of T_{eff}. Generally, the winter values are between 3-4 years, while the decrease is a little faster during summer due to generally shorter half-lives in grazed summer vegetation compared to lichens. Appearance of mushrooms may affect the long term decline of the reindeer radiocaesium concentration during late summer.

Table 16.8 Estimated effective ecological half-lives of ^{137}Cs in reindeer meat.

T_{eff}±SE	Year, season	Country	Reference
1.6	1987-1992, Aug.	Norway	Pedersen *et al.* (1993)
3.5	1987-1992, Nov.	Norway	Pedersen *et al.* (1993)
3.3	1987-1992, Apr.	Norway	Pedersen *et al.* (1993)
4-5	1987-1992	Finland	Rissanen and Rahola (1993)
3 2 ± 0.3	1986-92, Sep.	Sweden	Åhman and Åhman (1994)
3.2 ± 0.2	1986-92, Nov.-Dec.	Sweden	Åhman and Åhman (1994)
4.2 ± 0.4	1987-92, Jan.-Apr.	Sweden	Åhman and Åhman (1994)
3-5	1987-1994	Norway	Amundsen (1995)
2.8	1987-1995, summer	Norway	Strand and Hove (1996)
3.2	1987-1995, winter	Norway	Strand and Hove (1996)

The T_{eff} values reported in Table 16.8 are somewhat shorter than those observed after the nuclear bomb test fallout prior to the Chernobyl accident (e.g. Westerlund *et al.*, 1987). The difference is mainly due to continuous fallout which occurred over at least a decade after atmospheric bomb testing, compared to the single contamination event from Chernobyl (Strand, 1994).

17. Countermeasures

A wide variety of countermeasures are available for reducing intake of radionuclides by the population via animal products. They may be classified into three main groups:

I Restriction of foodstuffs from human consumption
- a) banning contaminated foodstuffs
- b) dietary advice

II Additives given to animals to reduce gut absorption of radionuclides
- a) given daily with concentrates
- b) within boli (radiocaesium only)
- c) within salt licks (radiocaesium)[1]
- d) stable analogue supplementation (radiostrontium only)

III Animal management
- a) providing clean fodder, or removing animals
- b) changing slaughter time
- c) changing from milk to meat production, or changing animal species

[1] Could also be used for ^{90}Sr by incorporating calcium, but this is not tested.

17.1 Restriction of foodstuffs from human consumption

17.1.1 Banning contaminated foodstuffs

Banning contaminated foodstuffs is a drastic countermeasure and is often the most costly alternative. However, it may, in some instances, be the only available measure, particularly in the early phase of an accident where full response procedures have not yet been developed. In addition, there may be inadequate information about the actual activity concentrations to make alternative decisions.

17.1.1.1 Effectiveness

This countermeasure is 100% effective in removing artificial radioactivity from the food chain. However, it also removes the product.

17.1.1.2 Practical aspects

The routine slaughtering system could be readily adapted to handle banned meat. The main difficulties in banning food may be the social and political consequences for an area which relies on these products and the subsequent economical problems of loss of public confidence in other products from the affected area. This may be important if the countermeasure needs to be applied over a longer period than only the early phase. The banning of food may be followed by restricted food production in the contaminated area, and this might have serious psychological and social effects in the community.

17.1.1.3 Cost

The cost of this countermeasures would comprise the value of the product which is being banned, and, if necessary, the provision of an alternative food. In Norway, condemnation of sheep and reindeer meat after the Chernobyl accident was estimated to cost 125,000 and 42,500 ECU per person-Sv respectively, and was the most expensive countermeasure applied (Brynildsen *et al.*, 1996).

A possible variation of this countermeasures is to divert animal products from human to animal feeding since contaminated animal products could be used as feed for other, possibly non-food producing animals. The use of animals products as animal fodder has previously been well established in countries where production in the agricultural system exceeds demand, but has been banned in some countries due to BSE. A moderate use of this countermeasures will probably be easy to implement, but limited in its capacity. The cost of this countermeasures is mainly the difference in value between animal products and animal fodder.

17.1.2 Dietary advice

Dietary advice would be in the form of guidance on upper limits for consumption of certain foodstuffs in a given period of time. The use of dietary advice may also be considered as a countermeasure for reclaiming abandoned areas. Application of this countermeasure, together with advice on food preparation in the household, can give significant reduction in individual doses, especially for critical groups. In the long term, particular attention should be focused on people with a high consumption rate of wild animals, as some of these products will have persistently high radiocaesium activity concentrations.

17.1.2.1 Effectiveness, practical aspects and costs.

It is difficult to assess the effectiveness, practical aspects and cost of this countermeasure. However dietary advice can be important in helping people to maintain their way of life and their work. This was observed in Norway after the Chernobyl accident for the reindeer breeding Sami people. By using dietary intake with food preparation advice, a yearly radiocaesium dose reduction of 50-85% was achieved in Norway (Strand *et al.,* 1992). Based on the experiences after the Chernobyl fallout, the cost effectiveness of this countermeasure in Norway has been estimated to be 5 ECU per person-Sv (Strand *et al.,* 1990).

17.2 Additives given to animals to reduce gut absorption of radionuclides

17.2.1 Radiocaesium binders

The addition of chemical binders to animal feedstuffs to make radiocaesium unavailable for gut uptake is a well-established method of preventing radiocaesium from contaminating the meat and milk of farm animals. The binders may be administered by mixing as a powder into concentrate feed, placing as a bolus in the stomach of the animal where it resides for a period of several weeks, or incorporating into salt licks for use with animals which graze in large open areas.

17.2.1.1 Effectiveness

Several clay minerals are known to be effective in reducing radiocaesium activity concentrations; bentonite is the most effective in reducing radiocaesium uptake in the gastro-intestinal tract of ruminants and non-ruminants in relation to the daily dosage (Voigt, 1993). A reduction of 50-60% (sometimes more) in the transfer of radiocaesium to the cows milk and meat, goat milk, and sheep and reindeer meat can be achieved with bentonite administration rates of about 0.5 g d^{-1} per kg of body weight (Hove and Ekern, 1988; Unsworth *et al.,* 1989; Hove *et al.,* 1991; Åhman, 1996). Higher reductions can be achieved by applying higher bentonite administration rates, but this is not recommended since the animals water requirements may increase (Åhman, 1996).

Hexacyanoferrate derivatives are more effective than bentonite in preventing radiocaesium uptake (Giese, 1989). Of these, the derivative ammonium-ferric (III)-cyano-ferrate (II) (AFCF or Giese salt) has been widely used. The reduction factors achieved using AFCF for transfer of radiocaesium to animal products in the literature, reviewed by Voigt (1993), are given in Table 17.1

Table 17.1 *Effectiveness of AFCF in reducing transfer of radiocaesium to different animal products (dosage 3 g d^{-1} for cows, 1-2 g d^{-1} for other animals (after Voigt, 1993).*

Animal	Product	Reduction factor
Cows	Milk	5-10
	Meat	4
Calves	Meat	11-12
Sheep	Meat	4-8
Lamb	Meat	2

The AFCF dose levels recommended for ruminants are generally 3 g d^{-1} to cows (6 mg kg^{-1} body weight) and 1-2 g d^{-1} (10-40 mg kg^{-1} body weight) for smaller ruminants. At these administration rates, reductions of tenfold or more can be achieved for both milk and meat (Hove, 1993), but it has been observed that doses as low as 1 mg kg^{-1} body weight can achieve a 50% reduction in radiocaesium uptake by reindeer feeding on contaminated lichens (Hove *et al.*, 1991).

The use of AFCF as a feed additive has recently been approved by the EC (EC, 1996).

17.2.1.2 Practical aspects

Bentonite or AFCF are fed to animals either as powder or included in concentrate mixtures; AFCF can also be included in a dilute form in salt licks or rumen boli. For housed animals, or for animals who are being kept under conditions where feeding is practical (i.e. for dairy animals which are milked daily), the easiest method of application is to incorporate the binder into concentrates. The binder can be incorporated into the concentrate without any significant change in the normal method of concentrate manufacturing. AFCF concentrations of 1 g kg^{-1} concentrate have been used in Norway. In addition, at a practical level on farms, it is not necessary to give any instruction to the farmer on how to use the binder since it is already in the concentrate and the normal use of concentrates is maintained.

The addition of bentonite or AFCF to concentrate is not always possible, for instance if concentrate is not available, or if it is not practical to administer because animals are freely grazing large areas over long periods of time. In such situations, sustained release boli or salt-licks may be used (Hove *et al.*, 1990; Hansen *et al.*, 1996). Of these two methods, the boli is the most efficient and control is achieved for all animals treated. After treating the animals, boli without a protective wax surface coating will last 4-8 weeks, whilst boli with a wax coating will last 10-12 weeks (Hansen *et al.*, 1996). The effectiveness of the two types of boli are similar; with 43-75% and 48-65% reduction in meat radiocaesium activity concentrations after 4-8 and 9-11 weeks respectively. Therefore the time of boli administration relative to the time of slaughter will determine which type of boli to use. Treatment may be done by the farmers, and it is recommended to administer 3 AFCF boli per animal to secure sufficient release rates of AFCF and to minimise the effect of losses caused through regurgitation. After administration, no further treatment is necessary.

Use of salt licks (with 25 g AFCF kg^{-1}) achieves less control of the effect on all animals since it is impossible to ensure that each individual animal uses the salt-lick sufficiently frequently and thereby ingests adequate amounts of AFCF. This causes a greater variation in the radiocaesium contamination of animal populations, but the average radiocaesium activity concentration in an flock can be reduced by about 50% (Hove *et al.*, 1990).

17.2.1.3 Costs

In Sweden, bentonite has been preferred as a feed additive as it is readily available and cheap (Åhman, 1996), whilst, due to the effectiveness, AFCF has been preferred in Norway since 1989. The costs associated with the use of AFCF in concentrates and salt licks are mostly incurred in the initial purchase, in Norway this has been estimated to be about 75 ECU per kg. One rumen bolus costs a maximum of 0.25 ECU (Hansen *et al.*, 1996). In addition, there is the cost of administration to animals, with the highest costs associated with the individual treatment of animals with boli. The compensation to the farmers for this work has been 5.6 ECU per animal (Brynildsen *et al.*, 1996). Overall, use of boli as a countermeasure for sheep was estimated to be 2.5 times as cost effective as special feeding (see next section). For reindeer, a compensation of 19 ECU was paid per animal.

Based on the price and the reduction effect, Brynildsen *et al.* (1996) estimated the cost per averted dose to the population. They found that the price per person-Sv was about 125 ECU for AFCF in dairy concentrates, 2,000-4,690 ECU for salt licks, and 12,500 ECU for boli in lambs.

A series of other hexacyanoferrate derivatives are also effective in reducing radiocaesium uptake in animals. AFCF is about twice as effective as these other chemicals, but is also considerably more expensive. The alternatives should be considered by relevant authorities for future use (Hove, 1993).

17.2.2 Radiostrontium analogue

In contrast to radiocaesium, currently no highly effective radiostrontium binder has been identified which can reduce radiostrontium uptake in the gastro-intestinal tract, although experiments in Norway have achieved a reduction of about 40% using Zeolite A (IAEA, 1994b). Furthermore, experiments by Hove *et al.* (in Howard *et al.*, 1995b) have shown that radiostrontium transfer to milk was not affected by administration of stable strontium.

Currently, calcium is one of the few practical dietary supplements which might be used as a countermeasure to reduce radiostrontium transfer to milk (Voigt, 1993; Howard *et al.*, 1995a). Published reduction factors, which predict the effect of giving additional calcium (reviewed by Voigt, 1993) are not generally applicable because the reduction achieved will depend on the current calcium intake and status of the animal. Howard *et al.* (1997) predicted the reduction factors which might be achieved in radiostrontium transfer to milk if different supplementary levels of calcium are added to diets of cows with different calcium intake rates. In the example shown in Figure 17.1, they estimated the fractional reduction in F_m for a cow with an unsupplemented dietary calcium intake of between 40 and 140g d^{-1}. This range of dietary calcium intake is representative of those to be expected in both Western and Eastern Europe.

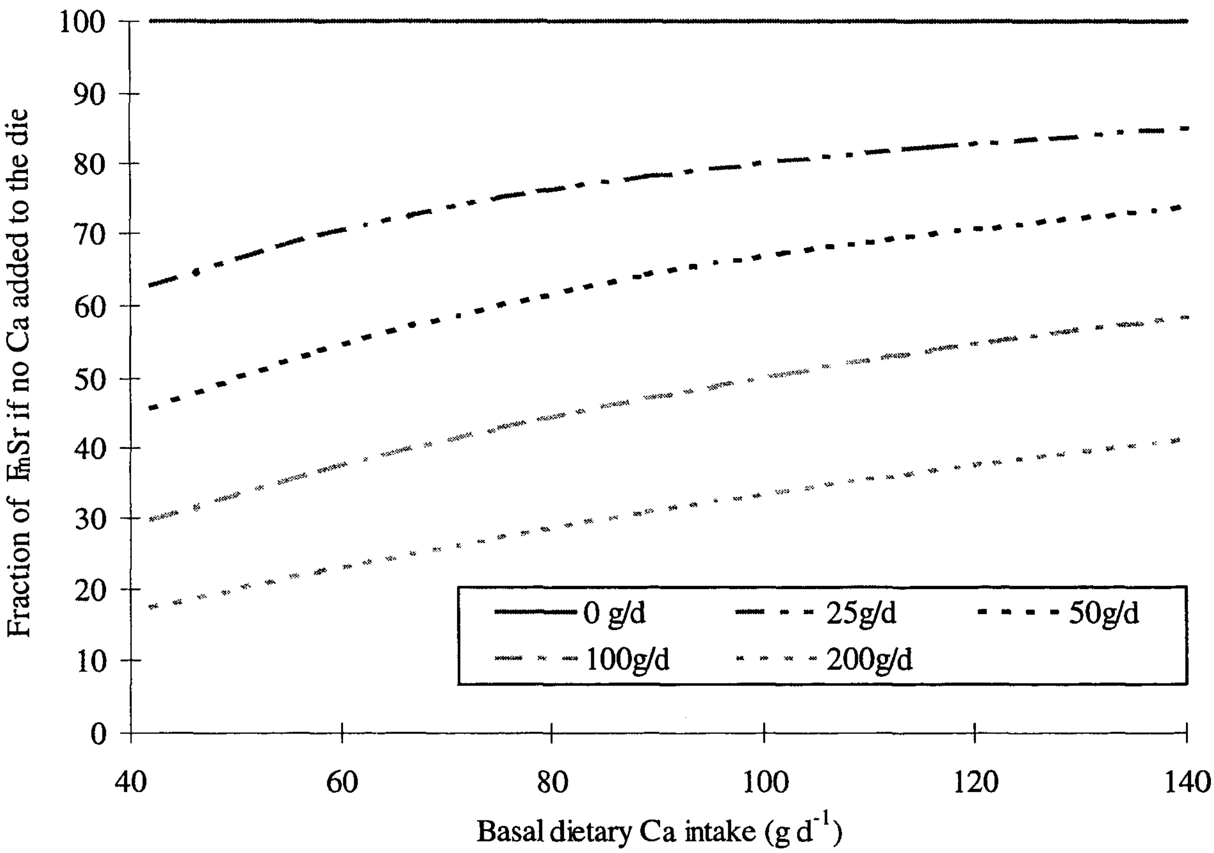


Figure 17.1 Effect of varying supplementary calcium intake on Fm for radiostrontium for dairy cattle receiving different basal dietary calcium intakes (Howard et al., 1997).

Supplementation rates of 100 to 200 g Ca d^{-1} are predicted to effectively reduce the transfer of radiostrontium to milk by at least 40-60 % for the range of dietary calcium intake rates considered. Advised maximum intake rates of calcium for ruminants are 1-2 % of the dry matter dietary intake; intake rates greater than this may reduce dry matter intake. A supplementation rate of 200 g for dairy cows would be at the upper limit of this range and would not therefore be advisable for prolonged periods. Maximum supplementation rates would obviously depend upon intakes from the basal diet. The availability or metabolism of other essential trace elements (eg. Cu, Zn and P) may be affected by too high a calcium supplementation.

17.3 Change in animal management

17.3.1 Providing clean fodder or removal of animals

Where animals become contaminated through grazing contaminated pastures, an effective countermeasure for all radionuclides is to provide uncontaminated ("clean") pasture or feedstuffs until the activity concentration of the radionuclides in herbage in contaminated areas has fallen to sufficiently low levels that milk and meat will not become too highly contaminated. The need for providing clean feeding will therefore be more of a long term problem in areas where agricultural practices involves the utilisation of unimproved pastures, since the decline in herbage contamination levels on these pastures is generally slower than in cultivated areas (Hove and Strand, 1990; Howard *et al.*, 1991).

17.3.1.1 Effectiveness

Providing clean fodder for stalled animals gives an almost 100% reduction in contamination levels. Also for grazing animals, such as free ranging sheep and reindeer, it is possible to achieve a reduction of almost 100% if clean feeding is continued for long enough after the animals are removed from the contaminated pasture. In this case the main factor determining the effectiveness of this measure will be the biological half-life of each radionuclide in the animals. For radiocaesium contamination the biological half-life when moving to non-contaminated pasture is about 3 weeks for both sheep and reindeer (Hove *et al.*, 1990; Åhman, 1996). In general, biological half times are longer for larger ruminants, such as cattle which have a biological half-life of roughly 30 days in meat (IAEA, 1994a). Thus, the reduction in contamination levels achieved depends on the time over which the clean fodder is used, and the species of animal.

Adding caesium binders (section 17.2.1) will enhance excretion of radiocaesium from the animals, thereby reducing the biological half-life. For reindeer adding binder reduced the half-life by about one week (Åhman, 1996).

17.3.1.2 Practical aspects

There are some limitations to this countermeasure. For instance, sufficient uncontaminated fodder may not be available in the event of widespread contamination. In addition, the provision of clean food may involve transport of food or animals and also management of the animals during the feeding period. There may also be limitations due to the climate (e.g. snow) and the lack of available housing (e.g. for sheep in late autumn/winter) or alternative pastures.

17.3.1.3 Costs

The duration of the clean feeding period is an important factor influencing the cost of this countermeasure. For grazing animals, two factors comprise the cost of this countermeasure; the original contamination level prior to clean feeding, and the biological half life for each animal species. In the event of widespread contamination clean feed may also cost.

In Norway a special feeding programme was established after the Chernobyl accident (Strand*et al.*, 1990; Brynildsen *et al.*, 1996). The compensation rates and arrangements for the farmers have changed over the years. Initially, they received 0.5 ECU d^{1} per sheep. From 1994, they received no compensation for the first week of special feeding of sheep and goats, then 0.63 ECU d^{-1} per animal from week 2 to 5, and 1 ECU d^{-1} per animal from week 6 to 8. The compensation is intended to cover extra expenditures connected to the feeding program, mainly additional work because the animals could not be slaughtered at the usual time. Extra compensation is paid to farmers with special difficulties in applying the feeding program, for instance, when alternative pastures are lacking or transport of animals is required to other areas. For reindeer, being fenced and fed, the compensation varied from 44 to 94 ECU per animal for the whole period, depending on the level of radiocaesium in the meat. For cattle 1 ECU d^{-1} per animal has been paid to carry out the feeding programme.

In terms of cost effectiveness in reducing dose to man, special feeding of sheep has been estimated to cost 35,000-41,000 ECU per person-Sv in Norway (Brynildsen *et al.,* 1996).

17.3.2 Changing slaughter time

In certain animal species radiocaesium contamination levels vary greatly with season. Therefore changing the normal time of slaughtering may have a pronounced effect on intake of radiocaesium by humans. The effectiveness and feasibility of this approach as a countermeasure is highly dependent on the environment and animal being considered.

One variant of this countermeasure is to collect the animals earlier than usual from unimproved pastures; this has been done with sheep in Norway in years when there has been a high abundance of mushrooms (Brynildsen *et al.,* 1996).

17.3.2.1 Effectiveness

The effectiveness of this countermeasure will depend on the animal species and season, and also on the time difference between the selected slaughter time and that of the normal slaughter. Hence, it is not possible to assign one general reduction factor for this countermeasure. In autumn, radiocaesium activity concentrations in reindeer can be expected to be 2-4 times lower than during winter (see section 16.2). In Sweden, a hunting season for roe deer buck in the spring was introduced which considerably reduced the transfer of ^{137}Cs to man. In 1990, the ^{137}Cs activity concentrations were about 5 times lower during the spring compared to the period for normal roe-deer hunting of August- September (Johanson *et al.,* 1991).

17.3.2.2 Practical aspects

For animals such as reindeer, it may be necessary to develop a new infrastructure when changing slaughter time, because of differences in arrangements for gathering and slaughtering animals during summertime compared with wintertime. Earlier collection of sheep from mountain and woodland pastures may also requires changes in management practices. Changing hunting seasons will have to take animal protection and hunting traditions into account.

17.3.2.3 Cost

There would be an income shortfall during the transition period. For herded animals, it may be necessary to compensate for the loss of increased body weight in the animals between summer and winter. For reindeer, there may be a roughly 10 kg difference between late summer weight for calf compared with winter weight, and in Norway early slaughter of reindeer calves was encouraged by paying 6.25 or 12.5 ECU per animal, depending on the time of slaughter (Brynildsen *et al.,* 1996). In addition, there will be the cost of providing the infrastructure. However this is a one time expense, which does not reoccur.

Normally, it should not be necessary to pay compensation for changing hunting seasons of wild animals.

17.3.3 Changing from milk to meat production, or changing animal species

The rate of transfer of radiocaesium from fodder to meat is generally 4-6 fold higher than that to milk (see chapter 16.2) so changing production strategies from meat to milk production could be considered. For radiostrontium, the transfer to meat is negligible compared to that for milk. Since transfer of radionuclides differs to different animal species (see chapter 16.2) altering the animal species, generally from small to larger ruminants, is also a method of reducing the transfer of radioactivity from plants to animals.

17.3.3.1 Effectiveness

The effect of changing from milk to meat production is a reduction in contamination of the end-product of more than 90% in the case of radiostrontium and up to 35% for radiocaesium.

17.3.3.2 Practical Aspects and Costs

Providing cattle or buying beef producing cattle cannot be accomplished rapidly, and requires significant changes in management practices. There will also be a difference between income between milk producing and meat producing system. Investment in the infrastructure needed for milk production is also lost. This countermeasure would not be feasible in all ecosystems.

18. An outline strategy for reduction of dose via animals.

Reducing dose via animals involves countermeasures that must be applied every year, because none of the measures described in Chapter 17 have any long term effects. The countermeasures involving reductions in contamination of vegetation which may be fed to animals, which often do have a long-term effect have been considered in Part 2. The different countermeasures and their effectiveness and cost are summarised in Table 18.1.

Table 18.1 *Effectiveness (% reduction) of countermeasures against radiocaesium contamination and cost (ECU/kg) for different animal products. The cost of banning is equal the monetary value of the product and assumes that there are other similar products easily available. The costs are mainly based on values given in Chapter 17, but some values have also been estimated from other Norwegian experiences with countermeasures after the Chernobyl fallout.*

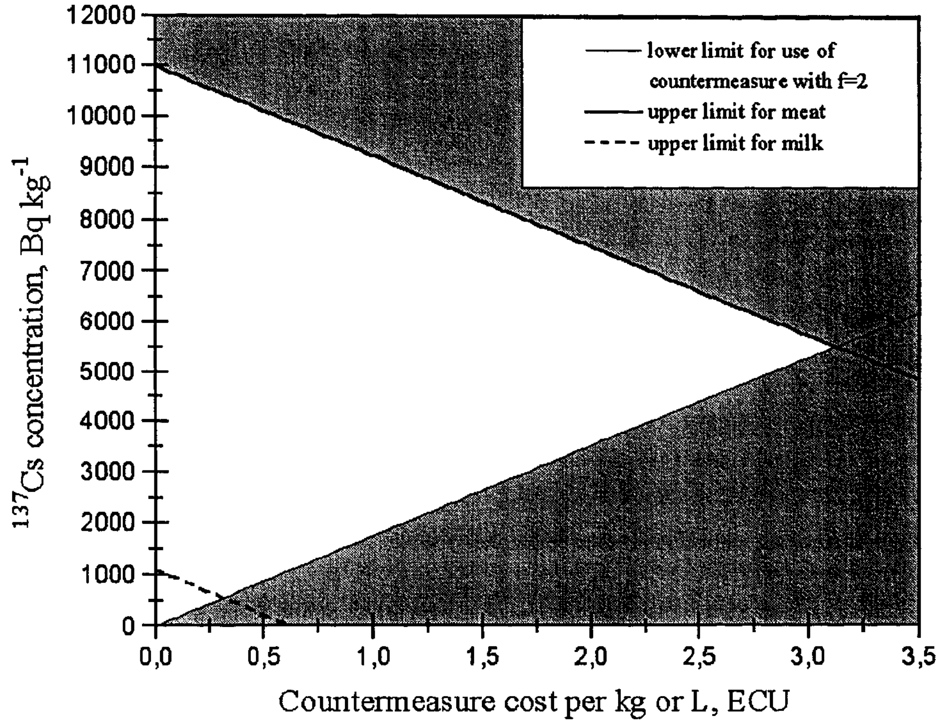
Food product	**Cattle milk**		**Cattle meat (beef)**		**Sheep meat**		**Goat milk**		**Moose meat**		**Roe deer meat**		**Reindeer meat**	
	Reduction %	**Cost** ECU/kg	**Reduction** %	**Cost** ECU/kg	**Reduction** %	**Cost** ECU/kg	**Reduction** %	**Cost** ECU/kg	**Reduction** %	**Cost** ECU/kg	**Reduction** %	**Cost** ECU/kg	**Reduction** %	**Cost** ECU/kg
Banning:	100	0.63	100	6.3	100	6.3	100	0.63	100	6.3	100	6.3	100	6.3
Special feeding:	100	0.059	50	0.12	50	0.66	100	0.10	-	-	-	-	50	6.0
Change slaughter/ hunting time	-	-	-	-	-	-	-	-	75	-	75	-	70	0.60
Binder:														
AFCF as powder or in concentrate	90	0.022	80	0.14	-	-	90	0.008	-	-	-	-	-	-
Bentonite as powder or in concentrate	50	0.004	45	0.024	-	-	50	0.001	-	-	-	-	-	-
AFCF in salt lick	-	-	-	-	50	0.076	50	0.004	50	0.081	50	0.078	50	0.077
AFCF in boli	70	0.030	70	0.22	50	0.54	50	0.031	-	-	-	-	50	0.49

Monetary cost of the countermeasures and the reduction achieved in radiological risk are important, but are not the only factors determining whether countermeasures should be applied. For monetary cost and the radiological risk reduction, Crick (1992) developed a theory showing that countermeasures should be applied for contamination activity concentrations in the range:

$$\left\langle \frac{P}{B} \cdot \frac{f}{f-1} \cdot \frac{B}{\alpha \bullet H}, f \cdot \left(1 - \frac{P}{B}\right) \cdot \frac{B}{\alpha \bullet H} \right\rangle$$

where P is the cost of the countermeasure per unit mass of the animal product (ECU kg^{-1})
B is the cost of providing alternative food when the product is banned (ECU kg^{-1})
f is the effectiveness of the countermeasure (the ratio of the dose without countermeasure to that with the measure)
α is the cost assigned to saving unit collective dose (in the Nordic countries the value of 100,000 USD per person-Sv is accepted (SSI, 1991)), and
H is the dose per unit intake (for ^{137}Cs - 1.3×10^{-8} Sv/Bq)

This may be used as a guide to when countermeasures could be applied. Using B values of 0.63 and 6.3 for milk and meat respectively (Table 18.1) it is possible to estimate the range of contamination levels where countermeasure use is justified. The range of contamination levels depending on the costs of the countermeasures are shown for countermeasures with differences in effectiveness in Fig. 18.1. For cases where all the contamination is removed (e.g. banning or special feeding for dairy animals) the countermeasure will be justified as long as the contamination levels in the products are above the value of P/(α H).



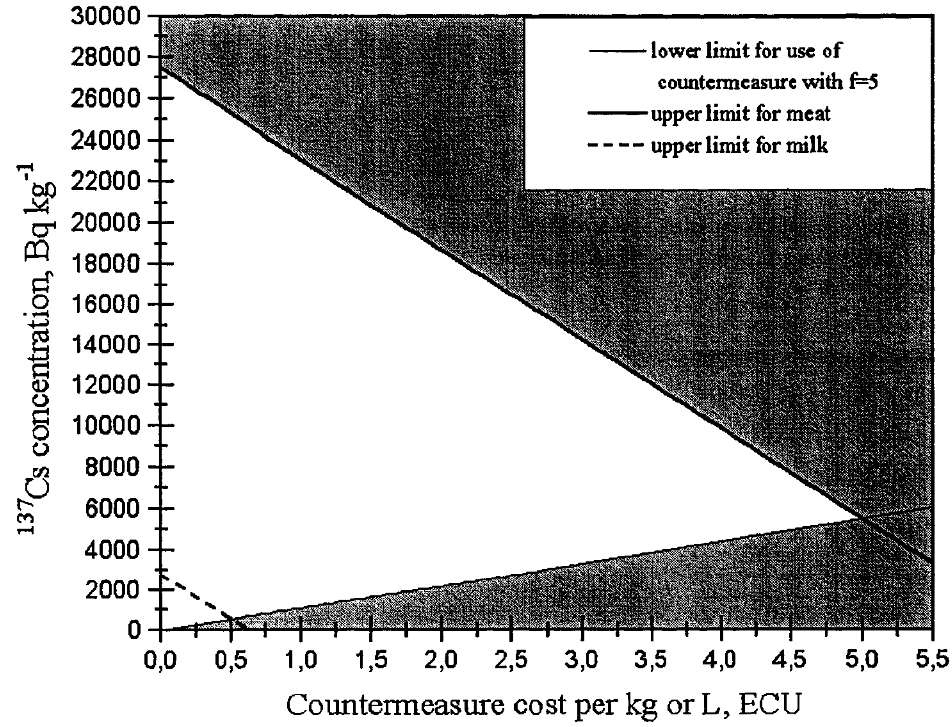


Figure 18.1 *Range of ^{137}Cs contamination levels in animal products and countermeasure costs (Table 18.1) where application of countermeasures are justified (white area) on the basis of monetary cost and achieved radiological risk reduction. The upper figure shows the situation for countermeasures with f=2 (i.e. 50% reduction, see Table 18.1), the lower figure for f=5 (i.e. 80% reduction). For meat ^{137}Cs concentrations and countermeasure costs within the whole white area are justified, while the values for milk should lie between the dashed line and the shaded area below.*

As an example of a strategy, we have assumed a fallout event with uniform contamination levels of 1 MBq m^{-2} of ^{137}Cs (in year 0). The first year after the fallout (year 1) measures will have been taken to reduce transfer to animals on most cultivated pastures as described in Part 2. Should radiocaesium activity concentrations above intervention levels in products from such farm animals persist, adding AFCF in concentrates to these animals would be recommended both because of practicability and costs.

For animals grazing unimproved pastures, estimates of radionuclide concentration activity concentrations and long term decline can be obtained using the aggregated transfer coefficients given in Chapter 16. This information is summarised in Table 18.2 for use in this example.

***Table 18.2** Summary of aggregated transfer values and effective ecological half-lives for radiocaesium in different animals and animal products from unimproved pastures (see Chapter 16). Some of the values given are intended for use in this example only. - indicates that the transfer is so low that contamination levels are not assumed to be cause problems.*

Animal and product	**Aggregated transfer coefficient, T_{ag} (m^2 kg^{-1})**	**Effective ecological half-life, T_{eff} (years)**
Cow milk	0.004	5
Cattle meat (beef)	0.024	5
Sheep meat	0.05[a]	8
Goat milk	0.01[b]	10
Moose meat	0.02	30
Roe deer meat	0.05[b]	30
Reindeer meat	0.8[c]	4

[a] This value is rather high for Denmark, Finland and the Faeroe Islands. On the other hand, it might also be higher in other parts of the Nordic countries when mushrooms are abundant.
[b] Varies considerable, also depending on mushroom abundance
[c] Not valid in Iceland

Applying these aggregated transfer coefficients, radiocaesium activity concentrations in animal products in the first year after the accident can be estimated. Table 18.3 gives the contamination levels both before and after different countermeasures have been used, and also shows the cost per person-Sv for the different countermeasures and products.

__Table 18.3__ Contamination activity concentrations (Bq kg^{-1} or L^{-1}) in animal products the first year (year 1) after deposition of 1 MBq/m^2. Also given are the achieved radiocaesium activity concentrations following application of different countermeasures, and the cost of the measure per reduced dose (ECU/person-Sv) obtained using the values in Table 18.1.

Food product	**Cattle milk**		**Cattle meat (beef)**		**Sheep meat**		**Goat milk**		**Moose meat**		**Roe deer meat**		**Reindeer meat**	
^{137}Cs (Bq kg^{-1} or L^{-1})	**4000**		**24000**		**50000**		**10000**		**20000**		**50000**		**800000**	
Countermeasure:	^{137}Cs level	Cost per person-Sv	^{137}Cs level	Cost per person-Sv	^{137}Cs level	Cost per person-Sv	^{137}Cs level	Cost per person-Sv	^{137}Cs level	Cost per person-Sv	^{137}Cs level	Cost per person-Sv	^{137}Cs level	Cost per person-Sv
Banning:	0	12000	0	20000	0	9600	0	4800	0	24000	0	9600	0	600
Special feeding:	0	1100	12000	740	25000	2000	0	770	-	-	-	-	400000	1150
Change slaughter/ hunting time	-	-	-	-	-	-	-	-	5000	-	12500	-	240000	82
Binder:	-	-	-	-	-	-	-	-	-	-	-	-	-	-
AFCF as powder or in concentrate	400	470	4800	550	-	-	1000	64	-	-	-	-	-	-
Bentonite as powder or in concentrate	2000	140	13200	170	-	-	5000	19	-	-	-	-	-	-
AFCF in salt lick	-	-	-	-	25000	230	5000	58	10000	620	25000	240	400000	15
AFCF in boli	1200	820	7200	1000	25000	1600	5000	480	-	-	-	-	400000	95

From the table, a priority order can be made for countermeasures for different products:

1. Applying bentonite as powder or as concentrate is the most cost effective countermeasure for daily «hand-fed» animals. However, with high radiocaesium activity concentrations bentonite may not be sufficiently effective. In such cases, AFCF as powder or as concentrate should be considered before special feeding.
2. For free ranging animals, salt licks with AFCF is most cost effective. When this is not applicable AFCF in boli should be considered before special feeding.

The cost effectiveness of a countermeasure varies between products, and a countermeasure strategy will also need to take contamination levels into account. It might, for instance, be necessary to combine different countermeasures to achieve contamination levels below intervention limits, for instance special feeding after animals have been grazing with boli or salt licks with AFCF.

As indicated in Table 18.3, radiocaesium activity concentrations can still persist above intervention levels after application of different countermeasures. Then dietary advice may be needed. Changing methods of preparing food (e.g. cooking, preserving) can significantly reduce radiocaesium concentrations, and reducing the intake of specific food products (like game, wild mushrooms) will also give lower radiocaesium intake by the population. These countermeasures have been shown to be very cost effective, but they also have larger impacts on peoples lifestyle.

Banning is the most expensive countermeasure, and would only be justified when contamination levels are so high that other countermeasures together with dietary advice cannot sufficiently reduce the radiocaesium intake in the population. And when you are not yet ready with other measures.

Since transfer and long term behaviour of radionuclides, as well as costs and effectiveness of countermeasures against the radionuclide contamination differs between animal products, the cost-effectiveness in terms of cost per saved population dose will vary with the passage of time. The cost per person-Sv will follow the equation:

$$R_{manSv} = \frac{P}{H \cdot E \cdot C_0} \cdot e^{\frac{\ln 2}{T_{eff}} \cdot (t-1)}$$

Here P is the cost of the countermeasure per unit mass of the animal product (ECU kg^{-1})
H is the dose per unit intake (for ^{137}Cs it is 1.3×10^{-8} Sv/Bq)
E is the effectiveness of the countermeasure (% reduction in radionuclide activity concentrations)
C_0 is the radionuclide activity concentration the first year after the fallout (year 1)
T_{eff} is the effective ecological half-life, and
t is years since the fallout occurred.

In the fallout scenario given here the 17th year after the accident will be the last year when countermeasures against caesium contamination of milk from unimproved areas will be needed. Two values can illustrate the changed cost effectiveness during these 16 years; compared to the values in Table 18.3 applying AFCF in concentrates for dairy cattle is 10 times more expensive, while the cost per person-Sv of salt licks for sheep is 4-5 times more expensive.

Wild animals
Radiocaesium contamination of wild animals can be a significant long term problem, and give higher doses to special groups in the population. Countermeasures appropriate for game have been discussed in the text earlier, and are also indicated in Table 18.1 and 18.2. In addition to the countermeasures shown in these tables, changing slaughter time and imposing restrictions on consumption of game animal products (including dietary advice and culinary preparation methods) are countermeasures that should be assessed.

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PART 4

FORESTS

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20. Contamination in Forests

Forests have the capacity to trap and retain radionuclides for a substantial period of time. The dynamic behaviour of nutrients, pollutants and radionuclides in forests is complex. The rotation period of a forest stand in the Nordic countries is about 100 years, whilst the time for decomposition of organic material in a forest environment can be several hundred years. This means that any countermeasure applied in the forest environment must have an effect for several decades, or be reapplied continuously for long periods of time.

The forest environment as a potential source of radiation to man has been emphasised in several reports. The nuclear weapons tests in the 1950s and 1960s initiated several studies on the transfer of radionuclides from lichen to reindeer and with that the impact on man (Ramzaiev et al., 1969; Mattsson, 1972). Higher external radiation doses to forest workers compared with agricultural workers have been observed in contaminated areas in the former USSR (Melin et al., 1991). There were also observations in the former USSR of damage to coniferous forests exposed to high radiation doses after the accident in Chernobyl (Kozubov et al., 1990; Arkhipov et al., 1995). In evaluating the consequences of radioactive contamination in a forest, it should not be forgotten that forests have a substantial economic value. Even if the radiation dose to man from forest products were infinitesimally small, a decrease in the trade of contaminated forest products cannot be excluded.

To mitigate the detrimental effect of a contaminated forest environment on man, and to minimise the economic loss in trade of contaminated forest products, it is necessary to understand the mechanisms of transfer of radionuclides through the forest environment. It must also be stressed that any countermeasure applied in the forest environment must be evaluated with respect to long, as well as short term, negative effects, before any decision about remedial action is taken.

In some forest-based industries (such as paper manufacturing), industry-specific processes will concentrate radionuclides by up to 200 times. These enrichment processes might cause radiological problems at certain points in the forest-based industries when using highly contaminated forest products (Holm et al., 1992).

20.1 Important radionuclides

Radionuclides that in the past have been considered in the forest environment are radiocaesium and radiostrontium. The root uptake of transuranic elements has been studied in forest stands growing on radioactive waste repositories, but the radiological consequences were not significant (Murphy, 1995).

In this part of the document, only options for reducing external dose will be considered. Foodstuffs from the forest, like berries and mushrooms, may be important sources of radiocaesium intake to man. Countermeasures to reduce radionuclide intake include restrictions on food gathering, dietary advises and food preparation.

Of the radionuclides studied in forests in the past, radiocaesium has been the main contributor to dose to man. In this document, only radiocaesium will be discussed since data on the impact of other radionuclides on man are too scarce for a proper evaluation.

The external exposure of man to radionuclides will diminish with time, partly because of radioactive decay. In addition, the translocation and circulation of radionuclides between soil, forest-floor vegetation and trees will contribute to the change in dose commitment to people living in a forest environment or working with forest products.

20.2 Deposition and retention of radionuclides

The distribution of intercepted radionuclides between different compartments in the forest environment depends largely on the density of the forest canopy. For instance, the canopy of a dense spruce stand has been shown to be able to intercept about 60-90% of the fallout (Melin et al., 1995; Tikhimirov, 1995). The corresponding figure for a leafless deciduous forest was much lower (Table 4.1.).

Table 4.1 *Relative interception (area) of dry deposited ^{137}Cs in different types of forest after the Chernobyl accident (time of sampling May 1986) (Melin et al., 1995).*

Stand	**Age of stand (year)**	**Stem density (stems ha^{-1})**	**Relative interception (m^{-2})**
Spruce	84	1336	1
Pine	139	1622	08
Beech (leafless)	87	687	04
Birch (leafless	68	573	04
Alder (leafless)	116	605	01

The high capacity of a coniferous forest to intercept fallout is independent of the season. However, the interception of radionuclides in a deciduous forest will depend on the stage of foliation, which in turn is governed by the season.

The interception of dry deposited radionuclides in forests will be higher than in grassed areas by about 25-35% (Tikhomirov, 1992). After wet deposition there was no difference between the two areas. A higher interception (2-5 times) at the edges of the forest compared to the inner part can be expected (Tikhomirov, 1995).

20.3 Long-term pathways of radionuclides in forests

During the first year after the initial interception there will be a rather fast transfer of radionuclides from the forest crown to the forest floor. This transfer is mainly caused by wash-off and by leaf fall. However, soil-to-plant transfer cannot be ignored during this period. In Table 4.2, available data on natural field losses from the aerial parts of the forest stand are summarised. It should be born in mind that the level of contamination in a tree is due to a combination of both interception and root uptake. The distribution of a radionuclide within the tree will vary according to the season.

In a later phase, the soil-to-plant transfer, together with the translocation of radionuclides in the plants, will be the dominant processes for radionuclide transfer in the forest system. A steady-state is established in the forest soil/plant community within ten years after deposition (Alexakhin, 1995, Tikhomirov, 1995).

Table 4.2 Initial transfer of Chernobyl radionuclides from a mixed birch/oak/pine forest stand to the forest floor (after Mamikhin et al., 1992)

Year after deposition	Distribution (%) tree stand/soil	
	Aerial parts of tree	Forest floor, soil
	border of the 30 km zone, Chernobyl condensed particles	
0 (initial)	60-90	10-40
0 (August)	17	83
1	8	92
2	6	94
3	6	94
4	5	95
	7 km from the Chernobyl NPP, fuel particles	
0 (initial)	60-90	10-40
1	0.3	99.7
2	0.2	99.8
3	0.6	99.4
4	2	98

The assimilation of radiocaesium and radiostrontium into the trees by leaves and roots is rather limited. It has been shown that 30 years after deposition (in the 1950s and 1960s) only 10% of recovered radiocaesium was found in the trees; 5% in the crown and 5% in the trunk. The corresponding figures for radiostrontium were 10% for total recovery and 8% in the trunk (Melin et al., 1995).

The incorporation of radionuclides into the trunk of the tree will increase with time from deposition until harvest. Apart from the assimilation of radionuclides into the trees, direct deposition on the trunk, branches and needles must be considered. The activity concentrations of radiocaesium and radiostrontium in the tree trunks with respect to the deposition rates are summarised in Tables 4.3 and 4.4 for a deposition of 1 kBq m^{-2}. For radiocaesium, the expected activity concentration in wood is in the range 0.1 - 3.9 Bq kg^{-1} and 0.1 to 4.2 Bq kg^{-1} for ^{90}Sr. For bark, the corresponding valves are 0.2 - 18.0 and 3 - 50 respectively.

Table 4.3 Relationship between deposition of ^{137}Cs and activity concentrations found in wood and bark

Species	Age of tree (years)	Deposition to ground ($kBq\ m^{-2}$)	Time elapsed since deposition (years)	Ratio: $Bq\ kg^{-1}/kBq\ m^{-2}$		Location	Reference
				(a) Wood	(b) Bark		
Birch	30	15	34	0.1	0.2	Kyshtym	Karavaeva (1995)
Birch	45	7	34	0.1	1.3	Kyshtym	Karavaeva (1995)
Birch	70	14	34	0.0	0.5	Kyshtym	Karavaeva (1995)
Oak			1	0.7		Chernobyl	Tikhomirov (1995)
Oak			5	0.8		Chernobyl	Tikhomirov (1995)
Pine			1	0.8		Chernobyl	Tikhomirov (1995)
Pine	17	126	2	0.5	14.0	Chernobyl	Vetrov (1995)
Pine	30	311	2	0.1	8.0	Chernobyl	Vetrov (1995)
Pine	85	3700	2	0.6	11.0	Chernobyl	Vetrov (1995)
Pine	85	2900	3	1.0	11.0	Chernobyl	Vetrov (1995)
Pine	85	3700	3	1.4	12.0	Chernobyl	Vetrov (1995)
Pine	17	126	4	2.5	14.0	Chernobyl	Vetrov (1995)
Pine	85	2900	4	0.7	10.0	Chernobyl	Vetrov (1995)
Pine	85	3700	4	1.0	10.0	Chernobyl	Vetrov (1995)
Pine			5	0.4		Chernobyl	Tikhomirov (1995)
Pine	17	126	5	3.9	18.0	Chernobyl	Vetrov (1995)
Pine	30	311	5	0.3	7.3	Chernobyl	Karavaeva (1995)
Pine	85	3700	5	1.5	11.0	Chernobyl	Vetrov (1995)
Pine	85	2900	5	1.9	8.0	Chernobyl	Vetrov(1995)
Pine	50	2	20	1.4	7.1	Sweden	Melin(1992)
Pine	30	15	34	0.1	0.4	Kyshtym	Karavaeva (1995)
Pine	45	7	34	0.1	1.6	Kyshtym	Karavaeva (1995)
Spruce			1	1.5		Chernobyl	Tikhomirov (1995)
Spruce			5	2.5		Chernobyl	Tikhomirov (1995)
RANGE				0.1-3.9	0.2-18.0		

Table 4.4 Relationship between deposition of ^{90}Sr and activity concentrations found in wood and bark

Species	Age of tree	Deposition ground (kBq m^{-2})	Time elapsed since deposition (years)	Ratio: Bq kg^{-1} kBq m^{-2}		Location	Reference
				(a) Wood	(b) Bark		
Birch			2		30	Kyshtym	Tikhomirov (1995)
Birch			3		15	Kyshtym	Tikhomirov (1995)
Birch			4		10	Kyshtym	Tikhomirov (1995)
Birch			13		4	Kyshtym	Tikhomirov (1995)
Birch	30	5	34	3.4	27	Kyshtym	Karavaeva (1995)
Birch	45	2	34	4.2	18	Kyshtym	Karavaeva (1995)
Birch	70	34	34	1.8	20	Kyshtym	Karavaeva (1995)
Pine			2	1.0	50	Kyshtym	Tikhomirov (1995)
Pine			3	0.4	40	Kyshtym	Tikhomirov (1995)
Pine			4	0.1	10	Kyshtym	Tikhomirov (1995)
Pine			13	0.5	3	Kyshtym	Tikhomirov (1995)
Pine	50	1	20	3.1	10	Sweden	Melin (1995)
Pine	30	5	34	1.8	9	Kyshtym	Karavaeva (1995)
Pine	45	2	34	1.5	11	Kyshtym	Karavaeva (1995)
Pine	70	34	34			Kyshtym	Karavaeva (1995)
RANGE				0.1-4.2	3-50		

20.4 Long-term changes in external exposure

The distribution of radionuclides will change from the time of deposition until the maturation and harvest of the stand. This will influence the external exposure to man. In addition, the physical half-life will contribute to external dose reduction up to the time of harvest, as illustrated in Figure 4.1 (based on the assumptions given in Table 4.5).

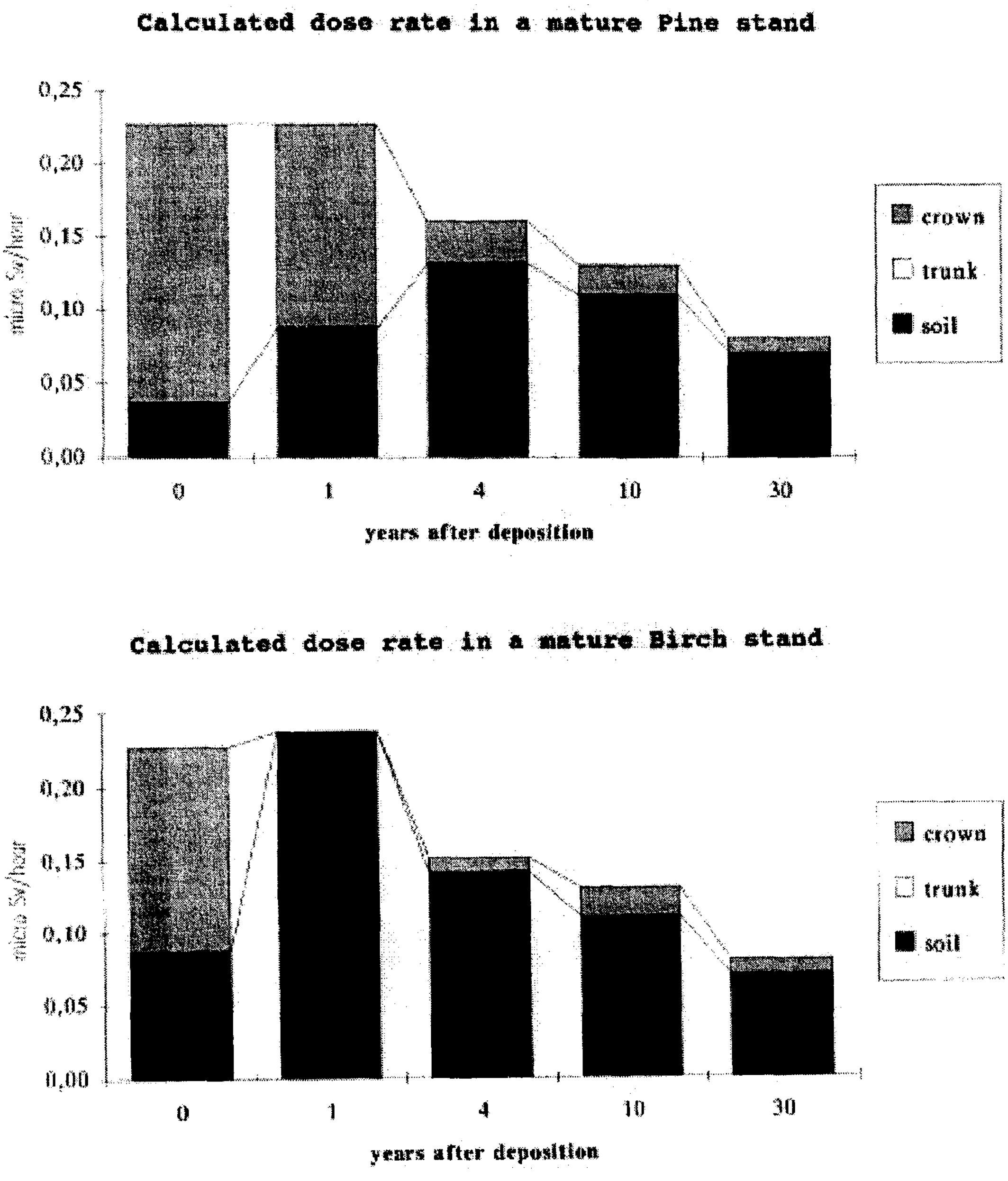

Figure 4.1 Calculated external dose rate in mature stands of pine and birch from deposited ^{137}Cs. Deposition rate 100 kBq m^{-2}.

Table 4.5 *The external dose rate calculated in Figure 4.1 are based on the assumptions given here.*

Stand density		Years after deposition, deposition occured when the trees were fully leafed							
		0		1		4		10	
		Cs-137 distribution, $Bq\ m^{-2}$	**Calculation of dose, source distribution[1)]**	**Cs-137 distribution, $Bq\ m^{-2}$**	**Calculation of dose, source distribution[1)]**	**Cs-137 distribution, $Bq\ m^{-2}$**	**Calculation of dose, source distribution[1)]**	**Cs-137 distribution, $Bq\ m^{-2}$**	**Calculation of dose, source distribution[1)]**
Pine, 1000 stems ha^{-1}	Crown	80	Plane surface[2)]	60	Plane surface[2)]	10	Plane surface[2)]	10	Plane surface[2)]
	Trunk	5	Point source	5	Point source	10	Point source	10	Point source
	Soil	15	Plane surface[3)]	35	Plane surface[3)]	80	Exponential depth[4)]	80	Exponential depth[4)]
Birch, 250 stems ha^{-1}	Crown	60	Plane surface[2)]	0	Plane surface[2)]	4	Plane surface[2)]	10	Plane surface[2)]
	Trunk	5	Point source	5	Point source	10	Point source	10	Point source
	Soil	35	Plane surface[3)]	95	Plane surface[3)]	86	Exponential depth[4)]	80	Exponential depth[4)]

[1] Finck (1992)
[2] Attenuation 0.9, 0.6 (primary photons) x 1.5 (build up factor)
[3] Linear relaxation depth 1 mm, soil density 1600 kg m^{-3}
[4] Linear relaxation depth 1 cm, soil density 1600 kg m^{-3}

21. Countermeasures in forests

When considering countermeasures in the forest environment, one should also bear in mind that they might also disturb the forest ecosystem and is likely to adversely affect productivity.

In this report only the influence of forest management procedures on the accumulation of radionuclides in different parts of the forest environment will be considered, together with radiation safety aspects for forest workers.

21.1 Clear-felling

Clear felling of a forested area within the first years after deposition can considerably contribute to a reduction in dose rate. For the examples given in Fig 4.1 a clear felling of the birch and pine stand could decrease the dose rate by 70 and 85% respectively over a period of 30 years if all the harvested material is removed. It has to be stressed that if clear felling is considered as a countermeasure it has to be carried out within the first year after deposition. If the forested area has to be decontaminated at a later stage other actions have to be taken such as removal of the litter layer, deep ploughing, covering with clean soil etc.

Clear felling and banning the harvested material will lead to a substantial loss in economical value of the forest. The economical loss is estimated to be within a range of 3000-12000 ECU ha^{-1} depending of stand age and productivity. In addition the felled material must be taken care of (see part 6).

The cost-effectiveness of clear-felling as countermeasure is dependent of the number of individuals concerned and the exposure time.

It must be stressed that clear-felling will increase the mineralisation of soil organic matter. Radionuclides bound in the soil organic matter may thus be released and eventually reach the ground water. There is also a risk of an increase in surface run-off following clear felling that might contaminate nearby surface water.

21.1.1 Ploughing and tilling of a clear-felled area

Ploughing and tilling the clear-filled area will further increase the rate of mineralisation of the soil organic matter. However, ploughing and tilling will usually decrease the radiation levels.

21.2 Processing of trees

Removing certain parts from the tree before processing to produce timber, board, paper etc., will reduce contamination levels in the products. For instance, taking off the bark can remove up to 50% of the contamination (Tables 4.3 and 4.4).

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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°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

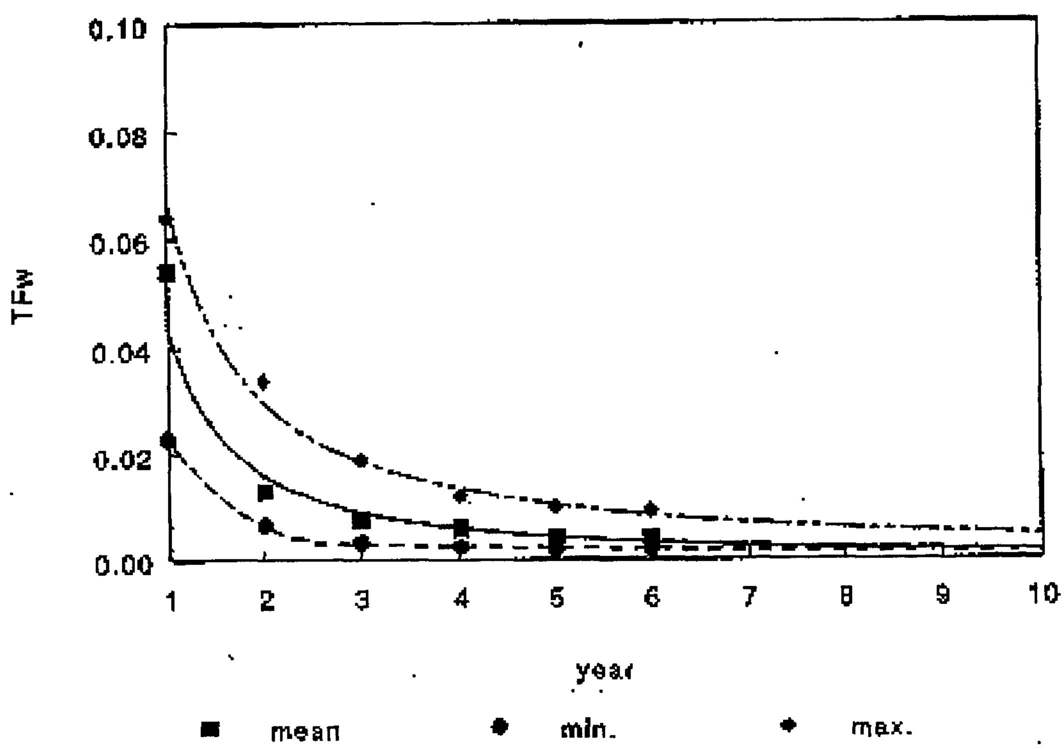

Figure 5.1 *Radiocaesium transfer coefficients with variation from deposition to surface water Bq m^{-3} / Bq m^{-2}.*

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):

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

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.

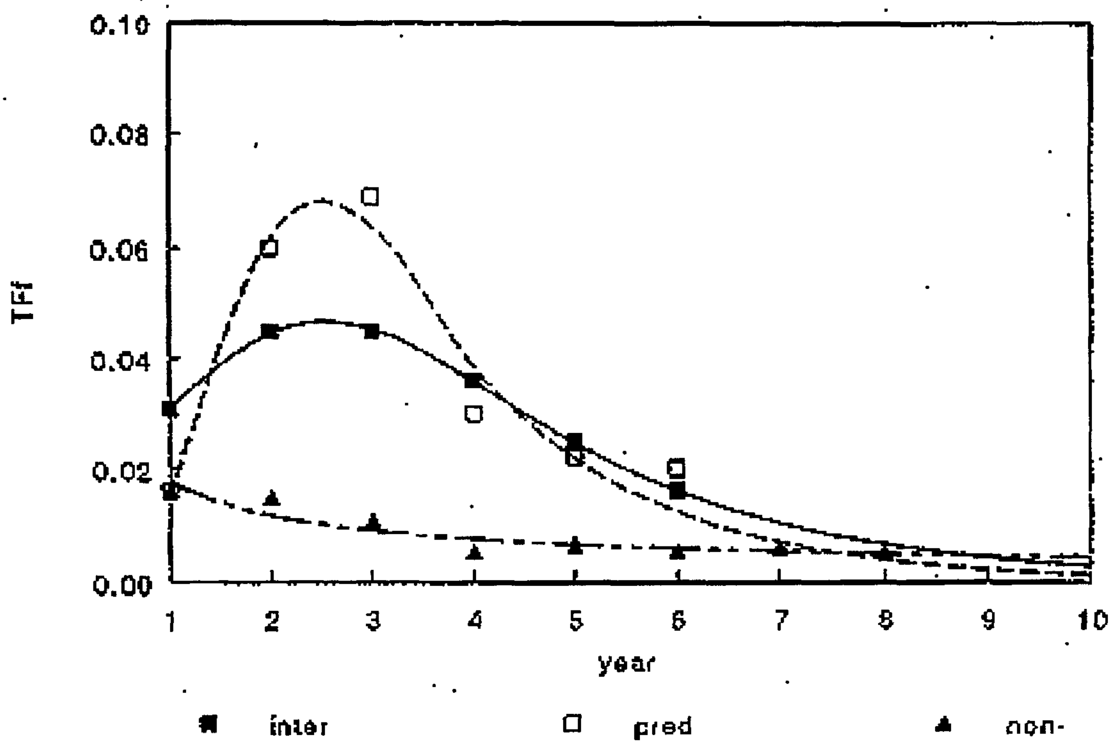

Figure 5.2 *Changes with time since deposition in radiocaesium transfer coefficients for three different types of fish (predatory, non-predatory and intermediate) in a large oligotrophic lake.*

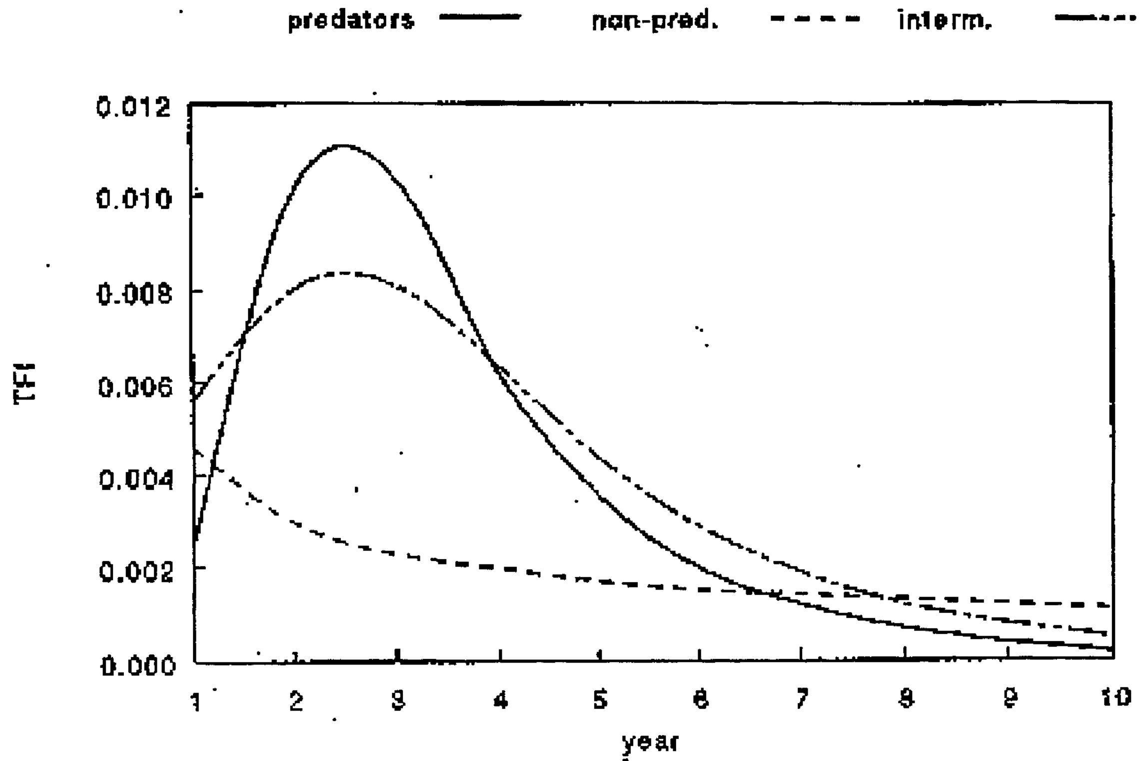

Figure 5.3 Changes with time since deposition in radiocaesium transfer coefficients for three different types of fish (predatory, non-predatory and intermediate) in a large eutrophic lake.

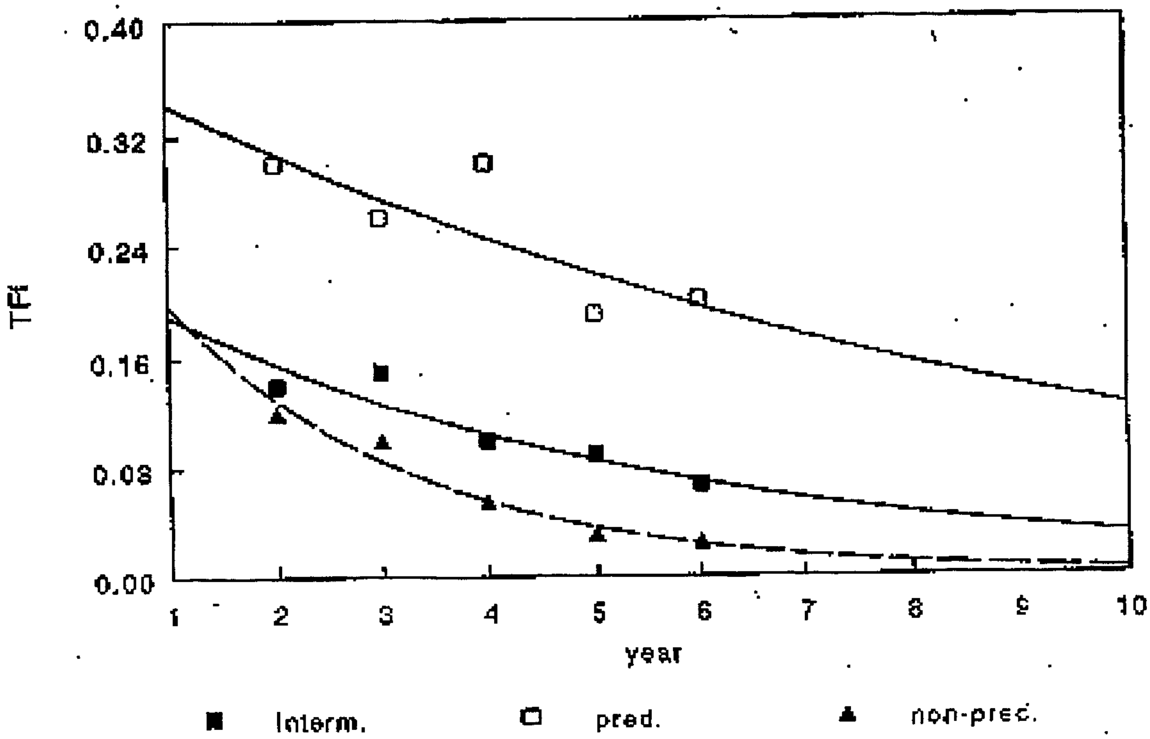

Figure 5.4 Changes with time since deposition in radiocaesium transfer coefficients for three different types of fish (predatory, non-predatory and intermediate) in a small oligotrophic lake with a long water residence time.

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 10^{th} 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.

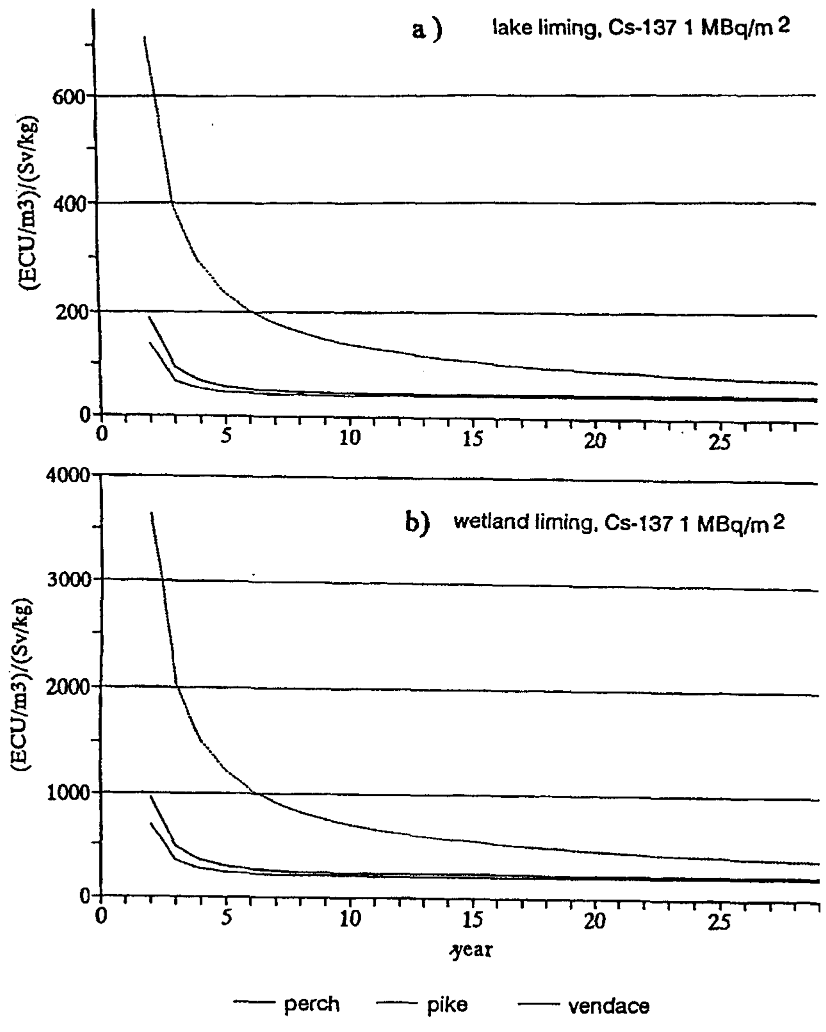

Figure 5.5 The estimated cost of the saved radiation dose as a function of time in three fish types: Predators (pike), non-predators (vendace) and intermediate (perch), in a case of a) lake liming, and b) wet land liming.

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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PART 6

MANAGEMENT AND DISPOSAL OF RADIOACTIVE WASTE FROM CLEAN-UP OPERATIONS

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26. Management and disposal of radioactive waste from clean-up operations

26.1 Introduction

Clean-up of large contaminated areas may create enormous amounts of radioactive waste which need to be safely disposed of. Disposal of the waste may include pre-treatment and transportation to a final repository.

There is much experience of the removal and disposal of large amounts of radioactive contaminated material from uranium mill tailings sites. For example, in Salt Lake City, USA, two million tons of radium-containing waste was transported 140 km by rail to a disposal site (US DOE, 1984). In Port Hope, Canada, 70,000 cubic meters of similar waste were moved by road to a disposal site 350 km away (Killey, 1985).

The disposal of the uranium mill tailings can be pre-planned, but an accident situation is quite different. In an emergency, decisions on how to deal with the waste from the clean-up may have to be made rapidly and disposal options may be limited. After the Chernobyl accident, large amounts of contaminated material (mainly soil and trees) were disposed of in shallow pits and surface mounds. Overall, approximately 4×10^6 m^3 of waste were distributed between about 800 disposal sites.

Because the amounts of waste after a major nuclear accident could be large, their final disposal may require large human and capital resources. Depending on the scale it is possible that the wastes will have to be placed in several final disposal sites. These are likely to be pits or surface mounds. Such repositories may need clay or concrete liners to prevent migration of the radionuclides from the disposal sites.

26.2 Amounts and activity concentrations of clean-up waste

In order to properly manage the reclamation procedure, the amounts and the activity concentrations of the wastes must be known in advance. In an actual accident, this will be done through a radiological survey of affected areas and evaluation of the clean-up required. To estimate these figures, different scenarios were considered in the project KAN2 of the Nordic Nuclear Safety Research Programme 1990-1993 (Lehto, 1993, 1994), in which all the most important types of wastes were considered. The following sections 23.3-23.7 summarises the results of this study.

26.3 Estimation of amounts and activity concentrations

The scenarios were based on a theoretical accident which simulated a worst case accident at a 700 MW boiling water reactor (BWR) plant (Lahtinen, 1988). This model accident gives three areas with different contamination levels for ^{137}Cs, ^{90}Sr and ^{239}Pu arising from both wet and dry deposition (Table 6.1). For the most heavily contaminated areas (Area 1), no estimation of the mean contamination level could be made and, therefore, the activity concentrations for these areas given later represent minimum values. Urban areas were not included in the calculations of Area 1 since, in the Nordic countries, there are no nuclear power plants in close proximity to towns or cities. In actual accident situations, the distribution of contamination will most probably be much more diverse than assumed in the model accident. Therefore, it should be borne in mind that the figures given below represent average values.

Table 6.1 *Contamination levels and areas of contaminated lands for dry and wet (in parenthesis) deposition after a major theoretical accident at a 700 MW BWR Plant*

Area and contamination level	**Estimated mean values of contamination levels in forest, urban and agricultural ereas**		
	Forest	**Urban**	**Agricultural**
Area 1. 1.8 km^2 ($21\ km^2$)			
^{137}Cs > 100 MBq m^{-2}	-[a]	-	-
^{90}Sr > 77 MBq m^{-2}	-	-	-
^{239}Pu > 22 $kBqm^{-2}$	-	-	-
Area 2. 32 km^2 ($403\ km^2$)			
^{137}Cs 10-100 MBq m^{-2}	20 MBq m^{-2}	4 MBq m^{-2}	8 MBq m^{-2}
^{90}Sr 7.7-77 MBq m^{-2}	15 MBq m^{-2}	3 MBq m^{-2}	6 MBq m^{-2}
^{239}Pu 2.2-22 kBq m^{-2}	5 kBq m^{-2}	1 kBq m^{-2}	2 kBq m^{-2}
Area 3. 2300 km^2 ($2210\ km^2$)			
^{137}Cs -10 MBq m^{-2}	2 MBq m^{-2}	0.4 MBq m^{-2}	0.8 MBq m^{-2}
^{90}Sr 0.8-7.7 MBq m^{-2}	1.5 MBq m^{-2}	0.3 MBq m^{-2}	0.6 MBq m^{-2}
^{239}Pu 0.2-2.2 kBq m^{-2}	0.5 kBq m^{-2}	0.1 kBq m^{-2}	0.2 kBq m^{-2}

[a] For the most heavily contaminated areas (Area 1), no estimation of the mean contamination level could be made.

The amounts of clean-up wastes were calculated in three ways:

a) Amounts per square metre or hectare for various clean-up methods. For example, removal of the upper 5 cm layer of soil from a field creates 600 m^3 of solid waste per hectare, including a 20% increase in volume due to handling.

b) Average amounts per square kilometre taking into account typical proportions of different types of areas in Nordic Countries. For example, in Western Norway 3.0 % of the land area is cultivated. Therefore, clean-up of all fields in this region by removing a 5 cm upper layer of soil from fields will create about 1800 m^3 of waste per square kilometre.

c) Total amounts of clean-up wastes for the areas obtained from the model accident. For example, the removal of 5 cm upper layer of soil from all the fields in an area of 403 km^2 in Western Norway creates 725,000 m^3 of waste.

Calculation of the activity concentrations are based on the following data:

a) Contamination levels obtained from the model accident.
b) Distribution of contamination on different surfaces.

c) Decontamination factors obtained with different clean-up measures.
d) Amounts of wastes.

In the following text, the activity concentrations of clean-up wastes are given for ^{137}Cs only. For ^{90}Sr and ^{239}Pu the activity concentrations can be calculated from the activity ratios $^{137}Cs/^{90}Sr$ and $^{137}Cs/^{239}Pu$ from Table 6.1.

There are a number of factors that affect the amounts and activity concentrations of clean-up waste. Most important of these are the contamination levels and the area of affected land. The level of the clean-up work, however, will finally determine the amounts of waste created. The time between the deposition and the clean-up may have an effect since weathering can displace the contamination from upper to lower layers. The season of the year when the clean-up takes place will also be an important consideration. These factors were taken into account in the scenarios by considering both dry and wet deposition and by considering clean-up both immediately after deposition and two years after. Wastes originating from the clean-up in winter time were only dealt with in a limited manner due to the lack of basic data for this situation.

In this study no evaluation of different clean-up measures was made regarding their clean-up efficiencies, costs and efficiencies to reduce doses. Instead nearly all possible waste forms were considered, and the data generated can be used as basic data for further cost-benefit and decision analysis.

26.4 Clean-up wastes from urban areas

All the clean-up measures in urban areas (described in Part 1), result in the formation of large amounts of radioactive waste. Table 6.2 shows the amounts and activity concentrations of clean-up wastes (per square kilometre) from a typical city area for a deposition of 4 MBq m^{-2} for a clean-up operation beginning one month after deposition. The most voluminous wastes are soil, asphalt and the effluents from fire-hosing streets and walls. The activity concentration is highest in cut grass and road dust if the clean-up is carried out soon after deposition.

Table 6.2 Projected clean-up wastes from a typical city area for a deposition event of 4 MBq m^{-2} of ^{137}Cs.

Clean-up method	Mass of waste (t km^{-2})	Volume of waste (m^3 km^{-2})	^{137}Cs concentration in waste	
			Dry Deposition	Wet Deposition
Soil removal (10 cm)	50,000	30,000	16.0 MBq t^{-1}	16.0 MBq t^{-1}
Grass cutting	25	60	32,000 MBq t^{-1}	6,400 MBq t^{-1}
Road planing	1,000	6,000	1,000 MBq t^{-1}	1,300 MBq t^{-1}
Firehosing roads (solid)	40	20	6,400 MBq t^{-1}	8,000 MBq t^{-1}
(liquid)	50,000	50,000	0.04 MBq t^{-1}	0.04 MBq m^{-3}
Firehosing + NH_4NO_3 walls	40,000	40,000	4.0 MBq m^{-3}	0.4 MBq m^{-3}
Sandblasting walls (solid)	2,000	1,000	64 MBq t^{-1}	6.4 MBq t^{-1}
(liquid)	4,000	4,000	2.0 MBq m^{-3}	0.2 MBq m^{-3}
Road sweeping	50	50	8,000 MBq t^{-1}	10,000 MBq t^{-1}

Table 6.3 shows the amounts of clean-up wastes calculated for the whole of Copenhagen and its closest suburbs.

Table 6.3 Estimated mass and volume of wastes from a projected clean-up of Copenhagen and its closest suburbs for a deposition of 4 MBq m^{-2} of ^{137}Cs (total area 250 km^2). Fifteen percent of the area are roads and other paved areas, 45% are large green areas of which one-fifth are forests, 11% are covered with buildings and 29% are gardens and other small green areas.

Clean-up method	Amount of waste	
	mass (tonnes)	volume (m^3)
Soil removal (gardens)	3,650,000	2,190,000
Grass cutting (gardens)	1,825	4,380
Digging (gardens)	0	0
Ploughing (larger green areas)	0	0
Firehosing roads (solid)	1,520	760
(liquid)	1,900,000	1,900,000
Sweeping roads	1,900	1,900
Planing roads	380,000	228,000
Firehosing walls with NH_4NO_3	3,300,000	3,300,000
Sandblasting walls (solid)	165,000	82,500
(liquid)	330,000	330,000

There was one important type of waste which was not dealt with in the scenario study, namely, sludges from municipal water treatment plants. These may contain so high levels of contamination that they may have to be treated as radioactive waste (Puhakainen, 1987).

26.5 Clean-up wastes from rural areas

Following clean-up in agricultural areas, the only important wastes to be disposed of are vegetation and soil, the latter giving the higher volume of waste (Table 6.4). Most of the radioactivity will be in the soil. If the clean-up work is done immediately or soon after the deposition, the vegetation removed will contain a large fraction of the intercepted activity, but if the vegetation is not removed all the activity will eventually be found in the soil. Since soil efficiently retains deposited radionuclides, it is necessary to remove only a shallow layer of soil (about 5 cm) to remove almost all the contamination. Even so, the amount of soil to be removed may well constitute a severe transport and disposal problem. Table 6.4 shows the volumes of soil resulting from skimming off various thicknesses of soil from cultivated fields.

Table 6.4 *Volumes of removed soil per km^2 and average volumes per km^2 taking into account the proportions of cultivated areas in different parts of Norway (20% increase in volume due to handling).*

	Thickness of soil layer removed		
	3 cm	**5 cm**	**10 cm**
Volume (m^3 km^{-2})	36,000	60,000	120,000
Volume (m^3 km^{-2}) in:			
• North Norway (0.73%)[a]	260	440	870
• Central Norway (3.9%)	1,400	2,300	4,400
• West Norway (3.0%)	1,100	1,800	3,600
• South Norway (1.7%)	610	1,000	2,000
• East Norway (5.8%)	2,100	3,500	7,000

[a] The percentages in parentheses are the proportions of cultivated areas.

If all the fields from the contaminated areas described in the theoretical accident are decontaminated by removing a layer of soil, the amounts of waste will be enormous. As an example, Table 6.5 gives the amounts of soil for an 403 km^2 area (Area 2 from the model accident, wet deposition, see Table 6.1).

Table 6.5 *Total amounts of removed soil from an area of 403 km^2 in different parts of Norway area (Area 2 from the model accident, wet deposition, see Table 6.1), depending on thickness of removed soil layer.*

Region	**Total soil removed (m^3) depending on thickness of soil layer**		
	3 cm	**5 cm**	**10 cm**
North Norway	106,000	176,500	353,000
Central Norway	565,800	943,000	1,888,000
West Norway	435,000	725,400	1,451,000
South Norway	246,000	411,000	822,100
East Norway	841,000	1,402,400	2,805,000

The activity concentrations of the removed soil are fairly high assuming a deposition of 8 MBq m^{-2} (Table 6.6).

Table 6.6 *Cs-137 activity concentrations in removed soil from an area contaminated with ^{137}Cs assuming that 90 % of the deposited radioactivity was removed in skimming.*

Contamination level MBq m^{-2}	**Activity concentration (MBq m^{-3}) depending on thickness of soil layer removed**		
	3 cm	**5 cm**	**10 cm**
0.8	20	12	6
8	200	120	60
>100	>2,500	>1,500	>750

Compared to the amounts of removed soil, the amounts of removed vegetation are much lower, as indicated in Table 6.7. These data refer to a growing season with a mature crops and are therefore maximum values.

Table 6.7 *Amounts of vegetation per hectare in different parts of Norway.*

Region	**Fresh weight (t ha^{-1})**		**Fresh volume (m^3 ha^{-1})**	
	Cereals	**Grass**	**Cereals**	**Grass**
East Norway	35	17	87	43
South Norway	31	19	77	48
West Norway	31	19	79	48
Central Norway	30	19	74	47
North Norway	17	18	43	45
Average	28	18	72	46

If the cleanup work is done immediately or soon after deposition the ^{137}Cs activity concentration of removed vegetation is, however, fairly high (Table 6.8).

Table 6.8 *Average concentrations of ^{137}Cs in freshly harvested vegetation in Norway. Figures are for dry deposition: for wet deposition the values should be multiplied by a factor 0.5.*

Contamination level MBq m^{-2}	**^{137}Cs Activity concentration**			
	MBq t^{-1}		**MBq m^{-3}**	
	Cereals	**Grass**	**Cereals**	**Grass**
0.8	120	180	48	69
8	1,200	1,800	480	690
>100	>15,000	>22,000	>5,960	>8,600

If contaminated domestic animals are slaughtered, 20-500 kg per carcass of waste will be created.

26.6 Clean-up wastes from forests

Clean-up of forest areas by removing contaminated material, such as vegetation, will create enormous amounts of solid radioactive wastes. Table 6.9 shows biomasses in pine, spruce and deciduous forest trees in both Northern and Southern Finland. Corresponding data for undergrowth, litter, humus, stumps and roots are shown in Table 6.10.

Table 6.9 *Biomasses in different types of natural forest trees stands in southern and northern parts of Finland (mass of stems without bark could not be estimated).*

Tree	Area	Type of biomass	Mass (t ha^{-1})	Stacked volume (m^3 ha^{-1})
Pine:	Northern	crown	17	44
		bark	1	13
		stems without bark		70
	Southern	crown	29	70
		bark	3	30
		stems without bark		140
Spruce:	Northern	crown	68	170
		bark	2	20
		stems without bark		110
	Southern	crown	75	190
		bark	3	40
		stems without bark		250
Deciduous:	Northern	crown	18	40
		bark	1	13
		stems without bark		70
	Southern	crown	24	50
		bark	3	30
		stems without bark		170

Table 6.10 *Biomasses of undergrowth, litter, humus, stumps and roots in pine, spruce and deciduous forests (masses of stumps and roots could not be estimated).*

Forest type	Type of biomass	Mass (t ha^{-1})	Stacked volume (m^3 ha^{-1})
Pine	undergrowth	0.7	4
	litter and humus	95	310
	stumps and roots		78
Spruce	undergrowth	0.8	4
	litter and humus	120	390
	stumps and roots		130
Deciduous	undergrowth	0.9	4
	litter and humus	120	400
	stumps and roots		90

Removal of the upper 5 cm or 10 cm layers of soil will create 600 or 1200 m^3 of waste, respectively. As an example, Table 6.11 gives the estimated average volumes and ^{137}Cs activity concentrations of clean-up wastes from an area of 2 km^2 with a ^{137}Cs contamination of 100 MBq m^{-2} (Area 1 in the model accident) in Southern Finland, where 72% of the land area is forest, of which 54% is pine, 35% spruce and 8% deciduous forest. In this example case it is assumed that the stems are barked and all the radioactivity is in the bark and the remaining stems are inactive.

Table 6.11 *Average amounts and ^{137}Cs activity concentrations of clean-up wastes from an area of 2 km^2 (0.77 km^2 pine forest, 0.5 km^2 spruce forest, 0.11 km^2 deciduous forest, 0.64 km^2 unforested) in Southern Finland. Dry fallout of 100 MBq m^{-2} is assumed and that clean up work starts immediately after the fallout is deposited.*

	Activity (TBq)	**Stacked volume (m^3)**	**Activity (GBq m^{-3})**
Contaminated pine forest 0.77 km^2, ^{137}Cs			
Needles	27	1,100	25
Other crown	27	3,600	7.5
Bark	0.4	1,700	0.2
Undergrowth	23	300	77
Total	77	6,700	
Inactive stems without bark	-	8,900	
Contaminated spruce forest 0.5 km^2			
Needles	20	2,800	7
Other crown	20	4,600	4.3
Bark	3.3	1,000	0.2
Undergrowth	10	200	50
Total	50	9,500	
Inactive stems without bark	-	9,800	
Contaminated deciduous forest 0.11 km^2, ^{137}Cs			
Leaves	1.7	170	10
Other crown	1.6	270	5.9
Bark	0.02	290	0.07
Undergrowth	7.7	48	160
Total	11	780	
Inactive stems without bark	-	1,500	

For all volumes together there are 17,000 stacked m^3 of radioactive waste (in addition to uncontaminated stems of 20,000 m^3).

The total activity for all forest types is 140 TBq (0.07-160 GBq m^{-3}).

26.7 Summary of the amounts and activity concentration

Vast amounts of radioactive waste will be generated in the clean-up of large contaminated areas. In all urban, agricultural and forest areas, soil will generally form the most voluminous part of the waste. In forest areas, vegetation will represent the bulk of the waste. Effluents from firehosing walls, streets and other paved areas in urban areas will also be considerable.

Activity concentrations of clean up waste will vary over a wide range. In urban areas, the highest levels will be found in cut grass and solid matter removed from walls and paved surfaces by sweeping and firehosing shortly after deposition. Years after deposition, soil and asphalt will have the highest activity concentrations. In forests and agricultural areas, the activity concentration of vegetation will be high shortly after deposition, but years later the activity will reside almost entirely in the soil and litter layers.

26.8 Transportation of clean-up wastes

The way of transport of radioactive wastes from the contaminated area to the disposal site depends on the distance between the disposal site and the area to be cleaned up. If the disposal site is not within a nearby controlled area, the wastes will have to be transported along public routes, and either international or corresponding national rules and regulations will have to be complied with when packing and transporting the waste. Observance of these regulations is not required if the transportation takes place inside a controlled area (IAEA, 1992).

26.8.1 Safety in waste transportation

General safety factors which need to be observed in the transport of radioactive wastes are (IAEA, 1992):

a) Preventing the spread of waste material during transport, e.g. by covering the loads.
b) Determining contamination levels on the external surfaces of packages and vehicles after loading and unloading, and decontaminating them if necessary.
c) Proper routing and timing of the transport.
d) Monitoring the transport route for contamination due to dropping and spilling.
e) Being prepared for transport accidents.
f) Monitoring and registering the quantity, physical and chemical form and activity of the transported wastes.
g) Protecting clean up workers and the general public against exposure to radiation, e.g. by additional radiation shields of the vehicles.

26.8.2 Transporting wastes within the controlled area

Solid wastes, such as contaminated soil and vegetation, can be removed short distances (100-200 m) directly into a storage mound, trench or natural basin using bulldozers or scrapers. However, a large number of waste disposal sites spread over a wide area is not easy to regulate or control over the longer term. It can only be used as an interim solution in the absence of adequate haulage equipment or an appropriate final disposal site, and for preventing the resuspension of contamination.

Centralised waste disposal includes the loading, transport and unloading of wastes. It is possible to use heavier haulage equipment in closed areas than it is on public roads. Big trucks or dump trucks can be used for transporting soil and other solid wastes, and different types of loaders can be used for loading, depending on their availability and suitability under the specific circumstances. The wastes can be dumped directly into the final disposal facilities from the trucks or dump trucks. Dry and dusty wastes can be sprayed with water to bind the dust. Also within the closed area, the dispersion of contaminated material to uncontaminated or decontaminated zones during transport should be prevented by covering the waste loads and decontaminating the vehicles.

In the Chernobyl area, the removed soil was first collected into mounds in the middle of the decontaminated area, and then loaded with wheeled or tracked loaders onto dump trucks, lorries and other vehicles. The platforms of the vehicles were covered with polyethylene film before and after loading, and the loads were covered with a tarpaulin. The vehicles were decontaminated and monitored at the boundary of the zones with lower activity levels (IAEA, 1992).

26.8.3 Transporting wastes outside the controlled area

Waste arising from the clean-up of a vast area contaminated by a nuclear accident is likely to be classified mostly as low-level waste which, in accordance with international regulations, must be transported in industrial packages. However, large amounts of waste can be transported under special arrangements under which the shortcomings in packaging and transport methods can be off-set by operational controls (IAEA, 1992). The transport of waste is supervised and accompanied by a radiation protection specialist who is also prepared for emergency situations. Emergency equipment must be made available for the escort. Where necessary, the convoy must also include the police or a fire engine.

Radioactive wastes can be transported outside a controlled area by road, rail or water. The choice of transport should be based on safety and cost-effectiveness considerations.

Large amounts of low-level waste have been transported safely over long distances by road and rail in connection with the dismantling of closed-down uranium plants and the clean-up of the surrounding areas. In Canada, almost 70,000 m^3 of soil contaminated with radium and arsenic, together with other solid wastes from the clean-up of areas where uranium mill tailings had been produced, were transported from the city of Port Hope over a distance of 350 km to be buried at the repository of the Chalk River Nuclear Laboratories. Trucks with a capacity of 20 m^3 were used in the transport operations. Before loading, a polyethylene film was stretched over the platform to prevent the loss of contaminated material during transport. After the loading, the vehicles were decontaminated, and the load was covered with a tarpaulin and weighed. The transport took place along public roads. The route was monitored regularly for possible contamination but there were no signs of any waste being lost during transportation. The loads were checked again at the unloading site and samples were taken to determine their activity concentrations. The waste was unloaded by dumping from the top of a ridge straight down into a natural valley. Before the return trip, the vehicles were washed and monitored (IAEA, 1992).

Radioactive wastes can also be transported by rail. The cost of rail transport may exceed that of direct road transport if double or triple loading and unloading is required, as in a truck-train-truck combination. Even for shorter distances, transport by train is less cost-effective than by road. However, transport by rail is often a safer option because public roads, or side roads which are not well-suited for heavy traffic, can then be avoided. It is also easier to shield the operational staff from exposure to radiation during rail transport.

In the United States, 1,880,000 m^3 of radium-contaminated soil and mill tailings were transported by rail from the area around the Vitro Chemical uranium plant, Salt Lake City, over 140 km to the South Clive desert area. At Salt Lake City, the waste was loaded with a backhoe loader onto dump trucks of 32 m^3 and 45 m^3. The waste was dumped via a loading ramp straight into the rail cars. Two rail units of 75 cars were used, one of which was being loaded while the other was being unloaded. At South Clive the waste was unloaded by turning over transportation rail cars. From the heaps the waste was transported to the storage site on dump trucks. The rail cars were washed and monitored after each loading and unloading (Rager, 1986). Loss of the contaminated material during transport was prevented by covering the loads with tarpaulin and the waste in the rail cars was sprayed with a polymer surfactant (IAEA, 1992).

26.9 Final disposal of clean up waste

26.9.1 Interim storage

In the absence of final disposal facilities, or during their planning and construction, it may be necessary to use temporary stores. Interim storage may also be needed forsludges and liquid wastes pending concentration and solidification. Interim storage may also be required for waste rich in organics whose volume may eventually be reduced by the decay and degradation of organic compounds.

In the Chernobyl area, where large amounts of radioactive waste had to be disposed of very quickly and before final disposal facilities were available, the waste was stored in surface mounds near the decontaminated areas. The storage sites were located remote from water systems and catchment areas. The base of the waste mounds were lined to prevent the washoff of liquids. The waste, which, in addition to the contaminated earth, contained large amounts of vegetation and other organic materials, was pushed into mounds with bulldozers. The mounds were first covered with a polyethylene film and later with clean earth. As the zone was closed off, a ditch was dug around it and warning signs were posted (IAEA, 1992).

26.9.2 Final disposal

This section is based on the IAEA documents given in the reference list (IAEA, 1981, 1982, 1984, 1985, 1992).

Selecting the final disposal site and the disposal method

When disposing of radioactive waste, the general principle is that the waste should be disposed of in such a way that there is no unacceptable detriment to humans and the environment. Therefore, the repository and the disposal method must be selected and planned so that the migration of radionuclides is prevented and intrusion by humans, animals or the roots of plants is made difficult. The waste must also be shielded against wind erosion. The disposal measures must be planned and carried out so that the workers and the general public will be protected from radiological hazards at all operational stages and contingency plans laid down to cope with possible accidents.

In a normal situation, selecting a disposal site for radioactive waste and designing the disposal facility and method is a complex and time-consuming process which involves geographical, ecological, climatic, structural, economic, social and safety analytical studies. The main considerations in the choice of a disposal site are the hydrogeological and ecological characteristics of the area, land use and future land needs, as well as social and economic factors. In general, the disposal site should be chosen in an area where natural shielding could be made use of. An ideal site is, however, rarely available. The disposal of wastes can normally be improved by engineered barriers and waste treatment methods, for example, by (i) compacting the waste, (ii) using immobilising materials as backfill and in the structures of the disposal facility, (iii) using flow barriers and drainage, and (iv) shielding the waste against erosion and intrusion. Following the closure of the repository institutional control is required to ensure that the requirements of radiation protection will be met. Low-activity wastes containing mainly ^{90}Sr and ^{137}Cs, usually require disposal which will last for a few hundred years.

Shallow ground disposal

Burial in the ground is a method commonly used for disposal of domestic waste. Burial of radioactive waste has also been widely used for several decades.

Waste burial trenches in the ground are usually shallow excavations with side-walls at an angle of 45° - 90°, depending on the type and properties of the soil. The bottom of the trench is usually above the water table. The surface of a covered trench may be at ground level or it may form a mound. The access of surface waters is prevented by digging ditches around the area.

The structure of a waste trench can be relatively simple in clay soils and under favourable conditions. A simple moisture barrier may be sufficient as a lining. The impermeability of the burial trench structure and the stability of radioactive waste material in the repository can generally be improved with engineering and chemical methods. The trench can be covered with a layer of clay, which will prevent the intrusion of rainwater, surface water, animals and plant roots into the waste. The clay layer can be covered against erosion with top soil and seeded with grass. Safety of the disposal sites can be further improved by increasing the thickness of these barriers or by using additional barrier and drainage layers, such as plastic sheets and gravel. Canalisation of groundwater, rainwater and meltwater is also important in preventing infiltration of water. A layer constructed of rubble, reinforced concrete or similar materials, to prevent the intrusion by humans, animals or plants can be omitted, provided that the burial trench is to be kept under control until the radioactivity has decreased to an acceptable level.

In humid areas, water may accumulate in the trenches if host soils are of low permeability. If the cap has developed cracks, the accumulation of water can lead to an overflow of the trenches on the ground. The accumulation of water at the bottom of the trench can be prevented by subsurface drainage and the moisture situation can be monitored through connecting pipes and collecting wells. The possible migration of radionuclides can also be detected at an early stage.

In the Chernobyl area, the low- and intermediate-level waste was piled into shallow trenches. One trench alone contains more than 10,000 m^3 of waste. The bottom and the walls of the trench were lined with clay, which was 1.0 m thick at the bottom. The waste was covered and levelled with a 0.6 m layer of local soil, on top of which was spread a 0.5 m clay layer. A 1.0 m layer of local soil was applied on top as an erosion barrier. Water was directed around the trench, and each trench was equipped with a sampling well. The disposal site was fenced off and illuminated (IAEA, 1992).

Surface mounds

Disposal of waste in surface mounds is as common as shallow ground disposal. Both methods are used, for example, at the Centre de la Manche disposal site in France (IAEA, 1985). Disposal of waste in surface mounds may however be a safer alternative in an area which does not adapt well to waste burial (for instance, where the water table is high).

In principle, the structure of a surface mound is similar to that of a waste burial trench, featuring buffer and intrusion barriers and water canalisation layers. Active wastes disposed of in surface mounds often require a fairly heavy shielding against weathering and intrusion. Sand, gravel, cobbles or similar materials can be used for the intrusion and moisture barrier. Surface waters can be directed into a collecting ditch around the mound by placing the cap in the right angle of gradient. Also the bottom should be bevelled towards the edges so that no moisture can gather under the waste mound. There are several alternatives for lining (clay, gravel asphalt etc.), depending on the soil type, its moisture and the climatic conditions.

In humid climates, where melt waters and frost heaving must be considered, the structure of a durable and safe mound may be rather massive. For example, the total thickness of the cap of a mound containing 13,000 m^3 of low-active waste at West Valley, New York, is nearly 5 m (Blickwedehl, 1986).

Mounds similar to those used for disposal of mill tailing wastes of uranium may also be suitable for waste from clean-up following a nuclear accident. As nuclear waste contains mainly ^{90}Sr and ^{137}Cs, the capping of the mound may be lighter than when disposing of uranium mill tailings where the predominant radionuclide is the long-lived ^{226}Ra (with a physical half life of 1600 y).

In the United States, the mill tailings of uranium and other clean up wastes (a total of 1,880,000 m^3), were disposed of in a big mound at South Clive, Salt Lake City, Utah (Rager, 1986; US DOE, 1984). The mound measures 671 m x 366 m x 7 m in height. The topsoil was first removed from the site to a depth of 2 metres and later used to cover the mound. No bottom lining was used. Instead, the waste was compacted against the ground. The composition of the underlying soil ranges from fine- and medium-grained sands to silty clays, and the water table is 7.6-10.7 m below the ground surface. The South Clive area is an arid desert. The 2.1 m thick layer which prevents the release of radon was constructed of clay, and the 0.6 m thick erosion and intrusion barrier of blasted rock.

Natural valleys and basins

Valleys with a suitable geological structure and natural basins have been used for disposing of waste, especially uranium mill tailings. Basins and valleys can also be used in emergencies where large amounts of solid low-level clean up waste must be disposed of, provided that there is a suitable valley in the contaminated area or in its vicinity.

The confinement basin can be formed by constructing an embankment across a valley at the downstream end. Where necessary, the base and the walls of the basin will be levelled, compacted and lined with low-permeability material, such as clay or clayey soil. The infiltration of surface waters into waste is prevented by drainage. The filled basin can be covered with normal capping materials. The covered area is likely to be vast and therefore exposed to erosion. The average depth of wastes in the basin is low, as the waste layer becomes shallower towards the edges. Since the valley is closed-off by natural barriers on three sides, a relatively short dam embankment is usually required, which cuts down the construction costs. Extra costs may arise from the uneven and irregular bottom and walls of the valley which may hamper the installation of any liner and the compaction of the waste into the basin.

Almost 119,000 tons of low-level waste soil, mostly uranium mill tailings, containing radium and arsenic, were buried into a valley in the Chalk River area, Canada. Apart from the contaminated soil, from Port Hope, concrete flagstones, concrete blocks, logs and tree roots were also buried. In the east, the valley bordered on the bedrock, and in the west on a ridge of dune sand. Sand was spread against the bedrock to isolate the waste from the rock. Contaminated soil was spread in the valley and compacted with a bulldozer in layers of 0.7 m. The waste layer was almost 12 m at the thickest, and the area covered by waste was about 1.5 ha. The area was first covered with native clayey silt to a depth of 0.3 m. On top of that layer was spread a 0.7 m sandy layer to prevent leaching of the clay layer and the intrusion of roots. A sandy topsoil layer of 0.15 m was applied as the upper layer. A ditch was dug on the side of the bedrock to direct the surface waters flowing down the rock to run around the waste basin (Killey, 1985).

Old mines and pits

Large amounts of low-level solid clean-up waste may also be placed in a closed-down underground mine or open pit, if they are situated nearby. The climate, the water table and its fluctuations, flowing surface waters and wall permeability are some of the factors to be considered when assessing the suitability of an old mine or pit for disposing of radioactive wastes.

Shafts and open pits are usually located deep below the normal groundwater level. Water and moisture often permeate into the caverns through cracks and breaks in the bedrock and through fractures in the walls caused by mining. Also, the migration of radionuclides can occur through these cracks. Rough walls may hamper the installation of liners and barriers and special techniques may be necessary for installing and compacting the waste. Old mines and pits usually require careful and time-consuming geological and hydrogeological studies before they can be used for disposal of radioactive waste.

In Germany, low- and intermediate-level wastes are intended to be disposed in the old iron ore mine at Konrad. The galleries are very dry, and they are situated in clayey soil at the depth of 1000-1200 m. It has been estimated that 500,000 m^3 of wastes can be disposed of in the mine.

Bedrock caverns
Disposing of low- and intermediate-level waste into galleries and caverns in the bedrock has been extensively studied and plans have been drawn up in many countries. Final disposal sites in the bedrock are usually designed for solid or solidified packed operational and decommissioning wastes from nuclear power plants.

In the Nordic Countries there have been two bedrock repositories which have been operational for a few years: at Forsmark in Sweden and at Olkiluoto in Finland. The Forsmark repository is placed 60 m below the sea bed and the top of the Olkiluoto repository, 60 m below ground level.

Wastes from accident situations such as washing fluids, sludges, or ashes from incineration, can be solidified and disposed of in the bedrock, provided that such a facility is available. Disposal of other types of waste, such as contaminated soil, into the bedrock will be a much more expensive alternative than disposal in surface mounds or in shallow pits.

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PART 7

RADIATION PROTECTION AND SAFETY OF WORKERS

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28. Radiation protection and safety of clea-up operators

This section briefly reviews the principles applicable to radiation protection and safety of workers, and methods that could be used to minimise occupational exposure in reclamation work.

In considering the clean up of areas shortly after an accident, a decision would have to be made whether to implement clean-up actions early and thus cause higher occupational doses, or wait until short-lived isotopes have decayed and/or weathering has reduced the radiation levels. For example, the decision may be to stabilise the contamination using sprays to prevent re-suspension followed by a delay before actual clean-up starts. The timing of such actions would depend on many factors, including weather conditions, the area involved, equipment available and the competence of the work force.

Means of reducing occupational exposure while carrying out the tasks should, as far as possible, be clearly defined in 'work procedures'. In general, reductions in occupational exposure during operational tasks can be accomplished by the use of shielding and limiting the time that workers spend exposed to radiation.

Protective clothing and respirators also enable radiation dose to be reduced.

28.1 Radiation protection and safety plan

As part of the emergency response plan for clean-up operations of radioactive contaminated areas, a preliminary radiation protection and safety plan should be formulated with the assistance of radiological experts. In the event of an emergency this plan would be adapted to meet specific requirements.

The radiation protection and safety plan should include a comprehensive radiation monitoring and data management programme that provides for the measurement, evaluation and recording of all exposures incurred by individuals through different pathways. The plan should also cover all practical aspects related to the implementation of this programme, including a list of equipment, facilities and personnel needed and it should state where and how these resources can be obtained.

The plan should also include classification of personnel, the duties, responsibilities and training programmes for various groups in all aspects of the clean-up operations (e.g. handling, transport and disposal), as well as the use of protective clothing, respirators and other means of reducing occupational exposure.

The radiation protection and safety plan would, in addition, have to consider site-specific factors.

28.2 Occupational dose limits

A major consideration in planning for any clean-up operation is the upper limit for occupational dose. The limits should be set by national authorities on the basis of international recommendations and take into consideration the accident situation.

Traditionally, the Nordic countries, Denmark, Finland, Iceland, Norway and Sweden, use the recommendations of the International Commission on Radiological Protection (ICRP) as a basis for national rules and regulations. The latest recommendations from ICRP adopted in 1990 (ICRP Publication 60) include the following recommendation (paragraph 225 in ICRP 60):

"In addition to the exposures resulting directly from the accident, there will be exposures of emergency teams during emergency and remedial action. Even in serious accidents, these can be limited by operational controls. The doses incurred are likely to be higher than in normal situations and should be treated separately from any normal doses. Emergencies involving significant exposures of emergency teams are rare, so some relaxation of the controls for normal situations can be permitted in serious accidents without lowering the long-term level of protection. This relaxation should not permit the exposures in the control of the accident and in the immediate and urgent remedial work to give effective doses of more than about 0.5 Sv except for life-saving actions, which can rarely be limited by dosimetric assessments. The equivalent dose to skin should not be allowed to exceed about 5 Sv, again except for life saving. **Once the emergency is under control, remedial work should be treated as part of the occupational exposure incurred in a practice.**" (Current authors emphasis)

This means that a clean-up operation will, in most cases, be considered as ordinary radiological work under fully controllable conditions and only in exceptional cases can annual dose limits be exceeded.

It is possible that decontamination operations might be dependent upon a small number of key individuals such as health physics personnel, and operations could be placed under severe pressure if such personnel are exposed to high doses early in a clean-up programme, which excludes them from future activities in a contaminated environment. As a consequence, management of dose exposures and manpower resources would be of crucial importance.

It is also important to consider doses to farmers working on their own contaminated land. In this case they would be occupationally exposed and should be treated as normal radiation workers and be subject to the same dose limits. In this respect it will not be any different if farmers are doing work on their own initiative or through recommendations from the authorities or if they are specially ordered and/or specially paid by the authorities to do the work.

Thus the authorities will normally not recommend anyone to receive doses in excess of the limits for this kind of work. It is, however, not a violation of the law or any other regulation that might lead to a penalty if the farmer himself decides to continue his work despite the fact that his dose has already reached the dose limit. The authorities can in this case only inform him of the risks and advise him to follow the recommendations. If possible, the authorities should also give him support with surveillance of his accumulated dose. On the other hand, it is not acceptable to the authorities if the farmer allows employees who work for him to exceed the dose limits.

The same principles apply to house-owners, landowners and contractors. There is no penalty if the recommended dose limits are exceeded by voluntarily actions undertaken by individuals that are not working under employment conditions (working for someone else).

28.3 The principles of justification, optimisation and dose limits

The international recommendations on radiation protection outline a formal system of dose limitations based on three ground principles, which in this field can be expressed as justification of an intervention, optimisation of protection and compliance with specified dose limits (ICRP Publication 60). This system is intended to apply whenever the source of radiation is under control. After a major nuclear accident that results in widespread contamination, a temporary loss of control may occur, during which time it may not be possible to comply fully with the specified dose limits. However, by the time the clean-up programme starts, the public would have been evacuated from highly contaminated areas and the authorities would have re-established control. Thus, the justification and optimisation principles would apply. The radiation doses accumulated by workers during a clean-up operation of an area need to be balanced against those that might result (to the general public) if clean-up is not undertaken. If the clean-up operation does not result in a net saving of radiation dose then the operation is not justified from a strictly radiation protection point of view. The optimisation process does, however, involve a further balancing of all kinds of detriment and benefits in the two cases, e.g. economic and social costs and not only a balance of radiation doses.

The optimisation process in radiation protection was suggested by ICRP (1977) and is perhaps the most important concept in radiation protection. Keeping doses as low as reasonably achievable (the ALARA principle) is considered to be an important tool in optimisation and the dose limits should be regarded as the upper limit in this process. Thus, for activities involving ionising radiation, where these activities are justified, it is not sufficient to meet the requirements of dose limits, there should also be efforts made to reduce individual doses further below those limits.

These considerations and the continually changing situation will place a particular burden of responsibility on health physics staff and decision-makers. They should ensure that adequate effort is put into planning and implementation of each stage of the clean-up to prevent unnecessary exposure of workers and the general public.

28.4 Training programmes for workers and decision-makers

An important way of reducing occupational exposure is to minimise the time which workers spend in radiation fields. Any technique that allows the clean-up workers to perform duties more efficiently, or in a shorter time, will reduce exposure. It would therefore be sensible to train operators in the necessary situation-specific decontamination procedures in a training-area with no contamination to give them the required skills before entering the clean-up area. Such training will, on the other hand, only be efficient if started at the time after a major nuclear accident, when it has been established what kind of problems the situation gives at hand. It will thus most likely not be very useful to perform practical field training in a general manner before an accident has happened, since the clean-up procedure to be used must always be incident specific. It is e.g. not much use to teach a large number of clean-up workers to do a special technique really quickly before you know which technique, of many available, that is going to be applied in the real situation after an accident. Theoretical training in choosing the right incident specific techniques etc. will on the other hand be useful to perform before an accident for planning personnel and decision-makers.

Special personnel with responsibility for radiation protection of the workers should supervise all clean-up actions. These supervisors should also have a good knowledge of the special methods to be used and how to modify and use them in the most efficient way.

After a major nuclear accident, large areas may have to be cleaned with a large number of workers involved. In this case the training of key workers to perform radiation protection actions themselves may be desirable.

28.5 Applied radiation protection and safety of clean-up operators

This section addresses general requirements on radiation protection and equipment for reclamation work.

28.5.1 General radiation protection principles in reclamation work

The work should be carried out in such a manner that the radioactive material is not spread more widely. Thus special clothing should be used which will not be used outside the contaminated area. In addition, personal precautions such as not smoking or consuming of food during the work would be advisable to avoid intake of radioactive material. Finally showering and changing to clean clothes after reclamation work would be advisable.

28.5.2 Equipment for radiation protection

The aim of using equipment for radiation protection is to accomplish three main goals:

1. To avoid inhalation of radioactive particles. This may be achieved by the use of respirators/breathing protection for workers out in the field and clean-breathing-air-devices (filtered air) for vehicle drivers. It may also be advisable to provide the workers with iodine tablets to further limit the uptake of radioiodine by the thyroid gland.

2. To avoid direct contact between radioactive material and skin. This may be achieved by using protective clothing.

3. To shield the working place (vehicle etc.) from the radiation source. This may be done by the use of shielding by lead matting etc.

28.5.3 Information and training on radiation protection topics

The following basic knowledge is necessary for all personnel involved in reclamation work:

1. Basic information on the nature of ionising radiation and its hazards, and
2. Basic information on how to protect oneself to minimise radiation dose

In addition key personnel and supervisors must also have:

3. Basic knowledge on the radiation protection organisation, the decision-making process, procedures for dose measurements, the reporting and recording of results, and
4. Basic ability to improvise and from time to time improve the working conditions to minimise the radiation dose

It is suggested that a minimum duration of one day is allocated for the provision of information and training for key workers and supervisors which includes a basic test of knowledge. The responsible authorities must decide the details in the educational programme and the minimum level of competence. For other categories of workers it is suggested that items 1 and 2 in the list above be covered during an information meeting arranged in co-operation with the radiation protection authority.

It is further suggested that the training programme includes the opportunity for workers to acquire the necessary skill in handling the equipment in a time-effective manner as discussed in section 26.4.

28.6 Estimation of radiation doses in reclamation work

The absolute level of radiation dose received in a reclamation work is dependent on the type of work and the level of contamination and radionuclides involved. It is therefore not possible to give any generally applicable table on doses. The following *examples* are intended as a guide in dose estimation for different reclamation work. All examples are based on fallout of 1 MBq m^{-2} of ^{137}Cs.

28.6.1 Examples of radiation doses in reclamation of urban and suburban areas

The relatively highest external exposure rates in urban and suburban areas are found in gardens. Deposition in dry conditions generally gives higher exposure rates compared to wet deposition. However, the difference is not large enough to require a separation in dose calculations taking all other uncertainties into account.

In urban areas with a deposition of 1 MBq m^{-2} the estimated mean external exposure rate in gardens is 4 $\mu Gy\ d^{-1}$ which is equal to 0.17 $\mu Gy\ h^{-1}$. This is a relatively low exposure rate and limitations in the exposure time for reclamation work would not be necessary.

The time required to do reclamation work in an urban area could roughly be estimated by the following rule of thumb: The person-power-days in reclamation = the number of persons living in the house to be decontaminated. The estimated doses from reclamation of a multi-storey building with 30 flats (for about 75 people) would, in this case, be roughly 300 μGy (J. Roed, pers. comm.)

Gardens in suburban areas are generally larger than in urban areas and this gives higher exposure rates. A mean value for dose calculations would be 20 μGy d^{-1} which is equal to 0.83 μGy h^{-1}. Limitations in the exposure time to avoid exceeding the equivalent dose limit of 0.5 Sv, however, would not be necessary.

A rule of thumb to estimate the time required doing reclamation work in a suburban area is: 5 person-power-days in reclamation work for each house. The estimated doses from reclamation of one house would, in this case, not be more than 100 μGy.

28.6.2 Examples of radiation doses in reclamation of agriculture areas

Case Study 1

The following ***first example*** assumes a case where relatively high exposure dose rates could be expected from agriculture reclamation.

Suppose we want to cut a contaminated crop on a field during the same year in which the fallout was deposited. The crop will contain a substantial fraction of the total contamination of the land at an early stage after deposition. A standard value for the intercepted fraction often used in calculations is 20%, but in some cases the fraction can be up to 50% for wet deposition and up to almost 100% for dry deposition, depending on the type of deposition, the type of crop and at what growing-phase the deposition occurred. It could therefore be considered that some contamination could be removed from the land by harvesting the crop and transporting it to a waste deposit area.

In this example the deposition is assumed to be 1 MBq ^{137}Cs m^{-2}. The external exposure dose rate one metre above the ground will then be about 2×10^{-6} Gy h^{-1}, depending on the shielding in the surface structure.

In a standard procedure the vegetation is first cut and left on the field. This is done by a single person and will take about 1 h ha^{-1}. The workers position on a tractor is one metre or more above the ground and with a shielding factor (transmission factor) of 0.4 or better (Eriksson, 1983) due to the tractor, this will roughly give an external exposure dose rate of $0.4 \times 2 \cdot 10^{-6}$ Gy h^{-1} which is equivalent to 0.8 μGy h^{-1} or 0.8 μGy ha^{-1}.

The next step in the operation will be to pick up the harvested vegetation from the field and transport it to the waste deposit. This will be carried out by a team of three workers; one at a tractor with a loader, lifting the vegetation into a hay-cart, one worker at a tractor driving the wagons from the field to the waste deposit area about 1 km from the field and one worker changing and emptying the wagons at the waste deposit area. The third worker will mostly work with a front loader. The working speed for this team is estimated to 1 ha h^{-1}. The worker on the loader in the field would receive approximately the same dose as the worker doing the cutting i.e. 0.8 μGy h^{-1} or 0.8 μGy ha^{-1}. About the same dose can be expected for the worker at the waste deposit area. In order to calculate the dose to the worker transporting the hay-carts, we have to make some additional assumptions. These are that the dry weight in the harvest is 25% and that the wagon will take 30 m^3 of load with a weight of approximately

6000 kg. Thus, with a specific weight of 200 kg m^{-3}, the dry weight will be 50 kg m^{-3} and 150 kg m^{-3} will be water.

Vegetation harvested during the growing season is expected to contain approximately 0.5 to 1.2 MBq kg^{-1} dry weight with a deposition of 1 MBq m^{-2} ^{137}Cs during rainfall (Eriksson, 1994). We have used the higher value here to calculate the total activity in the hay-cart, which gives 1.2 MBq kg^{-1} x 50 kg m^{-3} x 30 m^3 = 1.8 GBq ^{137} Cs.

The physical dimension of the load of the hay-cart is approximately 3 m high, 2 m broad and 5 m long. The distance from nearest wall to the driver is assumed to be 4 m.

A calculation with the program SuperShield (developed by Shonka Research Associates INC., 1992) gives an exposure dose rate at the drivers position of about 0.28 mR h^{-1} = 2.4 μGy h^{-1} with the assumption that the wall of the hay-cart does not give any shielding. Assuming the hay-cart is empty about 50% of the time gives the driver a mean value dose rate of
1.2 μGy h^{-1}.

A sensitivity analysis of the parameters in the calculation shows that it is not very important to consider the composition e.g. the fraction of water in the load as long as the correct specific weight in kg m^{-3} is used. The type of crop has thus little influence on the result. The amount of radioactivity in the load is, as expected, the most critical parameter. Therefore the parameter Bq kg^{-1} gives the major part of the uncertainty in this calculation.

The calculation for this example has shown that the dose from the transportation of a hay-cart between a field and a waste deposit area can be expected to be of the same order of magnitude as that received from ground radiation. With extra shielding between the wagon and the driver this dose can be further reduced. A shield equal to 1 cm of lead will give an exposure dose reduction factor of about 0.6. This means that the dose with shielding is going to be 40% lower than without shielding.

Conclusions of first example case study:

For ***wet deposition***, a team of four workers is assumed to be involved in moving the crops from a field to a waste deposit area. To be more precise, at least three persons need to work together as a team. The first step in the procedure, cutting the crops, might be made separately a few days ahead by some person in a team of three. In the calculations below the dose received during the cutting procedure is (for simplicity of expression) regarded as person number four. The individual dose will be slightly higher (ca 10%) if three persons instead of four carry out the work. The collective dose for the work will not change.

Each person (in a team of four) will receive about the same radiation exposure from the ground during the work. The worker transporting the hay-carts between the field and the waste deposit will receive an additional exposure from this harvest load that is of the same order of magnitude as that from the ground radiation. The team will together receive about (4 x 0.8 + 1.2) μGy h^{-1} = 4.4 μGy h^{-1}. After a full working day of eight hours, each worker will have received about 9 μGy, assuming that they interchange their tasks during the day. The collective dose for the team will be about 35 μGy per eight-hour working day.

For ***dry conditions,*** where a possibly much higher fraction of the fallout could be contained in the crop, the exposure from the hay-cart will be higher. If we assume that the fraction of deposited activity in the crop is 100%, then the exposure dose rate from the full hay-cart to the worker driving to the waste deposit area will be about twice the value calculated in the above example. The exposure to the other workers in the example will be about the same as before. Thus, the team will in this case receive about (4 x 0.8 + 2.4) μGy h^{-1} = 5.6 μGy h^{-1}. After a full working day of eight hours, each worker will have received 11 μGy, assuming that they interchange their tasks during the day.

If only three persons do the work - the cutting could be done some days ahead of transportation and thereafter using only three workers at the same time for the rest of the work - then the individual doses to each worker will be about 15 μGy per full working-day. The collective dose for the team will be about 45 μGy per eight-hour working day. With a dose rate of 15 μGy per full working day it would take more than 33 000 working days before an effective dose limit of 0.5 Sv per person would be reached.

Case study 2

For comparison, a ***second example*** is considered here assuming that a medium size hay-cart with a capacity of 18 m^3 is used instead of the large wagon of 30 m^3. The working team is in this case assumed to consist of at least two persons working simultaneously; one person at a tractor with a loader, lifting the harvest into the hay-cart and the other person driving a tractor with the hay-cart between the field and the waste deposit area where the wagon is emptied. The working capacity for this team is about 2.5 h ha^{-1}.

With other assumptions being the same as in the first example above with wet deposition, the activity in the wagon will be (1.8 x 18 /30) GBq = 1.08 GBq of ^{137}Cs.

The calculated exposure from the load on the wagon to the driver of the tractor will then be 0.22 mR h^{-1} = 1.9 μGy h^{-1} for wet deposition and (as before) about double at 3.8 μGy h^{-1} in the case of dry deposition. Assuming the hay-cart is empty about 50% of the time it will give the driver a mean value dose rate of 1 μGy h^{-1} with wet and 2 μGy h^{-1} with dry deposition.

Conclusions of second example case study

The radiation doses to the team with ***wet deposition*** will be (2 x 0.8 + 1) μGy h^{-1} = 2.6 μGy h^{-1} or 6.5 μGy ha^{-1}. After a full working day of eight hours, each worker will have received 10 μGy, assuming that they interchange their tasks during the day. The collective dose for the team will be about 20 μGy per eight-hour working day.

In the case of ***dry deposition*** double the radioactivity occurs in the wagonload, and the radiation doses to the team at dry deposition will be (2 x 0.8 + 2) μGy h^{-1} = 3.6 μGy h^{-1} or 9 μGy ha^{-1}. After a whole working day of eight hours, each worker will have received about 15 μGy, assuming that they interchange their tasks during the day. Hypothetically it would in this case take over 35 000 working days before an effective dose limit per person of 0.5 Sv would be reached. The collective dose for the team will be about 30 μGy per eight-hour working day.

Summary and comparison of the two examples from agriculture reclamation work
From the examples given above, the mission to move a harvested crop from a field to a waste deposit area have been compared in Table 7.1, which also gives a summary of the key parameters.

Table 7.1 *Summary of example cases where two teams of workers remove contaminated crops from a field deposited with 1 MBq ^{137}Cs m^{-2}, to a waste deposit area.*

Parameter	*Team of three workers*		*Team of two workers*	
Radiation from ground during work on a tractor (shielding 0.4)	0.8 μGy h^{-1} (to each worker)		0.8 μGy h^{-1} (to each worker)	
Specific weight of crop	200 kg m^{-3}		200 kg m^{-3}	
Hay-cart load	30 m^3, 6000 kg		18 m^3, 3600 kg	
Time per hectare	1 h		2.5 h	
	Wet deposition	*Dry deposition*	*Wet deposition*	*Dry deposition*
Activity in the load on the hay-cart	1.8 GBq	3.6 GBq	1.1 GBq	2.2 GBq
Exposure dose rate from load on hay-cart to driver	2.5 μGy h^{-1}	5.0 μGy h^{-1}	1.9 μGy h^{-1}	3.8 μGy h^{-1}
External collective dose for 8 h work	30 μGy	40 μGy	20 μGy	30 μGy
External dose per worker for 8 h work	10 μGy	13 μGy	10 μGy	15 μGy
External collective dose per hectare	3.6 μGy	5.0 μGy	6.5 μGy	9.0 μGy
External dose per worker per hectare	1.2 μGy	1.7 μGy	3.3 μGy	4.5 μGy

Note: A three-person-team is compared to a two-person-team. There was actually one more person involved in each of the two examples from the agricultural area above. Or, to be more correct, one more task was handled, namely the first step of cutting the grass prior to moving the crop from the field to the waste deposit area. The person doing this would, however, be doing the same job in the same period of time in both examples. Thus the doses received will be the same (0.8 μGy h^{-1} or 6.4 μGy d^{-1}) in both cases and this task will not influence the comparison. It was therefore excluded from the comparison table.

29. Conclusions on radiation protection and safety of clean-up operators

The external radiation exposure arising from reclamation work does not seem to lead to unmanageably high doses. With fallout of 1 MBq m^{-2} of ^{137}Cs the doses can be expected to be no more than 20 μGy d^{-1} as long as the reclamation work does not include handling of fallout in a more concentrated form.

The relatively highest exposure rates in urban and suburban areas are found in gardens and deposition occurring in dry conditions generally gives as a rule higher exposure rates compared to wet deposition. The difference is, however, in this case not large enough to require a separation in dose calculations taking all other uncertainties into account. Exposure rates are about 5 μGy d^{-1}.

The gardens in suburban areas are generally larger than in urban areas and this gives higher exposure rates. Reclamation work close to large trees and in the woods may also lead to higher doses, in some cases in excess of 20 μGy d^{-1}. However, limitations to the exposure time to avoid exceeding the equivalent dose limit of 0.5 Sv, will generally not be necessary with this level of fallout.

The time required to do reclamation work in an urban area could roughly be estimated by the following rule of thumb: The person-power-days in reclamation = the number of persons living in the house to be decontaminated. The dose from reclamation of a medium-sized multi-storey building has been roughly estimated to be 300 μGy. A rule of thumb to estimate the time required doing reclamation work in a suburban area is: 5 person-power-days in reclamation work for each house. The doses from reclamation of one house would in this case not be more than 100 μGy.

The doses from an agriculture reclamation work on a 100 hectare field that will take about 12 working-days has been estimated to be roughly 500 μGy.

Precautions to avoid inhalation and intake of radioactive material should always be considered to avoid doses from internal radiation. This should not be too difficult to achieve by using respiratory protection and keeping precautions such as not smoking or consuming food during the work in order to avoid intake of radioactive material, and finally showering and changing to clean clothes after reclamation work.

From a radiation protection point of view it can also be concluded that it is better to use a larger team in reclamation work if their work is more efficient per person-hour compared to a smaller team. If the working conditions are comparatively the same for a large team as for a small team then the shorter time it takes for the large team to complete the work leads not only to smaller individual doses but also to a smaller collective dose.

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PART 8

RESOURCES AVAILABLE IN SOCIETY

Sven Eric Berg

Swedish Rescue Services Board

31. Resources available in society

A decontamination operation will only be successful if cost- efficient methods are used. The cost-effectiveness depends, among many other factors, including the qualifications and training of the personnel and the capability of the equipment. The personnel must be able to handle the equipment in a professional way and should also know how to protect themselves. To fulfil these requirements they need courses in radiation protection. The equipment must be suitable for the selected countermeasure.

Societies planning and preparedness for reclamation should meet realistic demands for early actions and outline a cost-effective strategy that implies reasonable use of personnel and equipment resources.

Since the cleanup after fallout of radioactive materials will be accident specific it is difficult to forecast applicable countermeasures and their resource requirements. Therefore, it is difficult to forecast the size of the organisation and the need for equipment. On the other hand, there is a high possibility that ordinary resources can be used which are readily available from various organisations. Together, this would mean that planning of special resources and particularly planning of specific equipment, should not be necessary.

Planning for early cleanup actions is different from that of long term planning with respect to the available time and quantity and quality of available information on which to base decisions. Some fairly detailed short term planning before an accident happens is probably necessary, to be able to apply early countermeasures during the first few weeks after the deposition of fallout. A high level of contamination may require early decontamination of important communication links, industry, administrative buildings etc.

Detailed plans for long term remediation, before an actual contamination event, is complex. Strategies for long term remediation should be outlined in comprehensive planning. The most practical and helpful long term planning will be completed after fallout has been deposited, when the real needs for methods, staff and equipment are more fully known. But society must be prepared for accelerated and effective planning, over a short period, to cope with a possible release of radioactive materials.

To make sure that Nordic societies are adequately prepared to respond to a nuclear accident it is important that authorities with responsibilities for remediation ensure that adequate research is carried out and that there are sufficient personnel resources available. There should be reasonably experienced staff available, at all necessary levels in society, who are familiar with the planning process and emergency preparedness procedures.

Available resources vary, of course, between the Nordic countries, but in all countries there are organisations with both knowledgeable staff and suitable equipment accessible for decontamination operations. In some organisations there is a high level of competence, with highly qualified staff, whilst in others they have limited knowledge. Nevertheless, in all countries all relevant personnel would need some additional preparation, adapted to their present experience, before they can participate in decontamination operations.

31.1 Personnel resources

Examples of organisations in the Nordic countries with usable personnel are:

1. Emergency Preparedness Organisations
2. Rescue Services (fire brigades)
3. Private Companies (Contractors).
4. Nuclear Power Stations and other Nuclear facilities
5. Rescue Services Training Academies
6. Radiation Protection Authorities
7. Universities, high schools, institutes
8. Hospitals
9. Civil Defence Training Academies
10. Military Training Academies

Some examples are Statens strålevern in Norway, STUK in Finland, the National Emergency Organisation in Sweden, Risø Førsøgsanlæg in Denmark and Geislavarnir Rikisins in Iceland.

The public could also be an important resource if they are willing to cleanup their own houses and gardens, under appropriate direction.

31.2 Equipment resources

Examples of organisations in the Nordic countries with useful equipment include:

1. Private Companies
2. Rescue Services
3. Road Authorities
4. Rescue Services Training Academies
5. Civil Defence Training Academies
6. Military Training Academies
7. Agricultural Machine Pools

The resources of private companies with equipment for construction and cleaning are probably the best source of suitable equipment in all Nordic countries.
Agricultural machinery and equipment in private households could also be an important resource if farmers and the public are willing to cleanup their own property.

32. Organisation for decontamination operations

Society should only respond if it is considered necessary with regard to the importance and value of the event and the costs for the required operation and other circumstances that makes it necessary for the society to respond. Applicable authorities with responsibility for societies remediation measures vary among the Nordic countries.

Norway
In Norway, the Ministries are responsible for countermeasures after a nuclear accident within their specific area of responsibility. In the acute phase of an accident, however, overall authority has been delegated to the Crisis Committee for Nuclear Accidents. The Crisis Committee is a sub-group from the Advisory Committee for Nuclear Accidents. It consists of several authorities having special responsibilities in a nuclear accident. Both the Ministries and the Crisis Committee are supposed to make their decisions based on advice from the Advisory Committee. The Crisis Committee decides themselves for how long their actions are needed, but if time allows, all decisions and actions shall be clarified by the Ministries.

The following representatives from the Advisory Committee and their respective Ministries have tasks and responsibilities related to decontamination:

- Norwegian Radiation Protection Authority
- Agricultural University of Norway
- Directorate for Nature Management
- Directorate of Civil Defence and Emergency Planning
- Institute of Energy Technology
- Norway Military Headquarter
- State Pollution Control Authority
- Ministry of Health and Social Affairs
- Ministry of Environment
- Ministry of Agriculture

Finland
In Finland, the Ministry of the Interior is responsible for societies countermeasures. In the acute, initial phase the municipal fire department would carry out the first decontamination operations. After the acute phase responsibility would be transferred to environmental, agricultural, social- and welfare authorities.

Various national authorities such as STUK, defence forces, Meteorological Institute, Seismological Institute and Social- and Welfare Ministry and Agriculture and Forest Ministry are advisory boards for the Ministry of the Interior. These authorities and other most concerned national bodies would convene a co-operative group under the lead of the Ministry of the Interior to support a designated Rescue leader. The Ministry of the Interior is responsible for co-operation and supervision of emergency planning. Please check my alterations.

Sweden
In Sweden, County Administration Board is responsible for societies countermeasures. Various national authorities such as the Radiation Protection Institute, the Swedish Board of Agriculture, the National Food Administration, the National Board of Health and Welfare, the Environmental Protection Agency and the Nuclear Safety Inspectorate are advisory boards for the County Board.

A National Expert Group on decontamination (NESA), with representatives from the most concerned national bodies, was established at the end of 1993. The groups main duties are to support planning and to co-ordinate advice to the County Board. The group shall provide an operational and economic frame of reference for decisions concerning clean-up operations.

The Rescue Services Agency is responsible for co-ordination and supervision of planning at the regional level.

Denmark
In Denmark, Beredsskapsstyrelsen (Emergency Management Agency) is responsible for remediation, taking advice from national bodies and support from Forsøgsanlæg Risø (Risø National Laboratory).

Iceland
In Iceland, the responsibility for radiation emergency response is not clearly defined in the appropriate legislation. The Civil Defence Authority co-ordinates and supervises reclamation of land with advice and support from most concerned national and local authorities.

There is no need for a responsible authority to set up its own organisation with its own staff for direct management at the field level. It would be more effective to purchase such requirements from suitable contractors with experience of both field management and co-ordination of similar scenarios.

By transferring decontamination operations to contractors, administration for the authority will be confined to analysis, decision making and selection of methods, collaboration with other bodies and following up actions, whilst the contractor would be responsible for carrying out and co-ordinating operations. The principal responsibility for operations is of course still on society and the responsible authority.

In order to assure the effectiveness of the operations and the countermeasures it is important that responsibility assigned to individuals is defined before the decontamination operations start. Individuals needs to know not only their own responsibilities but also those of their colleagues and of their management unit, and how these responsibilities complement those of other groups. Persons with responsibilities for operation planning, administration including labour welfare, following up etc. should be clearly appointed. The management structure and reporting lines should be well known among the staff.

32.1 Education

To develop, maintain and improve the societies preparedness for remediation, decision makers, experts, chief staff and other key-personnel need specific information, education and periodic retraining. The key-staff should have some experience and competence for their duties. They should be familiar at least with planning for short term clean up and planning for long term reclamation, radiation protection, organisation matters, and how to purchase decontamination operations.

All staff in a newly established operation organisation, need some extra information or retraining just before the work commences. Firstly all operatives need general information on radiation protection principles, how to reduce exposure and how to use equipment for radiation protection (see Chapter 26).

Secondly they all need information on organisation structure and function, because the organisation will inevitably consists of personnel groups which do not have previous experience of working together. The training should therefore also give good knowledge about organisation and management structures, individual responsibilities, quality assurance measures etc.

32.2 Emergency Exercises

The need for large scale exercises at the field level is hard to justify because most of the operations in the field is normal work if the right staff perform it. However, there are reasons to conduct desk drills at national and regional level to maintain and improve emergency planning for reclamation and preparedness.

To maintain societies preparedness, consideration should be given to arranging suitable desk exercises for decision-makers and other key-staff at Nordic, national and regional level. Exercises of this kind should be concentrated on essential planning, decision-making and co-operation among decision-makers and other key staff at involved organisations.

As the intention is to use normally available equipment together with normal operators, it should not be necessary to train personnel on how to use equipment. Some key-personnel will need exercises on methods, before they start the work in the contaminated environment, to reduce radiation exposure.

33. Information

Information should be given in advance to the public about countermeasures that will be taken if radioactive materials are deposited. It will be easier to co-operate with the public if they are well informed and prepared to make their own decision to safeguard themselves and their dependants. The authorities should also give information and advise to the public on simple clean-up procedures they can use themselves to improve the situation.

Specific information should be given in advance to farmers on procedures in the agriculture system. It will be easier to co-operate with farmers if they are well informed and more prepared to make their own decision on their own safety and property.

Concise, understandable information about planned, ongoing and completed countermeasures should be given continuously to both the decontamination organisation personnel and the public.