Nuclear Safety

Cleanup of Large Radioactive-Contaminated Areas and Disposal of Generated Waste

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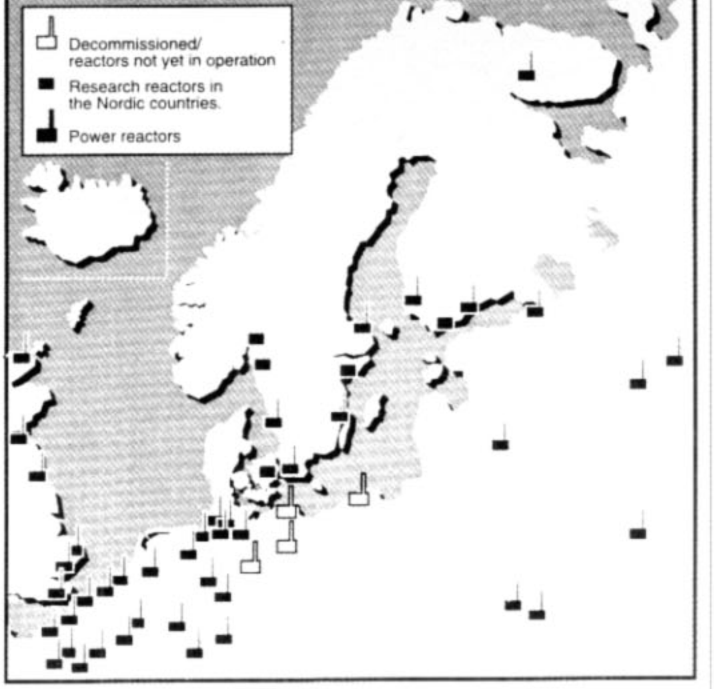

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Final Report of the KAN2 Project

February 1994

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The Nordic Council of Ministers
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The Nordic Council
was formed in 1952 to promote co-operation between the parliaments and governments of Denmark, Iceland, Norway and Sweden. Finland joined in 1955. At the sessions held by the Council, representatives from the Faroe Islands and Greenland form part of the Danish delegation, while Åland is represented on the Finnish delegation. The Council consists of 87 elected members - all of whom are members of parliament. The Nordic Council takes initiatives, acts in a consultative capacity and monitors co-operation measures. The Council operates via its institutions: the Plenary Assembly, the Presidium, and standing committees.

ABSTRACT

This report summarizes the results of the Nordic KAN-2 project. The purpose is to provide basic data and methodology to improve the planning of countermeasures for nuclear accidents. Special attention is paid to the tail-end topics of the cleanup process: Management and disposal of radioactive waste ensuing the cleanup of large areas after nuclear accidents. The results cover the following: A scenario of the amounts and activity concentrations of cleanup waste, description of the options for transportation and final disposal of the waste, costs of transportation and final disposal, justification-optimization and cost-benefit analyses. Experimental research was carried out in two areas: 1) scraping of surface soil layers and construction of a final disposal site for contaminated soil and 2) interactions of concrete with contaminated soil.

INIS descriptors:

CONCRETES; DECONTAMINATION; ENVIRONMENTAL TRANSPORT; GROUND DISPOSAL; RADIOACTIVE WASTE DISPOSAL; RADIOACTIVE WASTE FACILITIES; RADIOACTIVE WASTE MANAGEMENT; REACTOR ACCIDENTS; REMEDIAL ACTION; RURAL AREAS; SOILS; SURFACE CLEANING; URBAN AREAS; WASTE TRANSPORTATION.

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SUMMARY

A severe accident at a nuclear plant may result in a wide-scale contamination of the environment. If the contaminated areas are populated or cultivated, they may have to be cleaned up to protect the population from radiological hazards. A literature review performed in 1992 revealed that although experience is available from a number of accidents - Chernobyl, Kyshtym, Palomares, and so forth - the methodology needs improvement to facilitate decision making concerning the cleanup of waste after an accident. Therefore, the cleanup of urban, forest, and agricultural areas has been studied with special emphasis on the management and disposal of the radioactive wastes generated.

In the cleanup of wide radioactive-contaminated areas, large amounts of radioactive waste with varying activity will be generated. In order to make an estimation of the amounts and activity concentrations of a variety of cleanup wastes, a hypothetical accident at a nuclear power plant was used to calculate the radioactive releases. There are several factors, which will affect the form and activity of the cleanup waste. In the short term, i.e. if cleanup is done soon after deposition of the radioactive fallout, it is important whether conditions are dry or wet. In dry deposition more of the activity will be retained, for example in vegetation and on upper sufaces such as roofs. Subsequent rain will remove contamination from vegetation and upper surfaces and displace it into soil and water systems.

In the long term, contaminated soil, if removed, will form the most voluminous and active waste from cleanup operations, although the activity concentrations are likely to be low. Migration of most radionuclides in soil is rather slow and therefore they are retained in the upper layer of a few centimetres. If this upper layer is scraped off, the amounts of contaminated soil to be disposed of as radioactive waste will be very large. For example, if all gardens excluding parks from a metropolitan area of the size of Copenhagen were cleaned up by removing a 10 cm upper layer of soil, 2.2 million cubic metres of waste would be generated. Similarly, if all the agricultural fields in an area of 400 km^2 in Eastern Norway were cleaned up by removing a 5 cm upper layer of soil, 2.8 million cubic metres of waste would be generted. In other parts of the Nordic Countries with more intensive agriculture, the amounts of removed soil would be even higher.

Contaminated vegetation will also be an important cleanup waste, especially if the deposition takes place in the growing season and the cleanup work is done soon after the deposition. If vegetation harvested from agricultural fields is removed together with the upper 3-10 cm layer of soil, the vegetation may constitute three to twenty percent of the total waste volume from the fields. If contaminated grass is cut in urban gardens and parks, it will have a low volume compared to that of removed soil, but its activity concentration will be quite high if it is cut soon after the deposition. In forests and parks, the canopies would intercept deposited radionuclides very efficiently in the short term. If the trees are cut soon after the fallout, most of the radioactivity can be removed. In this case the amounts of waste will be high.

In urban areas a variety of waste other than soil and vegetation may also be generated. Most significant of these wastes would be dust from road sweepers and effluents arising if streets and other paved areas are decontaminated by firehosing. Sweeper dust will have a small volume but its activity concentration may be high. The volumes of firehosing effluents will be high, but their activity concentration will in any case be very low. Firehosing effluents cannot be regarded as waste as such, since firehosing merely results in the relocation of contamination, e.g. into open water systems and waste water treatments plants. Sewage sludges from waste water treatment plants may represent an important radioactive waste category after a nuclear accident.

Because contaminated soil usually forms most of the waste, it would be advantageous to minimize its volume by removing only a few centimetres of soil. Therefore, a variety of earth moving machinery was tested for this purpose in Northern Norway. Such extensive comparative studies have not been done previously. The performance of the machinery (minimum scraping depth, contamination removal efficiency, and time consumption) varied over a wide range. The performances depend on many factors such as the type of soil, whether the ground was dry or wet, the weight of the machine, etc. It is unlikely that one machine will be suitable for all types of situations. It could be shown, however, that in certain conditions at least 90 % of the contamination could be removed by scraping a few centimetres of soil. Furthermore, a new type of scraping process was tested: scraping of an only 2-5 cm thick layer of *frozen* peaty soil with a bulldozer. This resulted in removal of 90 % of the contamination. Scraping nonfrozen peaty soil was considerably less efficient.

In addition to the scraping tests, a "pilot-scale" disposal facility for removed soil was constructed. It includes three disposal trenches of 18 m^3 waste volume where about half of the waste is located 0.5 m below ground level. A radioactive tracer was used in order to determine the drainage. Only a very minor fraction of cesium has been eluted during the first year.

Pretreatment ("conditioning") of cleanup waste may sometimes be required or desirable before disposal. This may not be practicable with very large amounts of soil, but it could be of interest in connection with smaller incidents where moderate amounts of soil are contaminated and have to be stored or disposed of. In this project experimental work was carried out to investigate the possibilities for cement solidification of soil. A special additive may overcome the retardation effects caused by humic materials on the hardening of mixtures of cement and soil. It appeared that it is possible to prepare products with low cement content from many types of soil, but the products are porous, and leaching of cesium and strontium is in most cases faster than from pure soil. Access to carbon dioxide resulted in a considerable reduction in the leaching of strontium from cemented soil. This is due to precipitation of carbonates. Similar effects were noticed when contaminated soil was in contact with a concrete barrier as might be the case in a conventional burial facility. Clay and topsoil from the vicinity is commonly used for lining and covering shallow ground and surface mound repositories for cleanup wastes. This is excellent as far as cesium is concerned but it is questionable in case of strontium. Here a concrete barrier may be more efficient due to the carbon dioxide precipitation.

When cleanup waste is buried, it cannot be avoided that large amounts of vegetation will be mixed with the contaminated soil. The decaying organic materials have a profound influence on the short term water chemistry. Strontium, as well as cesium move faster in soil in the presence of decaying plant material.

In order to calculate the costs of various cleanup measures, three different final disposal options were considered: shallow ground trenches, surface mounds, and disposal in a natural valley. Shallow ground disposal was found to be somewhat more expensive compared to the surface mound option. Disposal in a natural valley was much more expensive. Disposal of 50,000 cubic metres of waste in one large disposal site instead of disposal of the same amount in five smaller sites was found to reduce costs considerably.

A cost-benefit analysis was made for urban areas. Here decontamination of roofs, walls, streets, trees, and gardens must be considered. Costs for cleaning these objects were compared with the reduction of radiation doses that can be obtained. Transportation and final disposal costs amount to 5 % of the overall cost for decontaminating roofs. They amount to 13 % for walls, 29 % for trees, and 30 % for streets. This is the first time that the overall cleanup costs including transportation and final disposal of ensuing waste have been taken into account in this kind of analysis. In case of wet deposition, it turned out that decontamination of gardens should be given first priorit and cleaning of streets second priority. Treatment of other surfaces would not be worthwhile. For dry deposition the same is true with the addition that also the cutting of trees is cost-efficient in reducing doses.

Remediation of contaminated forests has so far been studied to a limited extent only. Intuitively it has usually been considered that the cleanup of forest areas is not worthwhile. Therefore, a comprehensive study of the techniques, demands of costs and labour as well as the amounts of generated cleanup waste was carried out. A decision analysis was made for cleanup strategies of contaminated forests. In this analysis six different strategies were compared with each other. These strategies involved various cleanup measures, removal of trees, of undervegetation, and of soil from three areas considered in a hypothetical accident. In addition, control of access, control of wood material and "no action" were considered. In the analysis, collective dose to population, as well as individual doses to workers were calculated. The direct cost and capital cost of various actions was calculated. In addition to these measurable attributes, the effects of various actions on the stress to the population and on the quality of the environment were evaluated. According to the analysis, some cleanup actions appear to be quite reasonable. Removal of contaminated vegetation and soil from the two most contaminated forest areas together with "no action" for the third less contaminated area was found to be the most preferable choice of action.

SAMMANFATTNING

En allvarlig reaktorolycka kan leda till att stora områden i miljön kontamineras. Om de kontaminerade områdena är bebodda eller brukas kan de behöva saneras för att befolkningen ska kunna skyddas mot radiologiska risker. Projektet har studerat sanering av stadsmiljö, skog ock jordbruksmark. Speciell vikt har lagts vid hantering och förvaring av det radioaktiva avfall som uppkommer vid saneringen.

När utbredda radioaktivt kontaminerade områden saneras, uppstår stora avfallsmängder med varierande nuklidinnehåll. Projektet har gjort en omfattande uppskattning av vilka mängder som uppstår samt deras aktivitetsinnehåll för ett antal olika avfallstyper. På lång sikt gäller att det är kontaminerad jord, om den tas bort, som både har störst volym och högst aktivitetsinnehåll av de olika saneringsavfallen. Kontaminerad vegetation, som grödor och gräs, kommer också att vara betydande saneringsavfall, framförallt om depositionen sker under växtsäsongen och saneringen sker tidigt efter nedfallet. I stadsmiljö uppstår en stor mängd andra typer av saneringsavfall, till exempel stoft från gatsopning.

För att studera möjligheterna att minimera mängden avlägsnad jord från åkermark gjordes praktiska försök med ett antal jordskrapsmaskiner i Nordnorge. Maskinernas prestanda (minsta skrapdjup, effektivitet i dekontaminering, tidsåtgång) varierade inom ett ganska stort intervall. Det var dock möjligt att visa, under vissa förhållanden, att minst 90 % av kontamineringen kunde tas bort genom att skrapa bort några få centimeter av jorden. Förutom skraptesten konstruerades en förvarsanläggning i "pilotskala" för den borttagna jorden. Utläckaget av ^{134}Cs har därvid studerats.

Experiment har också genomförts för att studera möjligheterna att solidifiera jord med cement genom att utnyttja ett speciellt tillsatsämne med förmågan att förhindra humus materialens retardation av härdprocessen. Stabila produkter med lågt cementinneehåll kan tillverkas från de flesta jordtyper, men utläckaget av cesium och strontium visade sig iregel bli mycket snabbare än från den obehandlade jorden. Tillgång till koldioxid resulterade dock i en betydande minskning i utläckaget av strontium. Liknande effekter konstaterades när kontaminerad jord var i kontakt med en betongbarriär. Nedbrytande organiskt material visade sig starkt öka läckaget av strontium från kontaminerad jord. Detsamma gällde även cesium i en tidig fas, men ökningen avtog med tiden.

Beräkningar av saneringskostnader, samt transport och slutförvaring av saneringsavfallet visade att placera avfallet i jordhögar är något mera ekonomiskt än en iordningställd markdeponi för avfallet och mycket mera ekonomiskt än förvaring i en naturlig dalgång. Kostnaderna reduceras avsevärt om 50 000 m^3 förvaras på ett ställe jämfört med om samma mängd hade delats upp mellan fem olika ställen.

En kostnads/nytta-analys för stadsmiljö har genonförts där sanering av tak, väggar, gator, träd och parker har studerats. Det var första gången som alla saneringskostnader, inklusive transport och slutförvaring, beaktades i en sådan analys. Analysen visade att för våtdeposition skall sanering av parker ges första prioritet och sanering av gator andra prioritet. Behandling av andra ytor är inte meningsfullt. Om deponeringen skett under torra förhållanden är det även kostnadseffektivt att hugga ner träd att reducera doser.

En omfattande studie av teknik, behov av och kostnad för arbetskraft, samt resulterande mängder saneringsavfall har även gjort för skogsmark. Saneringsstrategier jämfördes i en beslutsanalys, vilken visade att sanering genom att ta avlägsna vegetation och jord från områden med högre kontamineringsnivåer skulle kunna vara ganska rimlig.

CONTENTS

1. INTRODUCTION

As a result of nuclear accidents large areas may be contaminated by radioactive fallout. Major accidents may contaminate thousands of square kilometers with an initial activity even higher than 10 MBq/m^2. The accidents that can cause a large-scale contamination are: a) melt-down and/or explosion and fire at nuclear power plants; b) explosion or fire at the storage sites for medium- or high-active waste solutions; and c) an accident involving nuclear weapons. Radioactive fallout deposited on the ground is a radiological hazard to humans living in the contaminated area. The radiation dose received is caused by external exposure from the radioactive material on the ground or in the air and by internal exposure from inhalation and from ingestion of contaminated foodstuffs.

The contaminated areas may have to be cleaned up to protect the population from radiological hazards. The cleanup measures may include the decontamination of houses, streets, gardens and fields, and the stabilization or isolation of contamination. Decontamination means the removal of radioactive material from contaminated surfaces or the removal of the contaminated surfaces in order to reduce the remaining activity on the surface. In stabilization the contamination is fixed on the surface to decrease its mobility. Radioactivity can also be isolated by covering it with a layer of clean material, such as concrete or soil or by deep ploughing to remove the contamination from the upper layer of soil.

Most of the cleanup measures result in the formation of radioactive waste that have to be disposed of in a safe manner. The removal of contaminated soil and vegetation generates huge amounts of waste. Prior to any decision on which cleanup measures will be taken, it should be known how to dispose of the resulting waste. There are many factors affecting the distribution of contamination and, consequently, also the volumes and physical forms of the waste. The distribution of contamination on different surfaces is very much dependent on whether the deposition took place in dry atmospheric conditions or whether it came down with rain. In dry deposition the contamination is mostly retained on upper surfaces (roofs, canopies) and on vegetation. In wet deposition a higher proportion moves to lower surfaces and soil. The effect of the season on waste is obvious: in the growing season the activity is retained by vegetation more readily than in other seasons. In the winter the contaminated snow and ice may be the most important source for waste. The time period between deposition and cleanup is also very important. If, for example, contaminated grass is cut immediately after dry deposition, most of the activity can be removed, but if not the subsequent rains and rotting of the grass will move the contamination into litter and soil. The distance of contaminated area from the release source may also have an important effect, since closer to the source greater part of the contamination is associated with particles of low solubility.

Typically the major waste products are vegetation and top soil, but large amounts of waste similar to ordinary municipal waste may also be produced, especially in urban areas. The waste coming from the cleanup work is mainly low-active. The experience of the management and disposal of low- and intermediate-level waste from nuclear power plants and other nuclear facilities is extensive. Experiences of the disposal of contaminated soil in hundreds of thousands of cubic meters have been obtained in the cleanup of uranium mill tailings sites and nuclear weapons test sites. This experience is, however, only partly useful when considering the waste problem in major nuclear accidents. First, the amounts of waste may be extremely large when vast contaminated areas are to be cleaned up after an accident. Second, the accident situation is always unexpected, although tentative planning for such situations may exist. Under normal circumstances planning and the actual operation may take years, whereas in case of an accident cleanup work may have to be started immediately.

2. REVIEW OF CLEANUP OF LARGE RADIOACTIVE-CONTAMINATED AREAS

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2.1. EXPERIENCE OF CLEANUP AND OF MANAGEMENT AND DISPOSAL OF CLEANUP WASTE AFTER NUCLEAR ACCIDENTS

Following the nuclear accidents, which have resulted in the contamination of the environment outside nuclear facilities, experience has been obtained on cleaning up large areas and disposing of large volumes of resulting radioactive waste. Especially this is true in the case of the Chernobyl accident. In additon to the accidents, experience on large-scale handling of radioactive waste has been obtained in the management and disposal of uranium mill tailings and the cleanup of a nuclear weapons test site. Transportation and final disposal of uranium mill tailings will be described later in sections 2.3 and 2.4. In this section only those accidents, which resulted in the formation of large amounts of radioactive waste, have been dealt with. Therefore, the Windscale accident in 1957 and the Three Mile Island accident in 1979 have been excluded.

2.1.1. The Kyshtym accident in 1957 and contamination of River Techa and Lake Karachai

In September 1957, a tank containing high-active waste solution, exploded at a plutonium separation plant Majak in Kyshtym, the USSR, about 100 km south of Sverdlovsk. About 10^{16} Bq ($2\cdot10^{6}$ Ci) of fission products was released. The short-lived nuclides ^{144}Ce and ^{95}Zr accounted for 91 % of the total released activity. The long-term radiological hazard was caused by ^{90}Sr, which accounted for 5.4 % of the activity, $4\cdot10^{15}$ Bq ($5.4\cdot10^{4}$ Ci). The radioactive deposition took place in dry atmospheric conditions. An area of a total of 15,000 km^2 was contaminated with ^{90}Sr activity higher than 3.7 kBq/m^2 (0.1 Ci/km^2) (Romanov 1991a).

Areas with contamination higher than 74 MBq/m^2 were interdicted. In the area of Chelyabinsk and Sverdlov a total of 106,000 hectares of agricultural land was taken out of production (Nikipelov 1990). About 500 severely contaminated animals were slaughtered, and the carcases were buried (Arkhipov 1991). Ploughing was, however, the main reclamation method used in rural areas. Altogether 20,000 ha of contaminated land was ploughed to the normal depth over the two years following the accident. During the subsequent two years, areas where the contaminated surface layer had not been adequately covered up by the ploughing, a total of 6200 ha, were deep ploughed to over 50 cm (Nikipelov 1990). This produced a 2-7 times reduction in the ^{90}Sr concentration of agricultural products. Using amendments and agents, which enhance the fixation of strontium in soil, such as calcium, sulphate, phosphate and silicates, decreased the ^{90}Sr concentration by a factor of 2-4. The removal of surface soil (5-10 cm), a cleanup method which was used only in a small area, reduced the ^{90}Sr concentration 5-15 fold (Romanov 1991b).

This so-called Kysthym accident is not the only, and not the most serious, contamination problem in the Majak plant area. The plant dumped from 1949 to 1956 their high-level radioactive waste solutions directly into River Techa. In total about 3 million curies (10^{17} Bq) were dumped. River Techa is running to the River Ob, the greatest Siberian river. As a result of the dumping, contamination was observed as far as 1000 km away at the Arctic Ocean. After stopping dumping into River Techa, the high-level waste was then from 1956 released into Lake Karachai. Today the contamination in Lake Karachai is about 120 million curies ($4 \cdot 10^{18}$ Bq), which is about 2.5-times as much as released in the Chernobyl accident in 1986. As a result of a drought in the area in 1967 the lake dried and a tornado distributed high amounts of radioactive material from the lake over a large area (Khodakovsky 1992). Planning of the remediation of the Majak area is going on, and it is clear that the actual remediation process will be very difficult and take a long time.

2.1.2. The Palomares accident in 1966

In January 1966, an American B-52 bomber, carrying four nuclear weapons, exploded in the air at an elevation of 10 km over Palomares, Spain. No nuclear explosion took place, but two of the four bombs the aircraft was carrying broke on impacting with the ground; one at a distance of 0.5 km from Palomares and the other at 2 km. Palomares had 2000 inhabitants at that time. A total of 255 hectares was contaminated with Pu: 2.2 hectares had a contamination level higher than 460 $\mu g/m^2$, 17 hectares between 54-460 $\mu g/m^2$ and the rest below 54 $\mu g/m^2$ (Iranzo 1968, DNA 1975).

A layer of about 5 cm was removed from the surface soil in areas where the contamination levels exceeded 460 μg of Pu m^{-2}. The ground was scraped into piles with graders, and was later on transported into an interim storage. The remaining plots, where the levels still exceeded the acceptable levels, as well as areas where the grader could not be operated, were removed with spades. A total of 900 m^3 was removed from an area of 2.2 ha (DNA 1975), whereafter the alpha activity of the soil was at the background level (Iranzo 1968). The plots whose contamination levels were 5-460 μg of Pu m^{-2} were irrigated and ploughed to 30 cm, after which the surface layer was homogenized by tilling. Plots with a contamination level below 5 μg of Pu/m^{-2} were only irrigated. In rocky waste lands, which could not be ploughed, the upper limit was 77 μg of Pu/m^{-2}. These areas were also irrigated with water. The ploughed area was 115 ha, as was the irrigated land. Plants with count rates over 200 cpm were removed and those with rates under 400 cpm were burnt in a dry riverbed near the coast on nights when the wind was blowing toward the sea. Plants with count rates over 400 cpm were treated as waste (DNA 1975).

The debris from the disintegrated aircraft and nuclear weapons were first collected into containers, then transported by truck to the nearest airport and flown to Texas, the United States. The rest of the waste to be transported out of Spain, i.e. the contaminated soil and vegetation, 1150 m^3 in total, were transported by a cargo ship to the port of Charleston in the USA, where they were moved onto 26 rail cars and transported to the Savannah River active waste storage site for final disposal in shallow-ground trenches (DNA 1975).

2.1.3. The Thule accident in 1968

In January 1968, an American B-52 bomber, carrying four nuclear weapons, crashed on the sea ice off Thule, Greenland. Both the aircraft and the weapons disintegrated, but no nuclear explosion took place. The aircraft debris and the plutonium from the disintegrated weapons were distributed over an area of about 22 hectars. A total of 3.2 kg of plutonium was distributed on the ice (Langham 1970).

The debris from the airplane and nuclear weapons were collected manually into sacks. The contaminated black snow and ice were gathered in mounds with a planer (Dresser 1970). The cleanup of the site whose area was about 6 ha (Langham 1970) produced altogether about 6700 m^3 of waste (Hunziker 1970). Approximately 93% of the plutonium had been removed in the cleanup (McRaney 1970).

The waste was transported by two cargo ships to USA. The debris from the aircraft and from the nuclear bombs was loaded into containers for transportation. The snow was transported in 95 m^3 tanks. From the port of Charleston in the United States the waste was moved by trains to the Savannah River site for disposal. The first rail transport comprized 147 cars. The waste was disposed of at Savannah River in shallow-ground trenches (Otten 1970, Dresser 1970).

2.1.4. The Ciudad Juárez accident in 1983

In December 1983, a teletheraphy unit, containing $1.7 \cdot 10^{13}$ Bq (450 Ci) of ^{60}Co in form of 6000 pellets, was dismantled and used as scrap metal in foundries in Ciudad Juárez, Mexico. This resulted in the contamination of thousands of tons of metal products, several foundries and streets, and hundreds of houses. The accident required a large-scale cleanup program both in Ciudad Juárez and Chihuahua (Molina 1990).

In the towns of Ciudad de Juárez and Chihuahua ^{60}Co pellets and metal pieces contaminated with ^{60}Co were collected and stored in concrete containers and steel barrels. Contaminated soil was removed and contaminated streets were cleaned up. 814 buildings were demolished totally or partially. The cleanup work produced 21,000 m^2 of radioactive waste (Molina 1990).

2.1.5. The Chernobyl accident in 1986

In April-May 1986, a melt-down and explosion of a 1000 MW graphite-moderated reactor took place in Chernobyl, the USSR. The activity released from the reactor was about $2 \cdot 10^{18}$ Bq ($5 \cdot 10^7$ Ci), of which about $4 \cdot 10^{16}$ Bq (10^6 Ci) was ^{137}Cs and 10^{16} Bq ($2 \cdot 10^5$ Ci) ^{90}Sr. The accident resulted in a serious contamination of thousands of square kilometers in the Ukraine, Russia and especially in Belorussia. In June 1986, the exposure rate was higher than 20 mR/h in the most severely contaminated area (870 km^2) close to the plant, and the corresponding surface activity was about 300 MBq/m^2 (IAEA 1989).

The cleanup of towns and populated centres was undertaken soon after the accident had taken place. Buildings were cleaned mainly by washing their surfaces with a decontamination solution containing hydrophilic surface active agents in water or acid solutions (Komarov 1990, IAEA 1989). Firehosing was also used for washing the buildings and streets. As a result, the radiation level in some buildings was reduced to the background level, however on other surfaces and roofs this method proved ineffective. After the buildings had been washed, radioactivity in the upper soil layer near the walls increased by 2-2.5 times (IAEA 1989). A 5-10 cm thick layer of topsoil was removed, in small garden areas manually (IAEA 1992). By summer 1987, about 600 population centres, a total of about 60,000 houses and other buildings, had been decontaminated (IAEA 1989). In 1987, the town of Chernobyl, which had not been as severely affected as the town of Pripyat, had been totally decontaminated and several buildings had been taken into use to house the cleanup personnel. In the town of Pripyat, the cleanup operations were performed twice, however in 1988 it was decided that the town of Pripyat would not be used any more (Varley 1988).

Large rural areas were ploughed, tilled and harrowed to bring down the activity level and to prevent material from being carried with the wind and rain (Komarov 1990, IAEA 1989, Nikipelov 1990). Production of agricultural products was prohibited in the areas, where the contamination level was higher than 1.5 MBq/m^2, and it was decided to afforest the fields in these areas. In the areas, where the contamination level was lower than 1.5 MBq/m^2, several reclamation measures were taken (Asmolov 1988). Hundreds of hectares were tilled, and sown with grass (Ko90). As a result, the contamination level of the grass was 8-10 times lower than in untilled lands (Izraehl 1988). After the grass has grown, mixed forest were planted on some of this land. The goal is to gradually sow grass and plant forests on waste land and less productive farmland in the Ukraine and Belorussia, altogether about 40,000 ha (Komarov 1990). After harvesting, the turf layer was removed from farmlands in the vicinity of the damaged nuclear power plant. In some areas a latex emulsion was used to fix the sod. In some other areas, also the surface soil layer was removed. The majority of the contaminated fields, hundreds of thousands of hectares of land, were, however, deep ploughed, adding lime, potassium and phosphate fertilizers, as well as a clay suspension or zeolite (IAEA 1989). After one year following the treatment, the activity concentration in the agricultural products was lower by a factor of 1.5-3 (Il'in 1988).

Near the Chernobyl NPP about 400 ha of forest, the so-called Red Forest, was dying in the autumn of 1986 due the high radiation doses it had been exposed to. At least 200 ha of this forest were felled. The trees, the understory vegetation and the litter were removed and buried into 1.5-2 meter deep ditches and the whole area was covered with a 0.5-1 meter layer of clean sand. These measures decreased the radiation level near the Chernobyl plant by a factor higher than 100 (Komarov 1990). In total about four million cubic metres of waste were placed in about 800 disposal sites (Komarov 1990). Most of these sites were surface mounds, covered with polyethylene film and clean earth. Some sites were, however, shallow trenches of about 10,000 m^3 volume, which were lined at the bottom with clay and on the top with clay and clean soil (IAEA 1992).

2.1.6. The Goiania accident in 1987

In September 1987, a cancer therapy unit, containing $5 \cdot 10^{13}$ Bq (1375 Ci) of ^{137}Cs, was opened by two scavengers and the metal parts were sold as scrap in Goiania, Brazil. The capsule containing the Cs salt was also damaged resulting in widespread contamination. Hundreds of people were contaminated in Goiania, which has one million inhabitants, of which 112,000 were monitored. About ten sites (houses, junkyards, hospitals etc.), in total about 5000 m^2, were contaminated to an extent that cleanup was necessary. Some of these had very heavily contaminated spots with a dose rate of 50 up to 2000 mSv h^{-1} (Rosenthal 1991, Vinhas 1990).

The most severely contaminated spots were first removed, after which all waste paper and scrap was removed from the area. The trees were felled, the apartments were evacuated, and all loose objects were removed and treated as radioactive waste. Loose materials had to be removed manually before heavy equipment could enter the area. On three sites the buildings were so badly contaminated that they had to be demolished with excavators, and their metal frames were cut into pieces. On other sites the buildings were cleaned up. Parts of the floors where contamination had fixed fixed were removed, and the floors were covered with a new concrete layer. The contaminated soil was removed from the sites, and the area was covered either with uncontaminated soil or concrete (Vinhas 1990). The contamination had spread with the wind, humans and animals from the areas in which the ^{137}Cs source had been broken up and the contaminated parts had been treated (Amaral 1991, da Silva 1991). Gardens and buildings within a radius of about 50 m around the actual site had to be cleaned up. The contaminated fruit were picked and the branches of the trees were cut. As 60% of the cesium on the residential sites had accumulated in the 1.5 cm thick surface soil, it was carefully removed manually in layers of about 1.5 cm until the acceptable dose rate level of 0.8 μGy h^{-1} or the acceptable concentration level of 22.5 Bq kg^{-1} was obtained (Amaral 1991).

Chemical and physical methods were used in decontaminating the buildings. The roofs were water-jetted and the ceilings vacuum swept. The roofs of two buildings had to be removed because the washing methods were inefficient. Paint was scraped off the walls with sandpaper. The floors, mainly of concrete, were treated with chemical substances, and washed with a suspension containing Prussian Blue, which acts as a fixation agent for cesium. The surfaces of the most heavily contaminated parts of the floors, were removed with a jewelry drill and electrical appliances. The most seriously contaminated areas were the bathrooms and other washrooms, and large quatities of active material had to be removed from them and their vicinity before the acceptable surface contamination level of 3.7 Bq cm^{-2} was reached. Altogether almost 50 buildings were decontaminated (da Silva 1991). Affected domestic animals, like dogs, pigs, rabbits and fowl were killed and treated as radioactive waste (Miaw 1990). The cleanup measures in Goiania produced a total of 3340 m^3 of active waste, of which 130 m^3 contained 72 % of the cesium (Mezrahi 1989).

2.1.7. Cleanup of the nuclear weapons test site at the Enewetak atoll

In the years 1948-58, a total of 43 nuclear weapons tests were carried out by the USA at the Enewetak atoll, in the Pacific Ocean. The atoll consists of 40 islands, with a total area of 713 hectares. Due to the tests, contamination levels on 4 islands were higher than 15 kBq/kg (400 nCi/kg) of soil and on 8 islands 1.5-15 kBq/kg (40-400 nCi/kg) of soil (DNA 1981).

In the years 1977-80, a large-scale cleanup program was carried out in the contaminated islands. Vegetation was first removed from areas requiring the removal of soil. When the piled up shrubs had become sufficiently dry, they were burnt. The ashes were either removed or left on the ground, depending on their activity levels. The soil was removed with a bulldozer (DNA 1981). As the distribution of radionuclides in the ground varied considerably in different areas (Gudiksen 1975), the soil was removed in layers of 15 cm and the operation was continued until the required concentration level was reached (DNA 1981). Some earth was removed from five islands. The trenches of active waste that existed on some islands were also removed. The $^{239,240}Pu$ content level 40 pCi/g, which was recognized suitable for human settlement, was reached on 30 islands, the level 80 pCi/g suitable for farming was reached on 7 islands, and the level 160 pCi/g suitable for growing fodder was reached on 2 islands. The island of Runit, where the waste was buried, was interdicted. The waste, altogether 80,000 m^3, was disposed of in a crater (7.5 high, 113 m wide) created by a nuclear explosion. The soil in the crater was solidified with cement. Finally the crater was closed with a 46 cm thick concrete cap (DNA 1981).

2.2. CLEANUP METHODS

There is wide variety of methods, which can be applied to the cleanup of radioactive contaminated surfaces. This is especially true for urban areas, where a broad range of surfaces may have to be cleaned up. Cleanup methods have been described in more detail in a literature survey done as a part of this project (Paajanen 1992). Recently reported methods, which were not covered in the survey, are described in Appendix 1.

2.2.1. CLEANUP OF URBAN AREAS

2.2.1.1 Distribution of Contamination in Urban Areas

In urban areas, radioactive fallout is distributed on a variety of surfaces, notably on the roofs, walls and interiors of buildings, on streets, and on trees, shrubs and lawns in parks and gardens. The relative distribution of contamination on various surfaces depends largely on whether the deposition took place with precipitation or under dry conditions. In the case of dry deposition the contamination is mainly deposited on roofs and streets, on the leaves of trees and shrubs, and on lawns. During wet deposition, roofs, streets, trees and shrubs are washed by precipitation, which causes more contamination to accumulate on the ground and into the sewage system. The ensuing precipitation moreover leaches contamination from the topsoil deeper down. The distribution of contamination following deposition depends further on particle resuspension, caused, for example, by the wind, vehicle traffic and possible cleanup measures.

2.2.1.2 Cleanup Methods in Urban Areas

Because of the wide variety of the surfaces to be cleaned up, there are several methods, which can be applied for this purpose. They can be classified into the following groups (Roed 1986).

a) Methods for removing contamination without damaging the surface - sweeping, vacuum sweeping, firehosing, water jetting and regular cleaning;

b) Methods for removing the top layer of the contaminated surface, parts of the surface or the whole surface - sandblasting, planing, scraping, spalling, cutting vegetation, removing surface soil, renewing roofs, or demolishing buildings;
c) Methods for fixing contamination onto the surface or introducing it deeper into the contaminated site - painting, ploughing, turning the soil, turning flagstones or repaving roads.

Taking into account the relative contributions of different surfaces to radiation doses to inhabitants in urban areas, the first priority in cleanup measures should be given to decontamination of gardens and trees, and the second to that of roads and streets. Cleaning up surfaces like roofs and walls with the methods available is relatively inefficient and rather expensive (Roed 1990). The cost-effectiveness analysis made by Bown et al. gave the highest priorities to the following cleanup methods: grass cutting, ploughing, sweeping, vacuum-sweeping and firehosing roads (Table 1) (Brown 1991). This analysis took into account the decontamination efficiences (decontamination factors), costs and contribution to dose reduction (dose reduction factors) of different cleanup methods.

Table 1. Cost-effectiveness of decontamination for ^{137}Cs in urban areas (Brown 1991).

Very beneficial	Grass cutting
	Ploughing
	Vacuum sweeping roads
	Sweeping roads
	Firehosing roads
Moderately beneficial	Road planing
	Soil removal
Unlikely to be worthwile	Firehosing buildings
	Roof replacement
	Sandblasting buildings
	Ammonium nitrate treatment
	Cleaning indoor surfaces

The authors note, however, that the results of their study may not be applied directly to all cases, because of the following limitations, which also have to be taken into account: "i) the feasibility of the techniques on a large scale, ii) the number of the decontamination workers required, iii) the doses received by workers during decontamination, iv) the effectiveness of the techniques for elements other than cesium, v) the costs of disposal of by-products of the decontamination processes" (Brown 1991). In the present study points ii) and v) have been estimated (Chapter 6) and taken into account in a cost-benefit analysis (Chapter 7).

2.2.2. CLEANUP OF FOREST AREAS

2.2.2.1 Distribution of Contamination in Forest Areas

In forests radioactive fallout is deposited onto trees, understory vegetation, litter and the soil. The relative distribution is largely dependent on the thickness and nature of the vegetation.

Under dry deposition and in areas with thick forestation the deposition mainly remains in the leaves and needles. Under wet deposition and in areas with a sparse tree stand, deposition leaches into the understory vegetation, litter and the soil in larger quantities. Shortly after the deposition most of the activity is in canopies. In deciduous forests, where the leaves of trees fall every autumn, most of the activity will be found in litter and underlying soil after the winter. Moreover, the leaves decompose in a short time, which results in a rather rapid transfer of the activity into the soil. In coniferous forests, however, the retention time in the canopy is longer and transfer of the activity into soil takes a longer time, because needles decompose more slowly than leaves. In winter time most of the radioactive contamination is deposited onto the snow laying either on the trees or on the ground. In the spring the contamination runs with melt-waters partly into the litter and soil and partly into the water systems.

2.2.2.2. Cleanup Methods in Forest Areas

Cleanup of forest areas is a very difficult task, because of the huge amounts of ensuing wastes. The most heavily contaminated forests close to populated areas or industrial plants are the most probable objects for decontamination. Forests play an important role in the retention of contamination and therefore, interdiction of contaminated forest areas and prevention of forest fires are probably the most important ways to reduce the radiation exposure to population.

If cleanup is, however, required the only way to carry it out is to remove partly or totally the contaminated vegetation, litter and soil. Trees can be removed with conventional harvesters or wood processors or in some cases manually. To lower the costs and the radiation doses to the cleanup workers manual work should be minimized. In difficult terrain and when finishing the cleanup work manual work may, however, be necessary. Removal of undervegetation, as well as litter and soil requires a lot of heavy-duty equipment such as bulldozers and diggers. Removal of trees only is effective in removing contamination immediately after the deposition, because the canopy can initially intercept as much as 60-90 % of the deposition (Tikhomirov 1992). This is especially true in summer time, when there are leaves on deciduous trees and in the case of dry deposition. Removal of undervegetation only, which should be probably done manually, can be applied for only very limited areas and it is effective in removing contamination in an intermediate time range after deposition. As early as a year after the deposition more than 90 % of the contamination lay under the trees, mostly in the litter. Mechanical removal of undervegetation together with litter, humus, stumps, roots and upper soil layer is effective years after the deposition, when more and more activity has transferred into the soil due to the decomposition of the litter. There is no data available on the effectiveness (decontamination factors) of different cleanup measures used in forest areas.

Later on, when considering the amounts and activity concentrations of the waste, as well as the costs of the cleanup work, the following scheme has been used for the cleanup of forests. In this sceme there are two subcases: either removed trees are transported out of the affected area for further use or final disposal, or they are disposed of on site. When transporting the waste off the area, they will be first felled with a harvester but not lopped. Non-lopped trunks will then be moved with forest tractors to road sides for chipping and transportation. Chipping can be done in order to reduce of the volumes to be transported. Chipping should be done so that the chipped wood material can be loaded directly from the hacker to the transportation trucks or trailers. If stumps also have to be removed, it should be done with stumps harvesters. Stumps will be moved with tractors to road sides and chipped for the transportation. Removal of undervegetation, litter and humus, as well as soil, should be done with bulldozers or diggers. Removed material is loaded with diggers or loaders to dumpers or directly onto trucks. Transportation off the area should be done by trucks and trailers. If the disposal is to take place in the affected site, moving of the material can be done with forest tractors and no trucks would be needed. Chipping of trees and stumps is also not necessary for transportation, but it will greatly help in their disposal, because it reduces the volumes and enhances the rotting. In the Nordic Countries there is large pool of technical knowledge and machinery to carry out a cleanup procedure described above.

2.2.3. CLEANUP OF RURAL AREAS

2.2.3.1 Distribution of Contamination in Rural Areas

In agricultural areas, radioactive contamination is deposited onto vegetation and soil. The relative distribution depends on the nature and density of the vegetation. During wet deposition, particularly in areas with scanty vegetation, most of the activity ends up in the soil, whereas during dry deposition and in areas of high and dense vegetation the opposite is true. Distribution is largely dependent on season, so that vegetation retains more activity during the growing period as foliar deposition. In a short time after dry deposition the activity in vegetation is attributed to surface contamination and harvesting represent an efficient cleanup measure. After decomposition of vegetation during winter time the activity is transferred into surface soil layers. Thereafter, the activity in the vegetation originates from the uptake by plant roots. Contamination in the soil is retained in the topmost layer of a few centimetres depth. Vertical migration of radionuclides in soil depends on their physico-chemical forms and interactions with soil components.

2.2.3.2 Cleanup Methods in Rural Areas

The reclamation methods used in agricultural areas affected by radioactive contamination can be divided into two main groups (van Dorp 1981, Menzel 1971):

a) Methods for cutting and removing the contaminated vegetation and contaminated surface soil.
b) Methods for preventing the uptake of radionuclides by plants, without removing the contamination - ploughing, fertilizing, liming and other soil amendments, irrigation and leaching, using cesium-binders for animals, changing the production and decontamination of foodstuff. These methods do not mainly create any radioactive waste.

Removing contaminated vegetation yields highly varying results, as the amount of radioactive material that remains in the vegetation depends on its density and the conditions under which the contamination took place. This method is most efficient in reducing radioactive contamination in cases of dry deposition soon after the fallout. Vegetation from fields can be removed efficiently with conventional crop-harvesters, but precautions to prevent the formation and spread of dust are needed. Radiation protection of the cleanup personnel is an important task, especially if any measures are taken prior to decay of ^{131}I.

Depending on the soil type and equipment, radioactive material can be mostly, 80-95 %, removed by scraping the surface 5-10 cm layer of soil. If the ground is covered with lush vegetation, it must first be cut, as it may interfere with the removal of thin layers of earth. Standard earthmovers, such as planers, bulldozers and various scrapers can be used for removing the surface soil. Vacuum sweepers are also suitable for removing particulate contamination from meadows and even from soil surfaces.

2.3. TRANSPORTATION OF CLEANUP WASTE

The transport methods used to radioactive waste from a contaminated area to a disposal site depends on the distance between the disposal site and the area to be cleaned up, and on the location of the disposal site. If the disposal site is not within the 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 radioactive wastes. Observance of these regulations is not required if the transportation takes place in the controlled area (IAEA 1992).

2.3.1. Safety of the waste transportation

General safety factors to be observed in the transport of radioactive wastes (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) Preparedness for transport accidents.

f) Monitoring and registering the quantity, physical and chemical form and activity of the transported wastes.

g) Protecting cleanup workers and the general public against exposure to radiation, e.g. by placing additional radiation shields on the vehicles.

2.3.2. Transporting wastes within the controlled area

Solid wastes, such as contaminated soil and vegetation, can be removed within short distances (100-200 m) directly into a storage mound, trench or natural basin with bulldozers or scrapers. However, having a large number of waste disposal sites in a vast area may not be a good solution as far as long-term control is concerned. It can be at least used as an interim solution in the absence of adequate hauling equipment or an appropriate final disposal site, and for preventing the resuspension of contamination.

Centralized waste disposal includes the loading, transport and unloading of wastes. It is possible to use heavier hauling equipment in closed areas than 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 available hauling vehicles. The loading platforms were covered with a polyethylene film before and after loading, and the loads were moreover covered with a tarpaulin. The vehicles were decontaminated and monitored at the boundary of the zones with lower activity levels (IAEA 1992).

2.3.3. Transporting wastes outside the controlled area

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

Radioactive wastes can be transported outside the controlled area by road, rail or water. The choice of transport method is based on its safety and cost effectiveness.

Large amounts of low-level waste have been transported safely even over long distances by road and rail in connection with the dismantling of closed-down uranium plants and the cleanup 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 cleanup 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 of 20 m^3 were used in the transport operations. Before loading, a polyethylene film was stretched over the platform to prevent the dropping of contaminated material during transport. After the loading, the vehicles were decontaminated, and the load was covered with a tarpaulin and weighed. The transport route was along public roads. The route was monitored regularly for possible contamination but there were no signs of any waste dropping during the journey. 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 to a natural valley. Before the return trip, the vehicles were washed and monitored for surface contamination (IAEA 1992).

Radioactive wastes can be transported also by rail, provided that there is a railroad between the cleanup and the disposal sites. The cost of rail transport may exceed those of direct road transport if double or triple loading and unloading is required, as for example in a truck-train-truck combination. Even in the case of shorter distances, transport by train is less cost-effective than by truck. However, transport by rail is often a safer alternative because the public roads, or side roads not well-suited for heavy traffic, can then be avoided. It is moreover 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 moved by rail from the area around the Virto 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 one was being unloaded. At South Clive the waste was unloaded by clamping one carful at a time with a turnover. 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). Spread of the contaminated material during transport was prevented by covering the truck loads with tarpaulin sheets, and the waste in the rail cars was sprayed with a polymer surfactant (IAEA 1992).

2.4. DISPOSAL OF CLEANUP WASTE

2.4.1. Interim storage

In the absence of final disposal facilities, or when planning and constructing them, it may become necessary to store the waste in interim storages. Interim storages may also be needed for sludges and liquid wastes whose concentration and solidification may take a long time. Interim storage may be applied also to wastes rich in organics and whose volume may reduce considerably as the result of the decay and degradation of the organics.

In the Chernobyl area, where great amounts of radioactive waste had to be disposed of very quickly at the initial stage when no final disposal sites or facilities were yet available, the waste was stored up in surface mounds near the removal sites. The storage sites were located as far as possible from water systems and their catchment areas. The bottoms 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 organics, was collected in mounds with bulldozers. The mounds were first covered with a polyethylene film and then with clean earth. As the zone was closed off, a ditch was dug around it and warning signs were posted (IAEA 1992).

2.4.2. Final disposal
(IAEA 1981, IAEA 1982, IAEA 1984, IAEA 1985, IAEA 1992)

2.4.2.1. Selecting the final disposal site and the disposal method

When disposing of radioactive wastes the general principle is that the wastes must 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 realized so that the migration of radionuclides from the waste with water is prevented, and intrusion by humans, animals or the roots of plants is hampered. The waste must moreover be shielded against 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 in case of 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 could be improved by engineered barriers and waste treatment methods, for example, by compacting the waste, by using immobilization materials as backfill and in the structures of the disposal facility, by using flow barriers and drainage, and by shielding the waste against erosion and intrusion. Following the closure of the repository, containing long-lived radionuclides, institutional control is required to ensure that the requirements of radiation protection will be met. Low-activity wastes, which mainly contain ^{90}Sr and ^{137}Cs radionuclides, usually require a form of disposal which will last for a few hundred years.

In the case of an accident where large amounts of radioactive waste must be disposed of promptly, the situation differs greatly from the normal. The disposal site or sites usually must be selected very quickly using data from regional studies made earlier. The costs of loading, packing and transporting large amounts of waste is a significant consideration. Time is another constraint when constructing disposal facilities, and therefore the materials needed in the structure of engineered barriers and backfill should be readily available. It should moreover be possible to carry out the constructing and the disposal with machines and equipment normally available. Therefore the use of a bedrock facility or concrete trenches, which may take several months or even years to construct, is likely to be very limited.

2.4.2.2. Shallow ground disposal

Burying the waste in the ground is a method commonly used especially in the disposal of community wastes, and it has been applied for several decades in many countries also for disposing of radioactive waste.

Waste burial trenches dug into the soil are usually shallow trench-type excavations and the angle of the slope of their walls varies between 45° - 90°, depending on the type and properties of the soil. The bottom of the trench is usually above the ground 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 clayey soils and under favourable conditions. A simple moisture barrier may be a sufficient lining. The impermeability of the burial trench structure and the stability of radioactive waste material in the repository can 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 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. Canalization of groundwater, rainwater and meltwater is also very important in preventing the infiltration of water. A layer constructed of rubble, reinforced concrete or similar materials to prevent intrusion by humans, animals or plants can be omitted, provided that the burial trench will be kept under control until the radioactivity of the waste 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 disposal method which was used for low- and intermediate-level wastes was shallow ground trenches, into which the waste was loaded also above the ground as mounds. That method allowed more than 10,000 m^3 of waste to be disposed of in one large trench. The bottom and the walls of the trench were lined with clay. The clay layer at the bottom was 1.0 m thick. The waste was covered and levelled with a 0.6 m thick layer of native soil, on top of which was spread a 0.5 m thick clay layer. A 1.0 m thick layer of native soil was applied as the erosion barrier. Each trench was surrounded by a moat and equipped with a sampling well. The disposal site was fenced off and illuminated (IAEA 1992).

2.4.2.3 Surface mounds

Disposing waste in surface mounds is as common method as shallow ground land disposal. Both methods can be used simultaneously or in combination, as at the Centre de la Manche disposal site in France (IAEA 1985). Disposing waste in surface mounds may moreover be a safer alternative in an area which does not adapt well to waste burial, for example, because of a high water table.

In principle, the structure of a surface mound is similar to that of a waste burial trench, as it also uses buffer and intrusion barriers and water canalization layers. Active wastes disposed of in surface mounds often require a fairly heavy shielding against wind, surface water, rain and other types of 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 at right angles to the horizontal. Also the bottom should be bevelled towards the edges so that no moisture would gather under the waste mound. There are several alternatives for lining (i.e. 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 with a similar structure have been used for disposing the mill tailing wastes of uranium. In the United States, the mill tailings of uranium and other cleanup 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, USDOE 1984). The external dimensions of the 7 m high mound are 671 m in length and 366 m in width. The topsoil was removed from the site and used as a covering material, and therefore part of the bottom of the mound is a couple of meters under ground. No lining was used on the bottom. 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. This kind of system may also be used for disposing large amounts of waste in an area contaminated by a nuclear accident. As nuclear waste mainly contains ^{90}Sr and ^{137}Cs, the capping of the mound may be lighter than that used when disposing uranium mill tailings where the predominant radionuclide is the long-lived ^{226}Ra (1600 a).

2.4.2.4 Natural valleys and basins

Valleys and natural basins with a suitable geological structure have been used for disposing of especially uranium mill tailings. Basins and valleys can also be used in emergencies where large amounts of solid low-active cleanup wastes 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 bottoms 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 up basin can be covered with regular 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 from 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-active waste soil, mostly uranium mill tailings, containing, in addition to radium, also arsenic, were buried in a valley in the Chalk River area, Canada. Together with the waste soil cleanup wastes from the Port Hope area were also buried. This waste consisted of concrete flags, blocks, logs and tree roots. In the east, the valley borders 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 whose thickness was 0.3 m. On top of that layer was spread a 0.7 m sandy layer to prevent the erosion of the clay layer and the intrusion of roots into it. 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).

2.4.2.5 Old mines and pits

Great amounts of low-active solid cleanup waste can also be emplaced in closed down underground mines or open pits, if they are situated relatively near. 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 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 present in the bedrock and through fractures in the walls caused by mining. The migration of radionuclides can also occur through these cracks. Rough walls may moreover hamper the installation of liners and barriers, and special techniques may be necessary for installing and compacting the wastes. Old mines and pits usually require careful and time-consuming geological and hydrogeological studies before they can be used for disposing of radioactive wastes. In Germany, low- and intermediate-level wastes have been disposed in the old salt mines in Asse and Morsleben.

2.4.2.6 Bedrock caverns

Disposing of low- and intermediate-level waste into galleries and caverns in the bedrock has been studied, and plans to implement this have been drawn up in many countries. Final diposal 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 operational for a few years: at Forsmark in Sweden and at Olkiluoto in Finland. The Forsmark repository is 60 m below the sea bed and the top of the Olkiluoto repository 60 m below ground level.

Waste from emergency situations, which may have been concentrated before solidification, such as washing fluids, sludges, or ashes from incineration, can be disposed of in bedrock, provided that there is a facility available. Disposing other types of waste, such as contaminated soil, into bedrock will be a much more expensive alternative than disposing them in surface mounds or shallow ground burial.

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3. A SCENARIO OF THE AMOUNTS AND ACTIVITY CONCENTRATIONS OF CLEANUP WASTE

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3.1. INTRODUCTION

The fallout from a major nuclear accident at a nuclear plant may result in a wide-scale contamination of the environment. Cleanup of contaminated areas is necessary if these are populated or cultivated. Most cleanup measures generate large amounts of radioactive waste, which have to be treated and disposed of in a safe manner.

There is a large body of experience in removing and disposing of large amounts of radioactive contaminated material from uranium mill tailings sites. For example, in Salt Lake City, USA, two million tons of waste containing radium was moved by trains to a disposal site at a distance of 140 km (USDOE 1985). In Goiania, Brazil, 3340 m^3 of waste, containing ^{137}Cs from the accidental dismantling of a radiotheraphy unit, was collected and transported to an intermediate storage facility 20 km from the town (Mezrahi 1989).

The cleanup of uranium mill tailings sites were, however, carried out in planned circumstances. When accidents occur the situation is quite different. Decisions on, how the waste obtained in the cleanup work will be treated and disposed of, have to be made in a short time and disposal options may be rather limited. After the Chernobyl accident large amounts of contaminated material, mainly removed soil and trees, were disposed of in shallow ground and surface mound disposal sites. Approximately $4x10^6$ m^3 of waste was placed in about 800 disposal sites.

In order to properly manage the reclamation procedure, the amounts and the activity concentrations of the wastes must be determined in advance. In an actual accident, this will be done through a radiological survey of affected areas and evaluation of the cleanup required. To estimate these figures, scenarios have been worked out in this project. These scenarios are for urban, agricultural and forest environments. All of the more important types of cleanup methods and resulting wastes were considered. In this work no comparative study of the cost-effectiveness or relevancy of the methods was made. This report summarises the results of these scenarios, the data are meant to give basic information for decision making in handling consequences of nuclear accidents. The reader should, however, bear in mind that these results show only an example of the amounts and activity concentrations, which may not be applied to other cases. More important is the methodology developed to estimate these figures.

3.2. CONTAMINATION LEVELS AND AREAS OF CONTAMINATED LANDS - MODEL ACCIDENT

In order to calculate the amounts and activity concentrations of cleanup wastes a model accident of Lahtinen et al was used (Lahtinen 1988, Lahtinen 1991). This model simulates a worst case accidents and, therefore, the amounts of the wastes, calculated on the basis of this model, represent the maximum values.

The model accident simulates a major accident at a 700 MW BWR reactor, in which 10 % of the fission products and 1 % of the transuranics are released into the envinronment. The height of the plume is 100 m and the roughness parameter 50 cm. Three areas with different contamination levels for both wet and dry deposition are given in Table 1 below. In the area 1 the values for the contamination levels represent minimum values. For the areas 2 and 3 the mean contamination levels in forest areas was estimated to 20 % of the maximum values. The mean values for urban and agricultural areas were derived from the mean values for forest area taking into account typical differences in deposition velocities, which are higher in forests than in the other areas.

Table 1. Contamination levels and areas of contaminated land after a major model accident at a 700 MW BWR reactor.

a) Dry deposition

Area 1.	1.8 km^2			
^{137}Cs	> 100 MBq/m^2			
^{90}Sr	> 77 MBq/m^2			
^{239}Pu	> 22 kBq/m^2			
Area 2.	32 km^2			
		estimated mean values		
		forest	urban	agricultural
^{137}Cs	10-100 MBq/m^2	20 MBq/m^2	4	8
^{90}Sr	7.7-77 MBq/m^2	15 MBq/m^2	3	6
^{239}Pu	2.2-22 kBq/m^2	5 kBq/m^2	1	2
Area 3.	2300 km^2			
^{137}Cs	1-10 MBq/m^2	2 MBq/m^2	0.4	0.8
^{90}Sr	0.8-7.7 MBq/m^2	1.5 MBq/m^2	0.3	0.6
^{239}Pu	0.2-2.2 kBq/m^2	0.5 kBq/m^2	0.1	0.2

b) Wet deposition

Area 1.	21 km^2
Area 2.	403 km^2
Area 3.	2210 km^2

Contamination levels as in the case of dry deposition.

3.2. METHODS

For the calculation of the amounts and activity concentrations of cleanup wastes the model accident described above was used.

3.2.1. Calculation of the amounts of cleanup waste

The amounts of cleanup wastes for the three different environments considered were calculated in three ways:

a) Amounts of wastes per square metre (urban areas) or per hectare (agricultural and forest areas) of a specific surface treated with a specific cleanup methods. For example, in icultural areas removal of upper 5 cm layer of soil from fields creates 600 m^3 of solid waste per hectar, including a 20 % increase in volume due to handling.

b) Amounts of waste per square kilometre of contaminated area. To calculate the amount of different cleanup wastes per square kilometre the amounts of waste per unit area (m^2 or hectar) were multiplied by a typical proportion of the surface of an area considered. For example, if fields form 7 % of the total area, the amount of solid waste generated in the removal of soil is: 600 m^3/ha x 100 ha/km^2 x 0.07 = 4,200 m^3/km^2.

c) Total amounts of cleanup wastes. Total amounts of waste were calculated by multiplying the amounts of waste created per square kilometer by the contaminated areas obtained from the model accident. For example, in the case of dry deposition the area with a ^{137}Cs contamination higher than 1 MBq/m^2 is 2,300 km^2. If all the fields in that area are cleaned up by removing the 5 cm upper soil layer, the total amount of waste is $8x10^6$ m^3. For urban areas this procedure is not relevant and, therefore, total amounts of waste were calculated for a typical large city, using Copenhagen as an example.

There are several factors that affect the amounts of cleanup waste. In this work the following, important factors were considered:

- Time between the deposition and the cleanup. After the deposition, rain and weathering move the contamination from upper to lower surfaces. For example, in grassed areas contamination from dry deposition is mainly retained by the grass, but the subsequent rains and the rotting of the grass during the following winter moves most of the contamination into the underlying soil. Therefore, cutting the grass, if not done very soon after the deposition, is an unreasonable cleanup measure and instead of grass the waste must be obtained in the form of soil and turf. In the scenarios two subscenarios for the amounts of waste were considered: 1) cleanup immediately or shortly after deposition and 2) cleanup two or three years after deposition.

- Level of the cleanup work. The level of the cleanup has a great effect on the amounts of the waste. For example, if in grassed areas only the removal of grass in required the amounts of waste is considerably lower than if the removal of a soil layer is also necessary. The level of the cleanup work has to be decided on the basis of a cost-benefit analysis and a justification-optimisation analysis.

- Season, in which the deposition takes place. Distribution of contamination is highly dependent on the season, in which the deposition occurs. There are three main classes of seasons, affecting the forms of waste obtained in the cleanup work immediately after the deposition: 1) winter 2) growing season and 3) nongrowing season without snow. For example, in rural areas the main types of waste for these three season are: 1) snow and ice, 2) vegetation and 3) soil and litter, respectively. This classification may not be as valid for Denmark and Southern Sweden as it is for the other parts of the Nordic Countries, where the winter is longer and colder. Winter time cleanup was, however, not dealt with in this study.

3.2.2. Calculation of the activity concentrations of cleanup waste

The calculation of the activity concentrations of cleanup waste is based on the following data:

a) Contamination levels.
Contamination levels were obtained from the model accident.

b) Distribution of contamination.
Distribution of contamination on different surfaces were obtained from the literature and from experimental measurements.

c) Decontamination factors.
Decontamination factors gained by using different cleanup methods were obtained from the literature and from experimental measurements.

d) Amounts of cleanup wastes.

In the following an example is given, on how the activity concentrations were calculated. Let us consider dry deposition and a contamination level of 4 MBq/m^2 in urban areas. Let us assume, that 90 % of the contamination in grassed areas is retained in the grass. If the grass is cut soon after the deposition, 65 % of the contamination in the grass is removed. The amount of removed grass is 0.1 kg/m^2. The activity concentration of the waste, removed grass, will then be:

$$\frac{4\ \mathrm{MBq/m^2}\ \times\ 0.9\ \times\ 0.65}{0.1\ \mathrm{kg/m^2}} = 24\ \mathrm{MBq/kg} = 24\ \mathrm{GBq/t}$$

3.4. AMOUNTS AND ACTIVITY CONCENTRATIONS OF CLEANUP WASTES FROM URBAN AREAS

Most cleanup measures in urban areas result in the formation of large amounts of radioactive wastes (Tables 2 and 3). The most voluminous of these are removed soil and asphalt, as well as the effluents from firehosing streets and walls. The activity concentration, however, is highest in removed grass and in the dust from road sweepers.

Table 2. Amounts of cleanup wastes from urban areas per square metre.

Cleanup measure	Type of waste	Amount of waste kg/m^2	l/m^2
Soil removal (10 cm)	Soil, turf, vegetation	200	
Grass cutting	Grass	0.1	
Road planing	Asphalt	20	
Firehosing	Water, dust	0.05-0.2	200
NH_4NO_3+Hosing	Water, dust		200
Sandblasting	Water, dust	5	10
Sweeping roads	Dust	0.05-0.2	
Vacuum sweeping	Dust	0.05-0.2	
Indoor cleaning	Dust etc.	0.05	
Roof replacement	Tiles, sheet metal etc.	20-100	
Cutting trees	Trunks, branches etc.	10	
Ploughing and digging soil		0	0

Table 3. Amounts and activity concentrations of cleanup wastes per square kilometre in a typical city area. Deposition 4 MBq/m^2. Cleanup work one month after the deposition.

Cleanup method	Amount (t/km^2)	(m^3/km^2)	^{137}Cs concentr.
Dry deposition:			
Soil removal (10 cm)	50.000	30.000	16 MBq/t
Grass cutting	25	60	32000 MBq/t
Road planing	1000	6.000	1000 MBq/t
Firehosing roads	40 (s)	20	6400 MBq/t
	50.000 (l)	50.000	0.04 MBq/m^3
Firehosing+NH_4NO_3 walls	40.000	40.000	4 MBq/m^3
Sandblasting walls	2.000 (s)	1.000	64 MBq/t
	4.000 (l)	4.000	2 MBq/m^3
Road sweeping	50	50	8000 MBq/t
Wet deposition:			
Soil removal (10 cm)			16 MBq/t
Grass cutting			6400 MBq/t
Road planing			1300 MBq/t
Firehosing roads (s)			8000 MBq/t
(l)			0.04 MBq/m^3
Firehosing+NH_4NO_3 walls (l)			0.4 MBq/m^3
Sandblasting walls (s)			6.4 MBq/t
(l)			0.2 MBq/m^3
Road sweeping			10000 MBq/t

Amounts of waste as in the case of dry deposition.
s = solid, l = liquid.

Table 4. Activity concentrations of cleanup wastes per square kilometre in a typical city area. Deposition 4 MBq/m^2. Cleanup work three years after the deposition. Amounts of waste are the same as in the Table 3.

Cleanup method	^{137}Cs concentration
Dry deposition:	
Soil removal	16 MBq/t
Grass cutting	0.004 MBq/t
Road planing	100 MBq/t
Firehosing roads	4 MBq/t (s)
	0.00004 MBq/m^3 (l)
Firehosing+NH_4NO_3 walls	2 MBq/m^3 (l)
Sandblasting walls	32 MBq/t (s)
	1.2 MBq/m^3 (l)
Road sweeping	4 MBq/t
Wet deposition:	
Soil removal	16 MBq/t
Grass cutting	0.004 MBq/t
Road planing	120 MBq/t
Firehosing roads	4 MBq/t (s)
	0.00004 MBq/m^3 (l)
Firehosing+NH_4NO_3 walls	0.2 MBq/m^3 (l)
Sandblasting walls	0.8 MBq/t (s)
	0.02 MBq/m^3 (l)
Road sweeping	4 MBq/t

s = solid, l = liquid.

Table 5 shows the amounts of cleanup wastes calculated for an area of 250 km^2 of a city with the characteristics of Copenhagen and its closest suburbs. 15 % of the area considered are roads and other paved areas, 45 % large green areas, of which one-fifth is forest land, 11 % is covered with buildings and 29 % is gardens and other small green areas. The waste amounts in Table 5 represent maximum values, because it is assumed that all the surfaces are treated with the cleanup methods mentioned. It is evident that these amounts cannot be summed up to get the total amount of all cleanup wastes, because some of the cleanup methods dealt with are optional. It is also evident, that if only those cleanup measures with the highest priorities (Roed 1990, Brown 1991) are taken, the amounts of waste will be radically reduced from those presented in Table 5.

Table 5. Total amounts of wastes for different cleanup methods from an area of 250 km^2 of a city with the characteristics Copenhagen.

Cleanup method:		Amount t	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 is one important type of waste not dealt with in Tables 2-5: sludges from municipal water treatment plants. After the Chernobyl accident in June 1986 the concentration of ^{137}Cs in the sludge of a sewage treatment plant in Helsinki was about 5 kBq/kg on a dry weight basis (Puhakainen 1987). Most of this activity is assumed to have originated from aluminium sulphate sludge from a drinking water treatment plant. The deposition in Helsinki and in the area, where the surface drinking water originates, was approximately one hundred to one thousand times lower than the highest deposition of 4 MBq/m^2 considered in the scenario for urban areas. Therefore, the ^{137}Cs activity concentrations of the sewage sludge may be roughly estimated to be 500-5000 MBq/t and 50-500 MBq/t for depositions of 4 and 0.4 MBq/m^2, respectively. The water treatment plants in Helsinki produce some 230 m^3 of sludge a day. This estimation gives an indication, that a considerable proportion of the contamination finds its way to sewage sludge. It is also possible that firehosing may increase the activity of the sludges, if at least a fraction of the effluents are directed into sewarage. Usually rain waters are, however, directed into lakes, rivers and seas, and therefore, also cleanup effluents go there. In general, firehosing cannot be considered as a final cleanup measure, because it only relocates the contamination.

3.5. AMOUNTS AND ACTIVITY CONCENTRATIONS OF CLEANUP WASTES FROM FOREST AREAS

Because there is no data available on the cleanup methods and the resulting decontamination factors, in this study it was assumed that all types of biomass, as well as soil, would be removed when cleaning up forests (Table 6). Because it was also assumed, that all the activity would be removed with the removal of vegetation, the activity concentrations given below represent maximum values. Distribution of contamination on different surfaces in forests are compiled in Appendix 2.

Table 6. Cleanup methods and resulting waste forms in forest areas.

Cleanup method	Waste
Removal of trees	Crown Bark if barking is used Stems with or without bark
Removal of undervegetation (manually)	Undervegetation
Removal of undervegetation litter, humus and soil (by machine)	Undervegetation Litter and humus Stumps and roots Soil

The removal of trees is effective in removing contamination only immediately or soon after the deposition. This is especially true in the summer time, when there are leaves on the deciduous trees. The second procedure, manual removal of undervegetation can be applied to only very limited areas and it would be effective in removing contamination in an intermediate time range after deposition. The third method, mechanical removal of undervegetation together with litter, humus, stumps, roots and upper soil layer is effective years after the deposition. Cleanup of bog areas is considered to be impossible and is not dealt with in this study.

The cleanup of forest areas will create enormous amounts of solid radioactive wastes. Table 7 shows the biomasses of undervegetation, litter, humus, stumps and roots in pine, spruce and deciduous forests in both northern and southern parts of the Nordic Countries. Table 8 shows the corresponding data for trees.

Table 7. Biomasses of undervegetation, litter, humus, stumps and roots in pine, spruce and deciduous forests.

Forest	Type of biomass	Mass (t/ha)	Stacked volume (m^3/ha)
pine	undervegetation	0.7	4
	litter and humus	95	310
	stumps and roots		78
spruce	undervegetation	0.8	4
	litter and humus	120	390
	stumps and roots		130
deciduous	undervegetation	0.9	4
	litter and humus	120	400
	stumps and roots		90

Table 8. Biomasses in different types of forest trees as mature stand in southern and northern parts of the Nordic Countries.

Tree		Type of biomass	Mass (t/ha)	Stacked volume (m^3/ha)
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

Removal of upper 5 cm or 10 cm layers of soil will create 600 or 1200 m^3 of waste per hectare, respectively. Probably the latter value is more reasonable for forest areas.

Amounts of waste and their activity concentration have been calculated for mature stand forests for various parts of the Nordic Countries. Table 9 shows one of these cases, while other cases are shown in Appendix 2.

Table 9. Average amounts and ^{137}Cs concentrations of cleanup 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) in Southern Finland. Dry fallout >100 MBq/m^2. Cleanup work immediately after the fallout.

Contaminated pine forest 0.77 km^2

	Activity TBq	Stacked vol. m^3	Activity MBq/m^3
Needles	27	1100	25000
Other crown	27	3600	7500
Bark	0.4	1700	200
Underveget.	23	300	77000
Total	77	6700	

(stems without bark 8900 m^3)

Table 9 continues

	Activity TBq	Stacked volume m^3	Activity MBq/m^3
Contaminated spruce forest 0.5 km^2			
Needles	20	2800	7000
Other crown	20	4600	4300
Bark	0.3	1000	200
Underveget.	10	200	50000
Total	50	9500	
(stems without bark: 9800 m^3)			
Contaminated deciduous forest 0.11 km^2			
Leaves	1.7	170	10000
Other crown	1.6	270	5900
Bark	0.02	290	70
Underveget.	7.7	48	160000
Total	11	780	
(stems without bark: 1500 m^3)			

All volumes together: 17,000 stacked m^3 of waste (+ stems without contamination 20,000 m^3). All Cs-137 activity together: 140 TBq.

3.6. AMOUNTS AND ACTIVITY CONCENTRATIONS OF CLEANUP WASTES FROM AGRICULTURAL AREAS

In agricultural areas only two types of cleanup wastes are important: removed soil from fields and removed vegetation, the former giving a higher volume of waste (Table 10). Most of the activity will be in the soil. In wet deposition approximately 20 % of the activity is retained in vegetation and in dry deposition 40 %. If the cleanup work is done immediately or soon after the deposition, removed vegetation contains an essential part of the activity, but later on practically all the activity will be in the soil. Because radionuclides are retained in the upper few centimeter layer of soil, it would be advantageous to remove only this in order to reduce the volume of waste.

Table 10. Volumes of removed soil per hectar and average volumes per square kilometre 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/ha	360 m^3	600 m^3	1200 m^3
Volume/km^2			
North Norway (0.73 %)	260	440	870
Central Norway (3.9 %)	1400	2300	4400
West Norway (3.0 %)	1100	1800	3600
South Norway (1.7 %)	610	1000	2000
East Norway (5.8 %)	2100	3500	7000

- the percentages in parantheses are the proportions of cultivated areas.

The figures for the removed soil in Table 10 and 11-13 have been calculated by taking into account the proportions of cultivated areas in different parts of Norway, where the percentage of agricultural land is lowest within Nordic Countries. In other Nordic Countries these percentages are considerably higher (Finland 9 %, Sweden 12 %, Dennmark 69 %), and correspondingly the figures for removed soil would be much higher than those presented in Tables 10 and 11-13.

The volumes presented in Table 10 are for pure soil, and they represent the amounts of cleanup waste in cases where there is neither vegetation nor snow cover on the fields. If all the fields from the contaminated areas described in the model accident were cleaned up by removing the upper soil layer, the amounts of waste would be enormous. Tables 11-13 give the total amounts for the three wet deposition areas from the model accident.

Table 11. Total amounts of removed soil from an area of 21 km^2 with a ^{137}Cs contamination level of >100 MBq/m^2 in different parts of Norway. (For the dry deposition scenario the values should be multiplied by a factor of 0.09).

	Thickness of soil layer removed		
	3 cm	5 cm	10 cm
North Norway	5,400 m^3	9,100 m^3	18,100 m^3
Central Norway	29,100	48,400	92,000
West Norway	22,400	37,300	74,500
South Norway	12,700	21,100	42,200
East Norway	43,200	72,000	144,100

Table 12. Total amounts of removed soil from an area of 403 km^2 with a ^{137}Cs contamination level of 8 MBq/m^2 in different parts of Norway. (For the dry deposition scenario the values should be multiplied by a factor of 0.08).

	Thickness of soil layer removed		
	3 cm	5 cm	10 cm
North Norway	106,000 m^3	176,500 m^3	353,000 m^3
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

Table 13. Total amounts of removed soil from an area of 2,210 km^2 with a ^{137}Cs contamination level of 0.8 MBq/m^2 in different parts of Norway. (For the dry deposition scenario the values are approximately identical).

	Thickness of soil layer removed		
	3 cm	5 cm	10 cm
North Norway	580,800 m^3	968,000 m^3	1,936,000 m^3
Central Norway	3,103,000	5,171,400	10,343,000
West Norway	2,387,000	3,980,000	7,960,000
South Norway	1,352,000	2,254,200	4,508,400
East Norway	4,614,500	7,690,800	15,381,600

The activity concentrations of removed soil are fairly high (Table 14).

Table 14. Concentration of ^{137}Cs in removed soil assuming that 90 % of the activity was removed.

	Thickness of soil layer removed		
	3 cm	5 cm	10 cm
Contamination level (MBq/m^2)			
0.8	20 MBq/m^3	12	6
8	200	120	60
>100	>2500	>1500	>750

If there is no vegetation nor snow on the fields during the deposition, the activity concentrations given above are independent of the type of deposition (dry or wet) and the time elapsed between the fallout and cleanup. If, however, there is vegetation and it is removed shortly after the deposition, the activity concentration of removed soil is reduced. In this study it was assumed that 40 % of the activity is retained in vegetation in dry deposition and 20 % in wet deposition. This means that for immediate cleanup of fields with vegetation the values for the activity concentrations have to be multiplied by factors of 0.6 and 0.8, respectively.

If the deposition takes place in the snowy winter conditions time and the snow is removed, there will be practically no contaminated soil. If, however, the snow melts before the cleanup work, a fraction of the contamination will run off the fields with the melt water. This fraction, which reduces the values in Table 14, is not known.

If the cleanup work after a deposition in the growing season were done soon after the fallout, removal of vegetation would be reasonable. Compared to the amounts of removed soil, the amounts of removed vegetation are much lower (Table 15). Correspondingly the activity concentrations of removed vegetation is fairly high (Table 16).

Table 15. Amounts of vegetation per hectare in different parts of Norway.

	Fresh weight (t/ha)		Fresh volume (m^3/ha)	
	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

Table 16. 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	Activity concentration			
MBq/m^2	MBq/t		MBq/m^3	
	cereals	grass	cereals	grass
0.8	120	180	48	69
8	1200	1800	480	690
>100	>15000	>22000	>5960	>8600

If contaminated domestic animals are slaughtered, 20-150 kg of waste are created per carcass.

3.7. SUMMARY

3.7.1. Amounts of cleanup wastes

Vast amount of radioactive waste will be generated in the cleanup of large contaminated areas. In all areas, urban, agricultural and forest, 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.

3.7.1. Activity concentrations of cleanup wastes

Activity concentration of cleanup waste will vary over a wide range. In urban areas, the highest activity concentrations will be found in cut grass and solid matter removed from walls and paved surfaces by sweeping and firehosing soon after deposition. Years after deposition, soil and asphalt will have the highest concentrations of activity. In forests and agricultural areas, the activity concentration of vegetation will be high soon after deposition, but years later the activity will be almost entirely in the soil and litter. The activity concentrations of different types of cleanup wastes from the three environments considered in the scenarios are compiled in Appendix 5.

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4. EXPERIMENTAL STUDIES ON THE REMOVAL AND DISPOSAL OF RADIOACTIVE CONTAMINATED SOIL FROM AGRICULTURAL AREAS

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This chapter reports the results of practical experiments in Norwegian agricultural areas, concerned with the decontamination (by scraping of soil layers) and disposal of the waste produced. Six different machines, commonly available in agricultural production, have been tested to establish their suitability for the removal of contaminated surface soil from agricultural areas after a severe nuclear accident. Three soil types have been tested, clayey, sandy and peaty soils, and the experimental fields were subjected to various pre-treatments, i.e. herbicides, cutting, raking, and stubble burning. In each case, the volume of waste produced for disposal and the time needed for the operation have been determined. The road-grader seems to be most useful on sand and clay, with a waste volume of 300-400 m^3/ha and a time consumption of ca. 100 min/ha. On peaty soils, the use of a road grader or bulldozer has proved to be very appropriate on frozen ground, after the thawing of the upper few centimetres of the surface. Otherwise the volumes produced are unnecessarily large.

Construction of the disposal sites is both time-consuming and space-demanding. Under optimal conditions (a road grader on dry, sandy soil), the construction of buried waste disposal ranks leads to a decrease in the production area of between 10 and 15%. Disposal sites have been constructed on the three soil types, and the transfer of ^{134}Cs tracer from soil into seepwater and uptake into plants have been followed. During the first year the amounts of ^{134}Cs in drainage water was less than 0.01 % and the concentration was decreasing with time.

The studies have been undertaken in collaboration with the Center for Soil and Environmental Research. Svanhovd Environmental Research Station and Tana Agricultural School have contributed to the practical experiments carried out in East Finnmark. In addition to NKS, the Ministry of Agriculture in Norway has contributed to the funding of the project.

4.1. INTRODUCTION

In the event of a nuclear accident, various decontamination measures could be employed depending on the character and nature of contaminated areas. In order to enable the implementation of prompt and efficient countermeasures, it is necessary to identify the most practical, cost-effective, and dose-reducing methods for the removal of radioactivity and the disposal of waste.

The feasibility of decontaminating agricultural land by the removal (by scraping) and disposal (by burying) of the top, surface layer of soil will depend on local conditions (e.g. equipment available, climatic conditions, agricultural systems). In order to obtain realistic estimates on the efficiency of potential scraping equipment in Norway, waste volumes produced, practical disposal methods and the time consumption involved, experiments have been carried out in East Finmark (Pasvik and Tana), Northern Norway. This site is representative of a potential risk area due to its proximity to the Kola reactors and Pasvik and Tana are the major arable productive areas in the region. Studies were carried out late in the growing season (August 1992) on sandy, clayey and peaty soils. A similar scraping experiment had previously been carried out at Selskapet for the Norges Vels farm in Hellerudsletta (Salbu et.al., 1992).

4.2. REMOVAL OF CONTAMINATED CULTIVATED SOIL

4.2.1 Background

4.2.1.1 Radionuclide retention in the surface layer

Both the composition and the physico-chemical forms of radionuclides in fallout will be dependent upon the accidental scenario, the distance from the accident site, and climatic conditions (wind direction, rainfall). In areas with a high deposition of activity, a significant fraction of the radionuclides is believed to be present in particulate forms. All the exposed surfaces will be covered with a thin layer of activity, with an inhomogenous distribution giving "hot spots". The retention of radionuclides will depend on species deposited, as well as on the characteristics, composition and structure of the surface layers, in particular, on the season (i.e. whether radionuclides are deposited onto snow cover, vegetation cover, or ploughed soil).

For deposition onto snow, the transport of radionuclides will be dependent on snowmelt. Removal of the snow before melting should be an effective countermeasure, as the majority of the activity (70-100 %) could be removed. For soils covered with dense vegetation, most of the radionuclides will be retained by the vegetation, however, washout due to heavy rainfall can transport the radioactivity from the vegetation to the surface soil. Removal of the vegetation prior to rainfall should be an effective countermeasure as between 60-90 % of the activity could be removed. After deposition to soils, the majority of radionuclides will be associated with soil components and vertical transport down the soil profile is, in general, rather slow. In porous soil, radionuclides associated with particles may be transported down the soil with rainfall, through capillary transport, cracks and poreholes. It should, therefore, be beneficial to decontaminate an area by the removal of the vegetation layer and a relatively thin layer of soil soon after deposition, i.e. after the decay of short-lived radionuclides (^{131}I) and before heavy rainfall. Within 7 half-lives of ^{131}I, i.e. in 56 days, the ^{131}I decreases to $<$ 1% of the initial activity.

It is assumed that the removal of the surface layer in agricultural areas will involve the use of harvesting (to remove vegetation), scraping (to remove to surface soil) or ploughing machinary (deep ploughing of the contaminated surface layer). Apart from deep ploughing, the countermeasures will produce radioactive waste. The amount produced will be dependant on the equipment used.

After the Chernobyl accident, removal of the surface layer (vegetation, soil, asphalt) was carried out within the 30 km zone around the reactor. The use of bulldozers produced large quantities of waste. Removed soil and vegetation were disposed of in surface mounds which were covered with polyethylene sheet and uncontaminated topsoil. Drainage systems were also constructed. However, wind transport resulted in resuspension and relocation of the radioactivity. Thus when evaluating removal techniques it is essential to ensure that adequate disposal methods can be applied.

4.2.1.2 Previous testing of machinery

After the atmospheric nuclear weapons' testing in the 1960s, a number of experiments were carried out to test the suitability of different equipment for removal of contaminated surface soil. In a extensive study by James and Menzel (1973), various types of scraping machines, sweeping machines, asphalt laying and removal, deep-ploughing, etc. were tested, with the following results:

- No one method could be considered "best" under all circumstances.
- By either scraping a 5 cm thick layer of soil, or using a road-sweeping machine, about 90% of the added contaminants could be removed. Results from the other methods were inconclusive.
- The measures should be carried out before rainfall, in order to reduce the potential for transfer of contaminants to other parts of the ecosystem.
- Removal of contaminated vegetation alone was not sufficiently effective.
- Under optimal conditions a road scraper could decontaminate 0.1 ha (1000 m^2) in 20 minutes, including simple disposal of the waste produced.

4.2.2 Methods

The equipment tested in East Finmark should represent easily availabile vehicles with a high expected effectivity. For these reasons, road-scraping machines: self-powered road scraper, bulldozer, tractor-mounted road scraper, tractor with rake and front-mounted shovel, and front-loader, were tested. Photographs of these vehicles are shown in the appendix at page 60. The experiments were carried out
in the last week of August 1992. As 1992 had rather heavy rainfall, the soil was wetter than usual for the time of year. Due to the wet conditions, all the vehicles were not tested at each locality.

The scraped fields were ca. 30 m long (Figure 2.1), with a relatively flat terrain (0-2 % slope). All the drivers practised the scraping techniques before performing the experiments. In general, two scraping experiments were carried out on each soil type: one on untreated pasture and one on pasture sprayed with herbicide. On the sprayed areas, glyfosat was applied about three weeks before the experiment.

Four parameters were used to assess the efficiency of the method: <u>percentage removal</u>, <u>mean scraping depth</u>, <u>increase in soil volume</u>, and <u>volume of waste soil</u> produced.

The <u>percentage removal</u> was estimated from the remnants of vegetation cover and untouched surface soil. Using surveying equipment, the height of the soil surface was measured between the machines' wheel tracks, at 1 m lengthways intervals, before and after scraping. Hence, the mean scraping depth was calculated. However, since the surface height measurements were taken in the centre of the fields, uneven scraping of the outer edges of the fields will not be represented in the figures below. In some figures the scraping effect may appear to be better or worse than the estimated percentage of removal. In order to determine the increase in soil volume due to scraping, soil samples from the experimental field and the mounds of removed soil were collected by insertion of pF cylinders. The increase in soil volume was determined by comparing the respective weights of soil removed. The total volume of waste soil produced is given in units of cubic metres per hectare (m^3/ha).

In order to calibrate the approximate percentage of removal (as estimated in the test runs), ^{134}Cs tracer was added to a small plot before scraping.

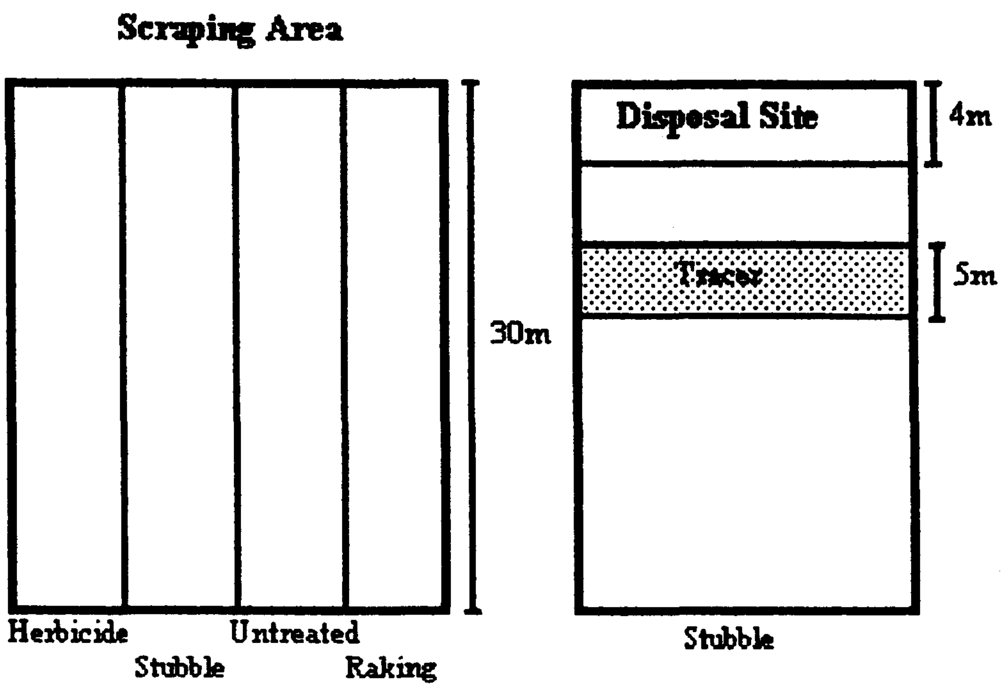

Figure 2.1. The experimental field. Scraping area (left): 30mx20m. Disposal site area (right): 30mx10m.

4.2.3 Results

The results of the scraping experiments are summarized in Table 2.1. Where relevant, the individual results from the various pre-treatment of the plots (sprayed, not sprayed, stubble etc.) have been combined to enable easier comparison of the machine performances on the different soil types. The various experiments and the performance of the different machines are discussed in detail below.

Table 2.1. Production of waste (m^3/ha) on three soil types after using different machines to remove the toplayer of agricultural soil.

Machine	Soil Type	Scraping depth (cm)	Removal (%)	Volume Increase (%)	m^3/ ha
Bulldozer	Clayey soil	20	90	6	2120
Bulldozer	Frozen peaty soil	2-5	90	6	530
Tractor with	Clayey (raked meadow)	10	30	24	1240
road scraper	Clayey (burnt straw)	< 1	50	23	40 ·
"	Clayey (straw)	-	Not	suitable	-
Road scraper	Clayey (burnt straw)	1	80	8	130
"	Clayey (straw)	3	90	10	320
Digger	Clayey soil	5	95	20?	550
Bulldozer	Peaty soil	6	60-80	16	700
Tractor with road scraper	Peat (raked meadow)	3	20	24	370
Front loader	Sandy soil	5	55	15	520
Road scraper	Sandy soil	3	50-90	30?	400
Tractor with shovel	Sandy (raked meadow)	4	50-70	25	500

All experiments were carried out at the Svanhovd environmental centre. The bulldozer, tractor-mounted road scraper and mechanical digger were tested on clayey soils; conditions were too wet to allow the front loader and the road scraper to be tested. Scraping efficiency was assessed on a meadow with both fresh vegetation and sprayed vegetation plots. Previous experiments, at Hellerud in 1991 were carried out on burnt and untreated stubble. For organic soils, the experiments were carried out on marshy ground, using the tractor-mounted road scraper and bulldozer. In addition to the sprayed and unsprayed meadow, an experiment was also undertaken in a barley-field. The crop had been harvested and the stubble height on this field was 5-10 cm. The front-loader, road-scraper and tractor with shovel were tested on sandy soils.

4.2.4 Machine Performance

4.2.4.1 Bulldozer

On clayey soil, the bulldozer was clearly incapable of performing with the precision needed. The slicer was a considerable distance from the driver and out of his line of sight, thus it was difficult to regulate the slicer with the necessary speed. The slicer removed a greater mass of soil than necessary, became quickly overloaded, and it was difficult to drive distances of greater than 15 meters at a time (Figure 2.2). Under drier conditions, the scraping depth could probably be reduced significantly. The bulldozer drove well on the marshy ground despite the high water content (68% of total weight). However, it was rather difficult to remove a thin layer of the soil. As it is shown in Figure 2.3 up to 20 cm of the topsoil was removed. Because of spillage, the mean scraping depth will probably be somewhat deeper that given in Table 2.1. The bulldozer compressed the soil whilst shovelling, thus the volume increase was relatively low.

A specific bulldozing technique has been developed for frozen ground. This was carried out during the spring 1993, after the upper few cm of clay soil have thawed. A relatively thin layer (2-5 cm) could be easily be removed by this powerful machine. However, bulldozing on frozen ground is of course only possible for a short period each year. The effect of the frozen ground bulldozing technique on non-frozen peaty soil is represented in Figure 2.4.

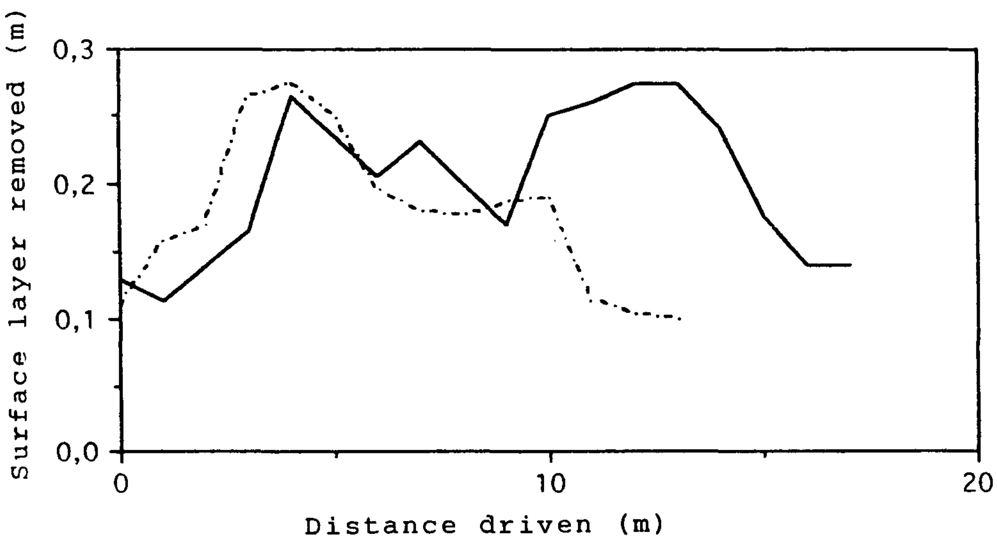

Figure 2.2. Scraping of a sprayed meadow, clayey soil, with a bulldozer, 90% removal.

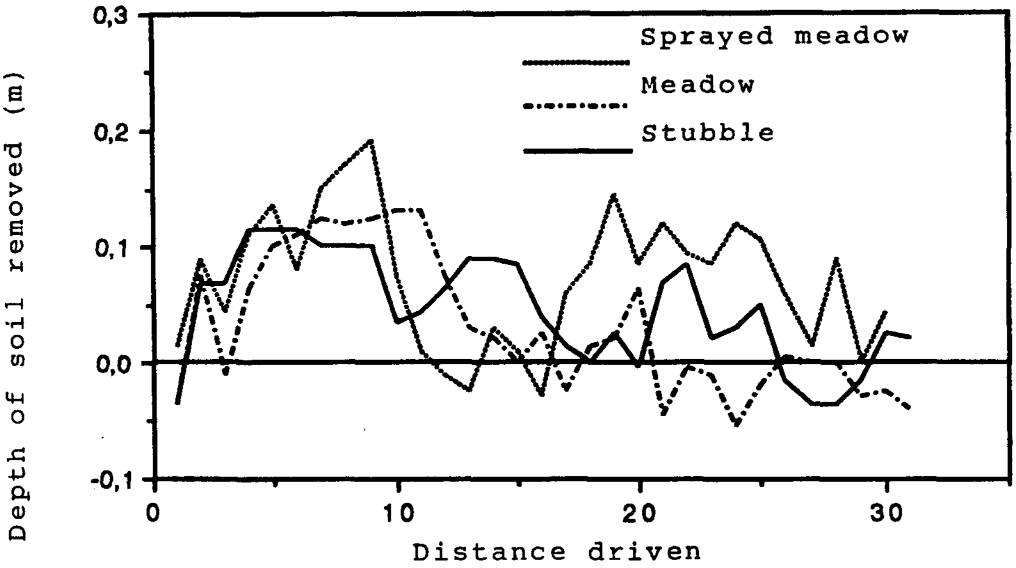

Figure 2.3. Removal of the toplayer of peaty soil with the help of a bulldozer. Negative values represent loss of soil due to spillage.

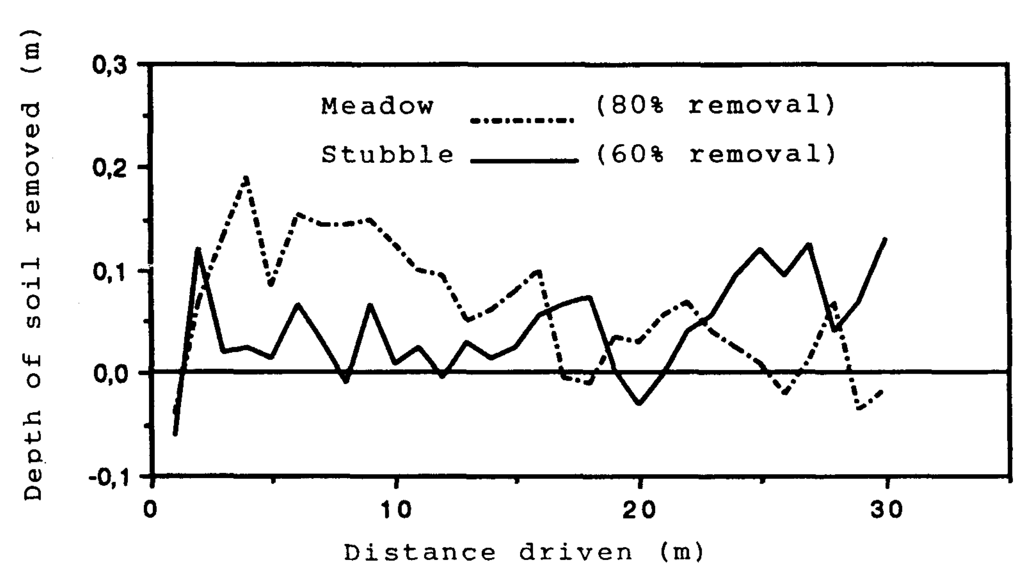

Figure 2.4. Removal of the toplayer of peaty soil with stage-by-stage bulldozer driving (frozen-ground technique). Negative values represent loss of soil due to spillage.

In order to reduce the spillage the bulldozer was driven over the plot twice, first from 15 to 30 meters and thereafter over the whole plot (0-30 m). On the last run (0-30 m), only soil from the first 15 m was scraped away. On average, the scraping depth was 6.5 cm on the meadow and 4.4 cm on the field with stubble. Even though the results after two runs are better than those after one run (Figure 2.4), more soil was removed in total. This technique should first and foremost be carried out on frozen ground. If carried out on non-frozen ground, there is a danger that the decontaminated soil could be recontaminated when the new soil masses pass over the treated areas.

4.2.4.2 Tractor-mounted road scraper

On clayey and peaty soil, the use of a tractor-mounted road scraper seems to be impossible without some kind of pretreatment, as the road scraper tended to glide on top of the grass, above the soil surface. A similar problem was experienced in 1991 when attempting to scrape clayey soil covered with straw stubble. Even after the soil was loosened by raking, the tractor-mounted road scraper was not a suitable machine for removal of contaminated vegetation and topsoil from stubble or vegetation covered meadow. The tractor-mounted road scraper did not have the capacity for removing large volumes of soil. The slicer became quickly overloaded; in the experiment represented in Figures 2.5 the slicer became full three times during the 30 m run. Furthermore, the slicer jumped up leaving behind mounds of soil. Spillage is depicted in Figure 2.6 as negative values. With more thorough raking, it is possible that a smaller soil volume can be removed.

An attempt was made to drive the soil together in ranks by setting the cutter at an angle. This was somewhat more successful, but soil could only be moved sideways a distance of two slicer-length's. Another instrument would have to be used to move the soil masses farther.

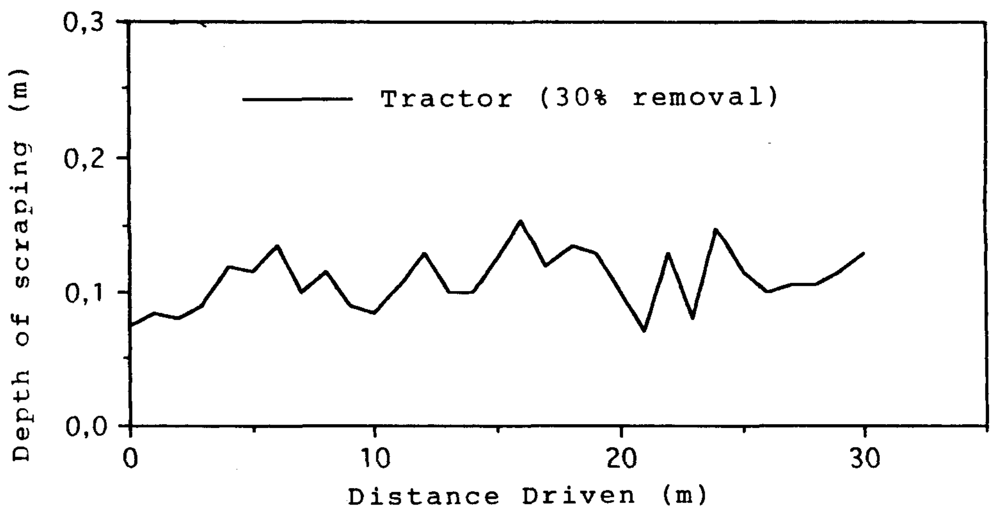

Figure 2.5. Removal of the toplayer of clayey soil with a tractor-mounted road scraper on sprayed meadow. The soil was first loosened by raking.

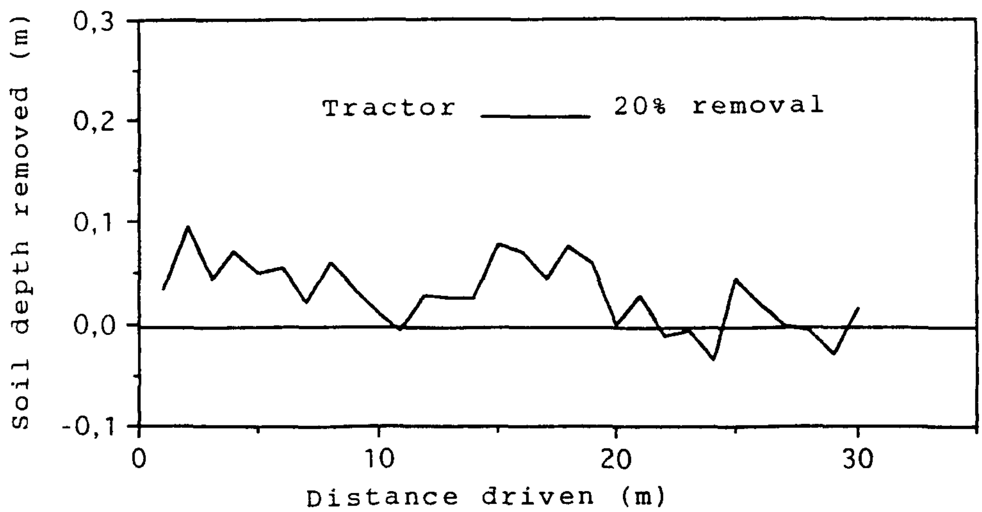

Figure 2.6 Removal of the top layer of peaty soil with a tractor-mounted road scraper on sprayed and raked meadow. Negative values represent loss of soil due to spillage.

4.2.4.3 Self-powered road scraper/road grader.

On clayey soils, the road grader was tested once after straw and stubble burning and once on untreated areas. The experimental site was on a slight slope, and the road scraper had problems driving against the gradient. This was because the tyres were designed for gravel roads, and had difficulty gripping on bare soil. Driving downhill posed no problems. On sandy soils, the results from the road scraper were variable. From Figure 2.7 it can be seen that the runs were easily disrupted. In certain places the soil was replaced by spillage, especially on uneven terrain. The overall impression was that this is an effective machine even though it was sensitive both to the water content of the soil and to irregular terrain.

The road scraper has one rigid slicer at the front and an adjustable slicer under the driver's cab. The driver used the movable slicer but, because this is placed after the front wheels and due to the weight of the vehicle, the front wheel compressed the soil. Because of this, the percentage removal was not as large as it could have been had the front slicer been used. It will be necessary to scrape deeper than 4 cm if full removal is to be achieved. The slicer was placed at a 45° angle, which resulted in a small rank of soil being left behind. The driver suggested that the machine could move these ranks sideways, 3 to 4 times, whilst scraping new soil, before the machine became overloaded and/or spillage became too large.

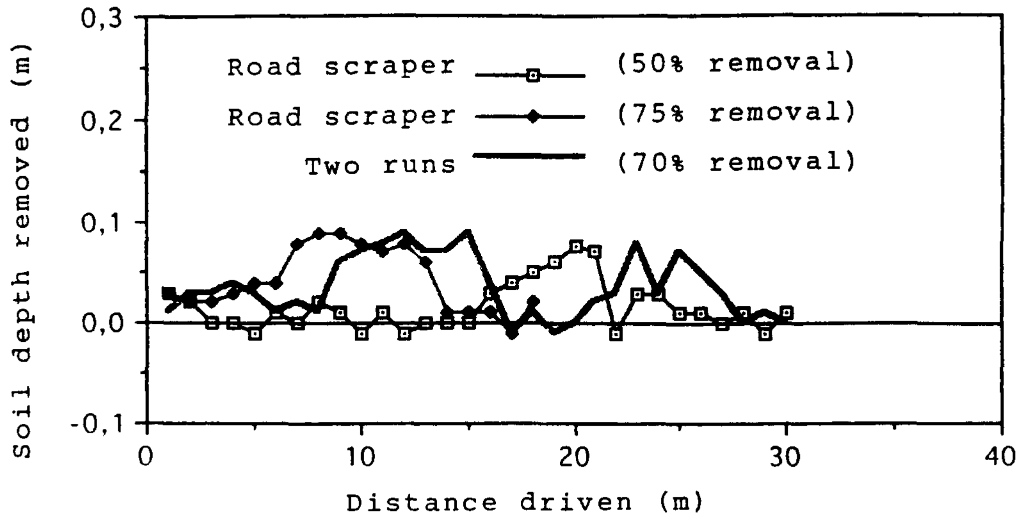

Figure 2.7. Use of a self-powered road scraper (road grader) to remove the surface layer of sandy soil from a meadow. In one case, two runs were made. Negative values represent loss of soil due to spillage.

When constructing the disposal sites (Section 3), the road scraper was used to transfer the soil into the ditches by repeated scrapings at high speed. When the road scraper was driven twice over one scraping field, the results were satisfactory (Figure 2.7).

4.2.4.4 Mechanical digger with a wide shovel.

An experiment was carried out on clayey soils using a belt-driven mechanical digger with a wide shovel of the same type as used by the highways' authorities for ditch clearing. The digger took 15 min to decontaminate a 10 by 10 m field, which is equivalent to 25 hr/ha. An important advantage of the digger is that the same machine can be used to bury the soil in the disposal site, or transfer contaminated soil directly to a lorry, which can save time. It should be noted, however, that the time consumption given above only refers to the soil removal process.

4.2.4.5 Front Loader

The use of a front loader appeared to be ineffective on sandy soils. Even though Figure 2.8 indicates a positive removal, the machine had problems cutting through the dense turf of the meadow. Progress was rather disrupted, the shovel became quickly overloaded and the run had to be interrupted after only 20 meters.

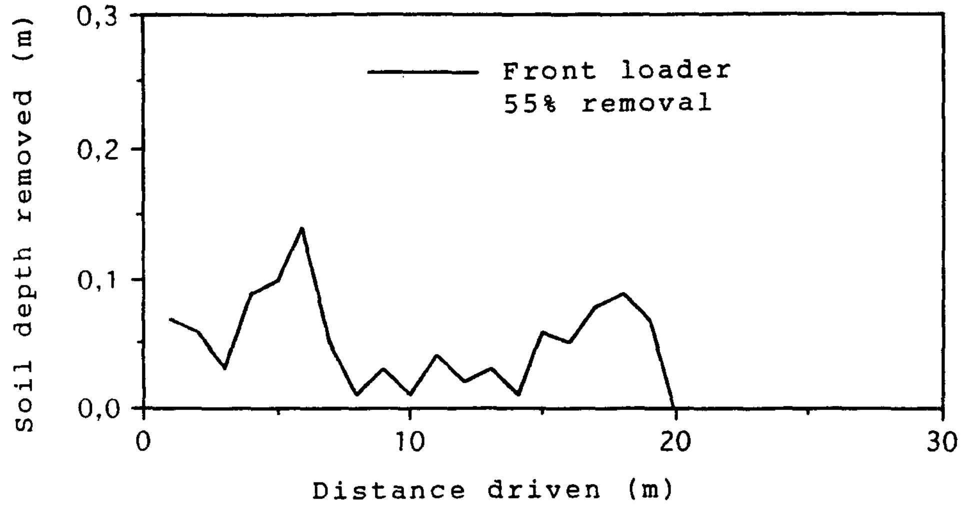

Fig 2.8. The use of a front loader to remove topsoil from sandy soil.

4.2.4.6 Tractor with snow plough

Initially, a road scraper was mounted at the back of the tractor, but it did not manage to cut through the vegetation cover. Figure 2.9 shows the results from an alternative procedure using a front-mounted snowplough after raking, using a tractor of ca. 65 horsepower. The grass mat was first loosened with a rotary hoe, then the snowplough was used to drive the soil together. The best result (75% removal) was achieved with the shovel cutting at the shallowest possible depth, on average 3.4 cm. Broken runs gave worse results. A possible alternative could be to combine the tractor-mounted rake or rotary hoe with the front loader. After first loosening the vegetation cover, the front loader could be used to push the soil together. Better results could be obtained by using a shovel which can tilt sideways, or a shovel with small support wheels, thus preventing the corners from digging down into the ground.

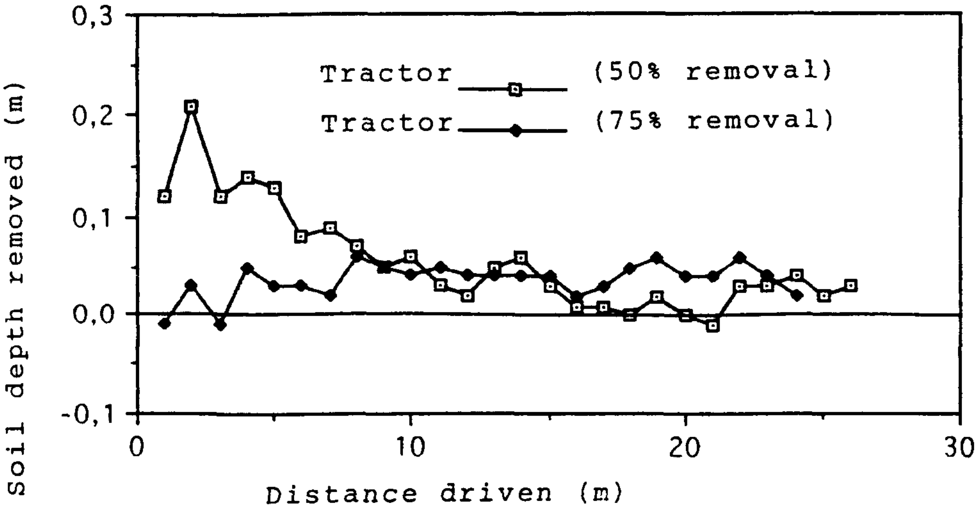

Figure 2.9. Two runs with a tractor on sandy soil. In both cases the grass cover was loosened by raking first, and then the toplayer was pushed together with a snowplough. Negative values represent loss.

4.2.5 Scraping and deep ploughing of an area dosed with radioactive tracer.

^{134}Cs was used as a tracer on a 5mx10m plot by the side of the burial trench (Figure 2.1). This plot was scraped with a bulldozer and a tractor-mounted road scraper. In general, these tests removed more of the radioactivity than estimated in the experiments reported above. This is due to a larger scraping depth, 10-15 cm. The small size of the plot meant that it was possible to remove a large soil mass, without spillage, in a single run. On clayey and peaty soils, a small plot was deep ploughed to 40-50 cm. This removed all ^{134}Cs activity from the surface layer. The same effect was achieved by turning the soil with the digger.

4.2.6 Pretreatments

It is difficult to say whether spraying of the grass cover had any significance. An experiment to rake the stubble was unsuccessful as the plant material was scattered.
On crop fields, straw or stubble can be burnt before scraping, after which the slicer is capable of removing a very thin layer of topsoil. However, it must be noted that stubble burning increases the concentration of airborne radionuclides and resuspension increases the risk of inhalation. Thus, for radiological protection reasons, burning is not recommended. After raking and burning, the soil volume increase was rather large.

4.2.7 Time consumption

Whilst assessing the different scraping measures, the time consumption was estimated from a 30 m run with the relevant machine on relatively flat terrain. The working breadth varied, but all numbers have been re-adjusted to represent the time consumption per ha. In addition to the time consumption for scraping, the time needed for turning, burial in trenches etc., must be taken into account. On the basis of these calculations, the measures have been ranked as follows:

> 100 min/ha = road grader << tractor with snowplough after raking < front loader < bulldozer < tractor-mounted road scraper after raking <<< mechanical digger = 25 hr/ha

This ranking merely indicates the time taken to carry out the measures, and the efficiency and waste volumes produced are not taken into account. Even though large differences in scraping efficiency were observed for the different soil types, there was little distinction between time consumption. It seemed that sandy soil was the easiest to work with, probably because it was drier and coped better with the weight of the machines.

4.2.8 Conclusions

A large variation can be seen in the effect and the volume of soil produced for disposal. On the basis of the experiments a number of general conclusions can be drawn:

i) Transport of large soil masses resulted in spillage. Equipment with the capability for precision work should be chosen.

ii) Under dry conditions, the self-powered road scraper (road grader) should be a valid alternative for sandy and clayey soils. However, the front wheels of the vehichle compress the soil such that either some of the contaminated material is not removed or the soil volume becomes rather large. Because of its weight, it seems unlikely that the road grader can be used on marshy ground.

iii) The tractor-mounted road scraper does not appear to be capable of moving large masses of soil. Only when the soil surface has been suitably conditioned, for example by stubble burning followed by raking, can this machine be used as a countermeasure. Nevertheless, significant amounts of the radioactivity remain behind, and the pretreatment can lead to increased resuspension.

iv) A mechanical digger with a wide shovel removed a large percentage of the surface soil whilst giving relatively small soil volumes, and the machine can also be used to combine scraping and disposal. However, the process is time consuming.

v) Bulldozing on frozen ground covered by a thin layer of thawed soil has proved to be most suitable also on peaty soil. Otherwise the use of a bulldozer produces large soil volumes for disposal.

vi) Special machines designed to remove plants may be suitable, but these machines are not widely available and require dry conditions and pre-treatment of the surface.

vii) These countermeasures could be carried out more easily on sandy soil than on clay or peaty soil.

viii) The volume increase after scraping seems to decrease with the scraping depth.

ix) No significant improvement in scraping performance was observed after spraying the meadow with herbicide.

x) "Practice makes perfect". It should be possible to improve on these results with more practice.

xi) Removal of the contaminated vegetation cover and the upper soil layer is a time consuming process, which often produces large volumes of soil for disposal. Better equipment should be found, constructed or developed.

xii) Under dry donditions the use of road scraper especially on sandy soil shòuld be an appropriate technique. The use of bulldozer on frozen ground covered with a thin layer of thawed soil is, however, most efficient technique for soil removal.

4.3. DISPOSAL OF CONTAMINATED SOIL

The removal of the contaminated vegetation and soil cover produces radioactive waste. The waste can either be driven off-site to a centralised disposal site or it can be disposed of in on-site disposal trenches. It will often be rather labour intensive and costly to remove the waste to an off-site disposal area. The actual volumes of waste have been discussed in Section 2, and can be rather large. A practical alternative is therefore to construct disposal sites and bury the radioactive soil on-site. However, the disposal sites must be constructed to cope with storage for a couple of hundred years. The alternative methods have been previously discussed in Salbu et.al. (1992). One of these was tested during the field-work in East Finmark, i.e. disposal of contaminated soil in buried trenches.

4.3.1 Construction of the experimental disposal facility

4.3.1.1 Methods

When constructing the disposal facility, three questions are of importance:
(i) How do the radionuclides migrate in the disposal trenches?
(ii) How large is the loss of radioactivity from disposal trenches in different soil types?

(iii) To what degree is extra protection against rainfall necessary to prevent wash-out?

Three burial trenches were constructed on clayey, sandy and peaty soil, respectively, in order to estimate the effects of on-site storage. ^{134}Cs tracer ($T_{1/2}$ 2.1 years) was added to the soil in the trenches. The tracer was first added to a small area of the soil surface and then scraped into the trenches, and in addition, tracer was added to soil layers in the trenches themselves. In total, ca. 102 MBq ^{134}Cs was added to each trench, which is equivalent to ca. 5700 kBq/m^3 soil. Figures 3.1.a - 3.1.b provide an overview of the construction of the ranks. The procedure was essentially the same for all soil types.

The trenches are 10 m long and ca. 3.5 m wide, with a sub-surface depth of ca. 0.5 m. This provides a storage capacity of ca. 18 m^3 contaminated soil. The rank top is shaped so that rainfall will run off without penetrating through to the buried, radioactive material. In order to evalute the need for extra protection against rainfall, each trench was split into two sections 5 m in length, segregated with the help of a plastic wall. In one section the contaminated soil was covered with a plastic sheet, the other was not covered, then both sections were covered with a layer of clean soil. A drainage pipe was laid in the bottom of the trench, and the pipe terminated in a collection bowl for the sampling of drainage water. In the section from which seepwater was be collected, the pipe was perforated, in the other section the pipe was water-tight.

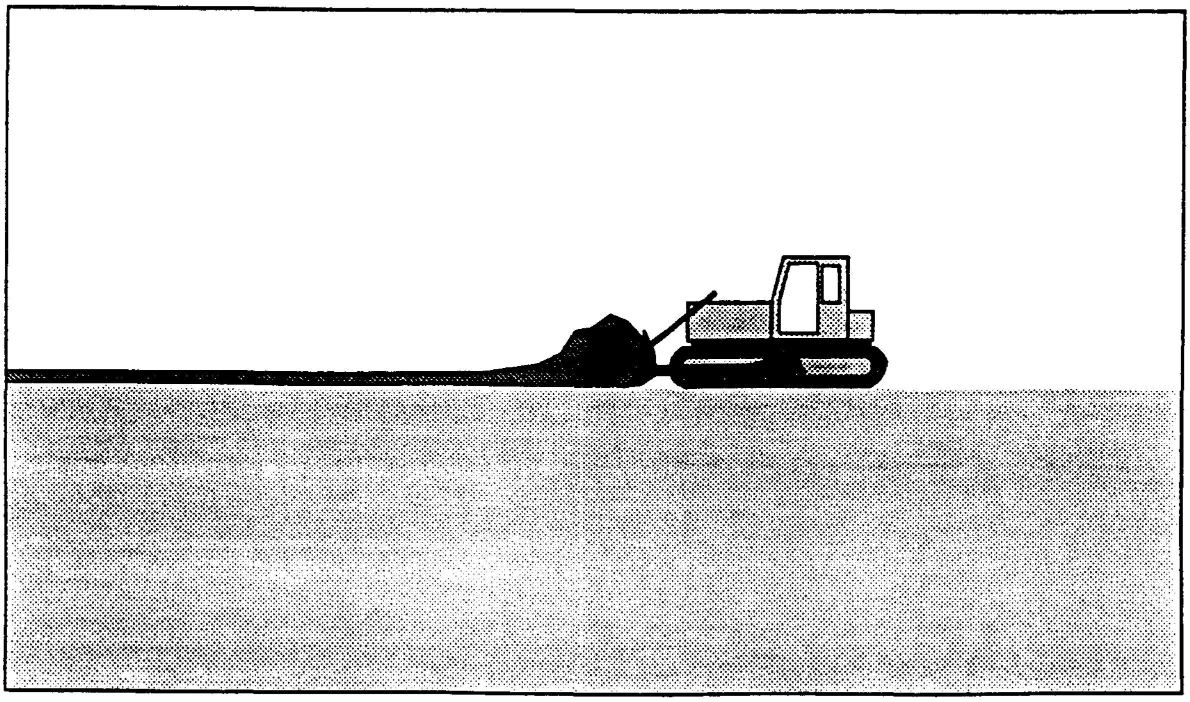

Figure 3.1.a. Preparation of the disposal site (buried ranks) by removal of the contaminated top layer.

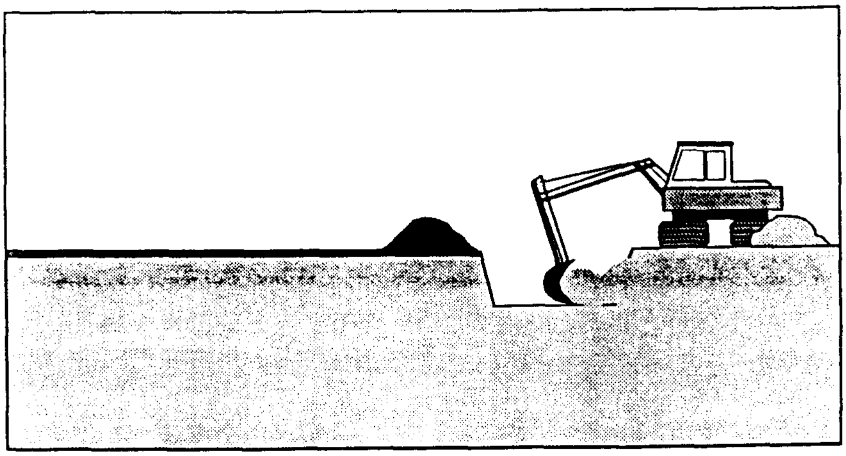

Figure 3.1.b. Excavation of the disposal trench: the clean soil should be removed and retained for the top cover of the rank.

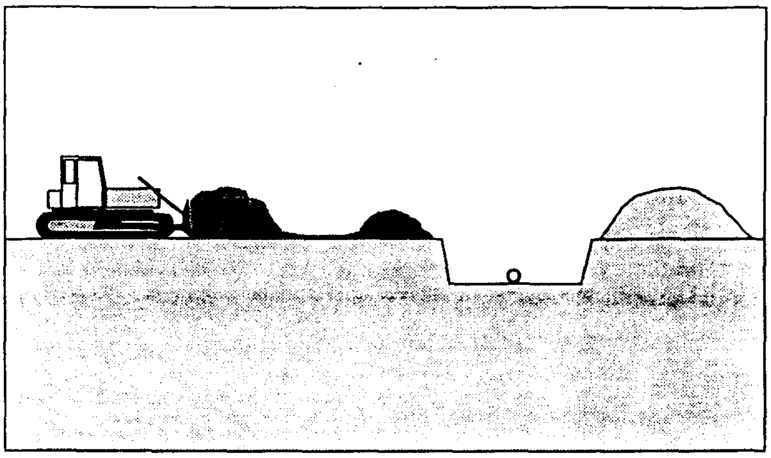

Figure 3.1.c. The contaminated soil is pushed into the disposal trench. A drainage pipe for inspection and sampling is laid at the bottom.

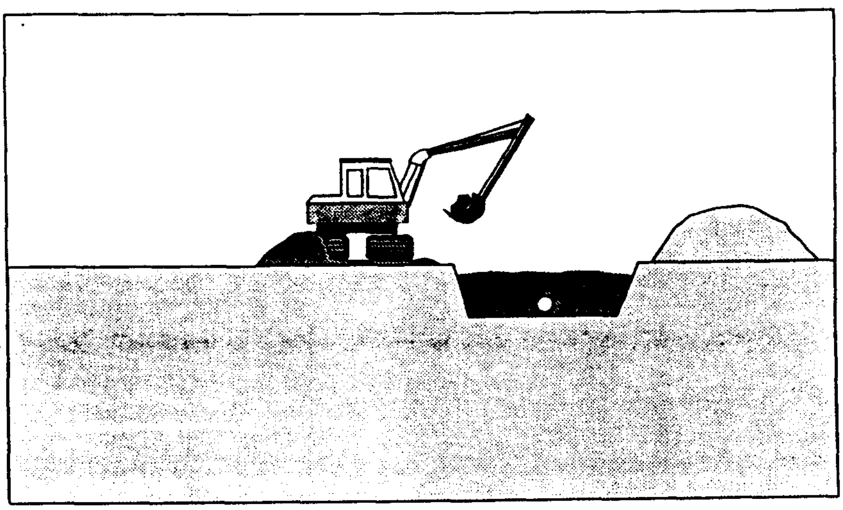

Figure 3.1.d. The remainder of the contaminated soil is placed in the disposal trench with the digger, which is also used to form the top of the rank.

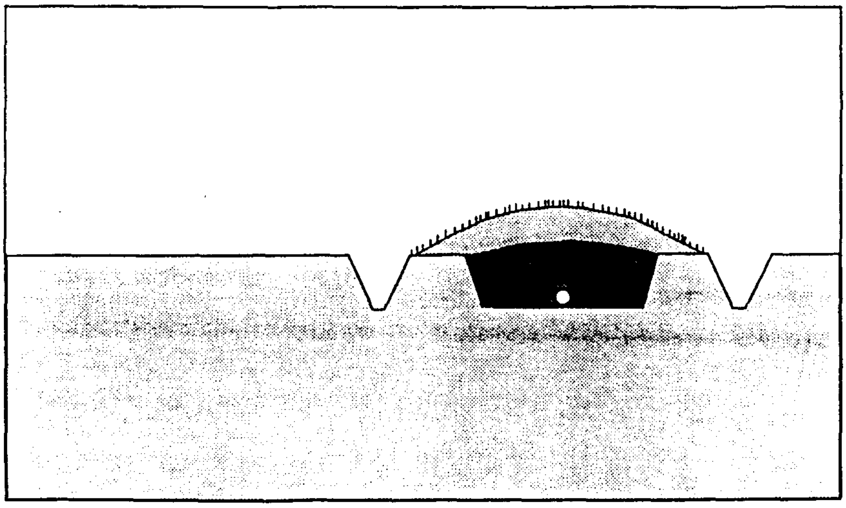

Figure 3.1.e. The finished disposal trench, with a top cover of seeded clean soil shaped such that water runs off easily. The height above the ground must be high enough to compensate for subsidence of the buried soil masses. Open or closed ditches around the trench helps to keep the groundwater low.

On sandy soils, because of the dangers of leaching, a plastic sheet was laid in the bottom of the trench. For disposal of contaminated peaty soils, a rank was dug on underlying clayey soil, since Cs is strongly bound to clay. A drainage pipe was placed at two levels: one along the bottom and one in the middle of the peat soil mass. Drainage from the upper pipe will give an indication of ^{134}Cs in seepwater from trenches on peaty soil, while that from the lower pipe will reflect the association of ^{134}Cs with clay. The migration of ^{134}Cs in the soil profile, and leaching from the trenches, have been followed since August 1992.

4.3.1.2 Migration of Cs in the trenches

In order to assess the migration of radiocaesium in the trench, a column of soil was taken from the centre of each section. The column was sectioned at 5 cm intervals and the activity concentrations were determined. Figures 3.2, 3.3 and 3.4 show the layered distribution of ^{134}Cs in the three soil types, and the two sections. It should be noted that the scales for activity concentrations are different in the two sections, and that the vertical height in the trench is on the x-axis.

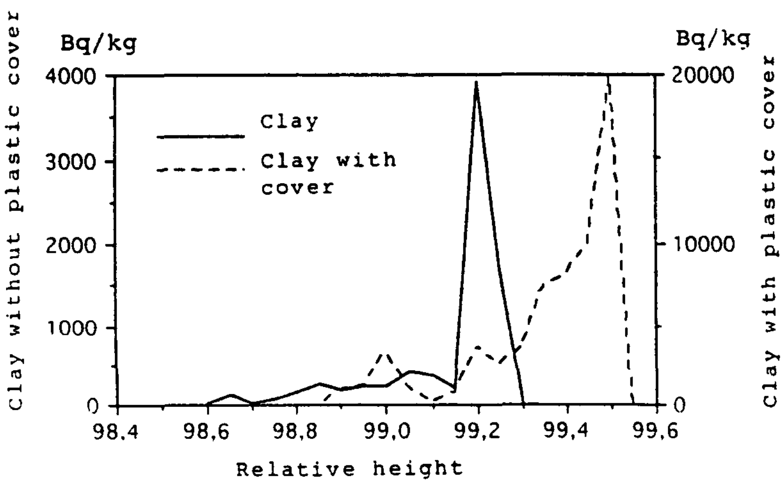

Figure 3.2. Vertical profile of a trench on clayey soil. The height is measured relative to a fixed marker.

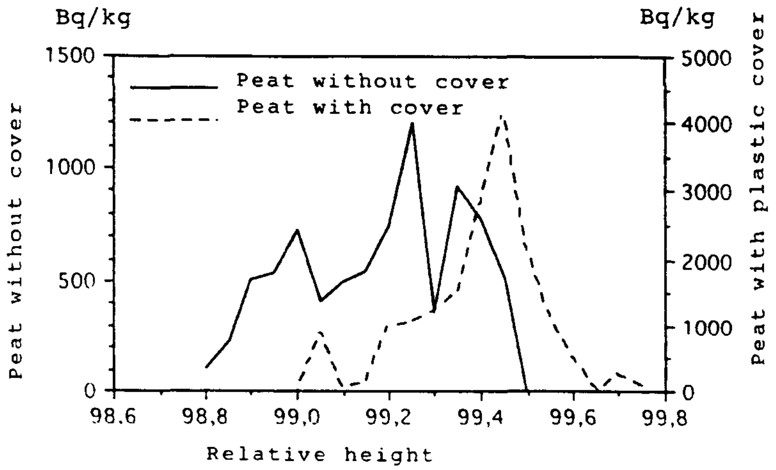

Figure 3.3. Vertical profile of a trench on peaty soil. The height is measured relative to a fixed marker.

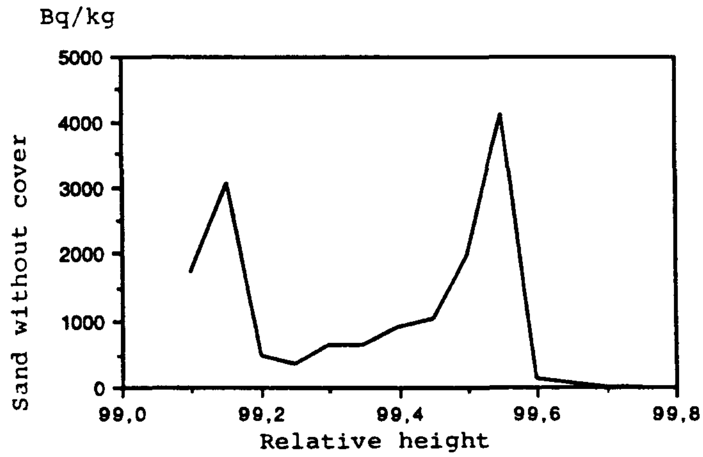

Figure 3.4. Vertical profile of a trench on sandy soil. Only the distribution from the section without the plastic cover is presented. The height is measured relative to a fixed marker.

The same activity of ^{134}Cs tracer was added to each trench. Even though the tracer was inhomogeneously distributed throughout the soil masses, this is of no relevance as only the relative movement of the most radioactive layer will be followed. In general, ^{134}Cs was added to each trench twice before covering with a layer (0.5 m) of clean soil.

4.3.1.3 Loss of seepwater from the trenches

Soon after construction, flooding in the sections on clayey and peaty soil trenches caused overflow in the collection bowl. Thus, open ditches were dug around the trenches to avoid such problems. Results to date indicate only a small loss of ^{134}Cs during the autumn months. It appears that the largest loss is from peaty soil trenches where the total ^{134}Cs activity in drainage amounts to less than 0.01 % (Figure 3.5.). The concentrations of ^{134}Cs in drainage decrease with time. When the peaty soil is, however, buried on clayey soils, the loss is much less.

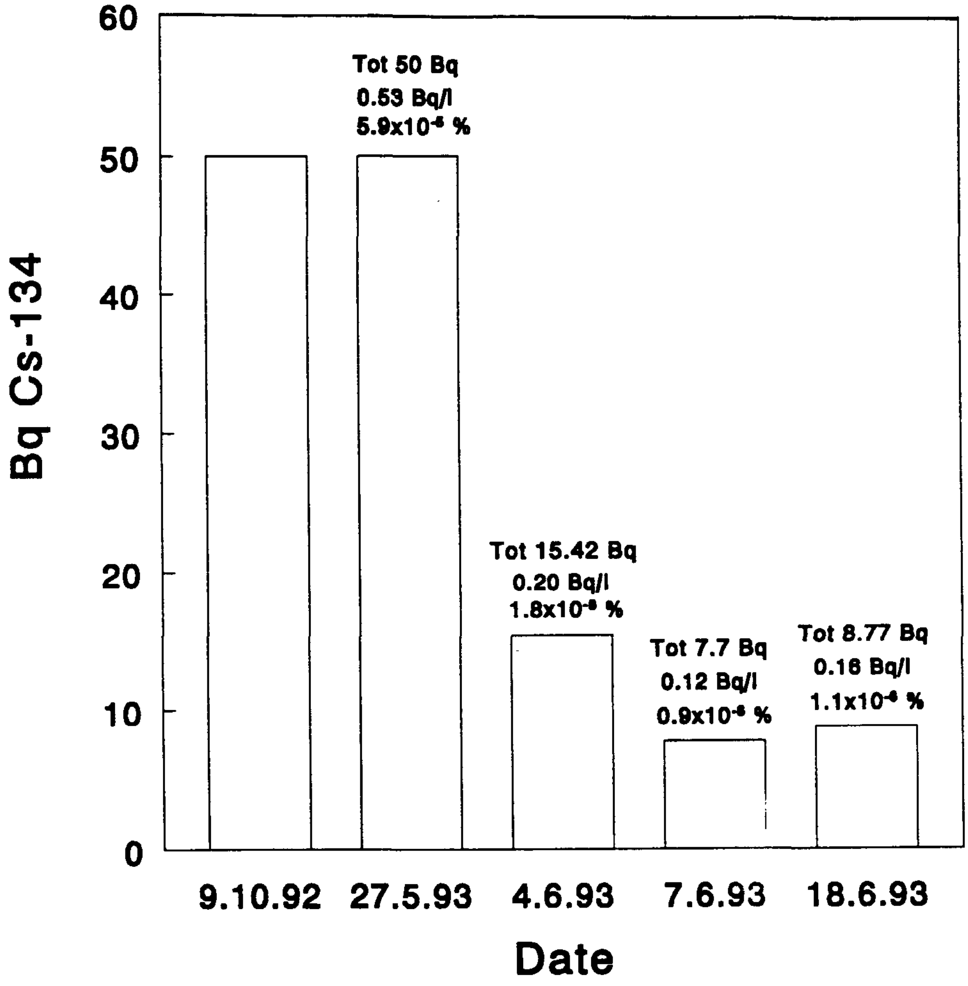

Figure 3.5. ^{134}Cs in drainage water from a peaty soil burial trench at Svanhovd.

4.3.2 Construction of disposal sites for contaminated soil after an accident

4.3.2.1 Background

After an accident leading to radioactive fallout, decisions on the introduction of specific countermeasures have to be taken. The following points should then be evaluated:

i) Which radionuclides are the most significant, in what concentrations are they present, and in what physico-chemical form.

ii) How much activity must be removed in order to reach acceptable levels in plants, animals and humans, such that agricultural production of foodstuffs for human consumption can be resumed.

iii) The choice of countermeasures will be dependent on the conclusions in the first two points. For example, "low" activities can be deep ploughed, whilst for "high" activities the surface soil must be scraped thoroughly and buried.

iv) Routine agricultural practices will almost certainly have to be suspended until the valid countermeasures have been evaluated, in order to prevent the dispersal of radioactivity through agricultural practices such as harvesting, ploughing etc. The possibility for subsequent deposition of new radioactivity must also be evaluated, i.e. decontamination should take place after woods and open land have ceased to represent possible sources of contamination due to resuspension of radioactivity.

4.3.2.2. Construction of the disposal trenches

The following points will be of importance for construction of the trenches:

i) The soil type must be suitable for the choice of scraping equipment and technique, and must possess the required retention properties for radionuclides.

ii) The available machinery for scraping will play a significant role in determining the volume of soil generated, and hence the space requirements for disposal, as well as the time needed and the cost-effectiveness of the operation.

iii) The climate, especially rainfall, will be significant with respect to the construction of the trench (i.e. whether it is necessary to have a plastic sheet over the waste), as well as for shape and slope of the mounds.

iv) The depth of the groundwater table will determine where the trench can be dug, and whether channels and ditches around the trench are necessary.

In principle, the disposal sites should follow the slope of the land, so that they do not dam surface water, and the trenches should be placed along the edges of the field. In large areas with high erosion, it is possible that the trenches could be placed crosswise to the slope and equipped with surface run-off channels to drain surface water. Soil erosion will be hindered because the length of the slope will be shorter. The distance of separation between each trench will be dependent on the soil volume which must be buried, and the maximum length which the scraping equipment can drive in a single run. This can, however, be regulated somewhat by making the width suit the situation in question.

4.3.3 Time consumption

The construction of the trenches can be split into a number of stages (see Figure 3.1). The total time needed will be dependent on the size of the trench, the equipment used, the proficiency of the drivers, the need for extra precautions against leaching, and the topography of the site. The time consumption was recorded in connection with the construction of the experimental trenches, and refers to a trench size of ca. 10 m length, 0.5 m deep and 3 m wide. A belt-driven and tractor-mounted digger was used to excavate and shape the burial trenches; a bulldozer and a road-grader were used to scrape the contaminated soil into the trech. Between 50 - 60 minutes were used to dig-out the trench, lay the drainage pipes, and fill with contaminated soil, and then cover with clean soil. This approximates to ca. 10 hour per 100 m length of the buried rank.

4.3.4 Conclusions

i) Construction of trenches is time and space demanding

ii) The necessity of precautionary measures against leaching, like draining (sinking the ground water table), and use of plastic sheeting over the contaminated soil masses, depend on local conditions.

iii) Where the size of cultivated land is small in comparison to the surrounding areas on which countermeasures cannot be carried out simpler trenches should be constructed, such as ranks along the edges of the fields seeded with grass to prevent wind and water erosion.

4.4. COST ANALYSIS FOR THE REMOVAL AND DISPOSAL OF CONTAMINATED SURFACE SOIL

After a severe nuclear accident, sophisticated cost-benefit analyses of the possible countermeasures (cost per man sivert) will be requisite in order to establish and implement the most effective methods of dose reduction in the exposed population. Scraping has been shown to be a viable method of decontaminating agricultural areas using equipment that is commonly available. However, a cost-benefit analysis of this method of decontamination must consider a large number of variables (i.e. levels of contamination, soil and crop type, etc.), and a reliable estimate of the cost of cleaning-up (e.g. removing surface soil) will be needed before a decision can be made. With this in mind, the cost, in % loss of productive area and machine hours per hectare, for removal and disposal of contaminated surface soil has been calculated.

The example is based on a one hectare area of agricultural land, and the use of a self-powered road scraper to remove the surface layer of sandy soils and a mechanical digger to excavate and construct the trenches. By making a number of runs, a road grader can manage a scraping length of 50 m. Therefore, a trench can be constructed at each opposite sides of the one hectare area/field (Figure 4.1).

It is assumed that the trench can be dug to a depth of 0.5 m without coming into contact with the ground water. In estimating the time needed for construction of the trench, no account has been taken of the need for extra coverage or other measures to prevent the loss of seepwater.

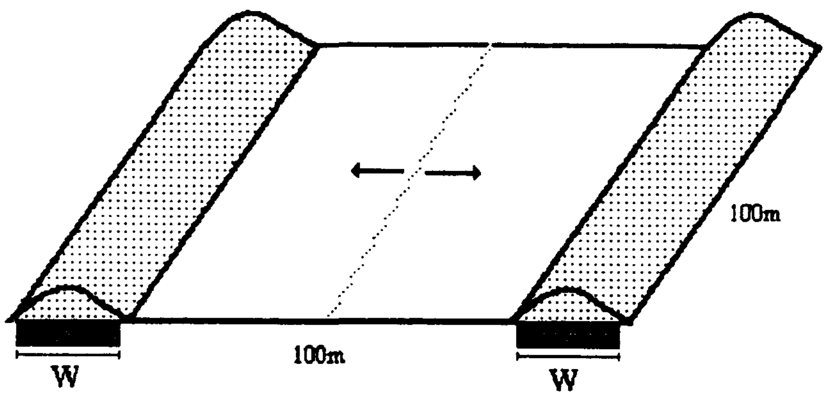

Figure 4.1. Plan of the one hectare field

4.4.1. Reduction in agricultural land

The amount of land removed from production as a result of the construction of disposal sites will be dependant on the volume of waste soil produced. For a 100 m long by 1 m wide scrape (100 m^2), the volume of soil produced, V, (m^3) will be dependant on the scraping depth, d (m), and the percentage volume increase, P, (%). Hence,

$$V\ (m^3) = d(1 + P/100)\ .\ 100\ m^2$$

For a scraping depth of 0.03 m and a volume increase of 30 %, the road grader will produce 3.9 m^3 of waste soil per 100 m^2.

Hence, the width, W, of each of the two 1 m long x 0.5 m deep buried ranks will be

$$W\ (m) = V$$

and the percentage reduction in productive land, L, allowing for a 0.5 m protusion of the top cover over the edges of the two ranks,

$$L\ (\%) = 2(W + 1) = 2[d(100 + P) + 1]$$

If extra drainage ditches and channels are necessary because of high groundwater, then larger area will be out of production.

On the basis of the above example, the width of each the two 0.5 m deep ranks will have to be 3.9 m plus 1 m covering edges, and the total area of cultivated land lost will be ca. 10 %. If the scraping depth had been 5 cm, the width of the ranks would have been 6.5 + 1 m and ca. 15 % of the land would be used for burial trenches.

4.4.2 Time consumption (machine hours)

As stated in section 3.4 approximately 1 hour is needed for the construction of each 10 m length of half-buried rank. Hence, for 1 ha the time needed for construction of the two 100m ranks and disposal of the radioactivity is about 20 machine hours. Between 1 and 2 hours are needed for scraping with the road grader, giving a total time of 22 machine hr/ha. If the scraping was carried out with another machine or on peaty soil, the scraping time would be more like 5 hr/ha, and the time needed would increase to 25 machine hours in total. The cost of one machine hour (i.e. driver's wage, fuel and other running costs) has been estimated at between 50 and 80 ECU/hour for the tested machines. Assuming 80 ECU/hour as the cost per machine hour, the machine cost for the above example is estimated at between 1760 and 2000 ECU/ha. The estimated cost does not include compensation to the drivers for increased risk (i.e. from exposure to enhanced levels of radiation relative to the other members of the community) or the future cost of additional monitoring of possible health defects in the exposed individuals.

In light of the considerable benefits of decontaminating agricultural land, this time seems to be reasonable, especially if at least two machines are working simultaneously. The time for scraping is calculated as effective time without empty driving and turning, and cost wise there is much to be gained by developing effective scraping methods. However, reduction in the time needed for construction of the ranks will have the most significant effect, and it is clear that the time needed for decontamination of a large region of agricultural land will be largely dependent on the number of available diggers.

4.5. REFERENCES

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APPENDIX

VEHICHLES USED FOR THE REMOVAL OF SOIL IN THE FIELD EXPERIMENTS IN THE EAST FINNMARK, NORWAY

Photographs below show the vehicles used in the field experiments in the East Finnmark, Norway, for testing their capability of removing surface soil layers. Numbers following the figure text in paranthesis refer to sections dealing with these vehichles.

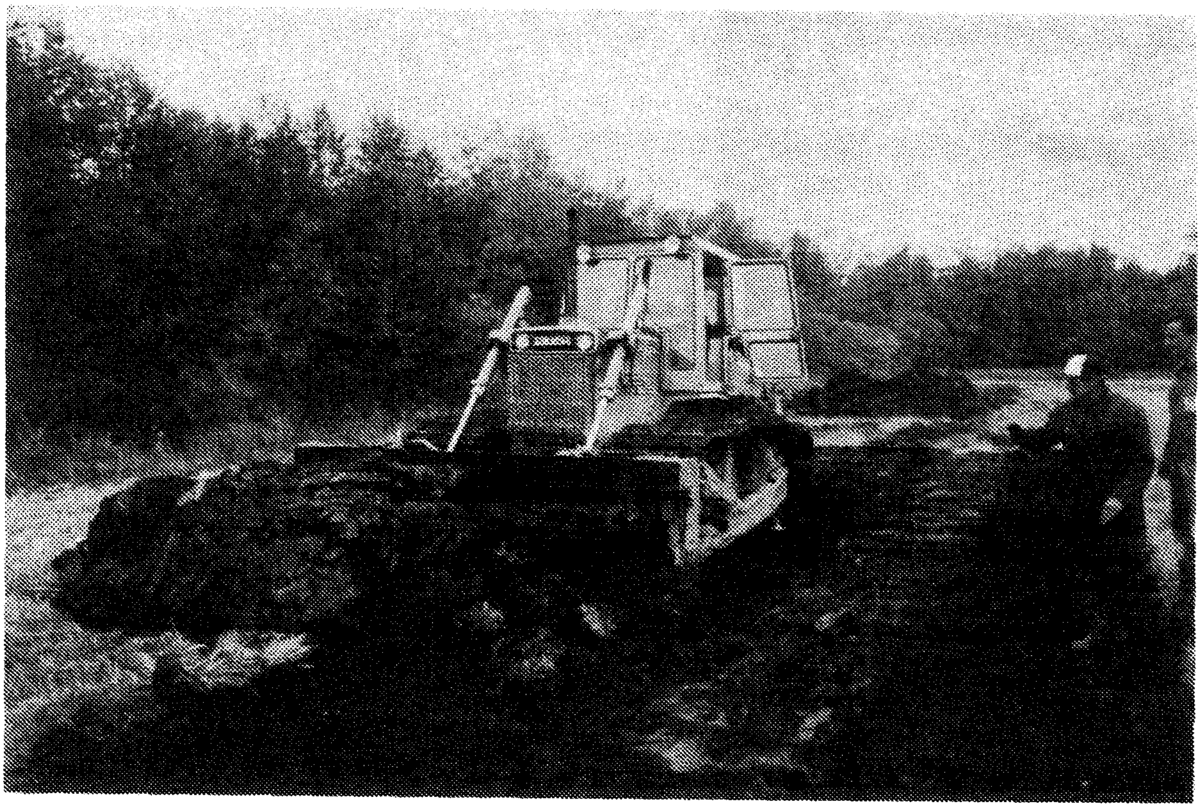

Figure 1. Bulldozer (4.2.4.1.).

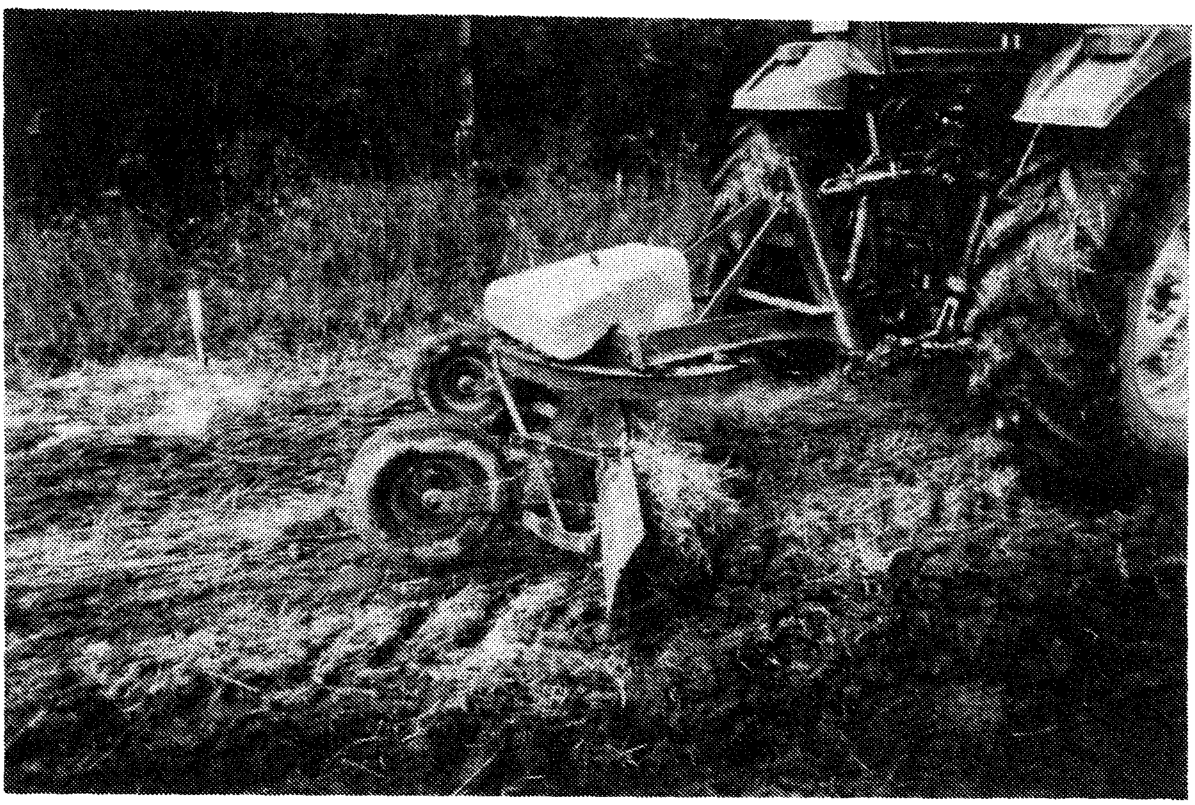

Figure 2. Tractor-mounted road scraper (4.2.4.2.).

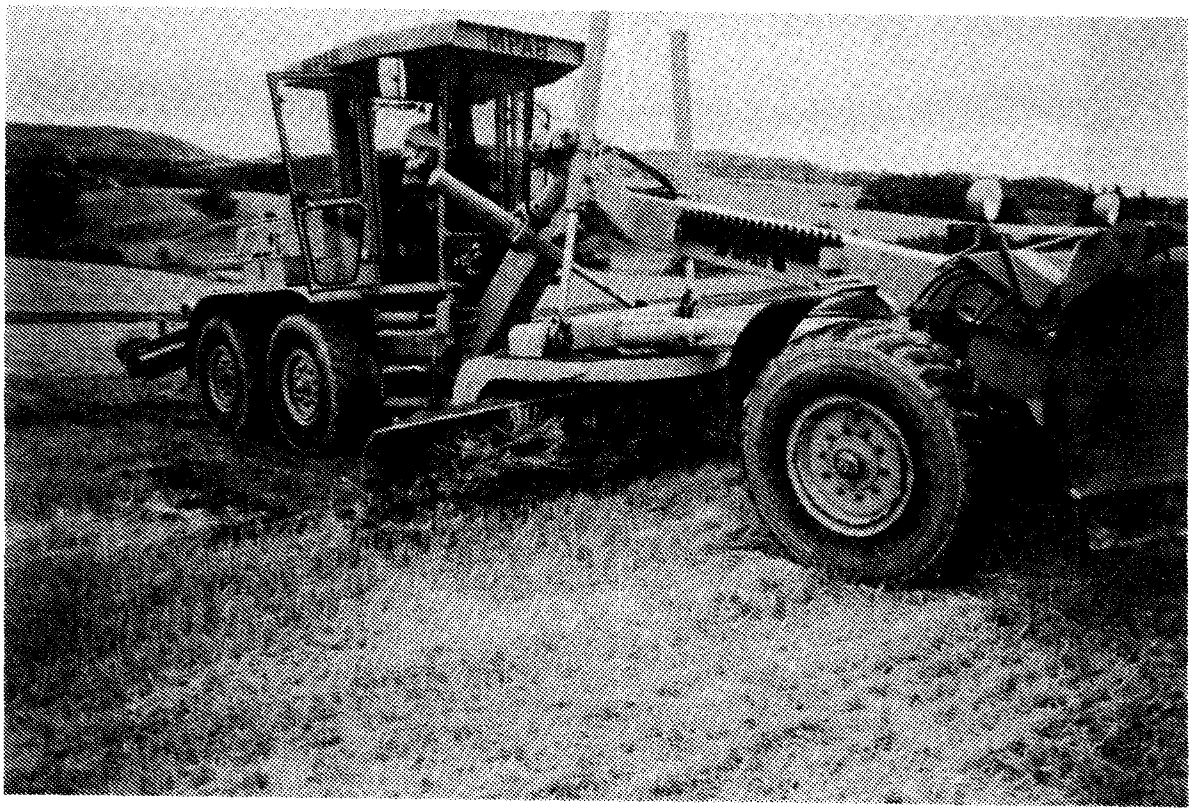

Figure 3. Self-powered road scraper/road grader (4.2.4.3.).

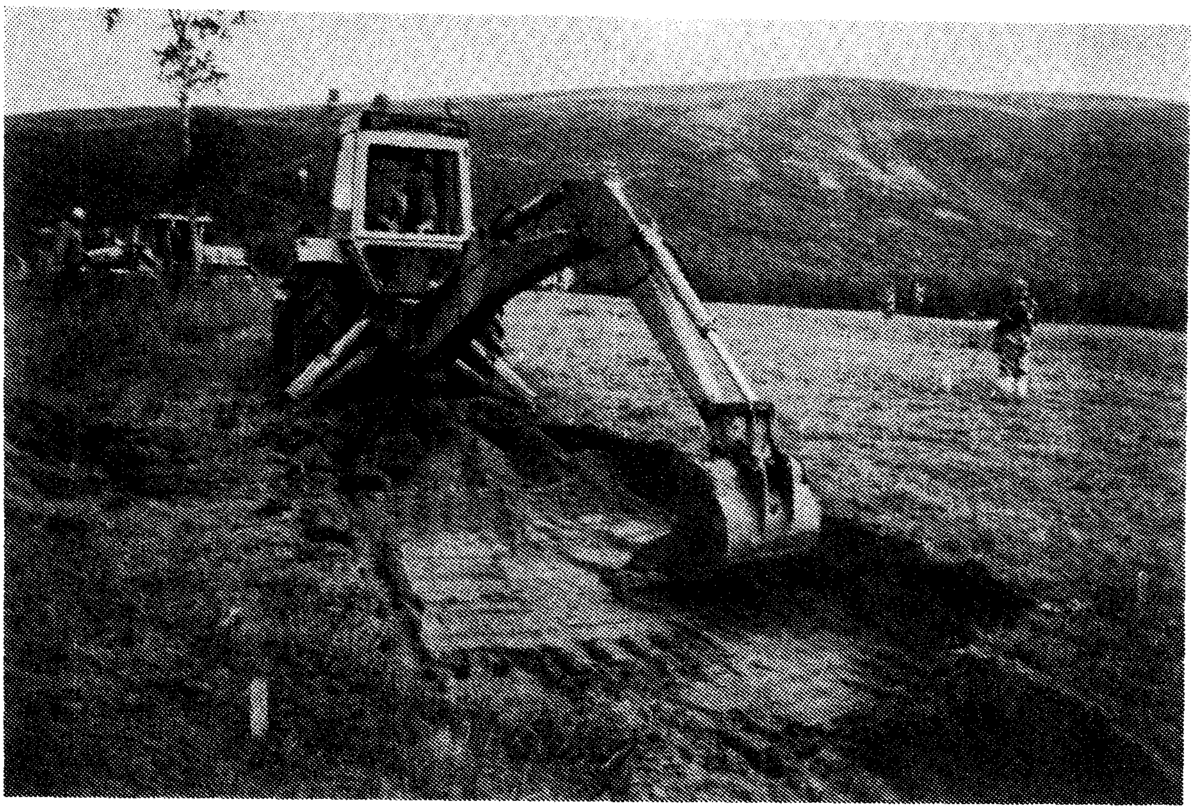

Figure 4. Mechanical digger (4.2.4.4.).

Figure 5. Front loader (4.2.4.5.).

Figure 6. Tractor with snow plough (4.2.4.6.).

5. EXPERIMENTAL STUDIES OF INTERACTION OF RADIOACTIVE CONTAMINATED SOIL AND CONCRETE OR DECAYING ORGANIC MATERIAL

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Cement solidification of radioactive contaminated soil was investigated. Humic materials retard the hardening reactions, but this was partly overcome by use of a special additive. However, the method was not usable for all soils. The retention of strontium and cesium radioisotopes was less efficient in the cemented soils than in untreated soils, although temporary improvements can be achieved for Sr when the product reacts with carbon dioxide from the environment. The low retention is caused by ion-exchange competition with alkali metal ions from the cement. Absorption and desorption of Cs and Sr in Ca or K solutions at high pH were investigated and confirm this assumption. When contaminated soil is in contact with concrete, used as barrier or for construction purposes, similar effects are observed, but they are probably of minor practical importance.

The migration of Sr, Cs and Eu with percolating water in soil under unsaturated conditions was investigated using a column technique. When the soil is mixed with decaying vegetable material the migration of Sr and Cs was considerably enhanced.

5.1. INTRODUCTION.

This part of the KAN2 report is a summary of experimental work carried out at the Risø National Laboratory, Denmark, from 1990 to 1993.

Two different topics were investigated. These were 1) the use of cement solidification and concrete barriers in the management of contaminated soil, and 2) the influence of decaying organic materials on migration in soil columns.

5.1.1. Soil contamination and management options.

The types and amounts of contaminated soil and other waste products which may result in connection with clean-up activities after a large nuclear accident are described in other parts of this report. In reactor accidents the most important long-lived radioisotope is ^{137}Cs, however, ^{90}Sr and α-emitters may also escape from a damaged core and be deposited on soil or vegetation, sometimes in form of difficult soluble 'hot' particles. Accidents at nuclear fuel cycle installations other than reactors can also result in soil contamination with fission products and/or α-emitters.

Accidents during transport of spent fuel, waste or other materials connected with the nuclear fuel cycle are not likely to result in releases, but should they occur, the releases and the size of the contaminated area will probably be small. Special transport accidents - including the historical cases of crashes of airplanes carrying nuclear weapons and the various incidents of return of nuclear powered satellites through the atmosphere - may give rise to more serious and widespread soil contamination. Mismanagement of the use of radioisotopes, especially big sources containing large amounts of long-lived isotopes, has also given rise to serious problems (see Chapter 3).

Contaminated soils are sometimes detected during the decommissioning of old nuclear installations and may have to be removed. A special case is pollution around operating or abandoned uranium mines. Waste disposal itself may also result in soil contamination, but in well sited and designed repositories the planned long-term releases to under-ground soil should be of no consequence and should not require any remedial actions.

Limited amounts of contaminated soil are produced in research projects where the radioisotopes are used to study migration behaviour, plant uptake etc.

The amounts and the specific activities of the contaminated soil and other materials produced in one of the above-mentioned contexts may vary over many decades. The amounts can be so huge that the methods selected for handling and disposal primarily are determined by the practical possibilities in the particular circumstances.

The second of the laboratory studies described in this part of the report is relevant in this context because large amounts of decaying organic material is likely to be present in primitive ad hoc systems for burial of contaminated soil, see Section 3. Further details are given in [1].

When the amounts of waste are smaller, and especially if the activity level in the contaminated soil is relatively high, it may be motivated to select a more advanced management and disposal system. Conditioning with cement might be a possibility in this context, and concrete is also used extensively as a barrier material in engineered burial systems.

The main part of the laboratory studies described here was carried out to investigate the feasibility and the possible problems associated with the use of cement in connection with contaminated soil. A summary of the experiments is given in Section 2. Further details are available in [2].

5.2. CEMENT SOLIDIFICATION OF SOIL AND INTERACTION BETWEEN SOIL AND CONCRETE BARRIERS

Conditioning of loose materials may make it permissible to use some types of transport, storage or disposal which would be undesirable for the unconditioned material. Improvement of the material properties may be desirable in general or it could be neccessary to comply with acceptance criteria for existing disposal systems.

Conditioning should result in products which are:

- Long-time stable in the disposal environment.
- Have a reduced rate of activity release compared with the unconditioned material.
- Are safe to transport (non-dusting and with a low fire risk).

And the conditioning must be possible:

- Without generating too much secondary waste.
- Without significant radiation doses to personnel.
- At a reasonably low cost.
- With some volume reduction or only a small volume increase.

If any real conditioning is to be done, collection and transport to a treatment facility is a prerequisite. The practical, dose related, environmental and economic consequences of such operations are not discussed here. In the following the contaminated soil is supposed to be available for conditioning at a suitable facility. The facility may be an existing one, a hired mobile unit or some installation build for the purpose, for example near the final disposal site.

5.2.1. Problems in cement solidification of soils.

Soil and especially surface soils contain considerable amounts of partly decayed organic materials in the form of humic acid, etc. These are known to retard the hydration reactions in cement, resulting in the slow development of mechanical strength or even in complete prevention of solidification. Standards and handbooks therefore specify a low content of humic material in sand and gravel used in ordinary construction concrete. The effects of humic materials in concrete are reviewed in [3] and [4].

Different soils will also have widely variable contents of clay minerals. These may to some degree influence the hydration via pozzolanic reactions [5]. A high clay content requires the addition of large volumes of water in the mixing procedure to obtain castable products. This will result in low-strength materials with a high internal porosity.

Interaction between soil and concrete constructions is a different topic. Information is available from studies of the interaction between concrete barriers and surrounding clay layers. Indications for decalcification of the concrete in the contact zone were obtained in a French study [6].

Cementitious materials are always somewhat porous. The volume fraction of pores, the pore-size distribution and the degree of tortuosity and interconnectivity of the pores determine to some degree how well the pollutants are retained within the material [7].

Chemical retention is also important and is much dependent on the chemistry of the specific pollutant. One posibility is ion-exchange, another is fixation in low-solubility minerals in the cemented material. Some pollutants - especially of the heavy metal type (including plutonium etc.) - are very well fixed in high pH cementitious materials. However, within the extended time frame of disposal, the pH is likely to decrease due to leaching and/or reaction with CO_2 from the environment. This may result in remobilisation of some pollutants.

A survey [2] of the literature concerned with combinations of soil and cement has been carried out. There is a large number of patents on cement and various additives used for soil conditioning and stabilization for construction purposes. A group of patents is concerned with solidification of sludges, for example ordinary waste-water purification sludge. A common feature appears to be the formation of ettringite acting as a water-binding component. This reaction could also be of interest for the present study, because contaminated waste-water purification sludge is likely to be a problem in connection with large accidents.

5.2.2. The GEODUR method.

The Danish firm GEODUR A/S has developed an additive which is supposed to improve the product quality and to overcome the retardation effects of humic materials in soil/cement mixtures. This proprietary additive is cheap and non-toxic*.

The additive is available as a white paste which must be mixed with water using a high-speed stirrer. The resulting 1 % suspension is what is meant by the GEODUR additive in the following. About 2 % by weight is used in preparing the soil/cement mixtures and the amount of solid material introduced with the additive is therefore low, only about 0.01 weight percent.

The additive should be mixed into the semi-dry soil for about five minutes. Then the dry cement powder is added and mixed with the soil for further five minutes. Finally water is added as required to obtain castable products. Products with less water can also be made, but must be rammed into position. Conventional high-shear concrete mixers are used in the mixing procedure.

Monolithic blocks with very low cement contents (~8%) and an adequate strength have been made from many types of soil or other materials using the GEODUR additive. This was demonstrated by the firm in 1988 using top soil from the Risø area. The appearance and touch of the dry product is like a grey-yellow sandstone.

Characterization of samples prepared from the same soil with 0, 2, 6 % GEODUR and 8 or 16 % cement was carried out in 1989-90. The work is reported in detail in [8,9], and has since been supplemented by further experiments done within the frame of the KAN2 project [2].

* Further information about the GEODUR additive may be obtained from:
S. Mortensen, Geodur A/S, Birkeroed Kongevej 49, DK-3460 Birkeroed, Tlf. +45-45 826900.

5.2.3. Soils used in the experiments.

The Risø top soil (marked Ri in the following) was obtained from the tillage layer of fields near the research center. It is developed from typical moraine material and is classified as a sandy clay.

Five other soil types were also employed. One (Ro) is a garden soil (0-5 cm) from Roskilde, very rich in organics. Another (Or) is a coarse sandy moraine soil from a tillage layer on the island of Orø. Two (UP and UR) are wind-shifted sand from the A-horizons in respectively a beech and a spruce plantation in Western Jutland. The last (Li) is from the A-horizon of a much washed-out moraine from the previous glaciation in a beech plantation at Lindet in Southern Jutland.

Texture analyses and other relevant parameters are given in Table 1. All the soils were sieved to remove > 2 mm particles: stones, roots etc.

Table 1. Texture and some chemical properties of the six soil types used in the experiments with soil solidification.

		Ri	Or	Ro	Li	UP	UR
Texture:							
Clay	<0.002 mm %	13	9	7	5	1	5
Silt	0.002-0.02 mm %	15	10	13	8	2	3
Fine sand	0.02-0.2 mm %	47	81	80	87	97	92
Coarse sand	0.2-2 mm %	24					
Dry material	%	99.2	90.1	97.4	99.2	99.6	97.4
Weight loss at 550°C	% of dry mat.	3.8	9.0	19.8	5.8	3.4	7.0
Chemical oxygen demand							
COD mg O/g dry mat.		45	72	135	74	56	40
Equivalent organics: $-CH_2O$ % of dry mat.		4.2	6.8	12.7	6.9	5.2	3.8
pH (1:2 soil/deionized water)		7.40	6.74	7.62	4.52	4.34	4.44
H^+ released to reach pH 11.5							
meq H^+/100 g soil			18	29	21	9	31

The weight loss on ignition (~20 h at 550°C) includes organics and some water loss from clay particles. The COD determination (cromate oxidation) represents oxygen consumption by the oxidation of organics. The ratio employed for the conversion to organic material, $CH_2O/2O = 30/32$, is likely to overestimate the humus content. The weight losses and the COD determinations indicate that a considerable range of organic content is covered by the 6 soil types.

The soil pH was measured in a solution obtained by centrifugation of 1:2 soil/deionized water suspensions. As expected the three soils from agricultural or garden areas has pHs about 7, while the three from forests were quite acidic.

The texture analyses were made after removal of humic material by treatment with H_2O_2. The clay and silt contents in UR and especially UP is seen to be very low, while the Ri soil has the highest clay content.

5.2.4. Ion-exchange capacities at high pH.

The cation-exchange capacity of soil is due to the clay content and the contents of humic materials and Al- and Fe-hydroxy compounds. The part of the CEC associated with the clay minerals is relatively independent of pH while the weakly dissociated hydroxy, carboxylic and phenolic groups present in the other materials give rise to a considerable increase in total CEC with increasing pH. This has been measured for all 6 soil types using a method described in [10]. The results are shown in Fig. 1.

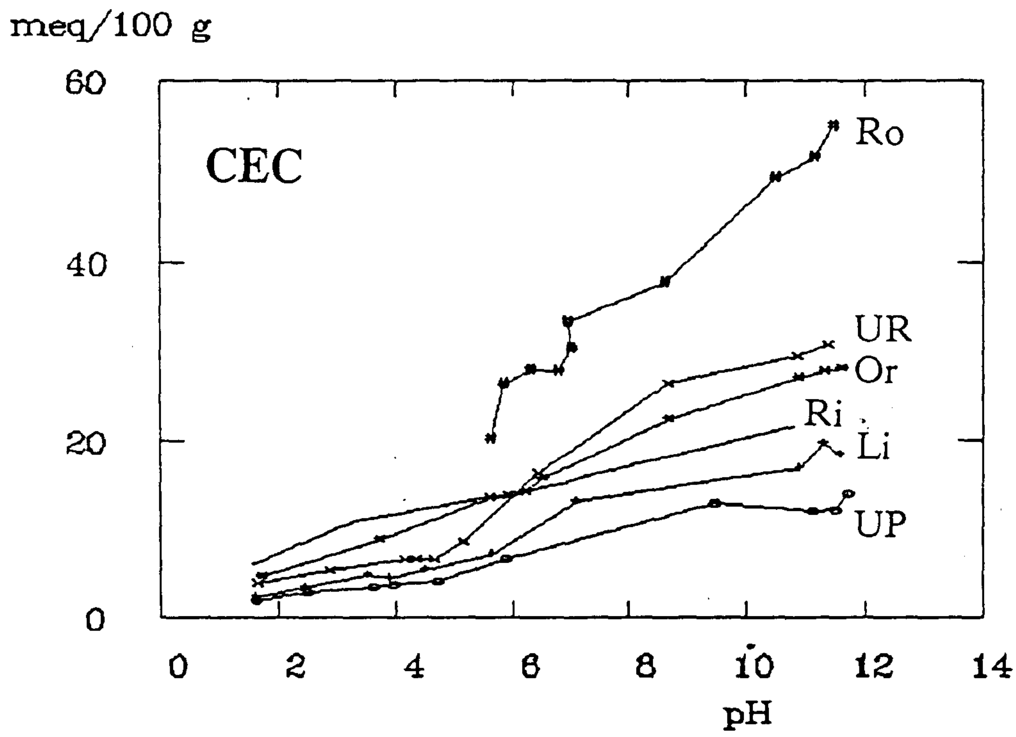


Figure 1. Cation exchange capacity as a function of pH for 6 different soils.

The amount of H^+ ions released from the soils when the pH is increased was also measured (see Table 1). A similar reaction takes place when cement is mixed with soil. The resulting change in pH from the initial values of 4-7 to pH ~11.5 involve the release of 9-31 meq H^+/100 g soil, equivalent to up to about 15 % of the total calcium added to a cemented soil containing 8 % cement.

5.2.5. Sorption/desorption of Cs and Sr for soil at high pH.

In a cemented soil most of the CEC sites are covered by Ca^{2+} or K^+ and Na^+ ions. The concentration of dissolved alkali-metal hydroxides in the pore water in cementitious materials is considerable, up to about 1 molar. Through the common OH^- ion effect this will suppress the Ca concentration, but when the relatively low alkali metal content is leached, the Ca^{2+} concentration rises towards 0.02 M as determined by the solublility of $Ca(OH)_2$ and the hydrated calcium silicate. The pH is typically about 11-13.

The high Ca or Na and K concentrations are likely to result in replacement of any minor components retained by reversible ion exchange on the soil particles. This has been investigated for the Risø soil at high pH. Measurements for ^{85}Sr and ^{134}Cs in Ca solutions and for ^{134}Cs in K solutions are available [2,10].

Absorption followed by up to 5 desorption steps were measured at various carrier concentrations and at two Ca or three K concentrations. It was found that the concentration of Cs in solution in equilibrium with the same content of Cs in the solid is higher by a factor of 5 to 10 in K than in Ca solution of the same molarity. The effects are known from the literature [11] but may not have been investigated before at high pH.

Incomplete desorption of Cs in weak K-solution was also demonstrated but the phenomenon is more pronounced in Ca solutions. The increased releases of Cs from cemented soil, which will be commented on later, may therefore primarily be caused by the small amount of potassium (and sodium) present in the cement.

Adsorption and desorption of Sr at high pH versus Ca concentrations were also studied. The behaviour was much more like ideal ion exchange maybe partly because Sr is present in much higher concentrations in the cement/soil mixture than Cs.

5.2.6. Sample preparation and physical properties of cemented soil.

Inactive as well as active samples of cement soil have been prepared from the six soils by mixing with various amounts of cement (SRPC, Danish low alkali sulphate resistant Portland cement), water and the GEODUR additive. Samples with cement, but without GEODUR, and samples of the pure soils were also prepared and used as controls. The active materials were cast in the form of blocks for use in the leaching experiments (Section 2.8) or were, after a few days initial setting, broken into small bits: < 5 mm and used in extraction experiments (Section 2.7).

The inactive samples were mainly used for the determination of physical properties. Results for the Ri soil are given in Table 2. The compression strength is low but increases with the cement content. The hydraulic permeability of the cemented soil is improved compared with untreated Ri soil. The permeability decreases with increasing cement and decreasing water content.

The hardening of the five other types of soil was unsatisfactory. The explanation could be a combination of the high humic contents and the high water contents necessary to produce castable products. Control materials without the GEODUR additive were also made and they behaved in the same manner.

This lack of effect of the additive was studied in an additional experiment where the Ri- and the Ro soils were combined in the proportions: 1/0, 0.75/0.25, 0.5/0.5, and 0.25/0.75, and then mixed with 8 % SRPC and 2 % GEODUR. Only the two first mixtures hardened, indicating that the lack of solidification is due to the very high content of humic materials in the Ro soil.

The physical stability of a barrier layer of soil, cemented soil or concrete is especially important when the material is used as protection around e.g. more strongly contaminated soil in near-surface burial systems. Wet-dry volume changes may result in crack formation. Studies of this type on cemented Ri soil were reported in [8]. Drying gave volume contractions up to 5 % for the 8-2 and 3 % for the 16-2 material, only partly compensated by reswelling when the samples were again exposed to water.

Table 2. Densities and compressive strength of cement-solidified soils measured using 4.4 cm diameter cylinders (h=8.8 cm) after 54 days of hardening at high humidity, and hydraulic conductivities measured for ~2 cm thick slabs in the case of the cemented product and a 3.8 cm thick layer in the case of the unconditioned soil (0-0-15).

	Product composition			Density (water saturated)	Compressive strength	Hydraulic conductivity
	Cement	GEODUR	Water	g/cm^3	MPa	m/sec
Reference cement mortar	30	0	13	2.3	~50	$2.5 \cdot 10^{-10}$
Soil type:	16	2	24	1.87	4.5	$0.6 \cdot 10^{-10}$
	8	6	25	1.83	1.3	$2.0 \cdot 10^{-10}$
Ri	8	2	24	1.89	1.5	$2.3 \cdot 10^{-10}$
	8	2	14	2.05	1.2	$0.6 \cdot 10^{-10}$
	0	0	15		-	$5.0 \cdot 10^{-10}$

5.2.7. Leaching from granulated cemented soil after various pretreatments.

Leaching of granulated cemented soil is somewhat similar to the extraction of soil at high pH as described in Section 2.5. Three series of experiments were run using samples of cemented Ri soil. The methodology and results are described in [2,8,9]. Most of the observed phenomena can be explained by a combination of precipitation and more or less reversible ion-exchange rections.

The finer details of the Cs-leaching is partly hidden by the formation of colloids in weak ionic-strength solutions, but there is no doubt that Cs is retained most efficiently in cemented as well as pure soil even at high pH and in rather strong calcium solutions.
Drying appears to result in a stronger fixation of the cesium especially in products with high cement content. The influence of CO_2 is uncertain.

The leached fractions for Sr are much larger than for Cs and the question of colloids is therefore of no practical concern. Strontium is probably mainly bound by reversible ion-exchange in the soil. It can be leached quite easily from cemented soil by deionized water (i.e. solution chemistry mainly determined by the cement content) and from the pure soil by calcium bicarbonate solution. However, when CO_2 or bicarbonate reacts at high pH in cemented soil, precipitation of nearly insoluble carbonates greatly reduces the strontium release. Resolubilization in the form of bicarbonate is possible at high CO_2 exposure.

The effect of the cement is mainly to modify the chemistry of the system by changing the composition of the leach solution. The resulting Ca concentrations varies between 1 and 10 mM/l with the lowest values for systems with pure soils. The leached fraction of Ca is highest in deionized water, but is strongly decreased for the carbonated material or when calcium is present in advance as in the leach solutions. From 50% to 90% of the Na and K contents are leached.

The pH varies from ~7 to ~12 with OH^- and CO_3^{2-} as the main anions at the high pH of the cementitious systems, HCO_3^- dominates at the lower pH values of the pure soil or the carbonated material in deionized or tap water.

The presence of the GEODUR additive does not change the behaviour of the systems. The results are obtained for a soil with high clay content. In cases where the retention is more due to humic materials the chemical behaviour of the radioisotopes may well be different.

5.2.8. Leaching from layers of soil or cemented soil.

Experiments with one-dimensional diffusive leaching from layers of soil or cemented soil were carried out using a method employed extensively in previous studies [13]. A layer of the contaminated soil or soil/cement mixture is placed at the bottom of polyethylene flasks and exposed to successive portions of leach solution. Only some of the solution is replaced at each sampling. Deionized water, 3 % NaCl or 0.001 M NaCl were used as leachants. Some of the systems were closed and some open to the atmosphere resulting in CO_2 uptake by the alkaline solutions.

The accumulative leached amounts of ^{134}Cs and ^{85}Sr are converted to the equivalent leached thicknesses L by division with the sample area and the initial activity concentrations in the products. Diffusion-controlled leaching is indicated when L plotted against the square root of time results in a reasonably straight line. The effective diffusion coefficient $D_{e(leach)}$ is then calculated from the equation:

$$L = 2\sqrt{(D_{e(leach)} \cdot t/\pi)}$$

Early results for cemented Ri soil are reported in [8,9] and were later supplemented with similar measurements on untreated Ri soil.

Cement solidification of the five other soils described in Table 1 was unsuccessful, but comparison of the leaching behaviour of the pure soils and the soft cement-soil mixtures is of interest as an illustration of interaction phenomena.

Results for four of the soils are shown as leach curves plotted against the square root of time in Fig. 2. Systems with 0 %, 8 % and 16 % cement mixed with the soil are included. The leachant was 0.001 M NaCl and all the systems were open to the atmosphere. The apparent diffusivities obtained from the slope of the later part of the curves are given in Table 3. The cation-exchange capacities at the pH of the systems are taken from Fig. 1 and give some indication of the different properties of the soils.

The pattern for diffusive leaching of Cs from the soils or soil/cement mixtures is somewhat complicated: For soils with a high content of clay or organic materials (and therefore high CEC values, Or, Ro and Li) a rapid initial release from the pure soil is followed by slow releases at a rate somewhat lower than that for the same soils mixed with cement. The pure soils may contain some of the Cs in a form easily exchanged with Na^+ from the leachant. In the soil/cement mixtures the pore water is initially of high ionic strength, and the following exposure to the 0.001 M NaCl leachant should not result in drastic changes in the conditions for release. The tendency for the soil/cement mixtures is therefore more likely to follow the same $\sqrt{t}$ law over the whole period.

For the soils with low clay contents (UR and UP, and for UP also a relative low content of organics) the releases are always highest from the cement mixtures. In the case of UP soil the cemented samples approach Cs depletion after about 100 days. The reason is probably that Cs is retained relatively inefficiently by the soils where the CEC is mainly due to organic materials. Although the CEC is increased in the alkaline soil/ cement mixtures this is not sufficient to compensate for the ion exchange caused by the high cation concentrations in the pore solution.

Diffusive leaching of Sr is different: In all cases the Sr ions are bound most efficiently in the pure soils. This may be due to preferential retention of Sr on the ion- exchange sites relative to the Na ions available for exchange from the leach solution.

In the soil-cement systems the amount of Sr carrier is much higher due to the strontium content in the cement. In addition the high ionic strength of the pore water and the availability of Ca ions result in high initial releases from all the systems. Later the release rates diminish, but this is probably exclusively an effect of CO_2 which in the highly alkaline systems results in the precipitation of low-solubility species.

For the UP soil with poor retention ability it should be noted that the retention of Sr is somewhat better than for Cs.

The D_e values in Table 3 for leaching from cemented Ri soil with or without CO_2 access and leached with deionized water or 3 % NaCl solution (~0.5 M) are taken from the earlier studies [8,9]. It is seen that the release of Cs increases with cement content and is considerably higher in the 3 % NaCl solution. Access to CO_2 from the atmosphere is of minor importance.

The release of Sr is also high from the cemented Ri soil, provided the systems are closed. With access to CO_2 the releases are considerably lower. For the sample with 8 % cement leached with 3 % NaCl the CO_2 uptake was sufficient to reach pH values where the bicarbonate ion is dominant. Resolubilisation of carbonate is probably the explanation of the increased Sr release observed in the later part of this experiment.

Calcium is leached quite easily from the cemented Ri soil in systems without CO_2 access. The calcium depletion is considerable, corresponding to 1-2 cm equivalent leached thickness in 100 days. The pH remained at about 12 in the closed systems, but dropped to values of about 8 in the systems open to the atmosphere.

The general conclusion of the leaching experiments is that cementation is not advantageous for the retention of Cs and Sr compared with the untreated soils. However, for Sr some temporary improvements are possible when cementation is combined with the precipitation of carbonates.

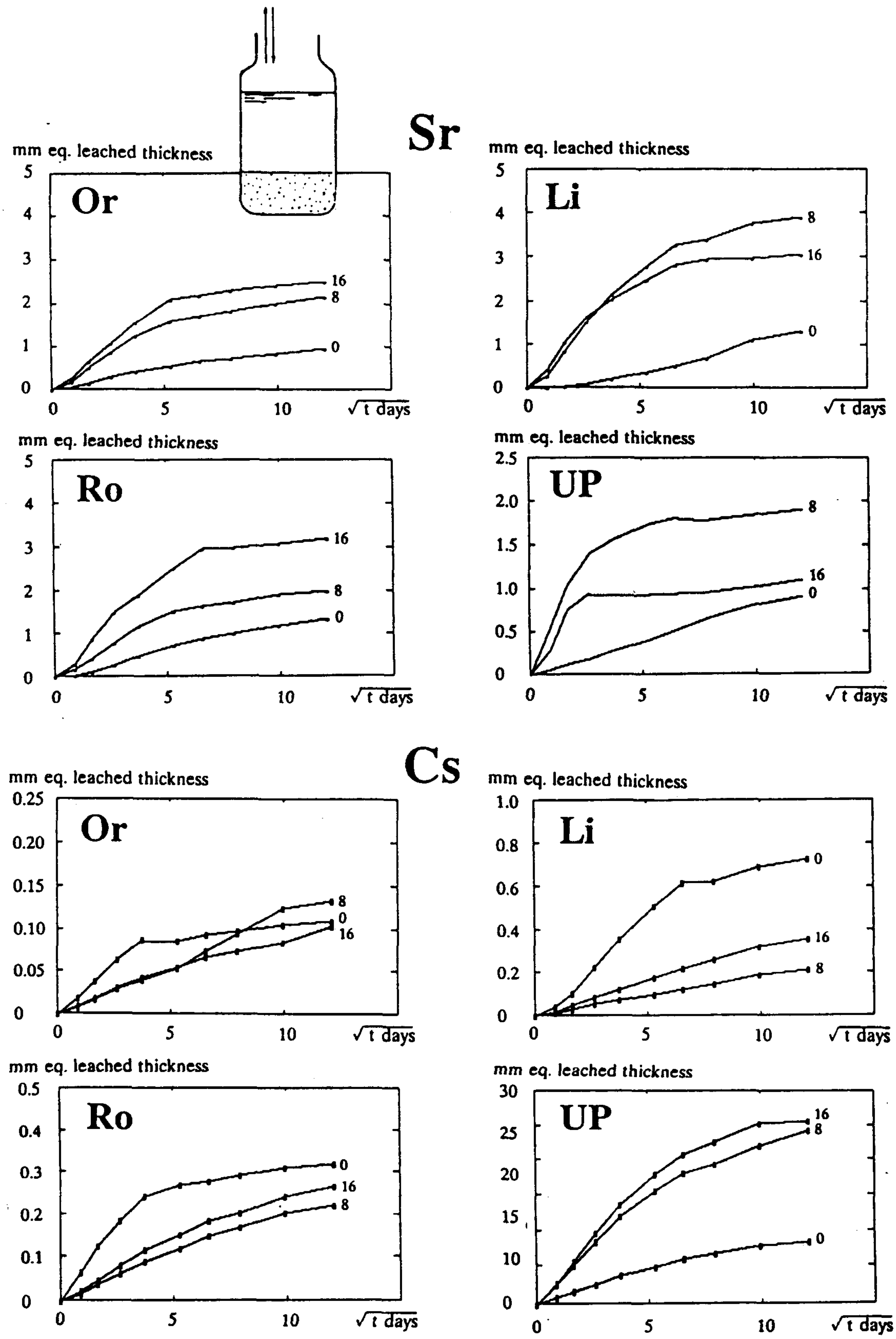


Figure. 2. Leach curves for ^{85}Sr and ^{134}Cs in 4 different types of soils mixed with 0 %, 8 % or 16 % cement and leached with 0.001 M NaCl with access to CO_2 from the atmosphere.

Table 3. Effective diffusion coefficients for leaching of ^{134}Cs and ^{85}Sr from different types of soils mixed with 0, 8 or 16 % cement. The values are obtained for the later part of the curves. The samples were leached with 0.001 M NaCl with access to CO_2 from the atmosphere. Some values for Ri soil obtained under various conditions are included for comparison.

Leach solution	Soil type	cement %	CEC meq /100g	at pH	$D_{e(leach)}$ unit: 10^{-15} m²/sec Cs	Sr
0.001 M NaCl	Or	0	13	6.7	0.08	31
		8			1.4	66
		16	27	~11	0.5	36
	Ro	0	33	7.6	7.9	64
		8			2.4	43
		16	50	~11	3.0	18
	Li	0	5	4.5	4.2	210
		8			2.4	130
		16	18	~11	5.8	12
	UR	0	7	4.4	1.9	69
		8			26	54
		16	30	~11	38	29
	UP	0	4	4.3	1900	59
		8			12000	11
		16	13	~11	16000	4.8
	Ri	0	15	7.4	0.0003	500
Deionized water	Ri open	8			0.33	24
		16	22	~11	26	140
	Ri closed	8			0.5	6400
		16	22	~12	35	5800
3 % NaCl ~0.5 M NaCl	Ri open	8			19	~1900
		16	22	~11	130	50
	Ri closed	8			24	13000
		16	22	~12	210	6100

5.2.9. Soil and cement interaction in barriers.

If soil or cement-solidified soil is used as barrier material around e.g. more strongly contaminated soil the rate of diffusion through the material is of relevance for safety assessments. This can be studied using classical diffusion cells or integral experiments.

5.2.9.1. Diffusion-cell experiments.

The effective diffusivity $D_{e(slab)}$ describing steady state migration through a slab is normally not identical with the diffusion coefficient $D_{e(leach)}$ obtained from leaching of the same material. Chemical retardation has in principle no influence on $D_{e(slab)}$ and the effect of pore structure is also different.

Values for $D_{e(slab)}$ can be obtained from the slope of the break-through curve using a simple version of Fick's first law:

$$D_{e(slab)} = \alpha \cdot c \cdot x/\Delta c$$

where α is the slope of the curve, c is the concentration in the strong solution and $\Delta c/x$ is the concentration gradient over the barrier.

A diffusion-cell experiment with 2 cm thick slabs made from cemented Ri soil has run for more than 3 years. The three slabs employed were tested for macro-defects by measurement of the hydraulic conductivity (the values given in Table 1) and were thereafter mounted in diffusion cells with a solution containing tritiated water (TOH) and $^{134}Cs^+$ on one side, and pure water on the other. The solution on the initially ianctive side was sampled regularly and analysed for tritium and ^{134}Cs.

The TOH break-through was very rapid with a flux corresponding to diffusivites between 87 and $114 \cdot 10^{-12}$ m^2/sec. A typical value for ordinary cement mortar is $3 \cdot 10^{-12}$ m^2/sec [13]. The higher values found for the cemented soil are an indication of the high porosity of these materials.

Cs break-through was only obtained for two of the samples. The diffusivities was found to be 0.6 and $0.2 \cdot 10^{-12}$ m^2/sec, respectively, which may be compared with $0.4 \cdot 10^{-12}$ m^2/sec for an ordinary cement mortar [13]. The last system showed nearly no Cs transport through the slab. This illustrates an often observed stochastic element in such diffusion experiments, and the values of the two cesium diffusion cofficients given above should therefore be regarded with some caution.

5.2.9.2. Integral experiments, transport through barriers.

In one type of integral experiments an originally inactive barrier layer is placed on top of a layer of contaminated soil. The system is water-saturated and the diffusive releases through the barrier is followed over a long period by sampling the free water on top of the barrier. 14 systems of this type have been studied.

An example of the set-up is shown as an insert in Fig. 2. A layer of active material consisting of moist Ri or Ro soil contaminated with ^{134}Cs and ^{85}Sr were covered by 3 cm thick barrier layers of inactive Ri or Ro soil, cemented Ri soil or ordinary cement mortar (w/c=0.38). The leachant was 0.001 M NaCl with 100 ml out of 200 ml substituted with fresh solution at each sampling. Some systems were closed and some had access to CO_2 from the atmosphere. In some cases the soil was sterilized by autoclaving to avoid internal generation of CO_2 in the systems.

Examples of release curves for Sr and Cs are presented in Figs. 3 and 5. The equivalent thickness of active material corresponding to the accumulative release is plotted versus time. A value for the barrier diffusivity cannot be estimated directly from the curves. Correct treatment requires that the results are fitted against a suitable diffusion model to obtain the relevant transport parameters.

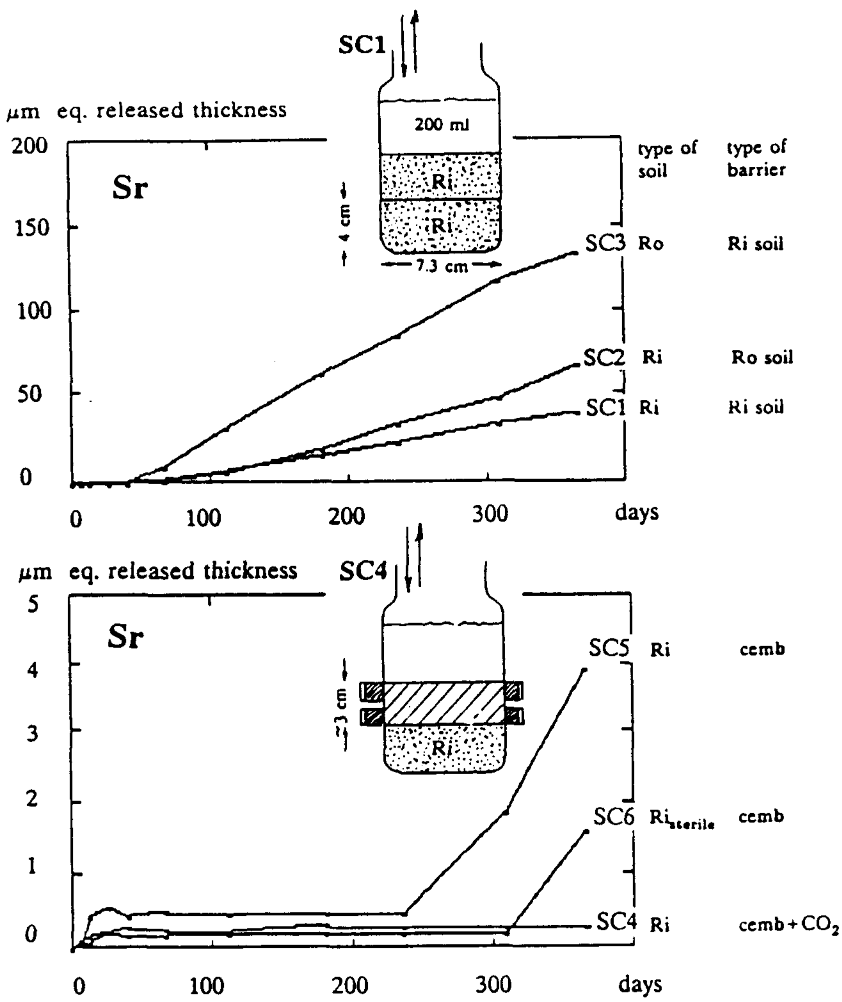


Figure 3. Release curves for ^{85}Sr for three systems with soil and three systems with cement mortar barrier.

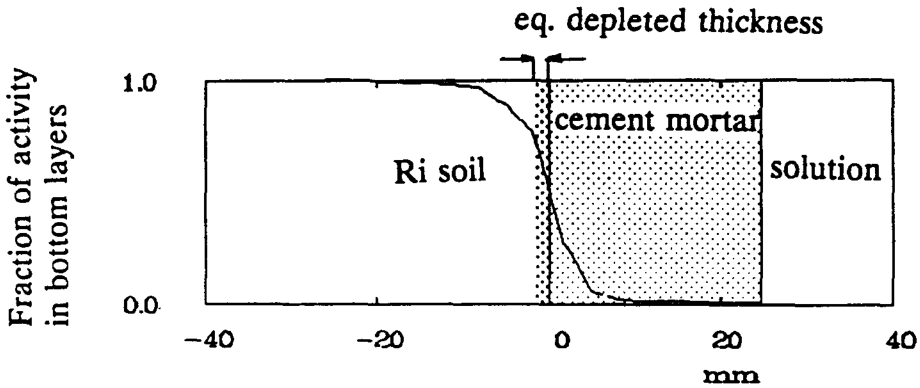
The fluxes out from the barrier surface are proportional to the slope of the curves. They give some idea about the rate of release at the end of the approximately one year long experiments, see Table 4. The releases does not comprise the spread of activity by diffusion into the originally inactive barrier layers. An idea about the concentrations profiles inside the systems were obtained by γ-scanning at the end of the experiment. The ^{85}Sr was too much decayed, but ^{134}Cs profiles are available [2], an example is shown in Fig. 4.

Table 4. Release rates for the later part of the integral experiments. Comparisons can be made between identical systems: SB2=SC1, SB6=SC4 and SB5=SC5.

Unit: μm equivalent thickness/year

System type	First set (387 days)		Second set (366 days)		
Sr					
Soil on Soil	SB2 o Ri/Ri 115		SC1 o Ri/Ri 48	SC2 o Ro/Ri 88	SC3 o Ri/Ro 150
Cemented soil on Soil	SB4 o CemRi/Ri 7	SB3 CemRi/Ri 28			
Cement mortar on Ri soil	SB6 o Cem/Ri 0.7	SB5 Cem/Ri 0.20	SC4 o Cem/Ri 0.03	SC5 Cem/Ri 10	SC6 s Cem/Ri 9
Cement mortar on Ro soil			SC7 o Cem/Ro -0.05	SC8 Cem/Ro -0.03	SC9 s Cem/Ro -0.01
Cs					
Soil on Soil	SB2 o Ri/Ri -0.15		SC1 o Ri/Ri 0.14	SC2 o Ro/Ri 0.09	SC3 o Ri/Ro 0.13
Cemented soil on Soil	SB4 o CemRi/Ri -0.18	SB3 CemRi/Ri 0.12			
Cement mortar on Ri soil	SB6 o Cem/Ri 0.60	SB5 Cem/Ri 0.46	SC4 o Cem/Ri 0.26	SC5 Cem/Ri 0.32	SC6 s Cem/Ri -0.02
Cement mortar on Ro soil			SC7 o Cem/Ro 0.22	SC8 Cem/Ro -0.01	SC9 s Cem/Ro 0.36

o indicates a system open to the atmosphere and s that the system was sterilized.



Figure, 4. Concentration profile measured by γ-scanning for ^{134}Cs in system SC4: Ri soil with cement mortar barrier, open. The equivalent depleted thickness corresponds to the activity present in the barrier plus activity released to the solution.

For soil with soil barriers comparison of the Sr release from the two identical systems SB2=SC1 (Ri soil with Ri soil barrier) does not indicate a high degree of reproducibility. Ro soil on top of Ri (SC2) is similar to Ri on Ri, but Ri on top of Ro (SC3) results in a considerably increased release rate.

The behaviour of Cs is more uncertain, but the releases in the first set of experiment appear to be higher by about a factor of 10 than in the second. However, this is entirely due to a curious sudden release of Cs which occurs after ~30 days, a phenomenon which is less pronounced but still visible in the later set of curves for these type of experiments (Fig. 3a). The long-term release rate for Cs is slight in all cases, about 0.1 µm/year. The presence of the Ro soil appears to give somewhat increased Cs releases. Systems with cemented soil barriers (SB3 and SB4) were only included in the first set of experiments. It appears to give releases between those of a soil barrier and a cement-mortar barrier.

For the soil with cement-mortar barriers the slope of the later part of the break-through curves for Sr was found to be slightly negative, indicating some re-absorption of initially released activity. Break-through may have occurred in case of SC5 and SC6 (see Fig. 3b). The reproducibility between the systems SB6=SC4 and SB5=SC5 are not too good.

The Sr release is likely to be reduced by the presence of CO_2, and this is the case for SC4 but not for SB6. The releases from the sterile systems SC6 and SC9 are initially somewhat less than from the corresponding biologically active systems. This could be an effect of the heat treatment during autoclaving of the soil. The two biologically active systems with Ro soil (SC7 and SC8) developed a slight overpressure during the experiment. The cement mortar barrier must have been rather gas tight and calcium carbonate precipitation must have occurred inside the barrier due to CO_2 from the soil.

For Cs a resonable value for the release rate through a cement mortar barrier of this type is about 0.3 µm/year. The influence of CO_2 appears to be slight and there is not much difference between the behaviour of the Ri and the Ro soil. The fixation of Cs in the Ri soil is possibly somewhat improved by the heat treatment during sterilization.

To some degree the above-mentioned experiments are inconclusive, mainly because they are too few and of too short duration. However, the relatively low release rates through the 3 cm thick barriers do not indicate a strong safety relevance of diffusive release through the much thicker concrete or soil barriers which would be used in practice.

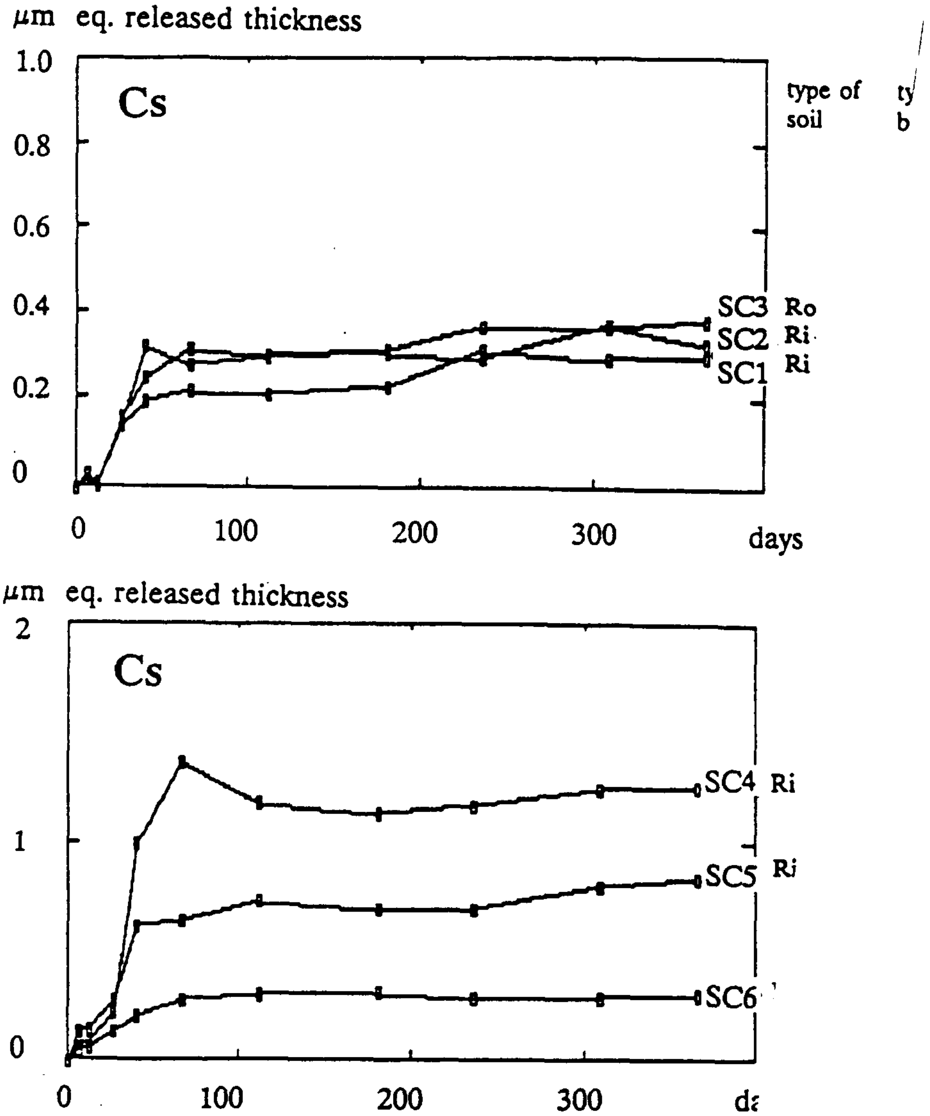


Figure. 5. Release curves for ^{134}Cs for three systems with a soil barrier and three systems with a cement mortar barrier. The equivalent released thickness is the layer of active soil corresponding to the accumulative activity found in the solutions.

5.2.9.3. Integral experiments, transport in soil near concrete.

Diffusive transport parallel to and near the surface of a concrete construction might also be of safety relevance. This was investigated by leaching experiments without barriers but with the solution chemistry influenced by the presence of a cement-mortar cylinder placed in the middle of the soil layer. This results in high pH and increased concentrations of cations in the system. $D_{e,leach}$ values are given in Table 5.

The Sr leaching is diminished by the presence of the cement cylinder especially for the open system (SC10) with access to atmospheric CO_2. The behaviour of the closed one (SC11) is approaching that of the pure soil system (SB1). The sterile system showed somewhat lower leaching. The Cs leaching is slightly increased by the presence of the cement cylinder especially in the sterile system.

Table 5. Effective diffusivities obtained from the later part of leach curves for ^{134}Cs and ^{85}Sr in layers of Ri or Ro soils and Ri soil with a central cylinder of type b cement mortar. The samples were leached with 0.001 M NaCl with access to CO_2 from the atmosphere.

Unit: 10^{-15} m^2/sec

	Pure soil systems		Soil layers with cement cylinder		
	SB1 o only Ri	Ro* o only Ro	SC10 o cyl/Ri	SC11 cyl/Ri	SC12 s cyl/Ri
^{85}Sr	500	64	48	220	48
	SB1 o only Ri	Ro* o only Ro	SC10 o cyl/Ri	SC11 cyl/Ri	SC12 s cyl/Ri
^{134}Cs	0.0003	8.0	0.0026	0.0014	0.05

* From Table 3.

5.2.10. Conclusions.

The following conclusions are drawn from the soil-cement interaction experiments:

Feasibility of producing cemented soil:

- Monolithic non-dusting blocks of cement-solidified soil can be produced from some sandy or clayey top soil using the GEODUR additive and a small amount of cement (8-16%). Soil with a high content of humic materials are problematic and test castings should always be made.

Properties of cemented soil:

- The compression strength is low and the volume stability on drying may not be satisfactory if the product is to be used as a barrier material.
- When produced from clayey soil the hydraulic conductivity is low although the porosity of castable products is high. Products with lower porosity can be made but must be handled as claylike plastic materials.
- Due to the high porosity, diffusion-controlled processes tend to proceed rapidly inside

the materials. Leaching is therefore also rapid unless ion-exchange or precipitation reactions inside the pore structure minimize the rate of transport.

- Cs is retained quite efficiently, but not as well as in the soil itself. Competition or blocking of the "Cs-specific" ion-exchange sites in the clay by K, Na or Ca ions from the cement is a probable explanation. The high concentration of alkali ions present in pore water in cementitious products is important and may prevent fixation of the Cs ions.
- Sr is leached quite easily from the cemented soil when CO_2 is absent, but the retention is considerably improved by carbonation of the material.
- Carbonation proceeds rapidly in the porous material as does the leaching of calcium from the structure. The capacity for maintaining a high pH pore solution is therefore limited with possible consequences for the retention of Sr or other radionuclides with a strongly pH- or carbonate-complex dependent behaviour.

In summary it is concluded that the incentive for cement solidification of Cs or Sr contaminated soil is rather limited. The additional expenses can only be justified when a monolithic structure is important. Safety gains in the form of improved retention of Cs and Sr radioisotopes cannot be expected. However, the situation may be different for other isotopes, for example the α-emitters.

Interaction between contaminated soil, high pH solutions, cement mortar and CO_2 from the atmosphere:

- The distribution behaviour of Cs and Sr ions in competition with Ca or K ions at high pH have been determined for the clayey Ri soil. Potassium was found to be more efficient than calcium for desorption of Cs ions.
- Sr is extracted efficiently from untreated Ri soil by equilibration with tap water (mainly a calcium bicarbonate solution with some Mg-, Na- and K-ions).
- Leaching of Sr and Cs from layers of untreated or cemented clayey soil is more efficient in a 3% NaCl solution than in deionized water (i.e. solution chemistry determined by the solids present). Only small amounts of Sr and Cs could be leached from untreated soil by a strong calcium hydroxide solution.
- Leaching from layers of five other types of soil were also studied. Their contents of organic material were higher and the clay contents lower. Solidified cemented products could not be made, but the leaching behaviour of the soft soil/cement mixtures compared with the pure soil shows some slight improvement in Cs retention when soils rich in organics are mixed with cement. This is also the case for Sr, but must be ascribed to CO_2 precipitation. The Cs retention in sandy soils with low clay content is poor, and gets even worse when the sandy soil is mixed with cement.

Interaction between contaminated soil and cement mortar barriers:

- The presence of concrete structures in contact with contaminated soil seems to improve the retention of Sr when CO_2 is available. The retention of Cs is somewhat decreased.

- The release rate for Sr through barriers of soil is relatively high, much higher than through a barrier of cement mortar. CO_2 is part of the explanation, but other mechanisms may also contribute to the high retention in cementitious materials.
- The release rate for Cs through barriers of soil is low, and releases through barriers of cement mortar is only slightly higher. The explanation could be effective retention in the contaminated soil even when it is in contact with the cementitious material.

The overall conclusion is that the modification of water chemistry caused by cement barriers are likely to influence the migration of radioisotopes in contaminated soil in contact with the barrier. The transport of Cs is somewhat enhanced while the transport of Sr is somewhat diminished, but only when CO_2 is present. The importance of the effects is dependent on the type of soil involved.

However, in most cases the changes are relatively minor, and concrete may well be an excellent barrier material for soil contaminated with Sr and Cs radioisotopes.

5.3. RELEASE OF Cs, Sr AND Eu FROM SOIL COLUMNS WITH DECAYING MATERIAL.

In large-scale accidents huge amounts of contaminated soil and vegetation may have to be handled and disposed of in a reasonably cheap and easy way. Simple near-surface burial without the use of concrete is the most likely solution (see Chapter 4), and it was therefore thought worthwhile to look for possible effects of decaying organic material on the migration of radionuclides in unsaturated soil.

Only a single set of 4 experiments has been performed and the results must therefore be regarded with some caution. The experiments are described in detail in [1].

5.3.1. Experimental method.

Four columns of the type shown in Fig. 6 were employed. The main part of the columns was filled with Ri soil mixed with an nearly equal amount of sand and - in three of the columns - with grass cut in about one cm long pieces. Grass was selected as an experimentally easily manageable sort of vegetation, and also as a type of waste likely to arise in large amounts in connection with cleanup activities after accidents. The sand was added to improve the permeability of the clayey Ri soil.

The columns contained 164 g soil mixed with 150 g sand. Column A_0 was a control without grass, in A_1 41 g grass (dry material) was present in the form of a separate layer in the upper part of the column, and A_2 and A_4 contained, respectively, 41 and 82 g grass mixed with the soil. The colums had a ~1 cm drainage layer of sand at the bottom and were topped by a ~1 cm layer of sand acting as a distributor for water added during the experiment. Further specifications are given in [1].

The soil, sand and cut grass needed for each column were mixed together in a mechanical mixer and then contaminated under continued mixing with 6 ml of a tracer solution containing ^{134}Cs, ^{85}Sr and ^{154}Eu. The activity was therefore not deposited primarily onto the grass, nor was it present in more or less particular form, as would have been typical for some real accident waste. The concentrations of inactive Sr, Cs and Eu were respectively

~100, 5 and $5 \cdot 10^{-6}$ mM/g soil + sand in the columns.

With the method employed the cationic tracers would mainly be fixed by ion exchange on the soil particles. Cesium and europium would be bound rather strongly while strontium would move more easily. A contributing feature in this connection is that the total carrier concentration (including the content naturally present in the soil) is considerably higher for Sr than for Cs and Eu.

Each column was closed by a polyethylene bag containing about 8 L air. A beaker with 50 ml 1 M NaOH was placed inside the bag for absorption of the CO_2 produced by the micro-organisms in the column. The air in the bag was replaced regularly with fresh air to maintain oxidizing conditions.

50 ml 0.001 M NaCl was added to each system on every weekday over a period of some months. The solution was introduced on top of the upper sand layer without opening the bag. The operation took about one minute and during the following few hours a slightly smaller amount percolated out through the bottom outlet.

The flow conditions are rather complex, with short (about one hour) drainage periods dominated by partial mixing and relatively rapid water flow and longer periods (~23 hours or more) where the conditions are stagnant and tending toward equilibrium determined by diffusion in the remaining pore water. The water samples obtained from the bottom outlets are therefore not in equilibrium with the bottom layer, nor can they be regarded as representing mean concentrations in the pore water down through the column. The procedure simulates in a coarse manner the effects of rain water percolating through unsaturated soil. However, the amount of water employed corresponds to a rain fall of 3190 mm/year, about 10 times what is typical for Denmark.

The percolates were weighed, their pHs were measured and the samples were analysed for ^{134}Cs, ^{85}Sr and ^{154}Eu by γ-spectroscopy, for K, Na and Ca by AAS and for organic material by the COD method. The activity measurements and some COD determinations were made on the percolates as such, and on samples filtered through 0.45 μm Millipore filters so that coarse impurities such as clay particles and micro-organisms were removed.

5.3.2. Percolates and respiration.

Each column had more or less the same dimensions and contained the same amount of soil + sand, and also the same amounts of the radioisotopes. Direct comparison between the leached amounts of radioisotopes is therefore meaningful.

The mean flow rate, when the amount of feed solution is distributed evenly over the sampling periods, varied between 1.45 and 1.52 ml/h. The corresponding outflow was in general about 20 % less. For systems A_0 and A_1 this must mainly have been due to evaporation, but for A_2 and A_4 the low values at the end of the experiments was due to clogging of the columns. These two experiments could only be run for a considerably shorter period than the other two. The reason for the clogging is thought to be the extensive growth of micro-organisms in the systems where the grass was mixed with the soil.

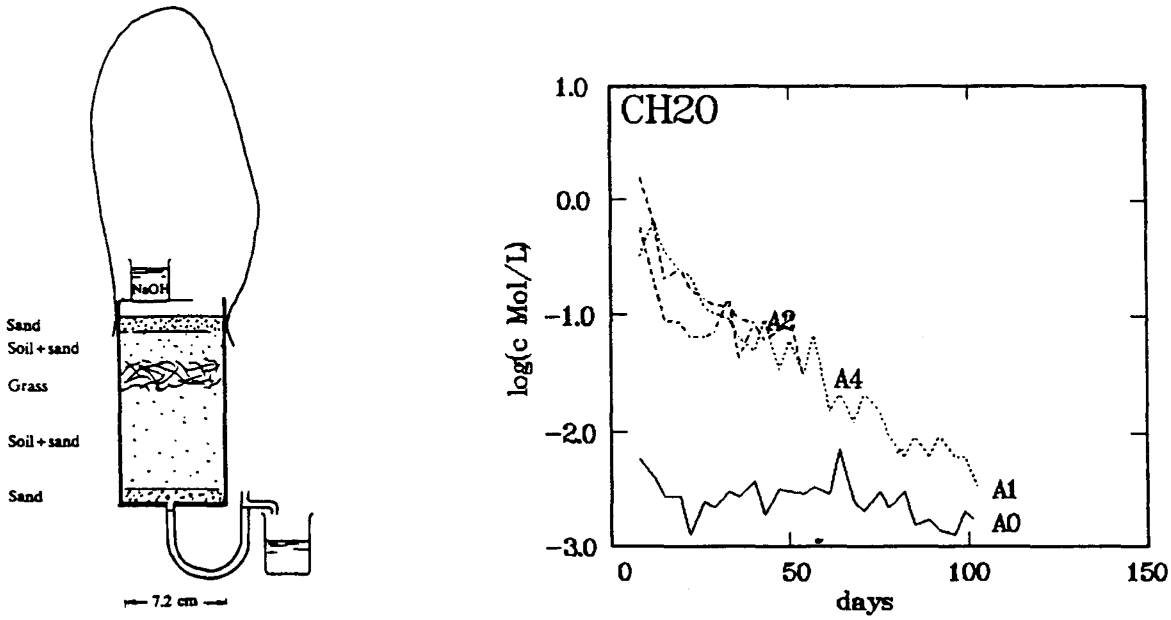


Figure. 6. The columns used in the experiments with leaching of radioisotopes from soil mixed with decaying organic material and curves for the concentration of organics in the pecolates.

The pH measured in the percolates varied from 8 for the pure soil and down to about 6 in the systems with grass. An initial drop in pH was followed by a return to 8 or even 9 followed by a new drop to values about 6.

In the systems with grass the concentration of potassium was high at the beginning, about ~0.01 M, but dropped after ~30 days towards the much lower values in the control system without grass. The fractional release of potassium reached 30 to 40 % in the initial rapid phase. Thereafter the release continued at a lower and nearly constant rate. The sodium concentration was mainly determined by the use of 0.001 M NaCl as the feed solution.

The behaviour of calcium was different. During the first ~20 days all four concentrations dropped from ~0.01 M to ~0.001 M, but then they suddenly increased again to ~0.01 M for the systems with grass. This occurred simultaneously with the second drop in pH mentioned above. The control system A_0 was losing calcium at a low and steady rate.

The COD analyses measured the amount of oxygen required for destruction of the organic material present in the percolates. The values are arbitrarily converted to an equivalent amount of carbohydrate as shown in Fig. 6. In reality the percolates contain many different and more complicated organic components including also bacteria and fungi. In some cases the percolates were also analysed after filtration through a 0.45 µm filter. It gave a reduction of only 10 to 20 % in the organic content, and most of the material must therefore be present as dissolved or colloidal degradation products.

Fig. 6 shows a large difference between the three systems with, and the one without grass. In the grass systems the concentrations are dropping very fast from values as high as ~1 CH_2O equivalent/l. The accumulative amount of organic material removed with the percolate is relatively modest, typically 10-15 % of the total dry weight of the grass.

The CO_2 releases from the top of the columns were followed by absorption in NaOH solution and titration. For the control system the respirative loss of carbon from the soil was about 5 times higher than the loss with the percolate. The opposite is true for the systems with grass, where the percolation loss were 4-5 times higher than the respiration loss. After the first ~20 days the rate of CO_2 release were nearly the same for all 4 systems.

5.3.3. Migration of radioisotopes.

The purpose of the study was to evaluate the possible influence of the decaying organic material on the downward movement of radioisotopes of strontium, cesium and europium.

The accumulative fractions of ^{85}Sr leached from the systems are shown in Fig. 7a. The curves are based on measurements of the unfiltered percolate. The Sr release from the control system A_0 is seen to be relatively low and constant and for this system there was no difference between the filtered and unfiltered percolate. The release from the systems with grass is very rapid especially at the beginning. For A_1 the accumulative release reached about 90% and similar values would no doubt have been obtained if continued operation of the A_2 and A_4 systems had been possible. The activity in the percolates was reduced some 20-30 % by filtering and some of the strontium must therefore be associated with rather coarse particles.

The results for ^{134}Cs are presented in a similar manner in Fig. 7b. It can be seen that the leached fractions (0.3 - 3 %) were much lower than for strontium. The Cs release was not influenced by the filtering procedure (discounting a single high value in the A_4 curve which no doubt was caused by a single or some few particles).

The behaviour of cesium in systems A_0 (no grass) and A_1 (a separate grass layer) is rather similar, while a considerable increase in releases can be seen for the two systems where the grass was mixed intimately with the soil. The increased releases occurred primarily during the initial 20-30 days, whereafter the slope of the curves for fractional release was levelling of, approaching the values for A_0 and A_1.

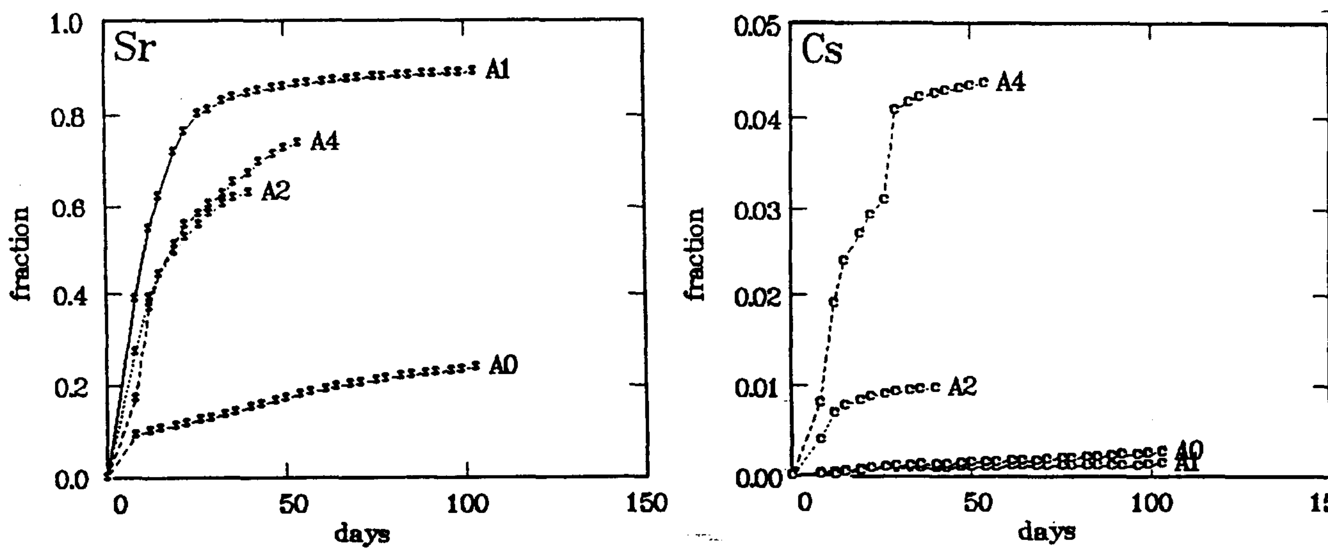


Figure 7. Accumulative fraction of ^{85}Sr and ^{134}Cs removed with percolate from the four columns.

The ^{154}Eu was bound very strongly to the soil resulting in percolate count rates too low to give reliable results. A tentative result is that there were no strong effects of the decaying organic material and that the slight releases appeared to be associated with particulate material.

Table 6 contains a summary of the total leached fractions for Ca, K and Na as well as for Sr, Cs and Eu based on the radioisotope measurements. The difference in behaviour of the radioisotopes described above are also seen here, but in addition it can be noticed that the fractional calcium releases are relatively low in systems A_1, A_2 and A_4 compared with the strontium results. It appears that the calcium is less affected by the presence of the decaying organic material.

Table 6. Duration of experiments, total feed solution and percolates and the fractions of the total contents of elements or radioisotopes found in the percolates from the four columns.

		A_0	A_1	A_2	A_4
		Control	Grass layer	Mixed with grass	
			41 g	41 g	82 g
Duration:	days	103	103	40	54
Feed solution:	ml	3650	3650	1400	1900
Percolate:	ml	2797	2553	802	1186
Fractional releases % of the total column contents found in the percolate	Ca	22	33	18	23
	K	48	47	38	32
	Na	59	19	22	7
	^{85}Sr	24	89	62	74
		24	66	53	44
	^{134}Cs	0.26	0.14	0.98	4.4
		0.09	0.11	0.97	3.4
	^{154}Eu	0.29	0.19	0.08	0.19
		0.08	0.09	0.03	0.02

The small figures are based on the analyses after filtering through a 0.45 μm filter.

After the experiment the two columns A_0 and A_1, which were functional through the whole period, were sectioned into layers and the contents of radioisotopes were measured to obtain the concentration profiles. A tendency to increased concentrations down through column A_0 was very pronounced for Sr, but could also be seen for Cs and probably Eu. In the last two cases it may be due to wash-down of clay particles in the column. In column A_1 concentration gradients could not be detected by the relatively coarse method employed.

5.3.4. Discussion.

Percolates with a high organic content are well known from ordinary municipal waste dumps and also from old shallow burial sites for radioactive waste. The conditions are often anoxic and the solution chemistry exceedingly complex. Percolates with increased organic contents will also occur in ordinary fields when for example a forest is cut down, or on a more limited scale after each autumn litter fall. In such cases oxic conditions may prevail.

Collection and near surface burial of large amounts of contaminated grass, leaves and other vegetable material, more or less mixed with soil, could be a part of the remedial actions after a large nuclear accident. The conditions inside such a burial site will probably be something between the two types mentioned above, largely determined by the size and thickness of the layers and of the load of organic material.

The column experiments reported here give some indication of the type of phenomena likely to be encountered in such a burial site:

- A percolate rich in organic materials and in general somewhat acidic is produced.
- High concentrations of Ca, Na and especially K will occur during the early period.
- ^{90}Sr present as contamination in the buried material will be leached and moved downward by the percolate at a rate much higher than in ordinary soil. The mechanism could be simple ion-exchange release caused by the increased cation concentrations in the percolate. Some fixation to moving relatively coarse particles may also contribute to this process.
- ^{137}Cs is much more strongly bound to the soil than the Sr-isotopes, but also for cesium there is - under some circumstances - a tendency to increased leaching when decaying organic material is present:

  When grass is mixed with soil there is an increase in the Cs-release corresponding to a factor 5 - 15 compared with pure soil. However, the phenomenon is mainly associated with the initial period, and the difference in release rate diminishes with time. The cesium in the percolate is in true solution or could be fixed on colloidal particles.

  The release caused by the organic materials appears to be a local phenomenon, because when the grass is present as a separate embedded layer - influencing the macro-chemistry in more or less the same way as when the soil and grass are mixed - no increased release from the system was observed. A possible explanation is that a weak complex or Cs-containing colloidal material are retained in the lower part of the column with soil, but this process is not as efficient in the other two systems where grass is present near the outlet.

  Experiments with diffusive leaching from layers of different types of soil indicate also that Cs is released more easily from soil with high organic contents, see Section 2.8.
- ^{154}Eu, acting as a stand-in for the α-emitters, is not seen to be influenced in any significant way by the decaying organic material. This could be due to the poor counting statistics, which may cover the possible effect. The low Eu-releases from the systems with soil mixed with grass confirm that the Cs-releases from the same

systems are not due to wash-out of small soil particles through the bottom outlet. This would have given similar results for Cs and Eu.

It is obvious from the effects found in the column experiments that undesirably contaminated percolate can occur when soil mixed with organic materials is buried. The activity concentration and the fraction leached in a given period will be high for ^{90}Sr and might also be unexpectedly high for ^{137}Cs.

The amount of percolate, and whether the contaminated solution flows to the ground water or to surface streams, will depend on the design of the burial system and the local conditions.

Comparison with preliminary results from the field tests described in Chapter 4 (Salbu et al) does not indicate that leaching of cesium is a problem. Results for strontium will unfortenately not be available from these field experiments, but there is no doubt that ^{90}Sr in buried soil represents a risk for ground water pollution.

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6. COSTS OF CLEANUP AND OF TRANSPORTATION AND FINAL DISPOSAL OF CLEANUP WASTE

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6.1. INTRODUCTION

To evaluate different cleanup methods the costs of their implementation have to be known. Evaluation can be done by either a cost-benefit analysis or by a more comprehensive justification-optimization analysis. Both these methods require a basic knowledge of the costs of cleanup methods. The tail-end of the cleanup procedure, transportation and final disposal of the ensuing waste, have been ignored in most previous analyses of this kind.

6.2. SCOPE OF THE STUDY

The following costs were calculated in this study:
- costs of several cleanup measures as a function of the area and the amount of waste
- costs of transportation of cleanup wastes as a function of the area, the amounts of waste and the transportation distance (for forest areas two cases are given: transportation within 50 km and transportation farther than 50 km)
- costs of final disposal of cleanup wastes
- in addition to the costs, the amount of working hours are also given

Cleanup of buildings were excluded from this work.

Most data for the calculations of the costs for urban and agricultural areas were received from the city of Espoo and the transportation company Espoon KTK Oy. Costs given below represent payment on a commercial contract basis to a company or a society. Therefore, these costs represent total costs, including also taxes, interest to the company etc.

When reading this chapter one should bear in mind that these calculations are examples of cleanup costs. In case of an actual cleanup process measures different from those concerned in these calculations may be taken and, therefore, also the costs are likely to be different.

6.3. URBAN AREAS (Salonen 1993a)

In a typical urban area the probable objects for cleanup are paved streets and parking lots, for which sweeping, firehosing and planing of asphalt can be used, and parks, where removal of vegetation and surface soil can be used.

6.3.1. Street sweeping

The most efficient way to clean up streets is to use both pressurised water cleaning and road sweepers. The whole width of a street (7-8 m) can be cleaned up by one treatment, if four road sweepers, which collect the dust and dirt, are followed in parallel by two cars with pressurised water cleaners. If the driving speed is 7 km/h, the area which can be cleaned up in one hour is about 5 ha. The volume of ensuing solid waste, mainly sand, is about 35 m^3, and the mass about 100 t. In order to keep the cars and sweepers cleaning continuously at least five trucks are needed to transport away the waste to a distance of about 20 km (8-8.5 m^3/load).

6.3.2. Firehosing streets

For street washing fire plugs, as well as, fire engines and tank trucks can be used. Fire plugs are usually located at distances of about 300 metres from each other. It is also possible, and also necessary outside population centres, to obtain the water with motor pumps directly from natural sources, such as lakes, rivers and seas.

The tank volume of a fire engine is 2000 l and that of a tank truck 12000 l. When washing the streets with normal pressure the water comsumption is about 300-400 l/min. At this speed a fire engine's tank is emptied in 5-7 minutes. The area which can washed with this amount of water is about 20x20 m when a wide water jet is used. If the tank of the fire engine is continuously reloaded with water from a tank truck, the fire engine can be used without reloading breaks. In this way 3400-4800 m^2/h can be washed and the water consumption is 18-24 m^3/h.

If the streets are cleaned using motor pumps or fire plugs, two men are needed to do the work. The working speed will be approximately the same as when sweepers and cars are used for cleaning. In this case the most important factor in the costs is the labour cost. It is not likely that the town would charge for the water.

Most of the water used for washing streets will go to rain water drainage systems, and finally to the sea through open drainage ditches. A smaller volume of wash waters will go through the drainage systems of buildings to waste water treatment plants. The fraction of waste waterwhich goes through the sewage system will vary from town to town and it is impossible to estimate the amount and activity of the effluents which end up in waste water treatment plants. Therefore, the activity of the waste waters has to controlled at the waste water treatment plants. If the activity in waste water sludge will be beyond a certain level, the sludge has to be transported for disposal as radioactive waste.

6.3.3. Planing asphalt

The thickness of asphalt pavements is normally 16-17 cm, from which a 1-5 cm layer can be planed for the removal of the contamination. The speed of a planer is 1 ha/15 h. The volume of the apshalt increases by a factor of 1.5 due to the planing and the the amount of the removed asphalt will thus be 150-750 m^3/ha. Loading onto trucks for the transportation is carried out continuously.

6.3.4. Removal of vegetation

In urban areas removal of vegetation - trees, bushes and grass - will be mainly used for cleaning up parks and gardens. In parks there are on average less than a hundred trunks per hectare and the amount of timber is about 100 m^3/ha. Felling, pruning, cutting and stacking by the roadside can be carried out by wood processors or harvesters in easy terrains. In difficult terrains the work has to be done manually by forest workers. Felling, cutting, pruning and stacking of trees from an area of one hectare takes 8 hours for a wood harvester, which is operated by only one man. The same area would take an unaided forest worker 55 hours to do. The removal of bushes can be done using a brush saw.

Pruned and cut trees and bushes can be transported away either in their natural state or as chipped. Trunks up to 20 cm in diameter can be chipped using a tractor-powered chipper. The result (m^3/h) will largely depend on the speed of trunk feed into the chipper, but it can be estimated to be 10-15 m^3/h.

For cutting grass it is sensible to use a mower with collecting tank and a movable platform, into which the collecting tank can be emptied. The working width of the mower is usually 1.2 m and the volume of the collecting tank 300-400 l. The speed of the mower is 6 km/h. When cutting grass of normal height the tank will get full every half a hectar and it will take some 45 minutes including the emptying of the tank.

6.3.5. Removal of soil

In urban areas the removal of upper soil layer can be used in parks and on gravel roads. Tractor excavators can be used for removal and loading and trucks for transportation. This kind of work is typically undertaken on contract basis and the unit prices (Table 1) include removal, loading and transportation of the waste within a distance of 20 km.

Removal of a 10 cm layer of soil corresponds to 1000 m^3/ha of dense soil and 1500 m^3/ha of loose soil. Removal of a 5 cm layer yields 750 m^3, correspondingly. The unit price of the soil removal is dependent on whether the work has to be done in easy or difficult conditions. Easy conditions mean, that the terrain is easy, and the ground is dry and resistant. Difficult conditions mean, that the terrain is difficult and the ground is wet and breakable. Because of the contract work nature of soil removal, the number of working hours needed can be estimated only for the entire process.

Table 1. Average costs (ECU/m^3 and ECU/ha) various cleanup measures and transportation of cleanup wastes in urban areas.

		Cleanup costs		Transport. < 20 km	
		ECU/m^3	ECU/ha	ECU/m^3	ECU/ha
Road sweeping		6.1	43	4.6	30
Road washing					
- tank truck or fire engine			340	7.6	
- fire plugs			53		
Asphalt planing					
- 1 cm		58	8760	3.8	570
- 5 cm		11			2860
Removal of trees					
- harvester		11	1140	4.6	460
- forest worker		23	2290	4.6	460
Chipping		2.3	230	4.6	460
Removal of bushes			76	4.6	
Cutting grass		44	30	4.6	3.2
Scraping of soil[x]					
- easy conditions	(5 cm)	7.6	5330		
-	(10 cm)	3.8			
- difficult conditions	(5 cm)	30	22900		
-	(10 cm)	15			

x) include transportation costs

When the transportation distance is greater than 20 km the transportation costs per cubic metre are calculated as follows: 3.8 ECU + 0.2 ECU for each kilometre more than 20 km.

Table 2. Amounts of working hours needed for cleanup and transportation of waste in urban areas.

		Cleanup (manh/ha)	Transportation < 20 km (manh/ha)
Sweeping roads		1.2	1
Washing roads			
- fire plug		5	
- tank truck or fire engine		10	
Planing asphalt			
- 1 cm		45	30
- 5 cm		45	75
Removal of trees			
- harvester		8	7
- forest worker		55	7
Chipping		8	7
Removal of bushes		5	7
Cutting grass		1.5	
Scraping of soil[x]			
- easy conditions	(5 cm)	100	
	(10 cm)	175	
- difficult conditions	(5 cm)	430	
	(10 cm)	750	

x) include transportation costs

6.4. AGRICULTURAL AREAS (Salonen 1993a)

In the following, cases in which 3 cm, 5 cm or 10 cm layer have to be removed from a field with an area of 100x100 m will be considered. After removal of the vegetation with a harvester soil will be scraped from the field, piled and loaded onto trucks for the transportation to a final disposal site.

For the removal of soil layer a tractor excavator will be used and, if the conditions are good, i.e. the ground is dry and resistant, the soil can be piled into one 100 m long pile. Removal and piling of field soil increases the the volume of the soil by 30 % and the volume to be transported away will be 1300 m^3/ha (10 cm), 650 m^3/ha (5 cm) and 390 m^3/ha (3 cm).

If the conditions are difficult, i.e. the ground is wet and breakable, the soil has to be piled into two 100 m long piles, in cases of 5 cm and 10 cm soil layer removal. If only 3 cm layer is to be removed, only one pile is needed. Due to the breakable ground a 3 m wide transportation road has to be constructed alongside the piles. The road costruction costs are made up of the cost of filtration fabrics and a 30 cm thick gravel layer.

Table 3. Average costs of removal and transportation of soil from fields.

	Easy conditions		Difficult conditions	
	ECU/m^3	ECU/ha	ECU/m^3	ECU/ha
Scraping of soil				
- 3 cm	4.3	1680	5.9	2290
- 5 cm	2.6	1680	3.5	2290
- 10 cm	1.3	1680	1.8	2290
Loading				
- 3 cm	0.5	180	0.6	240
- 5 cm	0.5	300	0.6	400
- 10 cm	0.5	590	0.6	790
Transportation				
- 3 cm	3.8	1490	8.5[x]	3310[x]
- 5 cm	3.8	2480	9.4[x]	3130[x]
- 10 cm	3.8	4950	6.9[x]	8610[x]

x) costs include the construction of roads

When the transportation distance is greater than 20 km the transportation costs per cubic metre are calculated as follows: 3.8 ECU + 0.2 ECU for each kilometre more than 20 km.

Table 4. Number of working hours needed for the removal of surface soil from fields.

	Easy conditions (manh/ha)			Difficult conditions (manh/ha)		
	3 cm	5 cm	10 cm	3 cm	5 cm	10 cm
Scraping	25	25	25	35	35	35
Loading	7	11	22	9	15	30
Transportation (< 20 km)	32	54	108	32	54	108
Road construction	-	-	-	13	26	26
Total manhours	64	90	155	89	130	199

6.5. FOREST AREAS (T.K.Ikäheimonen 1993)

Costs, manhours and machine-hours, required to carry out the cleanup measures described in chapter 2 (Cleanup methods) for forests, are presented in Tables 5 and 6.

Three examples of costs and an estimation of the amounts of men, machinery and working time to carry out a cleanup process are presented in Appendix 4.

Table 5. Machine-hours and costs of different steps of a cleanup procedure for various types of forests.

	trees		stumps	
	h/ha	ECU/ha	h/ha	ECU/ha
Felling and cross-cutting				
pine	8	500	3	170
spruce	14	870	5	290
deciduous	9	580	3	210
Local tractor transport (~250 m)				
pine	12	490	2	110
spruce	20	820	5	180
deciduous	15	610	3	140
Chipping and loading				
pine	3	460	3	120
spruce	5	840	5	200
deciduous	4	520	3	140
Short-distance transport ~50 km				
pine	10	560	2	90
spruce	19	1050	3	140
deciduous	12	640	2	110
Long-distance transport				
pine	18	980	3	150
spruce	34	1800	5	260
deciduous	20	1100	4	200
Removal of undervegetation and soil[1]		730		
Loading of undervegetation and soil[1]		460		
Transport of undervegetation and soil[1]		1220		

1) costs have been calculated for an average amount of undervegetation in forests of the Nordic Countries and for the removal of 10 cm upper soil surface layer

Table 6. Man-hours needed for the cleanup (h/ha) in different types of forest. Calculations are based on the amounts of biomasses in forests in the cenntral areas of the Nordic Countries.

	pine	spruce	deciduous
Felling and cross-cutting of trees	8	16	10
Local tractor transport of trees	13	23	17
Chipping and loading of trees	3	5	4
Transport inside the affected area (<10 km)	7	13	8
Long-distance transport (200 km)	20	37	23
Harvesting and logging of stumps	4	6	4
Local transport of stumps	4	6	4
Chipping and loading of stumps	4	6	4
Transport inside the affected area (<10 km)	4	6	5
Long-distance transport (200 km)	7	13	8
Removal of undervegetation and soil	18	18	18
Loading of -"-	11	11	11
Transport inside the affected area (<10 km)	38	38	38

6.6. FINAL DISPOSAL (Salonen 1993b)

6.6.1. Description of the final disposal options considered

To calculate the final disposal costs and the amounts of manhours to carry out the final disposal the following example cases were considered:
a) final disposal in surface mounds
b) final disposal in shallow ground trenches
c) final disposal in natural valleys or basins.

In the calculations there were two subcases for each of the above mentioned disposal options: the volume of the waste is either 10,000 m^3 or 50,000 m^3. In an actual accident case the amounts of waste to be disposed of in the final disposal facility, as well as the type of the facility will depend on many factors, which cannot be predicted accurately in advance. Therefore, the procedures and calculations given below, should be considered as suggestive.

All these final disposal options include same procedures. These are:

- First the waste, brought by truck to the site, will be spread and compacted with a caterpillar.

- Thereafter the waste will be covered with a 0.5 m layer of clay and 0.3 m layer of soil. This will be done with a caterpillar. A bucket loader and two trucks are needed to move the soil and clay to the site. The clay layer will function as a barrier against the infiltration of rain water and vegetation roots into the waste and the soil layer together with vegetation functions as an erosion barrier.

- Optionally a plastic sheet can be placed between the clay and soil layers to more efficiently prevent the infiltration of surface waters. Two men are needed to place the sheet manually.

- The disposal sites should also be underdrained. In the case of a surface mound the depth of the underdrain should not be more than 0.5 m, but in the case of trench disposal the underdrain has to be below the bottom of the waste layer. In the case of the valley or basin the depth of the underdrain depends on the type and depth of the underlying soil. Underdrainage must, however, be above the highest ground water level in the area. Underdraining requires one trench cutter and two men with spades.

There are, however, some special measures, which are characteristic for only one or two of these three final disposal options. These are described in the following:

a) Final disposal in surface mounds (Fig. 1): The angle of inclination of a surface mound, when touching to the ground, is in optimally about 20°. If the mound contains 10,000 m^3 (values for 50,000 m^3 volume are in paranthesis) of waste the mound will be 60 m (100 m) wide, 104 m (160 m) long and 4.8 m (8.1 m) high including the clay and soil layers above the waste. The volumes of these protective layers are 3240 m^3 (8890 m^3) and 2280 m^3 (5810 m^3), respectively. The need for underdrain will be 336 m (528 m) and in addition about 100 m to direct the drainage water into ditches.

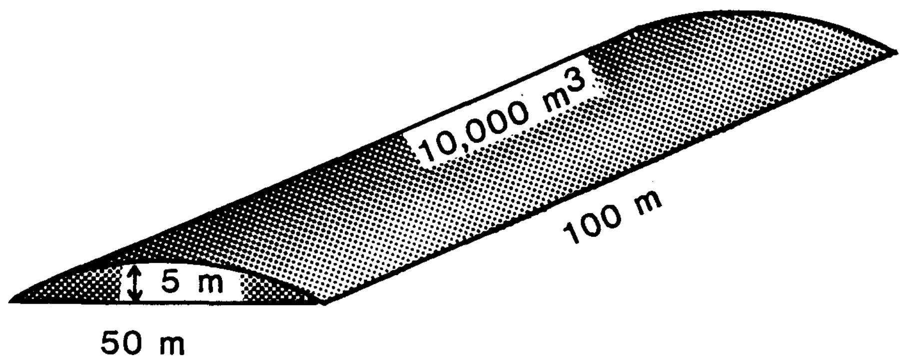


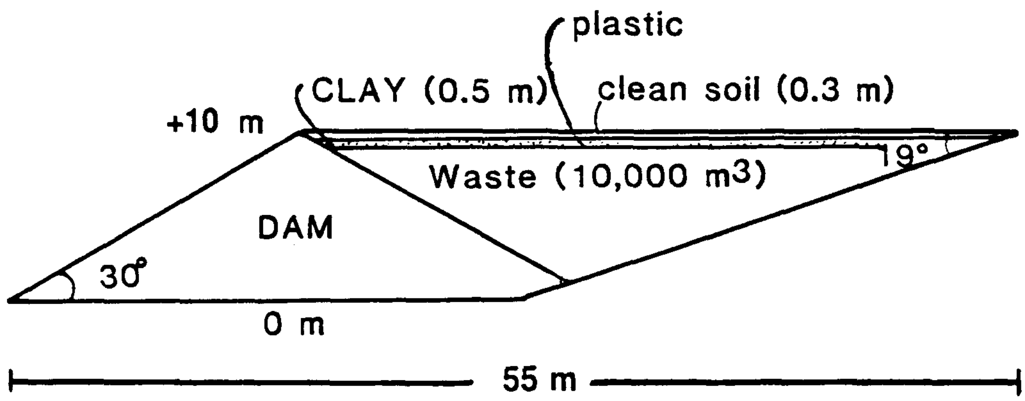
Figure 1. Disposal of cleanup waste in surface mound.

b) Final disposal in shallow ground trenches: The trenches cannot be very deep in the Nordic Countries, since the ground water level is rather high. At its highest the level in Finland is one metre below the ground. Therefore, a safe depth for a shallow ground disposal trench would be 0.8 m. If the final disposal site is located in a field, the depth can also be 0.8 m, since the underdrain pipes are at a lower level. Construction of the trench starts with the digging of the 0.8 m soil layer. This should be done in two phases. First, a 0.3 m fertile soil layer is removed and stored separately from the subsequently removed 0.5 m layer, which is in many cases clay, especially in the coastal areas. This clay and soil will be later used to cover the waste. Thereafter, the trench will be filled with waste and covered with clay and soil layers. If the volume of the waste is 10,000 m^3 (50,000 m^3), the size of the trench has to be 50 m x 72.5 m (90 m x 129 m), which means that 29 % (19 %), 2900 m^3 (9300 m^3), of the waste will be below the ground. The rest will be above the ground as a mound, the shape of which is the same as in case a). The amounts of clay (0.5 m) and soil (0.3 m) removed when the trench was constructed, are not sufficient to cover the waste mound. 27 % (23 %) of the needed clay and 38 % (30 %)

of soil have to be brought from other sources. The size of the ensuing mound will be 60 m x 82.5 m (100 m x 139 m) and the height 4.8 m (8.1 m). The need for the underdrain pipes will be 293 m + 100 m (486 m + 100 m).

c) Final disposal in natural valleys and basins: In these calculations it is assumed that there is a natural valley, which has a shape of a truncated cone with one side open. To make it suitable for the disposal of cleanup waste the valley has to be closed with a dam. For the waste volume of 10,000 m^3 the dam has to be 8 m high (15 m for 50,000 m^3). The length of the dam ridge should be 100 m (150 m) and the volume of the dam 6800 m^3 (43,900 m^3), when the inclination of the wall is 30°. Before costruction of the dam about 0.3 m layer of the surface soil from the valley should be removed and stored for use as the upper protective cover. Trees and bushes, if any, are also removed. The inner wall of the dam should be covered with a 0.5 m layer of clay. After placing the waste into the dammed valley, it should be covered with a 0.5 m layer of clay, plastic sheet (optional) and a 0.3 m layer of soil. The need for the underdrain pipes will be 330 m (525 m).

a)



b)

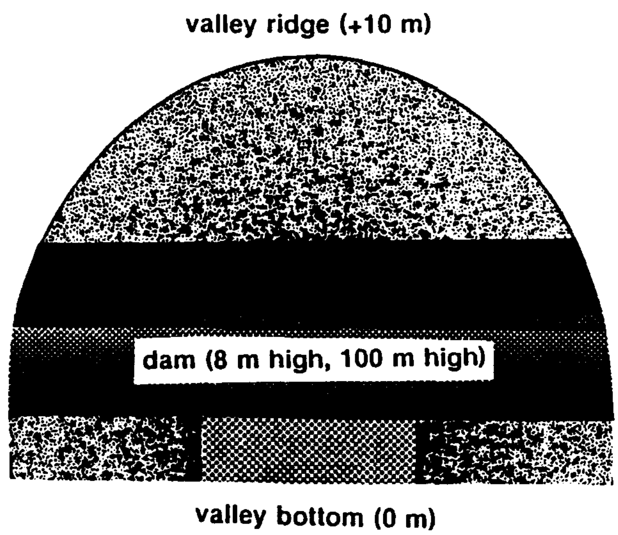


Figure 2. Disposal of cleanup waste in a natural valley, closed with a dam. a) cross section, b) plan view.

6.6.2. Costs and number of manhours

Tables 7 and 8 give the costs and number of manhours required to construct the final disposal facilities described above. As seen, depending on the amount of waste to be disposed of the shallow ground disposal is 10-20 % more expensive than disposal in surface mounds. Disposal in natural valleys is very expensive if large dams have to be constructed to close the valley. It is advantageous to construct large disposal mounds or trenches, e.g. construction of a 50,000 m^3 surface mound instead of five 10,000 m^3 mounds reduces costs by 39 %. When disposing of the waste in a natural valley increasing the amount of waste does not reduce the costs, since the construction costs of the dam, which are the main source of costs, increase correspondingly.

At the time these calculations were made, there was a severe economic depression in Finland, which certainly has caused a considerable decrease in the prices used in these calculations.

6.7. REFERENCES

Ikäheimonen, T.K. and Korhonen, E., 1993, Removal and Transportation of Forest Biomass After a Nuclear Fallout (in Finnish), NKS-KAN2 Report, 32 pages.

Salonen, P., 1993a, Costs of Pretreatment, Transportation and Final Disposal of Radioactive Waste from the Cleanup of Areas Contaminated in Nuclear Accidents (in Finnish), Part 1, NKS-KAN2 Report, 13 pages.

Salonen, P., 1993b, Costs of Pretreatment, Transportation and Final Disposal of Radioactive Waste from the Cleanup of Areas Contaminated in Nuclear Accidents (in Finnish), Part 2, NKS-KAN2 Report, 16 pages.

Table 7. Costs of different final disposal options.

	Surface mound		Shallow trench		Natural valley	
	$10000m^3$	$50000m^3$	$10000m^3$	$50000m^3$	$10000m^3$	$50000m^3$
	ECU	ECU	ECU	ECU	ECU	ECU
Scraping of surface layer[1]	-	-	-	-	1,200	3,800
Transportat. of soil[2]	16,800	44,800	18,300*	55,200*	51,800	334,000
Construction of dam	-	-	-	-	5,200	33,400
Spreading of waste[3]	7,600	38,000	7,600	38,000	7,600	38,000
Spreading of clay[4]	2,500	6,800	1,900	5,800	1,000	3,100
Spreading of soil[5]	1,700	4,400	1,400	3,800	610	1,900
Plastic sheet	2,600	6,900	1,700	5,900	1,400	4,300
Underdraining	3,300	4,800	4,200	6,200	2,500-3,500	4,000-5,600
Total costs	34,500	105,700	35,000	115,000	71,300-72,400	423,000-425,000

* include construction of the trench

1) Scraping of 0.2 m surface soil layer from a valley.
2) Truck transportation of soil used to cover the waste and to construct the dam.
3) Spreading and compacting the waste.
4) Spreading and compacting 0.5 m clay layer above the waste.
5) Spreading and compacting 0.3 m soil layer above the waste and clay.

Table 8. Number of manhours for different final disposal options.

	Surface mound		Shallow trench		Natural valley	
	$10000m^3$	$50000m^3$	$10000m^3$	$50000m^3$	$10000m^3$	$50000m^3$
	manhours	manhours	manhours	manhours	manhours	manhours
Scraping of surface layer[1]	-	-	-	-	23	71
Transportat. of soil[2]	264	705	226*	881*	735	4740
Construction of dam	-	-	-	-	109	702
Spreading of waste[3]	160	800	160	800	160	800
Spreading of clay[4]	52	142	40	121	21	66
Spreading of soil[5]	36	93	28	80	13	40
Plastic sheet	25	66	17	57	13	41
Underdraining	70-105	100-150	63-95	94-140	53-80	84-126
Total manhours	607-642	1906-1956	600-630	2033-2080	1125-1155	6543-6585

* include construction of the trench

1) Scraping of 0.2 m surface soil layer from a valley.
2) Truck transportation of soil used to cover the waste and to construct the dam.
3) Spreading and compacting the waste.
4) Spreading and compacting 0.5 m clay layer above the waste.
5) Spreading and compacting 0.3 m soil layer above the waste and clay.

7. COST-BENEFIT ANALYSIS ON CLEANUP OF RADIOACTIVE CONTAMINATED URBAN AREAS

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Investigations have been made of the cost and benefit of practicable reclamation and decontamination procedures for the cleanup of radioactively contaminated urban areas, taking into account the costs of transportation and disposal of the generated radioactive waste. In combination with calculations of the first year dose from different urban surfaces to people living in four different environments of varying population density, a strategy for cleanup after a contamination with precipitation has been developed. It was concluded that cleanup of green areas and streets is relatively highly cost-effective and would rank highly on a list of priorities, while cleanup of other surfaces would not be worthwhile in any of the examined scenarios.

7.1. INTRODUCTION

The ultimate goal of any reclamation/decontamination study is the provision of a nuclear contingency plan for reclamation/ decontamination, in this case for the urban environment.

In developing such a strategy, a wide range of factors need to be considered in order to provide the most cost-effective strategy for a given contamination scenario.

Some of the important factors to be considered in the development of such a strategy are:

1) The distribution of the deposited material with respect to the different outdoor surfaces.

2) The contribution of the different surfaces to the radiation dose in the area, as a function of time.

3) The decontamination or dose reduction achievable on the individual surfaces using the appropriate methods.

4) The evaluation of practicability and estimates of total costs of the different reclamation/decontamination procedures.

7.2. DEVELOPMENT OF A STRATEGY

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 components (e.g., walls, paved areas, roofs, grassed areas, etc.) can be recognized and the individual contributions to dose of each of these components will depend on their surface area, the energy of the radiations and the degree of shielding.

As part of the procedure for reducing radiation dose to the populace of a given urban or suburban area, it is necessary to first determine the physical characteristics of the area in some detail. For instance the size of the buildings, the thickness of the walls, the types of roofs, the extent of grassed and paved areas, the amount of trees, etc. need to be determined.

From a knowledge of the prevailing weather conditions during deposition, and of the content of the radioactive plume, the relative distribution of the deposited radioactive matter (in the urban environment the radionuclide of primary concern will be ^{137}Cs) on the different surfaces can be estimated (Roed, 1988a). An effective source strength can then be defined. Wet and dry deposition will, for instance, give different effective source strengths for the different surfaces in the urban area.

Having defined the source strength, the next step is to calculate the relative dose rate in different locations (indoors and outdoors) due to deposition on the different urban surfaces (roofs, walls, paved areas, trees, bushes, etc.). As these dose rate contributions change with time it is imperative to have a sufficiently detailed knowledge on how the deposited radioactive matter will migrate with time in an urban complex. Following the Chernobyl accident numerous field measurements of contamination on different urban surfaces have been carried out. From this data it was possible to estimate the dynamics of the radiocaesium retention / migration loss processes which typically occur in a contaminated urban environment. For this purpose the computer model URGENT (URban Gamma Exposure Normative Tool) has been developed.

The mean relative dose over a period of time (for instance the first year after contamination) to a member of the local populace can then be found taking into consideration the time that he will spend in the different locations in the environment.

The next step is to estimate the decontamination or dose reduction factors achievable for the different surfaces and from this to find the relative source strength after decontamination.

It is then possible to recalculate doses at the different locations and show the reduction in dose achievable through cleanup.

From the total costs of the various decontamination procedures and the corresponding achievable reduction in dose, the cost of a relative reduction of for instance the first year dose (e.g., 1 %) can be calculated. The data obtained will then indicate the most cost-effective means of dose reduction.

7.3. COST-BENEFIT ANALYSIS FOR CLEANUP: WORKED SAMPLE

In the following an example will be given of the development of a strategy for cleanup in four different urban or suburban areas. The areas consist of:

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

The total cost and efficiency of different decontamination procedures for cleanup of caesium contaminated areas will be evaluated in the following:

7.3.1. Roads

Where the equipment is available, vacuum sweeping of roads is a fast and inexpensive way of obtaining a decrease in the radiation level. According to the mode of deposition and the time at which the method is implemented, experiments have shown that a decontamination factor (DF - defined as the relation between contamination density before and after decontamination) of 1.7 - 2 can be expected. According to calculations made by Salonen and Lehto (1993), the total costs including transportation and final disposal at a distance of less than 20 km of the collected waste will be about 0.01 ECU per treated square meter.

As an alternative, firehosing of roads in the urban area might be considered. Here, similar figures to those by vacuum sweeping have been found experimentally for the DF. Where fire plugs are available, the method would only be slightly more expensive than the vacuum sweeping procedure. According to Salonen and Lehto (1993), the total costs amount to less than 0.02 ECU per treated square meter. However, if tank trucks or fire engines are used, the costs are estimated to about 0.06 ECU per m^2.

Another method which has been investigated for decontamination of roads is planing. This is a very effective method, as the upper layer of the road containing, and to some degree embedding, the radioactive matter is removed. The decontamination factor is very large, probably more than 20, but the method is expensive to carry out. According to Salonen and Lehto (1993), the cleanup costs alone would be in the order of 1 ECU per m^2. In addition to this, the transportation costs of generated waste would be about 0.4 ECU/m^2. The final storage costs would be in the same level. Also, the road would probably need further treatment to be usable after the operation.

7.3.2. Gardens and parks

Immediately following dry contamination, grass cutting can be very effective in removing much of the contamination on grassed areas (Roed, 1988a). The total costs would be low, probably about 0.005 ECU per treated square meter. It has been found that the transport process from grass to soil has a half-life of about 7-18 days (Krieger and Burmann, 1969).

As the downward migration of caesium in all undisturbed soils is exceedingly slow, the scraping of the top 5 cm of soil will be an effective way of decontaminating garden and park areas for years following contamination. The effect will depend on the time at which the procedure is carried out, the characteristics of the soil and to some extent the deposition mode, but a recent experiment made in the Chernobyl 30 km zone more than 7 years after the accident gave a DF of about 5. The problem is the large amounts of waste generated, and the total costs of the procedure including disposal of the waste has been estimated to about 0.6 - 3 ECU per m^2.

Digging up gardens might be an attractive alternative. Anyway, this would not generate any radioactive waste. The costs of digging the gardens would probably be about 0.2 - 0.5 ECU per m^2, depending on the cost of a man-hour. In a grassed garden area in Pripyat close to Chernobyl an experiment was recently carried out to investigate the effect of digging up gardens in a special, but uncomplicated way. The procedure was approximately the same as that which is performed by a trench plough, but can be performed very easily in urban areas, using only a shovel. The principle is shown in Figure 1. First, a two-spit deep trench with two levels is dug (1). Then the layer 3 is cut loose and placed in the bottom of a trench 1, with the turf facing down. Finally, the uncontaminated layer (2) is placed on top of layer 3. The procedure can be repeated. The procedure was found to give a dose reduction of more than 80 %.

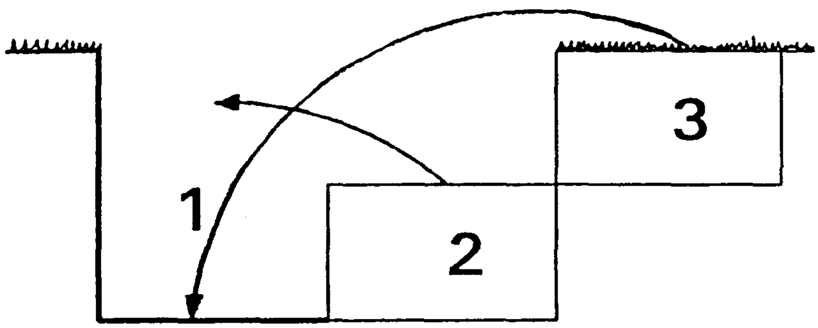

Figure 1. Sketch showing the principle of the special method which was followed for digging up a grassed area near Chernobyl.

Trees and bushes in garden areas have been shown to be effective interceptors of airborne particles (Roed, 1988b), and in the case of dry deposition they may contribute a highly significant fraction of the total dose to the local populace. Since trees in the vicinity of houses are often at window height, there is little attenuation of gamma rays entering the building since a thin sheet of glass gives much less shielding than bricks. The decontamination factor will be in the order of 10-50, according to the deposition mode. The costs of removal by forest workers of trees have been estimated to about 7 ECU per square meter covered by trees. The costs will almost be halved if it is possible to use a harvester for the operation (Salonen and Lehto, 1993).

7.3.3. Buildings

Decontamination of buildings will generally be relatively expensive to carry out, but in some cases, especially in dry deposition scenarios, the dose reduction can be substantial, although the decontamination factors are small compared with those found for countermeasures on other surface types. For roofs and walls, ion-exchange methods might be considered but due to practical difficulties associated with this procedure, the generally recommended method would be firehosing. The values given in Table 1 for this procedure for dry deposition are the results of in situ tests carried out in Pripyat near Chernobyl in the summer of 1992. The values given for wet deposition are the results of in situ tests on tiles and bricks in the Gävle area in Sweden.

Table 1. Achievable Urban Dose Reduction Factors and Costs of Dose Reduction in an Urban Area Contaminated by Wet/Dry Deposition.

Surface type	Roofs	Walls	Streets	Trees	Gardens
Efficiency (DRF):					
Dry Deposition	1.9	1.9	2	50	10
Wet Deposition	1.5	1.2	1.7	10	8
Costs : (ECU m^{-2})	2	0.8	0.01	7	0.5
Of this, the costs of transportation and final disposal of generated waste amount to :					
(ECU m^{-2})	0.1	0.1	0.003	2	-

By combining the data in Table 1 with computer estimates, made with the URGENT code, of the dose contributions from different types of surface to the total first year external dose to people living in one of the four investigated environments previously described it is now possible to estimate the actual costs and benefit of a cleanup programme. This has been done below for wet and dry contamination cases. The achievable percentage dose reduction by cleaning of different surfaces in the four types of environments of different population densities is given in Tables 2 and 3. The Tables also show estimates of the costs for each person living in the areas for a per cent reduction of the dose. These data indicate the most cost-effective means of dose reduction.

Clearly, the introduction of dose reducing countermeasures for gardens would be beneficial. In fact, in the suburban areas, the reclamation of garden areas alone would yield a dose reduction factor of more than 4 after a wet deposition of caesium. As the cost for this dose reduction would also be relatively low, cleanup of garden areas should be given first priority in all scenarios from a view point of cost-effectiveness. The achievable reductions of the dose from a wet caesium deposition on other surfaces in the urban and suburban environments are small. However, as the costs for street cleaning are very low, this procedure should be given second priority. In cases of wet contamination of cities, treatment of other surfaces would not be cost-effective, as walls and trees make relatively small contributions to the total dose and they are expensive and difficult to decontaminate. Wet deposition on roofs may contribute significantly to the population doses, but due to the lack of sufficiently cost-effective dose reducing procedures for such surfaces, decontamination of roofs would be given a low priority.

As shown in Table 3 for dry deposition scenarios, grassed garden areas are also here the most important decontamination target, and the contamination on trees will make the second largest dose contribution. In this case, cutting back trees as well as street vacuum-sweeping is cost-effective in terms of dose reduction per unit cost. The dose contribution from roofs and walls can be large, but decontamination of such surfaces should only be carried out in special cases.

Table 2. Estimates of cost and benefit of different procedures for cleanup in a wet-contaminated urban area.

Environment with single-storey detached suburban houses:

	Roofs	Walls	Streets	Trees	Gardens
% dose reduction by cleaning the different surfaces:	2.8	0.1	-	1.7	78.3
Costs per person per % dose reduction [ECU]:	22.50	218	-	237.36	1.01

Environment with semi-detached houses in two stories:

	Roofs	Walls	Streets	Trees	Gardens
% dose reduction by cleaning the different surfaces:	3.8	0.1	-	1.2	76.2
Costs per person per % dose reduction [ECU]:	16.57	458.08	-	160.87	0.59

Environment with rows of two-storey terrace houses:

	Roofs	Walls	Streets	Trees	Gardens
% dose reduction by cleaning the different surfaces:	2.2	0.1	5.6	1.5	68.1
Costs per person per % dose reduction [ECU]:	29.05	260.44	0.06	79.73	0.73

Environment with five-storey blocks of flats:

	Roofs	Walls	Streets	Trees	Gardens
% dose reduction by cleaning the different surfaces:	0.2	0.1	10.2	1.0	64.2
Costs per person per % dose reduction [ECU]:	179.47	145.27	0.02	6.28	0.16

Table 3. Estimates of cost and benefit of different procedures for cleanup in a dry-contaminated urban area.

Environment with single-storey detached suburban houses:

	Roofs	Walls	Streets	Trees	Gardens
% dose reduction by cleaning the different surfaces:	5.2	4.1	-	23.7	42.3
Costs per person per % dose reduction [ECU]:	16.21	7.43	-	11.97	1.87

Environment with semi-detached houses in two stories:

	Roofs	Walls	Streets	Trees	Gardens
% dose reduction by cleaning the different surfaces:	8.8	1.0	-	24.1	45.4
Costs per person per % dose reduction [ECU]:	10.71	45.31	-	7.12	1.05

Environment with rows of two-storey terrace houses:

	Roofs	Walls	Streets	Trees	Gardens
% dose reduction by cleaning the different surfaces:	6.0	1.1	4.2	23.9	42.7
Costs per person per % dose reduction [ECU]:	29.05	31.88	0.02	3.71	1.09

Environment with five-storey blocks of flats:

	Roofs	Walls	Streets	Trees	Gardens
% dose reduction by cleaning the different surfaces:	0.4	1.6	7.8	22.3	43.1
Costs per person per % dose reduction [ECU]:	55.92	47.33	0.01	0.22	0.24

7.4. CONCLUSIONS

The principles of development of a nuclear contingency plan for the cleanup of contaminated urban and suburban areas were demonstrated. A worked example was given. The data obtained from field measurements after the Chernobyl accident made it possible to model the behaviour of radionuclides (especially radiocaesium) when deposited under different weather conditions in urban areas. For wet- and dry-contaminated cities, calculations of the first year doses to the population were made using this computer model. A further analysis was made including the costs of carrying out cleanup procedures and of transportation and final disposal of the generated radioactive waste. This cost-benefit analysis for the wet deposition scenarios revealed that decontamination of gardens should be given first priority and street-cleaning second priority in any cleanup programme. Cleanup by simple means in the grassed areas could in some cases reduce the total external dose by almost 80 %. Treatment of other surfaces would not be worthwhile for any of the four examined wet deposition scenarios. In the dry deposition scenarios, the cutting of trees was found to be efficient and least costly in the urban centres. Again, the contribution from grass was found to be large, and the streets would also be worth doing something about, considering the low cost of vacuum-sweeping, but decontamination of roofs and walls would probably give too little benefit compared to the costs involved. Such calculations are valuable in pinpointing where cleanup would yield the maximum benefit.

7.5. REFERENCES

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8. DECISION ANALYSIS OF PROTECTIVE ACTIONS IN FOREST AREAS

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A nuclear accident itself and the introduction of protective action entails risks to the people affected, monetary costs and social disruption. As far as the society is concerned the values which enter decisions on protective actions are multidimensional. People have strong feelings and beliefs about these values, some of which are not numerically quantifiable and do not exist in monetary form. These problems, often including mutually conflicting objectives and uncertainties and are difficult to control simultaneously, cannot be undertaken without careful consideration of the essential consequences of decisions. Decision analysis can be applied in planning intervention, this helps in rendering explicit and apparent all the factors involved and evaluating their relative importance. In this study recovery operations to clean up a forest environment in the event of a hypothetical radiation accident in a nuclear power plant were analyzed and discussed to determine what would be appropriate intervention levels in protecting the public, workers and the environment. The values considered essential in the decision were included in the analysis and their importance on the decision making process is discussed.

8.1. INTRODUCTION

Situations in which the radiation sources, the pathways and the exposed individuals are already in place when the decisions on remedial actions are considered, are called *intervention* situations. In these situations the doses which are received or are likely to be received can only be reduced by remedial actions. The basic principle when implementing protective actions is that intervention should be *justified* and *optimized*, i.e., the introduction of a protective measure should achieve more good than harm and the net benefit should be maximized (IAEA91, ICRP91). Decision analysis is a suitable method for helping to solve societal problems of this type.

Research over the past 30 years has transformed the abstract mathematical discipline of decision theory to a potentially useful technology known as *decision analysis*, which can assist decision makers to handle large and complex problems together with their attendant flow of information. Decision analysis is not intended to solve problems directly. It's purpose is to produce insight and understanding. In the light of that understanding the decision maker can make better decisions. Those interested in the theory of decision analysis may consult the literature (Fr88, Go92, Ke76, Wi86). This report provides an application of how decision analysis can be used when planning protective actions.

As initially presented the background information is generally limited or incomplete in decision making. A careful analysis of the problem indicates what further information is needed to find the best course of action. Thus the aim of this study was not only to find the best protective actions, but also to indicate the information that should be catered for or revised. If in the light of revised information or gained insight, new feasible actions are identified, the analysis should be revised.

The following analysis deals with protective actions for contaminated forest areas. The actions which most probably have to be taken on cultivated or natural foodstuffs, were excluded, although they might have had an effect on the analysis. Because of the high contamination levels considered in this study there would certainly be restrictions on the use of natural foodstuffs.

8.2. ACCIDENT SCENARIO

For the purpose of the analysis it was assumed that a hypothetical accident had happened at a nuclear power plant in Finland leading to a core melt and to a very severe -presumably worst possible - contamination of the environment (cf. Chapter 3). 10% of fission products and 1% the transuranics were assumed to have released from a 700 MW BWR reactor. It was further assumed that the accident had happened in summer time and there had been only dry deposition. As a consequence of the accident the forest areas given in Table I were contaminated.

Table I. Fallout area in forest land and the contamination levels after a hypothetical reactor accident.

Nuclide	Area I 1.5 km^2	Area II 22 km^2	Area III 1660 km^2
^{137}Cs mean	> 100 MBq/m^2	10-100 MBq/m^2 20 MBq/m^2	1-10 MBq/m^2 2 MBq/m^2
^{90}Sr mean	> 77 MBq/m^2	7.7-77 MBq/m^2 15 MBq/m^2	0.8-7.7 MBq/m^2 1.5 MBq/m^2
^{239}Pu mean	> 22 kBq/m^2	2.2-22 kBq/m^2 5 kBq/m^2	0.2-2.2 kBq/m^2 0.5 kBq/m^2

8.3. CONCERNS AND ISSUES

From the radiation protection point of view the aim of protective actions is to reduce the individual as well as the collective doses to the public and workers, and also to reduce radiological impacts on the environment. Concerning the forest, the aim is also to keep the area in, or to bring it back into production by feasible decontamination.

Intervention will affect the exposure pathways, and it should be carefully considered, that the total detriment of the population is reduced and, e.g., the dose is not reduced in one group by increasing it in another group. For example, the decontamination and the reduction of exposure for the population can be achieved only by increasing the doses to workers, who are carrying out the intervention measures.

The use of dose limits as the basis for the deciding on intervention might involve actions that would be out of all proportion to the benefit obtained and would thus conflict with the principles of justification and optimization. ICRP therefore recommends against the application of dose limits or any predetermined limits for deciding on the need for intervention. However, the position of workers carrying out recovery operations is different. These actions, none the less even when they are in response to an accident, can be planned and optimized in advance and therefore it is recommended that workers undertaking recovery operations should be subject to the normal system of radiological protection, and dose limits should be applied.

The intervention measures also entail non-radiological risks to the population and the workers caused by various kind of accidents. The risks which are directly associated with remedial actions should be taken into account when making decisions about intervention. In addition, there might be radiological risks caused, for example by forest fires. Fires would result in the resuspension of radionuclides and a larger area than that contaminated without the fire would be contaminated. An increasing number of individuals would be subject to radiation.

Psychological stress could lead to health effects of a comparable nature to those arising from the contamination, while at the same time reducing the quality of life significantly. A majority of the population in a contaminated area may show varying degrees of stress reactions, but stress could also be a consequence of protective actions. Stress can be reduced by taking appropriate actions, such as actions which decrease the dose of population, but at the same time this would lead to an increase in exposure among the intervening workers.

Perceived risk, in addition of health effects, can have serious economical and social consequences, e.g., to the forest economy and industry. Public opinion and the perception of risk could result in consequences which reduce the benefit of actions or make their implementation impossible. For example, in a limited accident the population (and the forest industry) might not accept products made from wood grown in the contaminated area although, e.g., only the bark and branches of the trees were contaminated and the contamination could be removed very efficiently. The industry might think that the risk of being discredited by using contaminated materials was too great, and thus refuse to use even slightly contaminated raw raw materials.

Individual people and families own 75% of forests in Finland and the average area of a forest estate is 0.35 km^2. Thus, there would be nearly 5000 private forest estates in the contaminated area considered in this study. Land owners would be worried about their property and incomes, and so there would be considerable stress within this group of people.

The fallout would reduce the value of contaminated land for decades, but it would also reduce the value and the quality of the surrounding areas. The reduction in quality of the environment would take place also in the vicinity of disposal sites and around power stations burning radioactive wood.

Remedial actions would cause monetary costs to the individual land owners, industry and the society. The costs would include transportation costs, loss of income, costs of the control of area and costs of lost capital services. Also the question of reimbursig land owners for any remedial actions would arise. If any compensation is paid, either in full or in part, it means that the costs to individuals would now be costs to the society. None the less, would the cost of the actions be a limiting factor? The economic impact of an accident may not be entirely negative. The activities may have a positive effect on the economy, such as a generation of employment or the production of energy by burning wood produced in the contaminated area.

8.4. DECISION MODEL

8.4.1. Action alternatives

The essence of decision analysis is to break down complicated decisions into small components that can be dealt with individually and then recombined logically. The process of breaking something down into its constituent parts refers to the process of developing an overall analytic structure. The formulation of the problem is *to identify what can be done and what might happen as a consequence*. In this process construction of a *decision table* or a *decision tree* is a very helpful method. Figure 1 shows a decision tree used to analyze the remediation strategies of contaminated forests. Decision tree compactly represents a set of scenarios. Any path from left to right through the tree constitutes a scenario. We will discuss in more detail these scenarios when discussing the strategies which can considered for cleaning up the contaminated forest.

One main stage in the decision analysis is to identify the alternative courses of action. In considering an intervention, all feasible actions should be defined - including no action. When defining an action, its feasibility should also be considered; could it be implemented in practice as it has been planned? For instance, it should be taken into account that society is not neutral to the choice of action. The remedial actions which were considered in this study are no action, control of wood material, control of access and removal of various parts of vegetation, i.e., trees, stumps, undervegetation and/or soil.

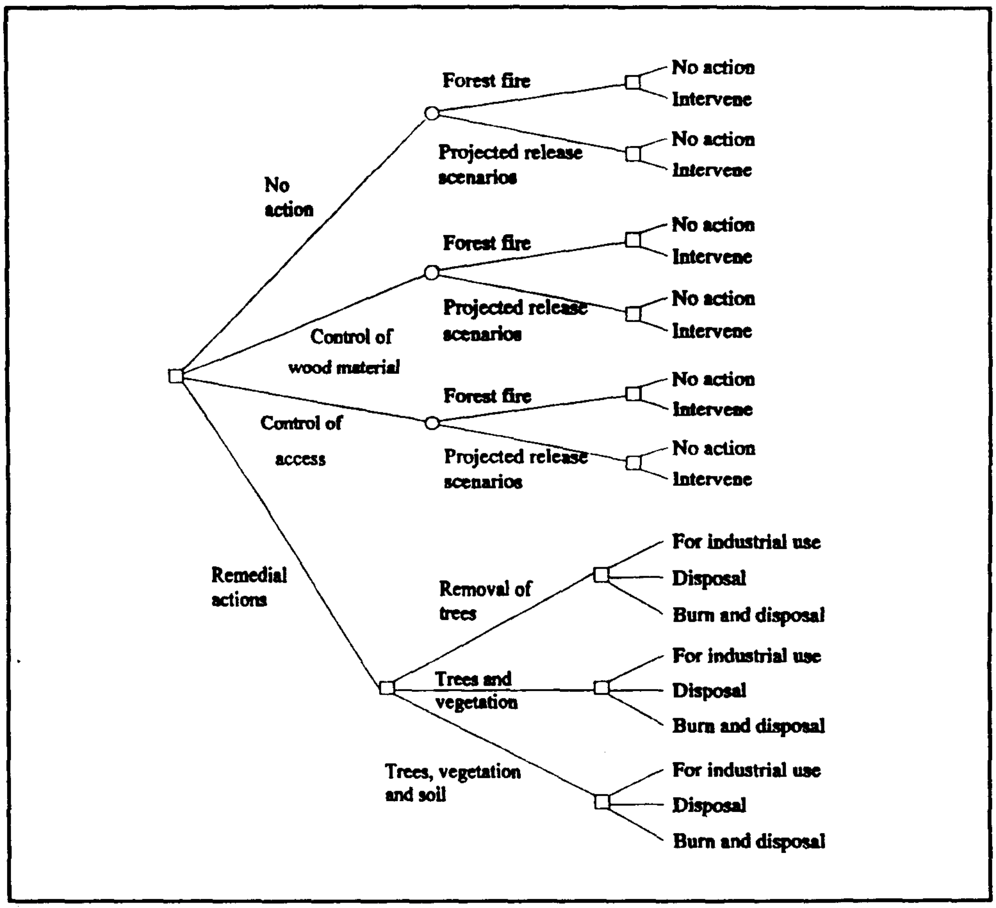

Figure 1. Decision tree to analyze the cleanup strategies of contaminated forest.

If in any of the defined areas (I, II or III in this analysis) no recovery operation, control of wood material or control of access is taken, the contamination is left in full in the forest. The amounts of radioactive materials will decrease with time through radioactive decay and by resuspension, which will cause transfer of radionuclides also to habilitated areas exposing unidentified people (projected release scenarios). The resuspension over 70 years is estimated to be 15%. Also, if the use of contaminated wood material is not restricted, its use will cause transfer of radionuclides to the living environment. It is estimated, that the total amount of transfer in this pathway will be 30% over 70 years.

A forest fire will cause a spread of radionuclides. According to the statistics there are a few hundred forest fires in Finland every year in which 0.05 km^2 of forest is burnt on average. The probability, that a forest fire would occur in the contaminated area is less than 0.01 in a year. By assuming that 50% of radionuclides in the forest will be released in a fire, gives the result that the expected collective dose to the public would be very low (a few ten's of mmanSv) as compared to the other pathways. Thus a forest fire is not an important scenario when considering the actions to be taken.

Decontamination of forest could be done by removing trees, stumps, undervegetation and/or soil. If all these are removed after two years it is estimated that 20% of radionuclides will remain in the contaminated area. When only trees and undervegetation are removed after two years the cleanup efficiency is estimated to be 60% in practice. During the first season the radionuclides are mostly in canopies, and by removing the trees the practical efficiency of decontamination is estimated to be 50%.

Based on the information mentioned above six strategies as defined in Table II were considered.

Table II. Strategies for recovery operations in forest areas defined in terms of their effects on the areas I, II and III.

Strategy	Removal of trees [a]	Removal of trees and undervegetation [b]	Removal of trees, undervegetation and soil [b]	Control of access [c]	Control of wood material	No action
1	II		I	I, II		III
2	II	I		I, II		III
3		I, II		I, II		III
4				I, II		III
5				I, II, III		
6			I	I, II	III	

a) Action is taken during the same season.
b) Action is taken two years after the fallout.
c) Projected period for control of access is 70 years.

Actually many more strategies could be considered by combining the areas and possible decontamination strategies. The limiting the analysis to those above would, however, not reduce the possibilities for evaluating the best course of action. Some strategies are not even feasible, e.g., it is not possible to remove the trees in area I during the same season, because the individual doses would be unacceptable high; the dose rate would be one mSv/h. Furthermore, as is indicated in the decision tree, there are three different methods to treat contaminated and removed trees:

1. Industrial use of trunks in sawmills or in pulp industry. Disposal of branches and barks as such or as ash after burning as fuel.
2. Chipping the trees, branches and stumps and burning the chips in power stations. Disposal of the remaining ash.
3. Disposal of trees as they are or in the chipped form.

These optional methods will be discussed below in more detail.

1. Industrial use. The method can not be easily applied. In principle, if trees are barked in the same or in the following season, the wood itself will be clean and the activity will be mostly still in the bark and in the branches. However, only big trees are barked in sawmills nowadays. There is a lack of machines suitable for this action to be done in the forest. Also, the distribution of contamination during the barking process would be unacceptably high.

If the trees are used in the chemical pulp industry, the pulp will be clean because during the process the radionuclides will be removed and they will remain in the waste sludge. The contamination of machines would be a problem.

The most serious problem is public opinion. Although it could be shown that the products made of contaminated material would be free from radioactivity, the industry and the population would in all probability not accept them. The public can be very suspicious about this kind of products as was demonstrated after the Chernobyl accident. Also, clean wood material would be available. Thus, the industrial use of contaminated trunks has to be rejected as a strategy.

2. Burning of wood before disposal. There are small (5 MW) and large (20 - 200 MW) power stations in every Nordic country, which are suitable for burning chipped wood material. If proper electrostatic precipitators are used, 95-99% of radionuclides will remain in the ash. The amount of waste ash to be disposed would be small. An ash content of 5% has been used in the calculation. However, there will be some suspicion of burning radioactive material, especially among the population in the vicinity of the power stations.

3. Disposal of all material as such. Undervegetation, litter, humus and soil have to be disposed as such. Wood material can also be disposed as it is, but chipping the trees, branches and stumps, however, will help in the disposal and rotting of material. In all cases, final disposal cannot be undertaken before the organic material is rotten.

Burning of wood before disposal and the disposal of all material as such are the two optional methods considered in the analysis.

8.4.2. Objectives and attributes

Having found the courses of action, the next main step is to identify all attributes (measures in Figure 2) relevant to the decision. One technique to identify an operational set of attributes, is to start by listing all the important objectives (goals in Figure 2), such as minimizing health detriment, monetary cost and social disruption. In order to check the list, the objectives can be divided into general categories: health, safety, social, political, psychological and economical effects. Many of these objectives will necessarily be part of the decision making process following radiological emergencies. Some of the objectives might be directly measured on a numerical scale and some should be further divided into sub-objectives in order to be measurable. This kind of numeric variable is called an attribute. An attribute is used to measure the performance of actions in relation to an objective. Natural attributes are, e.g., immediate deaths, cancer cases or reduction in lifespan. An *attribute hierarchy* (*value tree*) can be useful in defining attributes and objectives. Figure 2 shows the value tree for the remedial operations for the problem in hand.

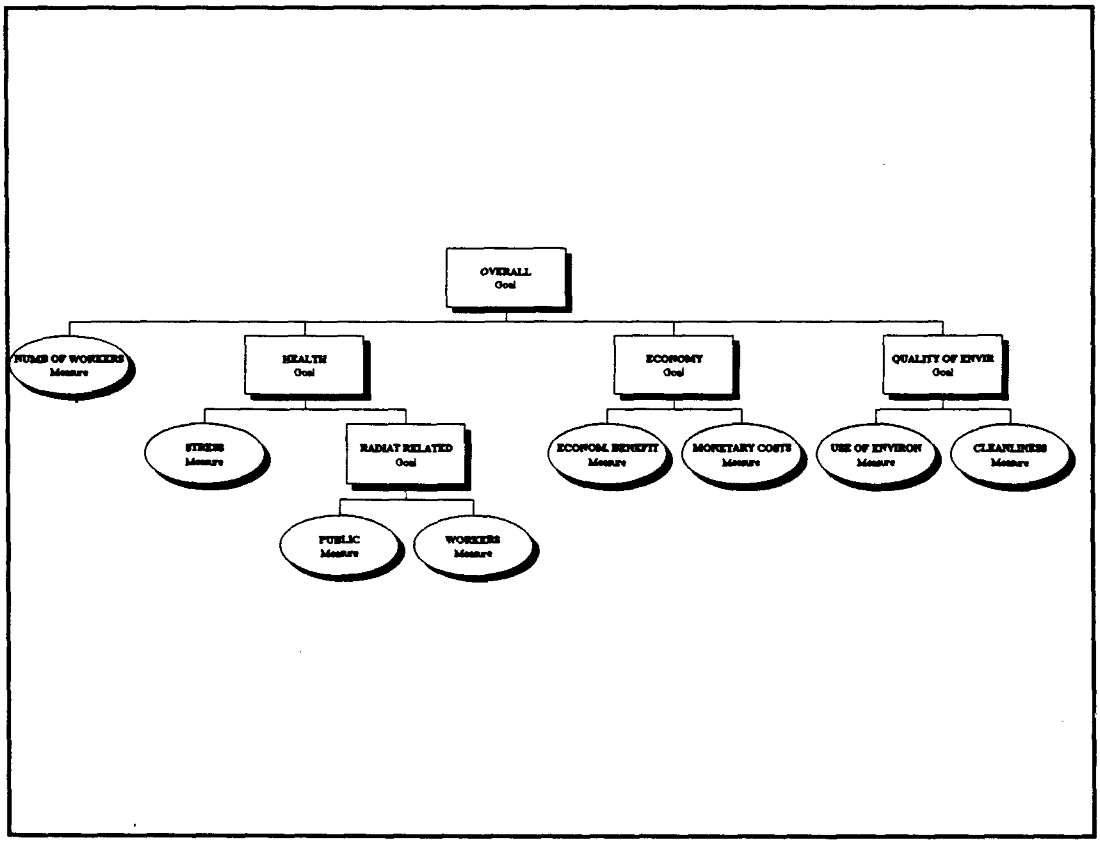

Figure 2. Hierarchy of attributes used in the decision model.

The attributes used in the analysis are defined as follows.

The effect on *health* is seen to have two components, of which radiation related health effects is further divided into two sub-attributes: these are doses to the workers and dose to the public.

Dose to the public. Because exposed individuals are not identifiable, the value of this attribute is assessed as the projected collective dose to the public (manSv). This relates to the expected number of fatal cancers caused by radiation, calculated by applying a risk factor of 5% to the dose in manSv.

Doses to the workers. Projected individual doses to the workers carrying out the recovery operations (mSv).

Number of workers carrying out the recovery operations. As initially planned the recovery operations will cause unacceptable high individual doses to the workers (several ten's of mSv).

To keep the doses acceptable, i.e., below the dose limits, more workers should be employed. Thus, an objective will be to keep the number of workers as low as possible.

Stress. Psychological stress caused by radiation both to the public and workers. Stress will be decreased or increased by protective actions and it reduces the quality of life. This attribute aims to capture the stress caused by unemployment of workers in the affect area and worry felt by land owners.

Quality of the environment. This attribute is seen to have two sub-attributes; *Cleanliness of the environments* and the *use of the environment* for refreshment. These attributes are aimed to capture the reduction in the quality of the contaminated areas and the living areas close to the contaminated forest, disposal site and around power stations burning radioactive wood.

Economic benefit. The monetary benefit to the industry and the society obtained by burning the wood as fuel (MFIM).

Costs. The monetary costs caused by implementing recovery operations. Total costs include direct costs of operations (harvesting, transportation, disposal), control of the area, loss of income and lost capital services (MFIM).

8.4.3. How the strategies perform for each attribute

Now the consequences of the actions can be assessed, i.e., how well the different actions perform for each of the lowest level attributes in the value tree. The consequences are the values of attributes in various actions, e.g., the assessed dose if action is taken and the action's monetary costs. The measurement of these two attributes is easy, because we can identify the variables representing them. However, for attributes, such as stress and quality of environment, it is more difficult to find a proxy attributes or variables that can be quantified. The techniques, which can be used to express the preferences over the values of an attribute, are *direct rating* and the use of *value functions*.

Direct rating can be used with attributes which cannot be represented by easily quantifiable variables. In this technique, the most preferred option for, e.g., stress, a value of 100 is given and the value of zero for the least preferred option. The other options are ranked between zero and 100, according to the strength of preference for one option over another in terms of stress. Although this technique seems to be robust it should be emphasized that there are methods to check the consistency of the elicited numbers. Also, numbers do not need to be precise. As will be pointed out later when discussing sensitivity analysis, the choice of an action is generally fairly robust, and often substantial changes in the figures are required before another option is preferred.

The preferences over values of an attribute can also be changed numerical by a value function. As in direct rating the most preferred option for an attribute, a value of 100 (or 1.0), is given, and the value of zero for the least preferred option. There are several methods which can be used to elicit the intermediate values to form a continuous value function. The simplest conversion, which is used in this analysis, is a straight line, where a unit change in the preference of an attribute corresponds to an equal change in value.

The assessed values of attributes for each action are given in Table III.

Table III. Values of the attributes for strategies defined in Table II.

Strategy	Collective dose to public (manSv)	Individual dose to workers (mSv)	Number of workers	Economical benefit (MFIM)	Costs of action (MFIM)
1a	5460	20	1270	120	121
1b	5380	20	1220	0	129
2a	5480	20	1230	120	119
2b	5400	20	1180	0	127
3a	5390	20	3550	166	151
3b	5370	20	3570	0	163
4	5560	0	0	0	66
5	2050	0	0	0	4710
6b	3140	20	170	0	3200

a) Wood material is burned as fuel before disposal
b) All material is disposed as such.

In assessing the collective dose to the public the external dose and intake of radionuclides in all relevant pathways were considered, i.e., the dose caused by resuspension, burning radioactive wood, using contaminated wood material and forest fire. Of these the use of contaminated wood would cause the highest doses, 3500 manSv. The inhalation dose over 70 years was considered to be small compared to intake and external dose. There are no models designed specifically for this kind of problem. Therefore, the dose calculations have to be based on expert judgements and as far as possible on the dispersion and dose prediction models developed primarily for accidents at nuclear power plant. The dose predictions were done by the ARANO software package (Sa77).

The doses to each worker group in each work phase were calculated separately; felling the trees, removal of undervegetation and soil, transportation of trunks, chip and ash, and disposal of wood material or ash. The software package MATERIA was used to assess the individual doses to workers (Ma93). In most cases the estimated doses were unacceptable high, several tens' of mSv, and in these cases the number of workers was increased to keep the individual doses below 20 mSv.

The use of wood will strongly be affected by public opinion if action to control wood material is taken (strategy 6b). Because the contaminated area would be commonly known, all the wood material from these areas would not be accepted by the industry. This will cause a reduction in collective dose, but at the same time will increase the monetary losses to land owners. It was estimated, that one third of wood otherwise used, i.e., if no action is taken, will be rejected.

The monetary costs of actions and also benefits were calculated using the information collected in another study of this project (cf. Chapter 6), Finnish statistics and similar monetary costs assessment methods as is presented in COCO-1 report (Ha91). The costs of lost capital service and removal of trees are the main costs components.

The scales for stress and quality of the environment attributes were developed judgementally and the values are given below in Tables IV and V. A high score represents a more preferred action.

Table IV. Scores of stress attribute.

Strategy	1a	1b	2a	2b	3a	3b	4	5	6b
Score	80	90	90	100	60	70	40	0	50

a) Wood material is burned as fuel before disposal
b) All material is disposed as such.

Strategy 2b was given the highest score because it treats the workers, the population and land owners fairly, offering a certain degree of decontamination and because the wood would not be burned there would be no local fallout. Also, the amount of disposed waste would be acceptable. It was felt that to reduce psychological effects it is important to take the actions shortly after the accident and at the same time to avoid excessive actions. Strategy 5 was given the lowest score. It treats area III differently than the others by offering reassurance only by controlling access to the area, but on the other hand, it would cause a lot of problems for individual land owners even if the cost of action would have to be borne by the society. Strategy 4 was seen as the next least acceptable. It offers no reassurance of decontamination, although there would be no doses to the workers. The scores for other strategies were assessed according their strength of preferences using similar arguments.

Table V. Scores of quality of the environment attribute.

Strategy	1a	1b	2a	2b	3a	3b	4	5	6b
Score	100	90	80	70	90	80	20	0	50

a) Wood material is burned as fuel before disposal
b) All material is disposed as such.

Although there are sub-attributes, the cleanliness and the use of environment, below the quality of environment attribute it was thought to be appropriate to assess the scores directly to the higher level attribute, i.e., to the quality. Strategies 4 and 5 were given the worst scores because the contamination would be left untouched in the environment. The objective, the use of the environment was also in its worst position in strategy 5. The control of access (with fences) would also impair the quality of the environment. Although there would be a small release of radionuclides into the environment in strategy 1a when burning contaminated wood, it was felt that this strategy offers the best outcome for the environment. The other strategies were felt to be less attractive and the assessed scores are seen in Table V.

In the analysis the figures given above, e.g., collective doses, were transformed linearly to 0 - 1 scales and their different relative lengths are taken into account in assessing the weights on the attributes (see below).

8.4.4 Trade-offs

Before we can combine the values for different attributes in order to obtain a view of the overall benefits which each action has to offer, we have to assess the weights on attributes. They represent the judgement of the decision maker of the relative importance of the levels of the attributes. For example, how much he/she is ready to accept doses to individual workers to avoid a certain dose to the population. When assessing a trade-off value, it should be noticed that the importance of an attribute is not only dependent on its conceptual value, such as health, but also on its *range of values*, such as the number of cancer cases. The range means the difference in values for various actions, e.g., the difference in dose when the action is taken or not taken.

Swing weighting is applied in the analysis as an assessment method for scaling constants, i.e., the trade-offs. In this method a decision maker is asked to compare a set of pairs of hypothetical actions which differ only in their values along two attribute scales until an indifferent pair of options is found. For example:

Option A: The individual dose is 20 mSv and the collective dose is 0 mmanSv.

Option B: The individual dose is 0 mSv and the collective dose is 100 mmanSv.

If options A and B are felt to be indifferent, it can be seen that it is preferable to avoid higher individual risks than individually low but collectively higher risk. It is estimated, that the individual doses to the population are far less than one mSv on average. If we set the weight of the collective dose to one, and taking into account the 'length' of collective dose scale, 3510 manSv, and individual dose scale, 20 mSv, this suggests a weight (5*0.001*20/3510) = 0.00003 for the individual dose scale relative to the collective dose scale.

The weights were set on other attributes using a similar process. For example, the following indifferent (marked with ~) pair of options was elicited for the number of workers and individual dose:

(1 men; 20 mSv) ~ (100 men; 1 mSv).

This assessment together with the fact that the number of workers scale has a length of 3570 men and the worker dose scale 20 mSv, means that number of workers scale is felt 36 times as important as the individual worker dose scale of 20 mSv. Altogether six trade-offs have to be made in order to have a complete set of weights. The following pairs of attributes were used to assess the weights, and the indifferent options are given below:

Collective dose/costs attributes:
(2 manSv; 0 MFIM) ~ (1 manSv; 0.25 MFIM)

Costs of action/monetary benefit attributes:
(0 MFIM; 170 MFIM) ~ (170 MFIM; 0 MFIM)

Collective dose/stress attributes:
(3000 manSv; 100 Stress) ~ (2000 manSv; 0 Stress)

Stress/quality attributes:
(50 Stress; 100 Quality) ~ (100 Stress; 0 Quality).

Based on these assessments the weights of attributes are given in Table VI. *Note:* The weights are normalized so that the sum of weights is one.

Table VI. Weights of attributes.

Attribute	Weight
Collective dose	0.15
Individual dose of workers	0.000004
Number of workers	0.0002
Monetary costs of action	0.76
Economic benefit	0.03
Stress	0.04
Quality of the environment	0.02

8.5. ANALYSIS OF THE MODEL

At this stage we are in position to aggregate the values to find out how well each strategy performs overall. The *additive model* was applied simply to add together an action's weighted value scores (weighted attribute values on each action) to obtain the overall benefit:

$$v(a) = \Sigma_i k_i v_i(a_i),$$

where $v_i(a_i)$ are single-attribute value functions, a_i are assessed values of attributes and k_i are weighting factors. *Note:* A sufficient condition for an additive decomposition of multi-attribute value function is the mutual preferential independence of the attributes. An attribute X is preferentially independent of attribute Y, if the two preference values of attribute X do not depend on the value of Y. The existence of preferential independence is normally verified during the analysis - and should, in principle, be verified. If the conditions for an additive function exist, the weights are assessed by making trade-offs between attributes as described earlier.

To make the calculations slightly easier the decision model was build using the software package LDW (Sm93). The overall scores and ranking of strategies are as is given Table VII. Strategy 1a, decontamination in areas I and II, and no restriction in area III, is just optimal. In fact, strategies 1-4 rank very close to each other. This is due to area III, which because of its large area has a strong effect on attribute values. In strategies 1-4 the same action is taken in area III.

Table VII. Overall scores for the initial analysis.

Strategy	1a	1b	2a	2b	3a	3b	4	5	6b
Overall score	0.834	0.819	0.833	0.819	0.829	0.805	0.790	0.195	0.41
Rank	1st	4th	2nd	4th	3rd	5th	6th	8th	7th

It is wise to be sceptical about the ranking of the actions, if the variation of figures used in the analysis is not analyzed with a sensitivity analysis. We have to examine how robust the choice of an alternative is to changes in the figures. In many cases sensitivity analysis also shows that the data do not need to be accurate. Large changes in these figures are often required before one action becomes more attractive than another. If this is the case, then it would be waste of effort and time to elicit the numbers accurately.

There are several techniques presented in the literature to perform a sensitivity analysis. The most straightforward analysis applied here examines the effects of varying one parameter at a time. Although the method is simple it clearly indicates which factors are important and require refined assessment.

There are lot of uncertainties in the assessment of the collective dose and monetary costs. The weights of these attributes are also high. The sensitivity analysis on the weight of costs is shown in Figure 3.

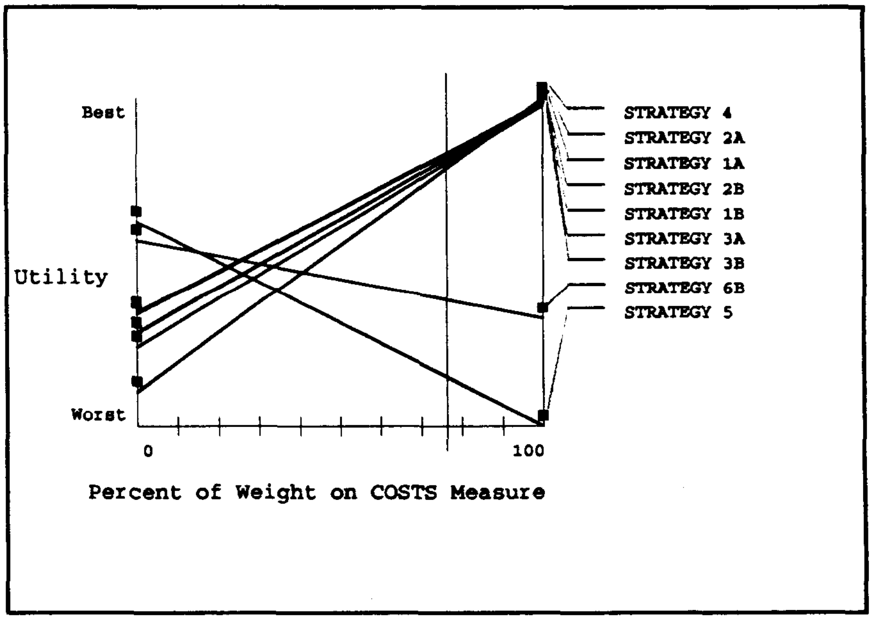

Figure 3. Sensitivity analysis on costs.

The weight on costs is about 76% of the total weight in the model and this value is marked with a vertical line in the figure. The overall score for each strategy against the percentage of total weight on costs are plotted with solid lines. The line with the highest intersection with the vertical line shows the optimal strategy, i.e., strategy 1a.

As the weight on costs is between 35% and 95% strategy 1a is just optimal, but below 35% strategy 6b and then strategy 5, and above 95% strategy 4 will be the best courses of action, respectively. Besides this range gives the accuracy needed in the weighting the costs attribute, it also reflects the required accuracy in the costs calculation because the 'length' of an attribute is taken into account when assessing trade-offs, on the assumption that there is consistency in costs calculation between strategies.

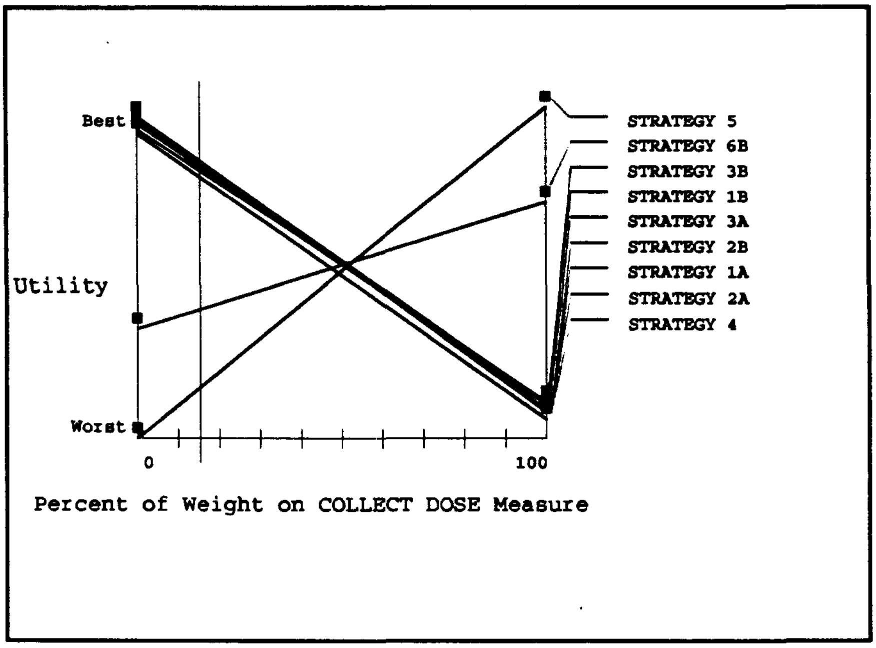

Figure 4. The sensitivity analysis on collective dose.

The sensitivity analysis on the weight of collective dose is shown in Figure 4. With the present weight (15%) on dose, strategy 1a ranks best. The highest value of the collective dose was obtained in the pathway where the contaminated wood is used without restrictions. It was felt that this might be too high. The analysis suggests that substantial changes would be required before strategies 5 and 6 are more preferred.

Because strategies 5 and 6 seem not to be the best course of action the analysis was revised omitting these strategies from the analysis. Doing this the effect of decontamination on decision is more clearly seen. The same trade-offs is used as earlier, but because the 'length' of scales are changed the weights given in Table VIII are obtained.

Table VIII. Revised weights of attributes.

Attribute	Weight
Collective dose	0.086
Individual dose of workers	0.00003
Number of workers	0.0014
Monetary costs of action	0.16
Economic benefit	0.23
Stress	0.34
Quality of the environment	0.17

The ranking of strategies for analysis based upon above-mentioned weights are given in Table IX.

Table IX. Overall scores for the revised analysis.

Strategy	1a	1b	2a	2b	3a	3b	4
Overall score	0.73	0.60	0.72	0.59	0.69	0.47	0.33
Rank	1st	5th	2nd	4th	3rd	6th	7th

The ranking of strategies is - as it should be - the same as in the initial analysis. However, the difference between the strategies is more clearly seen. Now the sensitivity analysis on the weight of collective dose is as is shown in Figure 5.

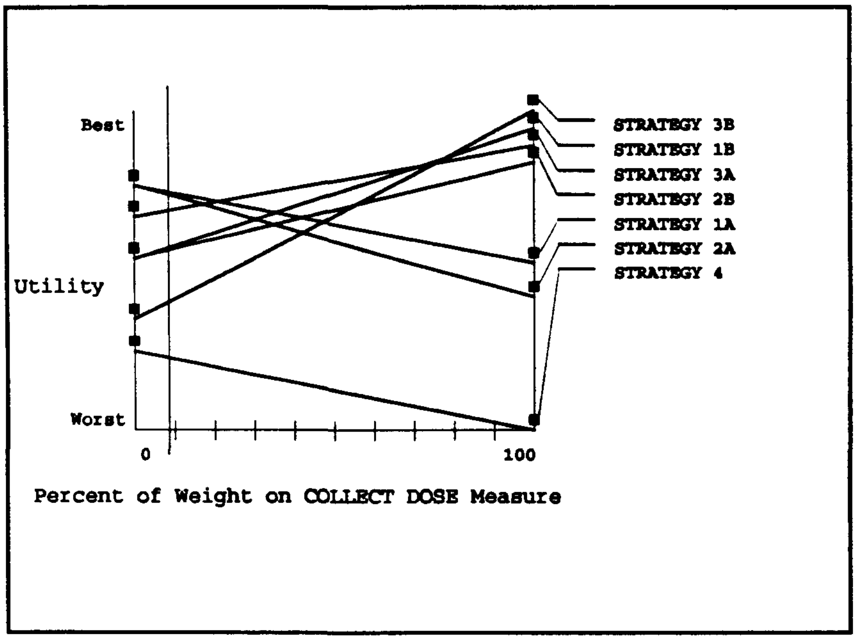

Figure 5. Sensitivity analysis on collective dose. Strategies 5 and 6 are omitted from the analysis.

The difference between strategies 1a, 2a and 3a is not large and the analysis suggests that the best course of action could be found in this group of strategies. There should be modifications in strategies 1b, 2b and 3b, or changes in numbers or trade-offs before this group of action will become more attractive. As is shown in Figure 5 strategies 2b and 4 are never optimal actions considering the values and the trade-offs used in the analysis.

Before giving final conclusion on the action it is useful to gain further understanding considering stress attribute. It was unpleasant to assess numbers on this attribute and because the weight on this attribute is also high, its effect on decision should be further considered. This could be done with Figure 6.

Figure 6 shows that increasing scores go with increasing preferences. The figure plots the overall utility for all other effects excluding stress against stress. In principle the strategy represented by a cross in the upper right corner is the most preferred. On the upper right boundary (Pareto or efficient frontier) lie strategies 1a, 2a, 3a and 2b in this diagram. The optimal choice depends on the value put on stress. As the value increases from 0 to 100 the optimality moves from strategy 3a to strategy 1a and through 2a to 2b. These strategies offer a better choice, i.e., they dominate strategies 1b, 3b and 4 which can never be optimal without changes in their scores and weights.

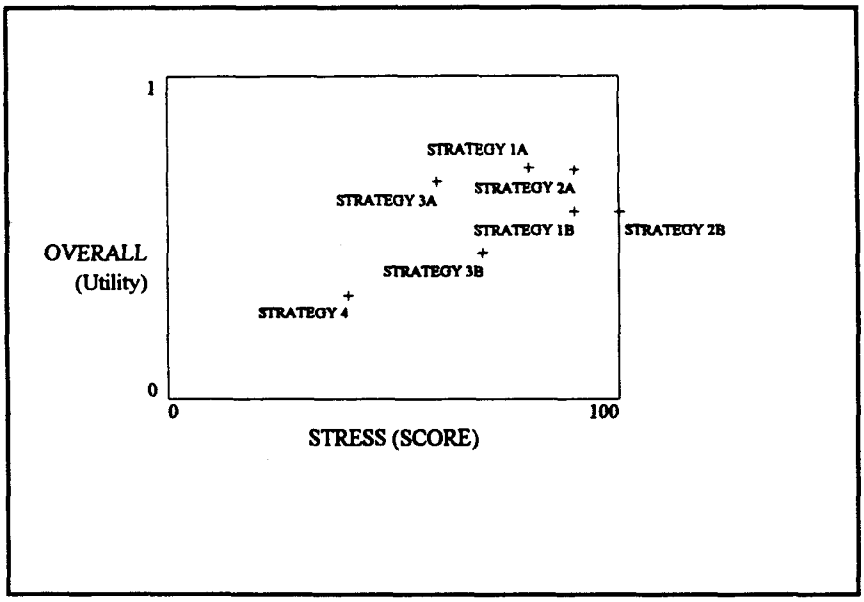

Figure 6. Plot of utility against stress.

8.6. CONCLUSIONS

The objective of this study has been to give an illustration of decision analysis and the application of the analysis when planning countermeasures for forest areas in order to mitigate the consequences of a nuclear accident. The basic principles of radiation protection are based on the justification and optimization of protective actions. Decision analysis, although closely entwined with these principles, does not interpret the results with this terminology. The aim of decision analysis is to find the best solution to a problem based on the rationality of the decision maker(s). However, the result of decision analysis can be translated to correspond to the basic principles of radiation protection.

At the beginning of a decision analysis all the feasible protective actions are defined, including the action of doing nothing. When assessing the justification of protective actions, the present situation forms the basis to which the actions are compared, with respect to the preferences of society represented by a decision maker. The preferences and trade-offs - the judgemental inputs to analysis - form the basis for justification. A protective action is justified if the values connected to it are greater than those of no action.

The optimization of the intervention is achieved by ranking all the feasible actions. The action with the highest ranking will produce the maximum benefit. In optimization it is thus assumed that all actions and attributes are defined at the beginning of an analysis. In practice, however, it is not possible to define all the actions before making some preliminary numerical assessments and running through some rough calculations to gain a feeling for what numbers are important and require refined assessment. The optimization of intervention means this iterative process of maximization of protection in all its essentials. The setting of an intervention level in an accident situation or in planning of the intervention levels is seldom a purely mathematical problem.

The decision analysis performed suggest that a strategy somewhere between 1a and 2a would be the best course of action to be taken in the given situation. There would be a few differences between these strategies. The treatment of areas are still the same: in area II trees would be removed during the same season and in area III 'doing nothing' would be taken in both strategies. No action in area III was deemed to be more preferable to the actions 'control of access' or 'control of wood material'. In area I the trees and undervegetation are removed and the only difference between strategies 1 and 2 is the removal of soil in strategy 1. Also, strategy 3a could be considered as an action. In all strategies the removed trees would be burned as fuel.

There are different preferences connected to the values of attributes. Therefore, *the values of attributes and trade-offs are subjective*, not objective. Expressing the value may be both unpleasant and difficult, but often it is very crucial when assessing an intervention level. Since the values are subjective, no universal values exist. The values are related to the unique problem, and in addition, they change according to opinions and resources. In addtion, people have strong feelings and beliefs about these values, which typically are not numerically quantifiable and do not exist in monetary form. Careful structuring of the problem is necessary to identify the underlying multidimensional values, attitudes to risk and trade-offs related to the problem. To create more insight more research is needed, specially on the less quantifiable factors.

The analysis represented above is based on a hypothetical accident. In real problem depending on prevailing situation where the fallout area could be located on the map, more strategies would have to be considered. Also, the factors entering the decision are dependent on the situation. Thus, the results of the performed analysis could not be applied to real situation as such, but the actions and factors should be revised and the calculations redone. The strategies found appropriate in the analyzed situation might turn out not to be the most preferred in the real problem, however, they might well indicate the course of actions to be considered.

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9. CONCLUDING REMARKS

The initial purpose of this project was to provide basic data and methodology to facilitate decision making when dealing with the management and disposal of cleanup waste after nuclear accidents. During the project it was found out that before methods to deal with the waste could be considered some gaps in the relevant data of the cleanup procedures themselves had to be filled. This was especially true for forest areas, however there were also considerable shortcomings with agricultural areas. After these gaps had been filled, the subsequent analysis has given a more sound and reliable background for decision making concerning the remedial actions. There are still, however, a large number of important topics which were not covered in this project. The topics, which deserve serious attention in the future studies, are: 1) Cleanup in winter time and the effects of winter time deposition on the distribution of contamination and on the cleanup procedures. This is obviously an important topic for the Nordic Countries, where the ground is covered with snow and ice for several months a year. It is probable, that experimental studies are needed to obtain reliable information. 2) Cost-benefit analyses and especially more comprehensive justification-optimization analyses should be made for various situations and areas to obtain insight and understanding of the entire problem. 3) Different parts of the rural environment (fields, indoors, gardens, forest etc.) should be evaluated both for their contribution to the doses and for the need and practicability of their cleanup. 4) Experimental studies on the behaviour of strontium in burial trenches for cleanup waste. Data on the leachability of radionuclides has so far been based mainly on the behaviour of cesium. Strontium is likely to be leached faster, especially when decaying organic material is present. 5) Demand and availability of machinery and techniques for performing cleanup processes.

10. APPENDICES

APPENDIX 1.

NOVEL METHODS FOR RECLAMATION OF CONTAMINATED SOIL

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There are several methods, which have been or are being developed as alternatives for the decontamination of soil by the removal of the upper surface layer. The purpose of the development work is mainly to reduce the volume of radioactive waste to be disposed of. These novel methods are briefly described below and their applicability for the decontamination of soil from very large areas, contaminated in nuclear accidents, is discussed. Most of these methods are being developed in the USA for the cleanup of contaminated DOE (U.S. Department of Energy) sites, such as Hanford and Los Alamos.

Mining Plutonium (Bramlitt 1993). Mining plutonium from soil, contaminated in nuclear weapons tests, has been performed at the Johnston Atoll in the Pacific Ocean. The process utilizes a mineral jig, which separates soil particles according to their densities. Mineral jigs has been for over a century to concentrate gold, however the process required a substantial amount of development to make it suitable for plutonium removal. The process is presently in full-scale operation at the Johnston Atoll, processing 1000 tons of soil per week. The volume of contaminated soil is approximately 80,000 m^3. The removal of plutonium has been 98 % and the volume of the waste, containing most of the plutonium, has been 2 % of the initial volume.

Using a mining process for the removal of radioactive contaminants from the soil is not generally suitable for all types of elements, since only heavier particles are separated. Particles containing, for example, cesium and strontium are of the lighter particle fraction and are, therefore, not separable with mineral jigs. Plutonium particles formed in nuclear weapons tests are larger than those formed in accidental releases from nuclear plants. The efficiency of mineral jig is better for larger particles, and therefore, it is questionable whether this process is suitable for the removal plutonium contamination from nuclear accidents.

Magnetic Separation (Avens 1993). Los Alamos National Laboratory together with the Lockheed Company is developing a High Gradient Magnetic Separation system for the removal of radioactive particles from contaminated soil. In a magnetic field paramagnetic elements and compounds can be retained on a magnetic filter, consisting of steel wool or nickel foam. The higher the field, the smaller the particles which can be removed. High gradients of magnetic field can be obtained with superconducting magnets. Most heavy elements, such as plutonium, americium and uranium, are paramagnetic. In a magnetic separator the soil is slurried in water and directed into a magnetic filter, which is surrounded by a magnet.

With a non-radioactive surrogate CuO (5 % in the slurry) a 90 % removal was obtained with one pass through the separator and a 99 % removal with two successive passes. Magnetic separator is being used in the kaoline clay industry for the separation of iron contaminants. The capacity of the process is fairly high, 30 tonnes of kaoline per hour. The process is still at the development stage for the removal of radioactive contaminants from soil.

Magnetic separation is not directly applicable for the removal of radioactivity from soil contaminated in nuclear accidents, since a wide variety of radioactive elements, with varying magnetic properties are present in the soil. For example, all strontium compounds and most cesium compounds are diamagnetic, and thus not separable with magnetic filters. In soil cesium and strontium are, however, bound to soil particles, some of which have favourable magnetic properties. To overcome the problem of varying magnetic properties Bradtec Ltd (UK) has developed a modification of the magnetic separation system. In this modification diamagnetic radionuclides are bound on magnetite mineral particles, which have been covered with an organic polymer containing selective groups to bind radioactive ions from the solution, e.g. with hexacyanoferrate for cesium. Magnetite support enables the retention of this composite in the magnetic filter. This modification has been developed mainly for the separation of radionuclides from solutions. If it were used for contaminated soil slurries, elements such as cesium should prior to absorption on magnetite composites be leached from the minerals into solution phase. It is probable, that this would make the process very slow and thus impractical.

Turf Harvesting (Jouve 1993). Turf harvesting methods utilizes a conventional turf harvester, used for producing turf mats for making lawns to gardens and parks. Turf harvester removes together with the turf 3-5 cm of soil, which is rather tightly bound to the root network. This upper soil layer, containing most of the radioactive contamination can be removed as rolls of turf mats, which are easier to handle than loose soil. On bare soil grass seeds are spread together with a mixture of peat and polysaccarides in water. This treatment both enhances the growing of the grass and prevents resuspension. In tests at the Chernobyl area with a turf harvester having a 40 cm blade, the removal efficiency for cesium and strontium was on avarage about 97 %. Anaerobic leaching of radionuclides from the turf mats is being investigated.

It is likely that turf harvesting requires a mature turf mat and is not very effective for sand and sandy soil. It is also likely that terrain of the area has to be rather flat. Therefore, it can not be considered as a generally applicable method for all types of soils and turfs.

Skim-and-burial-plowing (Roed 1993). A new type of plough for the treatment of radioactive contaminated soil has been developed by the Risö National Laboratory and a plough factory in Denmark. This so-called skim-and-burial-plough removes the upper 5 cm soil layer and buries it at a depth of about 50 cm without inverting the soil layer between. This plough has been tested at the Chernobyl area rather succesfully. In some cases 95 % of the contamination was found as a layer at a depth of about 50 cm after the treatment. With sandy soils, the performance was not as good, for example in one trial with a loamy sand the activity was removed to a depth between 10 cm and 30 cm.

Skim-and-burial-ploughing is an attractive method for treating radioactive contaminated soil.

First, it does not create any waste. Second, the depth of burial is enough to reduce the external dose very efficiently and prevent tranfer of radionuclides into plants, since the radioactivity lies beneath the root layer for most plants and conventional ploughing (< 30 cm) will not move the contamination afterwards back to the surface. A drawback of the method is that the plough is very heavy and requires a powerful tractor.

Soil Washing (Devgun 1993). Soil washing is mainly used for the decontamination of soil from organic contaminants, but its utilization for the radionuclide removal has also been studied. Soil washing consists of three phases: 1) contacting contaminated soil with a solution containing e.g. acid or chelating agents, 2) physical separation of soil from the solution e.g. by filtration and 3) stripping the contaminant from the solution e.g. by ion exchange. Because the strengths of radionuclide bonds in soil minerals differ within a wide range, no single elutant can be used. This is also true for the stripping phase. These limitations most probably make the process incapable of treating soil containing a wide spectrum of radionuclides.

Electrokinetic Migration (Bibler 1993). In electrokinetic migration electrodes are implanted in the soil and a direct current is imposed between the electrodes. Electric currect causes the cations and cationic particles to migrate towards the cathode and anions and anionic particles toward the anode. The process needs, however, careful control of the pH of the electrodes with rather complicated systems. In a laboratory test with a 3.5 kg soil sample 57 % of uranium was removed in ten days when the voltage used was 25 V. Electrokinetic migration is probably useful for only very limited soil masses.

In Situ Vitrification (Tixier 1993). In in situ virtification high temperatures (1400-2000 °C) are created with electrodes placed directly into soil. Due to the high temperature the soil mass between the electodes turns into a vitreous or ceramic monolith. The process has been tested in full-scale: monoliths of 800 tonnes (5 m deep, 12 m diameter) were obtained. In situ vitrification is a good alternative for e.g. old, improperly constructed waste pits and seepage areas for radioactive effluents. In situ vitrification is not, however, applicable in waste pits containing e.g. barrels and other containers.

Summary

Most of the novel methods described above are probably not applicable to cleanup of soil contaminated in nuclear accident. Fallout from an accident contains a variety of nuclides in several physico-chemical forms. Separation of these nuclides with, for example, washing soil with chemical agents would require several leachants and steps, which could make the process too complicated and expensive. From the methods described above skim-and-burial-plough and turf-harvesting look most promising for cleaning up large areas. They have been tested for actual contaminated field in the Chernobyl area, but are still at the development phase. Magnetic separation may also be applicable to cleanup of large soil masses if it can be modified in such a way, that it can separate also contaminants which do not have magnetic properties as such.

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APPENDIX 2

EXAMPLES OF THE AMOUNTS AND ACTIVITY CONCENTRATIONS OF CLEANUP WASTES FROM FOREST AREAS

There are nine examples below showing the amounts and activity concentrations of cleanup waste, which have been calculated in the following way:

1) The contamination levels and the areas of contaminated lands were obtained from the model accident (Chapter 3).

2) Deposition is either dry or wet.

3) All examples give the concentrations of ^{137}Cs in the waste, but examples 1-3 also those of ^{90}Sr and ^{239}Pu.

4) According to different forest types the Nordic Countries were been devided into three parts:
- Northern Finland, Northern Norway, Northern Sweden
- Southern Finland, Southern Norway, Middle Sweden
- Dennmark, Southern Sweden

Example 1.

Dry fallout in the northern area (Northern Finland, Norway and Sweden) in the summer time. Calculated for the Northern Finland, where 60% of the land area is forest (in Northern Norway 29%), of which 72% is pine forest, 17% spruce and 7% deciduous forest (4% treeless)

Fallout 100 MBq/m^2 Cs-137, area about 2 km^2
- 0.86 km^2 pine forest is contaminated
- 0.2 km^2 spruce forest "
- 0.08 km^2 deciduous forest "

Activities calculated according to the distribution of contamination immediately after the deposition

Contaminated pine forest 0.86 km^2, Cs-137

	Activity TBq	Weight tns	Stacked vol. m^3	Activity/Volume GBq/m^3
Needles	30	450	1100	27
Other crown	30	1000	2700	11
Bark	0.4	92	1100	0.4
Underveget.	26	60	340	76
Total	86	1600	5200	

(+ stems without bark: 6000 m^3)

Contaminated spruce forest 0.2 km^2 Cs-137

Needles	8	380	870	9
Other crown	8	980	2600	3
Bark	0.1	21	260	0.4
Underveget.	4	35	110	50
Total	20	1400	3800	

(+ stems without bark: 2200 m^3)

Contaminated deciduous forest 0.08 km^2, Cs-137

Leaves	1.2	52	110	11
Other crown	1.2	92	210	5.7
Bark	0.02	2.3	27	0.7
Underveget.	5.6	7.2	35	160
Total	8	150	380	

(+ stems without bark 560 m^3)

Total volume: 9400 stacked m^3 of waste
(+ stems without contamination 8760 m^3)
Total Cs-137 activity: 110 TBq (0.4-160 GBq/m^3).

Same area, Sr-90: 76.5 MBq/m^2, Pu-239: 21.7 kBq/m^2

Pine forest

	Activity Sr-90 TBq	Pu-239 GBq	Activity GBq/m^3 Sr-90	Pu-239
Needles	20	6	18	0.006
Other crown	20	6	7.4	0.002
Bark	0.3	0.1	0.3	0.0001
Underveget.	26	7.5	76	0.02
Total	66	20		

Spruce forest

Needles	5.3	1.5	6.1	0.002
Other crown	5.3	1.5	2.0	0.0006
Bark	0.08	0.02	0.3	0.0003
Underveget.	4.6	1.3	58	0.02
Total	15	4		

Deciduous forest

Leaves	0.9	0.3	8.2	0.003
Other crown	0.9	0.3	4.3	0.001
Bark	0.01	0.003	0.4	0.0001
Underveget.	4.3	1.2	120	0.03
Total	6	2		

Total Sr-90 activity: 87 TBq
Total Pu-239 activity: 26 GBq

In the winter time only about 20-40% of the deposition remain on trees in coniferous forests and almost nothing in deciduous forests. Contamination will end up onto the snow and transfer to the undervegetation, litter, humus and soil, when snow will melts. Removal of snow from the forests is seen to impossible in this study.

Area 2. 2/10 of the activities, amounts of waste 16 times higher
Area 3. 2/100 of the activities, amounts of waste 1150 times higher

Example 2.

Wet fallout in the same area as in the example 1.

Fallout 100 MBq/m^2 Cs-137, area 20.7 km^2

- 8.9 km^2 pine forest is contaminated
- 2.1 km^2 spruce forest is "
- 0.9 km^2 deciduous forest "

Activities calculated according to the distribution of contamination immediately after the deposition.

Contaminated pine f8orest 8.9 km^2, Cs-137

	Activity TBq	Weight tns	Stacked vol. m^3	Activity/Volume GBq/m^3
Needles	130	4600	12000	11
Other crown	130	10500	28000	4.6
Bark	1.3	1000	12000	0.1
Underveget.	530	620	3500	150
Litter + humus	89	85000	280000	0.3
Total	880	100000	330000	

(stems without bark: 62000 m^3, stumps 50000 m^3)

Contaminated spruce forest 2.1 km^2 Cs-137

Needles	32	4000	9200	3.5
Other crown	32	10000	27000	1.2
Bark	0.3	380	4600	0.007
Underveget.	130	170	830	150
Litter + humus	21	25000	82000	0.3
Total	210	40000	124000	

(+ stems without bark: 24000 m^3, stumps 21000 m^3)

Contaminated deciduous forest 0.9 km^2, Cs-137

Leaves	6.8	590	1200	5.7
Other crown	6.7	1000	2400	2.8
Bark	0.00008	100	1200	$7x10^{-8}$
Underveget.	68	81	400	170
Litter + humus	9	11000	36000	0.3
Total	91	13000	41000	

(+ stems without bark 6300 m^3, stumps 6000 m^3)

Total volume: 500 000 stacked m^3 of waste
(+ stems and stumps without contamination: 170 000 m^3)
Total Cs-137 activity: 1200 TBq ($7x10^{-8}$ -170 GBq/m^3)

Same area, Sr-90: 76.5 MBq/m^2, Pu-239: 21.7 kBq/m^2

Pine forest

	Activity Sr-90 TBq	Pu-239 GBq	Activity GBq/m^3 Sr-90	Pu-239
Needles	51	14	4.3	0.001
Other crown	51	14	1.8	0.0005
Bark	0.5	0.2	0.04	$2x10^{-5}$
Underveget.	510	140	150	0.04
Litter + humus	68	19	0.2	0.0001
Total	680	190		

Spruce forest

	Activity Sr-90 TBq	Pu-239 GBq	Activity GBq/m^3 Sr-90	Pu-239
Needles	12	3.4	1.3	0.0004
Other crown	12	3.4	0.4	0.0001
Bark	0.1	0.03	0.02	$7x10^{-6}$
Underveget.	120	34	140	0.04
Litter + humus	16	4.6	0.2	0.0001
Total	160	45		

Deciduous forest

	Activity Sr-90 TBq	Pu-239 GBq	Activity GBq/m^3 Sr-90	Pu-239
Leaves	3.4	0.9	2.8	0.0008
Other crown	3.4	0.9	1.4	0.0004
Bark	0.03	0.009	0.03	$8x10^{-6}$
Underveget.	55	15	140	0.04
Litter + humus	7	2	0.2	0.0001
Total	70	20		

Total Sr-90 activity: 910 TBq
Total Pu-239 activity: 260 GBq

Example 3.

Same area as in the example 1. Dry deposition. Cleanup two years after the deposition.

Activities calculated according to the distribution of contamination two years after the deposition.

Contaminated pine forest 0.86 km^2, Cs-137

	Activity TBq	Weight tns	Stacked vol. m^3	Activity GBq/m^3
Stems			7200	
Crown	4.3		3900	0.3
Stumps			4800	
Underveget.	8.6	60	340	25
Litter + humus	56	8200	27000	2
Soil (0-5 cm)	17		43000	0.4
Total	86		86000	

Contaminated spruce forest 0.2 km^2 Cs-137

Stems			2700	
Crown	1		3400	0.1
Stumps			2000	
Underveget.	2	16	79	25
Litter + humus	13	23000	7900	1.6
Soil (0-5 cm)	4		10000	0.4
Total	20		26000	

Contaminated deciduous forest 0.08 km^2, Cs-137

Stems			670	
Crown	0.16		320	0.1
Stumps			550	
Underveget.	0.24	7	35	7
Litter + humus	5.6	960	3200	1.8
Soil (0-5 cm)	2		4000	0.5
Total	8		8800	

Total volume: 120 000 stacked m^3 of waste
Total Cs-137 activity: 110 TBq

Same area, Sr-90: 76.5 MBq/m^2, Pu-239: 21.7 kBq/m^2

Pine forest

	Activity Sr-90 TBq	Pu-239 GBq	Activity GBq/m^3 Sr-90	Pu-239
Stems				
Crown	1.3	0.3	0.1	$3x10^{-5}$
Stumps				
Underveget.	2	0.6	5.9	0.002
Litter + humus	46	13	1.7	0.0005
Soil (0-5 cm)	16	5	0.4	0.0001
Total	65	20		

Spruce forest

	Activity Sr-90 TBq	Pu-239 GBq	Activity GBq/m^3 Sr-90	Pu-239
Stems				
Crown	0.3	0.09	0.05	$1x10^{-5}$
Stumps				
Underveget.	0.5	0.1	6.3	0.001
Litter + humus	11	3	1.4	0.0004
Soil (0-5 cm)	3.8	1	0.4	0.0001
Total	16	4		

Deciduous forest

	Activity Sr-90 TBq	Pu-239 GBq	Activity GBq/m^3 Sr-90	Pu-239
Stems				
Crown	0.1	0.03	0.1	$3x10^{-5}$
Stumps				
Underveget.	0.2	0.05	5.7	0.001
Litter + humus	4.3	1.2	1.3	0.0004
Soil (0-5 cm)	1.5	0.4	0.4	0.0001
Total	6	2		

Total Sr-90 activity: 87 TBq
Total Pu-239 activity: 26 GBq

Example 4.

Dry fallout in the middle area (Southern Finland, Southern Norway and Middle Sweden) in summer time. Calculated for Southern Finland, where 72% of the land area is forest, of which 54% is pine forest, 35% spruce and 8% deciduous forest

Fallout 100 MBq/m^2 Cs-137, area 2 km^2

- 0.77 km^2 pine forest is contaminated
- 0.5 km^2 spruce forest "
- 0.11 km^2 deciduous forest "

Activities calculated according to the distribution of contamination immediately after the deposition.

Contaminated pine forest 0.77 km^2, Cs-137

	Activity TBq	Stacked vol. m^3	Activity GBq/m^3
Needles	27	1100	25
Other crown	27	3600	7.5
Bark	0.4	1700	0.2
Underveget.	23	300	77
Total	77	6700	

(stems without bark 8900 m^3)

Contaminated spruce forest 0.5 km^2 Cs-137

Needles	20	2800	7
Other crown	20	4600	4.3
Bark	0.3	1000	0.2
Underveget.	10	200	50
Total	50	9500	

(stems without bark: 9800 m^3)

Contaminated deciduous forest 0.11 km^2, Cs-137

Leaves	1.7	170	10
Other crown	1.6	270	5.9
Bark	0.02	290	0.07
Underveget.	7.7	48	160
Total	11	780	

(stems without bark: 1500 m^3)

Total volume: 17 000 stacked m^3 of waste
(+ stems without contamination 20 000 m^3)
Total Cs-137 activity: 140 TBq (0.07-160 GBq/m^3)

Example 5.

Same area as in the example 4. Wet deposition.
Fallout 100 MBq/m^2 Cs-137, area 20.7 km^2
- 8 km^2 pine forest is contaminated
- 5.2 km^2 spruce forest "
- 1.2 km^2 deciduous forest "

Activities calculated according to the distribution of contamination immediately after the deposition.

Contaminated pine forest 8 km^2, Cs-137

	Activity TBq	Stacked vol. m^3	Activity GBq/m^3
Needles	120	11400	11
Other crown	120	37400	3.2
Bark	1.2	17700	0.07
Underveget.	480	3100	160
Litter + humus	80	250000	0.32
Total	800	320000	

(+ stems without bark 92 000 m^3
+ stumps 62 000 m^3)

Contaminated spruce forest 5.2 km^2 Cs-137

Needles	78	29300	2.7
Other crown	78	35000	2.2
Bark	0.8	19400	0.04
Underveget.	310	2100	150
Litter + humus	52	200000	0.26
Total	520	290000	

(+ stems without bark 100 000 m^3
+ stumps 66 000 m^3)

Contaminated deciduous forest 1.2 km^2, Cs-137

Leaves	9	1800	5
Other crown	9	3000	3
Bark	0.1	3100	0.03
Underveget.	90	530	170
Litter + humus	12	48000	0.25
Total	120	560000	

(+ stems without bark 16 000 m^3
+ stumps 10 000 m^3)
Total volume: 1 200 000 stacked m^3
(+ stems without contamination: 210 000 m^3)
(+ stumps " " :140 000 m^3)
Total Cs-137 activity: 1400 TBq (0.03-170 GBq/m^3)

Example 6.

Same area as in the example 4. Dry deposition. Cleanup two years after the deposition.

Fallout 100 MBq/m^2 Cs-137, area about 2 km^2
- 0.77 km^2 pine forest is contaminated
- 0.5 km^2 spruce forest "
- 0.11 km^2 deciduous forest "

Activities calculated according to the distribution of contamination two years after the deposition.

Contaminated pine forest 0.77 km^2, Cs-137

	Activity TBq	Stacked vol. m^3	Activity GBq/m^3
Stems		11000	
Crown	3.9	4700	0.2
Stumps		6000	
Underveget.	7.7	300	26
Litter + humus	50	24000	2
Soil (0-5 cm)	15	38500	0.4
Total	77	85000	

Contaminated spruce forest 0.5 km^2 Cs-137

Stems		12000	
Crown	2.5	7100	0.1
Stumps		6300	
Underveget.	5	200	25
Litter + humus	33	20000	1.6
Soil (0-5 cm)	10	25000	0.4
Total	50	71000	

Contaminated deciduous forest 0.11 km^2, Cs-137

Stems		1800	
Crown	0.22	450	0.1
Stumps		950	
Underveget.	0.33	48	6.9
Litter + humus	7.7	4400	1.8
Soil (0-5 cm)	2.8	5500	0.5
Total	11	13000	

Total volume: 170 000 stacked m^3
Total Cs-137 activity: 140 TBq (0.1-26 GBq/m^3)

Example 7.

Dry fallout in the southern area (Southern Sweden and Denmark) in the summer time. Calculated for the Southern Sweden, where 9.5% of the land area is forest, of which 5% is pine forest, 40% spruce and 55% deciduous forest

Fallout 100 MBq/m^2 Cs-137, area about 2 km^2
- 0.0095 km^2 pine forest is contaminated
- 0.076 km^2 spruce forest "
- 0.1 km^2 deciduous forest "

Activities calculated according to the distribution of contamination immediately after the deposition.

Contaminated pine forest 0.0095 km^2, Cs-137

	Activity TBq	Weight tns	Stacked vol. m^3	Activity/Volume GBq/m^3
Needles	0.3	6.5	17	18
Other crown	0.3	21	56	5.4
Bark	0.004	2.2	26	0.25
Underveget.	0.3	0.7	3.8	79
Total	0.9	30	93	

(+ stems without bark 160 m^3)

Contaminated spruce forest 0.076 km^2 Cs-137

	Activity TBq	Weight tns	Stacked vol. m^3	Activity/Volume GBq/m^3
Needles	3	230	530	5.7
Other crown	3	340	880	3.4
Bark	0.04	29	350	0.1
Underveget.	1.5	6	30	50
Total	7.5	600	1800	

(+ stems without bark: 2200 m^3)

Contaminated deciduous forest 0.1 km^2, Cs-137

	Activity TBq	Weight tns	Stacked vol. m^3	Activity/Volume GBq/m^3
Leaves	1.5	100	190	7.8
Other crown	1.5	140	320	4.7
Bark	0.02	27	320	0.06
Underveget.	7	9	44	160
Total	10	280	880	

(+ stems without bark: 2000 m^3)
Total volume: 2800 stacked m^3 of waste
(+ stems without contamination 4400 m^3)
Total Cs-137 activity: 18 TBq (0.06-160 GBq/m^3)

Example 8.

Same area as in the example 7. Wet deposition.

Fallout 100 MBq/m^2 Cs-137, area 20.7 km^2
- 0.092 km^2 pine forest is contaminated
- 0.79 km^2 spruce forest "
- 1.1 km^2 deciduous forest "

Activities calculated according to the distribution of contamination immediately after the deposition.

Contaminated pine forest 0.092 km^2, Cs-137

	Activity TBq	Weight tns	Stacked vol. m^3	Activity/Volume GBq/m^3
Needles	1.5	67	170	8.8
Other crown	1.5	210	570	2.6
Bark	0.01	23	270	4x10^{-5}
Underveget.	5.9	7	39	150
Litter + humus	1	930	3100	0.3
Total	10	1200	4100	

(+ stems without bark 1700 m^3
stumps 950 m^3)

Contaminated spruce forest 0.79 km^2 Cs-137

Needles	12	2400	5600	2.1
Other crown	12	3500	9200	0.1
Bark	0.1	300	3700	0.03
Underveget.	47	63	310	150
Litter + humus	8	9300	31000	0.3
Total	79	16000	50000	

(+ stems without bark: 23 000 m^3
stumps 12 500 m^3)

Contaminated deciduous forest 1.1 km^2, Cs-137

Leaves	8.3	1100	2100	4
Other crown	8.1	1500	3500	2.3
Bark	0.09	290	3500	0.03
Underveget.	83	99	480	170
Litter + humus	11	13200	44000	0.3
Total	110	16200	54000	

(+ stems without bark: 22 000 m^3)
stumps 12 000 m^3)

Total volume: 110 000 m^3 (+ stems without contamination: 47 000 m^3
+ stumps " " : 25 000 m^3)

Total Cs-137 activity: 200 TBq (4x10^{-5}-170 GBq/m^3)

Example 9.

Same area as in the example 7. Dry deposition. Cleanup two years after the deposition.

Fallout 100 MBq/m^2 Cs-137, area about 2 km^2
- 0.0095 km^2 pine forest is contaminated
- 0.076 km^2 spruce forest "
- 0.1 km^2 deciduous forest "

Activities calculated according to the distribution of contamination two years after the deposition.

Contaminated pine forest 0.0095 km^2, Cs-137

	Activity TBq	Stacked vol. m^3	Activity/Volume GBq/m^3
Stems		160	
Crown	0.05	72	0.2
Stumps		93	
Underveget.	0.1	4	26
Litter + humus	0.6	300	2
Soil (0-5 cm)	0.2	480	0.4
Total	0.9	1100	

Contaminated spruce forest 0.076 km^2 Cs-137

Stems		2200	
Crown	0.4	1400	0.1
Stumps		680	
Underveget.	0.8	30	27
Litter + humus	5	3000	1.7
Soil (0-5 cm)	1.5	3800	0.4
Total	7.5	11000	

Contaminated deciduous forest 0.1 km^2, Cs-137

Stems		2000	
Crown	0.2	500	0.1
Stumps		1100	
Underveget.	0.3	44	6.8
Litter + humus	7	4000	1.8
Soil (0-5 cm)	2.5	5000	0.5
Total	10	13000	

Total volume: 25 000 stacked m^3
Total Cs-137 activity: 18 TBq (0.1-27 GBq/m^3)

APPENDIX 3.

DISTRIBUTION OF CONTAMINATION IN FORESTS

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Imformation on contamination and distribution of nuclides in forests is very limited and rather contradictory. The estimations of distribution of nuclides in different parts of forests, given below, is based on the literature and knowledge on Nordic forests conditions. The estimations are very rough. Calculations of the activity concentrations of the cleanup wastes from forest environment are based on the data in Tables 1 to 3.

Table 1.Distribution of the activity on forest (% of total fallout)
a) in summer time. (Rest of the activity under trees)

I Immediately

	Dry fallout		Wet fallout	
	Cs-137	Sr-90 (Pu)	Cs-137	Sr-90 (Pu)
Spruce forest	80	70	30	15
Pine forest	70	60	30	15
Deciduous forest	30	30	15	10

II After three months (summer time)

	Cs-137	Sr-90 (Pu)
Spruce forest	35	20
Pine forest	30	10
Deciduous forest	15	10

Heavy rains and wind during the period decrease amounts

III After two years

	Cs-137	Sr-90 (Pu)
Coniferous forest	5	2
Deciduous forest	2	2

b) in winter time (if snow) immediately. (Rest of the activity is mostly in snow on trees or under trees)

	Cs-137	Sr-90 (Pu)
Coniferous forest	20-40	10-20
Deciduous forest	0	0

Table 2. Distribution of the activity on underlaying vegetation, litter, humus and soil (% of total fallout). Deposition in the summer time. (Rest of the activity in trees)

Undervegetation		Litter+humus		Soil		
I Dry fallout						
Conif.	Decid.	Conif.	Decid.	Conif.	Decid.	
30	70	0	0	0	0	Immediately
60	60	10	25	0	0	After 3 months
30	20	50	65	0	5	After 6 months
15	10	65	75	5	10	After one year
10	3	65	70	20	25	After two years
II Wet fallout						
60	75	10	10	0	0	Immediately

Table 3. Distribution of the activity (%), which remains in trees, in different parts of trees. Distribution depends much on the season.

I Immediately

Needles of leaves (if exist)	50 %
Other crown	49.5%
Bark	0.5%
Stem	0 %

II After six months

Coniferous forest

Needles	30 %
Other crown	40 %
Stem	30 %

Deciduous forest

Crown 60%
Stem 40%

III After two years

Coniferous and deciduous forests:

Crown	50 %
Stem	50 %

Most part of the activity in stems is in bark and vascular bundles.

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APPENDIX 4.

EXAMPLES OF CLEANUP AND TRANSPORTATION COSTS IN FORESTS

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There are three examples below of the costs and manhours needed to carry out cleanup in forest areas. These calculations are based on the calculations of the amounts of cleanup wastes (see Appendix 1) and on the unit costs given in chapter 7 (Tables 5 and 6).

Example 1.

See example 4 in Appendix 2:
Dry deposition in the middle parts of the Nordic Countries (Southern Finlnad, Southern Norway and Middle Sweden). Calculations are based on the data for Southern Finland, where 72 % of the land area is forest, of which 54 % is pine forest, 35 % spruce and 8 % deciduous forest. ^{137}Cs deposition 100 MBq/m^2, contaminated area 2 km^2, of which 0.77 km^2 pine forest, 0.50 km^2 spruce forest and 0.11 km^2 deciduous forest. Cleanup work carried out soon after the deposition. The following measures are taken: felling anf cross cutting of trees and removal of undevegetation and stumps, but no litter, humus and soil. There are two subcases:

1. Disposal of removed material will take place in the affected area. Therefore only local tractor transport and short-distance (within 10 km) truck transport is needed.

	stacked m^3	man-hours (h)	costs (ECU)
pine	15300	2000	102 000
spruce	18200	2300	116 000
deciduous	2200	340	17 000
all stumps	13300	2200	52 000
removal of underv.	550	2800	111 000
All together		9600 h	400 000 ECU

2. Disposal or use of removed trees and stumps will take place outside the affected area. Therefore, they require chipping and long-distance transport. Undervegetation will be disposed of in the affected area and requires only short distance transport.

trees	7300	442 000
stumps	2600	96 000
underveget. (~10 km)	2800	111 000
All together	12700 h	650 000 ECU

Amounts of men and machines to carry out the cleanup and transportation desribed above (subcase 2). It is assumed that the work will be done as three-shift work and in one shift one man works 6.7 hours.

Removal and transportation of trees:

4 x harvester	4 x 3 men
4 x forest tractor	4 x 3 "
2 x hacker	2 x 3 "
6 x truck	6 x 3 "
	48 men

Removal and transportation of undervegetation:

2 x (digger) bulldozer	2 x 3 men
2 x digger	2 x 3 "
4 x truck	4 x 3 "
	24 men

Removal and transportation of stumps:

1 x stump harvester	1 x 3 men
1 x forest tractor	1 x 3 "
1 x hacker	1 x 3 "
2 x truck	2 x 3 "
	15 men

The work can be carried out in 30 days and totally the following amounts of machines and men are needed:

87 men
4 harvester
5 forest tractors
2 hackers
12 trucks
4 diggers
1 stump harvester
1 stump hacker

Example 2

See example 5 in Appendix 2:
Wet deposition in the middle parts of the Nordic Countries (Southern Finlnad, Southern Norway and Middle Sweden). Calculations are based on the data for Southern Finland, where 72 % of the land area is forest, of which 54 % is pine forest, 35 % spruce and 8 % deciduous forest. ^{137}Cs deposition 100 MBq/m^2, contaminated area 20.7 km^2, of which 8.0 km^2 pine forest, 5.2 km^2 spruce forest and 1.2 km^2 deciduous forest. Cleanup work carried out soon after the deposition. The following measures are taken: felling anf cross cutting of trees and removal of undevegetation stumps, litter and humus, but no soil. There are two subcases:

1. Disposal of removed material will take place in the affected area. Therefore only local tractor transport and short-distance (within 10 km) truck transport is needed.

	stacked m^3	man-hours (h)	costs (ECU)
trees	370 000	48 500	2 453 000
stumps	140 000	23 300	533 000
underveget. + litter + humus	500 000	28 800	1 234 000
All together		100 000 h	4 200 000 ECU

2. Disposal or use of removed trees and stumps will take place outside the affected area. Therefore, they require chipping and long-distance transport. Undervegetation will be disposed of in the affected area and requires only short distance transport.

trees		76 500	4 525 000
stumps		24 100	1 005 000
underveget. + litter + humus	(~10 km)	28 800	1 234 000
All together		130 000 h	6 800 000 ECU

Example 3

See example 6 in Appendix 2:
Dry deposition in the middle parts of the Nordic Countries (Southern Finlnad, Southern Norway and Middle Sweden). Calculations are based on the data for Southern Finland, where 72 % of the land area is forest, of which 54 % is pine forest, 35 % spruce and 8 % deciduous forest. ^{137}Cs deposition 100 MBq/m^2, contaminated area 2 km^2, of which 0.77 km^2 pine forest, 0.50 km^2 spruce forest and 0.11 km^2 deciduous. Cleanup work carried out two years after the deposition. The following measures are taken: felling anf cross cutting of trees and removal of undevegetation stumps, litter, humus and soil. There are two subcases:

1. Disposal of removed material will take place in the affected area. Therefore only local tractor transport and short-distance (within 10 km) truck transport is needed.

	stacked m^3	man-hours (h)	costs (ECU)
trees	35 700	4 600	229 000
stumps	13 300	2 200	52 000
undervegetation, litter, humus and soil	120 000	8 400	335 000
All together		15 000 h	620 000 ECU

2. Disposal or use of removed trees and stumps will take place outside the affected area. Therefore, they require chipping and long-distance transport. Undervegetation will be disposed of in the affected area and requires only short distance transport.

		man-hours (h)	costs (ECU)
trees		7 300	434 000
stumps		2 600	96 000
undervegetation, litter, humus and soil	~10 km	8 400	335 000
All together		18 400 h	865 000 ECU

APPENDIX 5.

SUMMARY OF THE ACTIVITY CONCENTRATIONS OF CLEANUP WASTES

Activity concentrations of cleanup wastes for urban, agricultural and forest areas from the scenario described in Chapter 3. Values without parantheses: dry deposition, values with paratheses: wet deposition. Values for forest areas are for cleanup immediately after deposition in Southern Finland. Values for soil removed from fields are for plain soil, without vegetation cover.

Waste	^{137}Cs activity concentration (GBq/t)			
	>1GBq/t	0.01-1 GBq/t (10-1000 MBq/t)	0.0001-0.01 GBq/t 0.1-10 MBq/t	<0.0001 GBq/t <100 kBq/t
Area 1.				
Agricultural:				
Soil from fields				
3 cm	>1.3			
5 cm		>0.8		
10 cm		>0.4		
Cereals	>15 (>8)			
Grass	>22 (>11)			
Forest:				
Needles	>10 (>5)			
Other crown	>3 (>1)			
Bark		>0.02	(>0.006)	
Undervegetation	>14 (>28)			
Litter + humus		(>0.1)		

<u>Area 2.</u>				
Agricultural:				
Soil from fields				
3 cm		0.1		
5 cm		0.06		
10 cm		0.03		
Cereals	1	(0.6)		
Grass	2 (1)			
Forest:				
Needles	2 (1)			
Other crown		0.6 (0.2)		
Bark			0.004 (0.001)	
Undervegetation	3 (6)			
Litter + humus		(0.02)		
Urban:				
Soil (10 cm)		0.02 (0.02)		
Grass	32 (6)			
Removed asphalt	1 (1)			
Firehosing: solid	6 (8)			
liquid				$4 \cdot 10^{-5}$ ($4 \cdot 10^{-5}$)
Firehosing+NH_4NO_3			0.004(0.0004)	
Sandblasting: solid		0.06	(0.006)	
liquid			0.002(0.0002)	
Dust from sweepers	8 (10)			

Area 3.

Agricultural:				
Soil from fields				
3 cm		0.01		
5 cm			0.006	
10 cm			0.003	
Cereals		0.1 (0.06)		
Grass		0.2 (0.09)		
Forest:				
Needles		0.2 (0.1)		
Other crown		0.06 (0.02)		
Bark			0.0004(0.0001)	
Undervegetation		0.3 (0.6)		
Litter+humus			(0.002)	
Urban:				
Soil (10 cm)			0.002 (0.002)	
Grass	3	(0.6)		
Removed asphalt		0.1 (0.1)		
Firehosing: solid		0.6 (0.8)		
liquid				$4 \cdot 10^{-6}$ ($4 \cdot 10^{-6}$)
Sandblasting: solid			0.006(0.0006)	
liquid				$2 \cdot 10^{-4}$ ($2 \cdot 10^{-5}$)
Dust from sweepers	1 (1)			

APPENDIX 6.

KAN2 PROJECT REPORTS AND PUBLICATIONS

K.Brodersen, Cement Solidification and Interactions Between Cement and Radioactive Contaminated Material, 1991, KAN2(91)1, 43 pages.

E.Korhonen, Amounts and Removal of Biomass in Forests (in Finnish), KAN2(92)1, 13 pages.

A.Paajanen and J.Lehto, Disposal of Radioactive Wastes from the Cleanup of Large Areas Contaminated in Nuclear Accidents, Nordiske Seminar- og Arbejdsrapporter 1992:524, KAN2(92)2, 82 pages.

E.Korhonen, Removal and Transportation of Forest Biomass after Nuclear Accidents (in Finnish), KAN2(92)3, 32 pages.

B.Salbu, B.Braskerud and H.Brunstad, Radioactive Waste Mangement and Disposal in Agricultural Environements after a Nuclear Accident, KAN2(92)4, 24 pages.

J.Lehto, T.K.Ikäheimonen, B.Salbu and J.Roed, Amounts and Activity Concentrations of Radioactive Wastes from the Cleanup of Large Areas Contaminated in Nuclear Accidents, Proceedings of the 1993 International Conference on Nuclear Waste Management and Environmental Remediation, Praque, September 5-11,1993, p. 527, KAN2(92)6, 7 pages.

P.Salonen, Costs of Management, Transportation and Disposal of Cleanup Wastes after Nuclear Accidents (in Finnish), Part 1. KAN2(93)1, 13 pages.

P.Salonen, Costs of Management, Transportation and Disposal of Cleanup Wastes after Nuclear Accidents (in Finnish), Part 2. KAN2(93)2, 16 pages.

J.Roed, Urban Decontamination, KAN2(93)3, 7 pages.

J.Roed, Radioactive Wastes Generated by Cleanup of Large Contaminated Urban Areas, KAN2(93)4, 8 pages.

K.Brodersen, Release of Cesium, Strontium and Europium from Soil Columns with Decaying Organic Material, Report Risö-I-722(EN), 1993, KAN2(93)5, 23 pages.

K.Brodersen, Cement Solidification of Soil and Interactions Between Cement and Radioactive Contaminated Soil, Report Risö-I-721(EN), 1993, KAN2(93)6, 67 pages.

B.Salbu, B.Braskerud, G.Ostby and D.Oughton, Removal and Disposal of Radioactive Waste from Contaminated Agricultural Areas after a Severe Nuclear Accident, paper presented at the International Conference on Environmental Radioactivity in the Arctic and Antartic, Kirkenes, Norway, August 23-27, 1993, 4 pages.

Cleanup of Large Radioactive-Contaminated Areas and Disposal of Generated Waste

Should a serious reactor accident occur, a need may arise to remove contaminated material from the environment, from soil, forests or from urban areas. Hereby considerable amounts of radioactive waste material could be generated. In this report, waste quantities are calculated on a theoretical basis, and various methods for handling of such wastes are outlined. Different methods can be used in order to decide upon the best course of action if cleanup of large contaminated areas would actually become necessary.

The Nordic Committee for Nuclear Safety Research - NKS
organizes pluriannual joint research programmes. The aim is to achieve a better understanding in the Nordic countries of the factors influencing the safety of nuclear installations. The programme also permits involvement in new developments in nuclear safety, radiation protection, and emergency provisions. The three first programmes, from 1977 to 1989, were partly financed by the Nordic Council of Ministers.

The 1990 - 93 Programme
comprises four areas:

*	Emergency preparedness	(The BER-Programme)
*	Waste and decommissioning	(The KAN-Programme)
*	Radioecology	(The RAD-Programme)
*	Reactor safety	(The SIK-Programme)

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