PART 6

MANAGEMENT AND DISPOSAL OF RADIOACTIVE WASTE FROM CLEAN-UP OPERATIONS

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

26.1 Introduction

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

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

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

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

26.2 Amounts and activity concentrations of clean-up waste

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

26.3 Estimation of amounts and activity concentrations

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

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

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

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

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

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

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

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

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

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

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

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

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

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

26.4 Clean-up wastes from urban areas

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

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

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

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

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

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

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

26.5 Clean-up wastes from rural areas

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

26.6 Clean-up wastes from forests

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

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

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

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

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

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

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

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

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

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

26.7 Summary of the amounts and activity concentration

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

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

26.8 Transportation of clean-up wastes

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

26.8.1 Safety in waste transportation

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

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

26.8.2 Transporting wastes within the controlled area

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

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

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

26.8.3 Transporting wastes outside the controlled area

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

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

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

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

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

26.9 Final disposal of clean up waste

26.9.1 Interim storage

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

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

26.9.2 Final disposal

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

Selecting the final disposal site and the disposal method

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

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

Shallow ground disposal

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

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

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

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

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

Surface mounds

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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