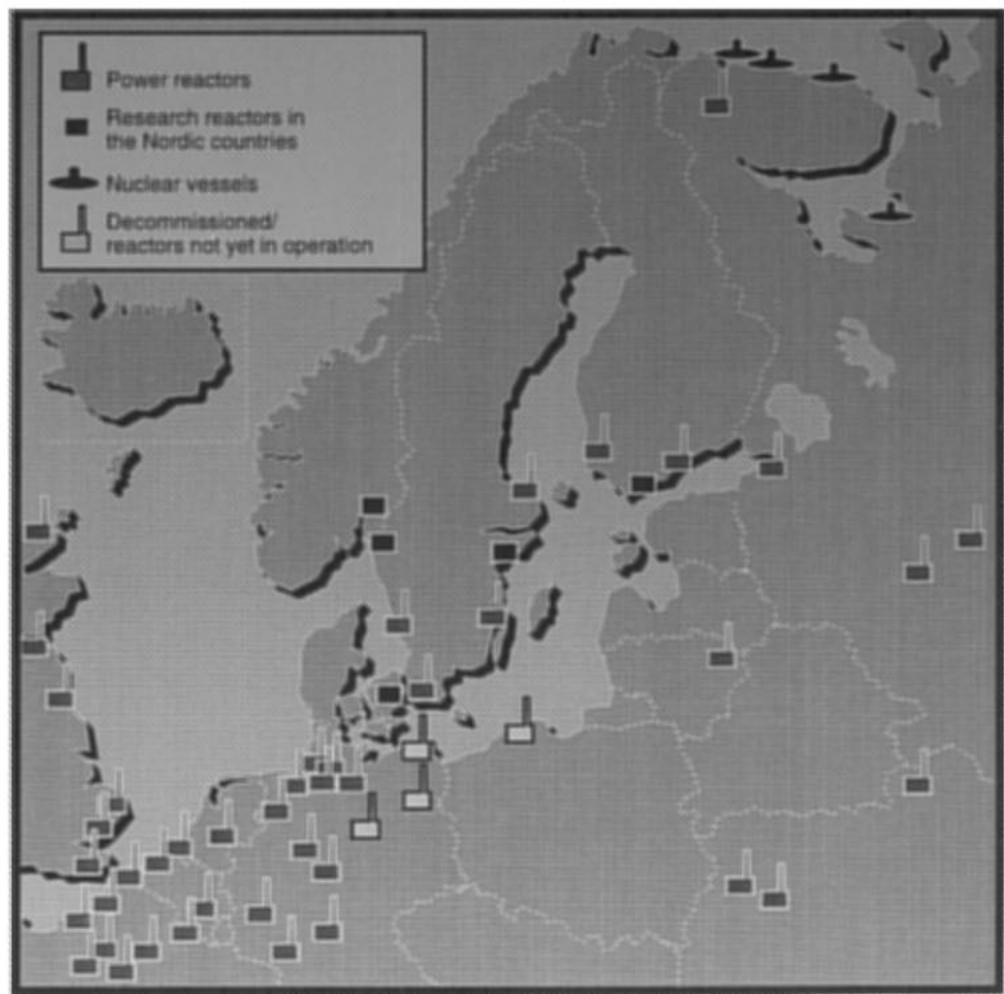


Guidance on Clearance from Regulatory Control of Radioactive Materials



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**Final Report of the Nordic Nuclear
Safety Research Project KAN-1.1**

**Esko Ruokola
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The Nordic Council of Ministers

was established in 1971. It submits proposals on co-operation between the governments of the five Nordic countries to the Nordic Council, implements the Council's recommendations and reports on results, while directing the work carried out in the targeted areas. The Prime Ministers of the five Nordic countries assume overall responsibility for the co-operation measures, which are co-ordinated by the ministers for co-operation and the Nordic Co-operation Committee. The composition of the Council of Ministers varies, depending on the nature of the issue to be treated.

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

Clearance from regulatory control involves that radioactive material of very low activity can be recycled, reused or disposed of like non-radioactive material. International recommendations for clearance and their implications for the applicability of various clearance options (conditional and unconditional clearance) are discussed. Guidance is given on the radiological consequence analysis and activity monitoring pertaining to clearance from regulatory control. Regulatory control and documentation are discussed. A summary of the national regulations and potential quantities of wastes to be cleared in the Nordic countries is also included in the report.

Descriptors - INIS:

DENMARK; FINLAND; ICELAND; LOW-LEVEL RADIOACTIVE WASTES; MONITORING; NORWAY;
RADIATION PROTECTION; RADIOACTIVE WASTE MANAGEMENT; RECOMENDATIONS;
REGULATORY GUIDES; SWEDEN; TRANSPORT REGULATIONS.

SUMMARY

A considerable part of the waste from installations where radioactive material is handled, contains only minor amounts of activity. In spite of this, regulatory requirements impose that all types of waste from a so called controlled area of a nuclear installation are considered radioactive until the contrary is shown. It can be difficult to prove that a certain quantity of such material is practically non-radioactive, especially if the volumes are large and the radioactive substances are distributed unevenly in the waste. Outside controlled areas, in nature, there are also vast amounts of material that are radioactive, but here the concentrations are considered to be so low that they are not subject to radiological control.

If material is classified as radioactive it has to be managed in accordance with complex and often expensive procedures compared to its treatment as non-radioactive. There is therefore a substantial incentive to reclassify the material as non-radioactive, i.e. to release it from regulatory control. However, the material must be treated and disposed of in accordance with other relevant regulations, such as those for disposal or recycling of conventional waste materials.

Certain categories of small radiation sources may be handled without any regulatory control. This option is called *exemption*. There is another option for waste from the nuclear fuel cycle or from other regulated facilities if the activities are sufficiently low. Such material may obtain *clearance* from regulatory control, which means that the material is no longer subject to any radiation protection control. The highest levels of activity concentration that are permitted for such waste are called *clearance levels*.

Cleared materials can be treated or disposed of in the same way as non-radioactive wastes. Thus, it can be used for landfill, or it can be incinerated, recycled or reused. It is not always possible to know the future fate of material that has been cleared, e.g. when metal scrap is transferred between countries. In such cases, a permit for clearance issued by the competent authority must not impose conditions limiting its further use. So there are reasons to define *unconditional* clearance levels that are internationally consistent. However, if it can be demonstrated that the cleared material will only be used for some well defined purposes, then the authorities may issue *conditional* clearances. This alternative allows higher activities to remain in the material. Such permits are normally issued on the basis of case-by-case evaluations.

Guidance for the clearance of radioactive materials, based on radiation protection principles, international organizations IAEA, NEA and CEC. Determination of unconditional clearance levels is being prepared in these organizations and some recommendations have already been issued. In the Nordic countries, there is a joint recommendation for the clearance of wastes from the use of radioisotopes, on which the national regulations are based. In Finland and Sweden, regulations for the clearance of nuclear wastes exist. However, there is a discrepancy between the clearance levels in the regulations of the two countries. Due to the trade of e.g. scrap between the countries, harmonization of the the Finnish and Swedish regulations would be desirable.

The international recommendations for unconditional clearance levels are quite stringent, in the order of 1 Becquerel per gram for the most restrictive nuclides. This clearance level has been derived in a conservative way in order to keep resulting individual doses below 10 microSievert per year. In many cases, it will be difficult to demonstrate compliance with such clearance levels. This is why the option of conditional clearance is attractive, though it requires that the fate of the cleared material can be adequately defined. A radiological impact analysis must be made to determine any effects that may be caused by the cleared material. Guidance for such case-by-case-analyses has been prepared in this project, so as to facilitate the issue of permits for conditional clearance in the Nordic countries.

In order to demonstrate that the radioactive content of waste material actually is below authorized limits, a monitoring programme is needed. Routine methods are available for direct measurements on waste batches as far as gamma emitting nuclides are concerned. Since pure beta and alpha emitters are difficult to measure, indirect methods must be used and they must ensure adequate consistence with the clearance levels.

One such indirect method is the use of scaling factors. This implies that the nuclide to be assessed is determined by relating it to a gamma emitter that can be measured directly. In this project, guidance has been prepared regarding the use of scaling factors, including a novel approach where the use of two reference nuclides for each difficult-to-measure beta or alpha emitter is suggested. This guidance, for the use of applicants and authorities in the Nordic countries, also includes description of radiochemical analysis and calculational methods that are needed for the determination of scaling factors.

Metal scrap can also be handled by melting in a licenced facility, as is the case in Sweden. The resulting ingots can either be disposed of as radioactive waste or cleared immediately or after a decay period. Clearance of ingots avoids some problems related to radiological impact analysis and activity monitoring, as the activity is homogenously distributed in the material.

Radioisotopes for use in medicine and industry on the scale of tens of terabecquerels are imported and produced each year in the Nordic countries. Only a small share of this activity consists of long-lived nuclides. Therefore, after a reasonable period of temporary storage most of this waste can be cleared in accordance with national regulations.

Of the few hundreds cubic meters of waste generated annually by the operation of a nuclear power plant, only a few tens of tonnes, or hundreds of megabecquerels in total activity, would normally be available for clearance. But during decommissioning of a nuclear power plant, more than tenfold quantities of materials that could be cleared would arise annually.

In view of future decommissioning of nuclear installations in the Nordic countries, activity levels of components and surfaces should be systematically recorded. This will facilitate documentation in view of obtaining a licence for clearance at a later stage. There can be a large economic advantage from the clearance of considerable amounts of waste from decommissioning, provided that this waste will have access to conventional landfill or recycling facilities.

SAMMANFATTNING

En stor del av det avfall som produceras vid kärnteckniska anläggningar eller vid andra anläggningar där radioaktivt material hanteras innehåller endast små mängder av radioaktiva nuklidor. Trots detta kräver lagstiftning och myndigheternas föreskrifter att allt avfall ifrån så kallat "kontrollerat område" vid en kärntecknisk anläggning betraktas som radioaktivt tills det undantagits ifrån lagens bestämmelser. Det kan vara svårt att bevisa att en viss avfallsmängd har så låga aktivitetsnivåer att den ska betraktas som inaktiv, speciellt om volymen är stor och de radioaktiva ämnena är ojämnt fördelade i avfallsmängden. Utanför kontrollerat område finns också stora mängder av naturlig förekommande radioaktiva ämnen, men här anses koncentrationen av de radioaktiva nukliderna vara så låga att de ej behöver underställas någon radiologisk kontroll.

Om material klassats som radioaktivt måste det hanteras på ett komplicerat och ofta dyrbart sätt jämfört med om det kunde hanteras som inaktivt. Det finns därför ett starkt incitament till att klassa om materialet så det kan behandlas som inaktivt, dvs att man kan undanta materialet från myndigheternas kontroll. Det måste dock fortfarande behandlas och omhändertas i enlighet med andra relevanta bestämmelser, t ex regler för återanvändning och hantering av konventionellt avfall.

Vissa typer av små strålkällor får hanteras utan myndighetskontroll. Man säger då att de är undantagna ifrån sådan kontroll. Också material inom kärnbränslecykeln eller annat material som reglerats av myndigheternas föreskrifter kan, om innehållet av radioaktiva nuklidor är tillräckligt lågt, undantas från vidare kontroll och friklassas. De högsta aktivitetsnivåerna som tillåts för sådant material eller avfall kallas friklassningsgränser.

Friklassat material kan hanteras på samma sätt som inaktivt material eller avfall. Det kan således användas som fyllnadsmaterial, förbrännas eller återvändas. Det är inte alltid möjligt att kärrna till hur material som friklassats kommer att användas i framtiden. Skrot hanterat t ex på en internationell marknad och förs över nationsgränserna. I sådana fall får ett friklassningsbeslut av en myndighet ej innehålla framtida villkor vad avser materialets användning. Det finns således ett behov att definiera internationellt överensstämmende gränsvärden för ovillkorlig friklassning. Om det kan visas att friklassade materialet bara ska användas för något begränsat, väl definierat ändamål, kan myndigheterna dock tillämpa villkorlig friklassning. Detta alternativ medger högre aktivitetsnivåer av de i materialet förekommande radioaktiva ämnena. Sådan friklassning görs normalt genom att det enskilda fallet studeras och utvärderas separat.

Anvisningar för friklassning av radioaktivt material, baserat på internationella strålskyddsprinciper, har utfärdats av de internationella organen IAEA, NEA och EC. Gränsvärden för ovillkorlig friklassning förbereds av dessa organisationer och några rekommendationer har redan formulerats. För de nordiska länderna finns gemensamma rekommendationer för friklassning av avfall som uppkommer när radioisotoper används och de nationella bestämmelserna och föreskrifterna baseras på dessa rekommendationer. Både i Finland och Sverige finns generella föreskrifter för friklassning av kärnavfall. De två ländernas gränsvärden

för friklassning är dock ej överensstämmende. På grund av handeln med skrot mellan länderna vore det önskvärt att Finlands och Sveriges friklassningsföreskrifter harmoniseras.

De internationella rekommendationerna för ovillkorliga friklassning är ganska restriktiva, gränsvärdet är i storleksordningen 1 becquerel per gram (1 Bq/g) för de mest begränsande nukliderna. Dessa gränsvärden har beräknats på ett konservativt sätt för att säkerställa att de resulterande individdoserna till allmänheten understiger 10 mikrosievert (10 μ Sv) per år. Det är dock i många fall svårt att visa att aktiviteten är så låg att man ej överstiger gränsvärdarna för friklassning. Därför är alternativet med villkorlig friklassning attraktivt även om materialets fortsätta användning i så fall måste kunna entydigt begränsas. En radiologisk analys måste utföras så att man kan beräkna de doser som det friklassade materialet kan ge upphov till. Inom detta projekt har rekommendationer utarbetats om hur sådana analyser kan utföras i enskilda fall för att underlätta villkorlig friklassning i de nordiska länderna.

För att demonstrera att det radioaktiva nuklidinnehållet i avfallsmaterial verkligen understiger fastställda gränsvärden krävs ett mätprogram. Metoder för rutinmässig bestämning av innehållet av gammastrålande nuklidor i avfall finns tillgängliga. Eftersom nuklidor som endast är alfa- eller betastrålande är svåra att mäta direkt måste indirekta metoder tillgripas för att säkerställa att man ej överstiger gränsvärdarna för friklassning för dessa nuklidor.

En sådan indirekt metod är användningen av skalfaktorer. Detta innebär att de nuklidor som är av intresse relateras till en gammastrålande nuklid som kan mäts direkt. I detta projekt har anvisningar utarbetats för hur skalfaktorer kan användas. En ny metod utnyttjas där två referensnuklidor används för varje alfa- eller betastrålande nuklid som annars skulle vara svårmätbar. Dessa anvisningar är avsedda att användas både av enskilda tillståndssökande och av myndigheterna i de nordiska länderna. De inkluderar också en beskrivning av radiokemiska analyser och beräkningsmetoder som behövs för att bestämma skalfaktorer.

Metallskrot kan behandlas genom smältning i en licensierad anläggning vilket är fallet i Sverige. De resulterande göten kan slutföras som radioaktivt avfall eller friklassas direkt eller efter avklingningsperiod. Genom smältning av metall till göt undviks vissa problem vid aktivitetsbestämningen av metallen eftersom de radioaktiva nukliderna är jämnt fördelade i göten. Även analysen av den radiologiska inverkan på allmänheten och miljön vid den fortsatta hanteringen av metalgöten underlättas.

Varje år importeras och produceras radioisotoper för användning inom medicin och industri med en sammanlagd aktivitet av tiotal TBq i de nordiska länderna. Endast en liten del av denna aktivitet härrör från långlivade radionuklidor. Efter en kortare lagringstid kan det mesta av det avfall som uppkommer härvid friklassas i överensstämmelse med nationella föreskrifter.

Av de hundratals kubikmeter avfall som årligen uppkommer vid driften av ett kärnkraftverk är det endast några tiotal ton med en sammanlagd aktivitet av några hundratal MBq som det är möjligt att friklassa. Däremot, när ett kärnkraftverk skall rivas är den mängd som årligen skulle kunna friklassas tiofaldigt större.

Ytaktiviteten på komponenter och deras aktivitetsinnehåll bör systematiskt registreras vid nordiska kärntekniska anläggningar, speciellt med tanke på den framtida rivningen av dessa. Ett sådant förfarande skulle underlätta den dokumentering som är nödvändig i syfte att senare erhålla en licensierad friklassning. Förutsatt att kan deponeras på ett vanligt avfallsupplag, omhändertas eller återvändas på något annat sätt så kan detta medföra stora ekonomiska fördelar, speciellt med tanke på de stora mängder avfall kommer att kunna vid rivning av kärntekniska anläggningar.

PREFACE

KAN-1.1 is a project of the Nordic Nuclear Safety Research Programme 1990-1993. The author has acted as the project leader. For the project, research contracts were made with ABB Atom AB, KEMAKTA Konsult AB, Technical Research Centre of Finland and the Radiochemical Institute of Helsinki University. Numerous experts from the Nordic authorities and companies have contributed to the project by providing pieces of information and comments and by hosting meetings in the course of the project. The author wishes to thank them all for their valuable contribution.

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- 1 Models and parameters for radiological assessment of various clearance options
- 2 Determination of some difficult-to-measure radionuclides

CHAPTER 1 INTRODUCTION

This is the final report of the Nordic Nuclear Safety Research Project KAN-1.1. The objective of the report is to provide guidance and background information on release from regulatory control ("clearance") of radioactive material with so low activity concentration that it is of no concern from a radiation protection point of view. The main target groups of the report are waste producers who make applications for clearance of radioactive materials, and competent authorities who review such applications and prepare rules and guides on the subject.

The basic principles are the same for all types of radioactive material and "practices", as long as the source is controllable. In the report, the main emphasis is put on solid materials from the operation and dismantling of nuclear facilities. Wastes from the use of radioisotopes in medicine, research and industry are also dealt with at some length. Wastes from mining and milling or from nuclear fuel fabrication involve some special problems and are not discussed here.

Most weight is given to the description of safety analysis and of monitoring methods that can be used for demonstrating compliance with the radiation protection criteria for clearance. Decontamination procedures that may be performed prior to clearance in order to reduce activity concentration levels, are not dealt with.

Though the radiation protection aspects are generally the crucial ones to be considered in context with clearance, also other aspects are of importance, such as conservation of raw materials and energy, non-radiological risk and environmental impacts, economic considerations and socio-psychological factors. Ideally, a generic optimization study taking account of all relevant factors would be desirable for the formulation of a clearance policy. The discussion in this report is, however, restricted to the radiation protection aspects.

CHAPTER 2 BASIC CONSIDERATIONS

2.1 Radiation protection principles

The basic radiation protection principles for release of radioactive materials from regulatory control are given in the International Atomic Energy Agency's (IAEA) Safety Series No 89 /1/. IAEA's Safety Guide was prepared in cooperation with the OECD Nuclear Energy Agency, thus it represents a wide international consensus on the subject.

The IAEA safety guide sets forth two basic criteria for determining whether or not a practice can be released from regulatory control:

- i Individual risk must be sufficiently low as not to warrant regulatory concern. In quantitative terms this criterion implies that individual effective doses to critical group from the practice should be of the order of 0.01 mSv in a year at the most.
- ii Radiation protection, including the cost of regulatory control, must be optimized. However, if a generic study indicates that the collective dose commitment resulting from one year of the unregulated practice would be less than about 1 manSv, the practice can be released from regulatory control without more detailed examination of other options.

The term "practice" is defined as a set of co-ordinated and continuing activities involving radiation exposure and aiming at a given purpose, or a combination of a number of similar such sets. A practice involving solid waste disposal should generally be defined as the disposal of waste at a given site (e.g. landfill or incinerator). If radioactive material is released for recycling or reuse, a source-related (e.g. a given nuclear facility) or material stream-related (e.g. steel or concrete) definition of the practice is the most appropriate. More guidance on definition of practices is included in Safety Series No 89 /1/.

The IAEA has also drafted a safety guide /2/, in which activity constraints for clearance from regulatory control are derived. These recommended unconditional clearance levels are summarized in Table 2-1. The safety guide will be published in 1994.

2.2 Exemption vs. clearance

In principle, there are two options for release of materials from regulatory control. One involves removal of restrictions from materials which have been a part of sources or practices subject to the requirements of regulatory control (registration, notification, licencing). The other option refers to a so small scale use of radioactive sources (e.g. calibration sources, low-level tracers, some consumer products) that it is excluded as a whole from the regulatory regime. To distinguish between these two options, the term **clearance** has lately been adopted for the former /2, 3, 4/ while the term **exemption** is nowadays devoted only to the latter option. The radiation protection principles given in the previous chapter are applicable to both exemption

Table 2-1: Unconditional clearance levels proposed by the IAEA /2/. Besides activity concentration, the numerical values are recommended to apply also for surface contamination in Bq/cm².

Ranges of Activity Concentration (Bq/g)	Radionuclides		Representative Single Values of Activity Concentration (Bq/g)	
0.1	Cs-134 Na-22 Na-24 Mn-54 Co-60 Zn-65 Nb-94 Ag-110m Sb-124	Cs-137 Eu-152 Pb-210 Ra-226 ¹ Ra-228 ¹ Th-228 ¹ Th-230 ¹ Th-232 ¹	U-234 ¹ U-235 U-238 ¹ Np-237 Pu-239 Pu-240 ¹ Am-241 Cm-244 ¹	0.3
< 1.0				
≥ 1.0	Co-58 Fe-59 Sr-90 Ru-106	In-111 I-131 Ir-192 Au-198 Po-210		3
< 10				
≥ 10	Cr-51 Co-57 Tc-99m Tc-99 I-123	I-125 I-129 Ce-144 Tl-201 Pu-241		30
< 100				
≥ 100	C-14 P-32 Cl-36 Fe-55	Sr-89 Y-90 Cd-109		300
< 1000				
≥ 1000	H-3 S-35 Ca-45	Ni-63 Pm-147		3000
< 10000				

Note

1. Radon-220 and 222 were not considered in this classification.

and clearance. However, the derived levels in activity units are not necessarily equal due to differences in types and quantities of materials of interest in each case.

In this report, only clearance of materials from regulatory control is considered. The terms **to clear**, **cleared waste/material** and **clearance level** are consequently used in the following chapters.

2.3 Unconditional vs. conditional clearance

The general concept of clearance from regulatory control implies a complete removal of controls so that the cleared materials are treated just as if they were not radioactive. This so called **unconditional** (or unrestricted) clearance of a practice or source requires that all possible exposure pathways are examined and taken into account in the derivation of the clearance levels, irrespectively of how that material is used and to where it may be directed. As the clearance levels must be widely applicable, they must be based on consideration of generic scenarios and data. Consequently, the unconditional clearance levels tend to be conservative, i.e. they are low and in most cases the actual doses received will be well below the dose constraints referred to in Chapter 2.1. Unconditional clearance is applicable in particular to such cases where the fate of material is not known once it is cleared.

Alternatively, the clearance may be constrained in some way, usually if the fate of material being considered is known. In this case, only a limited number of exposure routes has to be considered when deriving the clearance levels, and site specific data and realistic assumptions can be introduced in dose calculations. Hereby it is possible to end up with higher clearance levels as compared to unconditional clearance. Such options are called **conditional** (or restricted) clearances and the derived constraints are called conditional clearance levels. The main disadvantage of this option is that dose calculations must be revised whenever there is a significant change in the clearance practice.

2.4 Management of cleared materials

One of the main options for the management of cleared wastes is **landfill disposal**. It is applicable to almost all kinds of solid wastes. There are different kind of landfill disposal sites: municipal waste dumps and on-site shallow land burial sites. Some materials, such as concrete, may be used as filling material e.g. for roads. During decommissioning it is also possible that underground structures of a nuclear facility are not dismantled but are covered with soil. All of these options can be regarded as landfill disposal of cleared waste. More detailed descriptions of landfill practices are included in /5,6/.

Another main disposal option is **incineration**. There are various types of facilities: hospital incinerators, municipal incinerators and incinerators for hazardous wastes. Cleared oil might also be mixed with fuel oil and burned in a district heating plant. Large incineration plants have normally flue gas filtration systems. Incinerators generate secondary wastes (ash, slag, fly ash) which must be disposed of e.g. in a landfill. Descriptions of current incinerator practices are included in /5,6/.

Some wastes, metal scrap in particular, may be **recycled**. Recycling of metals involves melting and fabrication into new products. In context with this process, various kinds of secondary wastes (slag, dust, sludges) are generated and must be handled. Metal recycling practices are described in /3/.

A special type of recycling is melting of radioactive scrap metal in a licenced facility. The ingots that cannot be immediately cleared after melting, may be used as raw material for the nuclear industry or stored for later clearance after an adequate decay period. There are four such plants currently in operation in France, Germany, Sweden and the USA and more than 14,000 tonnes of metals scrap has been treated in those facilities.

During decommissioning or routine maintenance of nuclear or isotope handling facilities, discrete pieces of contaminated equipment may be salvaged and **reused** as such. Examples are tools, motors, pumps, tanks and containers. Also rooms with their fixed equipment might be used for purposes not involving radioactive substances.

CHAPTER 3

TYPES AND QUANTITIES OF MATERIALS TO BE CLEARED

Some examples are given below of the amounts and activities of radioactive wastes that might be cleared from regulatory control for disposal, reuse or recycling in the Nordic countries. The wastes considered arise from the operation and dismantling of nuclear power plants and research facilities as well as from the use of radioisotopes in industry, hospitals and research institutes.

3.1 Nuclear wastes

Operation of nuclear power plants

On the basis of experience gained at the Finnish and Swedish nuclear power plants, the following examples can be given of types and quantities of cleared materials arising from the operation of nuclear power plants /8, 9, 10/:

- Typically 10 m³ of contaminated oil per reactor unit is annually cleared for incineration or reuse. The activity concentration in oil has been about 100 Bq/liter on the average.
- The typical amount of very low level scrap metal from the operation a NPP varies from some tonnes to a few tens of tonnes per year. Occasionally the amounts can be much higher: as the copper-aluminium condensers of the Swedish and Finnish BWRs were replaced by titanium ones, totally about 2000 tonnes of scrap metal was cleared to be used as raw material for the metal industry. The average activity concentration in this scrap metal was some hundreds of Bq/kg.
- Annually about 15 tonnes of very low level trash waste from the Olkiluoto NPP is cleared for burial in an on-site landfill. Typical clearance levels are 2 Bq/g for Cs-137 and 10 Bq/g for Co-60. From the Loviisa NPP, trash with total activity concentration below 1 Bq/g is cleared for disposal in a municipal landfill.
- The Ringhals PWRs have a permit to dispose of ion exchange resins from condenser cleanup systems and bottom blow systems in an on-site landfill. The activity concentration is limited to 5 Bq/g and the total amount to 100 m³/a.

The most important nuclides in wastes from reactor operation with respect to clearance are

- activation products, such as Cr-51, Mn-54, Fe-55, Co-58, Co-60, Ni-63, Zn-65, Ag-110m, Sb-125 and
- fission products, such as Sr-90, Ru-106, Cs-134, Cs-137 and Ce-144.

The concentrations of actinides in reactor wastes are orders of magnitude lower than those of fission products, thus they are not normally the limiting nuclides with respect to clearance.

Dismantling of nuclear power plants

On the basis of studies made of decommissioning of the Finnish NPPs /11, 12/, Table 3-1 gives estimates of the main types and amounts of dismantling wastes (per reactor unit) that can potentially be cleared from regulatory control. The estimates are based on dismantling of systems and structures in the controlled areas, with activity concentration not more than 10 Bq/g.

Table 3-1: Quantities of waste from decommissioning of NPPs

Waste type	Olkiluoto NPP BWR 710 MWe	Loviisa NPP PWR 440 MWe
Reinforced concrete	35 000 m ³	25 000 m ³
Scrap steel	4 000 t	2 500 t
Other metals (copper, titanium, aluminium)	90 t	-
Insulation materials (e.g. mineral wool)	200 t	100 t

In addition, large amounts of low level trash waste, sand from off-gas delay tanks and contaminated liquids are generated during decommissioning.

For comparison, the dismantling wastes to be disposed of in a repository have been estimated to 15,000 m³ for a Olkiluoto NPP unit and to 10 000 m³ for a Loviisa NPP unit.

Swedish decommissioning studies indicate somewhat larger amounts of e.g. scrap steel to be cleared /13, 14/.

The most important nuclides in contaminated components and structures are the same as given above for reactor wastes. In activated structures, such as the biological shield, the dominant nuclides may include tritium, Cl-36, Ca-41, Fe-55, Co-60, Ni-63, Cs-134, Ba-133, Sm-151, Eu-152, Eu-154 /13, 14, 15/.

Dismantling of other nuclear installations

There are three large and several small research reactors in the Nordic countries. The large ones are situated in Risø (Denmark), Halden (Norway) and Studsvik (Sweden). A decommissioning study /16/ for the Halden reactor indicates that from the first decommissioning stage (safe storage), approximately 500 m³ of waste to be cleared will arise. The largest fraction of this waste is steel, insulation materials and water. Complete decommissioning would generate much larger amounts of wastes.

In Sweden, decommissioning of the small Ågesta NPP is underway. One small research reactor, the R1 reactor in Stockholm, has already been dismantled and the experiences have been reported /17/. During this decommissioning project, 750 tonnes of concrete (with activity concentration below 5 Bq/g) and 13 tonnes of scrap metal were cleared.

Decommissioning studies are available for the TRIGA reactor in Otaniemi (Finland) /18/ and for the JEEP II reactor in Kjeller (Norway) /19/. The estimated waste quantities to be cleared are reported only for the TRIGA reactor; those wastes constitute about 450 tonnes of concrete, 3 tonnes of graphite, 3 tonnes of steel and one tonne of aluminium.

At Kjeller, dismantling of a small reprocessing pilot plant (URA-facility) is being completed /20/. Large amounts of waste, mainly concrete and scrap metal, arose from the dismantling. Part of the waste was cleared on the basis of surface activity measurements; the applied limits were 2 Bq/cm² for beta-gamma emitters and 0.2 Bq/cm² for alpha emitters. The most important nuclides are Sb-125, Cs-134, Cs-137 and Pu-239.

Other significant future sources of dismantling wastes are e.g. hot cell facilities and waste treatment plants in the Nordic nuclear research centres.

In the licenced melting facility at Studsvik, 1500 tonnes of metal have been treated, e.g. the steam generators of the Ågesta NPP. About 230 tonnes of ingot have been cleared while the rest is stored for decay.

3.2 Wastes from the use of radioisotopes

Radioactive wastes from hospitals, industries and research institutes contain mainly radionuclides with short half-lives (less than one month for more than 90 % of waste). In Sweden the total activity of produced or imported radioisotopes is estimated to amount to 40-50 TBq in a year, excluding sealed radioactive sources and nuclides with very short half-life. In Finland the corresponding amount is about 10 TBq in a year. The most common radioisotopes are Mo/Tc-99, I-131, I-125, Xe-133 and Tritium.

After a sufficient decay period, most of the radioisotopes mentioned above are disposed into sewer systems or by landfill or are incinerated in accordance with the national regulations (see Chapter 7.2). In Sweden, about 300 tonnes of solid waste of this kind is cleared annually.

CHAPTER 4 RADIOLOGICAL ASSESSMENTS

Several analyses have been published regarding the radiological consequences of clearance of solid materials from regulatory control /3, 5, 6, 7, 15, 21, 22, 23, 24, 25/. These analyses consider some or all of the management options discussed in chapter 2.4 and assume a number of scenarios resulting in radiation exposure of persons involved with the cleared materials. As a result, the relationship between the activity level of the cleared material and the resulting radiation dose is obtained.

The following chapters contain a description of such analyses and summarize the typical results and the main conclusions that can be drawn from them. It seems that the ranges of results (dose for unit activity concentration in cleared material) are very large due to the varying assumptions and parameter values applied in the analyses. Consequently, decision on clearance can seldom be based on the results of such general analyses. Instead of that, the clearance levels for a given case can be derived by means of the calculational tools given in Appendix 1.

4.1 Landfill disposal

Scenarios and exposure pathways

In a radiological assessment of landfill disposal, it is appropriate to consider three different phases: operational, surveillance and long-term (post-surveillance) period. During the operational phase, two scenarios are important: the landfill operator scenario and the landfill fire scenario. The latter scenario is relevant only to such landfills where large amounts of combustible waste are in the open air for considerable periods, before the waste is covered with soil. Exposure via scavenging of objects from a landfill is considered unlikely in the Nordic conditions.

The landfill operator is exposed to external radiation from cleared waste and to internal radiation via inhalation of dust. In the landfill fire scenario, the potential exposure pathways of people living or working nearby are external exposure to airborne and deposited radioactive substances, inhalation of radioactive substances and ingestion of contaminated foodstuffs.

During the surveillance phase, activities that might easily cause radiation exposure are precluded. It is, however, possible that infiltrated water from the landfill is discharged to a lake, river or well outside the protection zone. Thereby people may be exposed by using the contaminated water as drinking water or for irrigation of vegetables.

During the long-term phase, when administrative control of the landfill site cannot be guaranteed anymore, a water well might be dug within the protection zone causing exposure of people as discussed above. It is also possible that a house will be constructed on a landfill or a road will be constructed through it. During the construction of a house or a road, workers may be exposed to external exposure or to internal exposure via inhalation of dust. Residents of the house might be subject to long-term external exposure.

An example of scenarios and exposure pathways is illustrated in Fig. 4-1.

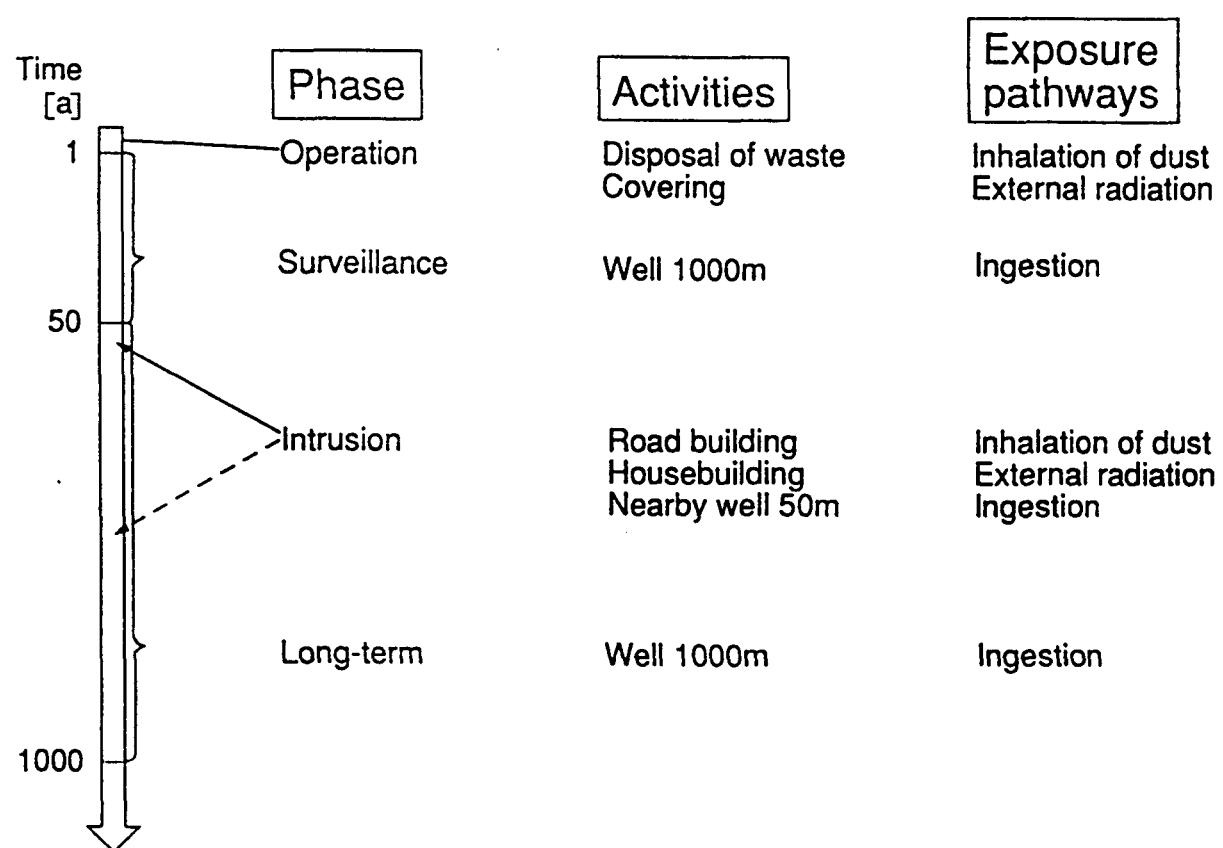


Fig. 4-1. An example of exposure pathways for landfill disposal /15/

Results of analyses

The **landfill operator scenario** has been analyzed in a number of reports /5, 6, 15, 22, 23, 24/. Table 4-1 shows a typical range of results. The wide range in the results of the different analyses is mainly due to the differences in parameter values, such as exposure duration, waste dilution factor, external radiation shielding factor and airborne dust concentration.

Table 4-1: Landfill operator scenario: range of doses ($\mu\text{Sv/a}$ for 1 Bq/g in cleared waste) and dominant pathways

Nuclide	Range of doses	Dominant pathway
C-14	< 0.01	Inhalation
Na-22	1 - 30	External
Co-60	1 - 100	External
Ni-63	< 0.01	Inhalation
Sr-90	0.03 - 1	Inhalation
Ru-106	1 - 30	External
I-131	0.3 - 10	External
Cs-137	1 - 30	External
Eu-152	1 - 100	External
Pu-239	3 - 300	Inhalation
Am-241	3 - 300	Inhalation

The **landfill fire scenario** is analyzed in an IAEA document /5/. It is assumed that 60 t of waste is involved in the fire. The resulting individual doses are in general much lower than the respective doses from the landfill operator scenario. Only for C-14 and Sr-90 the landfill fire might be a more limiting scenario.

In the **groundwater migration scenario** analyzed in /5/ it is assumed that the groundwater infiltrated through the landfill is discharged into a river, thus providing very high dilution of the released radioactive substances. Consequently, the resulting radiation doses are very low; for other nuclides than C-14 they remain below 0.0001 $\mu\text{Sv/a}$ for 1 Bq/g in cleared waste.

If the contaminated groundwater ends up in a well, the doses will be considerably higher, as indicated in /15/ and /23/. The results are very sensitive to some parameters specific to waste type and site, such as leach characteristics of the waste and hydrological properties of the landfill and its environs. Analyses indicate that the doses might vary from a few to hundreds $\mu\text{Sv/a}$ for 1 Bq/g of the most critical nuclides in waste. With respect to the well pathway the critical nuclides seem to be C-14, Cl-36, Sr-90, Cs-137 and Pu-239.

Analysis of the **road or house construction scenarios** is similar to that of the landfill operator scenario. The main difference is that intrusion is assumed to occur after the surveillance period; thus short-lived nuclides have decayed. The exposure time for a road or house builder is probably shorter than that for the landfill operator. Consequently the doses arising from road or house construction are likely to remain below the dose to the landfill operator.

Analysis of the **house residence scenario** is different. Exposure duration may be quite long but, on the other hand, the resident is more shielded against external radiation than a landfill operator. Besides external radiation, the resident may be exposed to internal radiation e.g. via

growing vegetables in his garden or via a well. The latter case was discussed above in context with the groundwater migration scenario.

The analyses /5, 15/ of the house residence scenario indicate individual doses varying from a few up to hundred $\mu\text{Sv}/\text{a}$ for 1 Bq/g for the most critical nuclides in the disposed waste. These nuclides include e.g. Sr-90, Cs-137, Eu-152 and Am-241.

In context with landfill disposal, the number of people exposed to cleared waste will probably remain low. Thus collective doses will not be crucial in determining the clearance levels, as indicated also in /5/.

4.2 Incineration

Scenarios and exposure pathways

Though incinerators are normally designed so that protection of its workers is ensured, the operators are to some extent exposed to penetrating radiation, and via inhalation of dust. Due to the high volume reduction in combustion, operations involving ash will probably lead to the highest exposures and consequently most consideration should be given to those operations.

Release of gases and particulates to the atmosphere as a result of combustion is another scenario to be considered. Consideration has to be given to the exposure pathways of external radiation caused by airborne materials and materials deposited on ground, inhalation of directly released and resuspended materials, and ingestion of radionuclides transported into food chains.

If incineration ash is disposed by landfill, the exposure scenarios and pathways are the same as for disposal of waste as such, except that the landfill fire scenario can be excluded. Apart from the volatile elements (H, C, Cl, I), the ratio between the concentration of radionuclides in ash and that in raw waste is the same as the mass reduction coefficient.

Results of analyses

Exposure from incineration is analyzed e.g. in references /5, 22, 23/. For the **incinerator operator scenario**, the range of doses given in Table 4-2 is obtained from these analyses. The reference incinerators are large municipal ones, thus the results are not applicable to small hospital-type incinerators. Analyses of small incinerators is included in /24/.

For the **atmospheric emission scenario**, an incinerator, having a 100 m high stack, is assumed to be situated in an urban area /5/. As incineration occurs frequently, average meteorological conditions are assumed. The results of analyses indicate that individual doses would be quite low. The maximum doses, about 0.01 $\mu\text{Sv}/\text{a}$ for 1 Bq/g in waste, would be caused by Pu-239 and Am-241 (plume inhalation) and by C-14 (ingestion of foodstuffs).

Also the collective doses from the incinerator residential scenario deserve some consideration, as the number of exposed people may be high. Some calculations /5, 24/, however, indicate that the incinerator operator scenario tends to be more limiting.

Table 4-2: Incinerator operator scenario: range of doses ($\mu\text{Sv/a}$ for 1 Bq/g in incinerated waste) and dominant pathways

Nuclide	Range of doses	Dominant pathway
Na-22	10 - 20	External
Co-60	5 - 20	External
Sr-90	0.1 - 1	Inhalation
Ru-106	0.5 - 1	External
I-131	0.1 - 1	External
Cs-137	0.3 - 3	External
Pu-239	5 - 50	Inhalation
Am-241	5 - 50	Inhalation

4.3 Recycling of metals

Scenarios and exposure pathways

In recycling of metals, the following main phases can be distinguished (as illustrated in fig 4-2):

- Storage and pretreatment
- Melting
- Manufacturing
- Use of products
- Use or disposal of byproducts

Storage of metal scrap causes external dose to workers at the scrap yard. Pretreatment may include sorting and fragmentation of scrap. This may give rise to external and inhalation doses to workers.

During the melting process, external radiation dose may be received by the workers transporting the scrap and those at the melting furnace. The melting will result in dust and fumes and, in spite of the ventilation system, occasionally high dust concentrations can occur in the melting hall. Thereby the workers are exposed via inhalation. Doses from ingestion of dust is expected to remain low in comparison to those from inhalation of dust.

During melting, radioactive substances in the scrap will be distributed between ingot, slag and fumes/dust depending on the chemical properties of each radionuclide. In slag and dust, the activity concentrations of some radionuclides (e.g. Cs-137) may be considerably higher than in the scrap wherefrom they originate.

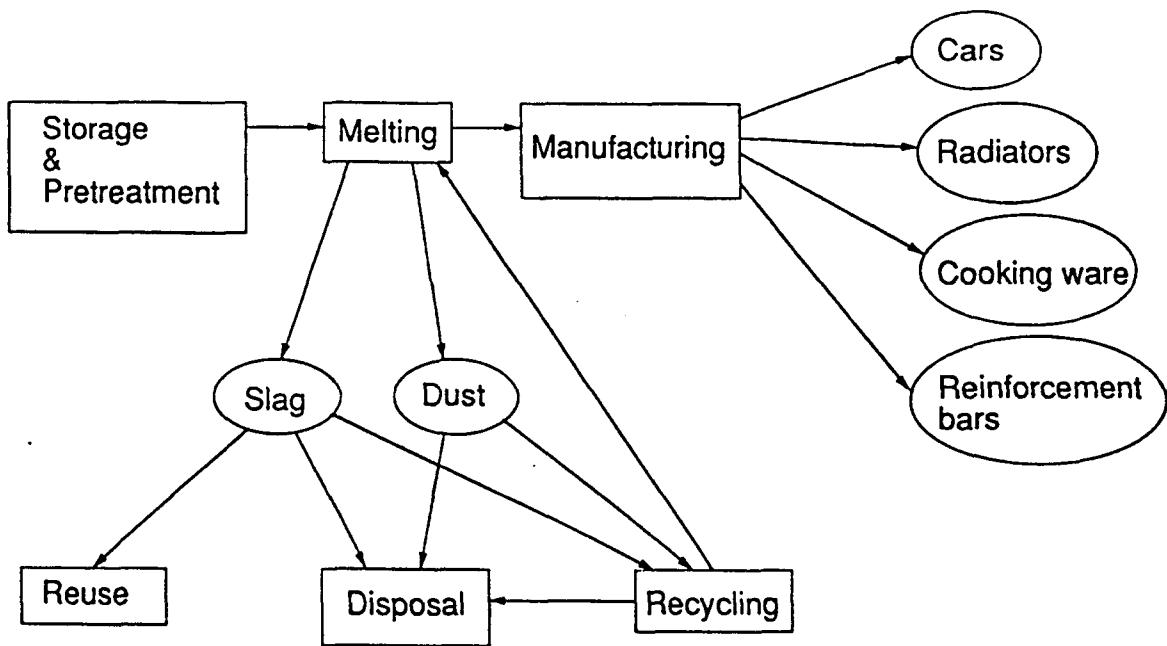


Fig. 4-2 A Scheme for recycling of metals /15/

After melting the metal will be cast in ingots, which will be further processed e.g. by hot and/or cold rolling into sheet metal and blanks. These items will be used as raw material for a variety of products. Casting and rolling are normally automated to a great extent and generate less dust than melting, thus both the external and inhalation doses from these operations are expected to remain below those from melting.

Due to the diversity of potential products, it is difficult to predict the conditions during the fabrication phase. The workers will be exposed to external radiation from the blank, sheet or coil of metal they use as raw material. Some dust may arise from cutting of metal but the resulting air concentrations will probably be fairly low.

In case of unconditional clearance, the scrap metal may end up as consumer products. Most of them cause mainly external exposure to their users. The most sensitive products in this respect include vehicles, reinforcement bars, thermal radiators, domestic appliances and furniture. Some internal exposure may arise if cooking ware is made of the cleared metal, as minute amounts of the metal will be dissolved in foodstuff and ingested.

In case of conditional clearance, at least the "profile" of potential end products is generally known, allowing a case-by-case definition of exposure scenarios and pathways and exclusion of the most sensitive end products from the radiological impact analysis.

The main byproducts from recycling of metals are

- slag from melting,
- sludges from further processing of metal and
- dust from the air cleaning system.

As illustrated in fig. 4-2, the byproducts may be disposed of in a landfill, reused as filling material or recycled. In the former cases, the exposure scenarios and pathways are the same as described in chapter 4.1. The scenarios and pathways for recycling of the byproducts are similar to those for recycling of metals.

If the air cleaning system of the steelworks is inefficient, dust and fumes are released in the environment causing exposure of nearby people.

In the licenced melting facilities, storage, pretreatment and melting of metal scrap is performed under radiation protection control and the byproducts (slag and dust) are treated as radioactive waste. Thus, only the exposures arising after clearance of the ingots (manufacturing and use of products) are relevant for this option.

Results of analyses

Radiation exposure arising from recycling of metal scrap is analyzed e.g. in the reports of the IAEA /7/, KEMAKTA /15/ and CEC /3/. They assume recycling of 100, 5000 and 10,000 tonnes of steel respectively, originating from large nuclear facilities. Consequently, the radioactive substances of interest are the long-lived nuclides most commonly encountered as contaminants of nuclear components. The calculated ranges of maximum individual doses from **storage, pretreatment, melting and manufacturing of steel scrap** are given in Table 4-3.

The variations in the results are mainly due to the different amounts of steel that are assumed to be recycled. Other parameters for which quite differing values have been selected in the studies are air dust concentration and distribution of radionuclides during melting.

Also exposure from the **use of products made of recycled metal** is analyzed in the reports mentioned above. The ranges of the highest individual doses for some important nuclides in metal scrap are given in Table 4-3. The large variations in the results are due to different assumptions about the exposure time and geometry, radionuclide distributions and dilution factors.

Table 4-3: Steel recycling scenarios: ranges of doses ($\mu\text{Sv/a}$ for 1 Bq/g in scrap) and dominant pathways

Nuclide	Ranges of doses (dominant pathway)			
	Steelworks		Consumer products	
Co-60	1 - 10	(ext)	10 - 100	(ext)
Sr-90	0.2 - 2	(inh)	< 0.1	(ing)
Cs-137	0.7 - 5	(ext)	< 10^{-2} - 20	(ext)
Pu-239	1 - 1000	(inh)	< 10^{-3} - 2	(ing)
Am-241	1 - 1000	(inh)	< 10^{-3} - 5	(ing)

ext = external inh = inhalation ing = ingestion

Due to the potentially great number of exposed people, the collective dose arising from products made of recycled steel may be of concern. The collective dose commitments for the public from recycling of 100 and 10,000 tonnes of steel scrap are analyzed in the reports of the IAEA /7/ and CEC /21/. Both studies end up with the same order-of-magnitude result, 0.1 mmanSv per tonne of cleared steel scrap with activity concentration 1 Bq/g. External exposure from strong gamma emitters, such as Co-60, is the dominating pathway.

Exposure from the use or disposal of byproducts from recycling, such as slag, sludge and dust, are also dealt with to some extent in the reports of the IAEA, KEMAKTA and CEC /7, 15, 3, 21/. For instance, the use of slag as a constituent of cement or asphalt is considered. For some nuclides (e.g. Cs-137) which tend to concentrate in slag during melting, the radiation exposure due to recycling or disposal of slag might be considerably higher than the corresponding exposure from the use of consumer products made of recycled steel.

Recycling of dust and sludges from steelworks might in some cases give rise to relatively high individual doses: e.g. tens of μ Sv/a for 1 Bq/g of Cs-137 in the original scrap /3, 25/. Valuable metals containing also radioactive isotopes (such as Zn-65 and Ag-110) might be recovered from the sludges and dust, resulting in further buildup of the concentrations of those radionuclides.

Recycling of other metals than steel, e.g. aluminium and copper, is also analyzed at some length in the reports of the IAEA and CEC /7, 21/. The results indicate roughly similar or lower doses (for 1 Bq/g in cleared metal scrap) than those from recycling of steel.

4.4 Reuse of components or buildings

Scenarios and exposure pathways

Components of interest include hand tools and larger items, such as motors, pumps, pipes, tanks and furniture. The most likely exposure pathway is external exposure (including skin dose from contact with the component) during the use of hand tools or during the installation, repair and maintenance of larger items. Secondary ingestion of contamination, transferred from the surfaces of the component to hands and then to mouth, might also occur. A cleared tank might be used as a water reservoir, thereby causing internal exposure via ingestion. Inhalation of resuspended radioactive substances may be an important exposure pathway in context with mechanical decontamination of the surfaces.

Two scenarios are of importance for reuse of buildings: normal occupancy and building renovation. The main exposure pathway for the normal occupancy scenario is external exposure to people working in the building after its unrestricted release. Sitting on a contaminated surface may cause significant skin doses. In the renovation scenario, the building is assumed to be modified e.g. by surface stripping or by removing walls. In that case, inhalation dose may be the dominating exposure pathway.

Results of analyses

Exposure from **reuse of cleared components** (hand tools, motors, pumps, tanks etc.) is analyzed in the reports of the IAEA and CEC /7, 21/. The results indicate that the highest individual external doses from strong gamma emitters would be in the range of 0.1-1 $\mu\text{Sv}/\text{a}$ for a concentration of 1 Bq/cm^2 on the surface of the component. The maximum individual internal doses, predominantly from alpha emitters, would fall in the range of 10-100 $\mu\text{Sv}/\text{a}$ for a surface contamination of 1 Bq/cm^2 .

Exposure from **occupancy and renovation of cleared buildings** is analyzed in the report of the IAEA /7/. The highest individual external doses from strong gamma emitters are calculated to fall in the range of 20-90 $\mu\text{Sv}/\text{a}$ for a surface activity of 1 Bq/cm^2 . The respective internal doses from Pu-239 and Am-241, arising from the removal of walls, would vary from 30 to 50 $\mu\text{Sv}/\text{a}$. From surface stripping (paint removal, surface grinding), somewhat higher inhalation doses might arise /2, 22/.

4.5 Conclusions

Radiological analyses indicate that doses arising from clearance of waste vary widely depending on nuclide. Provided that inhalation doses may occur, the most restrictive nuclides, per unit activity in cleared material, are long-lived alpha emitters, such as Pu-239 and Am-241. If external exposure is the dominating pathway, strong gamma emitters, such as Co-60 and Cs-137, are the most crucial nuclides. In some cases, when highest doses are received via ingestion of contaminated water or foodstuffs, the strong beta emitters such as Sr-90 may be the most restrictive nuclides. The doses arising from nuclides having only weak beta emission are, in general, orders of magnitude lower than the respective doses from strong gamma or beta emitters.

In addition to the radiological characteristics, also some other properties of radioactive substances are of importance for the determination of clearance levels. A long half-life contributes in particular to collective dose commitments and to long-term doses (i.e. doses beyond institutional surveillance). The volatility, leaching behaviour, solubility and mobility in ground of the substance may also substantially affect the radiation exposure depending on the scenario of interest.

The main options for management of cleared materials are landfill disposal, incineration, recycling and reuse. For these options, a variety of potential exposure scenarios and pathways can be identified, the relative importance of which depend on type and nuclide composition of cleared material.

Often it is unpractical to strive for "universal" clearance levels that cover all potential waste types, nuclide compositions, scenarios and exposure pathways. Such levels would be extremely low, even below the typical activity concentrations in natural materials. Narrowing of options to be considered in the determination of clearance levels is thus necessary.

Nuclear wastes contain a great number of nuclides. There are, however, approximate relations between the activities of different nuclides (see chapter 5.3). Thus, most of the nuclides can be neglected and the determination of clearance levels for nuclear wastes can be based on a few most restrictive nuclides, such as Co-60 and Cs-137. With this approach, an adequate control of the nuclide composition of wastes is constantly required.

Wastes from the use of radioisotopes normally contain one or a few nuclides, which are known if there are due records of the wastes. Thus it is practicable to set clearance levels so that they are specific to each nuclide or a group of nuclides having similar radiological characteristics. Due to the type and characteristics of non-nuclear wastes, recycling and reuse can normally be excluded, thus simplifying the determination of clearance levels.

In a draft report of the IAEA /2/, unconditional clearance levels for solid wastes are recommended (see in Table 2-1). The clearance levels suggested in the CEC report /3/ for recycling of metals are of the same order of magnitude for the critical nuclides in nuclear wastes.

In many cases, the unconditional clearance levels are too low as compared with the actual activity concentrations or possibilities to determine the activity in waste. In that case, one should consider the option of conditional clearance, where the activity constraints are determined on the basis of case-by-case evaluation. In Appendix 1, guidance is given for radiological assessments of such conditional clearances.

CHAPTER 5

ACTIVITY MONITORING

A prerequisite for clearance is that compliance with clearance levels has been shown with sufficient reliability. For this purpose, a carefully planned monitoring programme is needed that may include measurements of activity and dose rate, recordkeeping and often also theoretical calculations. Such monitoring programme may include the following elements: preliminary surveys, determination of scaling factors, monitoring for clearance and documentation.

This chapter describes a general approach for a monitoring programme in context with conditional or unconditional clearance. More information on the subject is given in a draft ISO standard /26/, two IAEA reports /27, 28/, a report made by ABB Atom /29/ and a CEC report /30/.

5.1 Decay characteristics of important radionuclides

An important factor affecting the selection of monitoring strategy is the decay characteristics of the nuclides of interest. Often nuclides are grouped into strong gamma emitters, weak gamma (or x-ray) emitters, strong beta emitters, weak beta emitters and alpha emitters according to their decay properties. Table 5-1 gives a summary for a number of relevant nuclides.

Besides for the selection of measuring instruments, decay characteristics are important from a self-shielding point of view. The half-value thickness for the unscattered flux of strong gamma emitters is 12 - 17 g/cm² while it is typically 3 - 5 g/cm² for weak gamma emitters (energy 100 keV). The penetration depth for strong beta emitters is several tens of mg/cm² while weak beta and alpha emitters are significantly attenuated even in a contaminant layer or a few centimeters of air.

5.2 Preliminary surveys

The preliminary surveys may have different objectives, viz:

- to identify important nuclides,
- to segregate materials so that batches well above the clearance levels are excluded from further monitoring,
- to provide a redundant control so that administrative or equipment failures do not result in the release of considerable activity quantities,
- to find out the activity distribution in material, in particular to locate "hot spots" and
- to find out scaling factors for nuclides which are difficult to measure.

Table 5-1 Decay characteristics of important nuclides in cleared materials

Nuclide	Half-life	Formation, Parent	Decay mode, Principal energy (MeV)
H-3	12 a		B, Li weak β (0.019)
C-14	5730 a	N	weak β (0.156)
Na-22	2.6 a	F	strong γ (1.274)
Cl-36	300 000 a	Cl	strong β (0.714)
Cr-51	28 d	Cr	fairly weak γ (0.320)
Mn-54	312 d	Fe	strong γ (0.835)
Fe-55	2.6 a	Fe	E.C., X-rays
Co-58	71 d	Ni	strong γ (0.811)
Co-60	5.3 a	Co	strong γ (1.17, 1.33)
Ni-63	96 a	Ni	weak β (0.067)
Zn-65	244 d	Zn	strong γ (1.116)
Sr/Y-90	28 a	fission	strong β (2.270)
Nb-94	20 000 a	Nb	strong γ (0.871)
Tc-99	200 000 a	Mo, fiss.	fairly weak β (0.292)
Ru/Rh-106	1 a	fission	strong γ (0.512)
Ag-110m	250 d	fission	strong γ (0.885)
Sb-125	2.8 a	Sb	fairly strong γ (0.428)
I-131	8 d	fission	fairly strong γ (0.365)
Cs-134	2.1 a	fission	strong γ (0.796)
Cs-135	3 000 000 a	fission	fairly weak β (0.210)
Cs/Ba-137	30 a	fission	strong γ (0.662)
Ce-144	284 d	fission	weak γ (0.133)
Eu-152	12.7 a	Eu	strong γ (1.408)
Pu-239	24 000 a	U	α (5.155)
Am-241	430 a	U	α (5.485), weak γ (0.060)

Preliminary surveys may include collection, maintenance and examination of records, samplings, activity or dose rate measurements or use of calculational methods.

Recordkeeping is the most essential part of a preliminary survey in connection with the use of radioisotopes, as only one or a few radionuclides are normally in use at each workplace or laboratory room. As soon as a waste package or a worn-out component arise, it should be furnished with a label identifying its origin and likely nuclide content.

At nuclear facilities, careful recordkeeping is of great help in the determination of nuclide contents of materials and scaling factors for nuclides that are difficult to measure. Worn-out components that arise from systems or areas where contamination is unlikely, should be marked or stored apart from potentially contaminated components. To facilitate activity monitoring

during decommissioning, records of the results from the operational activity monitoring should be systematically kept.

Field measurements in context with preliminary surveys are generally based on fast methods and instruments. Portable dose rate or surface contamination monitors or gate type measurement systems are used for coarse segregation of materials and for the determination of activity distribution.

Sampling and laboratory analyses are normally needed for the determination of nuclide composition of material. On a routine basis, only gammaspectrometric analysis of the samples is normally made. Alternatively, a mobile semiconductor/gammaspectrometer system can be used, particularly if sampling of material of interest is difficult.

In the determination of scaling factors for the socalled difficult-to-measure nuclides (e.g. C-14, Fe-55, Ni-63, Sr-90, actinides), complicated preparation of samples is needed. Appendix 2 includes a description of these methods. Scaling factors are discussed in chapter 5.3.

Calculational methods can be used to find out the nuclide content and to estimate activity levels in activated materials, such as the biological shield of a reactor. When the activities are calculated in this way, the results are often very approximate as some parameter values (trace concentration of parent elements, neutron flux outside reactor vessel) are not known with an adequate accuracy /37/.

Calculational methods can also be used for the estimation of surface contamination and distribution of activation products within reactor systems. An example of such method is the BKM-CRUD-code developed by the ABB Atom /31/. Such codes provide a supplementary method e.g. for the determination of scaling factors and for the estimation of activity levels in dismantling wastes.

5.3 Scaling factors

Use of scaling factors is an established method for the estimation of the content of difficult-to-measure nuclides in wastes. It involves finding out the activity ratio between the nuclide of interest and a suitable gamma emitter that can be directly measured from the waste batch. There are no universal scaling factor but they are specific to each plant (due to e.g. differences in cobalt content of materials) and they tend to vary somewhat between different waste types of the same plant.

In the determination of scaling factors, a suitable gamma emitting reference nuclide must be selected. Difficult-to- measure corrosion products, such as Fe-55 and Ni-63, are normally related to Co-60. With regard only to the parent elements, Fe-59 or Mn-54 would be more suitable reference nuclides for Fe-55, and Co-58 likewise for Ni-63. The relatively short half-lives of Fe-59 and Co-58, however, limit their applicability as reference nuclides if the "age" of the activated material is not known.

The difficult-to-measure fission products, such as Sr-90, Tc-99, I-129 and Cs-135 are generally related to Cs-137. From chemical point of view, an anionic reference nuclide, such as Ru-106, might be more appropriate for Tc-99 and I-129 /40/. The scaling factors seem to be different for fuel failure conditions and for non-fuel failure conditions (when tramp uranium on the fuel surface is the dominating source of fission products in reactor circuits). Preference should be given to the former scaling factors as most of the fission product activity in waste is expected to accumulate in context with fuel failures.

The birth and leakage processes of actinides resemble those of fission products. Experimental results, however, indicate that there is no perfect reference nuclide amongst gamma emitting fission products. Cs-137 is not the most suitable one because of its very different chemical behaviour in reactor circuits. From the chemical point of view, Ce-144 would be a more appropriate reference nuclide, but being a relatively weak gamma emitter, it is difficult to measure from waste batches. Co-60 is recommended as a reference nuclide for actinides /29/, though data with Cs-137 as a reference nuclide is also found in literature /32/.

As there are two potential reference nuclides for most of the difficult-to-measure nuclides, a conservative approach is the use of "a double scaling factor". The activity A_i of the difficult-to-measure nuclide of interest is then obtained as follows:

$$A_i = \text{MAX} \{ SF_1 \cdot A_1, SF_2 \cdot A_2 \},$$

where A_1 and A_2 are the activities of the reference nuclides 1 and 2 and SF_1 and SF_2 are the respective scaling factors.

On the basis of waste type, at least two sets of scaling factors should be defined: primary water related set and surface contamination related set. The former set should be applied for e.g. trash and contaminated concrete, as most of the activity in those waste types is expected to arise from process leakages. The latter set of scaling factors is applicable to e.g. contaminated metal scrap.

The scaling factors should be selected conservatively, i.e. so that in large waste quantities the actual activities of the difficult-to-measure nuclides remain, with high confidence, below the calculated values. In spite of this, the actual activity in a single waste package might incidentally exceed the one calculated by means of the scaling factors.

One might question if the scaling factor approach allows a "hot spot" of significant quantity of pure beta or alpha emitter inside a waste batch to pass the monitoring for clearance. In cleared materials from NPPs, the difficult-to-measure nuclides seldom seem to be the limiting ones and they almost without exception appear together with some gamma emitting nuclide. Thus violation of the radiation protection criteria for clearance due to the scaling factor approach seems unlikely.

More discussion of scaling factors is included in reference reports /29, 32, 33/. Table 5-2 gives a summary of scaling factors for some facilities and waste categories. It can be seen that except

Table 5-2

Examples of relations between nuclides in reactor waste

Nuclides	Activity ratio								
	NEA Quest. ¹		EPRI Quest. ²		ABB Atom ³		SFR ⁴	VLJ-rep. ⁵	Loviisa ⁶
	PWR	BWR					Resins	Coolant	
C-14/Co-60	0.001 - 0.1	0.05	0.004	-	0.003	0.001	-	-	
Fe-55/Co-60	-	1.2	0.9	1.7	0.4	-	-	-	
Ni-63/Co-60	0.05 - 1	0.35	0.38	0.1	0.35	0.1	0.85	0.7	
Sr-90/Cs-137	0.001 - 0.1	0.005	0.006	0.6	0.05	0.1	0.006	0.0005	
Tc-99/Cs-137	$10^{-4} - 10^{-3}$	0.004	0.0003	-	$5 \cdot 10^{-5}$	0.0008	-	-	
Cs-135/Cs-137	$10^{-5} - 10^{-6}$	-	-	10^{-5}	$3 \cdot 10^{-6}$	$3 \cdot 10^{-6}$	-	-	
Pu-239/Cs-137	< 10^{-3}	0.0003	0.0003	$5 \cdot 10^{-7}$	$7 \cdot 10^{-5}$	10^{-5}	$2 \cdot 10^{-6}$	10^{-5}	

¹ Data from 14 member countries of NEA /32/² Data from 52 US NPPs, log mean averages for low activity waste samples /33/³ Calculated surface contamination, one year after shutdown of plant /30/⁴ Waste in the Swedish SFR repository in the year 2010 /38/⁵ FSAR of the Finnish VLJrepository /39/⁶ Loviisa NPP in Finland, measured in ion exchange resins and primary coolant /40/

for Fe-55 and Ni-63, the activities of the pure beta and alpha emitters are one to several orders of magnitude lower than the activity of the gamma emitting reference nuclides.

5.4 Monitoring for clearance

Prior to monitoring for compliance with clearance levels, several factors must be considered. Firstly, the clearance levels with averaging rules and scaling factors (for the critical nuclides that are not measured) must be determined. The nuclide composition and preferably also the approximate activity distribution in the material of interest should be known on the basis of preliminary surveys.

Secondly, the monitoring methods and instrumentation should be carefully planned taking account of the material type and character of its radioactive content as well as the practical problems related to measurements or sampling. These issues are discussed below.

Direct gammaspectrometric monitoring of waste packages (drums, bales or sacks) is applicable to low density waste, e.g. trash, and for waste packages of higher density, if the activity is known to be uniform (e.g. drum of solidified waste or oil). At Swedish and Finnish NPPs, such measurement systems are used. The system consists of a collimated semiconductor detector with a multichannel analyser and a scanning mechanism (e.g. rotation and elevation) for the waste package /34/. By means of such a scanning mechanism, the results are made roughly independent of the activity distribution in the waste package. Experiences indicate that MDAs (Minimum Detectable Activity) below 10 kBq for waste packages of 10-100 kg can be reached with a counting time of 100 s, if the system is properly shielded with respect to background radiation. The energy and efficiency calibration and check measurements of the system can be made by means of a set of point calibration standards.

Mobile gammaspectrometric systems can be used for direct measurements of surface contaminated objects with simple geometries and uniform activity distribution, such as walls or metal plates. Calibration can be based on measurements with point source standards and supplementary calculations.

Sampling and laboratory analyses is the most common monitoring method for liquids and for surface contaminated materials. There are different sampling methods depending on type of surface and contamination: smear sampling for removable contamination, scrape sampling for rough surfaces or for fixed contamination, and an electrochemical method for metals (used e.g. at the Loviisa NPP /35/). Sampling by cutting pieces or by drilling can be used for massive objects, such as condensers, air filters and biological shields. If only non-fixed contamination is sampled and the ratio non-fixed/total contamination (removal factor) is not known, a value of 0.1 can be assumed /26/. Measurement systems normally used in laboratories can easily detect e.g. contamination levels included in the transport regulations (0.04 Bq/cm² for alpha, 0.4 Bq/cm² for beta-gamma emitters), if smear samples are taken from a surface covering 100 cm².

Preparation of samples and activity determination of the difficult-to-measure nuclides is discussed in appendix 2.

Sampling should be systematic and statistically representative. Large surfaces should be divided into grids to facilitate systematic sampling. If a random sampling approach is adopted, the number of samples n required for statistical acceptability (error of less than 25% at a confidence level of 90%) is given by /36/:

$$n > 45 \cdot s^2 / m^2$$

where s is the standard deviation and m the mean of the measured activity for n samples. If, however, the hot spots have been located e.g. by means of a portable surface contamination monitor, a much smaller number of samples is enough to obtain a conservative estimate for the contamination. Often it is advisable to reduce the number of radiochemical analyses by combining several samples.

Field instruments, such as gamma or surface dose rate meters or scintillation counters can only be applied to monitoring for clearance, if there is one dominating nuclide or a few nuclides with almost fixed activity ratios. The measured dose rates or count rates can then be converted into activity concentrations by means of calibration sources and simple calculations. To reach an adequate MDA, the background radiation level should be low.

If the surface to be measured is not accessible to field instruments or the background radiation level is too high, a combination of smear sampling and surface contamination measurement of the wipes can be used. The result indicates only the non-fixed activity on the surface.

Monitoring for clearance is often needed for levels close to the lower detection limit of the measurement system. Consequently, calibrations, functional checks and background level determinations of the system should be done more frequently than normally.

If the material to be monitored contains a mixture of nuclides and the clearance levels are nuclide specific, the following condition should be met:

$$\sum_i (A_i/C_i) < 1$$

where C_i is the clearance level for nuclide i and A_i is the measured (or calculated by means of scaling factor) activity concentration for nuclide i . The averaging rules define the maximum volume and surface area for the calculation of activity concentration.

If a facility to be decommissioned has, on the basis of monitoring, been divided in parts that are below and above the clearance levels, it should be remembered that dismantling activities may be contaminating as such. Consequently, a final survey should be made after dismantling in order to check whether the earlier monitoring for clearance is still valid. The monitoring should be scheduled in parallel with dismantling operations.

5.5 Documentation

The main documentation for activity monitoring consists of the implementor's guidelines and the records of measurements. They should contain at least:

- description of the material to be monitored with the records of its origin and earlier measurements,
- clearance levels with averaging rules,
- description of methods, instruments and locations used for monitoring,
- counting times, allowable background readings, detection limits,
- calibration and checking procedures with conversion factors,
- scaling factors for the critical nuclides that are not measured and
- the primary measurement results with date and the calculated results with the description of the calculation methods.

5.6 Economic considerations

The main economic aspects related to clearance are the savings in waste management costs, the recycling/reuse value of scrap material, component or building and the cost of monitoring programmes. The latter issue is discussed below.

The cost of a monitoring programme depends on the type and amount of the material and the composition and distribution of its radioactive content. These factors together with the applied clearance criteria to a great extent determine the methods, instrumentation and personnel needed for the monitoring programme. If the required equipment and services are readily available at the facility where the material arises (e.g. for the monitoring of wastes to be disposed of), the extra costs due to monitoring for clearance are relatively low.

By experience from nuclear facilities, monitoring for clearance of bulky materials, such as trash, heavy components and concrete structures, is laborious but still the costs are low in comparison with the savings. The same is true for materials like oil and chemicals that, in spite of the fairly small amounts, are difficult to condition for disposal. On the other hand, the monitoring costs for e.g. middle-sized metal objects (a few kg - 100 kg) of complex shape may be much higher than their disposal cost.

Due to the limited number of radionuclides, monitoring for clearance in context with the use of radioisotopes is generally easier than that at nuclear facilities. Thus most of the wastes that meet the clearance levels are generally cleared.

5.7 Conclusions

In principle, methods and instrumentation exist for monitoring all kind of wastes and nuclide compositions so that compliance with even very stringent clearance levels can be shown. But in practice, it is prudent to perform an overall monitoring of only strong gamma emitters from batches of wastes to be cleared. The activity concentrations of difficult-to-measure nuclides, mainly pure beta or alpha emitters, can be estimated by means of scaling factors. Such a factor

indicates the ratio between the activity of a nuclide of interest and that of a suitable strong gamma emitter, such as Co-60 or Cs-137. This ratio should be such that it leads to an overestimation of the activity of concern in large material quantities. For the determination of the scaling factors, regular sampling and demanding laboratory analyses are needed. These are described in Appendix 2.

A two-stage monitoring programme is generally required. The first stage, preliminary surveys, aims at segregation of materials on the basis of activity, identification of nuclide compositions and "hot spots", determination of scaling factors and ensuring that considerable activity quantities are not cleared by mistake. The second monitoring phase should demonstrate compliance with the clearance levels with high confidence. Thorough documentation of the monitoring procedures and results of measurements is necessary.

To facilitate monitoring for clearance in context with decommissioning of a nuclear power plant, a programme consisting of systematic activity measurements and documentation should be established during the operational phase.

It seems economically attractive to clear from regulatory control most of the materials that meet the clearance criteria, provided that sites or facilities for the disposal, treatment or recycling of those materials are available.

CHAPTER 6

REGULATORY CONTROL

All clearances of radioactive materials shall be based on national regulations. For small users of radioisotopes, the regulations and detailed regulatory guides are the main control methods. But for practices where large amounts of radioactive wastes arise, more effective regulatory instruments are needed. These are described below.

6.1 Application for clearance

Before clearance of radioactive material from regulatory control, the waste producer must submit an application to the competent authority. The application for unconditional clearance should contain at least the following information:

- Description of the origin, characteristics and the likely radionuclide composition of the material.
- Description of the methods to be used for the determination of the activity of the material. Such methods are described in chapter 5 of this report.

After approval of the application, the material that has been monitored to meet the clearance levels, can be released without regard to its future destination. The regulator controls by occasional inspections that activity monitoring and other clearance procedures are performed properly.

An application for conditional clearance must include the same information as that for unconditional clearance, and in addition:

- A general description of methods to be used for recycling or disposing of the material. An identification may be required of the organization or site to which the cleared material is transferred.
- An analysis of radiation exposure arising from the selected management option; the activity constraints are determined on the basis of this analysis. Calculational tools for such analyses are described in Appendix 1 of this report.

After approval of the application, the material that has been monitored to meet the clearance levels can be released. A permit for conditional clearance can apply to a single quantity of material or it can be continuously valid in case that the material arises repeatedly and its disposal or recycling method remains unchanged.

Besides the control of the clearance procedures, the regulator should occasionally assure that the fate of the released material complies with that described in the application. This control may also include sampling of materials arising e.g. in the recycling process and comparison of measured activity concentrations with the results of the radiological analysis included in the

clearance application. It should be remembered, however, that the receiver of the cleared material is not under regulatory control; thus the sampling should occur on voluntary basis or from products available on the free market.

The transport of cleared materials should be carried out in accordance with the regulations for the transport of radioactive materials, particularly with regard to surface contamination. If the material has also other hazardous characteristics than ionizing radiation, the selected clearance option should comply with the pertinent regulations for that type of material.

6.2 Documentation and quality assurance

The waste producer must describe in detail the clearance procedures in his guidelines. He shall also keep records of cleared materials and present, e.g. annually, a summary of them to the competent authority. The authorities should periodically review all clearance practices to ensure that assumptions behind the clearance authorizations are still valid. The authorities should also promulgate regulatory guides on clearance.

The waste producer should have a systematic program for keeping records of nuclide composition and activity concentration in materials that may be cleared later on. With regard to decommissioning of large nuclear facilities, such programmes should be established already during the operational phase.

All clearance procedures should be performed within the framework of a control system, set up in accordance with recognized quality assurance requirements. In the definition of such a system, the potential amounts of materials to be cleared and the complexity of the involved practice should be taken into account.

CHAPTER 7

RECOMMENDATIONS, REGULATIONS AND GUIDELINES

7.1 International recommendations

The Safety Guide /1/ of the International Atomic Energy agency on clearance principles was discussed in chapter 2.1. In addition, the agency has issued documents /2,5,7,24/ giving practical guidance for clearance of radioactive materials from regulatory control.

In 1988, a group of experts, set up on the initiative of the Commission of European Communities, issued their recommendations /21/ on radiological protection criteria for the recycling of materials from dismantling of nuclear installations. The proposed clearance levels for the recycling of steel scrap are:

- i 1 Bq/g beta-gamma activity, including fixed surface contamination, averaged over a maximum mass of 1000 kg; besides no single item may exceed 10 Bq/g.
- ii 0.4 Bq/cm² for non-fixed beta-gamma contamination on accessible surfaces and 0.04 Bq/cm² for alpha contamination on any surface, averaged over a maximum area of 300 cm².

The clearance levels were derived by means of a generic study assuming recycling of 10,000 tonnes of steel per year having the above clearance levels for various radionuclides. The study indicated that the maximum dose to any individual would be less than 0.01 mSv/a and that the collective dose commitment would be about 1 manSv. Thus the clearance levels recommended by the CEC expert group seem to be consistent with the radiation protection criteria recommended by the IAEA (see chapter 2.1).

In 1990, the expert group was reconvened to revise its recommendations in the light of the latest analyses on radiological consequences of clearance. The group has prepared a draft report /3/ that includes proposed new clearance levels for recycling of steel. Determination of clearance levels for recycling of copper and aluminium is underway.

7.2 Nordic countries

In the Nordic countries, the principles followed in the clearance of radioactive wastes from hospitals and research institutes are extensively based on the recommendations /42/ issued by the Nordic radiation protection authorities in 1986. There are no corresponding recommendations for nuclear wastes, but in Finland and Sweden, national regulations for clearance of nuclear wastes exist.

7.2.1 Denmark

In Denmark, clearance criteria have been formulated only for non-nuclear radioactive wastes (wastes from the use of radioisotopes). The criteria included in the governmental order no. 485/1985 /41/ are given below.

Solid waste, containing less than 100 Bq/g, is classified as non-radioactive and can be disposed of without any further ado.

Sealed sources, containing less than 100 kBq can be disposed of by incineration.

Packaged solid waste can be disposed of by incineration, if

- i the surface dose rate in any point does not exceed 5 μ Sv/h, and
 - ii the activity of the contents of each package does not exceed:
 - 5 MBq for nuclides in radiotoxicity 2 group,
 - 50 MBq for nuclides in radiotoxicity 3 group and
 - 500 MBq for nuclides in radiotoxicity 4 group
- (The radiotoxicity grouping refers to that in the EEC directive 80/836 (Euratom)).

The package must be impermeable, sealable and reasonably robust. In addition, the entire route of the waste from the producer to the incineration plant must be known and everybody concerned with the transfer and disposal of the waste must be informed about it.

Liquid waste can be disposed of through the ordinary laboratory sewage system in the following amounts:

- 5 MBq/month for radiotoxicity-group 2 nuclides,
- 50 MBq/month for radiotoxicity-group 3 nuclides and
- 500 MBq/month for radiotoxicity-group 4 nuclides

The activity concentration must not exceed 100 kBq/l.

No clearance criteria have been formulated for gaseous waste or for waste containing radionuclides of the radiotoxicity-group 1.

7.2.2 Finland

Waste from the use of radioisotopes

The Finnish Centre for Radiation and Nuclear Safety has issued a guide on clearance of radioactive waste from the use of radioisotopes /43/. According to this guide, solid waste may be disposed of by the public waste treatment systems, if

- i the total disposed activity from one laboratory is below 25 ALI_{min} per month and the activity disposed of annually is below 100 GBq, and
- ii no single package contains more than 2.5 ALI_{min} of radioactive material and the surface dose rate on individual waste packages does not exceed 5 μ Sv/h.

There is also a limit of 100 kBq for disposing of sealed sources containing mainly short-lived beta-gamma emitters in a dumping site.

Nuclear waste

According to the Finnish Nuclear Energy Decree, nuclear waste can be transferred to another holder having no authorization pursuant to nuclear energy legislation (i.e. the waste is cleared from regulatory control) provided that

- i the average activity concentration in the waste is less than 10 Bq/g, and the total activity in the waste received by a transferee in one year is less than 1 GBq and alpha activity less than 10 MBq, and
- ii due to the cleared waste, the estimated annual effective dose to any individual is less than 10 μ Sv and the total radiation exposure is ALARA.

Within the constraints given above, the Finnish Centre of Radiation and Nuclear Safety can make decisions on the conditional clearance of nuclear waste. The clearance levels are then determined on the basis of case-by-case analysis.

There is also an option for unconditional clearance of waste. In a guide /44/ issued by the Finnish Centre for Radiation and Nuclear Safety, the following activity constraints are given for such clearance (excluding natural radionuclides):

- i The total activity concentration, averaged over a maximum amount of 1000 kg of waste, shall not exceed 1 Bq/g of beta-gamma activity or 0.1 Bq/g of alpha activity. In addition, no single item or waste package weighing less than 100 kg may contain more than 100 kBq beta-gamma activity or 10 kBq alpha activity.
- ii The total surface contamination of non-fixed radioactive substances, averaged over a maximum area 0,1 m² for accessible surfaces, shall not exceed 0.4 Bq/cm² of beta-gamma activity or 0.04 Bq/cm² of alpha activity.

Unconditional clearance is not applicable to waste that may easily cause radiation exposure (e.g. highly volatile or flammable materials).

7.2.3 Norway

In Norway, decisions on clearance of nuclear wastes will be made by the Norwegian Radiation Protection Authority (NRPA) on a case-to-case basis. When applicable, the decisions will be based on the criteria given in the Nordic recommendations /42/.

Clearance of radioactive wastes from hospitals and research institutes is regulated by the document "Rules for the Use and Handling of Open Radioactive Sources" /45/ issued by the NRPA. In these rules, the radionuclides are divided in four radiotoxicity groups based on a modified version of the toxicity classification given by the IAEA (Technical Report Series No 15). The maximum amount of activity release from one laboratory to the sewage or public waste treatment systems are:

- 0.4 MBq/month for toxicity group 1 (0.04 for Ra-226,228),
- 4 MBq/month for toxicity group 2 (0.4 for Sr-90),
- 40 MBq/month for toxicity group 3 and
- 400 MBq/month for toxicity group 4.

Maximum releases per day shall be less than one tenth of the monthly release constraints given above.

Permission for disposal of sealed radioactive sources is given on a case-to-case basis by the NRPA, based on the Nordic recommendations /42/.

7.2.4 Sweden

Waste from the use of radioisotopes

In 1983, the Swedish Radiation Protection Institute issued a regulation /46/ concerning waste from the use of radioisotopes outside the nuclear fuel cycle. According to this regulation, a maximum amount of 10 ALI_{min} per month can be disposed of through the ordinary laboratory sewage system. The maximum amount released at each occasion is limited to 1 ALI_{min}, or 100 MBq at the most.

Urine and faeces from patients treated or examined by means of radionuclides are allowed to be disposed of through the ordinary sewage system. This amount of activity does not have to be included in the total activity mentioned above. Solutions from liquid scintillators need not to be treated as radioactive waste if:

- i the solution contains no alpha emitting nuclides and
- ii the activity is less than 10 kBq/liter, or if the solution only contains H-3 or C-14 with an activity concentration below 100 kBq/liter.

For solid waste, the total amount of activity disposed of at an ordinary dumpsite is limited to 10 ALL_{min} per month and laboratory. No single waste package is allowed to contain more than 1 ALL_{min} and the surface dose rate on individual waste packages must not exceed 5 $\mu\text{Gy/h}$. Dumping of sealed sources with a content of radionuclides exceeding 50 kBq is not allowed.

Nuclear waste

The Swedish Radiation Protection Institute has issued two regulations on clearance of materials from nuclear facilities. In one of these regulations /47/, the following activity constraints are given for unconditional use of materials removed from the controlled areas of nuclear facilities:

- i 4 Bq/cm² for beta-gamma surface contamination and 0.4 Bq/cm² for alpha surface contamination, averaged over an area of 100 cm².
- ii 0.1 Bq/g for activity concentration, excluding typical concentrations of naturally occurring radionuclides.

In the same regulation, the following activity limits are given for disposal of cleared waste in an on-site or a municipal landfill:

- i 5 Bq/g for beta-gamma concentration and 0.5 Bq/g for alpha concentration, excluding typical concentrations of naturally occurring radionuclides.
- ii 1 GBq/a for total activity cleared from all nuclear facilities in the region.

The other regulation /48/ deals with the clearance of used oil from nuclear power plants. After the oil has been treated to remove water and particulates, it can be disposed of in a large incenerator or in an oil destruction plant provided that it contains less than 5 kBq/liter beta-gamma activity and 50 Bq/liter alpha activity.

It should be noted that the Swedish unconditional clearance level for activity concentration is one tenth of that included in the Finnish regulations, but for surface contamination the Swedish unconditional clearance levels are tenfold in comparison with the Finnish ones.

7.3 Other countries

In Canada, the Atomic Energy Control Board has drafted a regulatory position on clearance of waste to unlicensed facilities. It includes an individual dose constraint of 0.05 mSv/a for case-by-case decisions on such clearances.

In France radioactive materials are currently disposed of via public systems after a case-by-case approval, given by the Central Department for Protection against Ionizing Radiation. Studies on general principles of clearances and suggestions for activity constraints for materials to be cleared have been performed.

In **Germany**, the Commission on Radiation Protection has established a recommendation on recycling and reuse of metal scrap from NPPs. It lays down as a first priority to reuse or recycle material within the nuclear industry. If this is not feasible, unconditional clearance with an activity concentration constraint of 0.1 Bq/g or general melting for materials with activity concentration below 1 Bq/g is allowed. In addition, the surface contamination must not exceed 0.05 Bq/cm² for alpha emitters, 0.5 Bq/cm² for beta-gamma emitters and 5 Bq/cm² for weak beta emitters.

In **Japan**, the Radiation Council has adopted an individual dose constraint of 10 $\mu\text{Sv/a}$ for shallow land burial of solid radioactive waste and suggested a similar approach in establishing the criteria for unrestricted reuse or recycle of radioactive wastes. The Science and Technology Agency is developing regulatory criteria for the recycling of materials from dismantling of nuclear facilities. The Agency has requested the Japan Atomic Energy Research Institute to do research on the subject.

In the **United Kingdom**, the Radioactive Substances Act provides for introduction of clearance orders for practices with negligible radiological hazard. The clearance orders frequently include limitations and conditions concerning the method of disposal. In general, clearance is considered against criterion that doses from waste disposal should be not more than a few microsievert per year.

In the **United States**, the Nuclear Regulatory Commission issued in 1990 a policy statement on radioactive material "below regulatory concern", i.e. clearance from regulatory control. The basic principles in that proposal were similar to those of the IAEA Safety Guide No 89 /1/, but the dose constraints are ten times higher. The policy statement received much criticism and was withdrawn later. A revised rule is under consideration, until then it is possible to make case-by-case clearances of nuclear wastes on the basis of procedures described in NRC's 10 CFR Part 20.3

CHAPTER 8 CONCLUDING REMARKS

Use of radioactive materials is generally subject to the system of radiation protection, including notification, registration and licensing of sources of radiation. Some sources of ionizing radiation, however, present such a low risk that it would be a waste of resources to exercise control by regulatory processes. It means removal of regulatory controls from materials that originate from the nuclear fuel cycle or from other regulated facilities and may contain low levels of radioactivity.

Cleared materials can be treated or disposed of in the same way as non-radioactive wastes, i.e. by landfill, incineration, recycling or reuse. In many cases, e.g. due to transboundary movements of metal scrap, it is not possible to be sure of the fate of the material once it is cleared. So there are strong reasons to define unconditional clearance levels that are internationally consistent. However, if the fate of cleared material can be predetermined and constrained, national authorities may issue conditional clearances on the basis of case-by-case evaluations.

In the Nordic countries, there is a joint recommendation for the clearance of wastes from the use of radioisotopes, on which the national regulations are extensively based. In Finland and Sweden, regulations for the clearance of nuclear wastes exist. There is a discrepancy in the clearance levels included in these two regulations. Due to the trade of e.g. scrap between the countries, harmonization of the the Finnish and Swedish regulations would be desirable.

Presently, the calculations of radiological consequences of clearance levels are generally based on conservative assumptions, leading to unnecessary stringent clearance levels. The dose levels accepted to arise from clearance of nuclear wastes are small as compared to doses caused by naturally occurring substances. It is now important to collect experience and to verify the actual concentrations of radionuclides that originate from the cleared wastes. Thereby the uncertainties involved with the current radiological consequence calculations can be reduced and higher clearance levels might be accepted.

While the amounts of radioactive materials to be cleared in the Nordic countries are small at present, they will strongly increase once the nuclear power plants are to be decommissioned. There is a considerable economic incentive to clear these materials and, in addition, recycling of materials would save raw materials and energy. Though the principles for clearance have been established by international organizations some years ago, there has been little progress in the implementation of clearance policies worldwide, mainly due to socio-psychological factors.

All clearances of radioactive materials should be subject to strict regulatory control that includes review of clearance applications and other documents, inspections of monitoring practices, promulgation of regulatory guides and quality assurance. Awareness of the due regulatory control is likely to improve public acceptance of clearance.

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APPENDIX 1

MODELS AND PARAMETERS FOR RADIOLOGICAL ASSESSMENTS OF VARIOUS CLEARANCE OPTIONS

This Appendix describes a generic assessment methodology for the radiological impact from disposal, recycling or reuse of cleared materials. Such assessments are needed when applications for conditional clearance of materials under regulatory control are made or reviewed. The methodology is based on simple models and tabulated parameter values so that the results can be obtained without sophisticated computer analyses. The accuracy of the applied models, excluding the uncertainties due to parameter values, is expected to be better than about 30 %. The given ranges of typical parameter values are large in many cases. To narrow these ranges, more specific data about the cleared waste and the clearance option would be needed. In the absence of such data, the parameter values should be selected from the "conservative" end of the range.

The models and parameter values have mainly been picked up from a number of relevant reports (see the literature list). The authors of the Appendix are: Vesa Suolanen, Technical Research Centre of Finland (drafting), Mark Elert, Kemakta (review) and Esko Ruokola, Finnish Centre for Radiation and Nuclear Safety (project leader, editing).

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1. LANDFILL DISPOSAL

1.1 Operator scenario

The landfill operator is exposed to external radiation from the buried cleared waste. This radiation may be attenuated by the cover soil and the machine used for the compaction of waste. The operator may also be exposed via inhalation of dust that arise in particular during compaction of waste.

External and inhalation dose rates to the landfill operator:

$$\dot{H}_{\text{EXT},i} = t \cdot c_{w,i} \cdot W \cdot E_{\gamma,i} \cdot S_i \cdot DF_{\text{EXT},i} \quad (1)$$

$$\dot{H}_{\text{INH},i} = \zeta \cdot t \cdot c_d \cdot c_{w,i} \cdot W \cdot F_{d/s} \cdot DF_{\text{INH},i} \quad (2)$$

where

$\dot{H}_{\text{EXT},i}$	is the annual external dose from radionuclide i, $[\text{Sv}\cdot\text{a}^{-1}]$
$\dot{H}_{\text{INH},i}$	is the annually committed inhalation dose from nuclide i, $[\text{Sv}\cdot\text{a}^{-1}]$
ζ	is the breathing rate of the site operator, $[\text{m}^3\cdot\text{h}^{-1}]$
t	is the annual duration of exposure for the site operator, $[\text{h}\cdot\text{a}^{-1}]$
c_d	is the concentration of respirable dust in air, $[\text{g}\cdot\text{m}^{-3}]$
$c_{w,i}$	is the initial concentration of radionuclide i in cleared waste, $[\text{Bq}\cdot\text{g}^{-1}]$
$F_{d/s}$	is the ratio between the activity concentration of nuclide i in dust arising from cleared waste and that in solid cleared waste
S_i	is the shielding factor for radionuclide i due to cover soil and compaction machine
W	is the dilution factor (fraction of cleared waste of the total waste amount handled by the site operator)
$E_{\gamma,i}$	is the average gamma energy per disintegration for nuclide i [MeV]
$DF_{\text{EXT},i}$	is the external dose conversion factor for activity concentration in ground, $[\text{Sv}\cdot\text{h}^{-1}\cdot\text{Bq}^{-1}\cdot\text{g}\cdot\text{MeV}^{-1}]$
$DF_{\text{INH},i}$	is the committed effective dose from inhalation of one Bq of radionuclide i, $[\text{Sv}\cdot\text{Bq}^{-1}]$

If cleared wastes are not buried in a landfill gradually but a large amount at a time, a convenient way to calculate the doses is to set $W = 1$ and determine t as the time needed for the burial operations until the wastes are covered with soil.

1.2 Fire scenario

At landfills, combustible wastes that are not yet covered with soil may catch fire. The resulting release of smoke and gases may lead to exposure of people living or working nearby. Activity concentration in air and deposition on ground at a particular distance downwind are obtained as follows:

$$c_{a,i} = c_{w,i} \cdot M_f \cdot f_i \cdot \chi \quad (3)$$

$$c_{g,i} = c_{a,i} \cdot v_{g,i} \quad , \quad (4)$$

where

$c_{a,i}$	is the time-integrated concentration of radionuclide i in air, [Bq·h·m ⁻³]
$c_{g,i}$	is the concentration of radionuclide i deposited on ground, [Bq·m ⁻²]
$c_{w,i}$	is the initial concentration of radionuclide i in cleared waste, [Bq·kg ⁻¹]
M_f	is the mass of cleared waste involved in fire, [kg]
f_i	is the fractional release of radionuclide i to atmosphere following a fire
χ	is the dispersion factor at a given distance, [Bq·h·m ⁻³ /Bq]
$v_{g,i}$	is the deposition velocity of radionuclide i, [m·h ⁻¹].

Doses from inhalation of airborne material (in plume and resuspended from ground), from external radiation (in plume and deposited on ground) and from ingestion of contaminated foodstuffs:

$$H_{INH,i} = \zeta \cdot DF_{INH,i} \cdot (c_{a,i} + c_{g,i} \cdot \frac{\xi}{\lambda_i + \lambda_r}) \quad (5)$$

$$H_{EXT,i} = c_{a,i} \cdot DF_{EXT,2} \cdot E_{r,i} + c_{g,i} \cdot DF_{EXT,3} \cdot E_{r,i} \cdot \frac{T}{1 + \lambda_i \cdot T} \quad (6)$$

$$H_{ING,ij} = c_{g,i} \cdot F_{i,j} \cdot Q_j \cdot A_j \cdot DF_{ING,i} \quad , \quad (7)$$

where

$H_{INH,i}$	is the committed inhalation dose from of radionuclide i during and after (resuspended particles) the fire, [Sv]
$H_{EXT,i}$	is the external dose in one year from radionuclide i in plume and deposited on ground, [Sv]
$H_{ING,ij}$	is the committed dose from intake of radionuclide i by ingestion of foodstuff j, [Sv]
ζ	is the breathing rate of an individual, [m ³ ·h ⁻¹]
ξ	is an empirical parameter in the resuspension factor, [m ⁻¹] (i.e. [Bq·m ⁻³ _{air}]/[Bq·m ⁻² _{ground}])
λ_i	is the radioactive decay constant for radionuclide i, [h ⁻¹]

λ_r	is a short-term resuspension constant, [h^{-1}]
T	is the time spent by an individual at the contaminated area, [$\text{h} \cdot \text{a}^{-1}$]
F_{ij}	is the time-integrated concentration of radionuclide i in foodstuff j per unit deposition on the ground, [$\text{Bq} \cdot \text{a} \cdot \text{kg}^{-1} / [\text{Bq} \cdot \text{m}^{-2}]$]
Q_j	is the annual intake rate of foodstuff j, [$\text{kg} \cdot \text{a}^{-1}$]
A_j	is the fraction of contaminated foodstuff j of its total intake
$DF_{\text{EXT},2}$	is the external dose conversion factor for air concentration, [$\text{Sv} \cdot \text{h}^{-1} \cdot \text{Bq}^{-1} \cdot \text{m}^3 \cdot \text{MeV}^{-1}$]
$DF_{\text{EXT},3}$	is the external dose conversion factor for surface deposit, [$\text{Sv} \cdot \text{h}^{-1} \cdot \text{Bq}^{-1} \cdot \text{m}^2 \cdot \text{MeV}^{-1}$]

For long-lived nuclides, the duration of exposure from the substances deposited on ground is longer than one year. Equations (5), (6), and (7) describe the committed dose only for the first year after the fire, when it is normally highest.

1.3 Groundwater migration scenario

After the failure of the containment of the landfill site, the radioactive substances in cleared waste are released in groundwater and may be migrated into a nearby well or water body, causing exposure of people living nearby via contaminated drinking water or foodstuffs.

Initial steady-state activity concentrations of leach water, the leaching times and the concentrations of radionuclides in groundwater can be estimated as follows:

$$c_{LO,i} = f_m \cdot f_L \cdot \frac{c_{w,i}}{K_d} \quad (8)$$

$$\tau_i = \frac{c_{w,i} \cdot M}{c_{LO,i} \cdot I} \quad (9)$$

$$c_{L,i}(t) = c_{LO,i} \cdot F(t) \cdot \text{EXP}(-\lambda_r t) \quad (10)$$

$$F(t) = \frac{1}{2} \cdot \left\{ \text{ERFC} \left(\frac{x - ut/R}{2 \cdot \sqrt{D/R} \cdot t} \right) + \text{EXP}(u \cdot x/D) \cdot \text{ERFC} \left(\frac{x + ut/R}{2 \cdot \sqrt{D/R} \cdot t} \right) \right\} \quad (11)$$

where

$c_{LO,i}$	is the initial concentration of nuclide i in leach water, [$\text{Bq} \cdot \text{m}^{-3}$]
$c_{L,i}(t)$	is the concentration of radionuclide i in groundwater, [$\text{Bq} \cdot \text{m}^{-3}$]
$c_{w,i}$	is the concentration of radionuclide i in cleared waste at the time when the containment of the landfill fails, [$\text{Bq} \cdot \text{kg}^{-1}$]
M	is the total amount of disposed cleared waste, [kg]

τ_i	is the release time of radionuclide i from cleared waste, [a]
f_m	is the mixing ratio of cleared waste
f_L	is the leachability factor
I	is the amount of water infiltrated through the area where cleared waste is buried, [$m^3 \cdot a^{-1}$]
$K_{d,i}$	is the distribution coefficient of radionuclide i between solid and liquid phase, [$m^3 \cdot kg^{-1}$]
t	is the time elapsed from the start of release, [a]
x	is the distance to the discharge point (e.g. well or water body), [m]
u	is the groundwater velocity, [$m \cdot a^{-1}$]
D	is the dispersivity, [$m^2 \cdot a^{-1}$]
R	is the retardation factor

To find the maximum concentration, the parameter t must be varied. If the maximum is not reached until the depletion of the releases from the landfill (time τ_i), the concentrations beyond that time are obtained by replacing F(t) in equation (10) with F(t) - F(t - τ_i).

Dose from ingestion of contaminated well water:

$$\dot{H}_{ING,i} = f_w \cdot c_{L,i}(t) \cdot Q_w \cdot DF_{ING,i} , \quad (12)$$

where

$\dot{H}_{ING,i}$	is the annually committed ingestion dose from nuclide i, [$Sv \cdot a^{-1}$]
f_w	is dilution factor (flow of contaminated water into the well/total flow of water into the well)
Q_w	is the ingestion rate of water from the well, [$m^3 \cdot a^{-1}$].

Ingestion doses assuming that all contaminated water enters a river, lake or sea:

$$\dot{H}_{ING,i} = \frac{I \cdot c_{L,i}(t)}{\Phi} \cdot [Q_w + B_{f,i} \cdot Q_f + \sum_j L_j \cdot P_{i,j} \cdot Q_j] \cdot DF_{ING,i} , \quad (13)$$

where

$\dot{H}_{ING,i}$	is the annually committed ingestion dose from nuclide i, [$Sv \cdot a^{-1}$]
Φ	is the flow rate of a river or the local flow rate to a lake or sea, [$m^3 \cdot a^{-1}$]
Q_w	is the ingestion rate of water (sea: $Q_w = 0$), [$m^3 \cdot a^{-1}$]
$B_{f,i}$	is the transfer factor of fish for radionuclide i, [$Bq \cdot kg^{-1}$]/[$Bq \cdot m^{-3}$]
Q_f	is the ingestion rate of fish, [$kg \cdot a^{-1}$]
L_j	is the daily consumption of water by animal j (sea: $L_j = 0$), [$m^3 \cdot d^{-1}$]
$P_{i,j}$	is the transfer factor of animal foodstuff j for radionuclide i, [$d \cdot kg^{-1}$]
Q_j	is the ingestion rate of foodstuff j, [$kg \cdot a^{-1}$].

1.4 Residential and intrusion scenarios

After the institutional control period, a house or road may be constructed at the landfill. The residents may then be exposed to external radiation and via consumption of vegetables from their garden. During the construction, workers receive external and inhalation dose.

Dose from external radiation and from ingestion of vegetables:

$$\dot{H}_{\text{EXT},i} = t_2 \cdot c_{w,i} \cdot W \cdot S_i \cdot DF_{\text{EXT},i} \cdot E_{\gamma,i} \cdot f_{r,i} \cdot \text{EXP}(-\lambda_i t_1) \quad (14)$$

$$\dot{H}_{\text{ING},i} = Q_v \cdot B_{v,i} \cdot c_{w,i} \cdot W \cdot DF_{\text{ING},i} \cdot f_{r,i} \cdot \text{EXP}(-\lambda_i t_1) \quad (15)$$

where

$\dot{H}_{\text{EXT},i}$	is the annual external dose from radionuclide i, [Sv·a ⁻¹]
$\dot{H}_{\text{ING},i}$	is the annually committed ingestion dose from nuclide i, [Sv·a ⁻¹]
t_1	is the time elapsed from the burial of wastes, [a]
t_2	is the residence time at the contaminated area in a year, [h·a ⁻¹]
$c_{w,i}$	is the initial concentration of nuclide i in cleared waste, [Bq·g ⁻¹]
S_i	is the total shielding factor for nuclide i due to soil cover and buildings
W	is dilution factor (amount of cleared waste/total waste amount)
λ_i	is the radioactive decay constant of radionuclide i, [a ⁻¹]
$f_{r,i}$	is the removal factor for nuclide i by leaching, [a ⁻¹]
Q_v	is the annual ingestion rate of vegetables, [g·a ⁻¹]
$B_{v,i}$	is the soil to vegetable transfer factor for radionuclide i, [Bq·kg ⁻¹ (wet plant)] / [Bq·kg ⁻¹ (dry soil)].

Inhalation and external dose from house or road construction:

$$H_{\text{INH},i} = t_3 \cdot \zeta \cdot c_d \cdot c_{w,i} \cdot W \cdot DF_{\text{INH},i} \cdot f_{r,i} \cdot \text{EXP}(-\lambda_i t_1) \quad (16)$$

$$H_{\text{EXT},i} = t_3 \cdot S_i \cdot c_{w,i} \cdot W \cdot DF_{\text{EXT},i} \cdot E_{\gamma,i} \cdot f_{r,i} \cdot \text{EXP}(-\lambda_i t_1) \quad (17)$$

where

$H_{\text{INH},i}$	is the committed inhalation dose from nuclide i during excavation, [Sv]
$H_{\text{EXT},i}$	is the external dose from nuclide i during excavation, [Sv]
t_1	is the time elapsed from the release of the material for disposal, [h]
t_3	is the duration of excavation, [h].
c_d	is the concentration of respirable dust in air, [g·m ⁻³]
ζ	is the breathing rate of the worker, [m ³ ·h ⁻¹]

2. INCINERATION

2.1 Operator scenario

The operator of the incineration plant may be exposed to external radiation and to inhalation dose from airborne dust. Most of the exposures are probably received in the ash removal area. They can be estimated as follows:

$$\dot{H}_{\text{INH},i} = m \cdot V \cdot t \cdot \zeta \cdot c_d \cdot c_{w,i} \cdot (1 - f_i) \cdot DF_{\text{INH},i} \quad (18)$$

$$\dot{H}_{\text{EXT},i} = t \cdot S_i \cdot V \cdot m \cdot c_{w,i} \cdot (1 - f_i) \cdot DF_{\text{EXT},4} \cdot E_{\gamma,i} \quad (19)$$

where

$\dot{H}_{\text{INH},i}$	is the annually committed inhalation dose from nuclide i, [Sv·a ⁻¹]
$\dot{H}_{\text{EXT},i}$	is the annual external dose from radionuclide i, [Sv·a ⁻¹]
$c_{w,i}$	is the initial concentration of nuclide i in cleared waste, [Bq·g ⁻¹]
c_d	is the concentration of respirable dust in air, [g·m ⁻³]
ζ	is the breathing rate of the worker, [m ³ ·h ⁻¹]
m	is the mass reduction factor (waste/ash mass ratio)
V	is dilution factor (cleared waste/total incinerated waste volume ratio)
t	is the duration of exposure for the operator, [h·a ⁻¹]
f_i	is the fraction of radionuclide i released through stack
S_i	is the shielding factor for nuclide i due to e.g. walls
$E_{\gamma,i}$	is the average gamma energy per disintegration for nuclide i [MeV]
$DF_{\text{INH},i}$	is the committed effective dose from inhalation of one Bq of radionuclide i, [Sv·Bq ⁻¹]
$DF_{\text{EXT},4}$	is the conversion factor for external dose from the ash storage, [Sv·h ⁻¹ ·Bq ⁻¹ ·g·MeV ⁻¹].

Equations (18) and (19) can also be used to consider exposure to the raw waste by setting $m = 1$ and $f_i = 0$.

2.2 Atmospheric emission scenario

The atmospheric gas and dust emissions of an incineration plant may expose the nearby people via inhalation of airborne substances, external radiation from ground contamination and via ingestion of contaminated foodstuffs. The maximum concentrations at the end of the operational life of the incinerator:

$$c_{a,i} = \chi \cdot f_i \cdot R_w \cdot V \cdot c_{w,i} \quad (20)$$

$$c_{g,i} = v_{g,i} \cdot c_{a,i} \cdot \left(\frac{1 - \text{EXP}(-(\lambda_i + \lambda_g) \cdot \Omega)}{\lambda_i + \lambda_g} \right) \quad (21)$$

$$c_{i,j} = v_{g,i} \cdot c_{a,i} \cdot F_{i,j} , \quad (22)$$

where

$c_{a,i}$	is the average air concentration of nuclide i at ground level, [Bq·m ⁻³]
$c_{g,i}$	is the concentration of nuclide i deposited on ground, [Bq·m ⁻²]
$c_{i,j}$	is the concentration of radionuclide i in foodstuff j, [Bq·kg ⁻¹]
χ	is the dispersion factor at the distance of interest, [h·m ⁻³]
R_w	is the average incineration rate of waste, [kg·h ⁻¹]
$v_{g,i}$	is the deposition velocity of radionuclide i, [m·a ⁻¹]
Ω	is the operational lifetime of the incinerator, [a]
λ_g	is the constant for removal of nuclides from ground surface, [a ⁻¹]
$F_{i,j}$	is the time-integrated concentration of radionuclide i in foodstuff j per unit deposition on the ground, [Bq·a·kg ⁻¹]/[Bq·m ²].

Doses from inhalation of airborne material (in plume and resuspended from ground), external radiation (from plume and ground contamination) and from ingestion of contaminated foodstuffs:

$$\dot{H}_{INH,i} = c_{a,i} \cdot \left(1 + v_{g,i} \cdot \frac{\xi}{\lambda_i + \lambda_r} \right) \cdot \zeta \cdot t \cdot DF_{INH,i} \quad (23)$$

$$\dot{H}_{EXT,i} = (c_{a,i} \cdot DF_{EXT,2} + c_{g,i} \cdot DF_{EXT,3}) \cdot E_{\gamma,i} \cdot t \quad (24)$$

$$\dot{H}_{ING,i,j} = c_{i,j} \cdot Q_j \cdot A_j \cdot DF_{ING,i} , \quad (25)$$

where

$\dot{H}_{\text{INH},i}$	is the annually committed inhalation dose from nuclide i, [Sv·a ⁻¹]
$\dot{H}_{\text{EXT},i}$	is the annual external dose from radionuclide i, [Sv·a ⁻¹]
$\dot{H}_{\text{ING},ij}$	is the annually committed dose from nuclide i by ingestion of foodstuff j, [Sv·a ⁻¹]
ξ	is an empirical parameter in the resuspension factor, [m^{-1}] ([Bq m ⁻³ air]/[Bq m ⁻² ground])
λ_r	is a short term resuspension time constant, [a ⁻¹]
t	is the residence time in the vicinity of the incinerator, [h·a ⁻¹]
Q_j	is the annual consumption of foodstuff j, [kg·a ⁻¹]
A_j	is the fraction of intake of foodstuff j that is contaminated by the released radioactive substances.

2.3 Ash reuse and disposal scenarios

Incinerator ash might be used as a base material for road or building construction. Since ash is likely to be only one component of materials used, it is reasonable to assume that exposures would normally be less than for the incinerator workers. If incinerator ash is disposed of in a landfill, the exposure scenarios are the same as those for non-incinerated waste, except for the landfill fire scenario. The concentrations of radionuclides in the disposed waste, $c_{w,j}$, in formulae (1), (2), (8), (9), (14), (15), (16) and (17) are then obtained by multiplying those of raw waste by a factor $m \cdot V \cdot (1 - f_j)$.

3. RECYCLING OF METALS

3.1 Steelworks worker scenarios

During storage, pretreatment and melting of contaminated scrap or manufacturing of products, the workers may be exposed to external radiation from scrap, ingot or slag bales, metal sheets/blanks, or recovered dusts. In addition, the dust arising during these activities may cause inhalation doses to the workers.

The concentrations of radionuclides in the various constituents:

$$A_{ij} = i \cdot c_{wj} \quad (26)$$

$$A_{sj} = s \cdot c_{wj} \cdot f_{is} \quad (27)$$

$$A_{dj} = d \cdot c_{wj} \cdot f_{id} , \quad (28)$$

where

A_{ij} , A_{sj} and A_{dj} are the concentrations of nuclide j in ingot, slag and dust, $[Bq \cdot g^{-1}]$
 i , s and d are the fractions of radionuclides which end in ingot, slag and dust
 f_{is} and f_{id} are the ingot/slag and ingot/dust mass ratios
 c_{wj} is the activity concentration of radionuclide j in scrap, $[Bq \cdot g^{-1}]$.

External and inhalation dose rates to the workers:

$$\dot{H}_{EXT,m,j} = t \cdot A_{mj} \cdot DF_{EXT,S} \cdot E_{\gamma,j} \quad (29)$$

$$\dot{H}_{INH,j} = \zeta \cdot t \cdot c_d \cdot A_{dj} \cdot DF_{INH,j} \quad (30)$$

where

$\dot{H}_{EXT,m,j}$ is external dose rate from radionuclide j, $[Sv \cdot a^{-1}]$
 $\dot{H}_{INH,j}$ is annually committed inhalation dose from radionuclide j, $[Sv \cdot a^{-1}]$
 c_d is the concentration of dust, $[g \cdot m^{-3}]$
 ζ is the breathing rate, $[m^{-3} \cdot h]$
 t is the duration of exposure, $[h \cdot a^{-1}]$
 A_{mj} is activity concentration of radionuclide j in material of interest (scrap, ingot, slag or dust), $[Bq \cdot g^{-1}]$
 $DF_{EXT,S}$ is the external dose conversion factor for sources at the steelworks, $[Sv \cdot h^{-1} \cdot Bq^{-1} \cdot g \cdot MeV^{-1}]$.
 $E_{\gamma,i}$ is the average gamma energy per disintegration for nuclide i $[MeV]$
 $DF_{INH,j}$ is the committed effective dose from inhalation of one Bq of radionuclide i, $[Sv \cdot Bq^{-1}]$

3.2 Atmospheric emission scenario

If the steelworks has inefficient filtration systems, dust and fumes are spread in the environs of the plant. The exposures of people living there are obtained in analogy with the landfill fire scenario: equations (5), (6) and (7). The source term is obtained by replacing in equation (3) f_i with fraction of each radionuclide in scrap that is released with the effluents and M_f with the mass of scrap [kg] that is melt.

3.3 Consumer product scenarios

The contaminated scrap metal may end up as consumer products such as reinforcement bars, household commodities or cars. External exposure from the use of these products can be estimated as follows:

$$\dot{H}_{\text{EXT},i} = \frac{t}{1 + \lambda_i \cdot t} \cdot \text{EXP}(-\lambda_i \tau) \cdot i \cdot W_p \cdot c_{w,i} \cdot DF_{\text{EXT},s} \cdot E_{y,i} , \quad (31)$$

where

$\dot{H}_{\text{EXT},i}$	is annual external dose to individual from radionuclide i , $[\text{Sv} \cdot \text{a}^{-1}]$
t	is duration of exposure, $[\text{h} \cdot \text{a}^{-1}]$
τ	is the delay time until the use of the product, $[\text{h}]$
$c_{w,i}$	is activity concentration of nuclide i in non-diluted scrap, $[\text{Bq} \cdot \text{g}^{-1}]$
i	is fraction of the radionuclides which end up in the ingot
W_p	is the dilution factor (fraction of metal in the product that originates from the cleared scrap)
$DF_{\text{EXT},s}$	is the external dose factor for the product of interest, $[\text{Sv} \cdot \text{h}^{-1} \cdot \text{Bq}^{-1} \cdot \text{g} \cdot \text{MeV}^{-1}]$.

If the contaminated scrap metal end up as cooking ware, the annual ingestion dose resulting from corrosion or erosion of it can be estimated as follows:

$$\dot{H}_{\text{ING},i} = \rho_p \cdot A_p \cdot v_p \cdot f_f \cdot i \cdot W_p \cdot c_{w,i} \cdot \text{EXP}(-\lambda_i \tau) \cdot DF_{\text{ING},i} , \quad (32)$$

where

$\dot{H}_{\text{ING},i}$	is the annually committed ingestion dose from nuclide i , $[\text{Sv} \cdot \text{a}^{-1}]$
ρ_p	is the density of cooking ware material, $[\text{g} \cdot \text{cm}^{-3}]$
A_p	is the effective surface area (area from which the released material is transferred into food during cooking), $[\text{cm}^2]$
v_p	is the corrosion or erosion rate, $[\text{cm} \cdot \text{a}^{-1}]$
f_f	is the fraction of corrosion or erosion products which are transferred into food during cooking.

3.4 Slag disposal scenario

If slag from a steelworks is disposed of in a landfill, the exposure scenarios are the same as those in chapter 1, except for the landfill fire scenario. The concentrations of radionuclides in the disposed waste, $c_{w,j}$, in equations (1), (2), (8), (9), (14), (15), (16) and (17) should then be replaced with $A_{s,j}$ in equation (27).

3.5 Collective dose scenarios

The collective dose commitment after many recycling sequences can be estimated as follows:

$$H_{c,i} = n \cdot H_i \cdot \frac{\tau_r}{1 + \lambda_i \tau_r} \cdot \sum_{j=0} K^j \cdot \text{EXP}(-j \lambda_i \tau_r) \quad (33)$$

$$n = \frac{M_c}{m_p \cdot W_p}, \quad (34)$$

where

$H_{c,i}$	is the collective dose commitment for radionuclide i, [manSv]
H_i	is the individual annual dose for nuclide i from the use of product of interest in the first year of recycling, [$\text{Sv} \cdot \text{a}^{-1}$]
n	is number of persons exposed in each recycling sequence
τ_r	is the length of recycling sequence, i.e. life-time of the product, [a]
K	is the recycling factor (the fraction of product that is recycled)
M_c	is the total mass of the cleared scrap, [kg]
m_p	is the mass of product of interest, [kg].
j	is the recycling sequence number
W_p	is the dilution factor (fraction of metal in the product that originates from the cleared scrap)

4. REUSE OF COMPONENTS AND BUILDINGS

4.1 Contaminated component scenarios

External exposure may occur after the release of a contaminated component (e.g. a pipe, tank or pump used in a nuclear facility) for unrestricted use during repair or storage of the component. The resulting external dose can be estimated by applying a line source - cylindrical shield approximation:

$$\dot{H}_{\text{EXT},i} = \frac{t}{1 + \lambda_i \cdot t} \cdot \frac{B \cdot A_{C,i} \cdot DF_{\text{EXT},6} \cdot E_{\gamma,i} \cdot D_C}{2 \cdot a_R} \cdot F(\Theta, b) , \quad (35)$$

where

$\dot{H}_{\text{EXT},i}$	is the annual external dose from radionuclide i to the worker, [Sv·a ⁻¹]
t	is the duration of exposure, [h·a ⁻¹]
λ_i	is the decay constant of nuclide i [a ⁻¹]
B	is the dose build-up factor
$DF_{\text{EXT},6}$	is the external dose conversion factor for a cylindrical source, [Sv·h ⁻¹ ·Bq ⁻¹ m ² ·MeV ⁻¹]
$E_{\gamma,i}$	is the average gamma energy per disintegration for nuclide i, [MeV]
$A_{C,i}$	is the surface activity of the component for radionuclide i, [Bq m ⁻²]
a_R	is the effective distance between the worker and the (hypothetical) line source, [m]
D_C	is the effective diameter of the contaminated surface, [m]
$F(\Theta, b)$	is the Sievert's integral (see Fig. 2., page 32).

For small objects, a point source approximation can be applied by assuming a cylinder with small dimension in relation to the distance a_R .

During the repair or use of a contaminated component, the worker may be exposed via inhalation of respirable dust or via ingestion of contaminant in hands. These doses are obtained as follows:

$$H_{\text{INH},i} = \zeta \cdot t \cdot c_d \cdot \frac{A_{C,i}}{\rho_s} \cdot DF_{\text{INH},i} \quad (36)$$

$$H_{\text{ING},i} = q_r \cdot t \cdot f_r \cdot A_{C,i} \cdot DF_{\text{ING},i} \quad (37)$$

where

$H_{INH,i}$	is the committed inhalation dose from radionuclide i, [Sv]
$H_{ING,i}$	is the committed ingestion dose from radionuclide i, [Sv]
ζ	is the breathing rate of the worker, [$m^3 \cdot h^{-1}$]
t	is the duration of exposure, [h]
c_d	is the concentration of dust in air, [$g \cdot m^{-3}$]
$A_{C,i}$	is the activity concentration of radionuclide i on the surface, [$Bq \cdot m^{-2}$]
ρ_s	is the mass of layer to be removed from surface, [$g \cdot m^{-2}$]
q_r	is the surface contamination ingestion rate, [$m^2 \cdot h^{-1}$]
f_r	is the removable fraction of the surface contamination
$DF_{ING,i}$	is the committed effective dose from ingestion of one Bq of radionuclide i, [$Sv \cdot Bq^{-1}$]
$DF_{INH,i}$	is the committed effective dose from inhalation of one Bq of radionuclide i, [$Sv \cdot Bq^{-1}$]

Use of a cleared, surface contaminated component (e.g. tool or equipment) may cause skin dose to e.g. hands. This dose can be estimated as follows:

$$H_{SKIN,i} = A_{C,i} \cdot t \cdot S_c \cdot f_{skin} \cdot w_{skin} \cdot DF_{SKIN,i}, \quad (38)$$

where

$H_{SKIN,i}$	is the effective dose from skin exposure to radionuclide i, [Sv]
t	is the time for the component in use or the time constant for surface contamination to wear off [h]
S_c	is a shielding factor due to e.g. clothing
f_{skin}	is the exposed fraction of the total skin area
w_{skin}	is the dose weighting factor for skin
$DF_{SKIN,i}$	is the skin dose rate from a unit surface activity of radionuclide i, [$Sv \cdot h^{-1} \cdot Bq \cdot m^{-2}$]

A cleared storage tank from a nuclear facility might be used as a water reservoir in a dwelling. Assuming that the surface activity of the contaminated tank is completely leached in domestic use, the resulting ingestion dose from drinking water is given by:

$$H_{ING,i} = \frac{1}{n_f} \cdot A_{s,i} \cdot S \cdot DF_{ING,i}, \quad (39)$$

where

- $H_{ING,i}$ is the committed ingestion dose from radionuclide i, [Sv]
- $A_{s,i}$ is the surface activity concentration for radionuclide i, [$\text{Bq}\cdot\text{m}^{-2}$]
- S is the surface of the tank, [m^2]
- n_f is the number of persons in the family.

4.2 Contaminated room scenarios

Buildings of a nuclear facility might be released for unrestricted use. External exposure from contaminated room surfaces can be estimated by assuming a wall as an equivalent disc source:

$$H_{EXT,i} = DF_{EXT,\gamma} \cdot E_{\gamma,i} \cdot \frac{t}{1 + \lambda_i t} \cdot \sum_{j=1}^p A_{s,j,i} \cdot \ln\left(\frac{A_{D,j}/\pi + a_j^2}{a_j^2}\right) \quad (40)$$

where

- $H_{EXT,i}$ is annual external dose to the worker from radionuclide i, [$\text{Sv}\cdot\text{a}^{-1}$]
- $DF_{EXT,\gamma}$ is the external dose conversion factor for a disc surface, [$\text{Sv}\cdot\text{h}^{-1}\cdot\text{Bq}^{-1}\cdot\text{m}^2\cdot\text{MeV}^{-1}$]
- t is the annual exposure time, [$\text{h}\cdot\text{a}^{-1}$]
- $A_{s,j,i}$ is the surface activity concentration of plane j for nuclide i, [$\text{Bq}\cdot\text{m}^{-2}$]
- p is the number of plane sources in the room
- a_j is the distance from the receptor to the plane j, [m]
- $A_{D,j}$ is the surface area of plane (wall) j, [m^2].

Reparation of contaminated room surfaces may cause inhalation or ingestion dose to the workers; these dose can be calculated by means of equations (36) and (37). Contact with contaminated surface may cause skin dose; it can be estimated by means of equation (38).

TABLE 1

Decay characteristics of typical radionuclides in waste.

Radionuclide i	Half-life	Decay constant λ_i [a ⁻¹]	Decay mode, principal energy [MeV]	Average gamma energy per disintegration $E_{\gamma,i}$ [MeV]
H-3	12 a	0.56	(β , 0.019)	
C-14	5730 a	1.2 E-4	(β , 0.156)	
Na-22	2.6 a	0.27	(γ , 1.274)	2.19
Cl-36	3 E5 a	2.3 E-6	(β , 0.714)	
Cr-51	28 d	9.1	(γ , 0.320)	0.033
Mn-54	312 d	0.84	(γ , 0.835)	0.835
Fe-55	2.6 a	0.27	(E.C. X-rays)	
Co-58	71 d	3.6	(γ , 0.811)	0.975
Co-60	5.3 a	0.13	(γ , 1.17 and 1.33)	2.5
Ni-63	96 a	7.2 E-3	(β , 0.067)	
Zn-65	244 d	1.0	(γ , 1.116)	0.584
Sr-90/Y-90	28 a	0.025	(β , 2.270)	
Nb-94	2 E4 a	3.5 E-5	(γ , 0.871)	1.570
Ru-106/Rh-106	1 a	0.69	(γ , 0.512)	0.201
Ag-110m	250 d	1.0	(γ , 0.885)	2.74
Sb-125	2.8 a	0.25	(γ , 0.428)	0.430
I-131	8 d	31	(γ , 0.365)	0.380
Cs-134	2.1 a	0.34	(γ , 0.796)	1.55
Cs-137/Ba-137m	30 a	0.023	(γ , 0.662)	0.596
Ce-144	284 d	0.89	(γ , 0.133)	0.021
Eu-152	12.7 a	0.055	(γ , 1.408)	1.140
U-238	4.5 E9 a	1.5 E-10	(α , 4.198; γ , 0.050)	0.0014
Pu-239	2.4 E4 a	2.8 E-5	(α , 5.155)	
Am-241	430 a	1.6 E-3	(α , 5.485; γ , 0.060)	0.032

TABLE 2

Parameter values for all landfill scenarios.

Parameter	Typical value
Landfill capacity, [t/a]	
- industrial landfill	$10^2 - 10^4$
- municipal landfill	$10^3 - 10^5$
Waste/soil mass ratio	0.5 – 1
Bulk density of waste, [kg m^{-3}]	
- trash, ash etc	500 – 1000
- concrete, metal, slag	1000 – 3000
Landfill surface area/waste mass –ratio [$\text{m}^2 \cdot \text{t}^{-1}$]	0.1 – 0.5

TABLE 3

Parameter values for the landfill operator scenario.

Parameter	Typical value
Duration of exposure, t , [$\text{h} \cdot \text{a}^{-1}$]	100 – 1500 (Site specific)
Breathing rate, ζ , [$\text{m}^3 \cdot \text{h}^{-1}$]	1.25
Airborne dust concentration, c_d , [g m^{-3}]:	
- unpacked trash, ash, dry slag	10^{-3}
- concrete, metal, trash in drums	10^{-4}
Dust/solid activity concentration ratio in cleared waste, $F_{d/s}$	
- homogeneous waste	1
- surface contaminated waste, air filters etc	1 – 10
Dilution factor, W	0.001 – 1 (Site specific)
Shielding factor, S_i :	
- for energies > 0.5 MeV	see Fig. 1
- for energies > 0.1 Mev but < 0.5 MeV	0.5 · value from Fig. 1
- for energies < 0.1 MeV	0.1 · value from Fig. 1
External dose conversion factor for activity in ground, $DF_{EXT,i}$, [$\text{Sv} \cdot \text{h}^{-1} \cdot \text{Bq}^{-1} \cdot \text{g} \cdot \text{MeV}^{-1}$]	$2 \cdot 10^{-7}$
$DF_{INH,i}$:	see TABLE 18

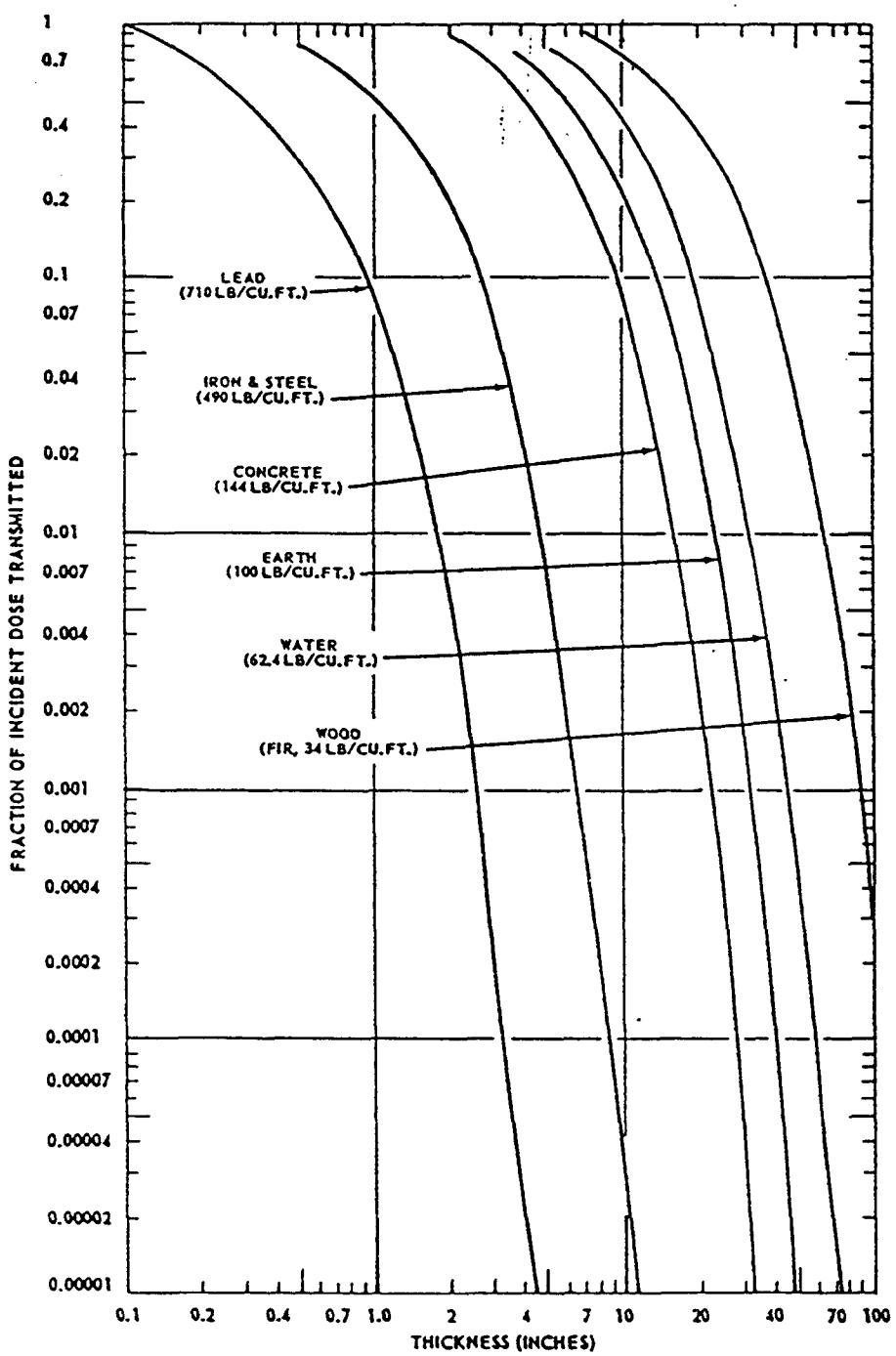


Fig. 1. Transmission of fission product gamma radiation in several shield materials.
 $(1 \text{ lb}\cdot\text{ft}^{-3} = 1.602 \cdot 10^{-2} \text{ g cm}^{-3}, 1 \text{ in.} = 2.54 \text{ cm})$

TABLE 4

Parameter values for the landfill fire scenario.

Parameter	Typical value
Mass of cleared waste involved in fire, M , [kg]	$10^3 - 10^4$ (Site specific)
Release fractions, f_i :	
H, C	0.5 – 1.0
I, Cl, Cs, Na, Ru, Ca	0.1 – 0.5
Fe, Co, Ni, Sb, Sr, Ce, Eu, Pu, Am	0.001 – 0.01
Dispersion factors in unfavourable weather conditions, χ , [$\text{Bq} \cdot \text{h} \cdot \text{m}^{-3} / \text{Bq}$]: (See also TABLE 12.)	
– source receptor distance 1 km	$10^{-8} - 10^{-7}$
– source receptor distance 5 km	$10^{-10} - 10^{-9}$
Deposition velocity, $v_{g,i}$ [$\text{m} \cdot \text{h}^{-1}$] :	
– H, C	0
– Other radionuclides	1 – 10
Breathing rate, ζ , [$\text{m}^3 \cdot \text{h}^{-1}$]	0.8
Time spent at the contaminated area [$\text{h} \cdot \text{a}^{-1}$]	100 – 3000
Parameters in the resuspension factor:	
ξ , [m^{-1}]	10^{-5}
λ_r , [a^{-1}]	3.65
External dose conversion factor for air concentration, $DF_{EXT,2}$, [$\text{Sv} \cdot \text{h}^{-1} \cdot \text{Bq}^{-1} \cdot \text{m}^3 \cdot \text{MeV}^{-1}$]	$1.5 \cdot 10^{-10}$
External dose conversion factor for surface deposit, $DF_{EXT,3}$, [$\text{Sv} \cdot \text{h}^{-1} \cdot \text{Bq}^{-1} \cdot \text{m}^2 \cdot \text{MeV}^{-1}$]	$2 \cdot 10^{-12}$
F_{ij} , Q_j , A_j :	see TABLE 17
$DF_{ING,j}$, $DF_{INH,j}$:	see TABLE 18

TABLE 5

Parameter values for groundwater migration scenario of landfill disposal.

Parameter	Typical value
Distribution coefficient, K_d , [$\text{m}^{-3} \cdot \text{kg}$]:	
H, Cl	0.001 (Assumed)
I, Zn, Sr, Ru, Sb	0.001 – 0.01
C, Cr, Mn, Fe, Co, Ni, Ag	0.01 – 0.1
Na, Cs, Zr, Ce, Eu	0.1 – 1
Pu, Am	1 – 10
Mixing ratio of cleared waste, f_m :	
– homogeneous mixing with other wastes/soil	W (in TABLE 3)
– no mixing with other waste/soil	1
Leachability factor, f_L :	
– porous, permeable waste material	1
– contaminated blocks of metal or concrete	5 – 10
– activated material	0.1 – 0.2
Infiltration trough the waste area , I, [$\text{m}^3 \cdot \text{a}^{-1}$]/[$\text{m}^2_{\text{ground}}$]:	
– penetrating cover layer	0.2
– isolating cover layer	0.05
Groundwater velocity, u, [$\text{m} \cdot \text{a}^{-1}$]:	$t \cdot p \cdot 10^8$
where: topography coefficient t:	0.01 (flat) – 0.05 (steep)
permeability coefficient p, [$\text{m} \cdot \text{s}^{-1}$]:	10^{-6} (clay) – 10^{-4} (sand/gravel)
Dispersivity, D, [$\text{m}^2 \cdot \text{a}^{-1}$]:	$0.1 \cdot u \cdot x$ (Peclet number 10)
where: groundwater velocity, u, [$\text{m} \cdot \text{a}^{-1}$]:	(see above)
distance to the discharge point, x, [m]:	100 – 1000 (Site specific)
Dilution factor for a well, f_w :	$1/x - 10/x$ (x as above)
Retardation factor, R:	
H, Cl, I	1 – 5
C, Zn, Ru, Sb,	5 – 50
Mn, Cr, Fe, Co, Ni, Sr, Ag, Eu	100 – 1000
Zr, Nb, Cs, Ce, Pu, Am	500 – 5000
Flow rate of freshwater body, Φ , [$\text{m}^3 \cdot \text{a}^{-1}$]	$10^5 - 10^9$ (Site specific)
Ingestion rate of water, Q_w , [$\text{m}^3 \cdot \text{a}^{-1}$]	0.7
Ingestion rate of fish, Q_f , [$\text{kg} \cdot \text{a}^{-1}$]	5 – 25 (Site specific)
Consumption of water by an animal, L_j , [$\text{m}^3 \cdot \text{d}^{-1}$]	0.001 – 0.05(Site specific)
B_{fj} , P_{ij} , Q_j :	see TABLE 16 and 17

TABLE 6

Parameter values for residential and intrusion scenarios of landfill disposal.

Parameter	Typical value
Annual ingestion rate of vegetables, Q_v , [$\text{kg} \cdot \text{a}^{-1}$]	10 – 20
Time elapsed from the burial of waste, t_1 , [a]	30 – 100
Residence time at the contaminated area, t_2 , [$\text{h} \cdot \text{a}^{-1}$]	
– outdoors	1000 – 2000
– indoors	3000 – 8000
Removal by leaching –factor, $f_{r,i}$,	
– no leaching occurred	1
– leaching immediately after burial of waste	$\text{EXP}(-t_2/\tau_i)$ (τ_i from eq. (9))
Duration of excavation, t_3 , [h]	20 – 200
ζ , c_d , W, S_i , $DF_{EXT,j}$:	see TABLE 3
$B_{V,j}$, Q_V :	see TABLE 16 and 17
$DF_{ING,j}$, $DF_{INH,j}$:	see TABLE 18

TABLE 7

Parameter values for incinerator operator scenario.

Parameter	Typical value
Mass reduction factor, m	2.5 – 3.5
Dilution factor, V	$10^{-4} - 10^{-1}$ (Facility specific)
Duration of exposure of worker, t , [$\text{h}\cdot\text{a}^{-1}$]	50 – 1500
Breathing rate of worker, ζ , [$\text{m}^3\cdot\text{h}^{-1}$]	1.25
Airborne dust concentration, c_d , [$\text{g}\cdot\text{m}^{-3}$]	$10^{-4} - 10^{-3}$ (Facility specific)
Fraction of radionuclides in ash, $(1 - f_i)$	
H-3, C-14	0 – 0.1
Other elements	1
Shielding factor, S_i	see TABLE 3 and Fig. 1
Conversion factor for external dose from the ash storage, $DF_{EXT,4}$, [$\text{Sv}\cdot\text{h}^{-1}\cdot\text{Bq}^{-1}\cdot\text{g}\cdot\text{MeV}^{-1}$]	$3\cdot10^{-8} - 10^{-7}$ (Facility specific)
$DF_{INH,i}$:	see TABLE 18

TABLE 8

Parameter values for the atmospheric emission scenario of incineration.

Parameter	Typical value
Dispersion factor, χ , [$\text{h} \cdot \text{m}^{-3}$], for $h_{\text{eff}}=150 \text{ m}$, distance 5 km, stability class D (neutral):	$10^{-10} - 10^{-8}$ (See TABLE 12)
Average incineration rate of waste, R_w , [$\text{kg} \cdot \text{h}^{-1}$]	100 – 10 000(Facility specific)
Operational lifetime of an incinerator, Ω , [a]	20 – 30
Dilution factor, V ,	see TABLE 7
Empirical resuspension parameter, ξ , [m^{-1}]	10^{-5}
Short term resuspension constant, λ_r , [a^{-1}]	3.65
Constant for long-term removal of radionuclides from ground surface, λ_g , [a^{-1}]	0.01 – 0.1
Fraction of radionuclides released from stack, f_i	
H, C	1
Na, Cl, Ru, Cs	0.01 – 0.1
Other nuclides	< 0.01
Deposition velocity, $v_{g,i}$, [$\text{m} \cdot \text{h}^{-1}$]	
H, C	0
Other nuclides	0.3 – 3
Breathing rate of an individual, ζ , [$\text{m}^3 \cdot \text{h}^{-1}$]	0.8
Duration of exposure, t , [$\text{h} \cdot \text{a}^{-1}$]	
– outdoors	1000 – 2000
– indoors	3000 – 8000
F_{ij} , Q_j , A_j	see TABLE 17
$DF_{INH,j}$, $DF_{ING,j}$	see TABLE 18
$DF_{EXT,2}$, $DF_{EXT,3}$	see TABLE 4

TABLE 9

Parameter values for recycling, the worker scenarios.

Parameter	Typical values	
Breathing rate, ζ , [$\text{m}^3 \cdot \text{h}^{-1}$]	1.25	
Airborne dust concentration, c_d , [$\text{g} \cdot \text{m}^{-3}$]: – in the steelworks – in the manufacturing	$5 \cdot 10^{-4} - 2 \cdot 10^{-3}$ $10^{-4} - 5 \cdot 10^{-4}$	
i, s, f :	see TABLE 14	
Mass ratio between ingot and slag, $f_{i/s}$, – arch furnace – induction furnace	5 – 10 10 – 50	
Mass ratio between ingot and dust, $f_{i/d}$, – arch furnace – induction furnace	50 – 100 100 – 500	
$E_{\gamma,i}$:	see TABLE 1	
$DF_{INH,i}$:	see TABLE 18	
Scenario / source material	Exposure time t [h] per tonne of cleared material	External dose conversion factor, $DF_{EXT,S}$, [$\text{Sv} \cdot \text{a}^{-1} \cdot \text{Bq}^{-1} \cdot \text{g} \cdot \text{MeV}^{-1}$]
Storage of scrap	0.01 – 0.1	$10^{-9} - 3 \cdot 10^{-9}$
Pretreatment of scrap	0.01 – 0.2	$2 \cdot 10^{-9} - 5 \cdot 10^{-9}$
Handling of ingot	0.01 – 0.3	$3 \cdot 10^{-9} - 10^{-8}$
Handling of slag	0.01 – 0.1	$10^{-9} - 2 \cdot 10^{-9}$
Manufacturing of metal blanks or coils	0.01 – 0.5	$3 \cdot 10^{-9} - 10^{-8}$
Recycling of dust	0.001 – 0.01	$5 \cdot 10^{-8} - 10^{-7}$

TABLE 10

Parameter values for recycling, consumer and collective dose scenarios.

Parameter	Typical value	
Dilution factor, W_p	$10^{-2} - 10^{-1}$	(Facility specific)
Delay time between release of scrap and use of the product, τ , [a]	0.5 – 1	
Product made of cleared scrap:	Exposure time t [h] in one year:	External dose conversion factor, $DF_{EXT,g}$, [Sv·h ⁻¹ ·Bq ⁻¹ ·g·MeV ⁻¹]:
Reinforcement bars	3000 – 5000	$3 \cdot 10^{-9} - 7 \cdot 10^{-9}$
Furniture, household appliances, thermal radiators	2000 – 5000	$2 \cdot 10^{-9} - 5 \cdot 10^{-9}$
Cars	2000 – 3000	$10^{-8} - 2 \cdot 10^{-8}$
Density of cooking ware material, ρ , [g·cm ⁻³]	7 – 8	
Effective surface area of cooking ware, A_p , [cm ²]	200 – 500	
Corrosion/erosion rate for stainless steel pans, v_p , [cm·a ⁻¹]	10^{-2}	
Fraction of corrosion/erosion products transferred into food during cooking, f_t	$10^{-2} - 10^{-1}$	
Length of the recycling sequence, τ_r , [a]	10 – 20	
Recycling factor, K	0.2 – 0.5	

TABLE 11

Parameter values for reuse and repair of contaminated components or rooms

Parameter	Typical value
Breathing rate of the worker, ζ , [$\text{m}^3 \cdot \text{h}^{-1}$]	1.25
Dust concentration during repair, c_d , [$\text{g} \cdot \text{m}^{-3}$]	$10^{-3} - 5 \cdot 10^{-3}$
Surface contamination ingestion rate, q_r , [$\text{m}^2 \cdot \text{h}^{-1}$]	10^{-4}
Removable fraction of surface contamination, f_r	0.1
Duration of exposure, t , [$\text{h} \cdot \text{a}^{-1}$]	
- working close to a contaminated component	100 – 1000
- working in a contaminated room	1000 – 2000
- repair of a contaminated component	5 – 20
- repair of a contaminated surface	5 – 50
- skin contact to a contaminated component	10 – 30
- body contact to a contaminated surface	100 – 300
Mass of surface layer to be removed, ρ_s , [$\text{g} \cdot \text{m}^{-2}$]	50 – 500
Skin dose shielding factor due to clothing, S_c	
- strong beta emitters	0.2 – 0.5
- weak beta emitters	0
Exposed fraction of skin area, f_{skin}	
- hands	0.02
- sitting on contaminated surface	0.1
Dose weighting factor for skin, w_{skin}	0.01
Conversion factor for external dose from a line source, $DF_{\text{EXT},6}$, [$\text{Sv} \cdot \text{h}^{-1} \cdot \text{Bq}^{-1} \cdot \text{m}^2 \cdot \text{MeV}^{-1}$]	$1.5 \cdot 10^{-12}$
Conversion factor for external dose from a disc source, $DF_{\text{EXT},7}$, [$\text{Sv} \cdot \text{h}^{-1} \cdot \text{Bq}^{-1} \cdot \text{m}^2 \cdot \text{MeV}^{-1}$]	$4 \cdot 10^{-13}$
B :	see TABLE 15
$F(\Theta, b)$:	see Page 29
$DF_{\text{INH},i}$, $DF_{\text{ING},i}$:	see TABLE 18
$DF_{\text{SKIN},i}$:	see TABLE 18

TABLE 12

Values for atmospheric dispersion factor and thermal plume rise.

			Typical value	
Eff. release height: [m]	Distance: [km]	Stability class:		
Some more detailed predictions for the dispersion factor, χ , [Bq s m ⁻³ /Bq]:				
10	1	A (labile)	$10^{-6} - 10^{-4}$	
10	1	D (neutral)	$10^{-5} - 10^{-3}$	
10	1	D + rain	$10^{-5} - 10^{-3}$	
10	1	F (stable)	$10^{-6} - 10^{-4}$	
150	5	A	$10^{-8} - 10^{-6}$	
150	5	D	$10^{-7} - 10^{-5}$	
150	5	D + rain	$10^{-7} - 10^{-5}$	
150	5	F	$10^{-8} - 10^{-6}$	
Some predictions for the thermal plume-rise (and thus the effective release height, $h_{\text{eff}} = h_{\text{ref}} + h_{\text{therm}}$) of the radioactive plume is given in the following:				
Heat content of the plume released: [MW]		Plume rise h_{therm} : [m]		
Mean wind velocity:				
		1 m s^{-1}	4 m s^{-1}	7 m s^{-1}
0.1		50	30	25
0.5		90	60	50
1.0		110	70	60
10.0		250	160	130
50.0		410	260	210

TABLE 13

Parameter values for recycling, composition of typical steels used in power reactors.

Element	Mean value	Minimum value	Maximum value
B	$2.75 \cdot 10^{-5}$	$< 5 \cdot 10^{-6}$	$< 5 \cdot 10^{-5}$
C	$1.3 \cdot 10^{-3}$	$3 \cdot 10^{-4}$	$2.5 \cdot 10^{-3}$
N	$4 \cdot 10^{-4}$		
O	$1.5 \cdot 10^{-4}$		
Al		$1 \cdot 10^{-4}$	$1.5 \cdot 10^{-2}$
Si	$3.5 \cdot 10^{-3}$	$2 \cdot 10^{-3}$	$6 \cdot 10^{-3}$
P	$9.8 \cdot 10^{-4}$	$1.6 \cdot 10^{-4}$	$4 \cdot 10^{-3}$
S	$3 \cdot 10^{-4}$	$1.5 \cdot 10^{-4}$	$7 \cdot 10^{-4}$
Ti		$< 5 \cdot 10^{-4}$	$1.5 \cdot 10^{-2}$
Cr		$5 \cdot 10^{-4}$	$1.9 \cdot 10^{-1}$
Mn	$9.7 \cdot 10^{-3}$	$2 \cdot 10^{-3}$	$2 \cdot 10^{-2}$
Fe		$6.5 \cdot 10^{-1}$	$9.7 \cdot 10^{-1}$
Co	$9.3 \cdot 10^{-4}$	$1.1 \cdot 10^{-4}$	$2.5 \cdot 10^{-3}$
Ni		$1 \cdot 10^{-3}$	$4.2 \cdot 10^{-1}$
Cu	$1.5 \cdot 10^{-3}$		
Zn	$1 \cdot 10^{-4}$		
Se	$< 2 \cdot 10^{-4}$		
Zr	$< 5 \cdot 10^{-4}$		
Nb	$3.3 \cdot 10^{-3}$	$1 \cdot 10^{-4}$	$9 \cdot 10^{-3}$
Ag	$1.13 \cdot 10^{-6}$	$1.1 \cdot 10^{-6}$	$1.17 \cdot 10^{-6}$
Sn	$< 1 \cdot 10^{-4}$		
Sb	$< 1 \cdot 10^{-4}$		
Sm	$1 \cdot 10^{-8}$		
Eu	$1 \cdot 10^{-6}$		

TABLE 14

Distribution of contaminants during the melting process.

Element	i (ingot)	s (slag)	d (dust/fumes)
Mn	1	0.1 – 0.01	0.1 – 0.01
Fe	1	0.01	< 0.01
Co	1	0.1 – 0.01	< 0.01
Ni	1	0.1 – 0.01	< 0.01
Zn	1 – 0.01	0.1 – 0.01	1 – 0.01
Sr	0.1 – 0.01	1	0.1 – 0.01
Cs	0.1 – 0.01	1	1
Pu,Am,Cm	1 – 0.1	1	< 0.01

TABLE 15

Taylor dose build up factor coefficients for iron and concrete.

$$B = A \cdot e^{-\alpha_1 b} + (1-A) \cdot e^{-\alpha_2 b} , \quad (40)$$

where

A, α_1 , α_2 are the Taylor coefficients (see below)

$$b = \mu \cdot x_s ,$$

where

μ is the macroscopic cross section (i.e. linear attenuation coefficient, $[cm^{-1}]$ (see Fig. 3.)

x_s is thickness of the shield, [cm].

IRON			
E_γ [MeV]	A	α_1	α_2
0.25	48.63800	-.07220	-.05130
0.50	40.43700	-.06410	-.03940
0.60	38.04000	-.06300	-.03680
0.70	35.64300	-.06180	-.03410
0.80	33.24600	-.06070	-.03140
0.90	30.84800	-.05960	-.02880
1.00	28.45100	-.05840	-.02610
1.25	25.80500	-.05540	-.02100
1.50	23.16000	-.05230	-.01580
2.00	17.97400	-.04630	-.00570

CONCRETE			
E_γ [MeV]	A	α_1	α_2
0.50	66.3460	-.08797	-.06717
1.00	28.5400	-.06635	-.02881
2.00	50.2754	-.03119	-.01472

Sievert's integral and linear attenuation coefficient.

The Sievert's integral (or secant integral) $F(\Theta, b)$ is given by:

$$F(\Theta, b) = \int_0^{\Theta} e^{-b \cdot \sec \Theta'} d\Theta' . \quad (41)$$

For small values of Θ :

$$F(\Theta, b) = \Theta \cdot e^{-b} , \quad (42)$$

where

$$\Theta = \arctan(L/2a_R)$$

$$b = \mu_s \cdot x_s$$

L is the length of the line source (cylinder etc.), [m]
 a_R is distance between the worker and the line source, [m]
 μ_s is linear attenuation coefficient of shield material, [m^{-1}]
 (See Fig. 3)
 x_s is thickness of the shield, [m].

Values of the Sievert's integral $F(\Theta, b)$ for different layer thicknesses b and angles Θ are presented in Fig. 2.

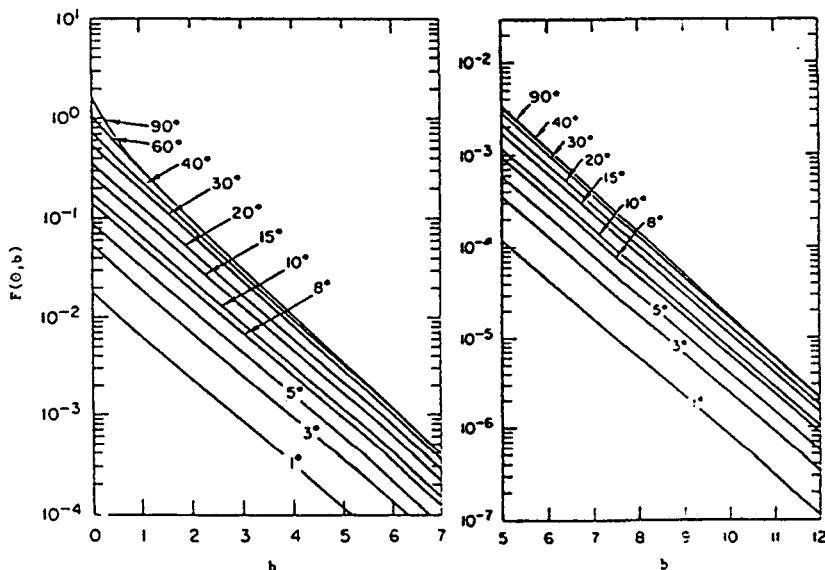


Fig. 2. Sievert's integral.

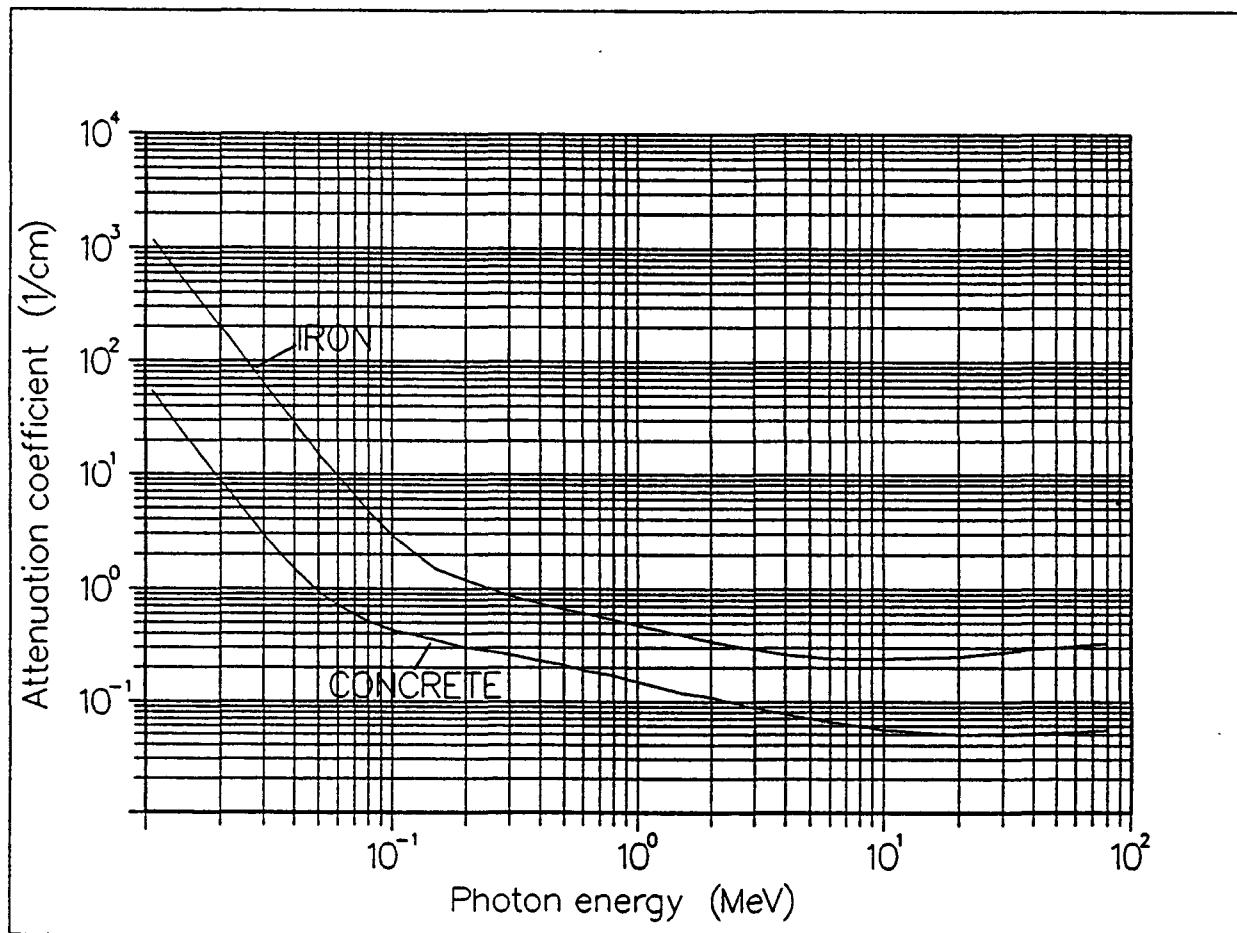


Fig. 3. Total linear attenuation coefficient μ_s [cm^{-1}] for iron and concrete.
($\mu_s = \sigma \cdot \rho$, $\rho(\text{Fe}) = 7.9 \text{ g cm}^{-3}$, $\rho(\text{concrete}) = 2.4 \text{ g cm}^{-3}$)

TABLE 16

Foodchain transfer parameter values for various elements.

Element	Freshwater fish concentration factor, $B_{f,i}$ [Bq·kg ⁻¹]/ [Bq·l ⁻¹]	Soil to plant concentration factor, $B_{v,i}$ [Bq·kg ⁻¹ _{f.w.}]/ [Bq·kg ⁻¹ _{d.w.}] (for vegetables)	Transfer coefficients to animal products, $P_{i,j}$ [d·l ⁻¹] or [d·kg ⁻¹]	
			Milk	Meat
H	1	1.5	0.02	0.03
C	5000	0.1	0.02	0.03
Na	100	0.08	0.04	0.03
Cl	30	1.5	0.02	0.01
Ca	20	1.5	0.01	0.001
Mn	100	0.02	0.001	0.001
Fe	3000	0.001	0.001	0.02
Co	300	0.02	0.01	0.005
Ni	100	0.02	0.01	0.005
Sr	20	1.5	0.001	0.001
Ru	100	0.02	10^{-6}	0.002
Sb	100	0.02	0.0001	0.001
I	30	0.2	0.01	0.01
Cs	1500	0.01	0.01	0.03
Ce	30	0.001	$5 \cdot 10^{-5}$	0.001
Eu	40	0.001	$5 \cdot 10^{-5}$	0.005
Pu	40	0.0001	$5 \cdot 10^{-6}$	0.001
Am	40	0.0001	$5 \cdot 10^{-6}$	0.001

TABLE 17

Time-integrated concentrations¹ in foodstuffs per unit deposition on the ground,

F_{ij} [$\text{Bq} \cdot \text{a}^{-1}$]/[$\text{Bq} \cdot \text{m}^{-2}$] or [$\text{Bq} \cdot \text{a} \cdot \text{kg}^{-1}$]/[$\text{Bq} \cdot \text{m}^{-2}$], food intake rates, Q_j [$\text{kg} \cdot \text{a}^{-1}$]

or [$\text{l} \cdot \text{a}^{-1}$] and fraction of contaminated foodstuff, A_j .

Radionuclide	Expected range of values for F_{ij}				
	Milk	Meat	Green vegetables	Wild berries	Mushrooms
H-3	$5 \cdot 10^{-4} - 5 \cdot 10^{-3}$	$10^{-4} - 10^{-3}$	$10^{-3} - 10^{-2}$	1 – 10	5 – 50
C-14	$10^{-3} - 10^{-2}$	$10^{-3} - 10^{-2}$	$10^{-3} - 10^{-2}$	$5 \cdot 10^{-1} - 5$	3 – 25
Na-22	$10^{-4} - 10^{-3}$	$10^{-4} - 10^{-3}$	$10^{-3} - 10^{-2}$	$10^{-2} - 10^{-1}$	$5 \cdot 10^{-2} - 5 \cdot 10^{-1}$
Cl-36	$10^{-3} - 10^{-2}$	$5 \cdot 10^{-4} - 5 \cdot 10^{-3}$	$10^{-3} - 10^{-2}$	5 – 50	25 – 250
Mn-54	$10^{-6} - 10^{-5}$	$10^{-6} - 10^{-5}$	$10^{-3} - 10^{-2}$	$10^{-3} - 10^{-2}$	$5 \cdot 10^{-3} - 5 \cdot 10^{-2}$
Fe-55	$5 \cdot 10^{-6} - 5 \cdot 10^{-5}$	$5 \cdot 10^{-5} - 5 \cdot 10^{-4}$	$10^{-3} - 10^{-2}$	$5 \cdot 10^{-4} - 5 \cdot 10^{-3}$	$10^{-3} - 10^{-2}$
Co-60	$10^{-4} - 10^{-3}$	$5 \cdot 10^{-5} - 5 \cdot 10^{-4}$	$10^{-3} - 10^{-2}$	$10^{-2} - 10^{-1}$	$5 \cdot 10^{-2} - 5 \cdot 10^{-1}$
Ni-63	$10^{-3} - 10^{-2}$	$10^{-4} - 10^{-3}$	$10^{-3} - 10^{-2}$	$10^{-1} - 1$	$5 \cdot 10^{-1} - 5$
Sr-90	$10^{-4} - 10^{-3}$	$10^{-5} - 10^{-4}$	$10^{-3} - 10^{-2}$	5 – 50	25 – 250
Ru-106	$5 \cdot 10^{-9} - 5 \cdot 10^{-8}$	$10^{-6} - 10^{-5}$	$10^{-3} - 10^{-2}$	$10^{-3} - 10^{-2}$	$5 \cdot 10^{-3} - 5 \cdot 10^{-2}$
Sb-125	$5 \cdot 10^{-7} - 5 \cdot 10^{-6}$	$10^{-6} - 10^{-5}$	$10^{-3} - 10^{-2}$	$5 \cdot 10^{-3} - 5 \cdot 10^{-2}$	$10^{-2} - 10^{-1}$
I-131	$10^{-4} - 5 \cdot 10^{-4}$	$10^{-5} - 10^{-4}$	$10^{-5} - 10^{-4}$	$10^{-5} - 10^{-4}$	$5 \cdot 10^{-5} - 5 \cdot 10^{-4}$
Cs-137	$10^{-3} - 10^{-2}$	$10^{-3} - 5 \cdot 10^{-2}$	$10^{-3} - 10^{-2}$	$5 \cdot 10^{-2} - 10^{-1}$	$10^{-1} - 1$
Ce-144	$10^{-7} - 10^{-6}$	$5 \cdot 10^{-7} - 5 \cdot 10^{-6}$	$10^{-3} - 10^{-2}$	$10^{-4} - 10^{-3}$	$5 \cdot 10^{-4} - 5 \cdot 10^{-3}$
Eu-152	$10^{-6} - 10^{-5}$	$5 \cdot 10^{-5} - 5 \cdot 10^{-4}$	$10^{-3} - 10^{-2}$	$10^{-3} - 10^{-2}$	$5 \cdot 10^{-3} - 5 \cdot 10^{-2}$
Pu-239	$5 \cdot 10^{-7} - 5 \cdot 10^{-6}$	$5 \cdot 10^{-5} - 5 \cdot 10^{-4}$	$10^{-3} - 10^{-2}$	$5 \cdot 10^{-4} - 5 \cdot 10^{-3}$	$10^{-3} - 10^{-2}$
Am-241	$5 \cdot 10^{-7} - 5 \cdot 10^{-6}$	$5 \cdot 10^{-5} - 5 \cdot 10^{-4}$	$10^{-3} - 10^{-2}$	$5 \cdot 10^{-4} - 5 \cdot 10^{-3}$	$10^{-3} - 10^{-2}$

Intake rates for the foodstuffs above, Q_j

75 – 365	30 – 70	10 – 20	1 – 5 ²	1 – 5 ²
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Contaminated fraction of the foodstuff, A_j

0.1 – 1	0.05 – 1	0.3 – 1 ²	0.3 – 1 ²	0.3 – 1 ²
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1) The time-integrated concentration values depend on the season when the deposition occurs.

2) Site specific.

TABLE 18

Dosimetric data.

Nuclide	Effective dose factors for different exposure types										
Committed effective dose per unit intake by inhalation and ingestion [Sv·Bq ⁻¹]	Ext. dose, infinite absorbing medium [Sv·h ⁻¹]/[Bq·g ⁻¹]	Ext. dose, infinite surface [Sv·h ⁻¹]/[Bq·m ⁻²]	Ext. dose, infinite plume [Sv·h ⁻¹]/[Bq·m ⁻³]	Skin dose, thickness 4 mg/cm ² [Sv·h ⁻¹]/[Bq·m ⁻²]	DF _{INH,i}	DF _{ING,i}	DF _{EXT,i}	DF _{DE,i}	DF _{PE,i}	DF _{SKIN,i}	
H-3	1.7·10 ⁻¹¹	1.7·10 ⁻¹¹	0	0	0	0	0	0	0	0	
C-14	5.6·10 ⁻¹⁰	5.6·10 ⁻¹⁰	0	0	0	0	0	0	0	9·10 ⁻⁷	
Na-22	2.2·10 ⁻⁹	3.2·10 ⁻⁹	4.4·10 ⁻⁷	3.5·10 ⁻¹²	3.5·10 ⁻¹⁰	2.5·10 ⁻⁶					
Cl-36	1.8·10 ⁻⁹	1.8·10 ⁻⁹	0	0	0	0	0	0	0	2.5·10 ⁻⁶	
Mn-54	1.7·10 ⁻⁹	7.3·10 ⁻¹⁰	1.7·10 ⁻⁷	1.5·10 ⁻¹²	1.3·10 ⁻¹⁰	6·10 ⁻⁸					
Fe-55	9.4·10 ⁻¹⁰	1.6·10 ⁻¹⁰	5.5·10 ⁻¹¹	4.3·10 ⁻¹⁶	4.3·10 ⁻¹⁴	2·10 ⁻⁸					
Co-60	4.1·10 ⁻⁸	7.0·10 ⁻⁹	5.0·10 ⁻⁷	4.0·10 ⁻¹²	4.0·10 ⁻¹⁰	2·10 ⁻⁶					
Ni-63	8.4·10 ⁻¹⁰	1.5·10 ⁻¹⁰	0	0	0	0	0	0	0	2·10 ⁻⁸	
Sr-90	3.4·10 ⁻⁷	3.6·10 ⁻⁸	0	0	0	0	0	0	0	5·10 ⁻⁶	
Ru-106	1.2·10 ⁻⁷	5.8·10 ⁻⁹	4.0·10 ⁻⁸	3.5·10 ⁻¹³	3.5·10 ⁻¹¹	3·10 ⁻⁶					
Sb-125	3.3·10 ⁻⁹	8.1·10 ⁻¹⁰	1.9·10 ⁻⁷	1.5·10 ⁻¹²	1.5·10 ⁻¹⁰	1.4·10 ⁻⁶					
I-131	8.8·10 ⁻⁹	1.4·10 ⁻⁸	7.9·10 ⁻⁸	7.6·10 ⁻¹²	6.1·10 ⁻¹¹	2.4·10 ⁻⁶					
Cs-137	8.7·10 ⁻⁹	1.4·10 ⁻⁸	1.1·10	9.7·10 ⁻¹³	9.0·10 ⁻¹¹	2.6·10 ⁻⁶					
Ce-144	9.5·10 ⁻⁸	5.3·10 ⁻⁹	1.0·10 ⁻⁸	9.7·10 ⁻¹⁴	7.9·10 ⁻¹²	4.6·10 ⁻⁶					
Eu-152	5.0·10 ⁻⁸	1.2·10 ⁻⁹	4.5·10 ⁻⁷	3.6·10 ⁻¹²	3.6·10 ⁻¹⁰	1.7·10 ⁻⁶					
Pu-239	1.2·10 ⁻⁴	9.7·10 ⁻⁷	0	0	0	0	0	0	0	-	
Am-241	1.4·10 ⁻⁴	1.2·10 ⁻⁶	4.0·10 ⁻⁹	4.0·10 ⁻¹⁴	3.2·10 ⁻¹²	-					

1) Factor of 0.7 has been used to convert absorbed dose in air to effective dose.

TABLE 19

Mass attenuation coefficient of energy absorption, μ_e/ρ , [cm²·g⁻¹]

Photon energy, [Mev]	Water	Air	Bone	Muscle
0.010	4.84	4.65	19.0	4.96
0.015	1.34	1.30	5.89	1.36
0.020	0.536	0.527	2.51	0.544
0.030	0.152	0.150	0.743	0.154
0.040	0.0680	0.0671	0.305	0.0677
0.050	0.0415	0.0404	0.158	0.0409
0.060	0.0315	0.0301	0.0979	0.0312
0.080	0.0258	0.0239	0.0520	0.0255
0.10	0.0254	0.0232	0.0386	0.0252
0.15	0.0276	0.0249	0.0304	0.0276
0.20	0.0297	0.0267	0.0302	0.0297
0.30	0.0319	0.0287	0.0311	0.0317
0.40	0.0328	0.0295	0.0316	0.0325
0.50	0.0330	0.0297	0.0316	0.0327
0.60	0.0328	0.0295	0.0315	0.0326
0.80	0.0321	0.0288	0.0306	0.0318
1.0	0.0310	0.0279	0.0297	0.0308
1.5	0.0283	0.0255	0.0270	0.0281
2.0	0.0260	0.0234	0.0248	0.0257
3.0	0.0228	0.0206	0.0219	0.0225
4.0	0.0206	0.0187	0.0199	0.0203
5.0	0.0192	0.0174	0.0186	0.0188
6.0	0.0181	0.0165	0.0178	0.0178
8.0	0.0166	0.0152	0.0165	0.0163
10.0	0.0157	0.0145	0.0159	0.0154

TABLE 20

Irradiation-rate constants Γ and kerma-rate constants Γ_δ for some radionuclides.

Radionuclide	$\Gamma \cdot 10^{-9} * \frac{(\text{C/kg})\text{m}^2}{\text{M}\cdot\text{Bq}\cdot\text{h}}$	$\Gamma_\delta \cdot 10^{-9} * \frac{\text{Gy m}^2}{\text{M}\cdot\text{Bq}\cdot\text{h}}$
Na-22	8.36	284
Na-24	12.8	435
K-42	1.39	47.2
Cr-51	0.111	3.77
Co-60	9.19	312
Zn-65	1.88	63.9
Sb-122	1.67	56.7
I-125	0.487	16.5
I-131	1.53	52.0
I-132	9.33	317
Cs-134	6.33	215
Cs-137	2.30	78.1
Ir-192	3.34	113
Au-198	1.60	54.4
Hg-203	0.905	30.7
Ra-226	5.78	196
Am-241	0.11	3.8

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APPENDIX 2

DETERMINATION OF SOME DIFFICULT-TO-MEASURE RADIONUCLIDES

This Appendix describes analytical methods for the determination of the activity of difficult-to-measure nuclides in nuclear wastes. Such nuclides are pure beta or alpha emitters like carbon-14, iron-55, nickel-63, strontium-90, technetium-99 and the transuranics. For the radioassay, careful and complicated preparation of samples and measurements with sensitive instruments are needed.

The procedures described in the report are based on the investigations performed for ion exchange resins and process waters from the BWR and PWR nuclear power plants in Finland. The methods for Fe-55, Ni-63, Sr-90, Tc-99 and for the transuranium elements (Pu, Am and Cm) have been developed at the Department of Radiochemistry, University of Helsinki /1/ and the method for C-14 at the Finnish Centre for Radiation and Nuclear Safety, surveillance Department and at the Technical Research Centre of Finland, Reactor Laboratory /2,3/.

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- 7. Determination of alpha-emitting transuranics**

REFERENCES

FIGURES

1. PREPARATION OF SAMPLES FOR ANALYSIS

Solid waste samples

The dried ion exchange resin sample (cation, anion or mixed resin) is transferred with water to a sintered-glass crucible. The sample is treated with nitric acid and hydrochloric acid in order to elute the radionuclides from the resin. The elution sequence is 4 M nitric acid, 2 M, 4 M and 8 M hydrochloric acids. The eluate is evaporated carefully to dryness, dissolved in 8 M hydrochloric acid and water is added to adjust the HCl concentration to 2 M. The resin is removed from the sintered-glass crucible with water to a filter paper (Whatman 40), magnesium nitrate is added and the filter paper (in crucible) is ashed in a furnace at 500 °C. The residue is dissolved in 8 M HCl and water is added to adjust the HCl concentration to 2 M. One ion exchange resin sample is divided to two fractions, the eluate and the residue, which is ashed and dissolved in 8 M HCl. From these solution fractions aliquots for radionuclide analysis are taken.

The ion exchange resin sample containing water is filtered through the sintered-glass crucible. The resin is treated like the dried resin sample. The water is radioassayed with gammascintrometry and depending of the results of the radioassay, analysis of the radionuclides is made with the same procedures as with the resin sample.

Carbon-14 content is determined from wet resin samples. To avoid losses of CO₂, CO or hydrocarbons the samples can't be dried. At sampling the wet resin sample is placed in airtight vial to be transferred to the reaction vessel for determination of C-14.

Liquid samples

Water sample is usually preserved by adding concentrated nitric acid to prevent adsorption of radionuclides on the walls of the sample vial. Aliquots for analysis can be taken directly from these solutions. Nitric acid is not added to the water samples for determination of C-14 in order to keep the CO₂, CO and hydrocarbons in solution. The C-14 sample solution is transferred to the sample preparation vial for analysis of C-14.

2. DETERMINATION OF CARBON-14

The long-living radioactive isotope of carbon, C-14, is produced during the operation of nuclear reactors due to ternary fission and neutron capture of different nuclides. The most important neutron reactions are the (n,γ) with C-13, the (n,p) with N-14 and the (n,α) with O-17. Carbon, nitrogen and oxygen exist as parts or impurities in the reactor system. Carbon-14 (half-life 5730 a) is a beta emitter ($E_{\beta\max} = 158$ keV) and can be assayed with liquid scintillation counting.

The studies on carbon-14 releases from the nuclear power plants have indicated that of the order of only a few percent of the stack releases is retained in the ion exchange resins used for purification of nuclear reactor process waters. However, the amount of C-14 absorbed in the resins can be relatively large and will be released during the treatment of waste resins.

Outline of the method

Carbon-14 exists as inorganic carbonate or in organic form in the sample. The procedure to measure the total C-14 content of the sample involves transformation of all carbon to carbon dioxide and collection of the evolved CO₂.

The main steps of the separation method used for determination of C-14 in process water and ion exchange resin samples is shown in the figure 1. The sample is transferred to the sample preparation vial (Figure 2). Hydrochloric acid (6 M) is added and air is bubbled through the solution to remove the evolved CO₂, which is absorbed in 2 M NaOH solution (first absorber). Then the gas is passed over an heated catalyst (CuO at 800 °C), in which the CO and hydrocarbons are oxidized to CO₂. This CO₂ is absorbed also in 2 M NaOH solution (second absorber). After 1.5 hours the collection of CO₂ is complete and the NaOH solutions from absorbers 1 and 2 are combined. Ammonium chloride and barium chloride solutions (2 M) are added to precipitate BaCO₃, which is washed and dried. The dry BaCO₃ is made to fine powder and mixed with scintillation coctail. The C-14 is radioassayed in rigid gel by liquid scintillation counter /2,3/.

3. DETERMINATION OF IRON-55

Iron-55 is produced by neutron activation of the stainless steel components of nuclear reactors. Therefore Fe-55 is commonly present in nuclear wastes. Iron-55 decays completely by electron capture into stable Mn-55. Its half-life is 2.7 years. The decay energy is 0.232 MeV. Most of the decay energy (about 97%) is carried off by neutrinos. The resulting excited state of the Mn-55 has, in the case of K electron capture, 6.54 keV of residual energy. The proportion of K captures is 89.7%. The modes of de-excitation are X-ray emission and Auger electron emission.

Outline of the method

The content of inactive iron in the solution is determined by atomic absorption spectrophotometry. The solution is evaporated to a small volume and made 8 M in regard to hydrochloric acid. The iron is then extracted from the solution into di-isopropyl ether. The iron in the organic phase is stripped into water. The iron is then precipitated as ferric oxide hydrate and the precipitate dissolved with 0.2 M hydrochloric acid. Following this, ion exchange is carried out. Iron is eluted from the column with 0.5% oxalic acid. It is then precipitated as ferric oxide hydrate. The precipitate is dissolved in hydrofluoric acid and the solution diluted to a given volume by water. From this solution a small aliquot is taken for determination of inactive iron and for calculation of the chemical yield. A suitable amount of the remaining solution is transferred to a liquid scintillation vial for radioassay of Fe-55 /4/. The separation scheme of the Fe-55 is presented in the figure 3.

The radioassay of Fe-55 is based on the determination of soft X-ray ($K_{\alpha}=5.9$ keV) and/or Auger electrons by liquid scintillation counting or by X-ray spectrometry. At present the liquid scintillation counting is preferred to X-ray spectrometry (e.g. proportional counter) due to its one order of magnitude higher counting efficiency. In addition the maximum amount of stable iron in sample prepared for radioassay of Fe-55 can be about twenty times higher for liquid scintillation counting than for X-ray spectrometry. This is an significant advantage especially

when the radioactivity of Fe-55 per mg stable iron in sample is low. The commercially available low background liquid scintillation counters developed have also increased the use of liquid scintillation method for radioassay of Fe-55.

4. DETERMINATION OF NICKEL-63

The radioactive isotope of nickel, Ni-63 (half-life 100 a), is mainly formed as a neutron activation product from stable nickel, which is in the primary coolant due the corrosion of the steel of the primary coolant circuitry. Nickel-63 is a low energy ($E_{\beta\max} = 60$ keV) beta emitter which can be radioassayed with a liquid scintillation counter.

Outline of the method

The separation procedure for nickel is based on carbonate precipitation, ion exchange and two successive dimethylglyoxime precipitations. Nickel, cobalt, manganese and zinc solutions are added as carriers to the sample solution. Concentrated nitric acid is added and the solution is evaporated carefully to dryness. The residue is dissolved in nitric acid and sodiumcarbonate and sodium hydroxide solutions are added to precipitate the metals as carbonates. The precipitate is dissolved in concentrated hydrochloric acid, the solution is evaporated to dryness and the residue is dissolved in 8 M hydrochloric acid. The solution is passed through an ion exchange column (Dowex 1X8). From the eluate nickel is precipitated as nickel dimethylglyoxime. This step is repeated. The precipitate is dissolved in concentrated nitric acid and the solution evaporated a few times to dryness. The residue is heated until a black nickel oxide is formed. Nickel oxide is dissolved in dilute hydrochloric acid and silver chloride precipitation is made to remove possible Ag-110m from the solution. The Ni-63 is radioassayed by liquid scintillation counting /1/. The procedure is described in figure 4.

5. DETERMINATION OF STRONTIUM-90

The fission product strontium Sr-90 is a beta emitter ($E_{\beta\max} = 0.5$ MeV, half-life 28.5 a) decaying to yttrium Y-90 ($E_{\beta\max} = 2.3$ MeV, half-life 66 h). Sr-90 can be radioassayed by liquid scintillation counting either assaying the Sr-90, from which the Y-90 is separated by precipitation or radioassaying a solution in which the Y-90 has reached radioactive equilibrium with Sr-90.

Outline of the method

The method used for separation of strontium-90 from other radionuclides is based on different precipitation steps. Strontium and barium are precipitated as nitrates with fuming nitric acid. The filtrate is discarded and the precipitate is dissolved in water. Barium is separated by precipitation as chromate. From the filtrate strontium is precipitated as carbonate, which is dissolved in nitric acid. The carbonate precipitation is repeated. Strontium carbonate is dissolved in nitric acid and yttrium-90 is removed by ferric oxide hydrate co-precipitation. From the filtrate strontium is precipitated as carbonate, which is dissolved in dilute hydrochloric acid. Strontium-90 concentration can be determined by liquid scintillation counting immediately after the separation of yttrium-90. The Sr-90 solution is mixed with scintillation cocktail and radioassayed with the liquid scintillation counter. Another method to determine Sr-90 content is to allow Y-90 to reach radioactive equilibrium with Sr-90 by storing the

solution at least 14 days. Yttrium-90 emits high energy beta particles which produce Cerenkov radiation in water solution. This radiation can be detected with the liquid scintillation counter and the concentration of the Sr-90 in the solution can be calculated /1/. The method is described in the figure 5.

In the case sample contains also Sr-89 ($E_{\beta\max} = 1.5$ MeV, half-life 50.5 d), for example in fuel leakage situation, the determination of Sr-89 and Sr-90 is performed by separating the Y-90 from the solution and detecting the Cerenkov radiation in water solution. Also the beta particles from Sr-89 produce Cerenkov radiation. When Y-90 has reached radioactive equilibrium with Sr-90, the solution is radioassayed again by Cerenkov counting to determine the sum content of Sr-89 and Sr-90. From these results the Sr-89 and Sr-90 concentrations in the sample can be calculated.

6. DETERMINATION OF TECHNETIUM-99

Technetium-99 ($E_{\beta\max} = 292$ keV, half-life 2.13×10^5 a) is a fission product, which decays to Mo-99. Technetium-99 is mainly produced by thermal neutrons in the reactor with a fission yield of 6%. The radioactivity of technetium can be determined with low background liquid scintillation counting.

Outline of the method

The separation of technetium is based on extraction into tributylphosphate (TBP). Technetium-99m is used to monitor the chemical yield of the separation procedure. Cobalt-60, iron-55 and silver-110m are the main radionuclides to be removed from the solution before the TBP extractions. The Ag-110m is separated by precipitation as chloride. The precipitation of ferric oxide hydrate removes Co-60 and Fe-55. Technetium is extracted into TBP in sulfuric acid solution. The TBP phase is washed with sulfuric acid and hydrochloric acid discarding the acid phases. The radioassay of Tc-99m in TBP phase is performed with a gamma counter (NaI detector). After five days (allowing the Tc-99m to decay) liquid scintillation cocktail is added to the TBP and the radioactivity of the solution is determined with a low background scintillation counter /1/. Figure 6 describes the separation procedure of Tc-99.

7. DETERMINATION OF ALPHA-EMITTING TRANSURANICS

Plutonium, americium and curium are actinide elements and are formed in the reactor by sequential neutron induced reactions and through alpha and beta disintegrations.

Table 1. The radionuclides of actinides determined. The half-lives, the main alpha-energies and their relative intensities (%) are given.

Nuclide	Half-life (a)	E_α (MeV)
Pu-238	87.7	5.498 (71.2%) 5.454 (28.7%)
Pu-239	2.41×10^4	5.155 (73.3%) 5.143 (15.1%)
Am-241	432	5.486 (86.0%) 5.443 (12.7%)
Cm-242	0.446	6.113 (74.2%) 6.069 (25.8%)
Cm-243	28.5	5.785 (73.5%) 5.741 (10.6%)
Cm-244	18.1	5.806 (76.7%) 5.764 (23.3%)

Outline of the method

The separation of Am and Cm from Pu is based on ion exchange in nitric acid. The chemical behaviour of curium is very close of that of americium. Thus the Am fraction also contains the curium isotopes. Concentrated nitric acid is added to the sample solution and the solution is evaporated to dryness. The residue is dissolved in 8 M nitric acid and passed through an ion exchange column (Dowex 1X8). Plutonium is absorbed on the ion exchanger and americium and curium stay in the effluent. The column is washed with 10 M hydrochloric acid. Plutonium is eluted with the mixture of ammonium iodide and 10 M hydrochloric acid solutions. The americium, curium and plutonium fractions are evaporated to dryness and the residues are dissolved in dilute nitric acid. The americium, curium and plutonium are coprecipitated with neodymium fluoride. The precipitates on filters are radioassayed by alphaspectroscopy with surface barrier detectors. The resolution of the detector is not high enough to separate the alpha-energies of Pu-239 and Pu-240 as well as the alpha-energies of Cm-243 and Cm-244 /1/. The results of these isotopes are presented as a sum indicated as Pu-239,240 and Cm-243,244 /1/. The method is described in the figure 7.

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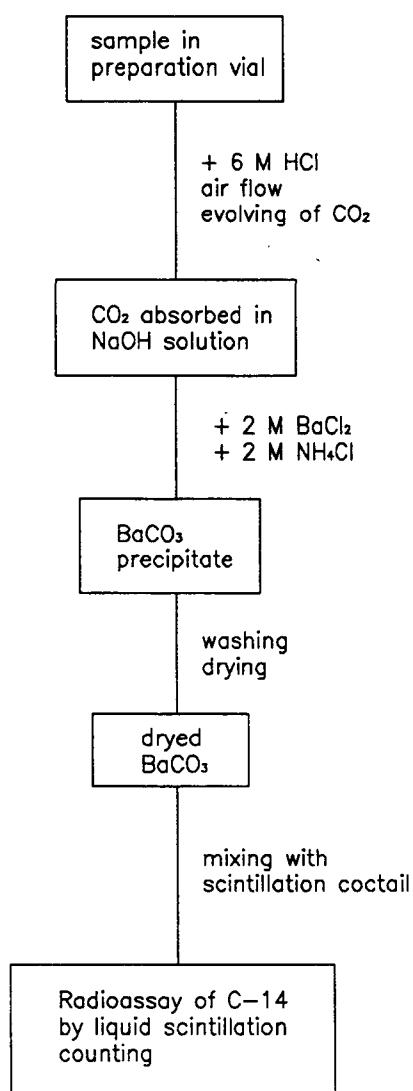


Figure 1. The separation scheme of C-14.

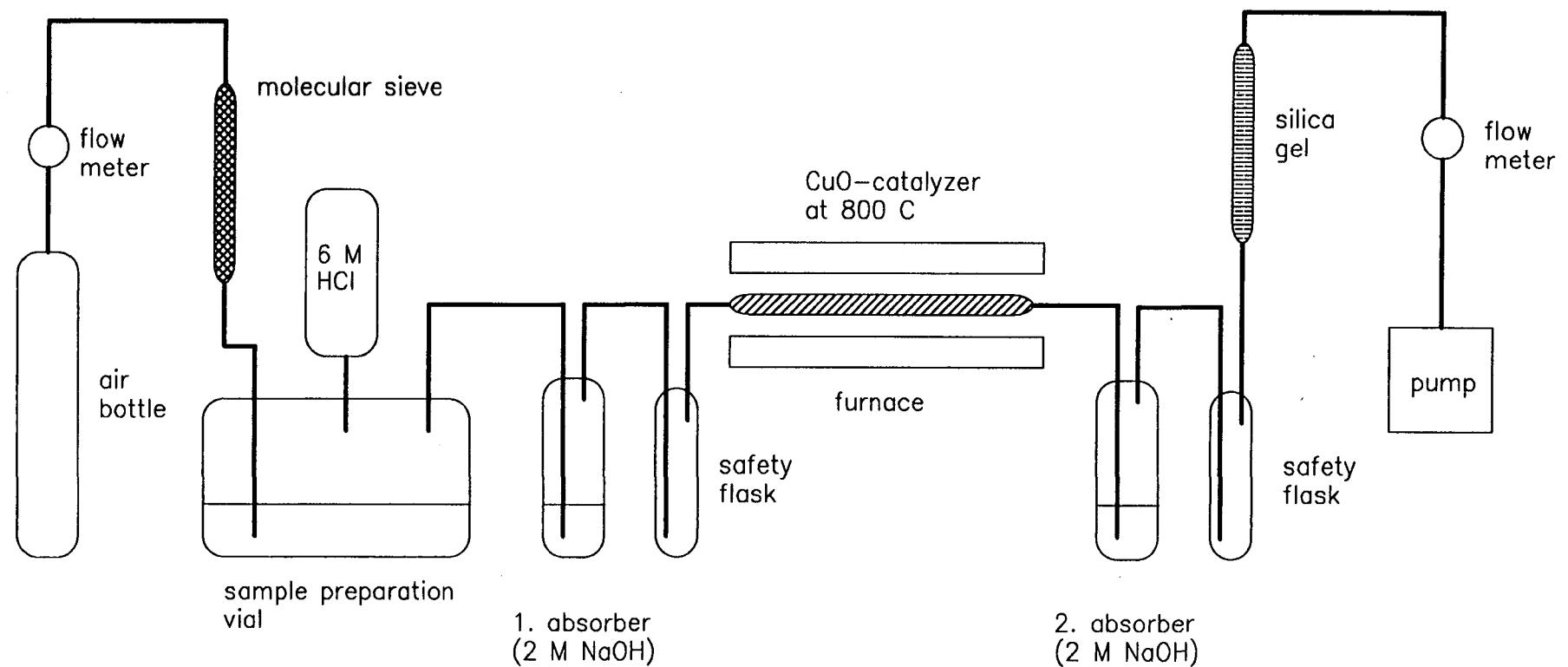


Figure 2. A schematic diagram of the system used for separation of C-14 from the sample.

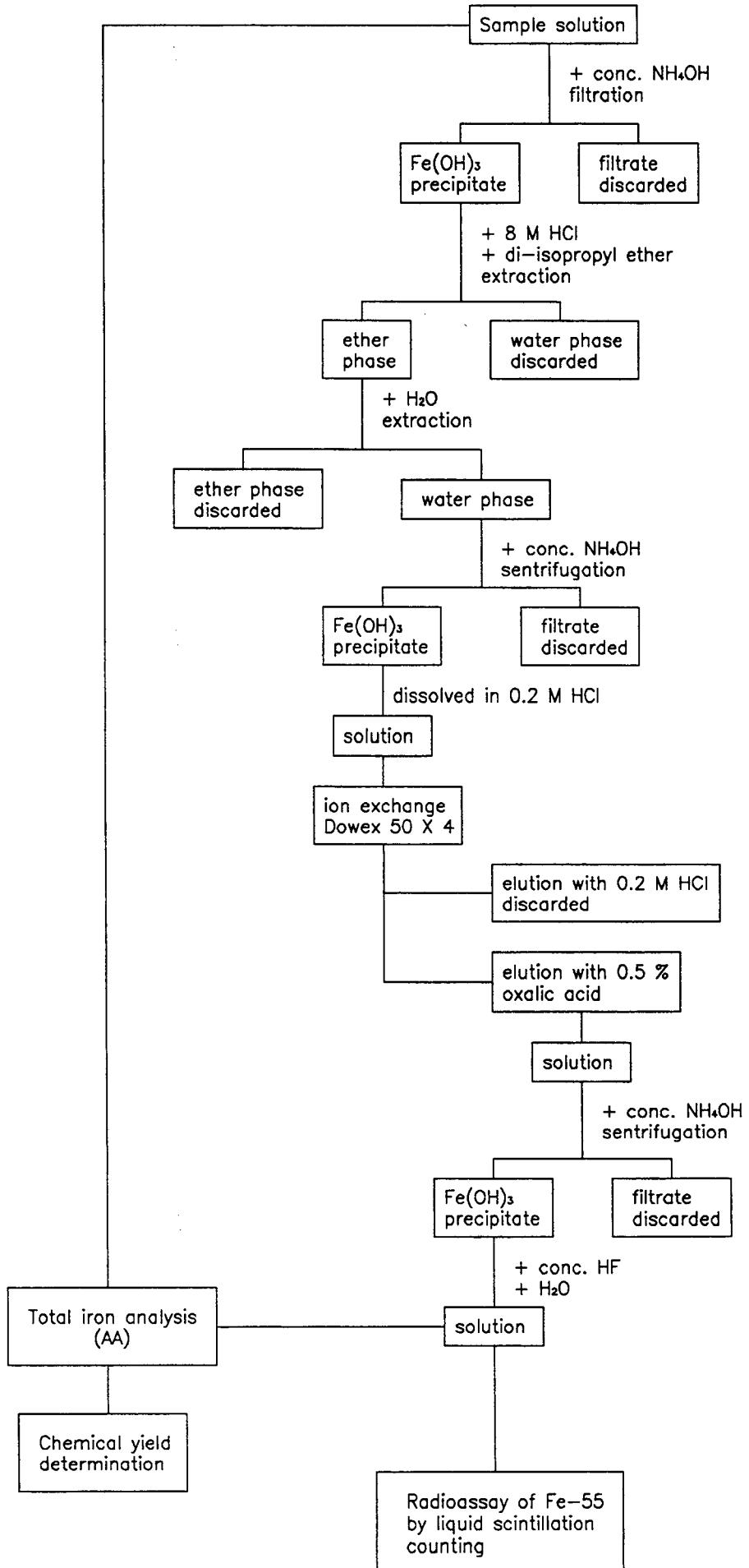


Figure 3. The separation scheme of Fe-55.

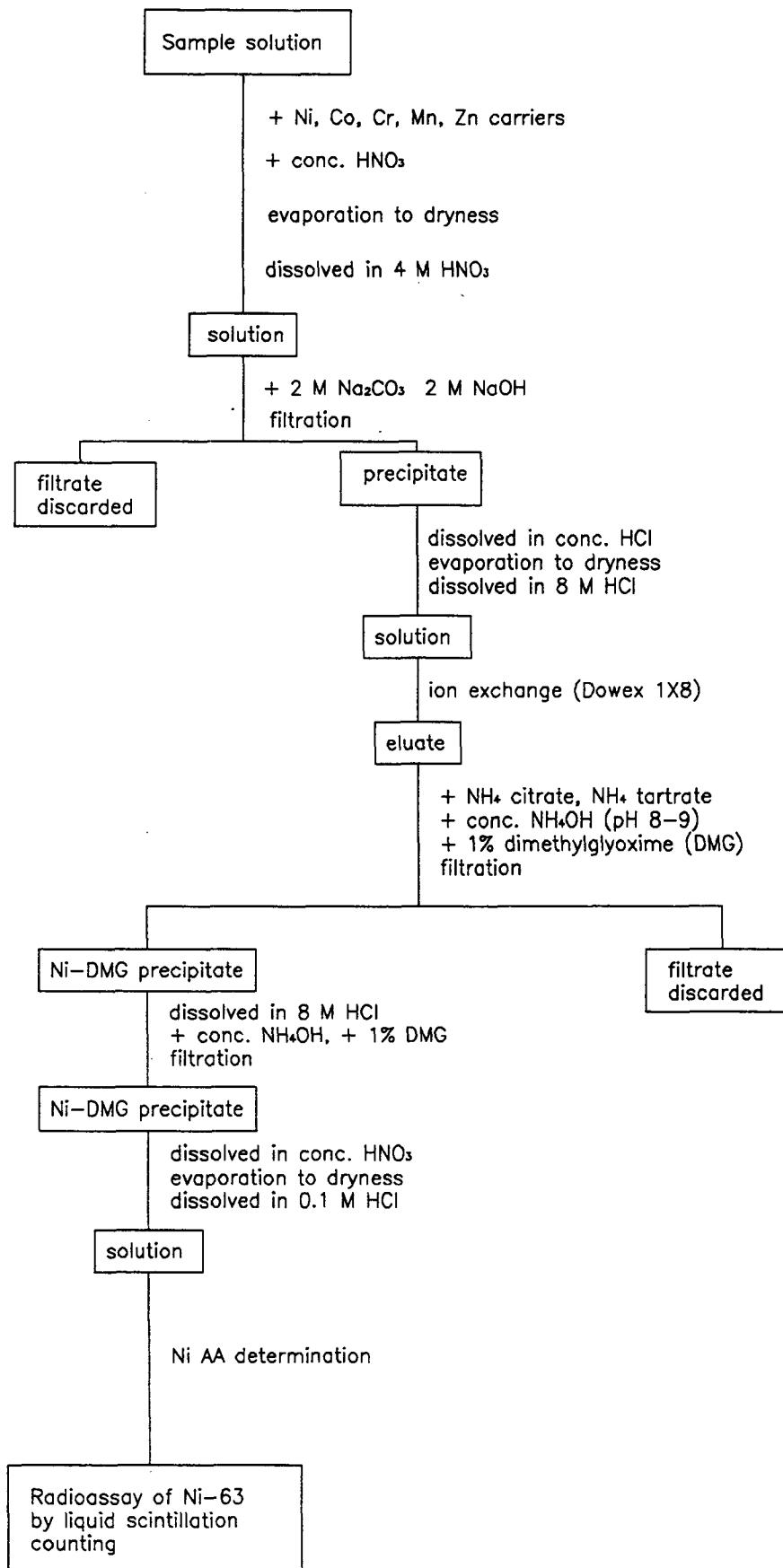


Figure 4. The separation scheme of Ni-63.

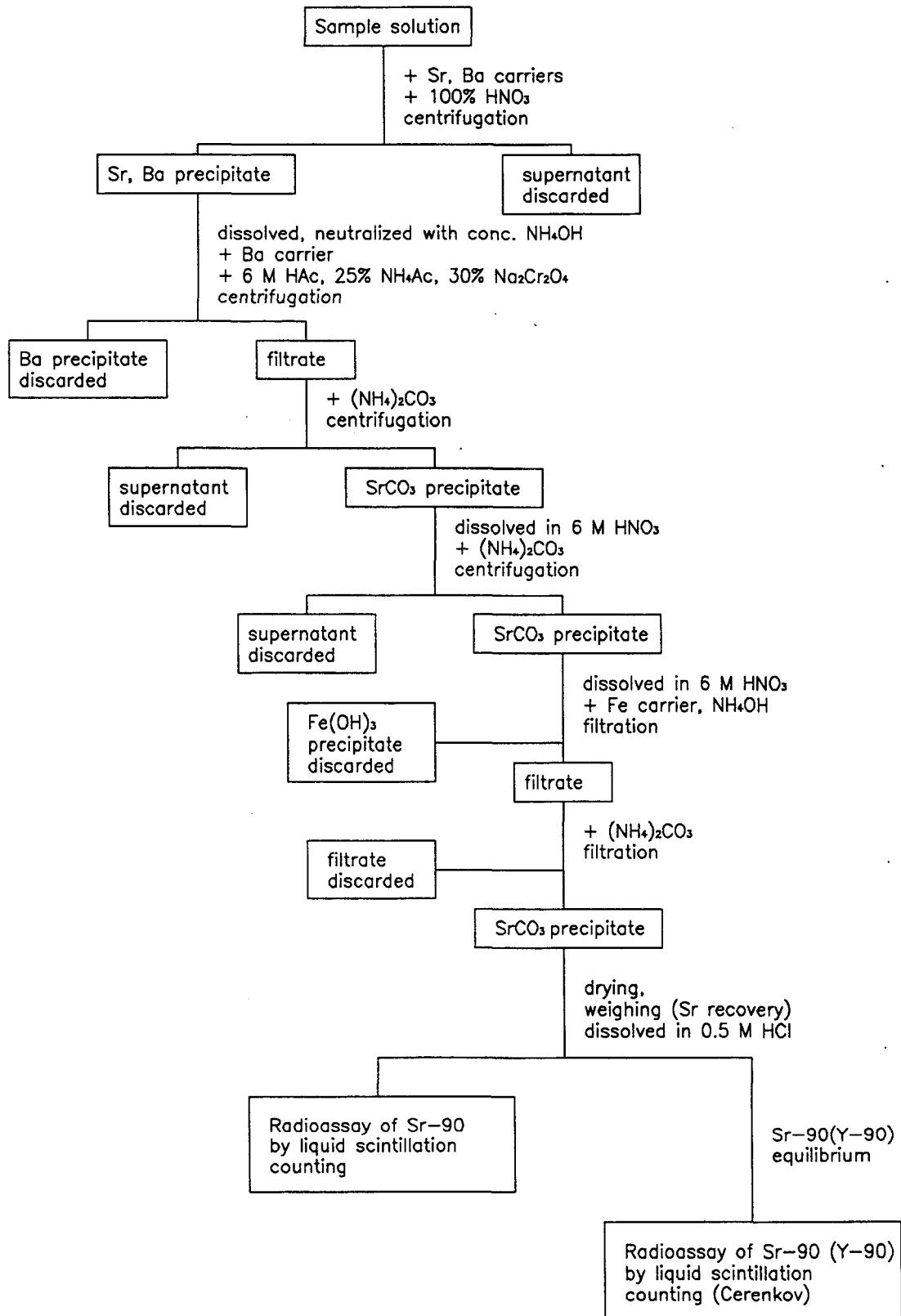


Figure 5. The separation scheme of Sr-90.

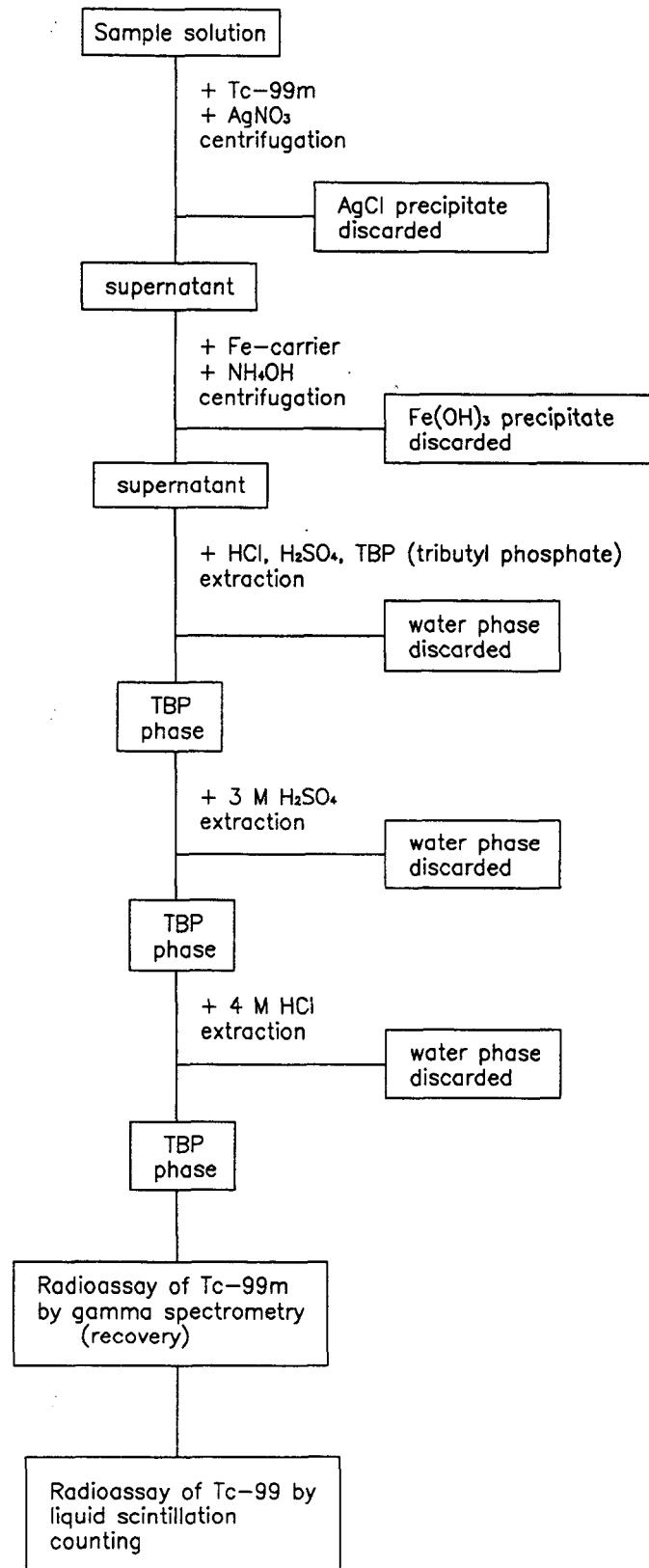


Figure 6. The separation scheme of Tc-99.

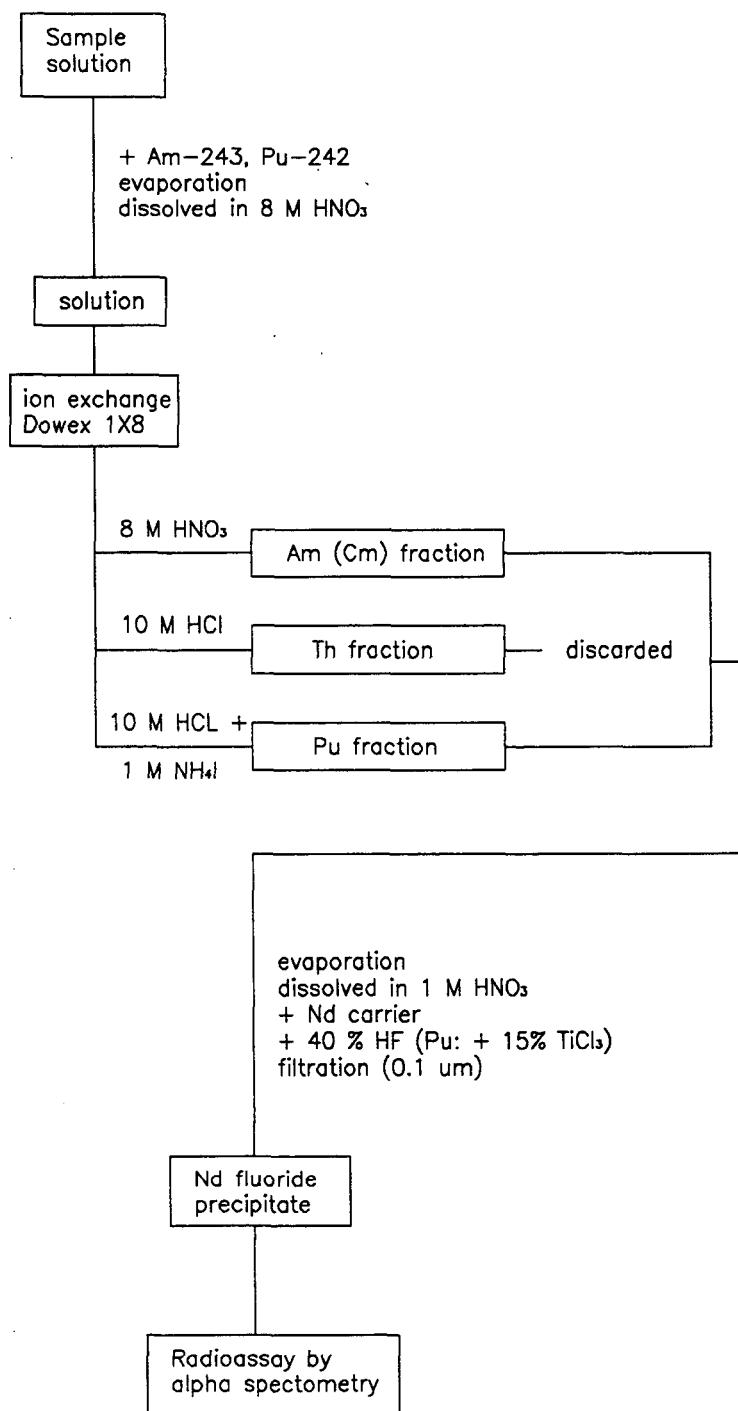


Figure 7. The separation scheme of transuranic nuclides (Pu, Am, Cm).

Guidance on Clearance from Regulatory Control of Radioactive Materials

The term radioactive waste is used for all waste generated within a nuclear installation. A considerable part of this waste contains so low activities that it can be recycled or disposed of without any radiological consequences. This report contains guidance to assist applicants and authorities to determine whether a certain part of such waste can be cleared from regulatory control in order that it can be used for other purposes. In the Nordic countries the highest amounts to be cleared will arise when the nuclear power plants are to be decommissioned.

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)

The programme is managed - and financed - by a consortium comprising the Danish Emergency Management Agency, the Finnish Ministry of Trade and Industry, Iceland's National Institute of Radiation Protection, the Norwegian Radiation Protection Authority, and the Swedish Nuclear Power Inspectorate. Additional financing is offered by the IVO and TVO power companies, Finland, as well as by the following Swedish organizations: KSU, OKG, SKN, SRV, Vattenfall, Sydkraft, SKB.

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