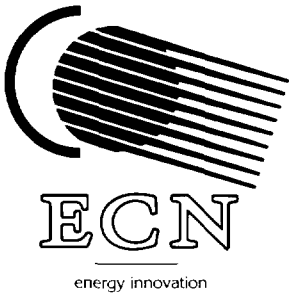
OCTOBER 1995

ECN-I--95-039



THE RADIOLOGICAL RISKS ASSOCIATED WITH THE THORIUM FUELLED HTGR FUEL CYCLE

A Comparative Risk Evaluation

D.H. DODD
J.F.A. VAN HIENEN

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OCTOBER 1995

KS002045665
R: FI
DE008891426

ECN-I--95-039

DE008891426

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Revision	
A	Draft: 16 August 1995: For Verification
B	Draft: 14 September 1995: For Review
0.0	Final: 28 September 1995
1.0	FINAL: 5 October 1995

Author: D.H. Dodd	**Approved By:** J. Prij	ECN Nuclear Energy Risk Analysis Section
Checked By: J.F.A. van Hienen	**Issued By:** C.D. Andriesse	

Preface

This report describes the results of a scoping study to evaluate the radiological risks to the general public associated with a nuclear fuel cycle based on a thorium fuelled high temperature gas cooled reactor. This work was performed for task B.3 of the "Technology Assessment of the High Temperature Reactor" research project. This project was carried out by the Netherlands Energy Research Foundation (ECN) under contract to the Dutch Ministry for Economic Affairs, ECN project number 17198.

The work for task B.3 was carried out by the Nuclear Energy Business Unit of the Netherlands Energy Research Foundation.

Contents

Summary

This report presents the results of task B.3 of the "Technology Assessment of the High Temperature Reactor" project. The objective of task B.3 was to evaluate the radiological risks to the general public associated with the sustainable HTGR cycle. Since the technologies to be used at several stages of this fuel cycle are still in the design phase and since a detailed specification of this fuel cycle has not yet been developed, the emphasis was on obtaining a global impression of the risk associated with a generic thorium-based HTGR fuel cycle. This impression was obtained by performing a comparative risk analysis on the basis of data given in the literature. As reference for the comparison a generic uranium fuelled LWR cycle was used.

The major benefit with respect to the radiological risks of basing the fuel cycle around modular HTGR technology instead of the LWR technology is the increase in reactor safety. The design of the modular HTGR is expected to prevent the release of a significant amount of radioactive material to the environment, and hence early deaths in the surrounding population, during accident conditions. This implies that there is no group risk as defined in the Dutch risk management policy.

The major benefit of thorium based fuel cycles over uranium based fuel cycles is the reduction in the radiological risks from uranium mining and milling. The other stages of the nuclear fuel cycle which make a significant contribution to the radiological risks are electricity generation, reprocessing and final disposal. The risks associated with the electricity generation stage are dominated by the risks from fission products, activated corrosion products and the activiation products tritium and carbon-14. The risks associated with the reprocessing stage are determined by fission and activation products (including actinides). For the final disposal stage, the reduction in the overall radiotoxicity in the waste which can be achieved using thorium fuels does not lead to a corresponding reduction in the radiological risks. For these three stages, significant differences in the radiological risks between uranium and thorium fuelled HTGR cycles are not expected.

1. Introduction

Modular high temperature gas cooled reactors (HTGRs) currently occupy a central position in the Dutch advanced nuclear reactor research programme. This central position is largely due to the generic HTGR design, which is to a large degree based upon passive safety principles. The design is intended to eliminate the possibility of core meltdown and the associated release of a significant amount of radioactive material to the environment during accident conditions. This feature is considered to make HTGRs an attractive nuclear option.

However, reactor safety is only one of the factors which will determine whether or not there is societal support for and acceptance of the HTGR technology. Eventual societal acceptance will depend upon a wide range of factors which includes: the technical suitability of the HTGR as a source of process heat in the chemical industry; the technical suitability of the HTGR as an electricity generating technology; the economic competitiveness of the HTGR; and the potential contribution of the HTGR to a sustainable energy policy.

The objective of the "Technology Assessment of the High Temperature Reactor" project is to assess the potential of the HTGR with respect to such issues. The project is divided into two parts. In part A an assessment is made of the technical and economic suitability of the HTGR for the chemical and electricity industries. In part B an assessment is made of the societal acceptability of a sustainable HTGR fuel cycle. The sustainable HTGR fuel cycle defined, is based upon thorium fuel and is one in which the spent HTGR fuel is reprocessed.

This report presents the results of task B.3 of the project which was to evaluate the radiological risks to the general public associated with the sustainable HTGR cycle. Since the technologies to be used at several stages of this fuel cycle are still in the design phase and since a detailed specification of the fuel cycle has not yet been developed, the emphasis was on obtaining a global impression of the risk associated with a generic thorium-based HTGR fuel cycle. This impression was essentially obtained by performing a comparative risk analysis on the basis of data given in the literature. As reference for the comparison a generic uranium fuelled LWR cycle was used. Where significant differences are to be expected in the risks associated with a thorium fuelled HTGR and a uranium fuelled HTGR these are also discussed.

The structure of the report is as follows. In Chapter 2 the general methodology for assessing the radiological risks associated with nuclear installations is described. An overview is given of the measures commonly used to quantify these risks. In Chapter 3 an overview is given of the different stages of the reference uranium fuelled LWR cycle and the thorium fuelled HTGR cycle. In Chapter 4 a stage-by-stage analysis is given of the radiological risks associated with the two fuel cycles. Finally, in Chapter 5 an evaluation is made of the radiological risks associated with the LWR and HTGR cycles and with thorium and uranium fuels.

2. Radiological Risk Assessment

As a result of the routine operation of the facilities at all stages of nuclear fuel cycles radioactive material will be released to the environment. Additionally, radioactive material may be released as a result of accidental situations. These releases can lead to increased exposures to ionising radiation for members of the general public. The individual levels of exposure and the total number of individuals exposed as a result of the releases from all stages of a nuclear fuel cycle determine the overall radiological risk associated with that fuel cycle. In this chapter the important aspects of the standard methodology used to assess the radiological risk associated with a nuclear fuel cycle are described. In Section 2.1 a description is given of the various types of release, of the pathways by which individual members of the public are exposed and of the models used to assess these exposures. In Section 2.2 an overview is given of the quantities used to express the increased exposure to ionising radiation as radiological risk. In Section 2.3 the approach used in this study is delineated.

2.1 Releases and Exposure Pathways

2.1.1 Normal Releases

As a result of normal operations most installations in the nuclear fuel cycle release radioactive material to the atmosphere and to surface waters. For deep geological repositories for the final disposal of radioactive waste, which are dealt with separately in Section 2.1.3, it is generally assumed that radioactive material is released to the surrounding ground water.

The radioactive material released to the atmosphere is transported downwind, dispersed as a result of normal atmospheric mixing processes and may be deposited on the underlying ground as a result of either "dry" or "wet" deposition processes. Whilst the material is present in the atmosphere individual members of the public may be exposed either internally, as a result of material inhaled, or externally. Once the radioactive material is on the ground individuals may be exposed externally to radiation emitted from the deposited material and internally as a result of the inhalation of resuspended material and of the ingestion of contaminated agricultural produce.

Radioactive materials may be released from nuclear fuel cycle installations either directly or via the sewage system to surface water bodies (i.e. rivers, lakes, estuaries and coastal seas). The material is then transported and dispersed due to general water flows and sedimentation processes. Individuals may be exposed either internally or externally by a number of exposure pathways. The internal pathways include drinking water, the inhalation of material which has become airborne and the ingestion of contaminated foodstuffs. The external pathways are due to the occupancy of contaminated beaches, lake shores or river banks and to the use of sediments for agricultural purposes.

Due to their longevity and mobility in the environment a limited number of radionuclides released either to the atmosphere or to surface water bodies

become globally dispersed in time. These radionuclides are: tritium (half-life circa 12 years); carbon-14 (half-life circa 5700 years); krypton-85 (half-life circa 10 years) and iodine-129 (half-life circa 16 million years). The global dispersion of these radionuclides leads to the long term exposure of the world population ("global dose").

In addition to the exposure pathways originating from the release of radioactive materials to atmosphere or to surface water bodies a limited number of individuals living in the immediate vicinity of an installation may be exposed to "direct shine" from buildings at a nuclear installation.

A range of models have been developed to assess the radiological consequences of routine releases to atmosphere and surface waters. Most models for assessing the local and regional consequences of atmospheric releases are based on the Gaussian plume dispersion model. The processes involved in the transfer of deposited radioactive material through the environment are often modelled using simple empirical models. Location-specific models are generally used for assessing the consequences of liquid releases. For global dispersion of radionuclides simple compartment models are used and the resulting exposure to man calculated using a specific activity approach. In the Netherlands guidelines have recently been developed for the analysis of routine releases of radioactive materials from all installations [1].

2.1.2 Accidental Releases

Accidental releases, leading to the exposure of the general public, could occur at all stages of the nuclear fuel cycle. Because of the magnitude of the consequences of severe accidents at nuclear power plants, in particular the possibility of early deaths amongst the surrounding population, a significant amount of work has been done into assessing the accident situations which could occur. A probabilistic safety analysis (PSA) now forms part of the licensing requirements for nuclear power plants.

A PSA for a nuclear power plant is divided into three levels. A level 1 PSA determines the probability of accidents which lead to core meltdown. A level 2 PSA determines the probability that the reactor containment will fail and the characteristics of the associated release of radioactive material (the source term). Based on the source term a level 3 PSA determines the offsite consequences of the release. For each accident scenario the consequences are analysed for a range of different weather sequences. In modern level 3 PSA computer programs weather sequences are sampled from a hourly meteorological data file in such a way that the full range of possible weather conditions are analysed. The transfer through other environmental compartments (e.g. through the food chain and through the urban environment) is modelled in a similar way to routine releases.

For other stages of the nuclear fuel cycle a full PSA is generally not carried out. It is common practice to define a number of design basis accidents (i.e. deterministic accident scenarios) for which the offsite consequences are analysed. The analysis is performed either for a pre-defined set of weather conditions or taking the variation in weather into account.

A number of internationally available computer codes have been developed

for performing level 3 PSA. In the Netherlands level 3 PSA should be performed in accordance with the recently developed guidelines [2].

2.1.3 Releases from Deep Geological Repositories

High level waste disposal facilities are designed to ensure that the radioactive material is contained for as long as possible. All facility designs are based on a multiple-barrier concept which means that a number of engineered barriers (e.g. the waste container, backfills, seals and dams) and natural barriers (e.g. the host rock and the geosphere) are present between the waste and man's immediate environment. Two types of exposure scenarios are often defined which result in the failure of these barriers and the exposure of the public to radioactive material released from the disposal facility: normal evolution scenarios and altered evolution scenarios (see for example [3]).

Normal evolution scenarios essentially account for the gradual degradation of the waste disposal facility by water and for the expected geological trends. For normal evolution scenarios it is generally assumed that radionuclides will be released from the disposal facility to the environment and the analysis of such scenarios concentrates on determining when this will occur and the consequences of the release. Processes which are not gradual are thought of as probabilistic and are included in altered evolution scenarios. Examples of processes which lead to altered evolution scenarios are seismic events and future human activity. The probabilities of such scenarios are either very low or impossible to estimate [3].

The analysis of the radiological consequences of releases from deep geological repositories is often carried out within an international framework. There are, as yet, no specific guidelines for performing such analyses in the Netherlands.

2.2 Measures of Radiological Risk

2.2.1 Basic Quantities

Exposure to ionising radiation can lead to both deterministic and stochastic health effects in humans. Deterministic health effects result when sufficient cells in a tissue are killed to impair the functioning of that tissue. Deterministic effects only occur above a certain threshold level of radiation exposure. During the normal operation of all facilities in the nuclear fuel cycle all individual exposures are below these threshold values but some accident scenarios could result in individual exposures high enough to lead to deterministic effects. Stochastic health effects can occur when an irradiated cell is modified rather than killed. If the modified cell is not repaired it may develop into a cancer. If the damage is to a germ cell this may lead to effects in the progeny of the individual exposed. It is generally assumed that there is no threshold level of exposure for the occurrence of stochastic effects and that the dose-response function is linear and begins at the origin.

Various dosimetric quantities are used to quantify exposure to radiation. The dosimetric quantities currently recommended by ICRP are macroscopic

quantities based upon the energy absorbed per unit mass (the mean absorbed dose) of tissue or organ [4]. From the absorbed dose the equivalent dose, which is calculated by multiplying the absorbed dose by the relevant radiation weighting factor, and the effective dose, which is formed by multiplying the equivalent dose by the tissue weighting factor and summing over all tissues, can be derived. In radiological risk assessment the absorbed dose is the basic quantity used to quantify deterministic effects. The effective dose - and the associated committed effective dose and the collective (committed) effective dose - are used to quantify stochastic effects.

2.2.2 Dutch Risk Management Policy

The policy document 'Radiation Protection and Risk Management' (ORS) [5] introduced a system of risk management for environmental radiation protection. The primary objective of the Netherlands radiation risk management policy is to *protect* members of the general public against the harmful effects of ionising radiation. Within this system of risk management two quantities are defined for which limits are set; the maximum individual risk and the group risk.

The maximum individual risk is designed to limit the risk of death that any one member of the public incurs as a result of a particular practice and limits are set for both the risk from routine releases and that from accidental releases. For routine releases the individual risk is equal to the effective dose received multiplied by the appropriate risk factor. For accidental releases deterministic effects could also contribute to the individual risk. The maximum individual risk is assessed using the reference group concept [6] and taking account of the *multifunctionaliteit* principle. The multifunctionaliteit principle essentially means that future possible uses of the ground immediately outside of the perimeter fence of the installation should not be constrained by the present value of the maximum individual risk.

The group risk is the probability that a given number of individuals will suffer an acute death as a result of an accident at an installation. The group risk is therefore determined entirely by deterministic effects which occur in the short term following the accident. As the group risk limits the number of early fatalities among the members of the public as a result of an accidental release it is only applicable to nuclear power plants and reprocessing plants (as releases following accidents at other stages of the nuclear fuel cycle are not large enough to result in early deaths).

The Dutch government has set a value of 10^{-6} per year for the maximum risk an individual may incur. The limit for the group risk is set at 10^{-5} per year for an accident which leads to 10 deaths. The maximum acceptable probability of accidents leading to a larger number of deaths decreases quadraticly with a linear increase in victims (e.g. 10^{-7} for 100 victims). These limiting values are for use in the licence application process and are of limited value when comparing the risks associated with different fuel cycles.

2.2.3 Collective Dose

The collective effective dose is defined as the integral of all individual effective doses in the population group of interest and is hence a measure of the total radiation exposure of that population. The collective equivalent dose for each tissue is defined in a similar fashion. Collective doses can be used to estimate the total number of stochastic health effects in the exposed population. Measures of collective dose should not be used to estimate the number of health effects when the individual doses are in the range where deterministic effects can be expected.

The concept of collective dose has been widely used to describe the total radiological impact of practices which involve radiation risks. Measures of collective dose have in particular been used in justification and optimisation assessments. Collective dose is therefore a useful quantity when comparing the radiological risks associated with different nuclear fuel cycles.

2.3 Approach used in this Study

The radiological risk assessment for this study has been performed upon the basis of a literature study. All values given in the literature are based upon the general approach given in Section 2.1. Where possible values have been used that were calculated in accordance with the relevant Dutch guidelines. This was generally the case for those stages of the LWR fuel cycle which take place in the Netherlands. For other stages this is not the case. The data is therefore not always strictly comparable but is sufficiently consistent to give an indication of the absolute and relative magnitudes of the various radiological impacts.

The radiological risks associated with normal operations have been expressed in terms of maximum individual dose (Sv) (as a measure of individual risk) and collective dose (manSv). For severe accidents at the electricity generating stage the consequences are expressed in terms of individual risk, group risk and collective dose. A qualitative discussion of the radiological impact of the accidents at other stages of the nuclear fuel cycle is given. In accordance with the approach used in the literature the values have been normalised with respect to the electrical energy generated. LWRs have a typical thermal efficiency of 33% whereas that for HTGRs is typically around 39% (direct gas driven generators may have a higher thermal efficiency). Given the global nature of this study and the uncertainties involved, normalisation with respect to the thermal energy generated would not alter the conclusions drawn.

3. Nuclear Fuel Cycles

The objective of this study is to perform a comparative assessment of the radiological risks associated with a thorium fuelled HTGR cycle. The basic reference for the comparison is a generic uranium fuelled LWR cycle. In this chapter the technologies used at the various stages of the reference LWR uranium cycle (Section 3.1) and the HTGR thorium cycle (Section 3.2) are described.

3.1 The Reference Uranium Fuelled LWR Cycle

The stages involved in the existing uranium fuelled LWR cycle are shown in Figure 1. After the electricity generating stage of the cycle, the spent fuel elements are either reprocessed or sent directly to the interim storage facility in preparation for final disposal. The dashed arrows represent the possible recycling of reprocessed uranium and plutonium as reactor fuel. Such recycling has already begun on a small scale in some nuclear power states. In this study it is assumed that the spent fuel is reprocessed.

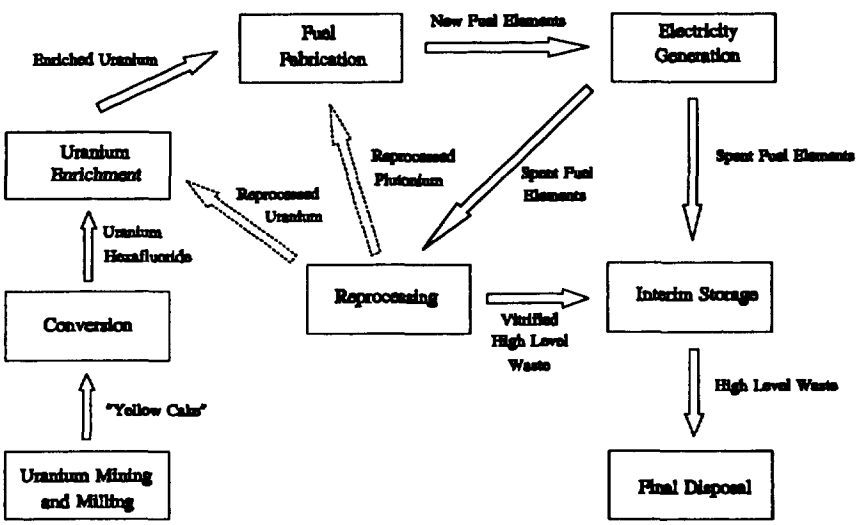

Figure 1 *Flowchart for the generic uranium LWR cycle*

A brief description of these stages is given below. For each stage an overview is given of: the process used; the inputs to and the outputs from that stage; and the important waste streams.

Uranium mining and milling. Uranium mining is defined as the extraction from the ground of ore containing between a tenth of a percent and several percent of uranium and its decay products [7]. Uranium mining is carried out by one of two general methods: open-pit mining - in which the surface layer of soils and rock is removed and the ore is extracted from an open pit - and underground mining. Milling is defined as the processing of the mined ores to extract the uranium in a partially refined form, known as yellow cake. Mining and milling operations result in releases of radioactive materials to the

atmosphere and to nearby surface waters. These releases consist entirely of naturally occurring radionuclides. Releases of radon from stockpiled mill wastes (tailings) form an important additional waste stream from this stage of the nuclear fuel cycle. Due to the long half-lives of the radioisotopes in the uranium decay chain the tailings form a long term source of radon.

Conversion. Conversion is defined as the series of physical and chemical transformations involved in converting uranium from uranium concentrate (from the mills) to the hexafluoride form (UF_6) [7]. The hexafluoride form is required for the enrichment stage. The conversion of the uranium concentrates to the hexafluoride form takes place via uranium trioxide and uranium tetrafluoride. The releases from the conversion stage of the nuclear fuel cycle consist primarily of the long-lived uranium isotopes and the first generation daughter radionuclides.

Uranium Enrichment. The enrichment process serves to enrich the ^{235}U content of the uranium hexafluoride produced by the conversion process from about 0.7% to the 3-4% needed for the fuel of a typical light water reactor. Uranium is enriched by one of two methods - the gas diffusion method or the ultracentrifuge method. In both processes the uranium hexafluoride arrives at the plant in solid form, is enriched in the gaseous form and is then resolidified. Releases to atmosphere and local surface water bodies consist of the long-lived uranium isotopes and the first generation daughter radionuclides. Reprocessed uranium contains trace amounts of fission products and therefore releases from the enrichment of reprocessed uranium will contain traces of fission products.

Fuel Fabrication. The fuel fabrication stage is defined as the production of the reactor fuel in the form of uranium oxide fuel elements. The enriched UF_6 is first converted into uranium oxide and then made into fuel pellets with the required fuel composition. These are then placed as columns in a zirconium alloy fuel rod cladding. Releases to atmosphere and local surface water bodies consist of the long-lived uranium isotopes and the first generation daughter radionuclides.

Electricity generation. The operation of the LWR to generate electricity is the central stage of the LWR fuel cycle. The majority of electricity generating reactors in the world are LWRs (either boiling water reactors or pressurised water reactors) and considerable experience has been gained in the operation of such reactors. Releases to the atmosphere as a result of normal operations consist to a large extent of noble gases. Aerosols, halogens, tritium and carbon-14 are also released. Releases to local surface water bodies consist primarily of fission products, activated corrosion products and the activation products tritium and carbon-14. In addition to the releases from normal operation the possibility exists that significant amounts of radioactive material may be released as a result of severe accident conditions. All high-level radioactive waste produced in the civil nuclear fuel cycle has its origins in the electricity generating stage. This waste consists of spent fuel elements and activated construction materials.

Reprocessing. At the reprocessing stage of the nuclear fuel cycle uranium and plutonium are extracted from the spent fuel elements for possible reuse in nuclear power reactors. Most reprocessing plants now make use of the plutonium uranium recovery by extraction (PUREX) process. The following

operations make up the PUREX process [8]: the head end plant (i.e. where the spent fuel elements are received and the fuel is separated from the fuel cladding in a dissolver); the extraction of uranium and plutonium from other actinides and fission products; the purification and concentration of the uranium and plutonium and their conversion to oxides; and the treatment of the various process waste streams. Radionuclides are released to both atmosphere and to surface water bodies from reprocessing plants. The most important radionuclides released are the long-lived global circulation nuclides, isotopes of caesium and transuranic elements. The possibility of significant releases as a result of accidents at reprocessing plants also exists.

Interim storage. The objective of the interim storage stage of the nuclear fuel cycle is to provide temporary storage facilities for the radioactive waste. The arrangements for the interim storage of radioactive waste differ between the different nuclear power states. In accordance with the Dutch government's radioactive waste management policy all radioactive waste will be stored for the next 50 to 100 years at one interim storage facility. Following interim storage, the remaining waste will be placed in a deep underground repository in the Netherlands. Releases from interim storage facilities vary depending on the facility operations (e.g. whether waste treatment takes place in addition to storage). In the Netherlands the interim storage facility encompasses facilities for the storage of the various categories of radioactive waste and facilities for the treatment of low and intermediate levels of radioactive waste.

Final disposal. The final disposal of high level radioactive waste is the last stage in all nuclear fuel cycles. The objective of final disposal is to isolate the radioactive waste from man's immediate environment for a long period of time and hence to ensure that the radiological risks to man from both natural release scenarios and human intrusion scenarios are suitably low. In most nuclear power states research, planning and preparatory work is being carried out into the disposal of radioactive waste in stable geological formations. In natural release scenarios radionuclides are released from the waste repository to the surrounding ground water as a result of natural processes. The most important radionuclides are those with a relatively high geomobility. Releases as a result of altered evolution (e.g. human intrusion) scenarios have also been postulated for deep geological repositories.

Transport. Since the various stages of the nuclear fuel cycle take place at different locations materials have to be transported between the various fuel cycle facilities. During normal operations the most important public radiological impact is generally via the "direct shine" from the transport containers. Additionally there is the possibility of releases of radioactive materials following transport accidents.

3.2 The Thorium Fuelled HTGR Cycle

Unlike the reference uranium fuelled LWR fuel cycle, the thorium fuelled HTGR cycle is very much still in the design stage. HTGR operating experience is limited to the Peach Bottom and Fort St. Vrain high temperature reactors in the United States of America and to the Atomversuchsreaktor and the Thorium High Temperature Reactor in the Federal Republic of Germany. All these reactors used thorium based fuel and all have now ceased operation.

Recent HTGR design efforts have concentrated on modular HTGR concepts: the American MHTGR concept and the German HTR-M concept. The reactor core of the MHTGR is based on the prismatic fuel block design used in the earlier American HTGRs whilst the HTR-M core is based on the spherical fuel element design used in earlier German designs.

In thorium based fuels the only naturally occurring isotope of thorium (^{232}Th) functions as the fertile material. Since no fissile isotopes of thorium are present in nature, thorium fuels have to contain externally supplied fissile material (known as the 'topping'). Three different fissile materials could be used [9]: uranium enriched in uranium-235; plutonium from reprocessed uranium fuel or from weapons inventories or; uranium-233 from reprocessed thorium fuel. When either uranium-235 or uranium-233 is used as the fissile material then the resulting fuel cycle is termed a thorium/uranium fuel cycle. Thorium/uranium fuels have been proposed with either circa 20% enriched uranium (MEU) or highly enriched uranium (HEU). The original motivation for introducing thorium based nuclear fuel cycles was to improve the use of relatively scarce uranium resources. In the framework of solid waste management, the application of thorium-based fuels has recently gained interest because of the favourable production of plutonium and other transuranic elements with respect to uranium based fuels.

The MHTGR reference design uses thorium oxide as fertile material with MEU carbide and oxide as fissile topping. The HTR-M reference design is based on 7.8% enriched uranium fuel. The four previous HTGRs used mixed oxide fuel. In this study a generic thorium/uranium cycle is assumed. The fissile uranium in the spent fuel is assumed to be recycled and any additional fissile uranium needed is assumed to be freshly mined. The implications of the choice for MEU or HEU are discussed where relevant for the radiological risk analysis.

Figure 2 gives an overview of the processes and the material flows in a thorium/uranium fuelled HTGR nuclear fuel cycle with reprocessing.

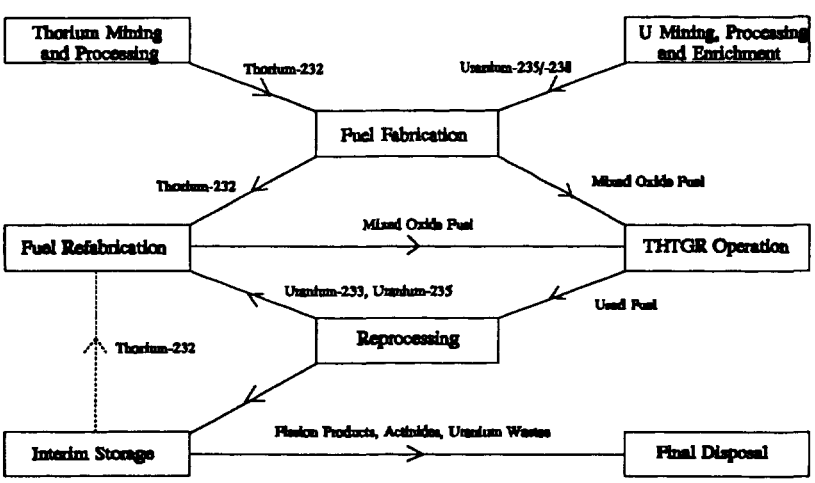

Figure 2 *Flowchart for the generic thorium HTGR cycle*

A brief description of the processes involved at each of the stages shown in the figure is given below. For each stage an overview is given of: the process used; the inputs to and the outputs from that stage and; the important waste streams from that stage.

Thorium mining, milling and processing. The principle thorium ore is monazite which is mined by the following methods [10]: the dredging of beach sands containing monazite; extraction by underground mining activities and; the leaching of thorium from uranium ore extracted in uranium mining activities. To date most thorium has been obtained as a by-product of rare earth metal production using the first method [9]. The waste streams from the thorium mining stage depend upon which method is used. If the monazite is obtained from the dredging of beach sands then there are no bulky mining wastes (as for uranium mining) since the sands are usually returned to the area from which they were obtained [10]. The monazite is then milled (i.e. ground) and then processed by dissolution in acid, selective precipitation and solvent extraction to obtain the rare earth metals, trisodium phosphate and thorium (in either the oxide or the nitrate form). Waste streams from the milling and processing stages consist of long-lived isotopes of uranium (typical monazite contains approximately 0.4% uranium oxide) and radioactive daughters from the three natural decay chains. Airborne releases are generated primarily at the milling stage whereas liquid and solid wastes are generated at the processing stage (e.g. filter wastes).

Uranium mining, milling, conversion and enrichment. The processes involved in the preparation of the ^{235}U fissile "topping" material are identical to the corresponding stages at the front end of the uranium nuclear fuel cycle (see description of reference uranium/LWR cycle given above).

Fuel fabrication. All fuel designs for HTGRs are based on the Triso coated fuel particle [11]. Each Triso particle consists of a fuel kernal coated with pyrolytic carbon, silicon carbide and pyrolytic carbon successively. The Triso particles are then surrounded by a layer of carbonaceous material and combined in the form of a fuel rod (American design) or a fuel sphere (German design). Releases to the atmosphere and to local surface water bodies from the fuel fabrication stage of the HTGR fuel cycle will consist of long-lived isotopes of uranium and the first generation radioactive daughters from the three natural decay chains.

HTGR operation. The operation of the HTGR to generate electricity and/or industrial process heat is the central stage in the fuel cycle. To date only limited experience has been gained in the operation of HTGRs. In the mid sixties the Dragon reactor project was initiated by the OECD. The Dragon reactor was the first HTGR and was a 20 MW(th) test reactor which used block type uranium oxide fuel. In the sixties the Peach Bottom HTGR and the Atomversuchsreaktor (AVR) were built in the United States of America and Germany respectively. With both reactors experience was gained of using mixed thorium/uranium fuel. In the United States the Fort St. Vrain Reactor (FSVR) was built as the successor to the Peach Bottom HTGR and in Germany the Thorium High Temperature Reactor (THTR) was built as successor to the AVR and both also were fuelled using mixed thorium/uranium fuels. Both reactors had a very limited operating life: the FSVR operated from 1978 to 1986 whilst the THTR operated from 1986 to 1989. Since the accidents at Three Mile Island and, in particular, Chernobyl there has been an emphasis

on the development of nuclear reactors with a high degree of passive safety. HTGR concepts have received significant attention because the integrity of the Triso particles is maintained up to high temperature (i.e. essentially all radiologically significant fission products remain within the particle). This implies that there will be low releases of radioactive materials during normal operations and it is maintained that core meltdown and the subsequent release of a significant fraction of core inventory cannot occur during accident conditions. During normal operations fission products will be released from "intact" particles (e.g. due to traces of thorium and uranium outside the silicon carbide layer) and from the very small fraction of particles with defects in the silicon carbide layer. During accident conditions additional defects and coating failure would occur should very high temperatures (i.e. above 1700 °C) be reached [12].

Reprocessing. At the reprocessing stage of a thorium/uranium HTGR cycle the thorium and uranium (and possibly plutonium) are extracted from the spent fuel for reuse in HTGRs. The THOREX process has been developed for the reprocessing of spent thorium fuels. The THOREX process follows the same basic steps as the PUREX process (for the reprocessing of spent uranium fuels) but is less advanced due to the limited use of thorium fuels. To date the emphasis in HTGR development has been on once through cycles and the reprocessing of spent thorium HTGR fuel has not yet been tested even on the pilot plant scale [9]. The following operations make up the THOREX process [9]: the head end plant (in the case of spent HTGR fuel this consists of the crushing and burning of the TRISO particles); the extraction of the thorium, uranium and possibly plutonium from the minor actinides and fission products by a solvent extraction process; the separation of the thorium and uranium streams and their subsequent purification by ion exchange; and the treatment of the various process streams. Significant amounts of gaseous activation (tritium and ^{14}C) and fission (isotopes of iodine and the noble gases) products will be released from the spent fuel at the head end plant stage of reprocessing. These can be treated with varying degrees of efficiency before the residuals are released to the atmosphere. The treatment of the fission product and minor actinide waste streams (i.e. concentration and vitrification) is similar to that used for uranium fuels [9].

Fuel refabrication. In the fuel refabrication stage of the fuel cycle new HTGR fuel elements are fabricated from the ^{232}Th, ^{233}U and ^{235}U from the reprocessing plant. Due to the intense γ-emitting daughter nuclides in the ^{232}Th / ^{233}U decay chain the fabrication of fuel from reprocessed materials must be done in hot-cell facilities. The reprocessed thorium and uranium will contain trace amounts of fission products. Therefore, in addition to long-lived uranium isotopes and the radioactive daughters from the three natural decay chains, releases will also contain small amounts of fission products.

Interim storage. The vitrified high level wastes from the reprocessing stage of the HTGR cycle would need to be kept at an interim storage facility until its decay heat had fallen below a given value and/or the final storage facility had been completed. The interim storage of vitrified wastes from the HTGR cycle would be similar to those from the reference uranium LWR cycle.

Final Disposal. The final disposal of high level radioactive wastes from the HTGR cycle will take place similarly to the disposal of high level wastes in the reference uranium LWR cycle.

Transport. The technologies and exposure pathways for the transport of materials in the thorium fuelled HTGR cycle are similar to those for the uranium fuelled LWR cycle (see Section 3.1).

4. Risk Comparison for Fuel Cycle Stages

In this chapter the results of the risk analysis for the different stages of the nuclear fuel cycle are given. Those stages for which the radiological risk (measured in terms of the radiological risk quantities given in Section 2.2) may make a significant contribution to the radiological risk from the fuel cycle as a whole are covered in detail. These stages are mining and milling, electricity generation, reprocessing and solid waste disposal. The risks associated with other stages are discussed briefly.

4.1 Mining and Milling

4.1.1 Risks associated with Uranium Mining and Milling

The releases to the atmosphere from uranium mining and milling operations consist of dusts and gases containing naturally occurring radionuclides. The atmospheric releases from open cast mines are very difficult to monitor whereas releases from underground mines and mills can in principle be monitored. In the 1993 UNSCEAR report [7] normalised releases are defined for a reference mine and mill site. These normalised releases were defined using release data from a number of existing facilities. It should be noted that considerable variation exists in the release data for the different mine and mill facilities around the world. The reference mine and mill was defined to have a population density of 3 people/km^2 out to 100 km and 25 people/km^2 from 100-2000 km. The collective effective dose to this population during the operation of the mine and mill was estimated to be 1.5 manSv/GW$_e$a. In [7] no estimates of individual doses are made. In [13] the individual doses for members of the public living near to a French mine are estimated: for example for individuals living 500m from the site the dose is estimated as 5 10^{-7} mSv/year at 500m from the site (i.e. 1 10^{-7} mSv/GW$_e$a). Using the approach delineated in [8] the maximum individual dose as defined in the Dutch risk management policy is estimated to be 8.8 10^{-5} Sv/GW$_e$a.

Releases from opencast mines to nearby surface water bodies are primarily in the form of liquid run-off from site operations. In underground mines water is pumped from underground and released. The liquid releases from the mill plant arise from the dissolving, filtering and drying operations needed to produce uranium oxide. In [7] the radiological impact associated with the liquid releases from the normal operation of the mine and mill facilities is not evaluated. In [13] this impact is shown to be negligible in comparison with the impact of the atmospheric releases.

Releases from the stockpiled mill wastes (mill tailings) form an additional public radiological impact from this stage in the nuclear fuel cycle. These mill tailings contain the decay products of ^{234}U and hence form a long term source of atmospheric radon (the radon release rate will remain essentially the same for the next 10 000 years and will only decrease by a factor of 2 in the next 100 000 years [7]). The magnitude of this impact will depend largely upon how the mill tailings are treated. In [7] it is assumed that a reasonably impermeable cover is placed over the tailings and this essentially remains intact. This leads to an estimate of the collective effective dose of

1.5 10^{-2} manSv/GW_ea per year of release. Other scenarios considered in [7] lead to collective doses in the range 9.6 10^{-5} to 9.6 10^{-2} manSv/GW_ea. In [13] it is assumed that the mill tailings are replaced in the mine and that this is sealed at the end of its working life. It is then assumed that there is no additional radiological impact as a result of the exposure pathway. Clearly such an approach is only possible for underground mines facilities.

Accident situations during the operational phase of a uranium mine and mill would not lead to a significant radiological impact on the general public. There have been examples of accidental releases resulting from the failure of mill tailing retention systems but to date these releases have not lead to significant radiological impacts [14].

4.1.2 Risks associated with Thorium Mining and Milling

Little information is available in the literature on the radiological risks associated with thorium milling and mining. However, it can be stated that the major difference, with respect to uranium mining and milling, is the absence of large quantities of tailings. In the case of uranium ore the majority of the ore material becomes waste in the form of tailings. Thorium ore (monazite) has a typical composition of 9.5% ThO_2, 60% rare earth oxides, 28% P_2O_5 and 0.3% U_3O_8 [10]. Most of the composite parts of monazite are extracted and used. The volume of radioactive wastes from the thorium milling stage is therefore relatively small and this waste can be treated as low-level radioactive waste.

4.1.3 Reduction in Uranium Requirements for Thorium Cycle

One of the reasons for introducing thorium based nuclear fuel cycles is to improve the utilisation of uranium resources. A significant amount of work was carried out on this subject in the 1960s and 1970s when a large expansion of the world's nuclear power base was anticipated and the efficient use of uranium resources was a real issue. In the context of this analysis the potential of thorium fuelled HTGRs to reduce the uranium ore requirements (with respect to uranium fuelled reactors) is of interest because of the radiological risks associated with uranium mining.

The uranium ore requirements of a number of fuel cycles were recently reviewed in [9]. Uranium ore requirements depend upon a number of factors including: reactor type; fuel type; fuel burnup; and whether or not fissile material is recycled. Representative values for LWRs and HTRs given in [9] are reproduced in Table 1.

As Table 1 shows, HTGRs with uranium recycling require a factor of circa 2 less uranium ore for a regular fuel reload of the reactor than a typical LWR with uranium and plutonium recycling. This reduction implies a corresponding reduction in the maximum individual dose per GW_ea and in the collective doses to present and future generations.

Thorium fuel cycles have been proposed for HTGRs with lower uranium ore requirements (near breeder cycles). However, at present such cycles require a

Table 1 *Uranium ore requirements*

Reactor Type	Fuel	Recycling	U-ore req^t (Mg/GWe.a)[1]
PWR	3.2% enriched U	No	200
PWR	3.2% enriched U	U and Pu recycle	130
HTR-M	8% enriched U	No	170
HTR-M	Thorium; HEU	U recycle	75
Large HTR	Thorium; MEU(20%)	U recycle	50

Note:
(1) Data taken from [9]. Data do not include the initial reactor inventory.

large reactor core and are therefore not possible in modular HTGRs [17]. The choice for a large HTGR implies that the inherent safety features of the modular design would be compromised.

4.2 Conversion

The emissions from the normal operation of the processes in the uranium conversion stage are generally relatively small. These releases consist primarily of the long-lived uranium isotopes (^{234}U, ^{235}U and ^{238}U) and the radioisotopes ^{234}Th and ^{234m}Pa [7]. In [13] the total collective dose to the general public from the reference facilities is estimated to be 3 10^{-4} manSv/GW$_e$a. The majority of this is a result of atmospheric releases. Using the approach delineated in [8] the maximum individual dose as defined in the Dutch risk management policy is estimated to be 2.1 10^{-7} Sv/GW$_e$a.

As for the uranium conversion stage releases from the processing (i.e. conversion to the oxide or nitrate form) of thorium ore (monazite) can be expected to be small.

No consideration is given in the literature to the possible radiological impact on the public as a result of accident situations at conversion facilities. The most serious accident which could take place at a uranium conversion facility would be categorised as 4 on the International Nuclear Event Scale (INES) [8]. The radiological impact of such accidents is not expected to be a major impact for the fuel cycle as a whole.

4.3 Uranium Enrichment

As for the conversion stage, releases from this stage of the uranium nuclear fuel cycle are relatively small. For a once through uranium cycle the collective dose to the general public out to a distance of 1000 km associated with the releases from the Urenco plant in the Netherlands has been assessed to be 7.1 10^{-7} manSv/GW$_e$a [15]. The maximum individual dose at the site perimeter was estimated to be 1.2 10^{-7} Sv/a for a plant capacity of with 2500 tSW/a) [16]. This corresponds to a maximum individual dose of 6.3 10^{-9} Sv/GW$_e$a for a once through LWR cycle. Reprocessed uranium will contain trace amounts of other actinides and of fission products. Releases of these radionuclides will lead to an additional radiological impact.

4.4 Fuel (Re)fabrication

The emissions from the normal operation of the processes in the fuel fabrication stage are generally relatively small. These releases consist primarily of the long-lived uranium isotopes (^{234}U, ^{235}U and ^{238}U) and the radioisotopes ^{234}Th and ^{234m}Pa [7]. In [13] the total collective dose to the general public from the reference facility is estimated to be 8.1 10^{-5} manSv/GW_ea. The majority of this is due to liquid releases. The maximum individual dose to individuals living near fuel fabrication facilities is also small. As for the reference uranium LWR cycle releases from this stage of a thorium fuelled HTGR fuel cycle should be relatively small.

Fuel refabrication (e.g. using reprocessed uranium in the thorium/uranium cycle and using reprocessed plutonium in the form of MOX fuel in the uranium cycle) will lead to releases of trace amounts of fission products and actinides present in the reprocessed fuel material. Whilst such releases will have an impact on the radiological consequences associated with this stage, the impact is not expected to be significant when considering the fuel cycle as a whole.

Little consideration is given in the literature to the possible radiological impact on the public as a result of accident situations at fuel fabrication facilities. In [8] it is estimated that the radiological impacts associated with accidents at fuel fabrication facilities are comparable to those associated with accidents at uranium enrichment facilities. These impacts are not significant when compared with severe accidents at the electricity generating stage.

4.5 Energy Generation

4.5.1 Normal Operations

The most important factors which determine the level of the routine releases from a nuclear reactor are: the production of radioactive material in the fuel elements; the release of this material from the fuel to the reactor coolant circuit; the production of radioactive material in the reactor coolant circuit (activation products); and efficiency of the radioactive waste treatment systems.

The production of radioactive material in the fuel depends primarily on the composition of the fuel, the neutron spectrum and fuel burnup. Both LWRs and HTGRs are thermal reactors and in [17] it is shown that the fission product yields for ^{233}U and ^{235}U are similar. For similar fuel burnups therefore, this factor is not expected to lead to essential differences in the release data. Likewise, it is reasonable to assume that the liquid and gaseous radioactive waste treatment systems (i.e. the use of filters) in an HTGR will be similar to those in an LWR. However, significant differences can be expected between the two reactor types in the release of radioactive material from the fuel and in the production of activation products. These factors are discussed below for LWRs and HTGRs respectively. Representative release data are given for each reactor type. Based on this release data an evaluation of the radiological risks is made.

LWR

In LWRs there are two principal mechanisms which lead to the release of fission products from the fuel elements to the reactor coolant: the leakage of gaseous fission products through small defects in the fuel cladding and the presence of trace amounts of uranium ('tramp uranium') on the outside of the cladding material. Additionally, small amounts of fission products will be formed from the trace uranium present in the cladding material.

There are two mechanisms which lead to the production of activation products in the reactor coolant circuit of LWRs: the direct activation of water atoms and of the impurities present in the water; and the activation of corrosion products which are formed from the chemical reactor of the coolant with primary circuit materials. The first mechanism results in the production of, for example, tritium and isotopes of nitrogen. The second mechanism results in the production of isotopes of cobalt, copper, iron, nickel, manganese and zinc.

Release data from LWR plants around the world is given in [7] for the five year period from 1985 to 1989. This data, normalised for electrical power production is reproduced in Table 2 below.

Table 2 *Normalised release data for the world LWR park*

	Radionuclide	Normalised Release ($Bq/GW_e a$)(1)	
		PWR	BWR
Atmospheric Releases	Noble gases	$8.1\ 10^{13}$	$2.9\ 10^{14}$
	Tritium	$2.8\ 10^{12}$	$2.5\ 10^{12}$
	^{14}C	$1.2\ 10^{11}$	$4.5\ 10^{11}$
	^{131}I	$9.3\ 10^{8}$	$1.8\ 10^{9}$
	Aerosols	$2.0\ 10^{9}$	$9.1\ 10^{9}$
Liquid Releases	Tritium	$2.5\ 10^{13}$	$7.9\ 10^{11}$
	Others	$4.5\ 10^{10}$	$3.6\ 10^{10}$

Note:
(1) Data from [7].

The release data given in [7] vary considerably for the different reactors and even for reactors of comparable design. For comparison, the monitored release data for the Borssele nuclear power plant are reproduced in Table 3 [18].

A comparison of Table 2 and Table 3 shows that the releases from the Borssele nuclear power plant are lower than those for the average PWR considered in [7]. The atmospheric releases of noble gases, tritium and ^{14}C are up to an order of magnitude lower whereas the releases of halogens and aerosols are significantly lower. The liquid releases are generally comparable.

Radiological risks associated with the normal releases from the Borssele plant are given in Table 4. The data in this table is based on the results given in [19].

Table 3 *Monitored normal release data for the Borssele PWR*

	Radionuclide	Release (Bq/a)[1]	Normalised Release ($Bq/GW_e a$)[2]
Atmospheric Releases	Noble Gases	$8\ 10^{12}$	$2.2\ 10^{13}$
	Aerosols	$< 10^{7}$	$< 2.7\ 10^{7}$
	^{131}I	10^{7}	$2.7\ 10^{7}$
	Other Halogens	$< 10^{7}$	$< 2.7\ 10^{7}$
	Tritium	$4\ 10^{11}$	$1.1\ 10^{12}$
	^{14}C	$8\ 10^{9}$	$2.2\ 10^{10}$
Liquid Releases	Tritium	$5\ 10^{12}$	$1.3\ 10^{13}$
	β and γ activity	$6\ 10^{9}$	$1.6\ 10^{10}$

Notes:
(1) Average releases for the period 1982-1991. Data taken from [18].
(2) Based on Borssele electical capacity of 450 Mw and an assumed plant availability of 80%.

Table 4 *Radiological risks from the normal operation of the Borssele plant*

Risk Quantity	Results[1]	Comment
Maximum Individual Dose	$1.0\ 10^{-7}$ $Sv/GW_e a$	Atmospheric Releases (55% ^{14}C; 34% ^{3}H; 10% rest)
	$3.6\ 10^{-8}$ $Sv/GW_e a$	Liquid Releases (>90% ^{60}Co)
Collective Dose - NL[2]	$5.7\ 10^{-3}$ $manSv/GW_e a$	Atmospheric Releases (50% ^{14}C; 35% ^{3}H; 15% rest)
	$7.0\ 10^{-3}$ $manSv/GW_e a$	Liquid Releases (>90% ^{60}Co)
Collective Dose - Global[3]	1.2 $manSv/GW_e a$	Atmospheric Releases

Notes:
(1) Average values for the period 1980 to 1987. Data from [19].
(2) Collective dose to Dutch population.
(3) Collective dose integrated to 10 000 years to a hypothetical global population of 10 billion.

HTGR

During normal operations the fission product concentrations in the HTGR coolant will depend upon the quality of the fuel particles. The fuel elements for both the MHTGR and the HTR-M are based upon the TRISO coated fuel particle. The mechanisms by which fission products are released from the TRISO particles to the helium coolant are analogous to those for the LWR. Firstly, there will be a small number of defective particles, and secondly the presence of trace quantities of thorium and uranium outside the silicon carbide layer. In the US stringent criteria were developed for the quality of MHTGR fuel and these criteria have been met by fuel produced on a laboratory scale [12]. Additionally high quality fuel was produced in Germany during the 1980s for the THTR. However, the consistent production of high quality TRISO fuel particles on a commercial production line has still to be demonstrated.

The levels of activation products in the reactor coolant will depend upon the purity of the helium used. Impurities such as air and steam result in the presence of activation products in the reactor coolant by the two mechanisms described above. Whilst no differences are to be expected in the production

of activation products between a thorium and uranium fuelled HTGR, this production will be significantly lower than that for an LWR.

For the MHTGR design, expected releases resulting from normal operations have been published [20]. These releases are reproduced in Table 5 below.

Table 5 *Estimated normal releases from MHTGR*

	Radionuclide	Release (Bq/a)(1)	Normalised Release ($Bq/GW_e a$)(2)
Atmospheric Releases	tritium	$3.7\ 10^{11}$	$7.6\ 10^{11}$
	^{41}Ar	$7.4\ 10^{11}$	$1.5\ 10^{12}$
	^{85}Kr	$1.5\ 10^{12}$	$3.2\ 10^{12}$
	^{133}Xe	$3.7\ 10^{11}$	$7.6\ 10^{11}$
Liquid Releases	^{131}I	$1.8\ 10^{5}$	$3.7\ 10^{5}$
	^{137}Cs	$8.1\ 10^{6}$	$1.7\ 10^{7}$
	^{140}Ba	$1.1\ 10^{4}$	$2.0\ 10^{4}$

Notes:
(1) Data taken from reference [20].
(2) Based on MHTGR electrical capacity of 540 MW and an assumed plant availability of 90%.

The atmospheric releases of tritium and noble gases for the MHTGR are lower than but comparable to those for the Borssele LWR. This implies that the maximum individual dose (risk) and collective dose associated with atmospheric releases from the HTGR will be of the same order of magnitude as for the LWR. One important point should be noted: no consideration has yet been given to releases of ^{14}C. The liquid releases of fission products from the MHTGR are anticipated to be circa two orders of magnitude lower than for LWR. The radiological risks associated with liquid releases from the LWR are dominated by releases of activated corrosion products. Therefore the radiological risks associated with the liquid releases from the HTGR can be expected to be much lower than for the LWR. The long term collective dose to global population cannot yet be assessed as carbon-14 releases from the HTGR have not yet been considered.

4.5.2 Accidental Situations

Severe accidents at nuclear power plants may be initiated from either internal or external events. Examples of internal events are equipment failure, human error and fires. External events may be natural or man-induced and include seismic events, floods and aircraft accidents. Different reactor designs incorporate different approaches to minimise the probability of occurrence of such events and to mitigate the effects of such an event should it occur. The differing safety approaches used in modern LWR and HTGR designs are outlined in this paragraph. Based on existing Safety Analysis reports for the two generic reactor types an overview of the radiological risks associated with accidental situations is given.

LWR
The safety philosophy behind modern LWR design is to a large extent based on the defense-in-depth concept. The defense-in-depth concept basically

provides for several layers of protection against accident progression so that should the first layer of protection fail, the additional layers will generally provide the protection required. An example of the application of the defense-in-depth principle to LWR design is the provision of several systems for the removal of reactor core heat. Many of such safety systems in LWRs are active systems - that is, they require external input for their operation. Another example of the defense-in-depth principle is the multiple barrier concept for the containment of radioactivity. These barriers include the fuel cladding, the primary cooling circuit boundary and the reactor building containment.

Since the late 1970s PSAs have been performed for numerous LWRs. Recently up-to-date analyses have been performed for the two Dutch LWRs: Dodewaard [21] and Borssele [18]. The radiological risk results from these studies are summarised in Table 6.

Table 6 *Summarised level 3 PSA results for existing Dutch LWRs*

Risk Quantity	Results(1)	Comment
Maximum Individual Risk	$4\ 10^{-6}$	Dodewaard (risk at 200m)
	$5\ 10^{-7}$	Borssele (risk at 350m)
Group Risk	Max no of fatalities = 1; Probability $5\ 10^{-8}$	Dodewaard
	10: $5\ 10^{-8}$, 100: 10^{-8}, 1000: $5\ 10^{-10}$ (number of fatalities:probability)	Borssele
Collective Dose	1000: $3\ 10^{-5}$, 10 000: 10^{-5}, 100 000: 10^{-6} (collective dose in manSv: probability)	Dodewaard (to 3000 km)

Note:
(1) Rounded values. Data taken from references [21] and [18].

The radiological risk results from PSA are always plant-specific since such results depend strongly on a number of plant design factors (such as containment behaviour) and plant location factors (such as the population distribution). However, the results in Table 6 give a general impression of the radiological risks resulting from severe accidents at LWRs

HTGR

Modern HTGR designs rely to a great extent on the fuel coatings to prevent any radionuclide release to the environment during accidental situations. These designs do not have an LWR-type reactor containment building to act as a final barrier for the release of radioactivity to the environment. It is therefore essential that the integrity of the TRISO coating is maintained at higher temperatures. Extensive testing has demonstrated that the coatings essentially remain intact up to temperatures of 1600 °C. Increasing coating failure and associated release of radioactive material will occur as the temperature rises from 1700 to 2200 °C.

A preliminary analysis of a range of design basis and beyond design basis accident scenarios has been carried out for the MHTGR design in the framework of the US DOE's advanced reactor programme [22]. This analysis showed that, for the scenarios considered, the maximum fuel temperatures reached were in the range 1330 to 1630 °C. The releases associated with these scenarios comprise essentially of radionuclide material "plated-out" in

the reactor coolant system and little additional particle failure was anticipated [12]. The maximum release estimated was associated with the emergency planning basis event 2 (EPBE-2) scenario. The EPBE-2 scenario was defined as "moisture inleakage with delayed steam generator isolation and without forced cooling" [22]. The cumulative releases to the environment associated with the EPBE-2 scenario are reproduced in Table 7. For comparison typical maximum releases to the environment for LWRs are $1\ 10^{18}$ Bq for noble gases and $1\ 10^{17}$ Bq for the fission products ^{131}I and ^{137}Cs. The releases of radioactive materials associated with the EPBE-2 scenario will not lead to any offsite deterministic health effects.

Table 7 *Maximum postulated releases to the environment for the MHTGR*

	Total Release to the Environment (Bq)			
	^{88}Kr	^{90}Sr	^{131}I	^{137}Cs
EPBE-2	$9.5\ 10^{10}$	$3.1\ 10^{9}$	$1.24\ 10^{11}$	$6.0\ 10^{9}$

This preliminary analysis has been reviewed by the US NRC [23] and by a working group of nuclear utilities [12]. These reviews generally agree with the conclusions of the preliminary analysis but emphasise that design uncertainties and lack of operating experience imply that the results should be treated with caution. The nuclear utility review further points out that the preliminary probabilistic safety analysis only includes events assumed to occur during normal operation and that a full PSA should include other operating modes such as start up and fuel reloading [12].

Preliminary risk analyses have also been performed for the HTR-M [24]. The results of these analyses are in general agreement with those for the MHTGR. The maximum offsite doses associated with the different accident scenarios considered are below the intervention levels for offsite countermeasures.

Clearly additional work is needed before a detailed quantitative risk assessment can be performed for the HTGR. However, a number of qualitative statements can be made on the basis of the existing information. The maximum offsite doses associated with the accident scenarios analysed to date are below the threshold values for deterministic health effects. This implies that there is no group risk, as defined in the Dutch risk management policy, associated with the HTGR. Additionally, it is clear that the maximum collective doses associated with HTGR accident scenarios are significantly lower (i.e. several orders of magnitude) than for the LWR severe accidents. The maximum individual risk and the CCDF for the collective dose will depend upon the probabilities of the various HTGR accident scenarios. Given the preliminary accident frequency ranges for these scenarios (e.g. in [20]) it can be stated that the maximum individual risk associated with accidental releases will be significantly lower than for LWRs.

4.6 Reprocessing

A wide range of radionuclides are released to atmosphere and to surface waters from PUREX reprocessing plants. Activation products and fission products determine the impact from atmospheric releases whilst transuranic

elements also play a role by liquid releases. In [13] an assessment is made of the collective doses to the local, regional and global populations as a result of releases from the UP3 plant in France. The results are summarised in Table 8.

Table 8 *Radiological impact of normal releases from reprocessing plant*

Risk Quantity	Results(1)	Comment
Collective dose - local pop^n	1.83 10^{-3} manSv/GW_ea	Due primaraly to Atm. rel. of activation and fission products.
Collective dose - regional pop^n	5.32 10^{-1} manSv/GW_ea	Due primaraly to liquid releases.
Collective dose - global pop^n	8.9 10^{1} manSv/GW_ea	Due primaraly to long term exposure to ^{14}C

Note:
(1) All data taken from [13].

The collective doses to the local and regional populations given in Table 8 are significantly lower that those reported in [7]. The reason for this is that the values in [7] are based on aggregated releases from all existing reprocessing plants including older facilities and facilities for reprocessing different types of fuel. According to [7] the maximum individual doses to individuals living near to reprocessing plants were of the order of 0.05 to 0.1 mSv/GW_ea in the late 1980s. The values are based upon existing facilities in France and England. The maximum individual doses for individuals living near to new PUREX facilities are expected to be significantly lower.

To date there has been little published on the releases of radioactive materials from either the reprocessing of HTGR fuels or the THOREX reprocessing process. Atmospheric releases generally have their origin in the head-end plant (i.e. the crushing and burning of the TRISO particles). The gaseous activation and fission products released from the head end plant can be treated in the same way (i.e. with the same degree of efficiency) as the releases from the head end plant of the PUREX plant. Differences in releases are therefore due to the differences in the incoming radionuclide inventory. The fission product inventories for thorium fuels will be similar to those for uranium fuels. There could be significant differences between the ^{14}C inventory in LWR and HTGR fuels. The principal production route for oxide LWR fuel is from ^{16}O. For oxide HTGR fuel the production from ^{12}C could play a significant role due to the large quantities of carbon in the fuel elements. This issue needs to be investigated further. For liquid releases lower quantities of higher actinides in thorium fuels could have an impact on the radiological impact. However, as for atmospheric releases the most important radionuclides are activation and fission products.

Accidents scenarios with a large public radiological impact (i.e. similar to the impact of severe accidents at LWRs) have been postulated for uranium fuel reprocessing plants [25]. It is reasonable to assume that similar accidents could occur at thorium fuel reprocessing plants. The radiological impact of accidents at reprocessing plants is generally dominated by fission product releases. The impact of accidents at both uranium and thorium fuel reprocessing plants can therefore be expected to be similar. The probabilities of accidents at reprocessing plants are estimated to be several orders of

magnitude lower than the probabilities of severe accidents at LWRs [25]. For the LWR cycle the radiological impact of accidents at reprocessing facilities are significantly less important than the impact of accidents at the electricity generating stage. For HTGR cycles the impact of accidents at the reprocessing stage are likely to be the most important accidental impact in the cycle.

4.7 Interim Storage

As stated in Chapter 3, the radiological impacts associated with the interim storage stage of the nuclear fuel cycle depend upon the facility design and functions. Radiological impact data related to the treatment and storage of LWR operating wastes at the COVRA interim storage facility is given in [26] and [15]. This data is summarised in Table 9 below. Additionally, there will be offsite radiological impacts associated with the wastes from the eventual decommissioning of the nuclear reactors. The atmospheric and liquid releases originate in the treatment of low and intermediate level radioactive wastes. The direct shine impact is due to the storage of low, intermediate and high level wastes.

Table 9 *Radiological impacts associated with interim storage in LWR cycle*

Risk Quantity	Results	Comment
Maximum Individual Dose[1]	5.5 10^{-7} Sv/yr (1.5 10^{-7} Sv/GW_ea)	Value at 500m from site. From direct shine pathway.
Collective Dose - NL[2]	5.9 10^{-5} manSv/GW_ea	Atmospheric releases (Imp nuclides: ^{60}Co & ^{137}Cs)
	2.7 10^{-5} manSv/GW_ea	Liquid releases (Imp nuclide: ^{60}Co)
	9.0 10^{-6} manSv/GW_ea	Direct shine

Notes:
(1) Based on data given in [26]. The impact of other pathways is negligible.
(2) Collective dose to Dutch population. Based on data given in [15].

The lower release values (two orders of magnitude) for non-gaseous fission products (e.g. ^{137}Cs) from the HTGR (see Section 4.5) imply that the quantities of these radionuclides in the HTGR operating wastes should be lower than for the LWR. The production of corrosion products (in particular ^{60}Co) is also much lower in the HTGR. In can be concluded that the atmospheric and liquid releases, and the associated radiological impacts, from the treatment of low and intermediate level operating wastes should be significantly lower for the HTGR than for the LWR. It is difficult to estimate the impact of the storage of HTGR wastes on the direct shine pathway. It may be reasonable to assume that the degree of shielding provided will depend on the gamma activity of the waste. If this is the case then the radiological impact via this pathway for the two cycles will be comparable.

The radiological impact from the interim storage stage of both fuel cycles is not a major impact when considering the cycle as a whole. The impact from the HTGR cycle is expected to be lower than that for the LWR cycle.

4.8 Final Disposal

A substantial amount of research has been carried out worldwide into the radiological risk associated with various final disposal options for radioactive waste. It is out of the scope of this study to perform a detailed quantitative or even qualitative review of this work and apply the results to the different fuel cycle options under consideration. Instead, the approach chosen for this study is to review the important similarities and differences in the wastes from the different fuel cycles with respect to a number of important factors which play a role in determining the risks associated with solid waste disposal.

The risks arising from the disposal of high level radioactive waste in deep geological repositories are generally categorised as being the result of either normal evolution scenarios or altered evolution scenarios (see Paragraph 2.1.3). For normal evolution scenarios the most important factors with respect to the radiological impact associated with the disposal of the waste are: the physical and chemical form of the waste; the repository design (including the choice of host rock); and the inventory of radioactive materials in the waste. For intrusion scenarios the most important factor is the probability of occurrence of the event. These probabilities are generally very low or impossible to quantify. The discussion below concentrates on normal evolution scenarios.

The vitrification of the high level waste from the THOREX process may be carried out in a similar fashion to that for the PUREX process wastes [9]. One major difference is that the vitrified waste volume is significantly (70%) larger for the THOREX process [9]. It is difficult to say what the effect of this increased volume will be upon the radiological impact. It is therefore reasonable to assume that the wastes from both uranium and thorium fuel cycles could be disposed of in repositories based upon the same design and located in the same host rock. These factors are therefore not likely to lead to significant differences in the radiological impact associated with the disposal of the wastes from the uranium and thorium fuel cycles.

The other major factor is the radionuclide inventory of the wastes. For normal evolution scenarios the collective doses and maximum individual doses in the period up to 1 million years are predominantly due to fission products: the most important being ^{99}Tc, ^{135}Cs and ^{79}Se. The fission product yields for isotopes ^{233}U, ^{235}U and ^{239}Pu are broadly similar (e.g. the procent yields for the fission product ^{99}Tc are 4.93, 6.21 and 6.20 respectively). This implies that the mass of each fission product in the vitrified waste per $GW_{th}a$ will not be significantly different for the uranium and thorium fuel cycles.

In the period beyond 1 million years the contribution of the actinides to the radiological risk from normal evolution scenarios becomes important. The radionuclide which makes the dominant contribution to collective dose and which leads to the maximum individual dose is ^{237}Np. There are four paths which lead to the production of ^{237}Np: from ^{235}U via neutron capture and β decay; from ^{238}U via neutron capture and β decay; via ^{241}Pu via β and α decay; and from ^{241}Am via α decay. In once through reactor systems the production of ^{237}Np depends largely on the quantity of ^{235}U in the reactor whereas the production of ^{241}Pu and ^{241}Am depends largely upon the quantity of ^{238}U and the burnup of the fuel. In [27] the net production rates

of actinides have been evaluated for a number of thorium based fuel cycles in heavy water reactors. It should be noted that the values for transuranic elements for HTGRs may differ significantly from those given in the table due to differences in fuel burnup.

Table 10 *Net actinide production rates*

Nuclide	Net Production Rate ($g/GW_e a$)[1]		
	Th-HEU[2]	Th-MEU[3]	Borssele[4]
^{237}Np	$1.3\ 10^4$	$1.6\ 10^4$	$1.4\ 10^4$
^{241}Pu	$7.1\ 10^2$	$4.0\ 10^3$	$5.5\ 10^4$
^{241}Am	$2.3\ 10^1$	$1.4\ 10^2$	$1.9\ 10^3$

Notes:
(1) All data taken from [27].
(2) Th-HEU in heavy water reactor with uranium recycling.
(3) Th-MEU in heavy water reactor with uranium recycling.
(4) PWR once through cycle.

The data for the relevant radionuclides and for a number of fuel cycles are summarised in Table 10 above. In order to be able to compare the data for the uranium fuel cycle with the thorium fuel cycle the impact of recycling the plutonium produced in the uranium cycle has to be considered. This would lead to a significant reduction in the production of ^{237}Np (due to less ^{235}U in the core) and to an increase in the production of ^{241}Am (due to the presence of the recycled plutonium in the core). It should be noted that ^{241}Am could make an important contribution to the risks associated with human intrusion scenarios [28].

Based on the data available it seems reasonable to make the following conclusions with respect to the relative radiological risks from normal evolution scenarios associated with the final disposal of the wastes from the two fuel cycles:

- The radiological risks in the period up to 1 million years associated with normal evolution scenarios will be dominated by the exposure to fission products. The fission product content in the vitrified waste will not differ significantly for the uranium and thorium fuel cycles.

- Actinides, in particular ^{237}Np, will make an important contribution to the radiological risks beyond 1 million years. It seems unlikely that the thorium cycle will lead to a large reduction in the ^{237}Np content (w.r.t. the uranium cycle).

Further analysis of these issues would require a more detailed definition of the fuel cycle parameters.

4.9 Transport

Members of the public may be exposed to increased levels of ionising radiation as a result of the routine transport of radioactive materials in a nuclear fuel cycle. This impact is essentially due to exposure via the external irradiation pathway and is incurred by those individuals living near to or using (e.g. other road users) the transport route. The public may also be exposed as a result of transport accident via internal (i.e. inhalation and ingestion) pathways.

There is little information available on the radiological risks to the public associated with integral nuclear fuel cycles. Data given in [13] indicates that the collective radiological impact to the local and global populations could be a significant impact when considering the fuel cycle as a whole. An assessment of the radiological risks associated with the thorium and uranium fuel cycles would require a detailed analysis of the material flows for both cycles. Such an analysis is out of the scope of this study. Analysis of the risks associated with the transport of fuel cycle materials is an area which needs further research.

5. Preliminary Conclusions

In this report a comparative analysis has been made of the radiological risks to the general public associated with a generic uranium fuelled LWR cycle and with a generic thorium fuelled HTGR cycle. This analysis was performed using data available in the open literature. It should be emphasised that to date only very limited work has been carried out on the radiological risks associated with the thorium cycle. Additionally many of the analyses related to the radiological risks associated with the uranium fuel cycle have been carried out using different assumptions with respect to model parameters. These analyses are therefore often difficult to compare. This analysis should therefore be seen as illustrative. Based upon the data and discussion given in Chapter 4 of this report the following preliminary conclusions can be made.

The major benefit with respect to the radiological risks of basing the fuel cycle around modular HTGR technology instead of the LWR technology is the increase in reactor safety. The general modular HTGR design is expected to prevent core meltdown and the associated release of a significant amount of radioactive material to the environment during accident conditions. This implies that there would be no early deaths amongst the surrounding population and therefore that there is no group risk as defined in the Dutch risk management policy. Additionally, the maximum collective doses associated with HTGR accident scenarios are several orders of magnitude lower than for LWR accident scenarios. The application of HTGR technology at the energy generating stage of the fuel cycle implies that the reprocessing stage of the fuel cycle becomes the only stage where accidents large enough to lead to early deaths in the surrounding population are possible. In addition releases of radioactive material from the HTGR during normal operations are expected to be lower than for LWRs.

The major benefit with respect to the radiological risks of thorium based fuel cycles over uranium based fuel cycles is the reduction in uranium mining and milling activity. Radon emissions from mill tailings make a major contribution to the long term collective dose from nuclear fuel cycle activities. A reduction in mining and milling activity leads to a corresponding reduction in this radiological impact. Significant differences between thorium and uranium based fuel cycles at other stages, in particular energy generation, reprocessing and the final disposal of radioactive waste, are not expected. The radiological risks associated with these stages are dominated by the risks from fission products, activated corrosion products and the activation products ^{3}H and ^{14}C. It is important to emphasise that the reduction in the overall radiotoxicity of the actinides in the waste which can be achieved using thorium fuels does not lead to a corresponding reduction in radiological risks. The actinide content of the waste is important when evaluating the radiological impact of human intrusion scenarios and makes a contribution to the very long term risks from normal evolution scenarios. The actinide content of the waste depends on the composition of the uranium topping.

In the framework of this study it was not possible to quantitatively address all the factors which play a role in determining the radiological risk associated with the thorium fuelled HTGR cycle (e.g. the production of ^{14}C in the HTGR fuel elements). In order to make a more detailed assessment of the

radiological risks these issues would need to be addressed and a detailed specification of the fuel cycle would have to be developed.

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