Final Report

NUCLEAR FUEL CYCLE

IMPLEMENTATION IN FRANCE

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SUMMARY

In 1991, the European Community (EC) and the US Department of Energy initiated a joint research project to assess the external costs of fuel cycles used to generate electricity. The intention of this project, called the EC-US External Costs of Fuel Cycles Project (ECFC), was to develop a conceptual approach, consistent methodology and identify future research in the assessment of the externalities.

A second phase of the project continued in Europe (with a new name "ExternE") and expanded to include the implementation of the consistent methodology in various EC countries.

This report presents the final results of the French Implementation for the nuclear fuel cycle.

WORK ACCOMPLISHED FOR THE NUCLEAR FUEL CYCLE

The French nuclear fuel cycle was broken down into 8 separate stages. Reference sites and 1990's technology were chosen to represent the total nuclear fuel cycle, as it exists today. In addition, the transportation of material between the sites was considered. The facilities are assessed for routine operation, except in the cases of electricity generation and transportation, where accidental situations are evaluated. The impacts of construction and decommissioning of a facility are included in the electricity generation stage. It is important to stress that this methodology does not employ a worst case scenario analysis, as is usually done for safety or regulatory compliance assessments, but intends to evaluate the impacts expected from the operations. In a few cases, however, when no reasonable alternative seemed possible, conservative values were used.

The impact pathway approach requires an inventory and assessment of all potential impacts, however, within the context of the ExternE project it has not been possible to consider all of these. Therefore, only the most important impacts, called priority impacts, have been included. Releases of radioactive material to the environment, which potentially impact public health, were given the highest priority. Occupational health impacts, from both radiological and non-radiological causes, were the next priority, even though the

extent to which occupational health impacts can be considered as externalities has not been addressed in this study.

Impact Assessment

Assessment of the impacts was organized by the type of routine emissions: atmospheric releases, liquid releases and solid wastes. The analysis of impacts of releases from severe accidents involves additional complex issues, therefore, it was treated as a distinct category. Health impacts to the workers - radiological and non-radiological - were also accounted for separately.

The most important choices for the assessment of the nuclear fuel cycle concern the definition of temporal and spatial boundaries. Due to the long half-life of some of the radionuclides, low-level doses will exist very far into the future. These low-level doses can add up to larger values when the total population dose is considered for thousands of years. The validity of the use of this type of modeling has been widely discussed. On one hand, there is a need to evaluate all the possible impacts if a complete assessment of the fuel cycle is to be made. On the other hand, the uncertainty of the models increases and the level of doses, that are estimated, fall into the range where there is no clear evidence of resulting radiological health effects.

If large distances and long time frames are included in the assessment of some fuel cycles and not in the assessment of others (due to lack of methodologies or lack of data) the direct comparison of the results becomes a problem. It is for this reason that the impacts estimated for the nuclear fuel cycle are presented in this report within time and space matrices. The short-term category includes immediate impacts, such as occupational injuries and accidents, medium-term includes the time period from 1 to 100 years, and long-term accounts for between 100 to 100,000 years into the future. The selection of a 100,000 year limit to the assessment was arbitrary, however the most significant part of the impacts are included.

The environmental impact from increased levels of natural background radiation due to the routine releases of radionuclides has not been considered as a priority impact pathway. The most important of these types of impacts could be expected to occur as a result of a major accidental release. These have been included in the economic damage estimates as the loss of land-use and agricultural products after a potential severe reactor accident. Possible long-term ecological impacts have not been considered at this time.

The final stage of the impact pathway methodology is the economic valuation of the impacts. The economists involved in the ExternE project set a common value of a statistical life, based on a literature review of contingent valuation studies, adopted for all the fuel cycle assessments in the project. Being that for almost all cases in the nuclear fuel cycle, low-levels of exposure are under consideration, a methodology has been developed for the valuation of radiological impacts. The stochastic nature of the effects and the expected delay time between exposures and manifestation of the health effects must be taken into account. Due to the lack of contingent valuation studies directly applicable for the monetary valuation of the morbidity impact indicators (radiologically-induced non-fatal cancers and occupational injuries), the best available information was used.

Severe Accidents

Accidents are one of the most controversial features of environmental assessment of the nuclear fuel cycle. Within the scope of this project, this type of assessment has been confined to the electricity generation stage and the transportation of radioactive materials between sites. Although facilities at other stages of the nuclear fuel cycle handle very large inventories of radioactive material, their activities are generally believed to be of a lower risk. The probabilistic assessment of the transportation of materials between all the fuel cycle facilities includes risks from both conventional traffic accidents and releases of radioactive material. These have been found to be relatively small.

At this time, there is no general consensus on a methodology to assess the external costs of severe nuclear reactor accidents. In this project, a risk-based approach has been adopted. Due to the complexity of the assessment and the difficulty in finding facility-specific or generally-accepted input data, the evaluation that was completed provides indicative results for this type of methodology. An accident consequence assessment code has been used to estimate the doses, costs of countermeasures and economic losses that would be expected after an accident.

The source term considered in this study corresponds to a release of about 1% of the core (ST21). This source term is in the same order of magnitude as the reference accident scenario used by the French national safety authorities. To illustrate the sensitivity of the results the impacts of three other source terms are presented. The largest can be considered as release that would occur after a core melt accident with a total containment breach. The fraction of the core released, based on a source term used in an international inter-comparison study, is about 10% of the core inventory. The smallest release can be considered to represent the situation after a core melt accident where all the safety measures have operated as planned and there is only leakage from the intact containment (0.01% of the core inventory).

The probability of a core melt accident, based on a French assessment of a major core melt accident at a 1300 MW PWR reactor, is taken to be 1.0E-5 per reactor.year. This is broadly consistent with other similar assessments based on engineering fault tree analysis, although a wide range of estimates have been proposed. The conditional probabilities of the large and small releases that would occur after a core melt accident are taken from a US Nuclear Regulatory Commission report, and are 0.19 for the three largest source terms and 0.81 for the lowest.

RESULTS

Doses

The total collective dose for all the stages of the fuel cycle, except for the severe accident analysis, integrated for a time period of 100,000 years into the future, is 13.1 man.Sv/TWh. A closer look shows that the total local collective dose is about 0.22 man.Sv/TWh and the total regional collective dose is 0.33 man.Sv/TWh, leaving over 95% of the public dose due to the global dispersion of certain radionuclides (C-14, I-129). If the global doses are not included, the occupational doses become a dominant contributor to the overall impacts (about 40% of the doses received).

When all the categories of the doses are considered, the reprocessing stage contributes the largest portion (79%) of the total collective dose (10.3 man.Sv/TWh), followed by the electricity generation stage (18% of the total). If the global collective doses are excluded, the reprocessing stage diminishes in importance and is replaced by the electricity generation (0.38 man.Sv/TWh) and the mining and milling (0.29 man.Sv/TWh) stages. The doses from the enrichment stage are the least important.

On a global scale, C-14 released from the electricity generation and reprocessing stages contributes the largest portion of the dose (more than 12 man.Sv/TWh). It must be stressed that even though this radionuclide is responsible of more than 90% of the total collective dose presented in this report, it is due to the aggregation of very small doses over a large time and space scale (a constant global population of 10 billion people is assumed for 100,000 years).

The average individual dose from the annual atmospheric release of C-14 from the electricity generation and reprocessing stages (8.5E4 MBq per TWh) has been estimated to be 2E-9 mSv/TWh. An individual dose of 1.4E-8 mSv/y is estimated for the operation of one 1300 MW PWR, assuming an electricity production of 7 TWh/y. It can be seen that this dose is insignificant when compared to the average individual dose of 1.2E-2 mSv/year due to natural C-14 or the 2.4 mSv/y average individual dose due to the natural background.

The collective dose of less than 1E-7 man.Sv/TWh due to potential transportation accidents is a very small part of the total 9.5 E-4 man.Sv/TWh public collective estimated for all transportation operations in France.

In case of a severe reactor accident, an indicative total collective dose for the population (for a radius of 3,000 km) for the four accident scenarios has been estimated. The impact of the reference scenario ST21 (core melt with a 1% of the core released) is a collective dose of about 58,000 man.Sv. For the other scenarios considered, the expected risk (consequences x probability of occurrence) varies between 0.001 and 0.08 man.Sv/TWh.

For the workers, the total collective dose for all the different stages of the fuel cycle is about 0.35 man.Sv/TWh. The electricity generation and the mining and milling stages are the operations where the occupational collective dose is the most important (0.2 man.Sv/TWh and 0.11 man.Sv/TWh, respectively).

Human Health Impacts

Routine Operations

The radiological health effects resulting from the normal operation of the nuclear fuel cycle are directly proportional to the total collective doses. The expected number of health effects were calculated assuming no lower threshold for radiological impacts, using internationally accepted data from Publication 60 of the International Commission on Radiological Protection. The total number of expected health impacts per TWh are: 0.65 fatal cancers, 1.57 non-fatal cancers, and 0.13 severe hereditary effects. These results include the long-term global dose assessment.

The number of estimated deaths for the European population due to the routine annual operation of one additional 1300 MWe PWR (about 7 TWh/y), integrated over 100,000 years, would be less than 1 fatal cancer (0.1). This can be compared to the approximate value of 800,000 fatal cancers reported in Europe each year.

It is estimated that the production of 1 TWh will result in 0.02 deaths, 0.96 permanent disabilities and 296 working-days-lost (non-radiological health impacts) in the work force for the nuclear industry. Worker accidents during the construction and the decommissioning of the reactor are the most important contributors to these values.

Accidental Situations

The transportation of the radioactive materials between the different sites and the transportation of the materials involved in the construction and the decommissioning of the reactor result in traffic accidents involving the general public. The number of non-radiological health impacts estimated are: 3E-4 deaths and 1.7E-3 injuries per TWh. Assuming an incremental annual production of 7 TWh, less than 1 death (0.002) can be expected per year. This is insignificant when compared to the nearly 10,000 traffic accident deaths that occur in France each year. In accidental situations occurring during the transportation of hazardous radioactive materials such as UF_6, the toxicological health impacts estimated are even smaller (2E-9 deaths/TWh and 7E-5 injuries/TWh).

The radiological health effects from reactor accidents can be divided into two categories: the immediate health effects (deterministic effects) and the stochastic effects as cancers or severe hereditary effects. For the four accident scenarios considered in this study, only the two most severe accidents lead to deterministic effects, but no deaths are expected for the reference scenario ST21. For the stochastic effects, as for normal operation, they are considered to be directly proportional to the collective doses. Depending on the scenario, the number of expected fatal cancers varies from 1E-4 to 3.9E-3 per TWh.

Monetary Valuation

Routine operation

The sub-total of the cost presented for all the stages of the nuclear fuel cycle is about 2.5 mECU/kWh, if no discount rate is applied. When 3% and 10% discount rates are used, the cost is reduced to 0.1 and 0.05 mECU/kWh, respectively. The current base load electricity generating costs in France are on the order of 35-40 mECU/kWh.

The dominant contributor to the total cost is the reprocessing stage (76%), followed by electricity generation (18%), when the 0% discount rate and the global impact assessment are implemented. When the 3% discount rate is applied, the construction of the reactor becomes the most important because discounting does not reduce the costs of the very short-term impacts assessed (40%). This is followed by mining and milling and electricity generation (19% and 17%, respectively). Six percent of the overall cost of the fuel cycle, at 0% discount rate, is due to occupational health impacts. This proportion increases in importance when a discount rate is applied (75% of sub-total for a discount rate of 3% and 95% of sub-total for a discount rate of 10%).

This sensitivity to the discount rate used is due to the relatively large portion of medium and long-term impacts associated with nuclear fuel cycle. For example, it can be seen that for no discount rate, the predominant costs are due to the global assessment, however, if a 3% discount rate is used, the short term, mostly occupational impacts dominate the final result. Another example is the waste disposal stage where the relatively small costs disappear if discount rates are applied. It has been for this reason that the use of this type of impact pathway methodology for the assessment of waste disposal and global impacts has been questioned.

Accidental Situations

The costs assessed for transportation impacts in general are extremely small. The portion attributed to accidental conditions can be considered insignificant due to the low probabilities and transportation packaging.

These results reported for the four accident scenarios are indicative of a risk-based methodology. The reference scenario, considered to represent a core melt accident followed by a release of 1% of the core, resulted in a 0.005 mECU/kWh cost. The risk for the other scenarios varies between 0.02 and 0.0005 mECU/kWh. The portion of these costs that might be internalised by nuclear accident insurance has not been addressed.

Further work must be done to evaluate other potential social impacts and external costs such as, *inter alia*, public perception, risk aversion, disruption of electricity supply, and decommissioning of the destroyed reactor. Besides the difficulty in assessing these impacts and costs, the partition of externalised versus internalised costs must also be evaluated.

Uncertainty

Each part of the impact pathway methodology employs different models and input data. The uncertainty of these calculations contributes to the overall confidence in the final results. In general the impacts of normal operations provide results that can be considered to have an uncertainty well within an order of magnitude, except for the estimates of human exposure and dose conversion where an uncertainty of an order of magnitude can be indicated in certain individual extreme cases. The propagation of the error estimates has shown that for the one standard deviation confidence interval, the results are considered to be correct within an order of magnitude.

Comparison with other Studies

For the most part, the earliest phases of comparative risk assessments of different fuel cycles options concentrated on deaths and injuries as the indicators for damages. In 1988, O. Hohmeyer published a report on the social costs of energy consumption. This was followed by a German-American workshop in 1990 on the external environmental costs of electric power and the "Pace University report" on the environmental costs of electricity. The results from these studies included impacts that had not been addressed in previous studies. The emphasis on social costs required that different impacts pathways be considered and it introduced the use of money as an impact indicator.

The range of results that have been reported for the external costs of the nuclear fuel cycle is from 0.1 to close to 100 mECU/kWh. This large range of results can be attributed to: (1) the different number of fuel cycle stages included in each assessment, (2) the methodology for the valuation of the health impacts, (3) the discount rate applied, (4) the inclusion of a global physical impact assessment, and (5) the methodology and assumptions used for the assessment of severe nuclear accidents.

One of the major disadvantages of these studies were that the boundaries of the assessments and methodologies applied varied between the different fuel cycles, therefore it was difficult to objectively compare the final results. The ExternE project set, as one of its main objectives, the goal of analysing different fuel cycles within the same consistent methodological framework in order to allow for direct comparison between fuel cycle options. Since the start of this project, a number of other projects have been initiated on the international and national levels.

Limitations of the Results

In attempts to include all impacts within a rather large time and space scale, the limits of valid assumptions have been stretched in order to be as complete as possible in the physical impact assessment. Current day conditions have been assumed to remain constant for 100,000 years - a truly unlikely event. However, with these assumptions it was possible to complete the assessment for the long-lived radionuclides.

No thresholds have been assumed in the calculation of the response to the doses received, so in many cases, the average individual doses to the public fall into a highly uncertain area of the dose-response relationship. The generally accepted collective dose approach, which integrates the average individual doses over the total population to be considered, was implemented. With this approach, the magnitude of the individual risk is masked when the results are presented. The most obvious drawback of this approach has been seen in the evaluation of the long-term global impacts of C-14, where the very small individual doses are summed to a large value over time and space. However, without this assessment, an important part of the overall potential physical impacts from the nuclear fuel cycle would not have been completed.

The assessment of a potential severe nuclear reactor accident was based on a risk-based approach. This methodology has not been accepted by everyone, however, it has been judged as an adequate basis for the calculation of the physical impacts within a range of uncertainty. This does emphasise the need for further methodological work to determine the additional social impacts and costs that have not been included in this type of approach.

Although there are limitations and uncertainties in the methods used for the assessment of the physical impacts, the key methodological issues that remain are for the monetary valuation stage. Before the monetary values of the impacts from the nuclear fuel cycle can be considered to be external costs the following issues must be addressed:

- If the same monetary valuation methodology is used for the evaluation of very small (and quite uncertain) individual risks to a large population and larger individual risks to a smaller population, does the final result really demonstrate the proper weighting of the real risks? Should occupational (voluntary) risks be valued in the same manner as risks to the general public (involuntary risks).

- If the use of a discount rate is not considered to be acceptable for the evaluation of far future impacts, what should be used in its place? The results of this phase of the project should be reviewed in order to determine how to provide a good representation of present day and far future risks.

- How can a method realistically incorporate the societal perceptions in terms of time and space keeping in mind the need for society to balance between the options available? For example, if C-14 is released today, it is diluted and results in low individual risks with no future disposal problems. If the releases are captured, waste repositories must be maintained causing increases in occupational risks and larger local population risks in the far future.

- How can the aversion of certain risks be equitably included in the assessment of external costs? This problem is clearly illustrated in the differences between expert and public perceptions of the risks of potential nuclear accidents and high-level waste disposal.

Conclusions

The systematic assessment of all the stages of the nuclear fuel cycle with a common methodology and the same boundaries has provided a good base of information with which to understand the health and environmental impacts of fuel cycle and preliminary assessment of the costs of these impacts. At this stage, no definitive evaluation has been made to determine to what extent the costs presented in this report are externalities.

In presenting these costs, care has been taken to indicate that they are considered to be a "sub-total" and are not intended to represent the absolute total of all the possible impacts. Within the constraints of available resources and existing methodologies, the priority impact pathways have been analysed. In some cases, the assumptions have been pushed to the limit of validity in an effort to be as complete as possible.

For nuclear accidents most of the work has addressed valuation of health impacts and costs of the countermeasures implemented to limit the possible health impacts. Further work is needed to better estimate some of the other social costs and the extent to which those costs can be considered to be externalities. More research is needed to continue to develop these ideas.

Even though some unresolved issues remain, this study has made important advances in reporting the physical impacts and monetary valuation of these impacts in a manner consistent with other fuel cycles. In addition, many remaining uncertainties in the methodology have been identified, and important parameters that should be considered in the decision making process.

1. INTRODUCTION

In 1991, the European Community (EC) and the US Department of Energy initiated a joint research project to assess the external costs of fuel cycles used to generate electricity. The intention of this project, called the EC-US External Costs of Fuel Cycles Project (ECFC), was to develop a conceptual approach, consistent methodology and identify future research in the assessment of the externalities.

External costs of the production of energy are not reflected in traditional energy prices and are passed onto the society as a whole or to future generations. During the past decade there has been increased interest in the assessment and integration of external costs into the decision making process due to the growing interest and concern over environmental issues. Externalities include the cost of damages to, *inter alia,* human health, natural ecosystems, and man-made urban and agriculture environments. On a project-wide basis, it was decided that the assessment would be limited to the most important impacts resulting from first-order processes, except for the cases where the largest impact would be missed if the boundaries were not expanded.

A second phase of the project continued in Europe (with a new name "ExternE") and expanded to include the implementation of the consistent methodology in various EC countries. The development of the methodology and the results of a practical implementation have already been presented for the nuclear fuel cycle (CEC, 1995, Dreicer *et al*, 1995). The French Implementation Project, presented in this report, is a summary of those results and an expansion of the assessment of the electricity generation stage to include various sites for a 1300 MW reactor.

This report is organised in a manner to present: the description of the nuclear fuel cycle in France with a brief description of the reference facilities, sites, and source term data; a summary of the methodology employed; and a discussion of the radiological dose calculations and the monetary valuation of the impacts.

2. NUCLEAR FUEL CYCLE

2.1. General Description of Present Cycle

The nuclear fuel cycle in France has been classified into the 8 different stages leading to the production of energy from the use of uranium: mining and milling, conversion, enrichment, fabrication of the fuel, electricity generation, reprocessing of spent fuel, and finally, disposal of two classifications of waste (low/intermediate level and high level waste). A specific technology is employed for each stage and in some cases the different process may be carried out at different locations. The main stages of the nuclear fuel cycle are illustrated on Figure 1. The transportation of radioactive materials and wastes between these sites has also been considered.

The uranium is mined in either open-pit or underground mines depending on the depth of the ore. The ore is processed into an usable form at mills which are usually located in close proximity to the mines for ease of transport. Milling involves crushing and grinding the ore, separation of the uranium from waste rock, and then refinement and purification. When the ore leaves the mill the uranium is in the form of uranium oxide (U_3O_8), known as 'yellowcake'.

From the mill, the yellowcake goes to be converted into uranium hexafluoride (UF_6). This is achieved through several chemical transformations. The UF_4 is produced first and then it is fluorinated to produce UF_6, thus completing the conversion step.

To fabricate nuclear reactor fuel, the UF_6 must be further enriched in the fissile isotope U-235. The enrichment process increases the percentage of U-235 to 3-4% from the approximately 0.7% naturally occurring U-235. Gaseous diffusion is currently the main commercial method used to accomplish this step.

The enriched UF_6 is transformed into uranium dioxide (UO_2) powder, and then pressed into pellets. The pellets are, in turn, inserted into thin tubes of zircaloy or stainless steel, which are sealed and assembled in bundles to form fuel assemblies. These fuel assemblies are placed in the core of a nuclear reactor.

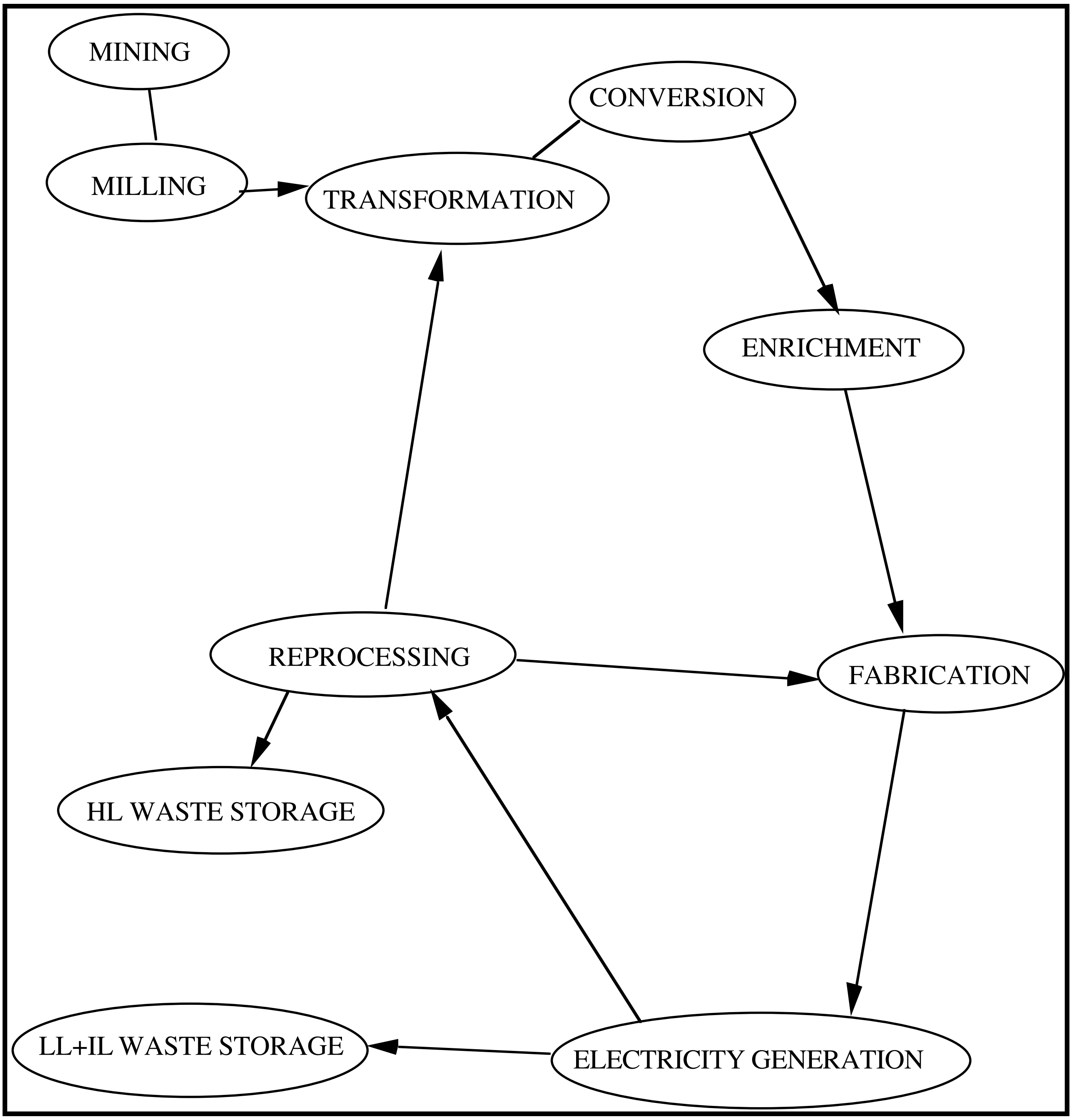

Figure 1. The nuclear fuel cycle

There are several nuclear reactor technologies in the world, pressurised water reactors (PWR) being the most common. These reactors use water coolant to dissipate the heat given off by the fission of U-235 atoms in the core. The heat produced is used to create steam (either in the reactor vessel itself in the case of boiling water reactors, or in a separate water circuit in the case of pressurised water reactors) which drives turbines for the generation of electricity, as in any conventional power plant. To maintain efficient power production, about one third of the fuel assemblies are replaced each year. The spent fuel assemblies reside in storage ponds on the reactor site for approximately one year to allow cooling and radioactive decay.

The spent fuel is then stored awaiting final disposal (as in USA) or it is reprocessed (as in France, the United Kingdom and Japan). The composition of the fuel at removal is 96% uranium, 1% plutonium and 3% fission products. Reprocessing allows for the recycling of the uranium and use of plutonium in the production of mixed oxide fuel. The waste products from reprocessing are highly radioactive and after vitrification are stored awaiting final disposal.

Radioactive waste is produced throughout the nuclear fuel cycle. Depending on the levels of radioactivity, these wastes are classified as high, intermediate, or low level. Low and intermediate surface waste disposal sites already exist, but at this time no final decisions have been taken for the disposal of high level waste (HLW). It seems possible that deep geological disposal will be used to dispose of high level waste, but as yet no sites are in operation.

2.2. Reference Sites and Technologies

For the purpose of assessing the external costs of nuclear energy production, a reference site and technology for each stage of the fuel cycle in France were considered, including a hypothetical site for the high level waste disposal facility. A reference technology for the 1990's was chosen for each stage. The locations of the sites are presented on Figure 2.

2.2.1. Mining and Milling

Several years ago, there were 4 large mining operations in France (Forez, Vendée, La Crouzille and Lodève), however, in 1994 there is only one large mine remaining in operation at the Lodève site in Hérault. This site, operated by the COGEMA since 1975, includes both open pit and underground mining operations and represents the most modern uranium mining techniques in France. From 1984 to 1992, the average annual production of uranium ore was 922.5 tonnes and 107.6 tonnes from the underground and open-pit sites, respectively. The average ore grade of 0.25% represents a relatively high uranium content in comparison with the average ore grade mined in France.

Since 1981, the milling operations at Lodève have used an alkaline leaching technology. The ore is purified by the precipitation of the uranium as sodium diuranate to produce uranium peroxide ("yellowcake"). The annual production capacity of the mill is 1100 tonnes of yellowcake.

For economic reasons, during the next few years the operations at Lodève will be closed. This is the expected fate of all mining operations because the French uranium production is not sufficient for domestic needs, and thousands of tons of uranium are being imported from Niger, Gabon, Canada and Australia.

The damages for this stage of the fuel cycle are normalised to an average annual production of approximately 46 TWh of electricity (22 tonnes of mined uranium ore are needed to produce 1 TWh at a PWR). The mill will be normalised by the same energy production because it treats the total annual production from the mine and does not import ore from other sites.

2.2.2. Conversion

The conversion of yellow-cake to the uranium hexafluoride (UF_6) needed for the fabrication of fuel, is carried out in two steps. At Malvesi near the city of Narbonne, 10 km from the Mediterranean Sea, the yellowcake is converted to uranium tetrafluoride (UF_4) by stages of dissolution, purification, precipitation, calcination, and fluorination. At a second plant at Pierrelatte, in the Rhône River Valley between the Alps and the Massif Central, the UF_4 is converted to UF_6 by fluorination, purification, recovery and distillation stages. The annual production for the first step is equivalent to the generation of 500 TWh of electricity generation per year. Some of this material is exported so the second step is normalised to an equivalent of 310 TWh per year. These plants are operated by COMURHEX.

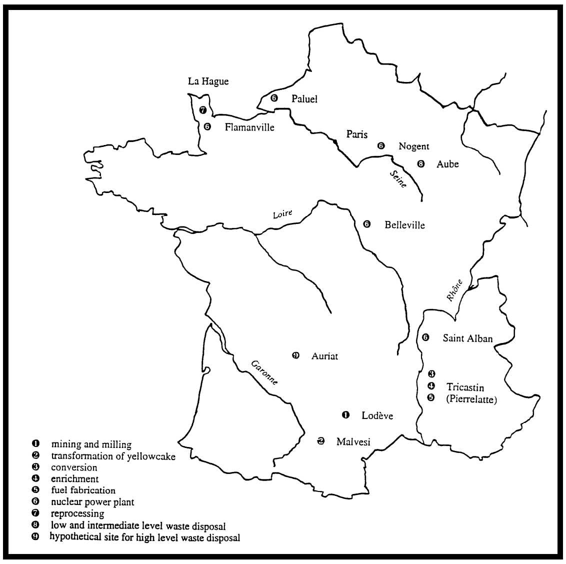

Figure 2. Locations of the French nuclear fuel cycle facilities

2.2.3. Enrichment

The enrichment stage increases the natural 0.7% of U-235 found in the UF_6 to approximately 3-4%. The only civilian enrichment plant, operated by EURODIF on the Pierrelatte site in the Rhône Valley since 1979, supplies over one-third of the world's enriched uranium. The total yearly production of enriched uranium at this plant is equivalent to 600 TWh of electricity at a PWR. This stage requires the largest amount of energy and the electricity is provided by the Tricastin nuclear power plant on the same site.

2.2.4. Fuel Fabrication

For the fabrication of fuel, the UF_6 is chemically transformed to metal oxide powder (UO_2) which is compacted and sintered into pellets that can be loaded into zirconium tubes. Two hundred and sixty four rods are then assembled into a fuel assembly. There are two fuel fabrication facilities, operated by FBFC (Franco-Belge de Fabrication du Combustible), at Romans and Pierrelatte in the south-eastern part of France which are almost equally representative. The Pierrelatte site has been evaluated for this project. It produces fuel for the equivalent of 140 TWh of electricity generation each year.

2.2.5. Electricity Generation

In France, 75% of the electricity is produced by nuclear power plants. This electricity is almost totally generated by the 54 PWR's currently operated by EDF (Electricité de France). The other reactors (two fast breeder reactors (FBR)) are not in full operation and the last gas-cooled natural uranium reactor (Bugey 1) was closed in 1994. Thirty four of the units are 900 MWe and 20 units that were built more recently are 1300 MWe (Table 1). There are also four 1450 MWe PWR under construction.

Table 1. French nuclear power plants

Type of reactor	Number of reactors	Number of sites	Nuclear net capacity (MWe)
900 MWe PWR	34	9	30,735
1300 MWe PWR	20	8	26,370
FBR	2	2	1,433
Total	56	19	58,538

Although the 900 MWe PWR technology represents more than 50% of the reactors in operation in France, the 1300 MWe reactors were considered to be representative of modern technology employed today; a 900 MWe reactor would not be built today. The evaluation of the external costs from the routine operation of the five following 1300 MWe reactors are presented in this report: Belleville, Flamanville, Nogent, Paluel and Saint-Alban (Table 2). There are two units on all sites, except for the Paluel, which operates four 1300 MWe units. The average yearly production per reactor is about 7 TWh.

Table 2. Main characteristics of the 5 electricity generation plants

Site	Number of reactors	Total net power (MWe)	Start of Commercial operation	Average (1) production (TWh/y)	Average (1) production per reactor (TWh/y)
Belleville	2	2,620	1987-88	14.02	7.01
Flamanville	2	2,660	1985-86	13.78	6.89
Nogent	2	2,620	1987-88	13.74	6.87
Paluel	4	5,320	1984-86	30.13	7.53
Saint-Alban	2	2,670	1985-86	12.88	6.44

(1) average value (1990-1992)

The 5 sites, shown on Figure 2, are located in different areas of France: Flamanville and Paluel are on the English Channel, Nogent is on the Seine River, relatively near the city of Paris, Belleville is on the Loire River, and Saint-Alban is located on the Rhône River, relatively near the city of Lyon. The general characteristics of the sites are presented on Table 3. The remaining three 1300 MWe facilities that exist in France (Golfech, Cattenom and Penly) were not included in this assessment because they have only been recently opened and the data on releases and effluents cannot be considered to be representative. The 5 sites included can be considered to represent the various types of sites currently in use.

The impacts of the 1300 MWe PWR, presented in this report, can be compared to the impacts of the Tricastin 900 MWe PWR, located at Pierrelatte (4 operational 900 MWe units) and that have been evaluated in a previous study (Dreicer *et al*, 1995). This site is situated on the Rhône River, and the average annual energy production of one unit, calculated, based on the operation of the four units from 1982-1991, is 5.7 TWh.

Table 3. Geographic characteristics of the 1300 MWe PWR sites

Site	Population 100 km (millions)	Aquatic releases to	Region	Agricultural main production	Climate	Predominant winds
Belleville	1.6	Loire River	Centre	cereals, meat	oceanic	SW-W speed: 93% < 10 m/s
Flamanville	0.48	English Channel	Basse-Normandie	milk	oceanic	W-NW average speed 7 m/s
Nogent	8.9	Seine River	Champagne -Ardennes	cereals	oceanic with continental influences	SW, NE average speed 3.8 m/s
Paluel	2.06	English Channel	Haute-Normandie	milk	oceanic	W-SW
Saint-Alban	4.65	Rhône River	Rhône-Alpes	milk	continental and oceanic	N-NE,S 35% weak speed (< 0.5 m/s)

For the electricity generation stage, in addition to the routine operation of a PWR, the construction and decommissioning have also been included in the assessment. The results for these phases of a 1300 MW facility have been normalised to the average production of 30 years of operation of a 210 TWh.

It has been assumed, in accordance with current French policy, that the decommissioning of the reactor will begin after a post-closure waiting period of 50 years. This will allow for the natural decay of the radioactivity before the dismantling and disposal process begins. The potential impacts of a severe reactor accident were evaluated for a representative location in the center of western Europe.

2.2.6. Reprocessing

In France, the reprocessing of spent fuel began at the UP2 facility at the La Hague site in 1966. This site is located on the north-western coast of France near the Flamanville power plant. The latest technology is in the newly opened UP3 facility, which began operations in 1990 using the PUREX process (Plutonium Uranium Recovery by EXtraction). This process involves the shearing of the fuel elements, followed by dissolution in nitric acid, extraction of the uranium and plutonium, and the final preparation of the oxides of Pu and

U. The facility was not operating at total capacity in 1991 (almost 44% of its capacity), so the source term that was used represents an equivalent of 81.4 TWh of electricity production per year. Data for 1993 show an increased production capacity at a lower rate of release per TWh.

2.2.7. Waste Disposal

The surface land disposal of low and intermediate level radioactive wastes at the Centre de l'Aube, located 180 km east of Paris, covers about 1 km^2 of land. The facility has 5 major components: the disposal structure, backfill, the floor and associated water collection system, the temporary cover and the final cover. This follows the multi-barrier concept for containment and allowing for easy leakage monitoring. The site has been designed to contain the waste of an equivalent of about 10,000 TWh of electricity production.

There is currently no existing permanent high level radioactive waste disposal site in operation in the world. For the purpose of this assessment, the results reported in the CEC PAGIS (Performance Assessment on Geological Isolation Systems) study (CEC, 1988) for a hypothetical site of a deep geologic disposal on the Massif Central in France have been used. This deep geologic granite disposal site is hypothesised to hold vitrified waste from 1,800 GWy of power production.

2.2.8. Transportation

The transportation of material between sites has been considered as a separate fuel cycle stage in this assessment. In France, the transport of radioactive material by road or rail follows the International Atomic Energy Agency (IAEA) approved safety practices. The distances vary between 5 km and 900 km. Material transport for the production of fuel, the actual fuel, and the waste generated during the cycle are considered for both routine conditions and potential accidents.

Table 4 presents the description of the transportation between the different sites. The data concerning the material transported per TWh are based on the 900 MWe PWR data (Dreicer *et al*, 1995). For the different 1300 MW PWR's considered in this report the new transportation routes have been considered. In Table 4, the distances presented associated with the electricity generation stage (fuel assemblies, spent fuel and reactor waste) are the average value for the 5 sites considered. The distances from Pierrelatte to the power plant vary from 130 km (Saint Alban) to 1020 km (Flamanville). For the transportation of spent fuel, the distance travelled between La Hague and the power plant varies between 20 km for Flamanville to 900 km for Saint-Alban. The range of distances for the transportation of waste to the CSA is from 100 km (Nogent) to 600 km (Flamanville).

Table 4. The mass and relevant packaging for the radioactive material transported in the nuclear fuel cycle

Material transported	Tons of material /TWh	From... to...	Mode	Packaging and capacity in tons	Transport index (2)	Average distance travelled
yellow cake (U = 70%)	3.18E+01	Lodève to Malvesi	rail	DV55 1-2.4 1.7 tons of concentrate	0.1	90 km
UF_4	2.93E+01	Malvesi to Pierrelatte	road	artic. truck silo 26 tons of UF_4	0.1	200 km
UF_6 natural	4.09E+01	Pierrelatte to Pierrelatte	road	48Y 8.3 tons of U	0.1	5 km
UF_6 enriched	6.21E+00	Pierrelatte to Pierrelatte	road	30B + shell 1.5 tons of U	0.1	5 km
fuel assembly (UO_2+assembly)	6.32E+00	Pierrelatte to Reactor	road	RCC (2xAF) 1.33 tons (2 assemblies)	1.6	620 km (1)
spent fuel assembly	6.32E+00	Reactor to La Hague	rail	4.585 tons (7 assemblies)	10	450 km (1)
$UO_2(NO_3)_2$ (from reprocessing)	1.67E+01	La Hague to Malvesi	rail	LR35(cap. 2.5m^3) 1.125 tons product (0.68 tons of U)	10	900 km
reactor wastes (concrete blocks)	7 blocks/TWh	Reactor to CSA (Aube)	road	3 or 4.5 tons	20	350 km (1)
	7	Reactor to CSA (Aube)	rail	3 or 4.5 tons	20	350 km (1)
reactor wastes (metallic barrels)	39 barrels/TWh	Reactor to CSA (Aube)	road	0.12 tons	5	350 km (1)
	44	Reactor to CSA (Aube)	rail	0.12 tons	5	350 km (1)

(1) average for the 5 electricity generation sites
(2) dose rate at 1m from the packages in 1E-2 Sv/h

2.3. Source Terms

The list of radionuclides and the amounts released to the air and aquatic environment for 6 of the stages of the nuclear fuel cycle are presented in Tables 5 and 6, respectively. The source terms are taken from the monitoring data reported by the facilities (Dreicer *et al*, 1995). A range of release data is presented for the electricity generation stage because 5 different 1300 PWR sites were considered in this project. More details of these data are presented later in this section.

Mining and milling releases are not measured due to the numerous and diffuse release points on the site. The source term used in this project is based on release estimates normalised to energy production reported by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR, 1993). Due to the fact that at Lodève mill tailings are disposed of in old mines and covered, it is assumed that the radon releases stop at the closure of the facilities. Other data that was not directly available was the C-14 releases from the reactors and the reprocessing plant. The UNSCEAR 1993 values were used for the electricity generation stage and for the reprocessing discharges of C-14 assumptions were taken based on information provided by the facility. The atmospheric C-14 source term was assuming to be 10,000 times lower than the gaseous Kr-85 releases, and the liquid releases of C-14 were assumed to be based on the ratio of gaseous to liquid releases reported by the Sellafield reprocessing plant (Dreicer *et al*, 1995).

The evaluation of the normal operation of the reactor and the transportation stage have been expanded to take into account the assessment of 5 different sites. The additional details are presented below. The details of the source terms for the waste disposal stages and the severe reactor accident assessment are also presented in this report.

Table 5. Source terms for the nuclear fuel cycle in routine operation, atmospheric releases in MBq/TWh

Radio-nuclide	Mining and milling	Conversion Malvesi + Pierrelatte	Enrich-ment	Fuel fabrication	Electricity generation	Reprocessing UP3
H-3	-	-	-	-	6.9E4 - 3.3E5	4.58E4
C-14	-	-	-	-	1.4E4	7.13E4
Co-58	-	-	-	-	2.5E-1 - 9.4E-1	-
Co-60	-	-	-	-	2.5E-1 - 9.4E-1	-
Kr-85	-	-	-	-	3.5E4 - 1.7E5	7.13E8
I-129	-	-	-	-	-	5.11E1
I-131	-	-	-	-	1.5 - 5.6	7.08E-1
I-133	-	-	-	-	3.0 - 1.1E1	3.13E-1
Xe-133	-	-	-	-	4.8E5 - 2.3E6	-
Cs-134	-	-	-	-	2.5E-1 - 9.4E-1	-
Cs-137	-	-	-	-	2.5E-1 - 9.4E-1	-
Rn-222	5.1E8	-	-	-	-	-
U-234	2.1E3	4.03E-1	1.9E-1	3.0E-3	-	-
U-235	8.9E1	1.75E-2	9.7E-3	2.0E-4	-	-
U-238	2.1E3	3.81E-1	1.0E-1	7.4E-4	-	-
Pu-238	-	-	-	-	-	1.02E-5
Pu-239	-	-	-	-	-	2.33E-5

Table 6. Source terms for the nuclear fuel cycle in routine operation, liquid releases in MBq/TWh

Radio-nuclide	Mining and milling	Conversion	Enrich-ment	Fuel fabrication	Electricity generation	Reprocessing UP3
H-3	-	-	-	-	1.4E6 - 2.9E6	2.89E7
C-14	-	-	-	-	-	4.55E4
Mn-54	-	-	-	-	2.4E1 - 9.5E1	-
Co-58	-	-	-	-	3.8E2 - 1.4E3	-
Co-60	-	-	-	-	1.2E2 - 7.3E2	9.1E3
Sr-90	-	-	-	-	-	1.46E5
Ru-106	-	-	-	-	-	8.77E4
Ag-110m	-	-	-	-	3.9E1 - 3.5E2	-
Sb-124	-	-	-	-	8.0E1 - 4.2E2	-
Sb-125	-	-	-	-	-	6.17E4
I-129	-	-	-	-	-	6.84E2
I-131	-	-	-	-	5.2 - 1.4E1	-
Cs-134	-	-	-	-	7.76 - 1.3E2	1.5E3
Cs-137	-	-	-	-	1.0E1 - 2.3E2	1.38E4
U-234	-	1.28E1	3.8E-2	4.33	-	-
U-235	-	5.49E-1	1.9E-3	2.86E-1	-	-
U-238	8.6 Bq/l *	1.21E1	2.0E-2	1.06	-	1.47E1
Pu-238	-	-	-	-	-	9.04E1
Pu-239	-	-	-	-	-	5.43E1
Am-241	-	-	-	-	-	9.21E1
Cm-244	-	-	-	-	-	4.42E1

* concentration in river water (downstream)

2.3.1. Electricity Generation

2.3.1.1 Routine operation

Tables 7 and 8 present the detailed data for the 5 electricity generation sites (EDF, 1992) evaluated. The data is a 3-year average (1990-1992), and considered to represent the release from a modern average 1300 MWe reactor.

Table 7. Atmospheric source terms for the 5 sites of the electricity generation stage in MBq/TWh

Radionuclide	Belleville	Flamanville	Nogent	Paluel	Saint-Alban
H-3	2.92E5	6.92E4	2.45E5	3.32E5	1.09E5
C-14	1.4E4	1.4E4	1.4E4	1.4E4	1.4E4
Co-58	9.4E-1	4.96E-1	3.51E-1	2.54E-1	3.30E-1
Co-60	9.4E-1	4.96E-1	3.51E-1	2.54E-1	3.30E-1
Kr-85	1.46E5	3.46E4	1.22E5	1.67E5	5.43E4
I-131	5.63	2.97	2.11	1.53	1.98
I-133	1.13E1	5.95	4.22	3.05	3.96
Xe-133	2.05E6	4.84E5	1.72E6	2.33E6	7.61E5
Cs-134	9.4E-1	4.96E-1	3.51E-1	2.54E-1	3.30E-1
Cs-137	9.4E-1	4.96E-1	3.51E-1	2.54E-1	3.30E-1
Total noble gases	2.46E6	5.88E5	2.09E6	2.83E6	9.24E5
Total halogens + aerosols	20.69	10.91	7.73	5.60	7.26

The data presented on Table 7 has been normalised to the production at each facility. Being that the legal release limits are site-specific, there is some variation between the different sites however in almost all cases, it is not significant. As indicated before, the C-14 releases are not reported by EDF, so an estimate of 1.4E4 MBq/TWh was used (UNSCEAR, 1993) for all sites.

For the sake of comparison, the releases seen from the Tricastin PWR (900 MWe) that was included in the first fuel cycle assessment (Dreicer *et al*, 1995) fall within the same range as the 1300 MWe source terms: 1.4E6 MBq/TWh for the liquid tritium releases, 1.66E3 MBq/TWh for the liquid activities excluded tritium, 10.84 MBq/TWh for the halogens and aerosols activities. Only in the case of the noble gases (8.9E6 MBq/TWh) are the releases greater than the average 1300 MWe releases.

Table 8. Liquid source terms for the 5 sites of the electricity generation stage in MBq/TWh

Radionuclide	Belleville	Flamanville	Nogent	Paluel	Saint-Alban
H-3	2.55E6	2.88E6	1.43E6	2.82E6	1.63E6
Co-58	4.44E2	7.91E2	3.81E2	1.39E3	8.23E2
Co-60	2.94E2	3.96E2	1.19E2	4.38E2	7.27E2
I-131	9.48	1.4E1	1.2E1	1.3E1	5.2
Cs-134	3.6E1	7.5E1	2.7E1	1.28E2	7.76
Cs-137	5.5E1	9.6E1	4.8E1	2.26E2	1.0E1
Mn-54	2.4E1	4.6E1	4.2E1	3.5E1	9.5E1
Ag-110m	1.02E2	3.53E2	3.9E1	3.04E2	2.10E2
Sb-124	8.0E1	1.57E2	8.7E1	2.70E2	4.19E2
Total without H-3	1.04E3	1.93E3	0.76E3	2.81E3	2.30E3

2.3.1.2. Severe Reactor Accident

For the assessment of the severe reactor accident, four different source terms, based on the radionuclide inventory of a 1250 MWe PWR, were considered. In all cases a core melt accident is assumed. The difference is in the degree of the breach of the containment. In the worst case, a total breach of containment occurs and in the best case all the safety control operate as designed.

The reference scenario for France, the ST21 source term, presented on Table 9, assumes a containment failure that results in the total release of 10% of noble gases from the core, 1% of the more volatile elements, such as caesium and iodine and smaller percentages of other elements (see Table 10). It is assumed to occur in a single release phase without energy release. This source term is in the same order of magnitude as the reference accident scenario considered by the French national safety authorities (Queniart *et al*, 1994).

To illustrate the sensitivity of the results the impacts of three other source terms are presented. The worst case, the ST2 source term, assumes a massive containment breach with about 10% of the core released. This is based on a source term presented in a CEC/NEA inter-comparison study (OECD, 1994). The lowest release (ST23), which is 2 orders of magnitude smaller than the ST21, represents a design-based leakage after the core melt accident has occurred.

The final results of this evaluation has been normalised to the electricity production of a 1300 MWe PWR.

Table 9. Radionuclides and activity released from ST21

Radionuclide	Activity (TBq)	Radionuclide	Activity (TBq)
Kr-85m	6.79E4	Te-131m	3.31E3
Kr-87	5.71E4	Te-132	4.76E5
Kr-88	1.44E5	I -131	3.40E5
Rb-88	1.58E5	I -132	4.92E5
Sr-89	3.37E3	I -133	6.43E5
Sr-90	1.65E2	I -134	1.63E5
Sr-91	3.77E3	I -135	5.24E5
Y -91	4.52E2	Xe-133	6.78E5
Zr-95	5.87E2	Xe-135	1.52E5
Mo-99	6.31E3	Cs-134	3.85E3
Ru-103	5.24E3	Cs-136	1.32E3
Ru-105	2.57E3	Cs-137	2.29E3
Ru-106	1.30E3	Ba-140	6.11E3
Rh-105	3.18E3	La-140	8.18E2
Sb-127	2.89E3	Ce-144	3.59E2
Sb-129	7.22E3	Np-239	7.14E3
Te-127m	4.37E2	Pu-238	0.32
Te-127	2.79E3	Pu-239	0.11
Te-129m	1.67E3	Pu-241	3.21E1
Te-129	7.64E3	Cm-242	6.62
		Cm-244	0.28

Table 10. Fraction of radionuclide groups released for ST21 source term [a]

Xe-Kr	0.1
organic I	0.0001
I	0.01
Cs-Rb	0.01
Te-Sb	0.01
Ba-Sr-Ru[b]	0.001
La[c]	0.001

Remarks:

a: all elements, apart from organic iodine and the noble gases, are assumed to be released in the oxide form and as an aerosol of size of 1 μm AMAD (Activity Median Aerodynamic Diameter)

b: Ru: includes Ru, Rh, Co, Mo, Tc

c: La: includes Y, La, Zr, Nb. Ce, Pr, Nd, Np, Pu, Am, Cm

2.3.2. Waste Disposal

The radionuclide inventory estimated to fill the low and intermediate level waste repository after the 30 years of operation is presented in Table 11.

The high level waste disposal site inventory (Table 12) is assumed to be the vitrified waste from the reprocessing of 1800 GWe.year of power production (30 years of French production) or about 48,000 tonnes of heavy metal. The amount of radioactivity is at the time of vitrification. The normal evolution scenario dose assessment has taken into account only 6 most important radionuclides, as is indicted on Table 12.

Table 11. Estimated inventory at the end of the operational phase for the surface waste disposal site Centre de l'Aube

Radionuclide *	Activity (MBq)	Radionuclide *	Activity (MBq)
H-3	4.0E9	Pd-107	3.0E9
C-14	4.0E8	I-129	3.0E6
Co-60	4.0E11	Cs-135	6.0E7
Ni-59	4.0E9	Cs-137	2.0E11
Ni-63	4.0E10	U-234	2.0E7
Sr-90	4.0E10	U-238	2.0E7
Zr-93	4.0E8	Pu-239	2.4E8
Nb-94	2.0E7	Pu-241	2.3E8
Mo-93	1.0E8	Am-241	3.5E8
Tc-99	1.2E7	Np-237	1.0E6

** The build-up of the following daughter products is also taken into account: Nb-93m, Th-230, Ra-226, Pb-210, Po-210, U-235, Pa-231, Ac-227, U-233 and Th-229. If any of these nuclides have daughter products these are also taken into account.*

Table 12. Radionuclide inventory considered for HLW disposal

Radionuclide	Activity (MBq)	Normal evolution scenario
Se-79	5.8E8	
Zr-93	5.1E9	X
Tc-99	2.5E10	X
Pd-107	1.8E8	
Sn-126	1.0E9	
Cs-135	2.7E9	X
Th-229	(*)	X
U-233	3.0E6	X
Np-237	5.4E8	X
Pu-239	2.0E9	
Pu-240	3.2E9	
Am-241	1.3E7	
Am-243	3.6E10	

(*) Daughter-nuclide having a concentration near zero at the vitrification time.

2.3.3. Transportation

For the routine transportation operations, the source of the radioactive exposure is direct irradiation at the exterior surface of the transported packages (see Table 4). For accidental circumstances, only the releases of UF_6 (natural and enriched) have been included because it is the most toxic of the potential releases. Depending on the hypothesised accident scenario, the potential releases vary between 0.1 % to 100 % of the total amount transported. This corresponds to between 8.3 and 8,300 kg of natural uranium in the form of UF_6, and between 1.5 and 1,500 kg of enriched UF_6.

2.4. Future Cycles

During the next twenty years further installations of PWR's are planned in France. Four 1450 MWe PWR are currently under construction. It is expected that for future PWR reactors, the fuel use and the core management will change to include: longer cycles between refuelling, improved enrichment levels, increase of burn-up levels, use of plutonium in mixed oxide fuel (MOX) with uranium, and the use of reprocessed uranium (Carle, 1994).

The use of MOX fuel has begun in some reactors and the French mixed fuel fabrication plant (MELOX) is under startup. These modifications will not change the reactor technology, but at this time, it is not known if the releases will change significantly.

In the long term, fast breeder reactors may play an important role in France, however, the design of the future generation of reactors will only be selected in about 20 years.

3. METHODOLOGY

3.1. Impact Pathway Approach

The impact pathway approach, which has also been called a bottom-up type approach, starts with the environmental releases or actions that could result in damages or benefits to human health or the environment. In the nuclear fuel cycle, the impacts are characterised in terms of radiological and non-radiological impacts in the public, occupational and environmental domains.

Within the context of the ExternE project it has not been possible to consider all the possible impacts, therefore, only the most important impacts, called priority impacts, have been included. The highest priority has been given to the releases of radioactive material to the environment which potentially impact public health. Occupational health impacts from both radiological and non-radiological causes are also considered to be a priority, even though the extent to which occupational health impacts can be considered externalities has not yet been addressed.

The impacts on the environment of increased levels of natural background radiation due to the routine releases of radionuclides have not been considered as a priority impact pathway. The most important impacts to the natural environment that could be expected would be the result of major accidental releases. This type of impact has been included in the economic damage estimates as the loss of land-use and agricultural products after a potential severe reactor accident. Possible long-term ecological impacts have not been considered at this time.

The final stage of economic valuation of the impacts has been challenging due to the stochastic nature and delay time of the manifestation of radiological health effects at low levels of exposure, and the lack of contingent valuation studies applicable directly to radio-induced cancer and occupational health in the long term. The economic data that is currently available for mortality and morbidity have been utilised.

3.1.1. Boundaries of the Assessment

In order to evaluate the incremental impact of an additional power station, the first-order processes for the routine operation of each stage of the nuclear fuel cycle have been included in this assessment. The construction and decommissioning are also evaluated for the electricity generation stage. Accidental situations for the reactor and transportation stages are addressed. Although there can be arguments made for the inclusion of second-order processes, this has not been done on a project-wide basis, except in the fuel cycles where significant impacts would be neglected.

The most important choices for the assessment of the nuclear fuel cycle concern the definition of time and space boundaries. Due to the long half-life of some of the radionuclides, low-level doses will exist very far into the future. These low-level exposures can add up to larger number when spread across many people and many years (assuming constant conditions). The evaluation was completed using the conservative assumptions that:

- lifestyles in the future would result in the same level of external and internal radiation exposure, as would exist today,
- a linear response to radiation exposure at very small doses does exist,
- the dose-response function of humans to radiation exposure will remain the same as today, and
- that the fraction of cancers that result in death remains the same as today.

The validity of this type of modelling has been widely discussed. The uncertainty of the models increases and the level of doses, that are estimated, fall into the range where there is no clear evidence of resulting radiological health effects. In addition, the very long time scale presents some problems in the direct comparison of the nuclear fuel cycle with the other fuel cycles which have considered mainly shorter term impacts. In spite of these drawbacks, it was decided that within the project-wide guidelines followed by all fuel cycles, this type of risk assessment methodology was required. The short-term time scale of 1 year was considered to include immediate impacts, such as occupational injuries and accidents. Medium-term includes the time period from 1 to 100 years and long-term from 100 to 100,000 years. The limit of 100,000 years is arbitrary, however the most significant part of the impacts have been included.

The assessment of the impacts for a wide range of distances is not as problematic but must also be taken into account. It has been shown that the distance at which the evaluation stops can have a large influence on the final costs. The partitioning of the spatial scale (local, regional, global) was defined at 100 km and 1000 km from the point of release. The results for each category are mutually exclusive.

It is for these reasons that the impacts estimated for the nuclear fuel cycle are presented in a time and space matrix. This form of presentation of results ensures that all the important impacts have been assessed and allow comparison of results in the categories that are appropriate. It also can be made clear that the uncertainty of the results increases as the scope and generality of the assessment.

3.1.2. Uncertainty

The level of uncertainty associated with the results are due to the models, input data and the lack of information for some pathways. Each part of the methodology contributes to the uncertainty of the final results. For example, in most cases generalised transfer coefficients and assumptions have been taken. It is not possible to determine the uncertainty of the final results by simple opeartion on the uncertainties mentioned in this chapter. The estimates of uncertainty presented in the methodology are based on expert judgements and the range of possible input values. As a general rule, the longer the time span and/or the larger the region considered in the model, the larger the uncertainty in the model and the input data.

A detailed uncertainty analysis of the results have not been carried out but indications of the levels of uncertainty have been completed and are discussed in the results section.

3.2. Priority Pathways

For the radiological impacts to the public and environment, independent evaluations must be done for each radionuclide in each mode of radionuclide release or exposure. The pathway analysis methodology presented by a CEC DGXII project for the assessment of radiological impact of routine releases of radionuclides to the environment has been used. Different models were required to evaluate the impact of accidents. The damages to the general population (collective dose) are calculated based on assumptions for the average adult individuals in the population. Differences in age and sex have not been taken into account.

Atmospheric, liquid and sub-surface terrestrial releases have been treated as separate pathways. Due to the different physical and chemical characteristics of the radionuclides, each nuclide is modelled independently and an independent exposure of dose can be calculated. The dose assessment methodology allows for the summation of all the doses before application of the dose response coefficients.

If the data were available, the releases to the environment, or the source term, used in this assessment were the average annual releases based on the data from a number of past years for the specific facility. Otherwise more general information was utilised. It is assumed that the annual release occurs at a constant rate and is representative for the 30-year operational lifetime of the facility.

Occupational impacts, radiological and non-radiological are based on published personnel monitoring data and occupational accident statistics. There is no modelling done for this part of the evaluation.

The evaluation of potential transportation and severe reactor accidents are treated separately due to their probabilistic nature and the need to use a different type of atmospheric dispersion model.

The Figure 3 illustrates a generalised flow of contaminants in the environment. In all cases the environmental releases with potential public health impacts fall into the three major categories of (1) atmospheric discharge, (2) liquid discharge into a river or the sea, and (3) land based waste disposal. It is not possible to evaluate all the possible environmental pathways, therefore priority was given to the pathways that are the most significant sources of impacts.

These priority pathways can be modelled in varying degrees of complexity taking into account the particular radionuclide released, the physico-chemical forms of the release, the site-specific characteristics, and receptor-specific dose and response estimates. With validated models of the transfer of radionuclides in the environment, many nuclide-specific parameters have been determined. Generalised values applicable to European ecosystems have also been developed. In this project, models and parameter values developed in Europe (NRPB, 1994), US (Till and Meyer, 1983) and by international agencies (UNSCEAR, 1993; ICRP, 1974; ICRP, 1991) have been the basis for the input parameters used. Site-specific data have been used for population, meteorology, agricultural production and water use.

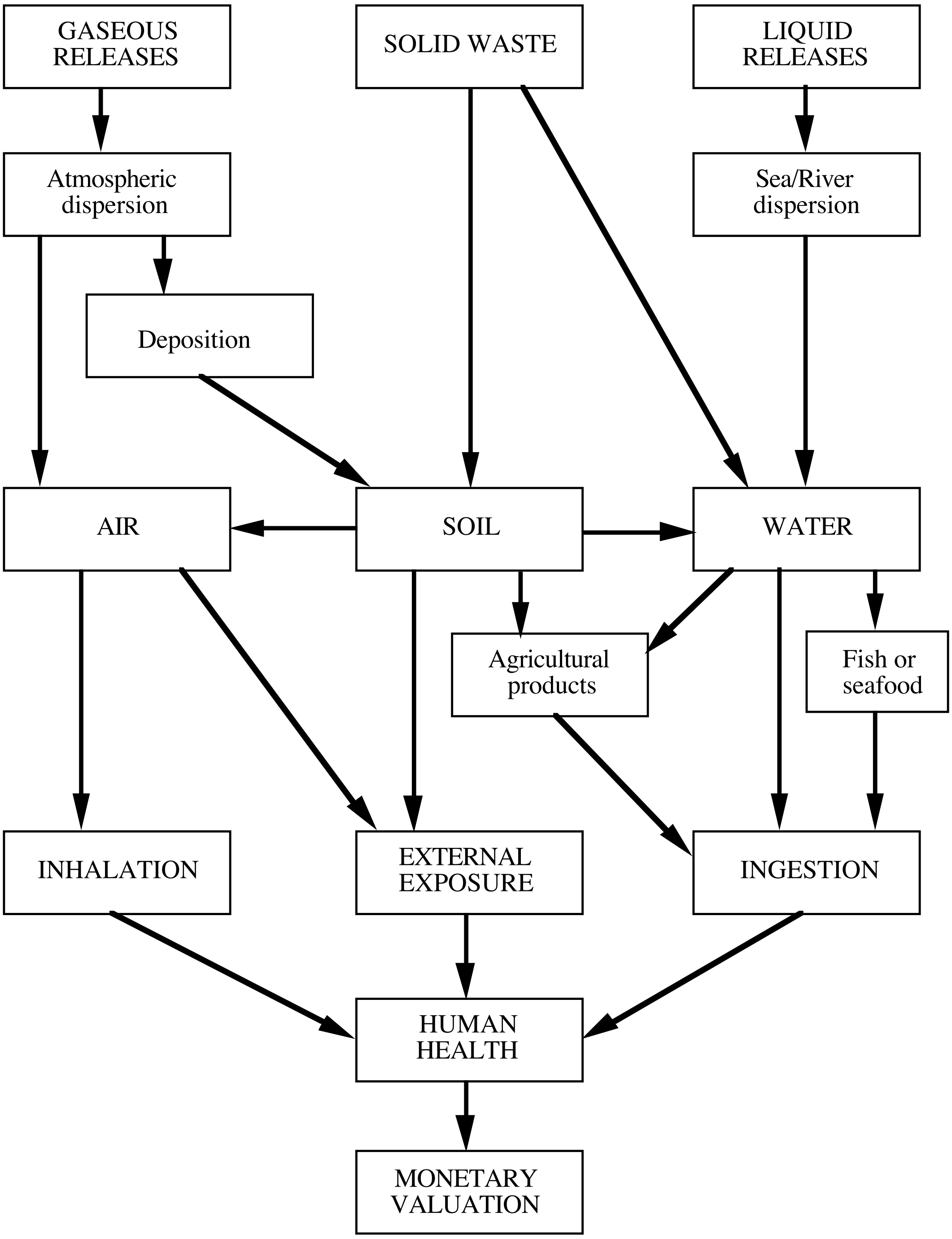

Figure 3. Impact pathways for the release of radioactivity in the environment

The result of the pathway analysis is an estimate of the amount of radioactivity (Bq) to which the population will be exposed converted to an effective whole body dose (Sv) using factors reported by the National Radiological Protection Board (NRPB, 1991). The method that has been applied does not accurately calculate individual doses or doses to individual organs of the body. It is intended to provide a best estimate of a collective dose (man.Sv) and an estimate of the expected health impacts as a result of those doses.

3.2.1. Impacts of Atmospheric Releases of Radionuclides on Public Health

The most important impact pathways for public health resulting from atmospheric releases are:

- inhalation and external exposure due to the radionuclides in the air,
- external exposure from ground deposition, and
- ingestion of contaminated food resulting from ground deposition.

3.2.1.1. Dispersion

Gaussian plume dispersion models are used for modelling the distribution of the atmospheric releases of radionuclides. Wind roses, developed from past measurements of the meteorological conditions at each site, represent the average annual conditions. This methodology is used for both the local and regional assessments. It is recognised that this is not the best method for an accurate analysis for a specific area, however, for the purpose of evaluating the collective dose on a local and regional level, it has been shown to be adequate (Kelly and Jones, 1985). For the global assessment, general boxes models for the dispersion of H-3, C-14, Kr-85 and I-129 have been used (IAEA, 1985).

3.2.1.2. Exposure

Inhalation doses to the population occur at the first passage of the "cloud" of radioactive material, and for the extremely long-lived, slow-depositing radionuclides (H-3, C-14, I-129), as they remain in the global air supply circulating the earth. Human exposure is estimated using the reference amount of air that is inhaled by the average adult (the "standard reference man" (ICRP 23) and nuclide-specific dose conversion factors for inhalation exposure in the local and regional areas (NRPB, 1991)).

External exposure results from immersion in the cloud at the time of its passage and exposure to the radionuclides that deposit on the ground. The immediate exposure of the cloud passage is calculated for the local and regional areas. The global doses for exposure to the cloud are calculated for I-129 and Kr-85. For external exposure due to deposition, the exposure begins at the time of deposition but the length of time that must be included in the assessment depends on the rate of decay and rate of migration away from the ground surface. For example, as the radionuclide moves down in the soil column, the exposure of the population decreases due to lower exposure rates at the surface. The time spent outdoors will also affect the calculated dose because buildings act as shields to the exposure and therefore diminish the exposure. This is a case where the conservative assumption that the population spends all the time outside is taken. The assessment of 100,000 years was considered to be a sufficient amount of time to include the major impacts from this pathway.

The human consumption pathway via agricultural products is due to direct deposition on the vegetation and migration of the radionuclides through the roots via the soil. Again, depending on the environmental and physical half-lives of each radionuclide, the time scale of importance varies but it is considered that 100,000 years take into consideration almost all the possible impacts from long-lived radionuclides.

In this assessment, a detailed environmental pathway model was not used. The environmental transfer factors between deposition and food concentration in different food categories, integrated over different time periods, assuming generalised European agricultural conditions was obtained from the NRPB agricultural pathway model FARMLAND. A constant annual deposition rate is assumed and the variation in the seasons of the year are not taken into account. The agricultural products are grouped, for this generalised methodology, as milk, beef, sheep, green and root vegetables and grains.

Cultivated vegetation is either consumed directly by people or by the animals which ultimately provide milk and meat to the population. The exposures received by the population are calculated taking into account food preparation techniques and delay time between harvest and consumption to account from some loss of radioactivity. An average food consumption rate for France and population size is used for calculating the amount of food that is consumed in the local, regional and global population. The collective doses are calculated assuming that the food will be consumed locally but if there is an excess it will pass to the regional population next, and afterwards, to the global population group. In this way the dose due to the total food supply produced within the 1000 km area included in the atmospheric dispersion assessment is taken into account, even if the exact location of consumption is unknown.

3.2.1.3. Dose Assessment

It is possible to report a calculated dose by radionuclide, type of exposure and organ of the body, but for the purpose of this project, an average individual whole body effective dose was used to calculate the collective (population) dose.

The relationship between the dose received and the expected radiological health impact is based on the information included in the recommendations of the International Commission on Radiological Protection (ICRP) in its publication 60 (ICRP, 1991). The factors, or dose response functions, used to predict the expected occurrence of cancer over a lifetime or severe hereditary effects in future generations per unit exposure received by the general public are 0.05 fatal cancers per man.Sv (unit of collective dose) and 0.01 severe hereditary effects in future generations per man.Sv. These factors assume a linear dose response function.

The fraction of cancers that would be expected to be non-fatal (0.12 non-fatal cancers per man.Sv) are calculated based on the expected number of fatal cancers and the lethality fractions reported for 9 categories of cancer in ICRP 60. This is reflected in the aggregated non-fatal cancer factor of 0.12 per man.Sv.

It is recognised that the dose-response functions that are chosen in the assessment of radiological health effects are extremely important. There is still controversy on the exact values to use and different models have been proposed. Within the context of this project, the internationally accepted factors have been used. The further details are presented in the main report of this project (Dreicer *et al*, 1995).

3.2.1.4. Time Distribution of the Expected Occurrence of Health Effects

The use of the dose response functions provides the estimate of the total number of health effects expected, however, the details on the expected time of occurrence of these effects has not been addressed. The deterministic health effects that occur after high doses of radiation (accidental releases) will occur in the short-term, but the distribution in time of the stochastic health effects is dependant on two factors: (1) the continued existence of radionuclides in the environment for years after deposition, and (2) the latency between exposure and occurrence of the effect.

The distribution of the total number of cancers is statistically predicted over the 100 years after 1 year of exposure, using data for the expected distributions of cancer in the average French population as a result of low-level radiation exposure (Figure 4). This curve is integrated over the 30-year operational lifetime of the facilities. After the shutdown of the facilities, except in the disposal stages, the releases do not continue and the level of radioactivity due to the releases will decrease dependant on their physical and environmental half-times.

Figure 4. **Relative frequency of occurrence of a radio-induced cancer for a French population after 1 year exposure**

For practical purposes in this assessment, the decreasing level of exposure is not integrated continuously through time but assumed to be constant in blocks of time of 0 - 1, 2 - 30, 30 - 50, 50 - 100, 100 - 200 and 200 - 100,000 years after the operational releases. The final accounting of potential cancers will range from the first year of release to 300 years into the future. This methodology may slightly underestimate the economic value of cancers due to the assumption of a constant exposure rate during the block time periods. In reality, the exposure will be greater at an earlier time in the block than at the end, however, the break down of time periods was chosen to minimise the difference. Estimates of the occurrence of severe hereditary effects during the next 12 generations were made using information presented in ICRP 60 (for further details, see the main report).

3.2.2. Impacts of Liquid Releases of Radionuclides on Public Health

Depending on the site of the facility, liquid waste is released into a river or the sea. The priority pathways for aquatic releases are the use of the water for drinking and irrigation, and the consumption of fish and other marine food products.

For the marine environment, the seafood and fish harvested for human consumption is the only priority pathway considered in this assessment. The other possible pathways involving the recreational use of the water and beaches do not contribute significantly to the population dose and have not been considered as a priority.

3.2.2.1. River

The dispersion of the releases in the river is modelled using a simple box model that assumes instantaneous mixing in each of the general sections of the river that have been defined. The upstream section becomes the source for the downstream section. River-specific characteristics, such as flow rate of water and sediments, transfer factors for water/sediments and water/fish, are needed for each section. The human use factors such as irrigation, water treatment and consumption, and fish consumption must also be taken into consideration.

The deposition of the radionuclides in the irrigation water to the surface of the soil and the transfer to the agricultural produce is assumed to be the same as for atmospheric deposition.

The ingestion pathway doses are calculated in the same way as described in the previous section on the atmospheric pathway. For aquatic releases, it is difficult to calculate independent local and regional collective doses without creating extremely simplified and probably incorrect food distribution scenarios. Therefore, the local and regional collective doses are reported in the regional category.

The estimation of health effects follows the same methodology as described in the section above.

3.2.2.2. Sea

To evaluate the collective dose due to consumption of seafood and marine fish, a compartment model which divides the northern European waters into 34 sections has been used. This model takes into account volume interchanges between compartments, sedimentation, and the radionuclide transfer factors between the water, sediments, fish, molluscs, crustaceans, and algae, and the tons of fish, molluscs, crustaceans and algae harvested for consumption from each compartment. For the regional collective dose, it is assumed that the edible portion of the food harvested in the northern European waters is consumed by the European population before any surplus is exported globally. Due to the difficulty in making assumptions for the local consumption, the local collective dose is included in the regional results. Marine dispersion at the global scale of H-3 and I-129 has been done using general global dispersion models (IAEA, 1985).

The risk estimates of this pathway use the same methodology as the other pathways.

3.2.3. Public Health Impacts of Releases of Radionuclides from Radioactive Waste Disposal Sites

The land-based facilities designed for the disposal of radioactive waste, whether for low level waste or high level waste, are designed to provide multiple barriers of containment for a time period considered reasonable relative to the half-life of the waste. It is assumed that with the normal evolution of the site with time, the main exposure pathway for the general public will be the use of contaminated ground water for drinking or irrigation of agricultural products.

The leakage rate and geologic transport of the waste must be modelled for the specific facility and the specific site. The global doses due to the total release of H-3, C-14 and I-129 are estimated assuming that ultimately the total inventory of wastes are released into the sub-surface environment under liquid form and using the same global dispersion models as for the liquid releases of H-3 and I-129 to the ocean (section 3.2.2.2.). As is done for the other pathways, it is assumed that the local population and their habits remain the same for the 100,000-year time period under consideration for the disposal sites. This time limit takes into account disposal of all the radionuclides except long-lived I-129.

3.2.4. Impacts of Accidental Atmospheric Releases of Radionuclides

The methodology used to evaluate impacts due to accidental releases was risk-based expected damages. Risk is defined as the summation of the probability of the occurrence of a scenario (P_i) leading to an accident multiplied by the consequences resulting from that accident (C_i) over all possible scenarios. This can be simply represented by the following equation:

$$\textbf{Risk} = \sum P_i \cdot C_i$$

3.2.4.1. Transportation Accidents

In the analysis of transportation accidents, a simple probabilistic assessment has been carried out. It had not been possible to evaluate all possible scenarios for the accident assessments but a representative range of scenarios, including worst case accident scenarios, have been included. The type of material transported, the distance and route taken by the train or truck, the probability of the accident given the type of transportation, probability of breach of containment given the container type, the probability of the different type of releases (resulting in different source terms) and the different possible weather conditions are taken into account. The site of the accident can play a key role in the local impacts that result so variation in the population and their geographic distribution along the transportation routes is considered.

The atmospheric dispersion of the release is modelled using a Gaussian plume puff model. The toxicological effects of the releases (specifically UF_6) are estimated using the LD_{50} (lethal dose for 50% of the exposed population) to estimate the number of expected deaths and a dose-response function for injuries due to the chemical exposure. The radiological impacts are estimated with the same methodology described for atmospheric impact pathway. The expected number of non-radiological impacts, such as death and physical injury due to the impact of the accident is estimated. These types of impacts are also calculated for the transportation of non-radioactive materials (concrete, steel) involved in the construction and decommissioning of the reactor.

The general methodology for the assessment of transportation risks is presented in Figure 5. The single cost value presented takes into account transportation between all the stages of the fuel cycle. The cost of clean-up and countermeasures that might have to be taken, have not been included.

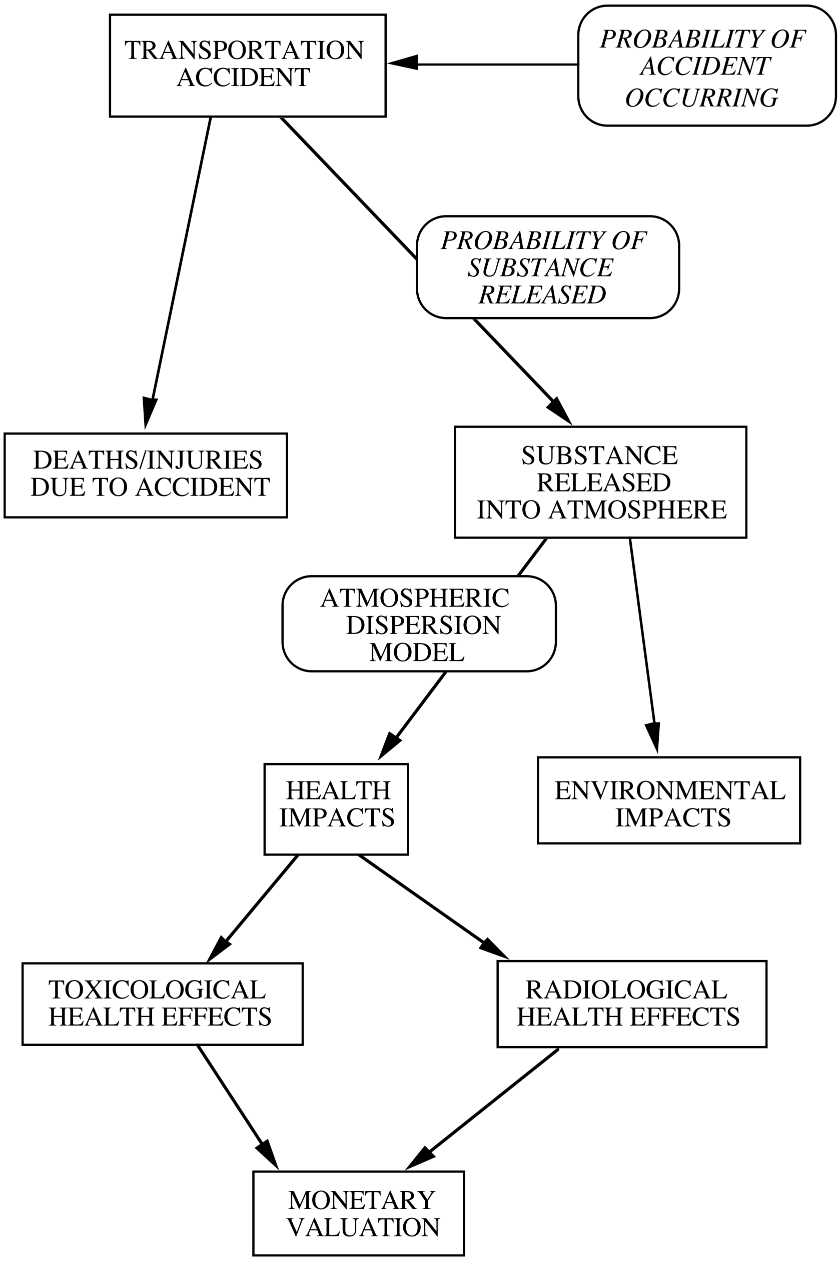

Figure 5. Pathway for the assessment of a transportation accident

3.2.4.2. Severe Reactor Accidents

A comprehensive probabilistic safety assessment (PSA) of potential reactor accidents does not fall within the scope of this project. In addition, the detailed data on potential source terms and associated probabilities for a multitude of potential scenarios for French nuclear power plants are not available. As result, four hypothetical scenarios were evaluated in order to demonstrate the range of results using a risk-based assessment methodology. The scenarios were assumed to take place at a hypothetical power plant in the centre of western Europe.

The more modern 1300 MWe reactors are considered to have a lower probability of occurrence for a core melt accident than the older 900 MWe models; so in this study a probability of 1E-5 per reactor-year was used (EDF, 1990). This is smaller than the estimated value of the NRC (NRC, 1990) but significantly higher than most of the probability values considered to be correct for a present-day European reactor (Wheeler and Hewison, 1994).

The magnitude and characteristics of radioactive material that can be released following a core melt will depend, *inter alia*, on the performance of the containment and its related safety systems. If the containment suffers massive failure or is bypassed, a substantial fraction of the volatile content of the core may be released to the environment, if the containment remains intact the release will be very small. For the purposes of this indicative assessment, it is assumed that the probability of massive containment failure or bypass conditional upon a core melt is 0.19, and the probability of the containment remaining intact is 0.81 (NRC, 1990). The same assumptions were made for the 900 MWe PWR assessment (Dreicer *et al*, 1995).

The reference scenario ST21 for France assumes a containment failure that results in the total release of about 1% of the core on the average (10% of noble gases from the core, 1% of the more volatile elements, such as caesium and iodine and smaller percentages of other elements). This source term is in the same order of magnitude as the reference accident scenario considered by the French national safety authorities (S3) (Queniart *et al*, 1994). The three other source terms evaluated to illustrate the sensitivity of the results are respectively 10 times larger than ST21 (ST2, massive containment failure with 10% of the core released) and 10 and 100 times lower than ST21 (ST22 and ST23).

The public health impacts and economic consequences of the releases were estimated using the EC software COSYMA (Ehrhardt and Jones, 1991). One hundred and forty-four different meteorological scenarios were statistically sampled to predict the dispersion of the releases. Due to the introduction of countermeasures for the protection of the public, the impact pathway must be altered, as is shown in Figure 6. The priority atmospheric release pathways, for local and regional areas out to 3,000 km from the site were assessed. Unfortunately the definition of time and space boundaries are not the same as those defined in the assessment of routine operations of the fuel cycle, so the results will be presented separately.

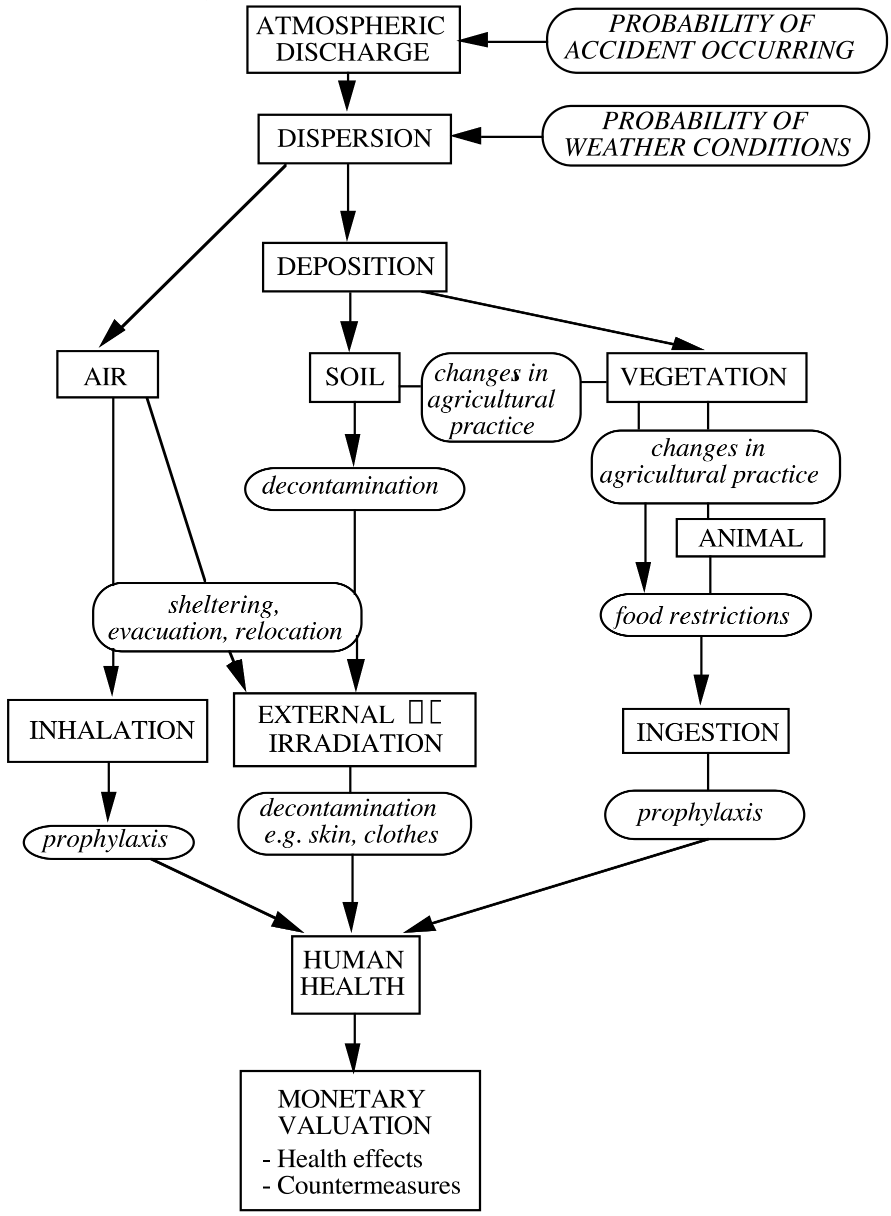

Figure 6. Pathways for a severe accidental release

The monetary valuation of the health effects arising from the collective dose is completed in the same manner for all the other parts of the assessment. The additional costs from the implementation of the countermeasures and the agricultural losses were calculated by COSYMA using estimates of the market costs.

The use of this type of methodology does not necessarily include all the social costs that might result after a severe nuclear accident. One important issue is the social costs of risk aversion. Further work is required before the external costs of a severe accident can be considered complete.

3.2.5. Occupational Impacts

The assessment of the radiological and non-radiological occupational impacts are very straightforward because modelling is not required for this pathway. The radiation protection of the workers requires direct monitoring and reporting of the doses received by the workers at each facility thereby providing measured data for the radiological impacts, except for waste disposal where the estimates of occupational doses were taken from an UK study (Ball *et al*, 1994).

The relationship between the dose received by the occupational population and the radiological health impacts are based on international recommendations of ICRP 60 (ICRP, 1991). The factors, or dose-response functions, used to predict the expected occurrence of cancer over a lifetime or severe hereditary effects in future generations per unit exposure received by the workers are 0.04 fatal cancers per man.Sv and 0.006 severe hereditary effects in future generations per man.Sv.

The fraction of cancers that would be expected to be non-fatal are calculated based the expected number of fatal cancers and the lethality fractions in the worker population reported for 9 categories of cancer reported in ICRP 60. The different age and sex distributions found in the working population compared to the general public slightly change the expected occurrence of disease of all three types of health impacts. This is not easily seen in the aggregated non-fatal cancer factor of 0.12 per man.Sv because the public and worker values are rounded. The methodology for the estimating the latency time before occurrence of the health effects has been presented earlier.

Non-radiological worker accidents data are obtained from the specific facility. If insufficient data are available for a representative value, the national accidents statistics (CNAM, 1991) reported by type of job are utilised. When it was not possible to find the data for a specific nuclear facility, the data for the chemical industry were used.

For the construction and the decommissioning of the reactor, the workforce is calculated based on the construction and decommissioning costs (respectively 910 ECU and 140 ECU per kW installed) associated with the worker productivity in each industrial branch involved in the works, and normalised by the electricity production of the plant over its expected lifetime (30 years).

3.2.6. Impacts of Transportation on Human Health

The priority impact pathway from accident-free transportation operations in the nuclear fuel cycle is external exposure from the vehicle containing the radioactive material. This has been estimated using a computer code by the International Atomic Energy Agency (INTERTRAN) which takes into account the content of the material transported, the type of container, mode of transport (road or rail), the distance travelled, and the number of vehicle stops at public rest stations along the highway (for road transportation).

3.3. Monetary Valuation

The results of the human health impact analysis are reported as deaths, fatal cancers, non-fatal cancers, severe hereditary effects, occupational accidental deaths and injuries (working-days-lost and permanent disabilities). The value of a statistical life (VSL = 2.6 MECU) is used for deaths, accidental deaths and fatal cancers. This is the same value that is applied in all the EC fuel cycles being assessed in the ExternE project (CEC, 1995).

Being that no values for the willingness-to-pay to avoid non-fatal cancers have been identified for France, the values reported in the US Nuclear Fuel Cycle report (ORNL, 1993) for direct (hospital, physician, drugs, etc.) and indirect (forgone earnings discounted at 6%) cost of different types of cancers have been used. Since the total average costs per cancer differ by about a factor of 2, it has been considered adequate to take the rounded value of a simple average of the total costs as the value of a non-fatal cancer, 0.25 MECU.

In the valuation for the nuclear fuel cycle, it will be assumed that the severity of a hereditary effect merits the same valuation as the VSL (2.6 MECU). The discounting of severe hereditary effects costs is much more complex because none of the impacts are seen during the lifetime of the population that is exposed. It is assumed that 15% of the cases that may occur are expected to be seen during the first generation, 12% during the second generation and the remaining 73% sometime in the future (ICRP, 1991). For the purpose of applying the 3 and 10% discount rates, the 73% remaining impacts are assumed to occur during the next 10 generations at a constant rate (7.3% per generation out to a total of 12 generations after exposure). The modified VSL-severe hereditary effect values for the 3% and 10% discount rates are 0.296 MECU and 0.039 MECU, respectively (Dreicer *et al*, 1995).

Non-radiological Health Effects

At this time, no willingness-to-pay based values are available in France for the non-radiological impacts, so estimates were made based on the available information. The cost that the national health insurance system pays for working-days-lost (WDL) and permanent disabilities (PD) for job-related accidents were used for this valuation (Dreicer *et al*, 1995; CNAM, 1991).

The value of one WDL is calculated based on the average cost of an accident with WDL (10,187 francs) and the average duration of days lost (35 days). Forty-five percent of this average cost of 291 francs per day is due to the cost of illness and 55% is compensation paid. The compensation cost is considered, at this time, to be equivalent to a value based on willingness-to-pay. The 160 francs per WDL is equal to 65 ECU.

The value assigned to PD is 19,000 ECU. The data available from CNAM, based on three years of data from 1988 to 1990, is:

	Deaths	PD > 10 %	PD < 10 %
Number per year	1240	18,600	48,200
Average cost per effect	2.25 MFrancs	292,573 Francs *	8775
Total cost per year	8230 MFrancs		423 MFrancs

* assuming that:
total cost of (PD>10% + death) = (# of PD>10% * cost of PD>10%) + (# of deaths * cost of a death)

Based on this, the cost of PD > 10% is estimated to be 292,573 francs. To have one general cost for PD, it is assumed that the ratio of the number of PD in the two categories is constant and that 72% of the PD are in the less serious category. The final PD value is weighted to take this into account to arrive at a cost of 88,239 francs (or 19,000 ECU).

Injuries from traffic accidents are valued at 15,000 ECU, based on statistics from the insurance company payments (Dreicer *et al*, 1995). Further work is needed to determine if the use of these values provides the external costs due to these physical impacts. The US team used a "revealed preference" analysis using hedonic wage techniques (ORNL, 1993). An industry average for the electricity generation sector of a value of a statistical injury (VSI) of 10,300 US$ (range of 8,000 - 34,000 US$) was used. It is assumed that in the mining industry, where the activities are more dangerous, the best estimate of a VSI would be 21,000 US$. The values used in the EC assessment can be generally considered to be the same within an order of magnitude. The discounting of these health effects costs are based on the expected distribution of occurrence through time.

4. RESULTS

4.1. Radiological Impacts

4.1.1. Routine Operation of the Nuclear Fuel Cycle

The total collective dose calculated for both the general public and workers, integrated for a time period of 100,000 years into the future, is 13.1 man.Sv/TWh taking into account all the stages of the nuclear fuel cycle. Over 97% of this dose is due to public exposures. The distribution of the public and occupational collective doses is presented in Table 13. The value for the electricity generation stage (public and occupational) is the average of the 5 sites considered; a more detailed presentation of the results for each site is presented a little later in this section.

A closer look shows that the total estimated local public collective dose is about 0.22 man.Sv/TWh and the total regional collective dose is 0.33 man.Sv/TWh, leaving over 95% of the public dose due to the global dispersion of certain radionuclides (as C-14 and I-129). This global dose contributes more than 93% of the total collective dose for the fuel cycle, as is illustrated in Figure 7.

If the global doses assessment is not taken into consideration, the occupational doses contribute over 38 % of the doses received (Figure 8).

The reprocessing stage is the most important contributor to the total collective dose (including the global impact and the occupational doses), with 79% of the total dose (10.3 man.Sv/TWh). The second contributor is the electricity generation stage (18% of the total collective dose). But, if only the local and regional collective doses are considered (including occupational), the most important contributors become the electricity generation (0.38 man.Sv/TWh) and the mining and milling (0.29 man.Sv/TWh) stages. It can be seen that the enrichment stage is the least important.

Table 13. Collective doses for the different stages of the fuel cycle

Collective doses (man.Sv/TWh)	Public local	Public regional	Public global	Public total	Occupational	Total
Mining and milling	8.50E-02	9.17E-02	1.05E-04	1.77E-01	1.12E-01	2.89E-01
Conversion	2.40E-05	1.00E-05	9.53E-07	3.50E-05	2.29E-03	2.32E-03
Enrichment	2.22E-05	4.27E-06	3.90E-07	2.68E-05	8.33E-06	3.52E-05
Fuel fabrication	3.50E-07	8.86E-06	5.18E-09	9.21E-06	7.14E-03	7.15E-03
Reactor construction	0	0	0	0	0	0
Electricity * generation 1300 MWe	1.42E-03	1.78E-01	1.98E+00	2.16E+00	2.02E-01	2.36
Decommissioning	1.45E-04	0	0	1.45E-04	2.16E-02	2.17E-02
Reprocessing	2.04E-04	6.07E-02	1.02E+01	1.03E+01	1.76E-03	1.03E+1
LLW disposal	1.27E-05	-	2.57E-02	2.57E-02	1.00E-04	2.58E-02
HLW disposal	1.36E-01	-	-	1.36E-01	6.00E-07	1.36E-01
Transportation	9.50E-04	0	0	9.50E-04	1.14E-03	2.09E-03
Total	2.24E-01	3.30E-01	1.22E+01	1.28E+01	3.48E-01	1.31E+01

* without major accident, average of the 5 sites

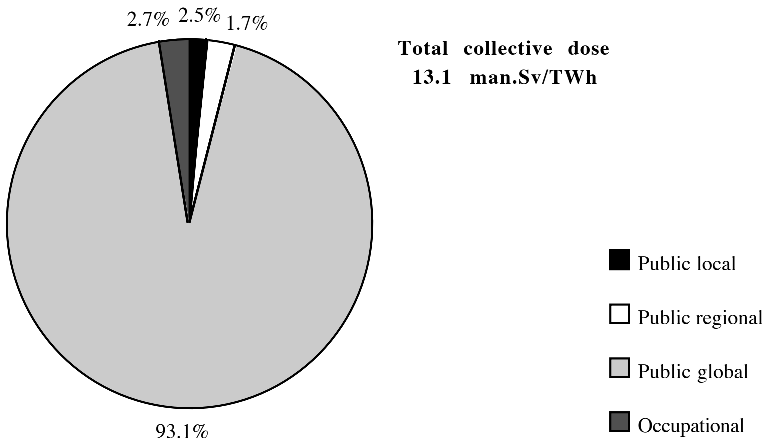

Figure 7. Distribution of the total collective dose

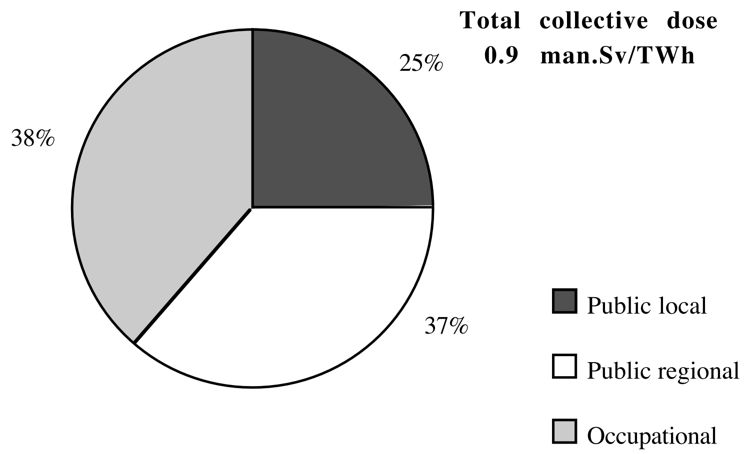

Figure 8. Distribution of the collective dose without global assessment

For the workers, the total collective dose for all the different stages of the fuel cycle is about 0.35 man.Sv/TWh. The electricity generation and the mining and milling stages are the operations where the occupational collective dose is the most important (0.20 man.Sv/TWh and 0.11 man.Sv/TWh, respectively). It is estimated that the workers of the HLW disposal stage will be the least exposed group of workers in the fuel cycle, with 6E-7 man.Sv/TWh. However, this estimate is not based on actual data, being that such a facility does not exist at this time.

The collective dose is calculated taking into account the various exposure pathways resulting from the liquid and atmospheric releases (i.e. inhalation, ingestion, external exposure). The contribution of each pathway to the total is very dependent on the radionuclide considered. Table 14 presents the results organised by type of exposure pathway.

For the public local and regional categories, it appears that the impacts of the atmospheric releases are two times lower than the impacts of the liquid radioactive releases: 0.19 man.Sv/TWh for the atmospheric discharges and 0.37 man.Sv/TWh for the liquid discharges.

Table 14. Distribution of the collective doses by pathways (man.Sv/TWh)

man.Sv/TWh	Local		Public local + regional (2)		Public global (2)	
	direct (1) exposure	occupational	atmospheric	liquid	from food products (3)	C-14, I-129, Kr-85, H-3
Mining and milling	0	1.12E-01	1.77E-01	1.54E-05	1.05E-04	0
Conversion	0	2.29E-03	3.35E-05	4.56E-07	9.53E-07	0
Enrichment	0	8.33E-06	2.63E-05	9.33E-08	3.90E-07	0
Fuel fabrication	0	7.14E-03	4.16E-07	8.79E-06	5.18E-09	0
Reactor construction	0	0	0	0	0	0
Electricity (4) generation 1300 MWe	0	2.02E-01	1.95E-03	1.77E-01	1.39E-05	1.98E+00
Decommissioning	1.45E-04	2.16E-02	-	-	0	0
Reprocessing	0	1.76E-03	8.42E-03	5.25E-02	2.75E-03	1.02E+01
LLW disposal	0	1.00E-04	0	1.27E-05	0	2.57E-02
HLW disposal	0	6.00E-07	0	1.37E-01	-	-
Transportation	9.50E-04	1.14E-03	0	0	0	0
Total	1.10E-03	3.48E-01	1.87E-01	3.67E-01	2.87E-03	1.22E+01

(1) direct exposure to the public (not due to releases)
(2) due to radioactive releases to the environment
(3) due to food products produced within the regional area but consumed elsewhere
(4) average value of 5 different sites

On a global scale, C-14 contributes the largest portion of the dose (about 12 man.Sv/TWh). It must be stressed that even though this radionuclide is responsible of more than 90% of the total collective dose, its global impact is based on a very large time and space scale (100,000 years for an assumed constant global population of 10 billion people).

The release of C-14 for the whole nuclear fuel cycle represented in this report is about 8.4E4 MBq per TWh. At about 10 years after the release, the high end of the range of individual dose rates (2.3E-17 Sv/y per MBq released (IAEA, 1985)) was used to estimate an average individual dose of 2E-9 mSv/y per TWh. Based on these figures, if one additional 1300 MW reactor were to be added to the nuclear park in France (an addition of about 7 TWh), 1.4E-8 mSv/y would be added to the average annual natural background dose of 2.4 mSv/y (1.2E-2 mSv/y of this is due to naturally-occurring C-14 (UNSCEAR, 1993)).

The radiological health effects resulting from the routine operation of the nuclear fuel cycle are directly proportional to the total collective doses. The expected number of health effects were calculated assuming no lower threshold for radiological impacts, using internationally accepted data from ICRP 60 (ICRP, 1991). Normalised by energy production the total number of expected health impacts are: 0.65 fatal cancers/TWh, 1.57 non-fatal cancers/TWh, and 0.13 severe hereditary effects/TWh.

These results include the global dose assessment estimates out to 100,000 years. Most of these impacts would be expected to occur in the public domain.

To provide a perspective, the number of deaths for the public in the local and regional areas can be multiplied by the average production of a 1300 MWe reactor in one year (about 7 TWh/y). In this case, in Europe over the next 100,000 years, less than one fatal-cancer can be expected to occur as a results of the releases of one year of operation of a reactor. This can be compared to the approximate value of 800,000 fatal cancers that are reported in Europe each year.

Five independent site-specific assessments were completed for the electricity generation stage of the nuclear fuel cycle (Table 15). The local public doses vary by about an order of magnitude, from 2.50E-4 to 2.80E-3 man.Sv/TWh due to the different releases from the facilities and the variation of the population (Table 3). The variation in the regional public doses at each site is from 3.4E-4 to 8.37E-1 man.Sv/TWh are even more dramatic. This is due to the aggregation of small average individual doses in large population centers in the prevailing wind direction.

A review of the characteristics of these sites provides some further explanations. The local and regional doses around Flamanville are the lowest doses of the 5 sites, because Flamanville is located on a piece of land jutting into the English Channel, in a rural area, and so the local population is very low compared to the other sites. Paluel is close to Flamanville, with the same kind of characteristics, and the differences existing between the doses of these two sites can be attributed to the releases of Paluel, which are larger on the average, especially the liquid Cs-137 and Xe-133 atmospheric releases, important in the dose evaluations.

The total collective dose to the public fluctuates very close to 2 man.Sv/TWh. The occupational doses vary a bit more but are still well within a factor of 2. The individual site values are so close that there is no problem in using the arithmetic average of 2.36 man.Sv/TWh to represent the collective dose from this stage of the fuel cycle.

Table 15. Collective doses for five 1300 MWe PWR's in France

Collective doses (man.Sv/TWh)	Public local	Public regional	Public global	Public total	Occupational	Total
Belleville	9.99E-04	2.77E-02	1.98	2.01	1.62E-01	2.17
Flamanville	2.50E-04	3.40E-04	1.98	1.98	1.87E-01	2.17
Nogent	2.80E-03	8.37E-01	1.98	2.82	1.59E-01	2.98
Paluel	1.06E-03	8.09E-04	1.98	1.98	2.88E-01	2.27
Saint-Alban	2.01E-03	2.26E-02	1.98	2.00	2.14E-01	2.22
Average	1.42E-03	1.78E-01	1.98	2.16	2.02E-01	2.36

The distribution of the doses by different environmental pathways are presented on Table 16. The variation of the doses due to the atmospheric releases illustrates the dependence on the variation in the population around the sites. Nogent has the largest local population and Flamanville has the smallest.

The liquid pathways are generally more significant than the atmospheric pathways, particularly when the reactor is located on a river, due to the use of water for consumption and irrigation of agricultural products. The doses from the liquid releases are the lowest for Flamanville and Paluel because the liquid effluents release into the sea are greatly diluted and not directly consumed by the population. Whereas a site such as Nogent, located on the Seine River, must consider the pathway of drinking water for the population of Paris. The larger result in this case is again due to the aggregation of low levels of radioactivity in the drinking water, and the resulting very small doses, over a large population group.

Table 16. Distribution of the collective doses by pathways (man.Sv/TWh)

Collective doses (man.Sv/TWh)	Local		Public local + regional		Public global	
	direct exposure	occupational	atmospheric	liquid	from food products	C-14, H-3 Kr-85
Belleville	0	1.62E-01	1.50E-03	2.72E-02	5.09E-05	1.98E+00
Flamanville	0	1.87E-01	4.10E-04	1.80E-04	8.20E-06	1.98E+00
Nogent	0	1.59E-01	3.71E-03	8.36E-01	4.66E-06	1.98E+00
Paluel	0	2.88E-01	1.66E-03	2.05E-04	2.81E-06	1.98E+00
Saint-Alban	0	2.14E-01	2.49E-03	2.21E-02	2.85E-06	1.98E+00
Average	0	2.02E-01	1.95E-03	1.77E-01	1.39E-05	1.98E+00

For the public global doses, the dose due to ingestion of food products (2.85E-6 to 5.09E-5 man.Sv/TWh) is insignificant compared to the main pathway for the atmospheric dispersion of C-14, H-3 and the Kr-85 (1.98 man.Sv/TWh). A same total public global dose of 1.98 man.Sv/TWh is reported for each site because the same estimated source term was used. This pathway represents more than 90% of the total public dose impact of each site, except for Nogent, due to the increased regional collective dose at this site.

4.1.2. Accidental Situations

4.1.2.1. Transportation

Due to the very low probabilities of occurrence of accidents for the transportation of radioactive materials, the collective dose that results (in term of expected risk) is less than 1E-7 man.Sv/TWh. This value is very small compared to the already low routine operation public average collective dose of 9.5E-4 man.Sv/TWh.

4.1.2.2. Reactor Accidents

In case of severe reactor accidents, an indicative total collective dose for the population (for a radius of 3,000 km) for the four accident scenarios has been estimated. Table 17 presents the collective doses for the scenarios. The expected risk varies between 0.08 and 0.001 man.Sv/TWh. The impact of the reference scenario ST21 is a regional collective dose of about 58,000 man.Sv.

Table 17. Expected collective doses for a major reactor accident (ST21: reference scenario for France)

Source term	Core melt probability (per reactor.year)	Conditional probability	Collective dose (man.Sv)	Collective dose x probability (man.Sv per reactor.year)	Risk man.Sv/TWh
ST2	1E-05	0.19	291,200	0.55	0.078
ST21	**1E-05**	**0.19**	**58,300**	**0.11**	**0.016**
ST22	1E-05	0.19	12,180	0.02	0.003
ST23	1E-05	0.81	1,840	0.01	0.001

The health effects from reactor accidents can be divided in two categories: the immediate health effects (deterministic effects) and the stochastic effects as cancers or hereditary effects. For the four accident scenarios considered in this study, only the two most severe accidents lead to deterministic effects, but no deaths are expected for the reference scenario. For the stochastic effects, as for routine operation, they are considered to be directly proportional to the collective doses. Depending on the scenario, the number of expected fatal cancers varies from 3.9E-3 to 1E-4 per TWh.

4.2. Non-radiological Impacts

The Table 18 presents the overview of the non-radiological impacts for the French nuclear fuel cycle. It is estimated that the production of 1 TWh will result in 0.018 deaths, 0.96 permanent disabilities and 296 working-days-lost (non-radiological health impacts) in the worker force for the nuclear industry. The construction and the decommissioning of the reactor are the most important contributors to these values. The Table 19 presents the detailed expected non-radiological impacts (occupational accidents) for the 5 electricity generation sites per TWh.

Traffic accidents involving the general public may result from the transportation of the radioactive materials between the different sites and the transportation of the materials involved in the construction and the decommissioning of the reactor. The expected number of health impacts has been estimated as 3E-4 deaths and 1.7E-3 injuries per TWh from all transportation included in this assessment.

If one is to estimate the number of deaths due to the transportation of material needed to run a 1300 MWe PWR for 1 year at an annual production rate of 7 TWh, a value of 0.002 deaths/year would be found. This value is insignificant, especially when compared to the nearly 10,000 deaths by road accidents occurring each year in France (INSERM, 1991). In accidental situations occurring during the transportation of hazardous radioactive materials such as UF_6, the toxicological health impacts estimated are even smaller (2E-9 deaths/TWh and 7E-5 injuries/TWh).

Table 18. Expected non-radiological impacts for the different stages of the French nuclear fuel cycle per TWh

	Number of deaths		Permanent disabilities	Transportation injuries	Working-days-lost
	occupa-tional	public	occupa-tional	occupational + public	occupa-tional
Mining and milling	3.8E-03	0	1.51E-01	0	32.2
Conversion	1.3E-04	0	7.9E-03	0	2.1
Enrichment	2.5E-04	0	1.47E-02	0	3.86
Fuel fabrication	1.71E-04	0	1.04E-02	0	2.72
Reactor construction	7.20E-03	9.9E-05	5.27E-01	6.3E-04	162
Electricity generation (1)	2.80E-03	0	-	0	14.33
Decommissioning	3.39E-03	9.9E-05	2.11E-01	6.3E-04	69
Reprocessing	6.1E-04	0	3.76E-02	0	9.87
LLW disposal	0	0	0	0	0
HLW disposal	0	0	0	0	0
Transportation (1)	(2)	1.15E-04	-	5.1E-04	-
Total	1.84E-02	3.13E-04	9.60E-01	1.77E-03	296.1

(1) average for the 5 sites
(2) included in public

Table 19. Expected non-radiological impacts (occupational accidents) for the 5 electricity generation sites per TWh

	Expected number of deaths	Expected number of working-days-lost
Belleville	2.87E-03	15.69
Flamanville	2.79E-03	22.06
Nogent	2.80E-03	7.35
Paluel	2.53E-03	15.27
Saint-Alban	3.00E-03	11.26
Average	2.80E-03	14.33

4.3. Environmental Impacts

4.3.1. Increase Levels of Ionising Radiation

4.3.1.1. Normal Operation

The impact from chronic low-levels of ionising radiation (corresponding to the routine operation of the facilities) can be investigated on the cellular, organism and population levels. Radiosensitivity of individuals and populations varies between terrestrial and aquatic species as well as between the various levels of species within the kingdom. It has been concluded, based on the review of a selection of data, that chronic doses of 1 mGy/d or less for terrestrial animals and 10 mGy/d for terrestrial plants and aquatic populations do not appear to cause observable changes in wild populations (IAEA, 1992; Warmer *et al*, 1993). The event of an acute dose level of 0.1 Gy or less was considered to be unlikely to produce persistent measurable deleterious changes in populations or communities of terrestrial plants or animals.

A preliminary assessment of the highest releases of the fuel cycle presented in this report has shown that the dose levels that could be estimated in the environment are below levels that have been indicated above. The maximum dose rates induced by the deposition on the ground, or by the dispersion in the aquatic environment of the radionuclides released from the reference nuclear facilities are estimated to be about 1E-3 mGy/day.

Environmental surveillance around nuclear facilities measures the incremental increase of radionuclides around the site. Under routine conditions, no environmental impacts have been reported. From the information cited and the lack of observed evidence, it can be concluded that no impacts on species can be estimated from the routine releases of radionuclides from the nuclear fuel cycle characterised in this assessment.

4.3.1.2. Accidental Situations

In the event of a major accident, where large amount of radioactive material may be released into the environment, there would, of course, be greater impacts. Two types of impacts need to be considered, those with direct economic impacts (such as the loss of land and land use which has been included in the consequence analysis of the accident), and impacts on the populations of the existing ecosystems. The economic impacts have been addressed in reference (Dreicer *et al*, 1995).

A Russian study on the effects on radioactive fallout on soil animal population in the condemned 30 km zone around the Chernobyl plant has shown that after 2-2.5 years, marked differences between population in the contaminated and control areas were no longer found (Krivolutzkii, 1992). However, after the accident, the contamination level led to an important decrease in mesofauna populations. The absorbed doses, 3 km from the power plant 2 months after the accident, were about 30 Gy, far exceeding the threshold for acute effects of 0.1 Gy, mentioned above. The main effects were seen in the reproductive processes of soil fauna but were not detected at distances greater than 30 km from the power plant.

After the Chernobyl accident, the effects were readily apparent in the pine trees close to the plant and less dramatic effects were seen in animals. However, it has been argued that it is debatable whether the long-term survival of the populations have been put at serious risk (Warmer *et al*, 1993).

In the accident analysis presented in this study, these kinds of environmental effects have not been included in the impact estimates. If indeed there are no long-term population impacts outside the condemned zone, it can be concluded that additional impacts (other than the loss of the condemned area) must not be included.

4.3.2. Releases of Heated Water

The release of heated water into the rivers and the sea have effects on the physical and chemical characteristics of the water, which in turn effect the life-forms dependent on the water. Impacts on individuals can be seen in the area close to the point of release. The indicators of the damages for aquatic populations and the ecosystems are considered in terms of species diversity, ecosystem stability, eutrophication, disease and host-parasite relationships and additive effects on the toxicity of different pollutants (IAEA, 1974).

It is difficult to clearly estimate the incremental temperature increase in the complex river and marine aquatic environments where these releases occur in the nuclear fuel cycle. Therefore it is not possible to establish a temperature increase over which populations and ecosystems would be affected. However, some observations can be cited.

In general, the changes in species diversity in temperate regions of the world show a decrease in species diversity near the release point, especially during the warmer summer months. In colder climates, the species diversity has been shown to increase in the region of higher temperatures. Indeed, there is also the possibility that some of the species prefer the warmer temperatures. In these cases the population thrives and there is increased migration into these areas. This could be considered as a benefit if the change in species diversity due to thermal releases does not de-stabilise the ecosystem. It has been shown that the stability of the populations in the areas with the greatest changes in temperature can be affected, but no evidence has been shown (due to limited research) that suggests that the whole body of water is affected (IAEA, 1974).

The increase in temperature may affect the development of pathogens as well as the resistance to the aquatic organisms to disease. For example, the parasites may develop twice as fast and the hosts will therefore suffer at a greater rate. However, there have also been other studies that show that fish populations develop an improved immunity under conditions of elevated temperature (IAEA, 1974). There is also the possibility of the transfer of the pathogens to other terrestrial based organisms, but there has been little research in this field.

Impacts from the release of heated water are expected to be quite localised as a result of a relatively rapid dilution in the sea, estuary or river. It is unlikely that an impact on the health of the total population would be seen, unless, either a large territory of the population is impacted, or the release occurs at an extremely sensitive area. In special conditions, where certain other pollutants exist, there is the possibility that the increase in temperature may cause an increase in the toxic effect. This synergistic situation would be very specific to the site, type of pollutant, and the species under consideration.

The impacts of releases of heated water have been studied around some of the nuclear power plants in France. It has been shown that in the sea or estuary, temperature in the water returns to normal beyond an area of about 10 km^2 (Gregoire *et al*, 1993). In the area affected, the temperature increase is about one degree for each power plant of 2000 to 4000 GW (2 to 4 reactors). Generally the most impacted individuals are the young and small aquatic organisms (Granier, 1986).

At Paluel and Flamanville, on the English Channel, the monitoring of the sea temperature around the point of release shows that the temperature increase is not detectable at distances farther than 500 m. The impacts of the heated water released from the Gravelines nuclear power plant (6 reactors), located on the east side of the English Channel, have been studied in more detail by EDF (Gregoire *et al*, 1993). The results of the hydro-biological monitoring show that:

- in the benthos close to the point of release (approximately 5 km^2), a polychaete species is replaced by less thermo-sensible species,
- there is no significant impact on the zooplankton, but the growth of the phytoplankton seems to be accelerated,
- there is an important increase of the amount of vibrion halophile (bacteria), but this is partly compensated by the mortality induced by the chlorine contained in the water,
- for the fish populations, the thermal impact is very limited. Some species, particularly during winter, are attracted by the areas where the water is warmer,
- the chemical impact of the chlorine contained in the effluents on bacteria and plankton is limited to an area less than 1 km^2.

In rivers, the variation of temperature depends greatly on the climate of the site and on the river characteristics. Therefore, site-specific limits for thermal discharges have been imposed by the local authorities to control the potential impacts of the release of heated water. In order to comply with these limits, some plants decrease their activity during summer periods. Temperature increases greater than 3 - 4 °C on the average seem to cause impacts on the fish and benthic populations for river releases (Granier, 1986).

The thermal impact of the Bugey power plant, located on the Rhône river, has been studied (Roger *et al*, 1991). It has been reported that the temperature rise is perceptible up to 10 km downstream the power plant. The mean temperature increase up to 5 km from the point of release is 5°C. This study has shown that when compared with the upstream site, the warmer station is characterised by a significant reduction of the species diversity of invertebrate communities. A study of the Loire, Seine and Moselle rivers reveals that the temperature rise of the water close to the nuclear power plants causes an increase of certain algae and that some fish species are replaced by more resistant species (RGN, 1981).

From the limited evidence cited, it appears that thermal releases into the aquatic environment result in observable levels of environmental impacts. The impacts have not been considered to be priority impacts due to the fact that they do not result in large scale impacts. However, if the assessment of all impacts were to be completed in the future, thermal releases should be investigated further.

4.4. Monetary Valuation

4.4.1. Results

The final results reported in this summary document are the monetary damages assessed for the impacts resulting from the releases of the nuclear fuel cycle activities. Except in the case of a severe accident, the damages are due to occupational and public health effects from radiological and non-radiological sources. Environmental impacts do not contribute a significant amount to the routine operation evaluation. They are taken into account in accidental situations where there is a market loss that can be estimated. The damages are presented in a matrix of time and space categories that will allow for more direct comparisons to be made with other fuel cycles and studies that may have different assessment boundaries.

Although a comprehensive study of physical impacts has been completed, it must be stressed that the final result should only be considered as a sub-total; it has not been possible to include the assessment of all possible impacts that should be included in a final total value. It is also important to stress that there has been very limited determination of the extent to which these impacts are to be considered externalities.

Tables 20 (a, b and c) present a summary of the costs of the physical impacts estimated for all stages of the fuel cycle, except for a severe reactor accident. The public health impacts are reported by the local, regional, and global population categories and three time periods. The occupational impacts are included in the short-term category. They are due essentially to non-radiological accident injuries. The radiological impacts occur in the medium- and long-term.

Tables 21 (a, b and c) present the detailed monetary valuation for the 5 electricity generation sites for the 3 discount rates. The use of a discount rate changes the importance of the values in the different cells of the matrix, as can be seen in Tables 20 and 21 (b and c).

The sub-total of the cost presented for all the stages of the nuclear fuel cycle is about 0.1 mECU/kWh if the 3% discount rate is applied. The range that is found by varying the discount rates between 0% and 10% is between about 2.5 mECU/kWh and 0.05 mECU/kWh, respectively. The base load electricity generating costs in France are on the order of 35 - 40 mECU/kWh (50% of this value is due to investment costs).

As has been stated, these results include the monetary valuation of the occupational health impacts. Over all, if no discount rate is used, the occupational impacts comprise of about 5.5% of the total cost. This proportion increases to over 75% when a 3% discount rate is applied, and over 95% with the 10% discount rate, due to the more immediate nature of some of the occupational impacts. The results of the local, regional and global public doses are illustrated in Figure 9 for the three discount rates.

Table 20. The monetary valuation of physical impact for normal operation

20a. Monetary valuation in mECU/kWh with no discount rate

DR=0%		Short term			Medium term			Lo te
mECU/kWh	local	regional	global	local	regional	global	local	regi
MINING AND MILLING	1.48E-02	0	0	3.23E-02	1.69E-02	1.94E-05	3.15E-04	1.82
CONVERSION	6.25E-04	0	0	3.43E-04	3.20E-07	1.77E-07	4.17E-06	1.54
ENRICHMENT	1.18E-03	0	0	1.46E-06	1.00E-07	7.25E-08	3.91E-06	6.94
FUEL FABRICATION	8.19E-04	0	0	1.07E-03	1.63E-06	9.64E-10	6.22E-08	1.09
ELECTRICITY GENERATION (1300 MW)								
- CONSTRUCTION	3.94E-02	0	0	0	0	0	0	
- OPERATION	8.21E-03	0	0	3.05E-02	3.36E-02	2.97E-02	1.13E-08	3.16
- DECOMMISSIONING	0	0	0	1.93E-02	0	0	0	
REPROCESSING	2.96E-03	0	0	2.98E-04	9.63E-03	1.60E-01	3.45E-06	1.67
LLW DISPOSAL	-	0	0	1.50E-05	0	1.24E-04	2.36E-06	
HLW DISPOSAL	-	0	0	8.98E-08	0	0	2.54E-02	
TRANSPORTATION	3.07E-04	0	0	3.47E-04	0	0	0	
Sub-total	**6.83E-02**	**0**	**0**	**8.42E-02**	**6.01E-02**	**1.90E-01**	**2.57E-02**	**1.85**

20b. Monetary valuation in mECU/kWh for 3 % discount rate

DR=3%	Short term			Medium term			
mECU/kWh	**local**	**regional**	**global**	**local**	**regional**	**global**	**local**
MINING AND MILLING	9.94E-03	0	0	5.58E-03	2.90E-03	3.34E-06	7.52E-1
CONVERSION	4.18E-04	0	0	5.97E-05	5.26E-08	2.78E-08	1.01E-1
ENRICHMENT	7.90E-04	0	0	2.54E-07	1.65E-08	1.25E-08	9.23E-1
FUEL FABRICATION	5.48E-04	0	0	1.86E-04	2.81E-07	1.51E-10	1.50E-1
ELECTRICITY GENERATION (1300 MW)							
- CONSTRUCTION	3.94E-02	0	0	0	0	0	0
- OPERATION	5.41E-03	0	0	5.31E-03	4.00E-03	1.99E-03	2.10E-1
- DECOMMISSIONING	0	0	0	6.91E-03	0	0	0
REPROCESSING	1.98E-03	0	0	5.10E-05	1.28E-03	1.06E-02	6.42E-0
LLW DISPOSAL	-	0	0	2.61E-06	0	5.76E-06	1.04E-1
HLW DISPOSAL	-	0	0	6.41E-09	0	0	0
TRANSPORTATION	2.06E-04	0	0	6.01E-05	0	0	0
Sub-total	**5.87E-02**	**0**	**0**	**1.82E-02**	**8.18E-03**	**1.26E-02**	**6.44E 08**

20c. Monetary valuation in mECU/kWh for 10 % discount rate

DR=10%	Short term			Medium term			Long	
mECU/kWh	**local**	**regional**	**global**	**local**	**regional**	**global**	**local**	**regi**
MINING AND MILLING	5.20E-03	0	0	7.00E-04	3.63E-04	4.19E-07	8.48E-13	4.97
CONVERSION	2.19E-04	0	0	7.48E-06	6.28E-09	3.14E-09	1.13E-16	4.21
ENRICHMENT	4.13E-04	0	0	3.18E-08	1.97E-09	1.56E-09	1.04E-18	1.89
FUEL FABRICATION	2.87E-04	0	0	2.33E-05	3.55E-08	1.71E-11	1.69E-20	2.98
ELECTRICITY GENERATION (1300 MW)								
- CONSTRUCTION	3.94E-02	0	0	0	0	0	0	0
- OPERATION	2.87E-03	0	0	6.65E-04	4.44E-4	1.27E-04	2.37E-11	9.43
- DECOMMISSIONING	0	0	0	9.26E-04	0	0	0	0
REPROCESSING	1.04E-03	0	0	6.29E-06	1.17E-04	6.79E-04	7.24E-09	2.58
LLW DISPOSAL	-	0	0	3.26E-07	0	8.44E-08	1.53E-15	0
HLW DISPOSAL	-	0	0	1.12E-10	0	0	0	0
TRANSPORTATION	1.13E-04	0	0	7.53E-06	0	0	0	0
Sub-total	**4.95E-02**	**0**	**0**	**2.34E-03**	**9.24E-04**	**8.07E-04**	**7.26E-09**	**2.58**

Table 21. The monetary valuation of physical impact for normal operation of electricity generation

21a. Monetary valuation in mECU/kWh with no discount rate

DR=0%	Short term			Medium term			
mECU/kWh	local	regional	global	local	regional	global	local
BELLEVILLE	8.49E-03	0	0	2.44E-02	5.15E-03	2.99E-02	1.66E-08
FLAMANVILLE	8.68E-03	0	0	2.81E-02	6.36E-05	2.96E-02	5.05E-09
NOGENT	7.76E-03	0	0	2.44E-02	1.56E-01	2.93E-02	1.69E-08
PALUEL	7.57E-03	0	0	4.33E-02	1.52E-04	3.00E-02	5.85E-09
SAINT-ALBAN	8.54E-03	0	0	3.23E-02	6.91E-03	2.97E-02	1.20E-08
Average	8.21E-03	0	0	3.05E-02	3.36E-02	2.97E-02	1.13E-08

21b. Monetary valuation in mECU/kWh for 3 % discount rate

DR=0%	Short term			Medium term			L te	
mECU/kWh	local	regional	global	local	regional	global	local	reg
BELLEVILLE	5.69E-03	0	0	4.25E-03	6.72E-04	2.02E-03	3.08E-10	1.1
FLAMANVILLE	5.81E-03	0	0	4.89E-03	1.08E-05	1.97E-03	9.39E-11	1.3
NOGENT	5.20E-03	0	0	4.24E-03	1.84E-02	1.95E-03	3.14E-10	7.4
PALUEL	5.07E-03	0	0	7.54E-03	2.60E-05	2.03E-03	1.09E-10	4.8
SAINT-ALBAN	5.27E-03	0	0	5.63E-03	9.09E-04	1.99E-03	2.24E-10	4.2
Average	5.41E-03	0	0	5.31E-03	4.00E-03	1.99E-03	2.10E-10	8.3

21c. Monetary valuation in mECU/kWh for 10 % discount rate

DR=0%	Short term			Medium term			
mECU/kWh	local	regional	global	local	regional	global	local
BELLEVILLE	2.97E-03	0	0	5.32E-04	5.72E-05	1.29E-04	3.47E-11
FLAMANVILLE	3.04E-03	0	0	6.12E-04	1.33E-06	1.26E-04	1.06E-11
NOGENT	2.72E-03	0	0	5.31E-04	1.37E-03	1.25E-04	3.54E-11
PALUEL	2.65E-03	0	0	9.44E-04	3.24E-06	1.30E-04	1.23E-11
SAINT-ALBAN	2.99E-03	0	0	7.05E-04	7.89E-04	1.27E-04	2.53E-11
Average	2.87E-03	0	0	6.65E-04	4.44E-04	1.27E-04	2.37E-11

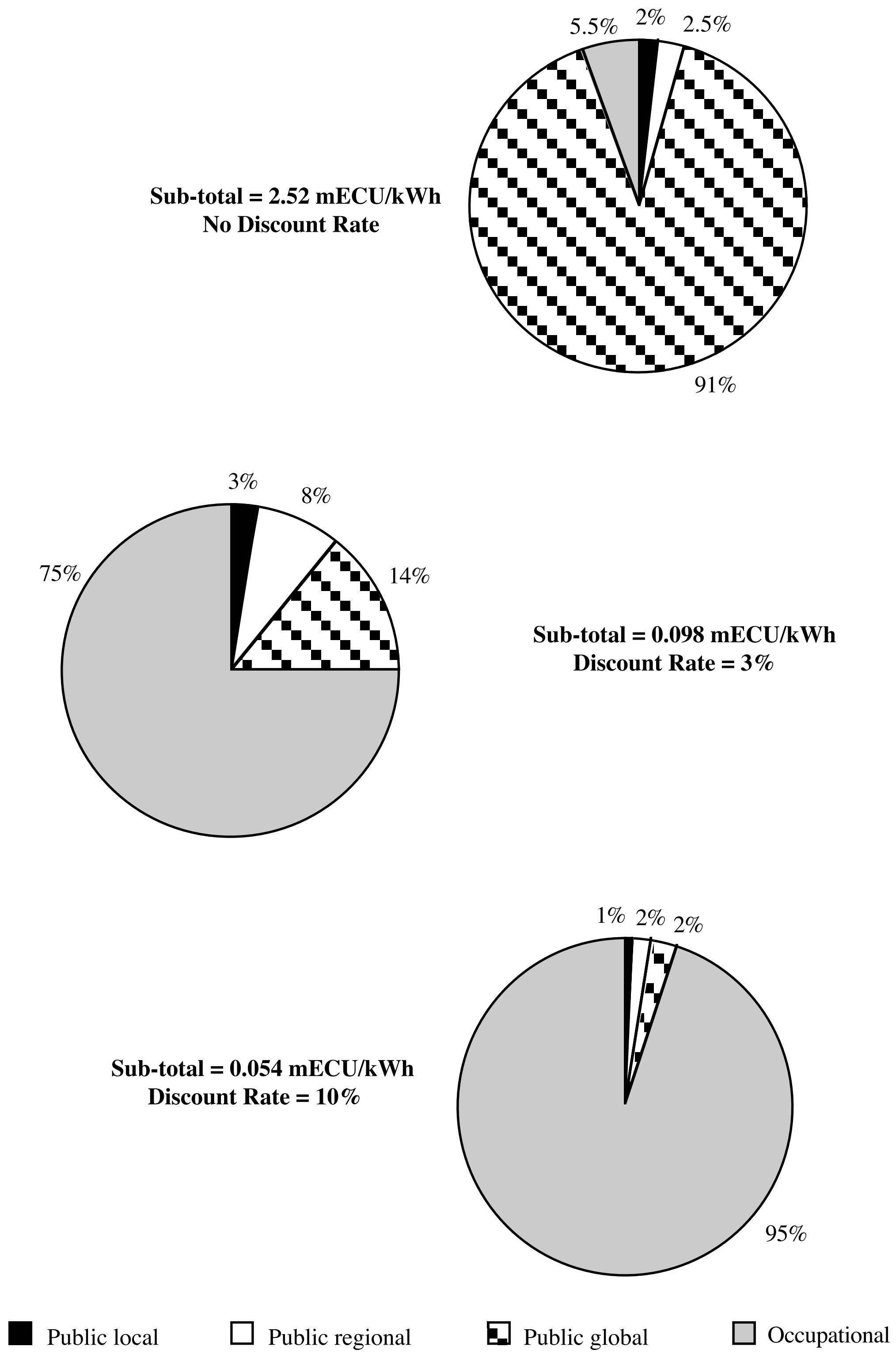

Figure 9. Distribution of the costs for the 0%, 3% and 10% discount rates

The results of the four accident scenarios assessed at a hypothetical PWR site in the centre of western Europe are presented on Table 22. These results are considered to be indicative of the impacts that would be found using a risk-based methodology. These results are lower than those presented for a 900 MWe PWR (Dreicer *et al*, 1995) because of the different probabilities of a core melt accident for the two types of reactors. For the reference scenario (ST21, core melt with 1% of the core released), a cost of about 0.005 mECU/kWh was estimated. The value for the smallest and the largest releases (ST23 and ST2) are estimated to be about 0.0005 and 0.02 mECU/kWh respectively. No discount rates were applied to the health effect costs, but were incorporated into the assessment of the countermeasure costs.

Table 22. Results of accident analysis at a 1300 MWe PWR, for 4 different scenarios including public health effects and costs of countermeasures

Source term (approximate % of total core released)	**Core melt probability** (per reactor.year)	**Conditional probability**	**Total cost** (MECU)	**Cost x probability** (MECU per reactor.year)	**Cost** (mECU/kWh)
ST2 (10%)	1E-05	0.19	83252	0.158	0.023
ST21 (1%)	**1E-05**	**0.19**	**17093**	**0.032**	**0.0046**
ST22 (0.1%)	1E-05	0.19	3339	0.006	0.0009
ST23 (0.01%)	1E-05	0.81	431	0.003	0.0005

4.4.2. Discussion

4.4.2.1. Normal Operation

General

The assessment of the priority impacts for all stages of the nuclear fuel cycle have been completed. A key factor in the interpretation of these results are the dimensions of time and space. These boundaries can have a profound effect on the final result, in terms of the populations considered and the adequacy of the monetary valuation methodology.

The results have been presented with a range of discount rates, to take into account the difference in social costing between present day events and possible future events. No single discount rate has been generally accepted, so three values encompassing the range of possible choices have been presented in this report (0, 3, and 10%). For the nuclear fuel cycle, the expected time of occurrence of an impact becomes very important when discount rates are applied due to latency time between exposure and disease and long-lived radionuclides.

Without a discount rate, it can be seen that the public health costs on the local, regional and global scale contribute 94.5% of the 2.5 mECU/kWh sub-total, the remaining 5.5% is due to occupational impacts. The global impacts contribute 91% of the total cost of the fuel cycle. The large global impacts are due to the risk assessment methodology employed that assumes no threshold in the dose response function and global dispersion for 100,000 years. This results in the aggregation of very small individual doses over large populations and time spans. The fact that this uncertain and insignificant additional risk to an individual ultimately dominates the assessment of the nuclear fuel cycle, opens the question of the adequacy of applying the monetary valuation methodology in an equal way to radically different levels of individual risk. The usefulness of the global costs presented in this report must be considered carefully before being used for decision-making.

The releases from the electricity generation stage contributes about 18% of the total cost, but the global impacts, predominately due to the long-lived C-14 releases, dominate the final result. The reprocessing stage of the fuel cycle is the largest contributor at 1.9 mECU/kWh (about 76% of sub-total 2.52 mECU/kWh).

On the local and region scale, the impacts of the atmospheric and liquid releases are in the same order of magnitude. Most of the impacts results from electricity generation (operation) and HLW disposal liquid releases and the mining and milling stage atmospheric releases. When a discount rate is applied, the HLW costs are reduced to zero, so the impacts are mainly due to the other stages. The large difference between the very small external cost estimated and public concern in many countries also raises the question of adequacy of the monetary valuation methodology for very long term impacts and technologies where risk aversion is an important factor in the social costing.

Influence of discount rate

Figure 10 illustrates for 0%, 3%, and 10% discount rates the distribution of the impacts with time and space for all stages of the nuclear fuel cycle. Looking at the results of the application of the 3% and 10% discount rates, one can see some trends. The cost contribution from the reprocessing stage with the discount rate of 3% (Table 20b) changes from 76% of the sub-total to 15%, and to as low as 4% when the 10% discount rate is used (Table 20c). This is due to the fact that the largest portion of the impacts are long term and therefore are greatly reduced if any discount rate is used.

When the 3% discount rate is applied, the most important contributor becomes the costs from the impacts from the construction of the reactor (40%) because discounting does not reduce these very short-term impacts. The next most important contributors are the operation phase of mining and milling and electricity generation (19% and 17%, respectively). These do not show as drastic a decrease due to the importance of the short term occupational health impacts.

For the higher discount rate of 10%, the cost from the construction of the reactor rises to 73% of the sub-total, followed by mining & milling (12%) and operation of the reactor for electricity generation (8%). The most dramatic drop in cost can be seen for the waste disposal operations where the values become essentially zero. Clearly this is due to the fact that the impacts from waste disposal will mostly occur in the far future.

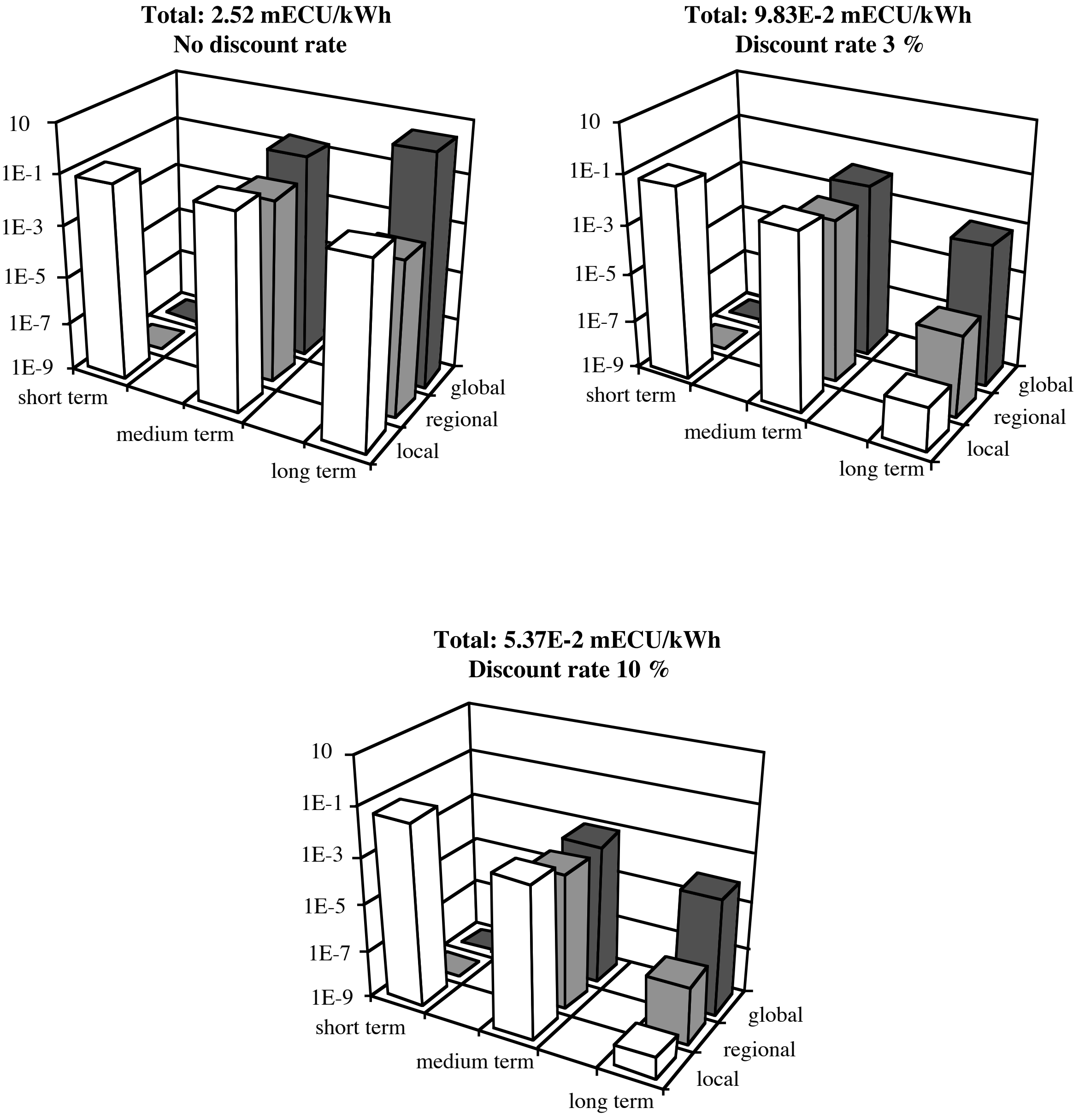

Figure 10. Distribution of the costs for the 0%, 3% and 10% discount rates with time and space (log scale)

Sensitivity of the site chosen for the electricity generation stage

The average cost of the electricity generation routine operation is 0.44 mECU/kWh for DR = 0%. If the construction and the decommissioning costs are taken into account, this average value becomes 0.5 mECU/kWh. As is shown in Figure 11, the total cost for each of the sites is essentially the same except for Nogent, due to the aggregation of low-level atmospheric and aquatic pathway doses affecting larger population centers.

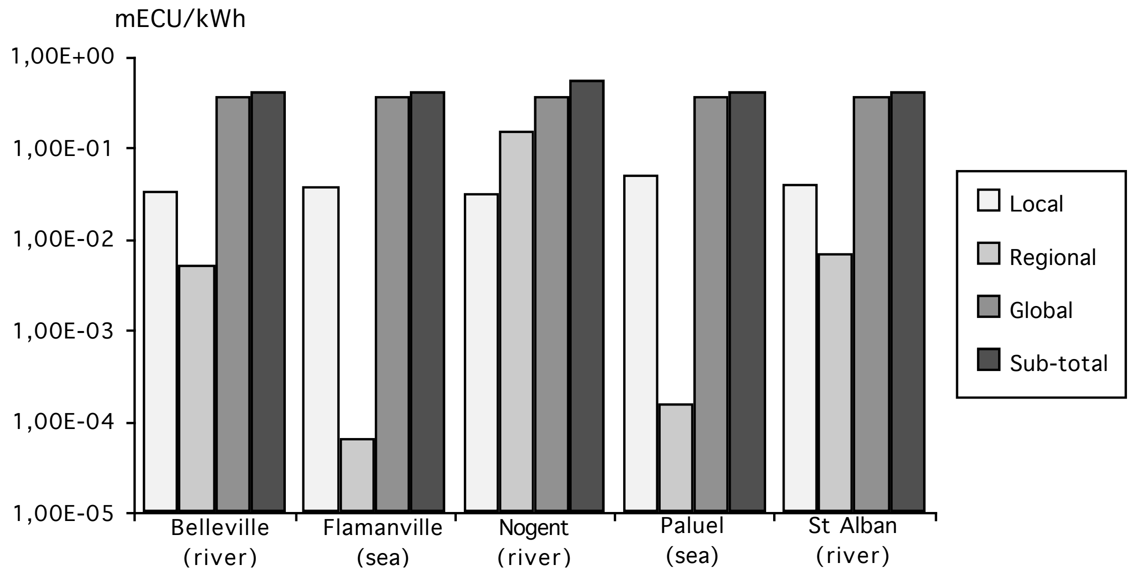

Figure 11. Costs of electricity generation for the different sites (no discount rate)

The average costs for the 1300 MW reactors can be compared to the monetary values that reported for the Tricastin 900 MWe PWR (Dreicer *et al*, 1995). Table 23 presents the differences between the construction, routine operation and decommission costs of the 2 types of reactors, as well as the differences that result from the change in transportation route.

The differences between the Tricastin reference site and the average of the five 1300 MWe PWR sites are essentially insignificant. The small changes seen in the transportation costs are due to the different distances travelled between the sites. It is important to note that, despite the differences existing in the characteristics of the sites and the releases, the costs of the two types of PWR are very similar.

Table 23. Monetary valuation of the physical impacts resulting from the construction, operation, decommissioning and transportation

	Electricity generation PWR			
	Construction mECU/kWh	Operation mECU/kWh	Decommissioning mECU/kWh	Transportation mECU/kWh
DR = 0%				
900 MWe (1)	3.37E-2	4.16E-1	1.70E-2	7.79E-4
1300 MWe (2)	3.94E-2	4.41E-1	1.93E-2	6.54E-4
DR = 3%				
900 MWe (1)	3.37E-2	2.03E-2	5.96E-3	3.11E-4
1300 MWe (2)	3.94E-2	1.68E-2	6.91E-3	2.66E-4
DR = 10%				
900 MWe (1)	3.37E-2	5.89E-3	7.93E-4	1.40E-4
1300 MWe (2)	3.94E-2	4.12E-3	9.26E-4	1.21E-4

(1) Tricastin site
(2) average of the 5 sites

4.4.2.2. Accidental Situations

Transportation

The contribution of risk of transportation accidents to the overall costs calculated for the nuclear fuel cycle can be considered to be negligible (0.0003 mECU/kWh). This is largely due to the fact that the most dangerous materials to be transported travel the shortest distances (about 5 km). This result has been included in the tables of results that have been presented.

Reactor accidents

Four accidental releases have been addressed during this phase of the work, but considering the results as beeing only indicative, they have not been included in the final 2.5 mECU/kWh sub-total.

The cost is 0.005 mECU/kWh for the reference scenario. The other scenarios, presented to illustrate the sensitivity of the results, were valued at costs 500% (ST2), 20% (ST22) and 10% (ST23) of the first estimate. If these costs were to be added to the routine operation results, the subtotal would increase by no more than 1%. Although it is acknowledged this assessment is only indicative, its results are broadly in accord with more comprehensive assessments.

The US team (ORNL, 1993) assessed the costs of severe nuclear reactor accidents using the same risk-based methodology and type of accident consequence computer code

(MACCS). Two types of accidents at two sites were included in their assessment. The results presented in the ORNL draft report are 0.06 mills/kWh for the Southeast site (SE) and 0.006 mills/kWh for the Southwest site (SW). In addition, the US team included the cost estimates for waste disposal, loss of utility assets, utility site clean-up, replacement power, and decommissioning after the accident. These additional costs will contribute about 0.04 mills/kWh, which would about double the SE site accident costs and dominate the SW site accident costs.

A Probabilistic Safety Assessment (PSA) for modern UK reactors completed a comprehensive assessment of 23 possible accident scenarios and their associated probabilities (Wheeler and Hewison, 1994). Another accident consequence computer code (CONDOR) was used to estimate the same costs that considered in COSYMA and MACCS. The cost of a severe accident taking into account all the scenarios was reported to be 0.0001 p/kWh or about 0.001 mECU/kWh. In a nuclear utilities group report to the UK government presenting these results, it was suggested that this value be increased an order of magnitude to 0.01 mECU/kWh for the sake of conservatism.

The range of costs that are found using this type of methodology, as shown by this assessment, the US team, and Wheeler and Hewison, fall within the range of 0.005 to 0.1 mECU/kWh. If this range of costs is accepted as the portion of the severe accident costs attributable to health effects and countermeasure implementation, further effort can be made to assess the source of other social costs that should also be included.

Pearce *et al* (Pearce *et al*, 1992) has also followed the same line of thought by concluding that the externality adders for health effects costs due to a severe nuclear accident assessment would be negligible after assuming an accident probability considered correct for the type of technology that would be built in the UK today (one in a million chance (10^{-6}) per reactor year). Preliminary proposals on possible mechanisms to estimate risk aversion were considered by Krupnick, Markandya and Nickell (Krupnick *et al*, 1993), however the data is not available to be able to complete this assessment.

4.4.3. Comparison with Other Studies

For the most part, the earliest phases of comparative risk assessments of different fuel cycles options concentrated on deaths and injuries as the indicators for damages. In 1988, O. Hohmeyer published a report on the social costs of energy consumption (Hohmeyer, 1988). This was followed by a German-American workshop in 1990 on the external environmental costs of electric power (Hohmeyer, 1991) and the "Pace University report" on the environmental costs of electricity (1990). The results from these studies included impacts that had not been addressed in previous studies. The emphasis on social costs required that different impacts pathways be considered and it introduced the use of money as an impact indicator.

One of the major disadvantages of these studies was that the boundaries of the assessments and methodologies applied varied between the different fuel cycles, therefore it was difficult to objectively compare the final results. The ExternE project set, as one of its main objectives, the goal of analysing different fuel cycles within the same consistent methodological framework in order to allow for direct comparison between fuel cycle options. Since the start of this project, a number of other projects have been initiated on the international and national levels.

Table 24 shows the rounded results for nuclear fuel cycle assessments of the EC and US partners in this project along with the other main studies in the field. Direct comparison of these results are not really possible due to the different approaches and coverage of the fuel cycle, but it is useful to illustrate the context in which the ExternE project results will be placed.

Table 24. Results for nuclear fuel cycle assessments of the EC and US teams

	Total cost (mECU/kWh)	Cost without accident
CEPN (DR=0%)	2.52 *	2.52
(DR=3%)	*nc*	0.1
(DR=10%)	*nc*	0.05
ORNL Draft 1993		
mid value SE site	0.3	0.2
mid value SW site	0.2	0.2
Pearce *et al* 1992	0.8 - 1.8	0.5 - 1.2
Friedrich & Voss 1991	0.1 - 0.7	0.1 - 0.4
Pace 1990	29	6
Hohmeyer 1991	15 - 88	*nd*
Hohmeyer 1988	5 - 50	*nd*

nc = not completed
nd = not done
* for the reference scenario considered for France

The large range of results can be attributed to:

- the different number of fuel cycle stages included in each assessment,
- the methodology for the valuation of the health impacts,
- the discount rate applied,
- the inclusion of a global physical impact assessment, and
- the methodology and assumptions used for the assessment of severe nuclear accidents.

A more detailed description of the past studies can be found in (Dreicer *et al*, 1995).

4.4. Uncertainty

Each part of the impact pathway methodology employs different models and input data. The uncertainty of these calculations contributes to the overall confidence in the final results. In general, each part of the assessment of the impacts of routine operations provides results that can be considered to have an uncertainty well within an order of magnitude, except for the estimates of human exposure and dose conversion where an uncertainty of an order of magnitude has been indicated in certain extreme cases.

A lower level of confidence can be assigned to the results of the global assessments for C-14, H-3, I-129 and Kr-85, due to the extremely general models that are used and the propagation of very small doses over a large population for very long periods of time. Some far-reaching assumptions were taken. The uncertainty in these estimates is probably greater than an order of magnitude except in the case of C-14, where the global carbon cycle is quite well known.

The estimates of the doses for the waste disposal stages are also considered to have a greater factor of uncertainty. The PAGIS study (CEC, 1988) completed a more comprehensive uncertainty analysis for high level waste disposal. For the normal evolution scenario, an uncertainty of the results of the granite disposal option in France of about 3 orders of magnitude was reported. The low level disposal site assessment employs the same type of hydrogeological models and exposure assumptions but the time scale of the possible impacts is quite shorter. Due to the different time scale, it is considered that the maximum uncertainty would be 3 orders of magnitude, but it is likely to be smaller.

For estimation of the uncertainty of the accident assessment results, one must consider the methodology for the assessment of the consequences, the probabilities that have been assigned, and the gaps in information and the methodology. It can be estimated that the uncertainty of the accident assessment results could range from a factor of 2 to as large as 3 or 4 orders of magnitude.

4.5. Limitations of the Results

In presenting the results, care has been taken to indicate that the "sub-total" costs are not intended to represent the absolute total of all the impacts possible. Within the resources available for this project, the priority impact pathways have been analysed and the most important impacts are included.

In attempts to include all impacts within a rather large time and space scale, the limits of valid assumptions have been stretched in order to be as complete as possible in the physical impact assessment. Current day conditions have been assumed to remain constant for 100,000 years - a truly unlikely event. However, with these assumptions it was possible to complete the assessment for the long-lived radionuclides.

No thresholds have been assumed in the calculation of the response to the doses received, so in many cases, the average individual doses to the public fall into a highly uncertain area of the dose-response relationship. The generally accepted collective dose approach, which integrates the average individual doses over the total population to be considered, was implemented. With this approach, the magnitude of the individual risk is masked when the results are presented. The most obvious drawback of this approach has been seen in the evaluation of the long-term global impacts of C-14, where the very small individual doses are summed to a large value over time and space. However, without this assessment, an important part of the overall potential physical impacts from the nuclear fuel cycle would not have been complete.

The assessment of a potential severe nuclear reactor accident was based on a risk-based approach. This methodology has not been accepted by everyone, however, it has been judged as an adequate basis for the calculation of the physical impacts within a range of uncertainty. This does emphasise the need for further methodological work to determine the additional social impacts and costs that have not been included in this type of approach.

Although there are limitations and uncertainties in the methods used for the assessment of the physical impacts, the key methodological issues that remain are for the monetary valuation stage. Before the monetary values of the impacts from the nuclear fuel cycle can be considered to be external costs the following issues must be addressed:

- If the same monetary valuation methodology is used for the evaluation of very small (and quite uncertain) individual risks to a large population and larger individual risks to a smaller population, does the final result really demonstrate the proper weighting of the real risks? Should occupational (voluntary) risks be valued in the same manner as risks to the general public (involuntary risks).

- If the use of a discount rate is not considered to be acceptable for the evaluation of far future impacts, what should be used in its place? The results of this phase of the project should be reviewed in order to determine how to provide a good representation of present day and far future risks.

- How can a method realistically incorporate the societal perceptions in terms of time and space keeping in mind the need for society to balance between the options available? For example, if C-14 is released today, it is diluted and results in low individual risks with no future disposal problems. If the releases are captured, waste repositories must be maintained causing increases in occupational risks and large local population risks in the far future.

- How can the aversion of certain risks be equitably included in the assessment of external costs? This problem is clearly illustrated in the differences between expert and public perceptions of the risks of potential nuclear accidents and high level waste disposal.

Even with these unresolved issues, the study has made important advances in reporting the physical impacts in a manner consistent with other fuel cycles, identifying remaining uncertainties, and highlighting the important parameters that must be considered in the decision making process.

4.6. Conclusions

This report is to be included as part of the ExternE Project French Implementation team. The majority of the results presented in this report have been taken from the Nuclear Fuel Cycle report, presenting the methodology for the assessment of the external costs (Dreicer *et al*, 1995). For the complete implementation of the accounting framework in France, a more recent and representative reactor type was required, so the evaluation of 1300 MWe PWR's were completed to replace the 900 MWe reactors evaluated previously.

The electricity generation stage is the only case where various civilian sites exist in France, so in order to determine if the results are significantly influenced by different site-specific characteristics five different sites were evaluated. In general, for the overall cycle, this has not proved to be an important factor.

As a result of the addition of the 5 different electricity generation sites, the evaluation of additional transportation routes were required, however the changes in the results were insignificant.

In keeping with the overall framework of the ExternE project, a consistent implementation of the monetary valuation methodology has been completed. The final results of this work have shown that in some cases, the values do not represent an adequate assessment of the external costs of some of the more difficult issues. Neither the application or omission of a discount rate have provided an adequate means of valuing very long term impacts. The consistent application of the monetary values regardless of level of individual risk also reveals what seems to be an imbalance in the evaluation. It has also been found that the level of costs attributed to low probability events do not seem to take into account the general public opinion. These are issues that must be addressed before the results can be used by decision-makers.

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