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A Comparison of Different Options for Electricity Production: Environmental Indicators for Application in a Multi Attribute Utility Analysis

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SUMMARY

The project "Comparative Assessment and Management of Radiological and Non-Radiological Risks Associated with Energy Systems", one of the projects in the research programme "Nuclear Fission Safety" of the European Commission DGXII, aims to develop a method for the comparison of environmental and human health impacts of electricity generating systems. Two energy systems serve as cases, the coal fuel cycle in Germany and the nuclear fuel cycle in France; they are compared on the basis of a Multi Attribute Utility Analysis (MAUA). RIVM is one of the participants in the study and investigates the use of environmental indicators in the MAUA to describe the impacts on the environment.

This report presents a set of indicators to describe the environmental impacts. The indicators are based either on the pressure exerted on the environment (pressure indicators based on emissions) or on no-effect levels (chemical-state indicators based on concentrations). The indicators are described in more detail for a limited number of environmental concerns, i.e. climate change, depletion of natural resources, acidification and eutrophication, contamination with radionuclides and with heavy metals, and illustrated with sample calculations.

The examples show that it is possible to define environmental indicators based on no-effect levels which can be easily calculated and intercompared. However, problems arise in the comparison of the pressure indicators, on the one hand, and chemical-state indicators on the other. This comparison should be executed in the MAUA, where the different indicators are weighted. It is therefore recommended to apply the indicators within the MAUA to discern the advantages and disadvantages of the indicators.

SAMENVATTING

In het kader van het onderzoeksprogramma "Nucleaire Veiligheid" van de Europese Unie (Directoraat-Generaal XII) wordt een project uitgevoerd, genaamd "Vergelijking van de risico's van electriciteitsproductie". Het doel van het project is een methode te vinden voor het vergelijken van zowel gezondheidseffecten als milieu-effecten van verschillende manieren van elektriciteitsproduktie. Hiertoe zijn twee voorbeelden gekozen, een kolencentrale in Duitsland en een kerncentrale in Frankrijk, die vergeleken worden op basis van een multicriteria analyse. RIVM is een van de deelnemers aan het project, en heeft als taak indicatoren te onderzoeken, die in de multicriteria analyse gebruikt kunnen worden voor het beschrijven van de milieu-effecten.

Dit rapport beschrijft een mogelijke set van indicatoren. Deze indicatoren zijn of gebaseerd op de druk op het milieu (*druk*-indicatoren op basis van emissies) of gebaseerd op concentraties, waarbij nog geen effect in het milieu wordt waargenomen (*chemische toestand*-indicatoren op basis van concentraties). Voor een aantal milieuthema's, verandering van klimaat, verspilling, verzuring en verspreiding (zware metalen en radionucliden) zijn de indicatoren verder uitgewerkt met voorbeeld berekeningen.

De voorbeeldberekeningen laten zien dat het mogelijk is indicatoren, die gebaseerd zijn op de concentraties waarbij nog geen effect optreedt, eenvoudig uit te rekenen en onderling te vergelijken. De vergelijking van de verschillende soorten indicatoren, de *druk*-indicatoren en de *chemische toestand*-indicatoren moet nog worden uitgevoerd in de multicriteria analyse. Het is daarom aan te bevelen de indicatoren nu toe te passen in de multicriteria analyse met de twee voorbeeldsystemen, en na te gaan wat de voordelen en nadelen zijn van de voorgestelde indicatoren.

1 INTRODUCTION

Electricity can be generated from a wide range of energy sources, like fossil fuels (coal, oil and natural gas), nuclear power and renewable energy sources (hydropower, biomass, sun and wind). The demand on electricity in a country can be realized by a combination of different sources. Electricity policy involves deciding which combination of sources best suits the need for electricity. In the decision-making process, various factors, like the economy, political preferences, infrastructure, access to resources, and public health and environmental concerns, have to be taken into account. To make a justified decision, the impacts of electricity generation on public health and the environment have to be explicitly taken into account. As different energy systems may have very different impacts, there is a need for data, as well as tools, to compare the advantages and disadvantages of various options. Such data and tools are, however, lacking, especially for the environmental and public health impacts (eg. [IAEA91]).

There are various international projects under way to improve information and methods for a comparative assessment of energy systems. Two projects which are relevant for this report are the EC/US Joint Project on the External Costs of Energy Fuel Cycles (External Costs) and the Inter-Agency Joint Project on **D**atabases and **M**ethodologies for **C**omparative **A**ssessment of **D**ifferent **E**nergy **S**ources for Electricity Production (DECADES). The External Costs project is a joint research project of EC, DGXII, and the US Department of Energy. The project aims to quantify and monetarize all the environmental and health impacts related to electricity production [EC94]. The monetarization will result in the so-called external costs, as they are not included in the current price of electricity. Different fuel cycles will be compared on the basis of total costs, including the external ones. The objective of the DECADES project is to improve the potential for a comparative assessment of energy chains for electricity generation [IAEA93]. A database with the technical, economic, health and environmental data of energy systems will be compiled; the resulting data will be made available, and methods, models and tools for a comparative assessment reviewed. The DECADES project is coordinated by the IAEA.

RIVM is one of the participants in a project in the research programme 'Nuclear Fission Safety' of the European Commission, DGXII, entitled 'Comparative Assessment and Management of Radiological and Non-Radiological Risks Associated with Energy Systems'. The other participants are CEPN (Centre d'étude sur l'Evaluation de la Protection dans le domaine Nucléaire) from France and IER (Universität Stuttgart, Institut für Energiewirtschaft und Rationelle Energieanwendung) from Germany. The project has to develop a method for comparing public health and environmental impacts of electricity generating systems.

The method has to be applied to two real energy systems, i.e. the nuclear fuel cycle in France and the coal fuel cycle in Germany.

It is recognized that in the comparison of energy fuel cycles the location-specific elements are important. Therefore two real sites have been selected for the project as reference sites. For the coal-fired power plant a 700 MW plant situated near the city of Lauffen, Germany, was chosen. For the nuclear power plant, the Tricastin 900 MW PWR at Pierrelatte, France, was selected. In the comparison, each stage of the fuel cycle, from mining and electricity production to waste management, had to be incorporated. For these stages real sites were selected as well. For example, coal is assumed to be taken from the large 'Ruhrgebiet' and Saarland coalfields, whereas the uranium is mined at Lodève in the south of France. Furthermore, in the comparison, the construction and decommissioning of the installations and the transport of fuel must also be included. As these reference sites were also selected in the External Costs project, efficient use of the search for location-specific data was assured.

For comparison of the fuel cycles, a Multi Attribute Utility Analysis (MAUA) was to be used [Be89, Ja91]. MAUA is a computer-based tool to aid in multiple-criteria decision-making. With this tool alternative options (e.g. electricity generation by coal combustion or by nuclear fission) are compared with respect to the scores for a number of criteria. The criteria are the impacts on public health and the environment.

Each institute had its own research tasks in the project. The tasks of the CEPN, co-ordinator of the project, were (1) to determine the impacts in the nuclear fuel cycle on workers and the public (2) to determine the emissions in the nuclear fuel cycle and (3) to develop and apply the MAUA tool to the comparison and test the tool on practical grounds.
The tasks of IER were (1) to determine the impacts in the coal fuel cycle on workers and the public and (2) to propose the indicators for the impacts on public health and workers to be used as criteria in the MAUA. RIVM, finally, would (1) propose the indicators for the impacts on the environment to be used as criteria in the MAUA and (2) discuss the uncertainty in the various steps of the impact assessment.

This report describes the results of RIVM in the development of the environmental indicators to be used as criteria in the MAUA. In Chapter 2, the starting-point of the RIVM part in the study is given, with emphasis on the general demands on the indicators. Chapter 3 describes the indicators that can be used for a limited number of environmental impacts. Sample calculations are shown for some of the indicators. The conclusions and recommendations are given in Chapter 4.

2 ENVIRONMENTAL INDICATORS FOR APPLICATION IN A MULTI ATTRIBUTE UTILITY ANALYSIS

2.1 Introduction

In developing the environmental indicators for use in a MAUA, various assumptions are made, based on the general starting-points of the study. In this chapter, the general starting-points and assumptions in the definition of the environmental indicators are described. Various types of environmental indicators are distinguished in Section 2.2, while in Section 2.3 the demands on indicators are presented. Section 2.4 describes the general form of an indicator based on sustainable development and Section 2.5 describes the selection of no-effect levels for different pollutants. Finally, in Section 2.6 some alternative indicators are introduced and discussed.

2.2 Application of indicators in an environmental impact assessment

Environmental indicators are used in various applications, e.g. to report on environmental quality, to evaluate the performance of environmental policy plans or to incorporate environmental concerns in policy planning (see, for example, [Ad93, SNV93, Ba94]). An environmental indicator is defined as a parameter which describes the state of the environment on a level beyond that directly associated with a parameter value. Key elements in the definition of an environmental indicator are [Ad93, SNV93]:

- Environmental indicators are aggregations of detailed information. Therefore, indicators are simplifications and do not provide complete information on environmental quality.
- Environmental indicators give quantitative information about the state of the environment and the changes in the state of the environment due to environmental impacts.
- Environmental indicators have reference values to assess the meaning of the indicator. Examples of reference values are quality objectives, historical data, natural background levels and no-effect concentrations.

Environmental indicators can be defined at each stage of the chain, from source via dispersion and effect to risk (see Figure 2.1). At least two different types of environmental indicators are distinguished, namely *stress* or *pressure* indicators and *state* or *effect* indicators [Ad93, SNV93]. *Stress* or *pressure* indicators describe the pressure by human activities on the environment. These indicators are related to the flow of materials: the use of raw materials, energy and space and the production of waste and emissions of pollutants, including radiation. Stress indicators are thus related to industrial processes rather than to the local environment. Stress indicators have several advantages: (1) they are

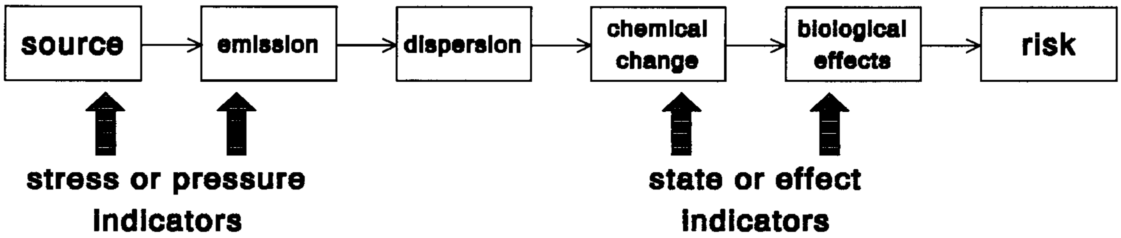

Figure 2.1 The different steps in a risk assessment and the various environmental indicators.

easy to define, and data are relatively easy to acquire and to interpret, and (2) figures can easily be used in abatement policies and international agreements. However, pressure indicators are confined to emissions and the relation with environmental effects is not obvious.

State or *effect* indicators describe the quality of the environment. Two different state indicators can be distinguished [SNV93]: chemical/physical and biological indicators. *Chemical-* or *physical*-state indicators are based on chemical/physical parameters describing the state of the environment. Examples of chemical-state indicators are atmospheric concentrations of greenhouse gases, concentrations of heavy metals in soil and the pH in soil and surface waters. *Biological*-state indicators relate to the biological state of the environment. Examples of biological indicators are: the number of species in an ecosystem, the size of the population of a species, and the quality of a forest expressed in terms of biomass, crown thinning or the percentage of dead trees. In comparison, chemical-state indicators are more easy to determine and directly correspond with the emissions. Chemical indicators also respond in general more quickly to long-term environmental pressure. However, it is difficult to relate a chemical change in the state of the environment to biological effects, since the ecosystem may have a buffering capacity, resulting in a time delay between the change in chemical state and the corresponding biological effects. Furthermore, biological changes are often due to a combination of environmental threats. The dose–response relationship is therefore difficult to assess. Biological indicators, on the other hand, are closely related to visible effects and to nature conservation. Consequently, biological indicators have a more direct appeal. Emissions, however, result generally in visible biological effects only after a long time period. Therefore the relationship between an emission and the changes in the environment is complex, resulting in large uncertainties in biological indicators.

In conclusion, the different types of indicators reflect the various stages in an impact assessment. Following an environmental assessment from emission via chemical changes towards biological effects, (1) indicators have a more direct appeal, but are more difficult to determine, and (2) the relationship between the indicator and the industrial activity becomes less clear.

2.3 Demands on environmental indicators in a Multi Attribute Utility Analysis

When defining environmental indicators for the comparison of the environmental effects of different energy fuel cycles, it is necessary to examine the demands imposed on the set of indicators by the user and by the intended application. For example, a policy decision-maker might be more interested in pressure indicators, which relate the emissions of a particular energy system to abatement strategies and national policy plans. The public, however, might be more interested in biological indicators, which describe local environmental effects. In this study a MAUA is primarily intended to be used by policy decision-makers. The information has to be on an aggregated level, and be concise and stripped of unnecessary details. Keeping this in mind, the following demands are distinguished:

- **Limited number - highly aggregated**
  A limited number of criteria should be weighted, otherwise the decision-support system is blurred with detail. The indicators should be as aggregated as possible. It is not useful to distinguish too many end-points and environmental effects. It is therefore suggested to combine all environmental effects into environmental issues, and to aim at the use of one single indicator for each environmental issue. A number of relevant environmental issues are given in Table 2.1. In the present stage of the attention is focused on contamination with radionuclides and heavy metals, depletion of natural resources, climate change, and eutrophication and acidification of water and soils. This is because these issues are significant in the quantification of the environmental impacts of the coal fuel and nuclear fuel cycles. Furthermore, this selection of environmental issues allows demonstration of the use of indicators for the whole range of effects, i.e. both location-specific as non-location-specific.

- **Sensitivity to environmental impacts**
  The indicator must be sensitive to the environmental impacts and reflect the changes in the environment due to the emissions from an energy fuel cycle. If the emissions of the fuel cycles are too low to cause measurable biological effects, the indicator must at least reflect the chemical changes.

- **Availability of data**
  To calculate the value of an indicator, it is important that all data with the required spatial and temporal resolution are available. Extrapolation or interpolation of data with different resolution introduces additional uncertainties.

Table 2.1 Environmental issues for use in the aggregation of environmental indicators [Ad93, SNV93]

Climate change (greenhouse effect)
Depletion of the ozone layer
Acidification of waters and soils
Eutrophication of waters and soils
Photochemical oxidants/ground-level ozone
Contamination with (organic) toxic substances
Contamination with radionuclides
Contamination with heavy metals
Use of renewable natural resources biomass, fish, wood, water, ...
Exploitation of land for construction and infrastructure
Use of finite natural resources metals, minerals, fossil fuels, ...
Soil degradation desertification, erosion, dessication

- **Related to the cause–effect chain**
  The indicator for the environmental impact must be clearly related to the emissions of the energy fuel cycle. For instance, if the relationship between a biological indicator of an observed environmental effect and the emission of the fuel cycle is not well established, it is better to define an indicator in the first part of the chain, i.e. a pressure indicator or a chemical-state indicator.

- **Easy to interpret, direct appeal, international consensus**
  The indicator must be easy to interpret: complex biological and chemical indicators lack direct appeal. An indicator which has a direct relationship with environmental policy planning, and on which international consensus exists, is preferred.

In the assessment of human health effects, common indicators are (acute) mortality, morbity (e.g. cases of chronic bronchites, emergency room visits, chronic cough) and workers time lost. The effects of a pollutant on a single human being is determined by using dose–response functions [CEC92]. To adapt this method to environmental indicators would mean that all possible impacts have to be calculated for all species. Such an approach is impracticable because of the lack of (species-specific) dose–response functions, the uncertainty in the effects and their possible delayed appearance. Therefore attention is focused on changes in ecosystems or populations in an ecosystem instead of quantifying effects on individual species.

The specific location of the reference power plant influences its total impact on the environment. If the reference plant is situated in a relatively unpolluted area, it is likely that its emissions will not lead to an exceedance of no-effect levels. However, if the no-effect levels in the vicinity of the reference plant are already exceeded due to emissions of, for example, other industries, traffic or agriculture, the environmental stress on an ecosystem is likely to increase. Site-specific information has therefore to be taken into account whenever possible.

In the following sections the possibility of defining environmental impact indicators for application in a MAUA will be investigated according to the requirements on environmental indicators as described in the previous sections. For environmental problems with a non-location-specific impact, like the depletion of natural resources, we propose using a pressure indicator. If the environmental impacts are location-specific, like acidification and pollution with heavy metals, we propose using an indicator based on no-effect levels.

2.4 General description of the environmental indicator

An indicator has to be sensitive to differences in the chemical and/or biological properties of a specific ecosystem or site. Therefore comparing the environmental burden with a no-effect level is proposed, since no-effect levels are often dependent on ecosystem properties. It is proposed to define the general form of the indicator as:

$$\text{Indicator}_p = \sum_i \frac{\text{additional pollution}_{p,i}}{\text{no-effect level}_{p,i}} \times \text{area}_i \qquad (1)$$

where:

Indicator_p = indicator for the total impact of a pollutant *p* on the environment

$\text{additional pollution}_{p,i}$ = the amount of a pollutant *p* in area *i* due to emissions by the installation or activity

$\text{no-effect level}_{p,i}$ = no-effect level for the polluting substance *p* in area *i*

area_i = area of grid cell *i*

The indicator is easy to interpret and calculate. However, it does not tell whether or not no-effect levels are exceeded. Moreover, for an exceedance of the no-effect levels, the effects on the ecosystem can only be described in a qualitative way.

When the indicator is summed over the area *i*, the outcome has to be evaluated carefully, since linearity of effects is assumed when using an indicator of this format. When the no-effect levels are exceeded, non-linear effects may occur. The exceedance of the no-effect levels in a small area cannot be compared with a lower value for a larger area. However, using geographical presentations, a geographical discrimination can be made. It is therefore proposed to use geographical presentations of the indicator defined as:

$$\mathrm{Indicator}_{p,i} = \frac{\mathrm{additional\ pollution}_{p,i}}{\mathrm{no-effect\ level}_{p,i}} \qquad (2)$$

2.5 No-effect levels

To compare indicators for the different environmental issues, the no-effect level for each impact has to be chosen in a way that the indicators are mutually comparable. The term no-effect levels is used here rather loosely, and is for acidification and radionuclides used as a level where no harmful effects are expected to occur at ecosystem level. This means that populations are protected, although for individuals some detriment in growth, age or reproduction may occur [IAEA92]. For toxic compounds, generally hazardous concentrations are used. Hazardous concentrations are indicated as HCp, being the concentration at which p% of the species is protected insufficiently. For instance, the protection level for ecosystems in the Netherlands is set at HC5, i.e. 95% of the species is protected.

The protection levels for metals and acidification are expressed in terms of concentrations. For radiation, the no-effect levels are given as doses to plants and animals. To make the definition of the no-effect level for radiation comparable with metals and acidification, the concentration in the environment has to be calculated for which 95% of the species receives a dose less than the no-effect dose for these species. This approach is in principle possible, however, due to lack of data the calculation for this project was restricted to the dose to grass, vegetables, sheep and cattle as reference end-points of the assessment. Next, the no-effect level of the most sensitive species was used for the calculation of the indicator value. Whether or not this result protects 95% of the species in the ecosystem is not clear.

The comparison of the impact of polluting substances can be based on the avoidance of changes in an ecosystem. Since plants and animals in an ecosystem are inter-dependent (food chain), a change in one species will result in a change in others. Unfortunately, this change can only be assessed qualitatively. Exceedance of no-effect levels for different pollutants in an ecosystem will affect the ecosystem in various ways. The effect will depend on the organism being the most sensitive end-point, and therefore on the pollutant.

It is assumed that the concentration of radionuclides in the environment is so low that the uptake of a radionuclide is not affected by the concentration of other radionuclides. When an ecosystem is polluted with several radionuclides, the total impact on the ecosystem is assumed to be equal to the sum of the single doses. On the other hand, when an ecosystem is polluted with a mixture of heavy metals, synergistic effects can be expected [DGM93]. By using an indicator as defined in Section 2.4, the possible synergism between different pollutants is not taken into account.

It is concluded that although the definition of the no-effect levels appears similar, differences exist in the no-effect levels used for radiation, on the one hand, and for example, for metals on the other. These differences relate to species used in the calculation of the no-effect concentration and the synergistic effects. Furthermore, how conservative the determination of the no-effect level is may differ for different substances, but for the purpose of this study the consequences of these differences are acceptable.

2.6 Alternative indicators

In this section two modifications to the indicator: the comparison with policy standards and the inclusion of the current levels of pollution, are discussed.

One of the options to make the indicator relevant for policy planning is to include environmental policy targets. This can be done by comparing the impact caused by the emissions of the reference plant with environmental policy targets. These targets are expressed in terms of emissions, depositions or concentrations. Usually different environmental policy targets for a pollutant are set for different periods in time. Comparing different pollutants, however, has to be done on equal time-scales, which depend on the impact considered. Moreover, environmental policy targets, if they exist, may vary between countries.

The additional impact of a plant can, for example, be compared to a current policy target (policy standard(t), see Equation 3) or to a future goal (e.g. a reduction in emission to be achieved in a time period $t^* - t$, i.e. policy standard(t) - policy standard(t^*), see Equation 4).

$$\mathbf{Indicator} = \frac{\mathbf{Additional\ impact\ (t)}}{\mathbf{Policy\ standard\ (t)}} \qquad (3)$$

or

$$\text{Indicator} = \frac{\text{Additional Impact}(t)}{\text{Current policy standard}(t) - \text{Future Policy standard}(t^{*})} \qquad (4)$$

In this way, policy decisions are directly related to the value of the indicator and might, for example, be useful in the evaluation of emission abatement strategies. An important disadvantage of this type of indicator is that environmental policy targets are usually related to emissions, and depend on time and not on location. Location-specific information can therefore not be taken into account. The indicator is more related to a pressure indicator than to an environmental state indicator. However, for the emission of CO_2 for example, this type of indicator might be useful.

In the case of an environmental-state or effect indicator, the comparison between the additional pollution and a policy standard only holds when the policy standard is related to impacts and not to emissions. Equations 3 and 4 are only valid when the policy standard and the environmental pollution have the same units.

To increase the location-specific part of the indicator, the indicator can be modified by taking the current level of pollution caused by, for example, emissions from traffic, agriculture or industry, into account. The release of a polluting substance into a polluted ecosystem, in which the no-effect concentration for that specific substance is exceeded, may have a more significant impact on the ecosystem than the release into a relatively unpolluted ecosystem in which the no-effect concentration is not exceeded. In this way, the burden from a reference plant can be compared with the environmental consumption capacity, defined here as the difference between the no-effect level and the current level of pollution (see Figure 2.2).
The modified indicator has the form:

$$\text{Indicator}_p = \sum_i \frac{\text{additional pollution}_{p,i}}{\text{no-effect level}_{p,i} - \text{current pollution level}_{p,i}} \times \text{area}_i \qquad (5)$$

where:

current pollution level$_{p,i}$ = the amount of a polluting substance p which is emitted into area i due to emissions of current installations or activities

When the current level of pollution is taken into account in this way, the outcome strongly depends on the difference between the no-effect level and this current level. If for one single area i, the difference between the no-effect and the current levels of pollution approaches zero, the indicator value in this area i will go to infinity and the sum over the areas will be dominated by the indicator value in area *i*. In this way, the calculation of the indicator makes the exact value of the no-effect level strongly weighted, whereas a smooth transition between no-effect and effect is more likely. Furthermore, when the difference between the no-effect and the current levels of pollution is small, the uncertainty will lead to a large uncertainty in the value of the indicator. One way to solve this problem is by setting a maximum for the indicator value in a single area. It should, however, also be noted that the calculation only holds for the areas where the no-effect levels are not exceeded; the areas where exceedance occurs have to be dealt with separately.

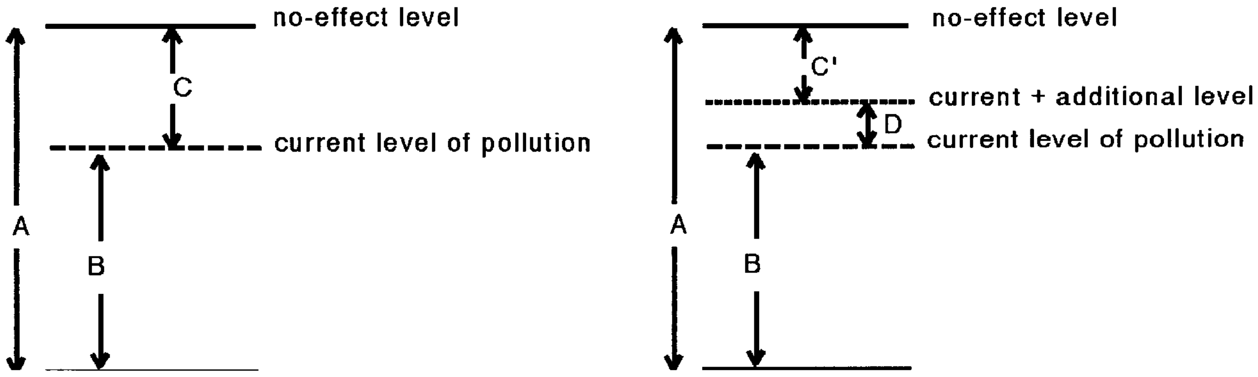

Figure 2.2 The environmental consumption capacity (C) is defined as the free space between the no-effect level (A) and the current level of pollution (B). The environmental consumption capacity is decreased to C' due to the additional burden (D) from the reference activity or installation.

For some of the environmental issues like acidification, and eutrophication and radiation, it is very likely that the additional impact caused by the reference plant is much less than the impact due to the current level of pollution from all other sources. It is unlikely that this additional impact can be distinguished from the fluctuations in the current environmental impact.

An example of an indicator modified with the inclusion of the current level of pollution is given for acidification in Section 3.3.3.6.

2.7 Conclusions

It is proposed to use geographical presentations of the indicator as defined in Equation 2 of Section 2.4. These presentations will be discussed for the environmental issues which can be described using a chemical state indicator. The indicator clearly shows whether or not the no-effect level of an ecosystem will be exceeded due to emissions of the reference plant. By including the current level of pollution caused by emissions of current industrial activities, see Section 2.6, the location- specific character of the indicator is improved. The main disadvantage of this modification is that the value of the indicator will go to infinity when the difference between the no-effect level and the current level of pollution approaches zero. Due to these calculational problems and since it is likely that the additional impact caused by the reference plant will be much less than the impact due to the current level of pollution from all other sources, we propose to use only the indicator as defined in Equation 2 of Section 2.4. The environmental issues which cannot be calculated by using a chemical-state indicator, e.g. depletion of natural resources (see Section 3.1) and the greenhouse effect (see Section 3.2), will be assessed by using environmental pressure indicators.

3 EXAMPLES OF ENVIRONMENTAL INDICATORS

In this chapter a number of environmental indicators based on the no-effect levels will be worked out. For each indicator, a sample calculation is done, using the emission figures as reported by the participants in the project [Dr93, CEC92]. The calculation only serves as an illustration: if emissions were not reported, like radionuclides in the coal fuel cycle and heavy metals in the nuclear fuel cycle, they were not included in the calculation.

Successive indicators are described for non-location-specific environmental problems, depletion of natural resources (3.1) and climate change (3.2), and location-specific environmental problems, acidification and eutrophication (3.3), as well as radionuclides (3.4) and heavy metals (3.5).

3.1 Use of finite resources

3.1.1 *Introduction*

The production of electricity is the conversion of one form of energy (heat, solar radiation) into another (electric energy). Two categories of primary energy sources can be distinguished: renewable and non-renewable sources. Renewable energy sources make use of an almost infinite, constant flux of energy to the earth's surface, part of which can be used efficiently. Tidal, solar and wind energy, and hydroelectric power, biomass and geothermal energy are included in the group of renewable sources. Non-renewable energy sources make use of a finite supply of resources. The combustion of fossil fuels (coal, oil, natural gas) and nuclear energy are obvious examples of non-renewable sources. It is clear from these examples that the terms 'renewable' versus 'non-renewable' are relative, and based on the time-scale involved. For fossil fuels, it takes millions of years to replenish the earth's resources, whereas biomass resources are replenished on a time-scale of a few years. Practically speaking, therefore, the resources of fossil fuels are considered to be finite, whereas the resource of biomass is considered to be renewable.

Use of finite resources will reduce the availability of these resources in the future, finally leading to exhaustion of the resource. However, exhaustion is a relative term. In practice, scarcity of a resource gradually leads to the exploration of lower grade ores, as long as it is economically profitable. But then again, the use of lower grade ore results in general in an enhanced consumption of land area, energy, and the like.

To allow the next generations to make use of the same resources, a minimal demand on the resources should be made. However, in the extreme this means that use of any finite resource is not allowed, since any use will diminish the available supply. There are several suggestions to avoid this pitfall:

- It is assumed that all resources can be substituted in the course of time. For electricity generation, this would mean that in time the non-renewable sources can be substituted by the renewable sources. However, a buffering capacity of a number of years is necessary to allow the substitution. Assuming a buffering time of 50 years, and a safety factor of 4 to allow for variation in demand and supply this would result in the exploitation of 0.5% annually.
- The known amount of utilizable material must be constant in time. This means that only the amount added annually to the reserves, through developments in technology, may be consumed. If addition of material to the reserves is not to be expected, it could be decided to use the known amount in a finite time period, say 100 years.
- The ratio between the known amount of utilizable material and annual consumption could be regulated. In this way, if no additional resources are discovered, both the amount of available material and the annual consumption will gradually diminish to zero.

Besides the use of the finite resources like coal and uranium, the available area of land can also be considered as a finite resource. After all, the available area of land is limited, and a competition exists between different land uses. Especially renewable energy sources, like hydroelectricity, solar power and wind energy may require large areas. It is therefore proposed to include the acreage required by an energy source in an indicator for the use of finite resources.

3.1.2 *Definition of the indicator*

As argued before, an indicator for the use of finite resources is required for:

- use of material, like coal, lime, uranium and
- use of land area.

The indicator for the use of a material can be based on the ratio between the amount of required material and the total amount of available material:

$$\boldsymbol{Indicator} = \frac{R_{material}}{S_{material}}$$

R_{material} = the amount of material required to produce a defined amount of electric energy
S_{material} = the total amount of available material

The amount of required material, R_{material}, is easy to determine with the technological data of the processes involved in the electricity production. The total amount of available material, S_{material}, is more difficult to determine, since this amount depends on technological and economic considerations. As stated previously, materials occur on earth in ores with different grades. The exploitation of a high-grade ore can be economically profitable, whereas the exploitation of low-grade ore is not. However, either due to increasing scarcity of high-grade ores or to technological developments, exploitation of low-grade ores might become profitable in the future. Furthermore, due to continued exploration of the mining industry and increasing knowledge on geology and technology, the known amount of utilizable material may increase in time.

For practical purposes, we can distinguish three classes for the natural resources, namely 'reserve', 'resource' and 'resource base' [An93]. The classification is based on three criteria: occurrence, economy and technology.

The term reserves is used for stocks with proven occurrence, which can be exploited economically at current market prices and with current technology. This term denotes the amount of material that is certain to be available. For the example of uranium, this would include all the uranium present in high-grade ores (defined as uranium contents more than 0.05%) and exploitable at a price level of less than the market price. However, it should be noted that due to large stocks, the prices on the spot market are currently low, the exploitation of high-grade ore is diminished at the moment [Se93,Un93]. Therefore the use of a specified price level of (for example) $80/kg U is more appropriate.

The term resources is used for stocks with an occurrence both proven and unproven, which can be exploited at certain specified costs with either current technology or technology that is likely to be available in the future. This term therefore denotes the amount of material that is likely to be available. For the example of uranium, this class comprises the estimated additional resources likely to be present and exploitable at a cost price lower than (for example) $260/kg U. Also, the uranium in sea water, present in a concentration of 3 mg m^{-3} and presumably exploitable at exploitation costs of $500-$800 per kg U, might possibly be included in the resources.

The term resource base is used for stocks with an occurrence both proven and unproven, and exploitable irrelevant of the expenses and technology. This term is more-or-less a theoretical notion, and denotes all the material present, irrespective of economic and technological considerations. For uranium, all the atoms present in the earth's crust, with an average concentration of 3-4 milligram per kg, is included in this term, although it is clear that it is not possible to exploit uranium.

It is clear that an indicator for the use of finite materials is based on either the reserves or the resources. An indicator based on the reserves has the drawback that it is not exactly an indicator of the occurrence of a material, but actually an indicator of the efforts of the industry and the operation of the economic market. If a material becomes scarce and the market price rises, low-grade ores become profitable and new ores are explored, resulting in an increase in the reserves. The alternative is an indicator based on resources. However, the number of resources depends on expectations with respect to technological developments and/or economic markets: what is expected to be economically profitable in future ?

We decided to base the indicator on the resources, as they appear to provide a better indication of the total amount of stocks available.

An alternative option is to use this indicator in combination with a threshold value: if the total consumption is annually less than about 1% of the total reserves, exhaustion of the material is not considered to be a problem at all. It is assumed that the reserves are large enough and no efforts are undertaken to explore new resources and new technologies.

Information on an indicator for land use is scarce. The indicator should be related to the amount of area needed for a fuel cycle. The area needed can be either the area in direct use or the area in indirect use. The first area represents, for instance, the area used for the power plant and storage of fuel and waste. The area in indirect use comprises the whole area where the fuel cycle causes restrictions in use. As data on land use for the fuel cycle are currently not available, we discard the calculation of land use.

3.1.3 *Calculation of the indicator*

As stated previously, the indicator can be defined as the ratio between the amount of required material and the total resources of the material:

$$\mathit{Indicator} = \frac{R_{material}}{S_{material}}$$

R_{material} = the amount of material, required to produce one unit of electric energy and
S_{material} = the total amount of available material, resources.

The input required in the calculation of the indicator is:

- the required amount of material per unit electricity (kg/kWhe) and
- the resources in the material (kg).

For the reference coal fuel cycle, the required amounts of materials are 1.0 x 10^{-2} kg lime and 3.2 - 3.8 x 10^{-1} kg coal per kWhe [CEC92]. The amount of uranium is not easily derived from [Dr93]. The required amount of fuel is given as 23.9 ton UO_2 per year. However, this figure must be corrected for the losses in the process mining, milling and enrichment.

Reserves and resources for coal and uranium and the relative consumption in 1990 are given in Table 3.1 [An93, EZ93]. The amount is expressed in either energy (J) or mass (kg). The indicator value can be calculated once the required amount of uranium per unit electricity is available.

Table 3.1 Reserves, resources and expenditure rate for coal and uranium [An93, EZ93]

Fuel cycle	Reserves	Resources	Expenditure of reserves e)
Coal	$1\cdot10^{15}$ kg a) 30500 EJ a) 22900 EJ f)	50500 EJ f)	0.3 % d)
Uranium	$4\cdot10^{9}$ kg b) 4000 EJ b) 600 EJ f)	$20\text{-}30\cdot10^{9}$ kg c) 800 EJ f)	1.6 % d)

a) Proven stock [EZ93]. 1 EJ = 1 x 10^{18} J
b) Proven stock. The conversion tons into exajoules depends on the market price for uranium [EZ93]
c) The limit for exploitation is $260 / kg U [EZ93]
d) [EZ93]
e) Consumption per year divided by reserves
f) [An93]

3.1.4 *Remarks*

At the moment the indicator is shown for the primary fuel, i.e. coal and uranium. However, as stated previously, it has to be investigated whether other materials are needed in the fuel cycles, e.g. exceptional materials in the construction of the nuclear power plant or the fuel enrichment process. In that case, these materials will have to be included in the calculation of the indicator value.

3.2 Climate change

3.2.1 *Introduction*

Radiatively active gases are present in the earth's atmosphere . These gases, the 'greenhouse' gases, play an important role in the radiation balance of the earth, as they absorb and re-emit part of the outgoing infra-red thermal radiation of the earth's surface. Consequently, the earth loses less heat to space by radiation than it would in the absence of the greenhouse gases and the earth's surface is therefore warmer. The mean temperature of the earth's surface is about 33 °C higher than it would be in the absence of the natural greenhouse gases [IPCC90, IPCC92].

The most important greenhouse gases are water vapour, carbon dioxide, methane, ozone, nitrous oxide and chlorofluorocarbons (CFCs). Measurements clearly indicate that the earth's temperature is closely correlated to the amount of carbon dioxide and methane in air [IPCC90, IPCC92]. The concentration of water vapour in the earth's atmosphere is determined within the climate system itself, and is not directly influenced by human sources and sinks. The concentrations of other greenhouse gases are influenced by human activities and have been increasing since pre-industrial times (ozone is excluded from this statement due to lack of data) [IPCC90]. For instance, the concentration of carbon dioxide increased from 280 ppmv (parts per million by volume) in the pre-industrial time 1750-1800 to 353 ppmv in 1990 [IPCC92]. The increased concentration of greenhouse gases in the atmosphere is likely to affect the radiation balance of the Earth and consequently the climate: the effect of emissions of greenhouse gases is climate change.

An important source for the emission of carbon dioxide to the atmosphere is the combustion of fossil fuels, whereas coal mining may lead to the release of methane. The coal fuel cycle is therefore an important source for the emission of the greenhouse gases carbon dioxide and methane [CEC92]. In the nuclear fuel cycle, the emission of greenhouse gases occurs due to, for example, construction and transportation [Bij89].

3.2.2 *Impacts and their quantification*

The emission of greenhouse gases in the energy fuel cycles results in an increase of their concentration in the atmosphere. The consequences of increased levels of greenhouse gases are calculated on the basis of general circulation models (e.g. in [IPCC92]). A calculation consists of two steps. First, a scenario is constructed which describes the emissions of greenhouse gases in time, based on social, technical and economic developments. These scenarios can extend over 100 years. Next, the general circulation models are used to calculate the climate change in time and (some of) the associated environmental impacts.

However, the prediction of the climate change and the environmental impacts has large uncertainties, due to:

- uncertainties in the various emission scenarios.
  Scenarios describing the release of greenhouse gases to the atmosphere are necessary to determine the anthropogenic impacts on the climate system. As the residence time of greenhouse gases in the atmosphere is large, long-term emission scenarios are necessary. On the time-scale involved, the uncertainties, for instance, in economic development, population growth etc. are large, and usually a number of different scenarios are studied.

- uncertainties in the processes and feedback mechanisms in the climate.
  The understanding of a number of important processes is still insufficient to determine conclusively the impacts of the enhanced greenhouse concentrations. Especially the long time-scales involved and the natural variability of the earth's climate hamper conclusive observations for the anthropogenic influences on the climate [IPCC92]. Especially the impacts of the changes in climate on a regional scale have large uncertainties.

3.2.3 *Policy*

It has been recognized that the anthropogenic emissions of greenhouse gases may result in climate change. International agreements have been made aiming at the limitation of the emission of greenhouse gases. Targets for reduction are set in the Netherlands. In the year 2000, the annual emission of carbon dioxide should be reduced to 173-177 x 10^9 kg (in 1990, the emission was 184 x 10^9 kg). For methane, the annual emission should be reduced to 970 x 10^6 kg (in 1990, the emission was 1080 x 10^6 kg) [RIVM93].

3.2.4 *Indicators*

In the definition of an indicator for climate change due to the emission of greenhouse gases in the energy fuel cycles, environmental effect indicators and pressure indicators are considered. In defining an effect indicator for climate change we encounter some important difficulties.

- The location of the source is not important for the assessment, since the greenhouse gases circulate worldwide. One single source contributes therefore worldwide. However, the impacts of climate change vary considerably on a regional scale. The calculation of an environmental effect indicator would therefore need the assessment on a regional scale of all impacts worldwide, and would have a large uncertainty.

- The environmental impacts of the emissions of a single power plant depend on the 'background', i.e. the total emission of greenhouse gases worldwide. Therefore a long-term scenario is needed to assess these background emissions. Such a scenario has large uncertainties.

Since it is not possible to determine the environmental impacts of the emissions of a single power plant conclusively, using a pressure indicator in the Multi Attribute Utility Analysis is suggested. The pressure indicator describes the emission of greenhouse gases in kg per kWh. It is proposed to weigh up the emission of each greenhouse gas with its capacity to contribute to climate change, the Global Warming Potential (GWP) [IPCC92]. The definition of the indicator for greenhouse gases would be:

$$Ind_{abs} = \sum_i E_i \cdot GWP_i$$

Ind_{abs}	indicator value in kg per kWh
E_{i}	emission of greenhouse gas i in kg per kWh
GWP_{i}	Global Warming Potential of greenhouse gas i

The value of this indicator has no direct relation with environmental effects. However, as policy plans and international agreements are stated in terms of reduction of greenhouse gas emissions, the value of the indicator is of importance to the policy decision-maker.

To relate the value of the indicator more directly to current emissions, an alternative definition might be:

$$Ind_{rel} = \frac{\sum_i E_i \cdot GWP_i}{\sum_i E_i^{total} \cdot GWP_i}$$

Ind_{rel}	indicator value as a fraction of the total emission
E_{i}	emission of greenhouse gas i in kg per year during operation
$E_{\mathrm{i}}^{\mathrm{total}}$	total emission of greenhouse gas i in kg per year
GWP_{i}	Global Warming Potential of greenhouse gas i

In this way the indicator gives the relative contribution of the fuel cycle under study to the total emission of greenhouse gases.

3.2.5 *Calculation of the indicator*

The emissions of the coal fuel cycle are given in [CEC92]. The relevant figures are:

CO_2 0.880 kg/kWhe
CH_4 0.0029 kg/kWhe
N_2O 0.0001 kg/kWhe

The reference power plant produces 689.5 MWe with 4010 full load hours per year.

For the Global Warming Potential the values for carbon dioxide, methane and nitrous oxide are 1, 11 and 270, respectively (the direct component for a 100-year time horizon) [IPCC92]. The total emission is therefore:

$Ind_{\mathrm{abs}} = 0.94$ kg/kWhe

In one year, the power plant produces $2.8 \cdot 10^9$ kWhe, resulting in

$Ind_{\mathrm{abs}} = 2.6 \cdot 10^9$ kg per year

Using the emission figures for the Netherlands for carbon dioxide and methane in 1990, the relative value of the indicator for the coal fuel cycle would be 0.013. The reference coal fuel cycle thus contributes about 1.3% to the total emission of greenhouse gases in the Netherlands (reference year 1990).

3.3 Acidification and eutrophication

3.3.1 *Introduction*

Deposition of NO_x, SO_2 and NH_3 is regarded as the cause of forest decline, although the mechanisms are incompletely known. Together with the possible interactions with other abiotic stress factors such as drought, frost, unfavourable habitat and all kinds of plant diseases and plagues, the effects of air pollution on ecosystems are specified as the 'acid rain syndrome' [St91].

There are three main hypotheses to explain the observed impacts [St91, St88].

1 The acid rain hypothesis: the impacts are caused by additional acidification of the forest soil due to wet and dry deposition of acidifying pollutants.

2 The ozone hypothesis: the effects are the result of interactions between photo-oxidants such as ozone and acid rain and/or acid fog.

3 The stress hypothesis: the effects are caused by a complex of abiotic and biotic stress factors, collectively the cause of the negative process.

First, an introduction to the acidification and eutrophication process will be given, followed by a brief description of the precursors and the effects. In Section 3.3.2, a definition of the proposed environmental indicator, the critical load, will be given and alternatives discussed. Furthermore, the relations between environmental impacts, the critical load and the (critical) chemical values, such as the base cation/aluminium ratio, will be reviewed. In Section 3.3.3 the calculation method, the results of the calculations, the uncertainties in and the main drawbacks of the proposed method will be presented. Limitations, and conclusions and recommendations, will be given in Sections 3.3.4 and 3.3.5, respectively.

3.3.1.1 Main causes

Soil acidification is a process which occurs naturally at rates which depend on the type of vegetation, soil parent material and climate. Currently, the acidification rate is influenced by human activities such as growing certain tree species, the use of fertilizers and the drainage of soils [CEC92]. Contributors to the acidification process are also the inputs of sulphur and nitrogen oxides produced by the burning of fossil fuels, and by agriculture and traffic.

Determination of the effects of acidification is complicated by the complex relationships that exist between the precursor species (like SO_2 and NO_x) and other atmospheric compounds. It is therefore difficult to distinguish between the environmental effects of acid rain and those from photochemical oxidants such as ozone [CEC92].

Within the fuel cycles, acidifying deposition is mainly due to the emission of SO_2, NO_x and NH_3. Eutrophication can be related to total nitrogen deposition of NO_x, being a mixture of nitric oxide (NO) and nitrogen dioxide (NO_2). In the reaction with O_2 and O_3, NO is converted into NO_2. In air, soil or water, SO_2 and NO convert to sulphuric acid (H_2SO_4) and nitric acid (HNO_3), respectively. Ammonia also converts to nitric acid (HNO_3).

The acidification equivalent Aeq is the unit which is commonly used to express the damage potential of acidifying pollutants [Do93, Ad93]. One mole of sulphur dioxide (64 g) corresponds to two acidic equivalents (Aeq), one mole of nitrogen dioxide (46 g) or one mole of ammonia (17 g) corresponds to one acid equivalent. In a similar way an eutrophication equivalent can be defined, but this unit will not be used further in this report.

3.3.1.2 Effects of acidification

Impacts of aerial SO_2 and NO_x on terrestrial ecosystems are likely to be rather unimportant relative to the impacts of deposited SO_2 and NO_x. Total deposition of nitrogen and acid are likely to be more important because of their long-term consequences [CEC92].

The primary effect of acidifying deposition on soil is a decrease in pH, especially for soils having a low buffering capacity. Leaching of Ca, Mg, K etc. increases, as well as the concentration of potentially phytotoxic metal ions such as Al^{3+}. High Al concentrations in soil can inhibit root growth [Sv93].

Many effects of acid rain and photo-oxidants on food crops are possible, although in practice the effects of acid deposition on soil of managed agricultural systems are probably minimal, since management practices such as liming counteract acidification and often override functions which are normally performed by soil organisms. Direct effects are a reduced yield, loss of quality of produce and an increased sensitivity to insect pests [CEC92].

The full impact of any increased deposition of acidic pollutants on water quality, and hence on fish populations, may not occur prior to a number of years after the beginning of the increase. Research has demonstrated major reductions in fish populations in some areas which have been impacted by acidic deposition [CEC92].

3.3.1.3 Effects of eutrophication

Eutrophication involves the deregulation of ecological processes in surface waters and soils due to an excessive supply of plant nutrients such as phosphates or nitrogen compounds. The principal sources are manure, fertilizers and waste water. In general, eutrophication leads to a change in competition between species within an ecosystem. This often results in lower biodiversity. Plant species that thrive in low-nutrient environments are disappearing. High nitrate levels in surface waters result in (undesirable) large quantities of algae in ponds and lakes, and a periodic shortage of oxygen. Currently, nitrate levels in groundwater in the

Netherlands, for instance, are rising to such an extent that in a growing number of places, the preparation of drinking water is threatened [Ad93]. Within the energy fuel cycles, eutrophication effects caused by phosphates can be assumed to be ignored [CEC92].

3.3.2 *Definition of the indicator*

Impacts of acidification and eutrophication take place at ecosystem level; the structure of the ecosystem is changed. By using state or effect indicators, these impacts on ecosystems can be assessed. As stated previously (see Chapter 2), two different state indicators can be distinguished: (1) chemical-state indicators, like pH level or concentration of heavy metals, and (2) biological indicators, like number of species or crown thinning of trees, which relate to the biological state of the environment. Although biological indicators are more directly related to visual damage, their quantification is often very difficult. Translation of chemical changes to actual damage is complicated due to a lack of dose–response functions, the influence of the buffering capacity of the system, the complexity of biotic interactions and possible time lags between the impact and occurrence of the damage. Therefore, the impacts of acidification and eutrophication will be assessed by using a chemical state indicator [Ba94].

A widely used chemical-state indicator for acidification and eutrophication is based on the critical load concept [RIVM93, Do93, Bu92, Ba94]. In this section the meaning of the critical load, the relation to acidification and eutrophication, the alternatives and the environmental effects above the critical load will be discussed.

3.3.2.1 Critical load

The critical load concept is an approach for estimating the quantity of pollutants that sensitive ecosystems can absorb on a sustainable basis, i.e. without measurable degradation. Different groups of experts have used a variety of definitions of differing complexity when deliberating on critical loads and levels. The term critical load has been defined by the United Nations Economic Commission for Europe (UN-ECE) (1990) as [Bu92]:

> *the highest deposition of acidifying compounds that will not cause chemical changes leading to long term harmful effects on structure and function of the most sensitive ecological systems according to present knowledge.*

Harmful effects are defined as:

> *chemical changes in forest soils and surface waters which may cause damage to ecosystem.*

Thus, a critical load is an indicator for the sustainability of an ecosystem in that it provides a value for maximum allowable load of one or more pollutants at which risk of damage or ecological changes to the most sensitive part an ecosystem is negligible.

The determination of critical loads has to be based on findings on the tolerance levels of aquatic or terrestrial organisms. These tolerance levels are influenced by a number of interacting processes. The influence of environmental factors (e.g. temperature, hydrology) as well as the status of the receptor (e.g. age, life stage) and the type of exposure (e.g. chemical form, exposure length) should all be taken into account in the determination of the critical chemical value which corresponds to the tolerance level of an organism [Bu92].

The method first focuses upon quantifying a pollutant threshold at which harmful effects occur to particular sensitive receptors (usually biological species). To calculate a critical load, biological effects are usually 'translated' to critical chemical values, e.g. harmful effects to fish 'translate' to alkalinity or aluminium concentrations in water. Therefore critical load calculations may be based upon the chemistry of a system [Bu92]. Since each element in the procedure depends on the preceding element, a change in indicator species or chemical limit will automatically change the critical load.

In this phase of the project, we will focus on the impacts on terrestrial ecosystems in general, and especially on forest ecosystems, since they are regarded as the most sensitive ecosystems with respect to the impacts of acidification and eutrophication [Sv93].

3.3.2.2 Environmental effects above the critical load

Due to complex relationships between different species within an ecosystem, it is difficult to assess the impacts on (the organisms of) an ecosystem quantitatively if the critical load of the ecosystem is exceeded. Impacts on vegetation and organisms may vary between ecosystems, depending on their characteristics. Areas of 'exceedance' indicate where present levels of pollutant deposition increase the risk of damage to ecosystems. By definition, exceedances of critical loads may lead to 'harmful effects', and the temporal aspect of these exceedances (i.e. estimating *when* these harmful effects will occur, or the 'damage time lag'), becomes important [Do93]. Therefore, only general qualitative statements about the expected impacts of acid pollution can be made.

Particular plant species will be affected by soil acidification when acidification changes the soil chemistry beyond the critical chemical value. The effect is manifested as decline in growth and increased mortality. Differences in the response arise from differences in the way that plants actually take up nutrients from the soil solution [Sv93].

The relation between root biomass and stem biomass for a number of trees indicates that the tree reallocates important nutrients to leaves and gives less priority to new root biomass production, when the tree is under stress of acidity. During dry periods, tree growth is already significantly reduced through unfavourable water conditions. In the case of soil acidification, additional stress by aluminium may push a tree below the minimum uptake limit required to sustain life. Conditions can become very unfavourable through low base- cation/aluminium ratio in the lower part of the rooting zone. The tree will try to compensate by relocating most of its root activity to the top layers of the soil. This implies that the tree would become more sensitive to water stress and more susceptible to damage by wind as its root mass becomes less in relation to crown mass, and as the roots do not penetrate the ground so well [Sv93].

3.3.2.3 Alternative indicators

Although the critical load is a widely used environmental indicator which has already proven its worth in abatement strategies and reduction protocols (see, for example, [Do93] or [RIVM93]), there are some alternatives. An alternative indicator could be the total deposition of (acidifying) pollutants. This indicator is a so-called stress or pressure indicator (see Section 4.3.1.2) and is sometimes used by policy-makers. A major disadvantage of a pressure indicator is that site-specific elements are not taken into account. Although the indicator is easily defined, the relation with environmental effects is not immediately visible. A pressure indicator such as the total amount of acidifying deposition seemed less suited for this project compared to a state or effect indicator like the critical load, since site-specific information has to be taken into account if possible.

3.3.3 *Calculation of the indicator*

In view of the previous discussion, we have chosen to base the indicator for acidification and eutrophication on the critical load concept. A complicating factor in calculating a critical load for acidity, is that nitrogen not only contributes to the acidification process, but also plays a role with respect to eutrophication. An assessment of the effects of nitrogen compounds on ecosystems must consider both acidification and eutrophication effects. The ecosystem is neutral with respect to whether its protection results from sulphur or from nitrogen deposition limitation. Trade-off between emissions of these two pollutants should be considered without exceeding the critical load of acidity **and** eutrophication [Do93]. Therefore a critical load has to be based on nitrogen deposition and sulphur deposition together, in this way protecting the ecosystem for both the impacts of acidification and eutrophication.

3.3.3.1 Calculation models

The assessment of critical loads can be based on calculations with several kinds of models of differing complexity. In general, a distinction can be made between (1) statistical and empirical analysis, (2) steady-state modelling and simple-process models, and (3) integrated process or dynamic modelling [Ho93]. While steady-state models provide the means for calculating critical loads for a variety of different ecosystems and provide results suitable for mapping purposes, dynamic models enable time response of systems to be considered. Dynamic models are also useful for investigating the consequences of deposition scenarios which exceed the calculated critical load values. Long-term effects, caused by the changes in soil chemistry due to (excess) acid deposition, and the corresponding damage time lag can only be assessed using these complicated and data intensive models. In addition, it is also possible to assess recovery time of an ecosystem (the 'recovery time lag') once emissions have sufficiently been reduced [Ho93, He92].

3.3.3.2 Calculation of the critical load

In 1993, a status report on the calculation and mapping of critical loads in Europe was published by the Coordination Center for Effects (CCE) of the RIVM [Do93]. The report summarizes the work of the CCE and National Focal Centers (NCFs) for mapping over 1991 and 1992. The report was produced for the United Nations Economic Commission for Europe's (UN ECE) Convention on Long-range Transboundary Air Pollution (LRTAP). The work was designed to support the negotiations within UN ECE on developing further protocols for reducing emissions of sulphur and nitrogen compounds.

One of the primary tasks of the CCE is to assist in the development and production of methods and maps of critical loads on a European scale in cooperation with the Task Force on Mapping. National Focal Centers for Mapping of 15 European countries have produced national maps of acidity and sulphur critical loads. A calculation method for critical loads of acidity for forest soils and surface waters was developed and is still being improved. Attention is focused on calculating critical loads for forest soils and surface waters because it is assumed that forest ecosystems and surface waters are the most sensitive ecosystems with respect to acidification [Sv93, Do93].

By using (part of) the results of the LRTAP project, a calculation method was adopted which is used by all of the National Focal Centers of the countries that participate in the project. In practice, this means that critical loads are determined for Europe, according to a uniform, well-defined and generally accepted calculation method [Do93]. This reduces the time needed to calculate the indicator value for acidification and eutrophication considerably, since results of the LRTAP project cover our purposes to a large extent.

The steady-state mass balance

The critical load is calculated using a steady-state mass balance method (see also Section 3.3.3.1). All sources of acidity have to be in balance against all sources of alkalinity in the ecosystem. At steady state, when no acidification occurs, the alkalinity input must counterbalance the acidification process. The assessment end-point is given by the base-cation/aluminium ratio [Sv93, Do93]. This BC/Al ratio depends on the uptake of base cations by plants or trees, chemical weathering, base-cation and acid deposition. Uptake will tend to decrease the base-cation (Ca+Mg+K) concentration; weathering and atmospheric deposition will tend to increase the base cation concentration, whereas acidification will tend to increase the Al concentration. The weathering rate is influenced by mineralogy, texture and hydrological properties of the soil [Sv93, La92]. A more thorough description of the calculation method is given in [He92] and [Do93]; in this report only a general overview of the main processes will be given.

The critical load of actual acidity, CL(A), is calculated according to [Do93]:

$$CL(A) = ANC_w - ANC_{l(crit)}$$

where:

ANC_w	=	the acid neutralizing capacity (ANC) produced by weathering
$ANC_{l(crit)}$	=	the acid neutralizing capacity (ANC) consumed by the maximum accepted alkalinity leaching at critical load.

The ANC is defined as the amount of strong acid (mole H^+) necessary to restore the pH to the reference pH [St91]. The general equation of the exceedance of the critical load of acidity, Ex(A), is given by:

$$Ex(A) = A_{dep} + A_{soil} - CL(A)$$

where:

A_{dep}	=	the total acidity deposition
A_{soil}	=	the total acidity produced by soil processes.

Different combinations of nitrogen and sulphur depositions are possible to achieve Ex(A) = 0. Therefore the equation Ex(A) = 0 is called the 'exceedance indifference curve'. In Figure 3.1 an example is given of a hypothetical exceedance indifference curve.

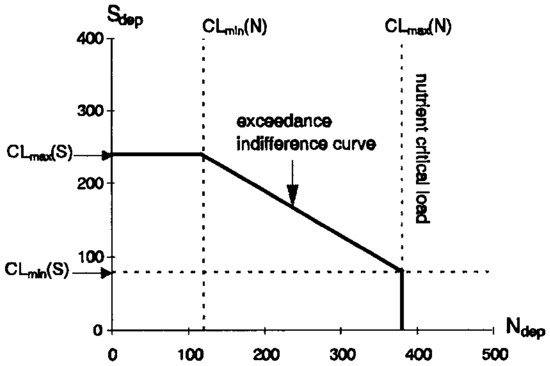

Figure 3.1 Relationship between the exceedance function and various sulphur and nitrogen depositions [CEC92].

If the combinations of sulphur and nitrogen depositions are found in the area below the exceedance indifference curve, the ecosystem is protected with respect to eutrophication and acidification. The critical load of sulphur ranges between a maximum $CL_{max}(S)$, which is the critical load of acidity including base-cation deposition less base-cation uptake, and a minimum $CL_{min}(S)$. The critical load for nitrogen ranges between a minimum $CL_{min}(N)$, which is based on the capacity of the ecosystem to remove nitrogen (uptake, immobilization, denitrification) and, by including nitrogen leaching, a maximum $CL_{max}(N)$ [Do93].

Critical load maps

By using the calculation method which is briefly described in this section, the critical loads for forest ecosystems and surface waters in Europe are calculated by the National Focal Centers (NFC) on a grid size ranging from 1 km^2 to approximately 225 km^2. For the Netherlands, critical loads and critical depositions for nitrogen and sulphur are calculated on a 100 km^2 grid size [Sa93], see Figures 3.2 and 3.3.

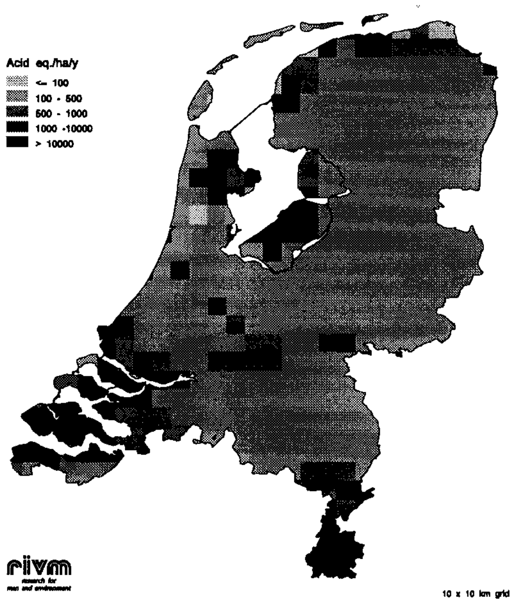

Figure 3.2 Critical depositions for sulphur [$Aeq.ha^{-1}.a^{-1}$] in the Netherlands calculated on a 100 km^2 grid [Sa93].

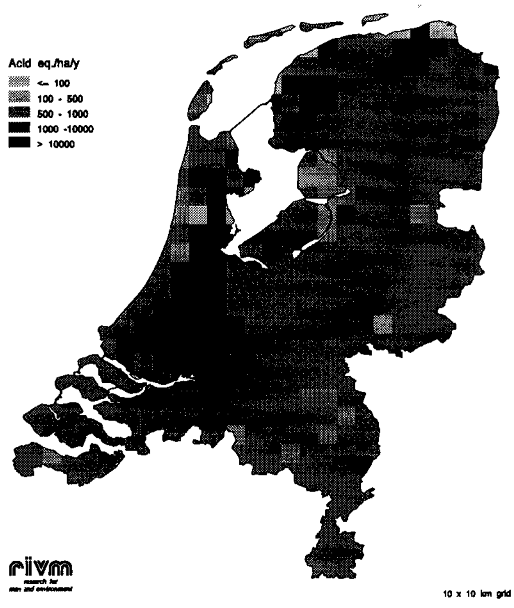

Figure 3.3 Critical loads for nitrogen [$mol.ha^{-1}.a^{-1}$] in the Netherlands calculated on a 100 km^2 grid [Sa93].

3.3.3.3 Calculation of the indicator value

Critical load maps are not directly suitable for use in a MAUA. The impacts on all grid cells have to be translated into a single value. First, a single value for one grid cell is calculated. It is proposed to calculate this indicator value of a certain grid cell according to:

$$\mathrm{Ind}_{CL,i} = \frac{\mathrm{deposition}_i}{\mathrm{critical\ load}_i} \times \mathrm{area}_i$$

where:

$\mathrm{Ind}_{CL,i}$	=	the value of the indicator [km^2] for acidification and eutrophication with respect to grid cell *i*
$\mathrm{deposition}_i$	=	the deposition of acid equivalents per unit area per year [Aeq.km^{-2}.a^{-1}]
$\mathrm{critical\ load}_i$	=	the critical load of grid cell *i* [Aeq.km^{-2}.a^{-1}]
area_i	=	the area of grid cell *i* [km^2]

Next, the indicator value for all grid cells is calculated by summing over the values of the single grid cell:

$$\mathrm{Ind}_{CL,tot} = \sum_i \frac{\mathrm{deposition}_i}{\mathrm{critical\ load}_i} \times \mathrm{area}_i$$

The deposition of acid equivalents can either exceed the critical load or can be below the critical load. Both cases can be summed separately. Since the advantage of this calculation method is not clear yet, it is proposed not to make a distinction between the two cases.

As explained in the previous section, each critical load 'value' consists of an exceedance indifference curve. The ratio between the deposition and the critical load could be calculated by taking the minimum 'distance' between the exceedance indifference curve and the total amount of acidifying deposition. The total amount of acidifying deposition, expressed in Aeq, is equal to the combined sulphur and nitrogen deposition. In this way, total effects of acidification and eutrophication due to nitrogen **and** sulphur depositions are taken into account.

This calculation process can be simplified by using separate critical loads for sulphur and nitrogen. By taking the nitrogen deposition or the sulphur deposition equal to zero, the critical loads for sulphur and nitrogen, respectively, is found (see Figures 3.1, 3.2 and 3.3). The ratio between the sulphur deposition and the critical load for sulphur, and the ratio between the nitrogen deposition and the critical load for nitrogen, are calculated separately. However, it should be noted that when the critical loads for sulphur and nitrogen are not exceeded, it is not necessarily true that the critical load for total acid deposition is not exceeded.

Since computation of the indicator using exceedance indifference curves is complicated and very time consuming, we decided to evaluate the impacts of nitrogen and sulphur separately. It should be noted, however, that using exceedance indifference curves instead of independent critical loads for sulphur and nitrogen is a better approach.

A modification of the indicator is the inclusion of the current level of SO_x and NO_x deposition (see also Section 2.6). In this case, the indicator is of the form

$$\mathrm{Ind}_{\mathrm{cl},i} = \frac{\text{deposition reference plant}_i}{\text{critical load}_i - \text{current deposition}_i}$$

where:

deposition reference plant$_i$ = SO_x or NO_x depostion [Aeq] in grid cell *i* resulting from the emissions of the reference plant

current deposition$_i$ = SO_x or NO_x deposition [Aeq] in grid cell *i* caused by the emissions of current acidifying sources (e.g. industry, road traffic, farming)

It should, however, be noted that simply summing over the grid is not reasonable, since both negative and positive numbers occur, indicating areas where effects are expected and where effects are not expected, respectively. Furthermore, problems arise if the current deposition becomes comparable with the critical load: a small change in either the critical load or the deposition results in large changes in the value of the indicator.

3.3.3.4 Available data

Critical load data on an ecosystem scale are not available yet. Critical load maps for NO_x and critical deposition maps for SO_x, calculated by the Dutch NFC, are available on a 10 x 10 km^2 grid [Sa93, Do93]. Current deposition levels (in $mol.ha^{-1}.a^{-1}$) of NO_x and SO_x for 1993, which can be used to compare the additional impact caused by the reference plant with the current level of pollution, are calculated by the Air Research Laboratory at the RIVM [Er95] (see Figures 3.4a and 3.4b). Main sources of emissions are industry, agriculture and traffic.

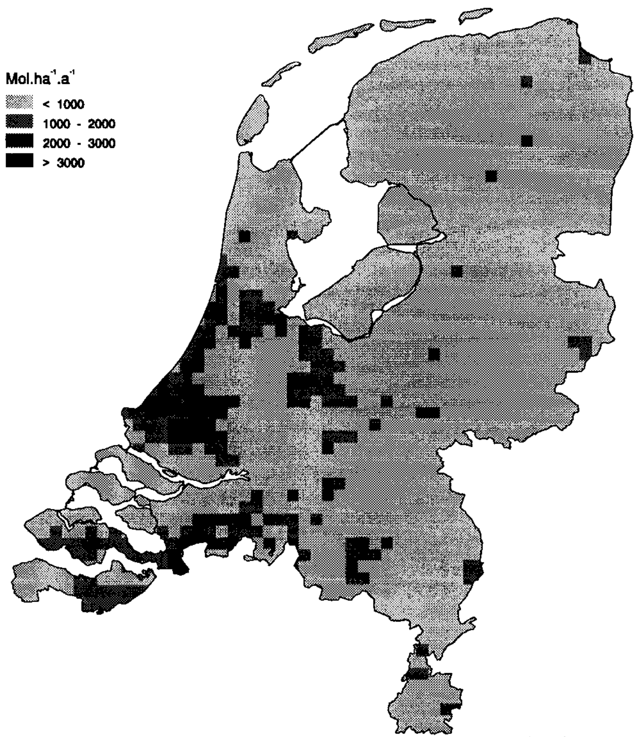

Figure 3.4a: SO$_x$ deposition [mol.ha^{-1}.a^{-1}] on a 25 km^2 grid due to emissions from industrial activities [Er95].

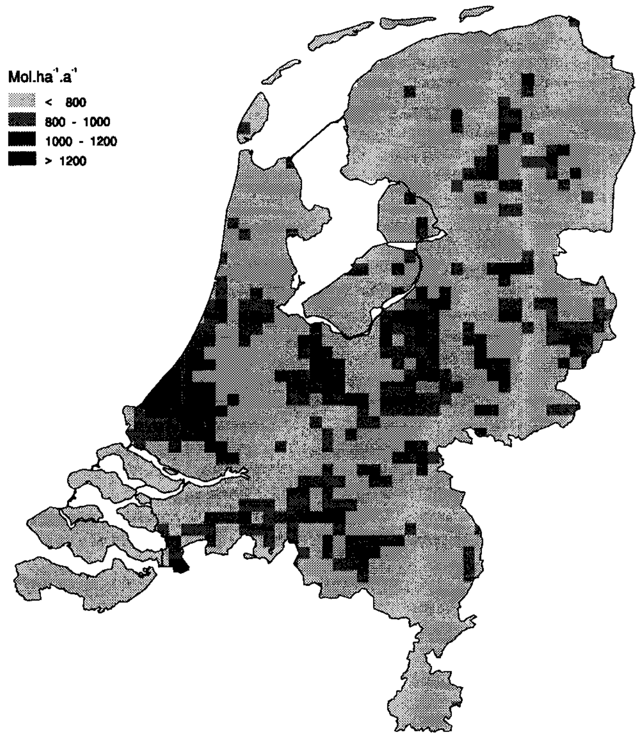

Figure 3.4b: NO$_x$ deposition [mol.ha^{-1}.a^{-1}] on a 25 km^2 grid due to emissions from industrial activities [Er95].

3.3.3.5 Calculation of the additional deposition

SO$_x$ and NO$_x$ depositions resulting from the releases by the reference power plant used in the coal fuel cycle are calculated with the OPS dispersion model (see Appendix A) [Ja91a]. To determine whether or not the location of the reference plant is important for the numerical values, the calculation is done for two reference sites. The first site is located at Amersfoort, situated in the centre of the Netherlands, and the second site in the north of Rotterdam.

Emissions are based on the emissions of the reference plant at Lauffen. About 0.8 g kWh$_{el}^{-1}$ SO$_2$, 0.8 g kWh$_{el}^{-1}$ NO$_x$ and 0.02 g kWh$_{el}^{-1}$ NH$_3$ are emitted by the 689.5 MW reference power plant [CEC92]. HCl is also emitted in small quantities, but the emission factor is still unknown. Emissions in the other stages of the coal fuel cycle are assumed to be negligible [CEC92]. Emissions of acidifying pollutants are not reported within the nuclear fuel cycle [Dr93]. Deposition of SO$_x$ and NO$_x$ for the two reference sites are shown in Figure 3.5.

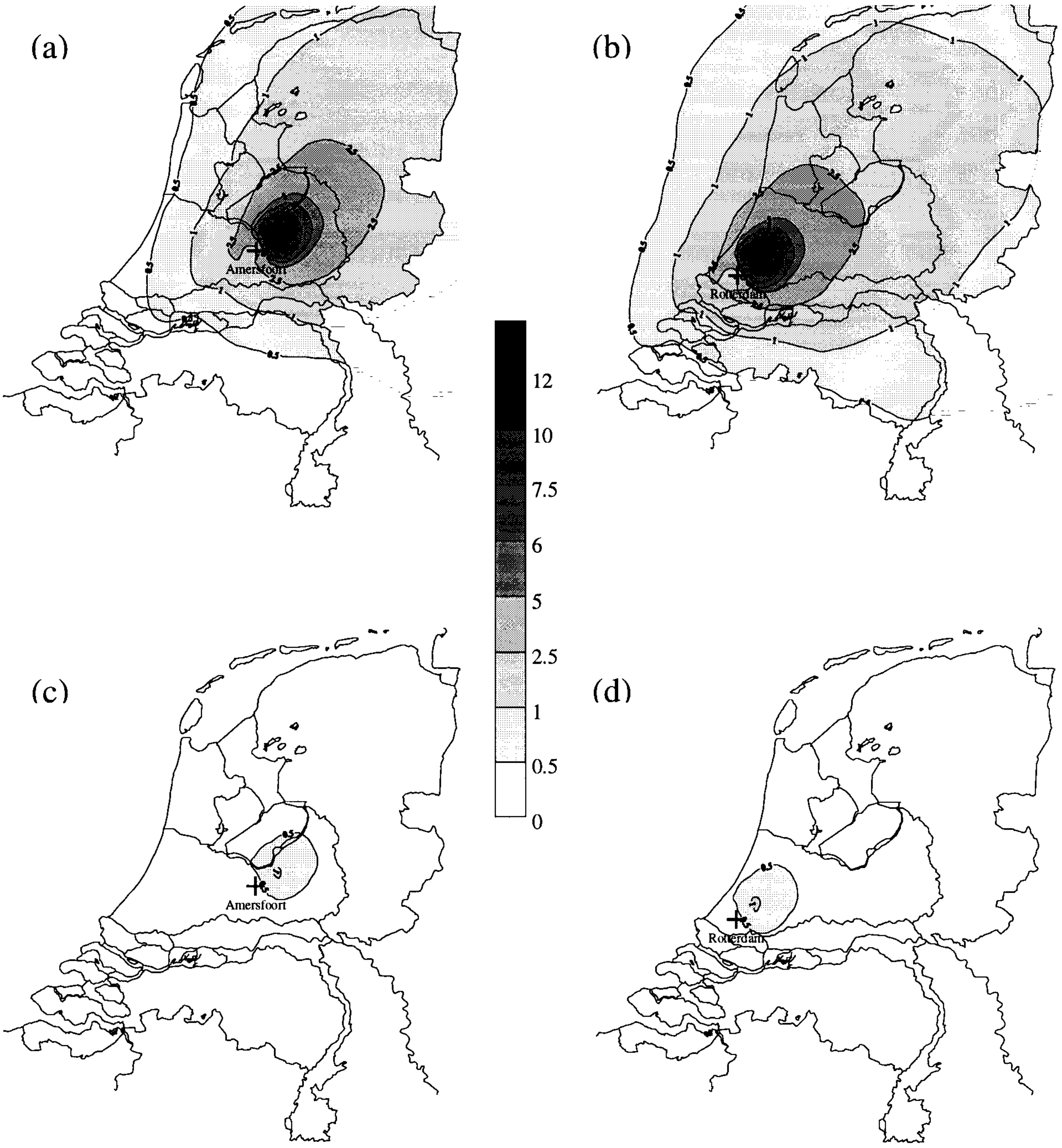

Figure 3.5 Deposition of SO_x [Aeq.ha^{-1}.a^{-1}] (a, b) and NO_x [Aeq.ha^{-1}.a^{-1}] (c, d) for a plant located at Amersfoort and Rotterdam.

3.3.3.6 Comparison with the critical load and critical deposition

To determine whether or not the critical depositions for sulphur and the critical loads for nitrogen are exceeded for each 10 x 10 km^2 grid cell, the ratio between the SO_x deposition and the critical sulphur deposition, and the ratio between the NO_x deposition and the critical load for nitrogen, are calculated. The results of the calculation are shown in Figures 3.6a and 3.6b.

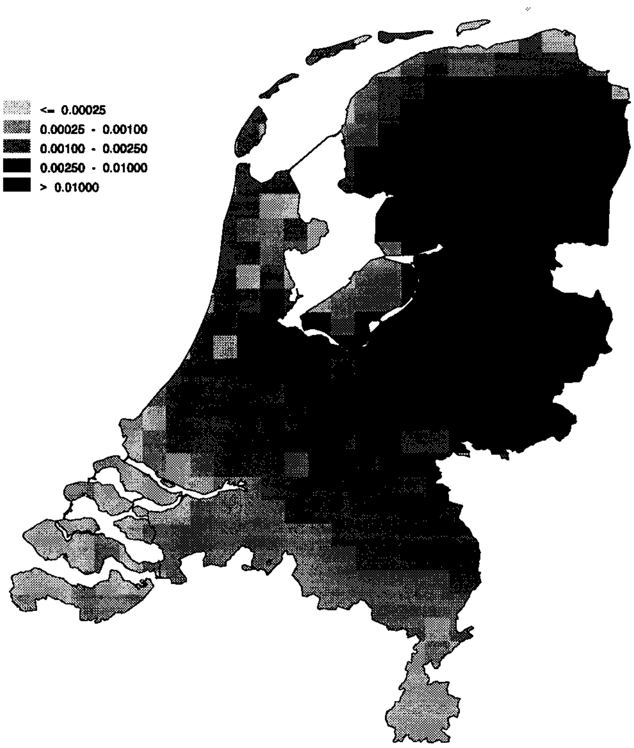

Figure 3.6a Ratio between the additional deposition and critical deposition for SO_x (location Amersfoort).

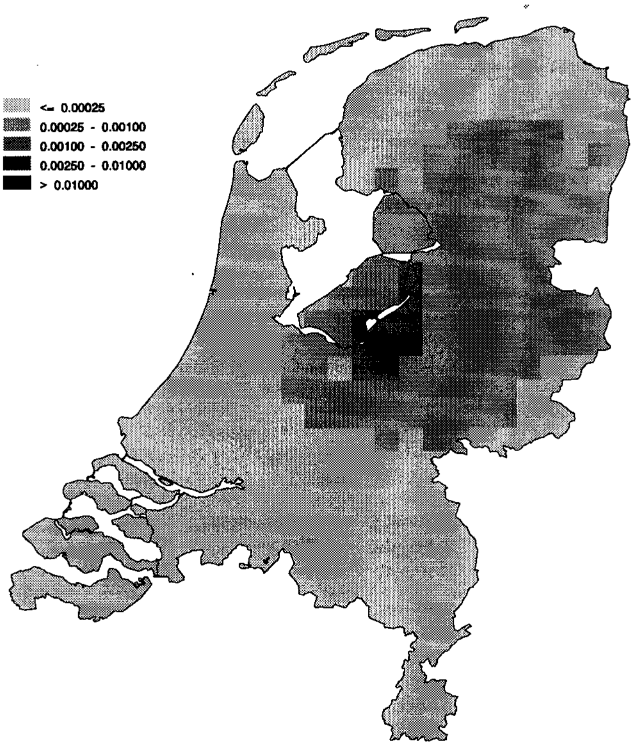

Figure 3.6b Ratio between the additional deposition and the critical load for NO_x (location Amersfoort).

It can be concluded that in each grid cell the deposition of SO_x and NO_x does not exceed the respective critical depositions for SO_x and the critical loads for NO_x. The indicator for SO_x is calculated at 2.1×10^{-3} km^2 MW^{-1} and for NO_x at 1.2×10^{-4} km^2 MW^{-1} for the reference plant at Amersfoort, and at 2.0×10^{-3} km^2 MW^{-1} for SO_x and at 1.1×10^{-4} km^2 MW^{-1} for NO_x with respect to the reference plant at Rotterdam.

The current level of SO_x and NO_x deposition is included in the alternative indicator (see Section 3.3.3.3). The indicator value is calculated for each grid cell. The results of the calculations for the reference plant at Amersfoort are shown in Figures 3.7a and 3.7b. Comparable illustrations can be made for the reference plant near Rotterdam but are not shown in this report.

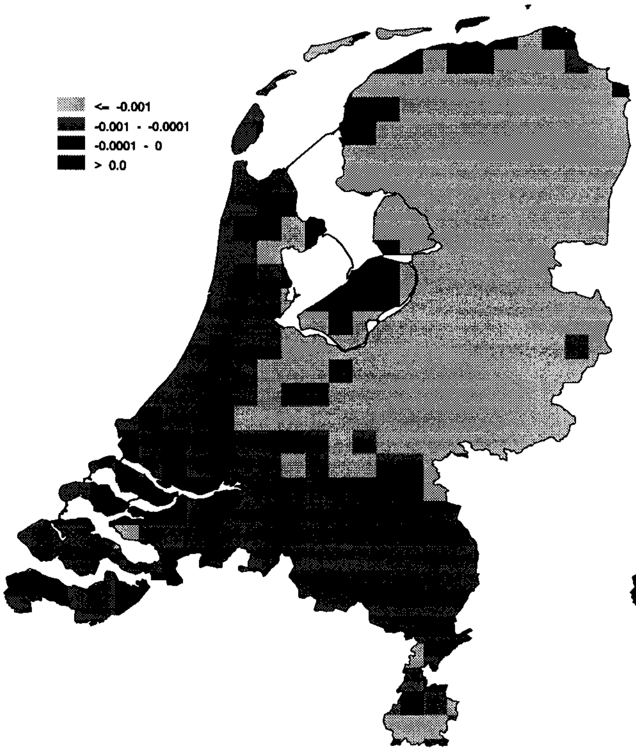

Figure 3.7a Ratio between the additional and critical deposition for SO_x, with the current level of pollution included.

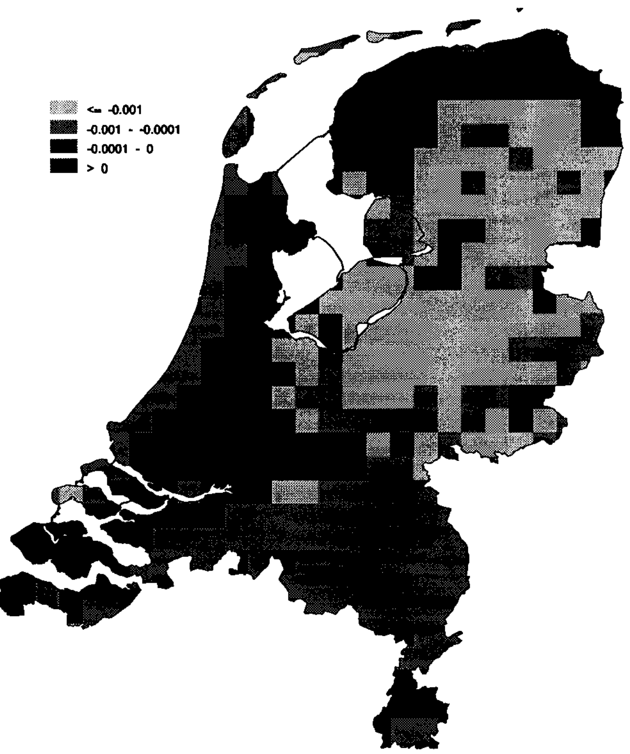

Figure 3.7b Ratio between the additional deposition and critical load for NO_x, with the current level of pollution included.

If the current level of pollution exceeds the critical load the value of the indicator is negative. As can be seen in Figures 3.7a and 3.7b this is the case for SO_x as well as NO_x for a significant part of the Netherlands. It is clear that the figure illustrates only the areas where the critical load is exceeded (negative nos.) or not (positive nos.). The figure is not suitable to deducing the contribution of the reference coal-fired power plant. Hence the indicator, based on current pollution, is not easily applicable.

3.3.3.7 Uncertainties

The confidence in modelled results and the reliability of the predicted output is influenced by various factors such as quality of the applied model, quality of the sampled data and the extrapolation and numerical treatment of the results [He92, Do93]. One of the causes of uncertainty is directly related to the problem of aggregation, i.e., to what level of detail should a system's process be modelled. Models should be designed to reflect the properties of the system. This means that data should be sampled such that ecosystem processes can be properly monitored and reflected. The number of sites to monitor, the stratification of the sites, and the level of detail of the system parameters to be sampled, may all contribute to potential errors in understanding a system.

The influence of environmental factors (e.g. temperature, hydrology) as well as the status of the receptor (e.g. age, life stage) and the type of exposure (e.g. chemical form, exposure time) should all be taken into account in the assessment of the critical chemical value. Because of this, there is inevitably a degree of uncertainty linked to any critical chemical value. Such values must be seen therefore as the best estimates from current scientific understanding. These estimates should always be considered subject to revision as scientific knowledge is updated. Complex synergistic (biochemical) processes caused by interactions with other pollutants such as NO_x and O_3 are not completely understood yet, but have to be incorporated [St93]. It is also important to consider the ecosystem response time-scale and the influences of temporal variations within the ecosystem. This process cannot be considered in steady-state models and is one of the major disadvantages in using this type of model [Fo92, St93].

Within the LRTAP project (see Section 3.3.3.2), an uncertainty analysis is performed using Monte Carlo simulation [Do93]. The weathering rate and the water flux are concluded to be the most important variables, which is consistent with other, independent, findings [Do93]. The current knowledge on the quality of the data for the two variables is not pessimistic about the reliability of the critical deposition computation results, although current knowledge on the quality of the data for base-cation deposition needs to be improved [Do93].

3.3.4 *Limitations*

Although the critical load concept is still in development, it has already proven to be a powerful tool in evaluating abatement strategies and protocols with respect to sulphur and nitrogen depositions [He92a]. Impacts on ecosystems can be assessed in a quantitative way. The ecosystem recovery time or the time lag in which harmful effects will occur can be calculated, but complex dynamic models and intensive data requirement are needed.

The biochemical processes controlling the chemical response of soil and surface water to sulphate deposition are reasonably well-known. In contrast, a consensus on biological consequences of surface-water acidification has not yet been developed, but new and better information is constantly becoming available. Both the chemical and biological implications of elevated nitrogen deposition for terrestrial ecosystems remain a matter of debate, largely because of the effects resulting from the complex set of biochemical transformations among nitrogen species [Ho93]. Synergistic processes and long-term effects of nitrogen need to be studied in more detail [St93].

3.3.5 *Conclusions and recommendations*

By adapting the calculation method and results of the LRTAP project described in Section 3.3.3.2 the impacts of the reference site on the environment can be estimated. By comparing the deposition resulting from the power station with the critical loads calculated by the Co-ordination Center for Effects and the National Focal Centers, potential areas at risk can be identified.

In the next phase of the project, it is recommended to quantify, where possible, the effects and the 'damage time lag' if the critical load is exceeded. This can be done by using dose–response functions and dynamic models. Exceedance indifference functions may be used as well. In this way the effects of acidification and eutrophication can be examined together.

3.4 Radionuclides

3.4.1 *Introduction*

In assessing the risk of the emission of radionuclides, calculating the impact is usually restricted to the damage to human health. The detrimental effects to the environment are neglected according to the principle cited in the *Recommendations of the international Commission on Radiological Protection* 'the standard of environmental control needed to protect man to the degree currently thought desirable will ensure that other species are not put at risk. Occasionally, individual members of non-human species might be harmed, but not to the extent of endangering whole species or creating imbalance between species.' [ICRP90].

This assumption has been generally accepted and adapted as policy on radiation protection. For example, cited in *Radiation protection and risk management*, the Dutch policy paper on the protection of the public and workers against ionizing radiation: 'Protecting individual human beings from radiation provides a sufficient safeguard for populations of plants and animals. Consequently, a separate risk limit is not required for this form of damage. Consideration should, however, be given to situations where plants and animals may be subjected to extremely high exposures.' [VR90]. Notable examples of regions where the environment can be exposed to high radiation levels without exceeding the dose limit for humans are very localized sites and regions normally not encountered by humans, like deep-sea dumping sites of radioactive waste.

The validity of the assumption is sometimes questioned, e.g. [IAEA92]. However, if the exposure of the most exposed humans is limited to 1 $mSv.a^{-1}$, as concluded, this will not lead to dose rates to plants and animals larger than 1 $mGy.d^{-1}$. Therefore specific radiation protection standards for non-human biota are not needed. Some situations, such as cases of prolonged exposure approaching 1 $mSv.a^{-1}$ of a human critical group, combined with the existence of specific ecological conditions such as the presence of rare or endangered species or combined stresses, may require site specific analysis [IAEA92]. In this section, only the impacts on the environment will be assessed.

In general, no-effect levels for irradiation of biota are based on field experiments. The no-effect level is set at a level such that no harmful effects will occur at a population level, although on an individual level some detriment in growth, age or reproduction may occur [IAEA92].

The emission of radionuclides in the energy fuel cycles is usually in inhabited regions and the radiation dose to humans ascribed to the continuous emissions of the fuel cycles is, in general, well below 1 $mSv.a^{-1}$. Nevertheless, the emission and accumulation of radionuclides emitted within the energy fuel cycles do contaminate the environment, which may lead to a decreased utility value for future generations. Therefore it is important to quantify the contamination of the environment and to develop an indicator to determine the impact on the environtment associated with the releases within the energy fuel cycles.

This chapter presents an environmental indicator for radionculides. The methods which have been developed for the calculation of the dose to humans will be used. Only the end-points in the dose assessment are different. The calculation process is described and discussed in Sections 3.4.2 and 3.4.3 and worked out for ^{129}I, one of the most important nuclides according to [Dr93]. The sample calculation is performed for an imaginary reprocessing plant located in the centre of the Netherlands, with a source term equal to the source term of Cap de la Hague, France. Finally, the results and conclusions are given in Section 3.4.4.

3.4.2 *Definition of the indicator*

In this section the general form of the environmental impact indicator for radionuclides is presented. The indicator is based on no-effect levels. However, other methods are discussed in literature to establish the acceptability of emissions of radionuclides to the environment [Am93]. The different parts of the indicator are reviewed briefly and several possible modifications of the indicator are discussed.

3.4.2.1 General form of an indicator based on no-effect concentrations

Environmental impact indicators which describe all significant effects within the nuclear and the coal fuel cycles should be mutually comparable. Therefore an environmental impact indicator for radionuclides of the following form is proposed:

$$\mathbf{Ind_{rad}} = \sum_{i}\sum_{j} \frac{\mathbf{Dose_{i,j}}}{\mathbf{NEL_j}} \times \mathbf{Area_j}$$

where:

Ind_{rad} = value of the indicator for radionuclides [km^2]

$\text{Dose}_{i,j}$ = dose to biota by radionuclide *i* in grid cell *j* resulting from the release from a plant [Gy.a^{-1}]

NEL_j = no-effect level (dose) in grid cell *j* [Gy.a^{-1}]

Area_j = area of grid cell *j* [km^2]

As previously shown, the indicator is based on the ratio between the dose to (concentration in) the environment and the no-effect level (concentration), weighted by the contaminated area. A sample calculation of the indicator is given for the radionuclide ^{129}I in Section 3.4.3.

3.4.2.2 No-effect levels for radiation

In [IAEA92] different dose limits are given for terrestrial and aquatic ecosystems. The dose limits are set for populations and not for individuals. It means that occasionally individual members of non-human species might be harmed, but not to the extent of endangering whole species or creating imbalance between species.

In [IAEA92] an acute exposure is described as an exposure 'which is delivered in a time period which is short compared with the time over which any obvious biological response develops', a chronic exposure is described as an exposure 'which could continue over a large fraction of the natural life of the organism'. Acute doses up to 0.1 Gy are very unlikely to produce persistent, measurable deleterious changes in populations or communities of terrestrial plants or animals [IAEA92]. Within the fuel cycles acute exposure will only occur during accidental situations. Chronic dose rates up to 1 $mGy \cdot d^{-1}$ do not cause observable changes in terrestrial animal populations. Up to this level adequate protection would therefore be provided. Chronic dose rates of 10 $mGy \cdot d^{-1}$ or less do not cause observable changes in terrestrial plant populations. Limitation of the dose rate to the maximally exposed individuals in the population to 10 $mGy \cdot d^{-1}$ provides adequate protection for populations in the aquatic environment [IAEA92].

On an IAEA report (Technical series no. 332, [IAEA92]), the suggestion is made to set the no-effect level for terrestrial animal and plant populations, and the maximally exposed individuals in the aquatic environment, at 1 $mGy.d^{-1}$, 10 $mGy.d^{-1}$ and 10 $mGy.d^{-1}$, respectively, for chronic irradiation. With respect to acute exposure, a dose limit of 0.1 Gy appears to be a sensible level for populations or communities of terrestrial plants or animals.

3.4.2.3 Some possible modifications of the indicator

In the indicator presented in Section 3.4.2.1 the only location-specific information contained is the ecosystem used, e.g. terrestrial or aquatic. However, a location-specific background radiation is present, which can be included in the indicator. Taking the dose resulting from current background levels into account, the indicator can be calculated as:

$$\mathrm{Ind}_{rad,bkgr} = \sum_i \sum_j \frac{\mathrm{Dose}_{i,j}}{\mathrm{NEL}_j - \mathrm{Dose}_{background,i,j}} \times \mathrm{Area}_{i,j}$$

where:

$\mathrm{Ind}_{rad,bkgr}$ = value of the indicator for radionuclides with inclusion of current background levels [km^2]

$\mathrm{Dose}_{background,i,j}$ = dose to biota in grid cell *j* resulting from current levels of background radiation of radionuclide *i* [$Gy.a^{-1}$]

Background radiation levels of the radionuclides released within the fuel cycles are not included in the definition of the indicator (see Section 3.4.2.1), since they are low compared to the no-effect level.

3.4.3 *Example calculation of the indicator*

The first step in the calculation of the indicator is to establish the source term. Next the dispersion of the radionuclides has to be modelled. Finally, the dose to biota has to be assessed. To illustrate the different steps, the calculation of the release of ^{129}I to air is worked out as an example. Grass, plants, cows, sheep and soil organisms are chosen as endpoints in the dose assessment. Grass, plants, cows and sheep are selected, since parameters in the uptake process of radionuclides by these organisms are relatively well known. Soil organisms are very suitable as bio-indicators [Kr92]. Further attention has to be paid to the representativeness of these organisms for an ecosystem. Finally, the results are compared to the calculations of [IAEA92].

3.4.3.1 Calculation of dispersion and deposition

The first step in the determination of the radioactivity indicator is calculating the concentration in air and the deposition rates of the radionuclides by using a suitable air dispersion model.

The emissions in the nuclear fuel cycle are given in [Dr93]. At the UP3 reactor of the reprocessing plant at Cap de la Hague 2.9 x 10^8 Bq ^{129}I per 900 MW reactor was emitted to air in 1991. Emissions of ^{129}I at other stages of the nuclear fuel cycle can be ignored [Dr93]. The concentration in air [$Bq.m^{-3}$] and the deposition rate of a radionuclide [$Bq.m^{-2}.a^{-1}$] are calculated with the OPS dispersion model [Ja91a]. The model parameters used in the calculation of the dispersion and deposition of ^{129}I are listed in Appendix B (Table B.1).

The concentration of ^{129}I in air [$Bq.m^{-3}$] and the deposition on soil [$Bq.m^{-2}.a^{-1}$], as calculated with OPS, are shown in Figures 3.8 and 3.9. The maximum air concentration is 5.7 x 10^{-7} Bq m^{-3} and the highest deposition is 9.2 x 10^{-2} $Bq.m^{-2}.a^{-1}$.

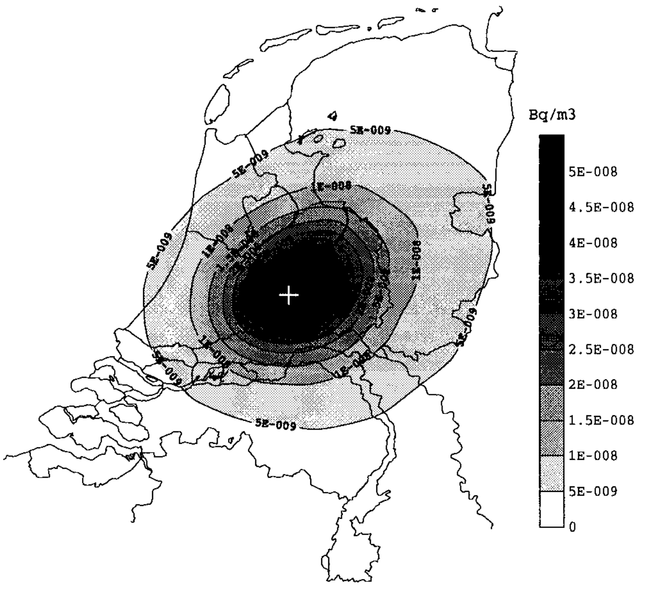

Figure 3.8 Concentration of ^{129}I in air [Bq.m^{-3}].

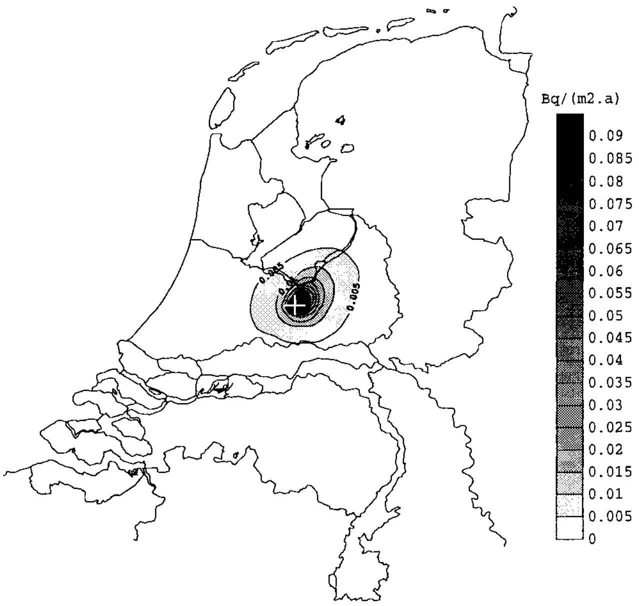

Figure 3.9 Deposition of ^{129}I on soil [Bq.m^{-2}.a^{-1}].

3.4.3.2 Calculation of the concentration in soil

Several models of different complexity are available to derive soil concentrations from depositions rates. The soil concentration is calculated by a relatively simple one compartment model as used in [IAEA82, La94].

It is assumed that the release of radionuclides into the air is continuous over a period of 30 years [Dr93]. The concentration of a radionuclide in soil Conc$_{soil}$(x,y,t) [Bq.kg^{-1}] is then described as in [IAEA82, La94]:

$$0 < t_{release} < 30 \text{ year} \qquad Conc_{soil}(x,y,t) = \frac{Dep(x,y)}{\rho \cdot \lambda \cdot h}\left(1 - e^{-\lambda t}\right)$$

$$t_{release} > 30 \text{ year} \qquad Conc_{soil}(x,y,t) = Conc_{soil}(x,y,30) \cdot e^{-\lambda t}$$

with

$$\lambda = \lambda_0 + \lambda_1$$

and

$$\lambda_1 = \frac{P}{\theta \cdot h \cdot \left(1 + \frac{K_d\, \rho_{soil}}{\theta}\right)}$$

where:

Dep(x,y) = the amount of a radionuclide deposited on the soil [Bq.m^{-2}.a^{-1}]
λ = removal of a radionuclide from the soil [a^{-1}]
λ_0 = radioactive decay [a^{-1}]; λ_0 = 4.41 x 10^{-8} a^{-1} for ^{129}I [ICRP83]
λ_l = removal from the soil by leaching [a^{-1}]
P = precipitation surplus; P = 0.35 m.a^{-1} [La94]
K_d = distribution coefficient [Bq kg^{-1}] x [Bq m^{-3}]$^{-1}$; K_d = 0.0 m^3.kg^{-1} for iodine [La94]
θ = relative moisture content [-]; θ = 0.2 [La94]
h = thickness of the soil layer [m]; h = 0.2 m [La94]
ρ_{soil} = dry density of the soil [kg.m^{-3}]; ρ_{soil} = 1300 kg.m^{-3} [La94]

The processes considered in the removal of radionuclides from the soil are radioactive decay and leaching only. Consequently, the concentration of long-lived radionuclides with high K_d-values can become very high after prolonged deposition times. For long time periods other processes like bioturbation and paedogenesis may become important. It is then useful to set a lower limit for λ_l, e.g. 10^{-2} a^{-1} [IAEA82]. However, since the time period for the calculation of the concentrations is limited to 30 years (see further), the use of a lower limit to λ_l is not necessary.

With the above equations, the soil concentration of a radionuclide can be calculated at different times. The concentration of ^{129}I in the soil is shown in Figure 3.10. The concentration reaches equilibrium within one year.

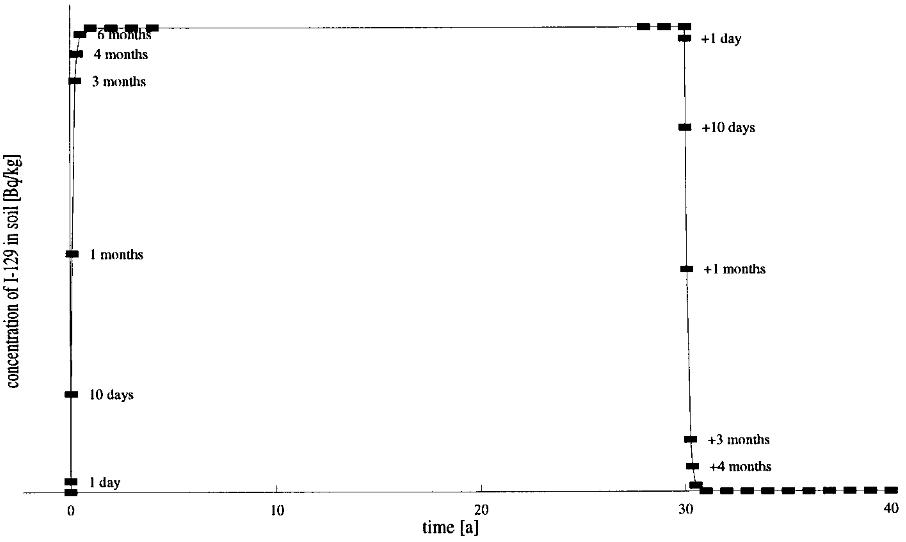

Figure 3.10 Concentration of ^{129}I in soil.

The concentration in soil for long-lived radionuclides depends strongly on the value of K_d. In [IAEA94], for example, a K_d value of about 2 x 10^{-3} $m^3.kg^{-1}$ for iodine is given; this enlarges the concentration of ^{129}I in soil with a factor of 85.

The value of the indicator can be calculated at different times. One could, for example, take an average value or calculate the value of the indicator at equilibrium. We have chosen to calculate the value of the indicator at t = 30 a, since the maximum concentration (in soil) due to 30 years of release will occur at that time. The concentration of ^{129}I in soil after 30 years of deposition is shown in Figure 3.11. The maximum concentration of ^{129}I in soil is 9.2 x 10^{-2} $Bq.kg^{-1}$.

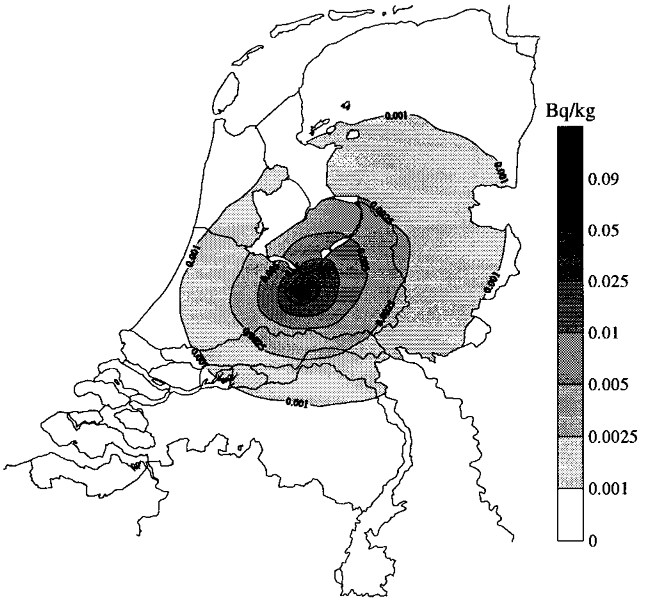

Figure 3.11 Concentration of ^{129}I in soil [$Bq.kg^{-1}$] after 30 years of deposition.

3.4.3.3 Calculation of radionuclide concentration in biota

Concentration of radionuclides in vegetation

Pathways contributing to the contamination of vegetation with radionuclides are foliar deposition and root uptake [IAEA82]. A straightforward way of calculating the concentration in vegetation is the use of a soil-to-plant transfer factor B_v, which integrates all interactions between soil and vegetation, together with a term which describes the contribution caused by foliar uptake [IAEA82, La94].

The concentration in vegetation caused by the transport of radionuclides from the soil to the plant is calculated as [La94]:

$$Conc_{veg,root} = B_v \times Conc_{soil}$$

where:

$Conc_{veg,root}$ = concentration in vegetation as a result of root uptake [$Bq.kg^{-1}$]
$Conc_{soil}$ = concentration in soil [$Bq.kg^{-1}$]

B_v = soil-to-plant transfer factor, for iodine 2 x 10^{-3} (Bq kg^{-1} fresh weight plants)(Bq kg^{-1} dry weight soil)$^{-1}$, based on the transfer factor 2 x 10^{-2} (Bq kg^{-1} dry weight plants)(Bq kg^{-1} dry weight soil)$^{-1}$ [IAEA94] and 10% dry weight, and 3.4 x 10^{-3} (Bq kg^{-1} dry weight grass)(Bq kg^{-1} dry weight soil)$^{-1}$ [IAEA94]

The concentration factor, B_v, depends on the radionuclide and vegetation considered. Special attention has to be paid to the dimensions of the concentration factor, since B_v can be based on either the fresh or the dry weight of the plant. The concentration in the plant on the basis of dry weight can be obtained by dividing the fresh weight concentration by the dry matter content. For grass and plants an average 10% dry matter is taken [IAEA94].

The concentration in a plant by foliar uptake of radionuclides is described by [IAEA82, La94]:

$$\mathrm{Conc}_{veg,fol} = \frac{\mathrm{Dep} \times \mathrm{R} \times \left(1 - e^{-\lambda_v t_e}\right)}{\mathrm{Y} \times \lambda_v}$$

where:

$\mathrm{Conc}_{veg,fol}$ = concentration in vegetation as a result of foliar uptake, fresh weight [Bq.kg^{-1}]

Dep = deposition of a radionuclide [Bq.m^{-2}.d^{-1}]

λ_v = effective decay rate for reduction of radionuclides deposited (d^{-1}); $\lambda_v = \lambda_0 + \lambda_w$ where λ_0 describes the radioactive decay and λ_w the removal by weathering

λ_0 = radioactive decay rate [d^{-1}]; $\lambda_0 = 4.41 \times 10^{-8}\ a^{-1} = 1.21 \times 10^{-10}\ d^{-1}$ for ^{129}I [ICRP83]

λ_w = weathering rate [d^{-1}]; $\lambda_w = 4.6 \times 10^{-2}\ d^{-1}$ for particulates (used in the calculation of the concentration in a plant), $\lambda_w = 6.9 \times 10^{-2}\ d^{-1}$ for iodine on pasture vegetation (used for the calculation of the concentration in grass) [IAEA82]

t_e = period of exposure [d]; 30 days for grass [La94]; the exposure time for plants is estimated at one year

R = fraction intercepted, $Y = 1 - e^{(-\mu Y)}$ [La94]

Y = standing crop biomass [kg m^{-2}]; Y = 7.4 for grass, 3.8 for plants (value used for vegetables) [La94]

μ = interception coefficient [m^2.kg^{-1}]; 2.9 for grass, 0.36 for plants (value used for vegetables) [La94]

Concentration of radionuclides in animals

Pathways contributing to the concentration of a radionuclide in animals are the consumption of food (grass with soil attached), water and inhalation. Although the inhalation pathway is taken into account for completeness, it can be ignored, except for accidental situations, due to the low contribution to the total dose [La94]. For a release of radionuclides into the air, the contamination of drinking water is a minor contributor to the total dose and is therefore not taken into account.

Cows and sheep ingest soil with the consumption of food. The intake of soil is estimated to be 4% of the dry food consumption for cows and 20% of the dry food consumption for sheep [La94]. The concentration in beef is then given by:

$$\mathbf{Conc_{fl,cow} = F_{fl,cow} \cdot Q_{veg,cow} \cdot \left(Conc_{grass} + 0.04 \times Conc_{soil}\right)}$$

and the concentration in mutton is given by:

$$\mathbf{Conc_{fl,sheep} = F_{fl,sheep} \cdot Q_{veg,sheep} \cdot \left(Conc_{grass} + 0.20 \times Conc_{soil}\right)}$$

where:

$Conc_{soil}$ = radionuclide concentration in soil [$Bq.kg^{-1}$]

$Conc_{gras}$ = radionuclide concentration in grass [$Bq.kg^{-1}$] (dry weight)

$Q_{veg,cow}$ = daily consumption in beef cattle [$kg.d^{-1}$]; $Q_{veg,cow} = 7.2$ $kg.d^{-1}$ dry matter [IAEA94]

$F_{fl,cow}$ = transfer coefficient to beef for iodine [$(Bq.kg^{-1})(Bq.d^{-1})^{-1}$]; $F_{fl,cow} = 4 \times 10^{-2}$ $d.kg^{-1}$ [IAEA94]

$Q_{veg,sheep}$ = daily consumption in sheep [$kg.d^{-1}$]; $Q_{veg,sheep} = 1.1$ $kg.d^{-1}$ dry matter [IAEA94]

$F_{fl,sheep}$ = transfer coefficient to mutton for iodine [$(Bq.kg^{-1})/(Bq.d^{-1})$]; $F_{fl,sheep} = 3.0 \times 10^{-2}$ $d.kg^{-1}$ [IAEA94]

The concentration in meat due to inhalation of contaminated air is calculated by [La94]:

$$\mathbf{Conc_{fl,inh} = F_{fl} \times Q_{air} \times Conc_{air}}$$

where:

$Conc_{fl,inh}$ = concentration in meat (from cattle or sheep) [$Bq.kg^{-1}$]

$Conc_{air}$ = concentration in air [$Bq.m^{-3}$]

F_{fl} = transfer coefficient for iodine [$(Bq.kg^{-1})(Bq.d^{-1})^{-1}$]; $F_{fl} = 4 \times 10^{-2}$ $d.kg^{-1}$ for cattle, $F_{fl} = 3.0 \times 10^{-2}$ $d.kg^{-1}$ for sheep [IAEA94]

Q_{air} = inhalation air [$m^3.d^{-1}$]; $Q_{air} = 200$ $m^3.d^{-1}$ for cattle [La94]

3.4.3.4 Conversion of concentrations into dose

The total dose delivered to biota consists of external and internal irradiation. External irradiation is caused by γ-emitters deposited on the ground and γ-emitters in the air. The contribution of α-emitters and β-emitters to external irradiation is neglected because of their short mean free pathway in air.

The contribution of ^{129}I to the external irradiation dose [Gy.a^{-1}] is calculated by [La93, Ko83]:

$$\mathbf{Dose_{ext,{}^{129}I} = 6.92 \cdot 10^{-12} \times \rho_{soil} \times Conc_{soil} + 1.16 \cdot 10^{-8} \times Conc_{air}}$$

where:

$Conc_{soil}$ = concentration in soil [Bq.kg^{-1}]
$Conc_{air}$ = concentration in air [Bq.m^{-3}]
ρ_{soil} = density of the soil [kg.m^{-3}]; ρ_{soil} = 1300 kg.m^{-3} [La94]

The formula is designed for the calculation of the radiation dose to humans. The crude assumption is made that the dose to animals and plants is comparable to the dose to humans. In the calculation of the internal dose, it is assumed that 100% of the energy of the α-particles and β-particles is absorbed by the biota. With respect to γ-radiation, it is assumed that plants and animals absorb 10% and 30% of the photon energy, respectively [IAEA92]. The internal dose $Dose_{int}$ [Gy.s^{-1}] to biota is now given by [IAEA92]:

$$\mathbf{Dose_{int} = Conc_{biota} \times E_{nuclide} \times f_{abs,nuclide}}$$

where:

$Dose_{int}$ = internal dose [Gy.s^{-1}]
$Conc_{biota}$ = concentration of the nuclide in biota [Bq.kg^{-1}]
$E_{nuclide}$ = decay energy of the nuclide (α, β en γ separately) [J]
$f_{abs,nuclide}$ = absorbed fraction of the radionuclide (α, β en γ separately) [-]

For ^{129}I, the internal dose to biota [Gy.a^{-1}] is therefore calculated by:

$$\mathbf{Dose_{int,{}^{129}I} = Conc_{biota} \times \left[f_{abs,\beta} \times 2.5 \cdot 10^{-7} + f_{abs,\gamma} \times 1.2 \cdot 10^{-7} \right]}$$

where:

$f_{abs,\beta}$ = absorbed fraction of the β-particle; $f_{abs,\beta} = 1$
$f_{abs,\gamma}$ = absorbed fraction of the γ-particle; $f_{abs,\gamma} = 0.1$ in plants and $f_{abs,\gamma} = 0.3$ in animals

3.4.3.5 Calculation of the dose on soil organisms

In the calculation of the impact of radionuclides on soil organisms, for example, a worm, it is assumed that the concentration in the soil organisms is equal to the concentration in the soil. It is assumed that the radionuclides are homogeneously distributed over the first 20 cm of the soil. Since part of the γ-radiation will not be absorbed in this soil layer, a correction was made, which, however, did not influence the total absorbed energy significantly. It is assumed that 100% of the energy of the α- and β-particles is absorbed by the organism. The dose on a soil organism is calculated by:

$$\mathbf{Dose}_{\text{soil org.}} = \mathbf{E}_{\text{soil org.}} \times \mathbf{Conc}_{\text{soil}} = \left(\mathbf{E}_{\alpha} + \mathbf{E}_{\beta} + \mathbf{E}_{\text{abs},\gamma}\right) \times \mathbf{Conc}_{\text{soil}}$$

where:

$\text{Dose}_{\text{soil org.}}$ = dose delivered to a soil organism [Gy.s^{-1}]

$E_{\text{soil org.}}$ = energy absorbed by the soil organism [J]

$\text{Conc}_{\text{soil}}$ = soil concentration [Bq.kg^{-1}]

E_{α} = energy of the α-particle [J]

E_{β} = energy of the β-particle [J]

$E_{\text{abs},\gamma}$ = energy of the γ-particle absorbed in the soil layer [J]

The dose conversion factors, based on a soil concentration of 1 Bq.kg^{-1} and calculated with the method described in this section, are listed in Appendix B (Table B.2).

3.4.3.6 Determination of the indicator value

The dose to grass, plants, cattle, sheep and soil organisms resulting from the emission of ^{129}I is calculated using the method described in Sections 3.4.3.1 to 3.4.3.5 (see Figures 3.12 to 3.16).

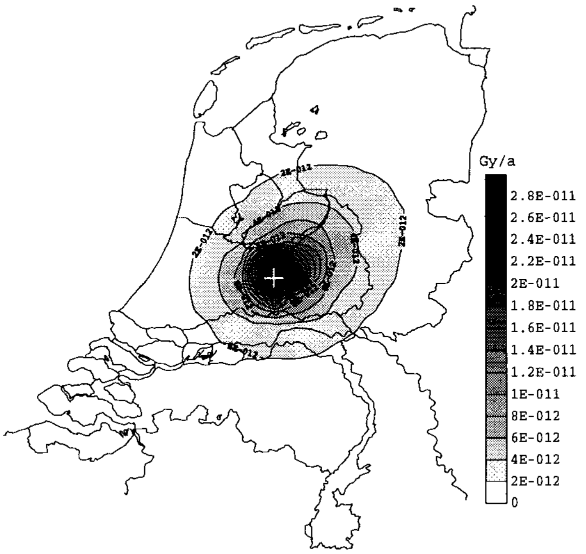

Figure 3.12 Dose of ^{129}I to grass [Gy.a^{-1}].

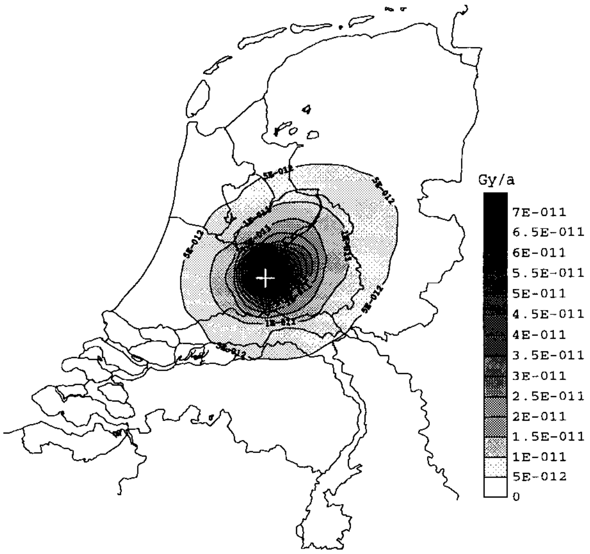

Figure 3.13 Dose of ^{129}I to plants [Gy.a^{-1}].

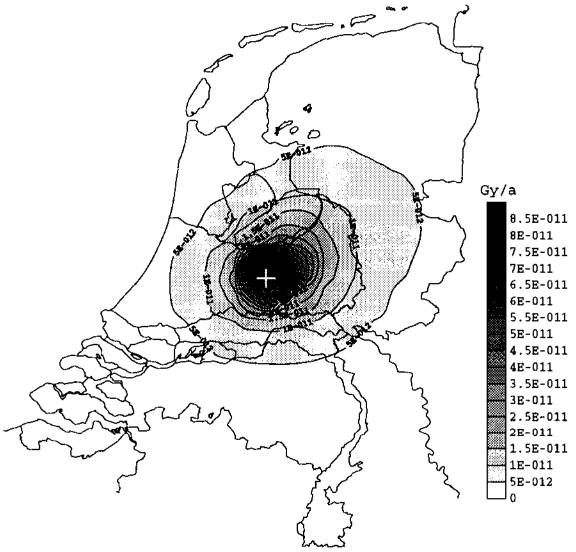

Figure 3.14 Dose of ^{129}I to cattle [Gy.a^{-1}].

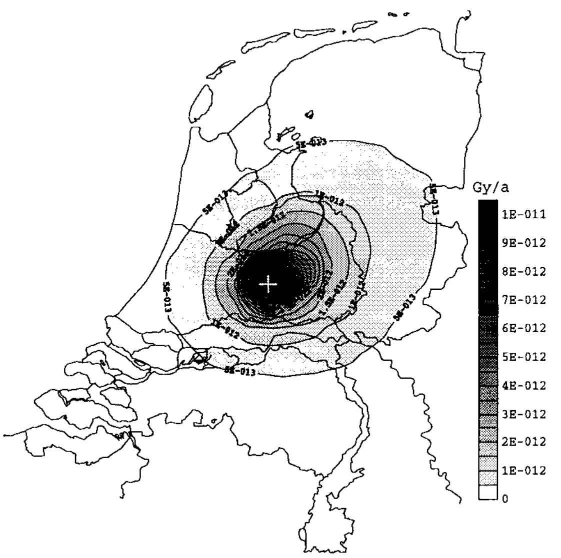

Figure 3.15 Dose of ^{129}I to sheep [Gy.a^{-1}].

The maximum dose to grass, plants, cattle, sheep and soil organisms is 1.1 x 10^{-10}, 2.9 x 10^{-10}, 3.4 x 10^{-10}, 4.2 x 10^{-11} and 1.1 x 10^{-15} [Gy.a^{-1}], respectively. This is well below the no-effect level of 0.37 Gy.a^{-1} [IAEA92]. The low value for the dose to soil organisms is due to the low K_d value of ^{129}I. By using the equation given in Section 3.4.2.1 and the no-effect concentrations presented in Section 3.4.2.2, the value of the indicator for ^{129}I is calculated at 1.3 x 10^{-7} km^2.MW^{-1} for plants and 1.7 x 10^{-6} km^2.MW^{-1} for cattle.

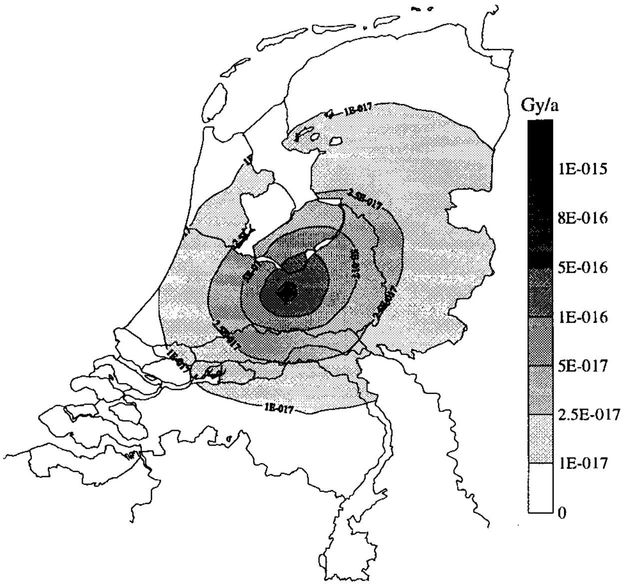

Figure 3.16 Dose of ^{129}I to soil organisms [Gy.a^{-1}].

3.4.3.7 Calculation method for ^{3}H and ^{14}C and noble gases

For some specific radionuclides, like ^{3}H and ^{14}C, the calculation of the radiation dose to biota can be based on the specific activity approach. We used the method outlined in [IAEA82].

^{3}H

In the specific activity approach described in [IAEA82] it is assumed that there is steady-state equilibrium between the environment and the biota considered. Therefore the concentration ratio between the radionuclide, ^{3}H, and the corresponding non-radioactive nuclide, ^{1}H, in biota is the same as in the environment. We assume that the release, transport and incorporation of ^{3}H is in the form of water. The average concentrations in air are calculated for ^{3}H with the OPS model. Since we assume that the relative concentration in biota is equal to the relative concentration in air, the concentration of ^{3}H in biota, C_{biota}, is equal to:

$$\mathbf{Conc_{biota}} = \frac{\mathbf{C_{air,H\text{-}3}}}{\mathbf{C_{air,H_2O}}} \times \mathbf{f_w} \times \mathbf{f_a}$$

where:

C_{biota} = the concentration in biota [Bq.kg^{-1}]

$C_{air,\ H\text{-}3}$ = the average concentration of ^3H in air [Bq.m^{-3}], calculated with OPS

$C_{air,\ H2O}$ = the concentration of water in air [kg.m^{-3}]; default 6 x 10^{-3} kg.m^{-3} [IAEA82]

f_w = the fraction of body water to the body weight; default 0.6

f_a = the fraction of the water content in the body attributed to the water in air; a conservative estimate is made by taking $f_a = 1.0$

The dose calculation is similar to the calculation for ^{129}I.

<u>^{14}C</u>

The ^{14}C released into the atmosphere is associated with CO_2 molecules and is subsequently fixed within plant tissues during photosynthesis. The organic molecules thus formed are transported along with stable ^{12}C through biota.

The concentration in biota $Conc_{biota}$ [Bq.kg^{-1}] is calculated by [IAEA82]:

$$\mathbf{Conc_{biota}} = \frac{\mathbf{Conc_{^{14}C,air}}}{\mathbf{Conc_{^{12}C,air}}} \times \mathbf{f_{biota}} \times \mathbf{f_{diet}}$$

where:

C_{biota} = concentration of ^{14}C in biota [Bq.kg^{-1}]

$Conc_{C\text{-}14,\ air}$ = concentration of ^{14}C in air [Bq.m^{-3}]

$Conc_{C\text{-}12,\ air}$ = concentration of airborne carbon [kg.m^{-3}], $Conc_{C\text{-}12,\ air} = 1.8 \times 10^{-4}$ kg.m^{-3} [IAEA82]

f_{biota} = carbon fraction in biota [-], f_{biota}, is estimated at 0.18 [Ca87]

f_{diet} = fraction of total dietary carbon [-], f_{diet}, is assumed to be 1.0

The dose calculation is similar to the calculation for ^{129}I.

<u>Noble gases</u>

For noble gases like ^{85}Kr and ^{133}Xe, external radiation is considered to be the major pathway. Other pathways are ignored. Calculation of the external radiation is similar to the calculation of the external radiation from ^{129}I. Air immersion coefficients are obtained from [Ko83].

3.4.3.8 Nuclear accidents

Apart from regular releases within the nuclear fuel cycle, releases of radionuclides during accidental situations have to be considered. Releases resulting from a nuclear accident are characterized by a low probability of occurrence and considerable consequences. The probability of emission of radionuclides during normal operation, however, is equal to one and consequences are few. As stated before, only impacts on biota are assessed.

Accidental and regular releases could be combined by summing the consequences (the amount of radionuclide released) multiplied by the probability of occurrence. The main disadvantage of such a probabilistic approach is that the real impact of a heavy nuclear accident is not assessed. It remains unknown whether no-effect levels will be exceeded, and if so, to what extent. To obtain an impression of the environmental impact of a release of radionuclides during an accidental situation, we performed a sample calculation of the consequences to the environment of a severe nuclear accident.

Source term

In [Dr93], four different source terms and their conditional probabilities are given for the reference nuclear power plant. The largest source term, ST_2, corresponds to a release of about 10% of the core of a 1200 MW_e reactor and has a probability of occurance of 4.3×10^{-6} a^{-1} [Dr93]. The radionuclides and activity released from ST_2 are given in Appendix B (Table B.3).

Since calculation of the environmental impact caused by all radionuclides released from ST_2 would be very time consuming, a radionuclide reduction programme, NUR-2, was used to select the radionuclides in ST_2 which are likely to contribute most to the environmental dose [Bl93]. The computer program is developed for calculating the acute dose to humans. Impact pathways are external radiation from the cloud, and from the soil and inhalation. The nuclides which contribute most to the dose delivered to humans are ^{132}I (35%), ^{135}I (27%), ^{133}I (15%), ^{134}I (10%), ^{131}I (5%), ^{88}Rb (3%) and ^{132}Te (1%).

Real-time calculation of the environmental impact

As an example, total deposition and air concentration, based on source term ST_2, are calculated at a distance of 1.15 km with the PC-COSYMA model [Jo93]. Exposure pathways, with respect to acute exposure, are external radiation from the cloud and soil (animals and plants), interception (plants), inhalation (animals) and ingestion (animals). Since a release height of 10 m and a heat content of the plume of 0 MW was used [Dr93], maximum concentration and deposition are found close to the source (within 1 kilometer). Default values were used for weather conditions. Source parameters, model parameters and model results are given in Appendix B (Table B.4 and B.5).

The total acute dose is calculated by:

$$\mathrm{Dose}_{\mathrm{tot,acute}} = \mathrm{Dose}_{\mathrm{ext,cloud}} + \mathrm{Dose}_{\mathrm{ext,soil}} + \mathrm{Dose}_{\mathrm{interc}} + \mathrm{Dose}_{\mathrm{ingest}} + \mathrm{Dose}_{\mathrm{inhal}}$$

with

$$\mathrm{Dose}_{\mathrm{ext,cloud}} = \mathrm{Conc}_{\mathrm{air}} \times \mathrm{Im}_{\mathrm{air}}$$

$$\mathrm{Dose}_{\mathrm{ext,soil}} = \left[\int_0^{t_{\mathrm{acute}}} \mathrm{Dep}_{\mathrm{soil}} \cdot e^{-\lambda_0 t}\, dt\right] \times K_{\mathrm{soil}} = \frac{\mathrm{Dep}_{\mathrm{soil}} \cdot \left(1 - e^{-\lambda_0 t_{\mathrm{acute}}}\right)}{\lambda_0} \times K_{\mathrm{soil}}$$

$$\mathrm{Dose}_{\mathrm{interc}} = \int_0^{t_{\mathrm{acute}}} \left(\frac{\mathrm{Dep}_{\mathrm{soil}} \times R}{Y} \times e^{-\lambda_0 t} \times E_{\mathrm{nucl,abs}}\right) dt$$

$$= \frac{\mathrm{Dep}_{\mathrm{soil}} \times R}{Y} \times \left(\frac{1 - e^{-\lambda_0 t_{\mathrm{acute}}}}{\lambda_0}\right) \times \sum_{\alpha,\beta,\gamma} \left(E_{\mathrm{nucl},\alpha,\beta,\gamma} \times F_{\mathrm{abs}}\right)$$

$$\mathrm{Dose}_{\mathrm{ingest}} = \left(\int_0^{t_{\mathrm{acute}}} \int_0^{t'} \frac{\mathrm{Dep}_{\mathrm{soil}} \times R \times Q_{\mathrm{grass}}}{Y \times \mathrm{Mass}_{\mathrm{cow}}} \times e^{-\lambda_0 t'}\, dt\, dt'\right) \times E_{\mathrm{nucl,abs}}$$

$$= \frac{\mathrm{Dep}_{\mathrm{soil}} \times R \times Q_{\mathrm{grass}}}{Y \times \mathrm{Mass}_{\mathrm{cow}} \times \lambda_0} \times \left[\frac{1}{\lambda_0} - \frac{e^{-\lambda_0 t_{\mathrm{acute}}}}{\lambda_0} - e^{-\lambda_0 t_{\mathrm{acute}}}\, t_{\mathrm{acute}}\right] \times \sum_{\alpha,\beta,\gamma} \left(E_{\mathrm{nucl},\alpha,\beta,\gamma} \times F_{\mathrm{abs}}\right)$$

$$\mathrm{Dose}_{\mathrm{inhal}} = \int_0^{t_{\mathrm{acute}}} \left(\frac{Q_{\mathrm{air}} \times \mathrm{Conc}_{\mathrm{air}}}{\mathrm{Mass}_{\mathrm{cow}}} \times e^{-\lambda_0 t} \times E_{\mathrm{nucl,abs}}\right) dt$$

$$= \frac{Q_{\mathrm{air}} \times \mathrm{Conc}_{\mathrm{air}}}{\mathrm{Mass}_{\mathrm{cow}}} \times \left(\frac{1 - e^{-\lambda_0 t_{\mathrm{acute}}}}{\lambda_0}\right) \times \sum_{\alpha,\beta,\gamma} \left(E_{\mathrm{nucl},\alpha,\beta,\gamma} \times F_{\mathrm{abs}}\right)$$

where:

$\mathrm{Dose}_{\mathrm{ext,acute}}$ = acute dose due to external radiation [Gy]
$\mathrm{Dose}_{\mathrm{ext,cloud}}$ = dose due to external radiation from the cloud [Gy]
$\mathrm{Dose}_{\mathrm{ext,soil}}$ = dose due to external radiation from the soil [Gy]
$\mathrm{Dose}_{\mathrm{interc}}$ = dose to plants due to foliar uptake [Gy]
$\mathrm{Dose}_{\mathrm{ingest}}$ = dose to cattle due to ingestion [Gy]
$\mathrm{Dose}_{\mathrm{inhal}}$ = dose to animals due to inhalation [Gy]

$Conc_{air}$ = time-integrated concentration of the radionuclide in air $[Bq.s.m^{-3}]$

Dep_{soil} = deposition on the soil $[Bq.m^{-2}]$

Im_{air} = effective dose-rate conversion factor for air immersion $[(Gy.s^{-1}).(Bq.m^{-3})^{-1}]$

K_{soil} = effective dose-rate conversion factor for ground-surface exposure $[(Gy.s^{-1}).(Bq.m^{-2})^{-1}]$

$E_{nucl,abs}$ = total energy energy absorbed in biota (α-, β-, and γ-energy) [J]

$E_{nucl,\alpha,\beta,\gamma}$ = total emitted decay energy of the nuclide (α-, β-, and γ-energy) [J]

F_{abs} = absorbed fraction of the decay energy; $F_{abs} = 0.3$ for γ-particles, $F_{abs} = 1$ for α and β particles

Q_{air} = inhalation rate $[m^3.s^{-1}]$; $Q_{air} = 200\ m^3.d^{-1}$ for cattle [La94]

Q_{grass} = consumption of grass by cattle $[kg.s^{-1}]$ (fresh weight); $Q_{grass} = 70\ kg.d^{-1}$

λ_0 = radioactive decay $[s^{-1}]$

t_{acute} = period of exposure [s]; for acute exposure t_{acute} is set at 1 day

$Mass_{cow}$ = mass of a cow [kg]; $Mass_{cow} = 500$ kg [IAEA92]

Y = standing crop biomass $[kg.m^{-2}]$

R = fraction intercepted [-]

It is assumed that the intercepted deposition is distributed homogeneously in the plants and that the radionuclides inhaled and ingested are distributed homogeneously in animals. This is a conservative estimation and may lead to an overestimation of the total dose. It is also assumed that the biological half-life is larger than the integration time t_{acute} and can be neglected.

Air immersion factors and conversion factors for external radiation from the soil are again obtained from [Ko83]. E_{nucl} is obtained from [ICRP83]. The total acute dose, based on deposition values and air concentrations as given in Appendix B (Table B.5) is presented in Table 3.1. the most relevant exposure pathways are interception for plants, and ingestion and external radiation from the cloud for animals.

As for chronic exposures, the effects of acute exposure will be evaluated by comparison with a no-effect level. As stated before, the no-effect levels we used for radiation are based on population protection and not on protection of individuals. In [IAEA92] it is stated that acute doses of 0.1 Gy or less are very unlikely to produce persistent, measurable deleterious changes in populations or communities of terrestrial plants or animals. The total dose delivered to biota can, in the case of a severe nuclear accident, be about a factor of 200 larger than the no-effect level.

Table 3.1 Total acute dose due to inhalation and external radiation

Radionuclide	Acute dose delivered to animals [Gy]	Acute dose delivered to plants [Gy]
^{88}Rb	1.7×10^{-2}	6.5×10^{-2}
^{132}Te	1.1×10^{-1}	7.6×10^{-1}
^{131}I	4.1×10^{-1}	3.6
^{132}I	4.1×10^{-1}	3.2
^{133}I (1)	9.3×10^{-1}	9.3
^{134}I	4.2×10^{-2}	2.2×10^{-1}
^{135}I	6.1×10^{-1}	5.4
Total	2.5	2.3×10^{1}

(1) The contribution of ^{133}Xe, a daughter product of ^{133}I, is not taken into account.

3.4.4 *Discussion*

In [IAEA92] dose calculations are performed for ^{131}I released to air. The equilibrium concentration in the soil and an upper estimate of the dose rate to plant and animal tissues are calculated. To compare the results of the dose assessment of [IAEA92] with the method described in Section 3.4.3, a calculation for ^{131}I has been performed. The deposition rate is normalized to 1 Bq.m^{-2}.a^{-1}. The results of the calculations are shown in Table 3.2.

Table 3.2 Chronic dose rates to plant and animal tissues calculated according to [IAEA92] and the method described in Section 3.4.3, with results for a deposition of 1 Bq m^{-2} a^{-1}

Nuclide	Deposition rate [Bq.m^{-2}.a^{-1}]	Dose rate to plant tissues [Gy.a^{-1}]	Dose rate to animal tissues [Gy.a^{-1}]	Calculation method
^{129}I	1	3.0×10^{-9}	3.9×10^{-9} (c)	Section 3.4.3
^{131}I	1	4.7×10^{-9}	9.8×10^{-9} (c)	Section 3.4.3
^{131}I	1	2.9×10^{-8} (*)	1.40×10^{-9} (*)	[IAEA92]

(*) upper estimate
(c) cattle

Differences between the dose delivered by ^{131}I and ^{129}I can be attributed to differences in half-life which results in a change in removal rate (λ) of the radionuclide from the soil, and to a difference in energy of the emitted β-particles and γ-radiation. The dose rate to plant and animal tissues, calculated with the method described in [IAEA82], differs by a factor of 6 and 0.14, respectively, compared to the dose calculated using the method in Section 4.3.4.3. These differences are probably due to the parameter values selected, and give a first impression of the uncertainties associated with these calculations.

It is likely that the plants or animals taken as the assessment end-points (grass, plants, cattle, sheep and soil organisms) in the method described in Section 3.4.3 are not the most exposed species, which will lead to an underestimation of the maximum dose delivered to biota. Nevertheless, the end-points in the assessment are diverse and the calculated doses for regular releases are well below the no-effect level.

3.4.5 *Conclusions*

The dose to biota resulting from a release of 2.9×10^8 Bq a^{-1} of ^{129}I to air does not lead to an exceedance of the no-effect level. The maximum dose is about a factor of 10^9 below the no-effect level. Although releases during accidental situations still not taken into account and the calculations are performed for ^{129}I only, it is very likely that within the nuclear fuel cycle the chronic emission of radionuclides to air during normal operation will not significantly affect the environment.

In the case of a major nuclear accident calculations indicate that no-effect levels are likely to be exceeded in an area close to the reference plant. The total dose delivered to biota at a distance of about 1 km from the source is estimated at about 20 Gy, which means that the no-effect level is exceeded by a factor of 200. It is recommended to focus more on comparing the consequences of a major nuclear accident with emissions during normal operation.

3.5 Heavy Metals

3.5.1 *Introduction*

Emission of heavy metals into the environment takes place as a result of different anthropogenic processes, such as burning fossil fuel and waste, and refining petrol, as well as traffic. The release of heavy metals as a result of electricity generation may lead to undesirable high concentrations in soil, surface or drinking waters.

Hard coal is originally rich in heavy metals. When it is burned, some of the heavy metals, like Hg, Se and Pb, evaporate and are released into the air. After their deposition, some leach into the soil and surface water. They may accumulate in food chains, thus affecting the health of animals and humans [St91]. Since the radiotoxicity of radioactive heavy metals is always higher than their chemical toxicity and since the amount of radioactive heavy metals is very small compared to the total of heavy metals emitted, consequences of their emissions are incorporated into the indicator for radionuclides only [Eg93].

First, an overview of the heavy metals which may be released to the environment is given. Next, processes on which the indicator can be based are reviewed. In Section 3.5.4 target values on which the indicator value is based are given, while Section 3.5.5 presents the calculation method for the indicator with respect to soil contamination. The limitations of the proposed method, and the conclusions and recommendations are given in Sections 3.5.6 and 3.5.7, respectively.

3.5.2 *Emissions of heavy metals in the fuel cycles*

The coal fuel cycle

Emissions of heavy metals within the coal fuel cycle are not given [CEC92]. Nevertheless, it is likely that a certain amount of the heavy metals present in the hard coal, is released into the environment [Bl91]. The amount of trace elements in German hard coal as used in the reference plant is given in Appendix C. To reduce the emission of trace elements, the reference plant is equipped with electrostatic dust precipitators. The percentage with which the emission is reduced is not specified.

The emission to air by the power plant, leaching and drainage of heavy metals from refuse piles of hard coal and mine water, which is pumped into surface waters, are the major impact pathways. No data is available regarding the contamination of ground water and surface waters caused by the release of heavy metals at the mining site.

No data are presented in [CEC92]. In this stage of the project only the impact of heavy metals emitted to air will be assessed because of a lack of data with respect to emissions to groundwaters and surface waters from mining.

The nuclear fuel cycle

No non-radioactive emissions of heavy metals within the nuclear fuel cycle are reported [Dr93]. It is likely that contamination of ground water and surface waters caused by leaching and draining of heavy metals from refuse piles or mining sites will take place if no countermeasures, such as leakage trays, are taken. Contamination of the environment by radioactive elements is discussed in Section 3.4.

3.5.3 *Definition of the indicator*

In Europe, generally accepted indicators describing the impacts of heavy metals on soil and ground water have not yet been developed. Several countries have standards for the maximum amount of heavy metals in agricultural products or legislation with respect to the cleanup of polluted soils. The starting-point, however, may differ. Legislation can, for example, be developed for protection of all environmental species, including humans, or for the protection of human health only, without avoiding environmental hazards [Vi93].

It is widely accepted that (negative) impacts of heavy metals on plants, animals or humans do not occur below a threshold value, the so-called no-effect level (NEL). For each ecosystem, a different NEL can be found, since it depends, for example, on the properties of the soil [Me90, Be91a, DGM93, Be95]. Comparison between the concentration of heavy metals in surface waters or soil and a concentration below which no harmful effects occurs, seems an acceptable starting-point for the calculation of the indicator value. In this way, site-specific information is incorporated and the impact on ecosystems or humans can be assessed. Furthermore, the indicator is comparable with the indicator for acidification and contamination with radionuclides.

An alternative strategy would be to define an indicator for heavy metals based on emission equivalents, the total emission within the fuel cycle. Although the calculation of this indicator is very straightforward compared to other options, site-specific elements are neglected and the relation to environmental damage is not obvious. Therefore, the impacts of heavy metals will be assessed using an chemical state indicator based on soil concentrations which are set to the corresponding no-effect level.

3.5.4 *Target values*

In the Netherlands, legislation has recently been developed on target and intervention values for heavy metals in soil, sediment and surface waters. The underlying idea is sustainable development, thus avoiding environmental damage now and in the future [VR91, DGM93, Ja94, Be95].

For human beings, the maximum tolerable concentration is equal to the threshold value below which no harmful effects occur. In the assessment of the maximum tolerable concentration for an ecosystem, an extrapolation model is used. Since the relation between the concentration of a harmful substance and the percentage of protected species is asymptotical, a protection limit is set, such that 95% of the species is protected [DGM93].

Until we have come across a more consistent or accepted approach, Dutch target values based on maximum tolerable risk levels will be used in the calculation of the indicator value. Dutch target values for concentrations of heavy metals in a reference soil are given in Table 3.3.

Table 3.3 Target values for some heavy metals in a reference soil in mg.kg^{-1} dry matter [DGM93]

Heavy metal	Target value
Cadmium	0.8
Mercury	0.3
Copper	36
Nickel	35
Lead	85
Zinc	140
Chromium	100
Arsenic	29

Necessary ecotoxicological data are, with respect to surface waters, generally available in sufficient quantity. For sediment and ground water, protection standards can be derived from standards for surface waters by using an equilibrium partition method when sufficient ecotoxicological data are not available. For soil, present background concentrations in relatively unburdened areas may exceed the no-effect concentrations for some heavy metals [Me90, DGM93]. Under these conditions environmental quality standards for soil can be based on background concentrations.

Due to complex relationships between species in an ecosystem, it is difficult to assess the impacts quantitatively when the concentration of a heavy metal exceeds the no-effect level. Therefore only general qualitative statements about the expected impacts can be made.

As stated before, at least 95% of the ecosystem will be protected if the concentration of a heavy metal is below the target level. This does not necessarily mean that the most sensitive species within an ecosystem are sufficiently protected [Be91a].

3.5.5 *Calculation of the indicator*

In this phase of the project, the only indicator derived is that for soil contamination. The indicator for contamination of sediment and surface waters can be calculated using a comparable assessment procedure.

A separate value for the indicator can be calculated for each heavy metal. The calculation method described in [Ja92] to calculate the soil concentration based on a deposition rate is equal to the calculation method used for radionuclides (see Section 3.4). K_d values can be found in, for example, [La93, La94, Sh90]. It should be noted that the uncertainty in the K_d values may considerably influence the outcome of the assessment [Sh90]. Impacts on an ecosystem are strongly related to the K_d value, since it is assumed that the impact of a substance is determined by the concentration in the water phase [Be91a].

To compare soil concentrations with target values, the soil concentration of metal *k* in area *i* must be transformed into a soil concentration for the standard soil [Be91a]. This standard soil is defined as a soil which contains 25% lutum and 10% organic dust. Transformation into concentrations for a standard soil can be done using [DGM93]:

$$\textbf{Soil conc}_{i,\text{stand},k} = \textbf{Soil conc}_{i,k} \times \frac{a_k + b_k \times 25 + c_k \times 10}{a_k + b_k \times \%\textbf{lutum}_i + c_k \times \%\textbf{org.dust}_i}$$

where:

Soil $\text{conc}_{i,\text{stand},k}$	=	corrected concentration of heavy metal *k* in the standard soil (25% lutum and 10% organic dust) of grid cell *i* [mg.kg^{-1}]
$\%\text{lutum}_i$	=	% lutum in the soil of grid cell *i*
$\%\text{org.dust}_i$	=	% organic dust in the soil of grid cell *i*
a_k, b_k en c_k	=	constants which depend on metal *k*, see Table 3.4

Table 3.4 Constants for soil correction [DGM93]

Heavy metal	a_k	b_k	c_k
Cadmium	0.4	0.007	0.021
Mercury	0.2	0.0034	0.0017
Copper	15	0.6	0.6
Nickel	10	1	0
Lead	50	1	1
Zinc	50	3	1.5
Chromium	50	2	0
Arsenic	15	0.4	0.4

For each grid cell *i*, the indicator value of heavy metal *k* can be calculated by:

$$\mathrm{Ind}_{i,k} = \frac{\text{Soil conc}_{i,\text{stand},k}}{\text{Soil conc}_{\text{ref,stand},k}} \times \text{Area}_i$$

where:

$\mathrm{Ind}_{i,k}$ = value of the indicator for metal *k* with respect to grid cell *i* [km^2]

Soil $\text{conc}_{\text{ref,stand},k}$ = target concentration of metal *k* in the standard soil [$mg.kg^{-1}$], Table 3.3

Area_i = area of grid cell *i* [m^2]

The indicator value for heavy metal *k* is obtained by summing over the area *i*:

$$\mathrm{Ind}_{\text{tot},k} = \sum_i \frac{\text{Soil conc}_{i,\text{stand},k}}{\text{Soil conc}_{\text{ref,stand},k}} \times \text{Area}_i$$

where:

$\mathrm{Ind}_{\text{tot},k}$ = the value of the indicator for metal *k* summed over all grid cells [km^2]

After that, the indicator value for all heavy metals is obtained by summing over k:

$$\mathrm{Ind}_{\text{tot}} = \sum_k \mathrm{Ind}_{\text{tot},k}$$

where:

$\mathrm{Ind}_{\text{tot}}$ = the value of the indicator summed over all metals and areas

3.5.6 *Limitations*

Since emissions of heavy metals to the environment have not been reported yet and are likely to be low because of the use of electrostatic dust precipitators, concentrations in the environment have also not been calculated yet. Although the amount of heavy metals in hard coal gives an indication of the expected impacts on the environment, a reliable assessment of environmental impacts is only possible if it is based on environmental concentrations.

To what extent the calculated target concentrations actually protect 95% of the species has not been validated yet. Uncertainties may in particular arise from combination toxicity (synergistic effects), the lack of representative ecotoxicological data and the extrapolation of laboratory data to field conditions. The ecological consequences of 5% of the species not being entirely protected is not clear. Accumulation of toxic substances in the food chain is not taken into account, but may cause harmful effects to predators [VR91, Me90, St91].

3.5.7 *Conclusions and recommendations*

There is no uniform legislation in the European Community for soil and ground water protection. Legislation relevant to soil protection is primarily issued by directives. These directives aim to harmonize the national legislation, for which certain time limits are set. Each Member State can choose the most appropriate form and method of implementation for these directives [Vi93]. As a result, no generally accepted environmental standards, on which the calculation of the indicator can be based, have been established.

Emission rates of heavy metals have also not been reported yet. Without this information, only general statements about possible effects can be given; an accurate assessment of the impacts of heavy metals in which site-specific information is included, is not possible.

In this phase of the project, the assessments of environmental impacts of heavy metals emitted in the energy fuel cycles is restricted to impacts on soil. In the next phase of the project attention should be paid to the impacts on ground water, sediment and surface waters.

4 CONCLUSIONS AND RECOMMENDATIONS

The project 'Comparative Assessment and Management of Radiological and Non-Radiological Risks Associated with Energy Systems' in the research programme of the European Commission aims to develop a method for the comparison of impacts of electricity generating systems. As an example two energy systems, the coal fuel cycle in Germany and the nuclear fuel cycle in France, are being compared on the basis of a Multi Attribute Utility Analysis (MAUA). This report describes possible indicators for environmental impacts to be used in the MAUA.

In an early stage of the project it is decided to investigate indicators for environmental issues based on no-effect levels. The reasons for using only a limited number of (grouped) environmental effects and no-effect levels, are: (1) with a limited number of criteria the MAUA would remain transparent and (2) quantifying the real environmental impacts in time is extremely difficult and uncertain due to natural variability in ecosystems and uncertainty in dose–response relations.

Possible indicators for a MAUA presented in this report are worked out for a limited number of environmental concerns, i.e. climate change, depletion of natural resources, acidification and eutrophication, contamination with radionuclides and with heavy metals. These environmental issues are, on the one hand, the most important for the fuel cycles considered, and on the other, they give a clear view of the problems and possibilities in defining indicators.

Suggested indicators for the MAUA are pressure indicators (based on emissions only), and chemical-state indicators, based on a comparison of concentrations in the environment and no-effect levels. For some environmental problems the impacts do not depend on the location of the installation, like climate change and depletion of natural resources. For these indicators it is therefore proposed using pressure indicators: the emissions can be directly related to reduction scenarios and policy planning.

For environmental problems with impacts near the location of the installation it is proposed to use indicators based on no-effect levels. The values of the indicators can be shown to be easily calculated and intercompared. For example, the indicator value for the contamination with radionuclides is several orders of magnitude less than the indicator value for acidification.

Problems arise when different types of indicators have to be compared, for example, the indicators for climate change and for acidification. This comparison should be executed in the MAUA, where the different criteria are weighted. It is therefore recommended to apply the indicators in the MAUA, and to see what the advantages and disadvantages of the indicators are.

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APPENDIX A DATA USED FOR THE INDICATOR FOR ACIDIFICATION

Table A.1 Model parameters used in the calculation of transport and deposition of SO_x and NO_x using the OPS model [Ja91a].

Model parameter	**Parameter value**	**Remarks**
Substance	SO_x, NO_x	calculation of secondary compounds SO_4^{2-} and NO_3^- are included
Emission height	240 m, heat content 99 MW	stack height and internal energy of plume [CEC92]
Diameter of source	0 m	point source
Type of emission	continuous	emission is constant over the year
Location of the source	x = 155 km, y = 463 km (Amersfoort) x = 86 km, y = 440 km (near Rotterdam)	location in RDM coordinates (not of specific importance)
Grid resolution	5 km	grid over the Netherlands

APPENDIX B DATA USED FOR THE INDICATOR FOR RADIONUCLIDES

Table B.1 Model parameters used in the calculation of transport and deposition of ^{129}I (regular emissions) using the OPS model [Ja91a].

Model parameter	**Parameter value**	**Remarks**
Substance	^{129}I	gaseous, modelled as I_2
Emission height	100 m, heat content 0 MW	stack height and internal energy of plume [Dr93]
Diameter of source	0 m	point source
Type of emission	continuous	emission is constant throughout the year
Location of the source	x = 155 km, y = 463 km	location in RDM coordinates (not of specific importance)
Grid resolution	5 km	grid over the Netherlands
Dry deposition	$0.003867\ m.s^{-1}$	average dry deposition velocity [Bu90]
Wet deposition	scavenging: 16.88 % per hour	average scavenging coefficient during precipitation events (± 6% of the time) [La94]
Conversion	5×10^{-10} % per hour	radioactive decay in % per hour [ICRP83]

Table B.2 Dose conversion coefficients for soil organisms for several nuclides based on a soil concentration of 1 Bq.kg^{-1}, as calculated with the procedure outlined in Section 3.4.3.5.

Nuclide	[Gy.a^{-1}] Dose
^{129}I	1.2 x 10^{-14}
^{131}I	8.6 x 10^{-14}
^{133}I	1.5 x 10^{-13}
^{234}U	7.6 x 10^{-13}
^{234}U	7.3 x 10^{-13}
^{238}U	6.7 x 10^{-13}
^{238}Pu	8.8 x 10^{-13}
^{239}Pu	8.2 x 10^{-13}
^{228}Th	8.6 x 10^{-13}
^{230}Th	7.5 x 10^{-13}
^{232}Th	6.4 x 10^{-13}
^{234}Th	8.4 x 10^{-15}

Table B.3 Radionuclides and activity released for an accidental release, source term ST$_2$ [Dr93]

Radionuclide	Activity [Bq]	Radionuclide	Activity [Bq]
^{85m}Kr	6.8 x 10^{17}	^{131}I	3.4 x 10^{17}
^{87}Kr	5.7 x 10^{17}	^{132}I	4.9 x 10^{17}
^{88}Kr	1.4 x 10^{18}	^{133}I	6.4 x 10^{17}
^{88}Rb	1.6 x 10^{18}	^{134}I	1.6 x 10^{17}
^{89}Sr	3.4 x 10^{16}	^{135}I	5.2 x 10^{17}
^{90}Sr	1.7 x 10^{15}	^{133}Xe	6.8 x 10^{18}
^{91}Sr	3.8 x 10^{16}	^{135}Xe	1.5 x 10^{18}
^{91}Y	4.5 x 10^{15}	^{134}Cs	3.8 x 10^{16}
^{95}Zr	5.9 x 10^{15}	^{136}Cs	1.3 x 10^{16}
^{99}Mo	6.3 x 10^{16}	^{137}Cs	2.3 x 10^{16}
^{103}Ru	5.2 x 10^{16}	^{140}Ba	6.1 x 10^{16}
^{105}Ru	2.6 x 10^{16}	^{140}La	8.2 x 10^{15}
^{106}Ru	1.3 x 10^{16}	^{144}Ce	3.6 x 10^{15}
^{105}Rh	3.2 x 10^{16}	^{239}Np	7.1 x 10^{16}
^{127}Sb	2.9 x 10^{16}	^{238}Pu	3.2 x 10^{12}
^{129}Sb	7.2 x 10^{16}	^{239}Pu	1.1 x 10^{12}
^{127m}Te	4.4 x 10^{15}	^{241}Pu	3.2 x 10^{14}
^{127}Te	2.8 x 10^{16}	^{242}Cm	6.6 x 10^{13}
^{129m}Te	1.7 x 10^{16}	^{244}Cm	2.8 x 10^{12}
^{129}Te	7.6 x 10^{16}		
^{131m}Te	3.3 x 10^{16}		
^{132}Te	4.8 x 10^{17}		

Table B.4 Source parameters for the calculation of total deposition and air concentration calculated with PC-COSYMA [Dr93]

Parameter	Value
Released activity:	
^{88}Rb	1.6 x 10^{18} Bq
^{132}Te	4.8 x 10^{17} Bq
^{131}I	3.4 x 10^{17} Bq
^{132}I	4.9 x 10^{17} Bq
^{133}I	6.4 x 10^{17} Bq
^{134}I	1.6 x 10^{17} Bq
^{135}I	5.2 x 10^{17} Bq
Release height	10 m
Heat content	0 MW
Duration of release	1 h
Distance source - receptor	1.15 km
Wind speed	5 $m.s^{-1}$
Stability class	D
Precipitation	0 $mm.h^{-1}$
Height of the mixing layer	560 m
Deposition velocity	0.001 $m.s^{-1}$
Deposition velocity elemental iodine	0.01 $m.s^{-1}$
Deposition velocity organic iodine	5 x 10^{-4} $m.s^{-1}$

Table B.5 Air concentration and total deposition calculated at 1.15 km from the source for the source term in Table B.4

Radionuclide	Air concentration [$Bq.s.m^{-3}$]	Deposition [$Bq.m^{-2}$]
^{88}Rb	4.6 x 10^{11}	4.6 x 10^{8}
^{132}Te	1.7 x 10^{12}	1.7 x 10^{9}
^{131}I	1.2 x 10^{12}	4.4 x 10^{9}
^{132}I	1.7 x 10^{12}	6.6 x 10^{9}
^{133}I	2.2 x 10^{12}	8.1 x 10^{9}
^{134}I	2.5 x 10^{11}	9.2 x 10^{8}
^{135}I	1.6 x 10^{12}	6.1 x 10^{9}

APPENDIX C DATA USED FOR THE INDICATOR FOR HEAVY METALS

Table C.1 Trace elements in German hard coal [CEC92]

Beryllium	0.6 - 2 mg/kg
Vanadium	10 - 55 mg/kg
Cobalt	4 - 15 mg/kg
Nickel	10 - 40 mg/kg
Copper	10 - 30 mg/kg
Zinc	10 - 200 mg/kg
Arsenic	2 - 25 mg/kg
Strontium	20 - 75 mg/kg
Molybdenum	0.6 - 5 mg/kg
Cadmium	< 1 mg/kg
Barium	40 - 450 mg/kg
Mercury	< 0.4 mg/kg
Uranium	0.2 - 2 mg/kg
Lead	7 - 100 mg/kg