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Risk assessment to man and the environment in case of exposure to soil contamination. Integration of different aspects.

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A memorandum on the status of this report

In the period 1990-1994 the RIVM produced a series of reports on risks for humans, ecosystems and risks due to contaminant migration, under commission of the Dutch Ministry of Housing, Physical Planning and Environment (VROM). This report belongs to this series. The objective of this report is to show integration of the different aspects depicted in the reports released in the period up to 1991.

Since December 1993 several Working Groups for the Implementation of the Remediation Procedure, initiated by the Ministry, have been operational in revising the present Soil Clean-up Act. One ministerial circular was released in May 1994 ("Intervention values") and was followed by another in December 1994 ("Priority-setting of remediation"). Both ministerial circulars will, after a technical evaluation by the Technical Commission on Soil Protection and after testing the practical use by the environmental authorities and political discussion by the Dutch Parliament, be implemented in the future in the Dutch Soil Clean-up Act. Revisions to the circulars warrant the following remarks on the report of 1991.

The circulars include three relevant topics:
1. A two-step procedure for assessing and passing judgement on soil and groundwater quality and deciding on a the execution of a soil and/or groundwater remediation.
2. Soil and groundwater limit values for "clean soil" and "serious (soil and groundwater) contamination".
3. A procedure to determine remediation priority.

Ad 1. The procedure (Chapter 1.1) has undergone some major changes: First of all, the term C-standard value has been replaced by *intervention value*. When one of the contaminants exceeds the intervention value, the case is considered a "*serious contamination*", replacing the term "serious hazard to public health or the environment" (Chapters 1.1 and 3). In the case of a "serious contamination" the remediation *priority* has to be determined. As a consequence, exceeding the intervention value no longer automatically implies the necessity of remediation as stated on page 1 of the report!

Ad 2. The proposed procedure for derivation of the intervention values has been accepted. However, several minor and major changes were adopted in Tables 2, 11, 12, 13, and 14. The adopted values, as depicted in the enclosed Annex 2, were implemented in the ministerial circular of May 1994.

Ad 3. The procedure for priority-setting in cases with serious soil contamination (shortly mentioned in Chapters 2.4 and 5) based on actual (i.e. location-specific) risks for humans and environment and due to contaminant migration, has been released in the ministerial circular of 22 December 1994. A manual for calculating these actual risks has also been released.

Foreword

The Directorate-General for Environmental Protection commissioned the RIVM to provide the technical-scientific basis for certain elements of Part II (the technical part) of the Guideline Soil Protection.
The results will be used in the periodical revision of the Guideline. In this context, the criterion 'serious hazard to public health or the environment' takes a prominent place.
The RIVM brief comprises the following three aspects: hazards to the public health and the environment respectively as a result of exposure to soil contamination and views on potential exposure as a result of dispersion of contaminants via air and groundwater.
This report incorporates the contents of earlier reports published in the framework of this brief (report Nos. 725201001 up to and including 725201006 and report No. 725201008).
Thus, apart from the contribution by the authors, this report also includes contributions by the authors of the reports mentioned above. Moreover, this report also incorporates the results of discussions with individual experts, conducted as this report was written.

SUMMARY

This report concludes a series of reports which provide the technical-scientific base for revision of the guidelines for soil and groundwater in the Dutch Soil Protection act. The present study integrates the results of earlier studies in this context (Nos. 725201001 up to 725201006 and No. 725201008). Relevant information from the earlier reports concerning the derivation of intervention values is presented. This information takes into account the determination of human exposure and the risk of damage to (human) health or the environment. Furthermore, proposals are being made to clarify and where necessary adjust toxicologically based intervention values. Finally, attention has been given to the uncertainties in the derivation of the intervention values.

SAMENVATTING

In het kader van de herziening wet bodembescherming zijn voorstellen gedaan voor humane interventiewaarden voor bodem en grondwater, berekend met behulp van humaan toxicologische grenswaarden en humane blootstelling. Ten behoeve van dit doel wordt het humane blootstellingsmodel CSOIL gepresenteerd en bediscussieerd. Dit model is ontwikkeld met het doel om de humane blootstelling ten gevolge van bodemverontreiniging te kunnen kwantificeren. Bovendien worden de onzekerheden van de modelberekeningen besproken. Het CSOIL model wordt niet alleen gebruikt bij de afleiding van de interventiewaarden, maar wordt tevens toegepast, in combinatie met metingen in contact media, voor de berekening van het actuele humane risico, ten behoeve van de bepaling van de saneringsurgentie.

1 INTRODUCTION

1.1 Research framework

The Directorate General for Environmental Management (VROM, DGM) has commissioned RIVM to provide the technical-scientific basis of parts of Part II (the technical part) of the Guideline Soil Protection (VROM, 1990a). The results of this study are used in the periodical revision of the Guideline. The criterion "serious hazard to public health or the environment" is crucial in this context. In the incorporaton of the Provisional Soil Clean-up Act into the Soil Protection Act, proposals were put forward to change the test criterion into "serious danger of a deterioration of the functional properties of the soil in relation to human, animal or plant life". As this is an editorial adjustment according to the explanatory memorandum, this report also applies to the section on clean-up in the Soil Protection Act (Legislative Proposal 21 556; VROM, 1990c).

The concept "serious hazard to public health or the environment" is defined by two aspects: the nature and the concentration of the contaminants, which gives an indication of the extent of the contamination (of environmental compartments) as well as of the potential effects of the contamination (on man or the environment), and the local circumstances concerning the contamination, which determine the possibility and the extent of dispersion or contact. The test criterion indicates whether or not an intolerable risk to man (public health) or the environment might occur if all potential exposure routes were to apply.

If such a "serious hazard" occurs, it means that the site will be examined and cleaned up (clean-up necessity), based on the existing legal regulations. The present soil-use is a determining factor for the extent to which "actual risks to man or the environment as a result of exposure to soil contamination" occurs. By means of a so-called actual exposure risk analysis, based on the local soil condition, hydrogeology and soil-use, it must be determined whether there is a high priority for clean-up (research) of that particular site (clean-up urgency).

The C-standard value is a general soil-quality criterion for the assessment of "serious hazard to public health or the environment" as a result of soil contamination.
In the framework of the instruction to RIVM explained above, the test criterion "serious hazard" has been further clarified. The existing figures for the C-standard values from the test table are evaluated in the present Guideline. Moreover, the two aspects of the criterion for the assessment of "serious hazard" are integrated.
The C-standard value does not provide information on the actual risks referred to above. It is based on the assumption that all exposure routes operate and does not take account of specific local circumstances to which this assumption need not apply.

The instructions to RIVM concerned the risks to public health or the environment in cases of soil contamination and a further examination of (potential) exposure as a result of dispersion of contaminants via air and groundwater.

So far, six reports have been published within the context of these instructions.
The concept "serious hazard to the environment" has been elaborated by Denneman and Van Gestel (report No. 725201001, 1990) and translated into proposals for ecotoxicologically-based C-standard values. Following an external consultation round, Denneman and Van Gestel compared these values, derived mainly from soil ecotoxicological data, to aquatic toxicity data (report no. 725201008, 1991).
Linders (report no. 725201003, 1990) and Vermeire et al. (report no. 725201005, 1991) contributed towards the derivation of human-toxicologically-based C-standard values. Linders developed a model procedure for the assessment of the risks to man in cases of exposure to high concentrations of substances

in the environment. Van den Berg integrated this procedure into the CSOIL exposure model (report no. 725201006, 1991). This calculation model was also based on extensive and detailed studies of other models and background literature. Report no. 725201006 gives a justification of the choice of parameters, routes, etc. This model is used in order to assess the potential and actual exposure and the related risk of damage to human health and the environment.

Both in cases of potential and actual exposure, the extent of the exposure must be assessed in the light of the question whether the exposure is tolerable or not. Report no. 725201005 (Vermeire et al., 1991) defines the toxicological assessment framework, the risk levels related to the concept "serious hazard to public health", the specification of this concept in terms of carcinogens and non-carcinogens and the human-toxicological recommended values for different substances. Van den Berg (1991) presented proposals for human-toxicologically-based C-standard values, based on these data and estimations.

The other two reports concern the dispersion of the contaminiation. Lagas et al. (report no. 725201002, 1990) were mainly concerned with the dispersion of substances via groundwater. The mobility of contaminants in the soil and the estimation of the direction of the soil-water flux were important elements in this study. Kliest (report no. 725201004, 1990) provides information on the dispersion via the gaseous soil phase and the effects on indoor and outdoor air in relation with the inhalatory exposure route.

The reports referred to above provide basic knowledge. In addition, the protocols referred to below are presently available.
Commissioned by the Directorate General for Environmental Protection, the Netherlands Organization for Applied Scientific Research (TNO) developed protocols for the determination of the nature, concentration and extent of soil contamination (Lamé and Bosman, 1991a and b). Kliest (1990) includes a protocol for the implementation of (indoor) air research. Van den Berg (1991) provides a protocol for the estimation of the exposure of man to soil contamination. Other protocols will be developed in due time.

1.2 Objective

The objective of the present report (no. 725201007) is divided into three parts:
- providing a structure for the design of the Further Research, which now includes the (determination of) potential and actual exposure;
- presenting proposals for the clarification and where necessary adjustment of the C-standard values. To this purpose, the proposals for ecotoxicological and human-toxicologically-based C-standard values referred to above are integrated;
- providing instruments with which the actual exposure and the related risk of damage to human health can be assessed and based on which the urgency for Clean-up (research) may be determined.

1.3 Structure of the report

The present report is structured as follows.
Chapter 2 deals with the design of the Further Research. Chapter 3 contains relevant ecotoxicological and human-toxicological background information. Chapter 4 describes the integration of the proposed ecotoxicological and human-toxicological standard values. In addition, chapter 4 presents an adjusted test framework (the test table) for contaminations reaching the level where Clean-up (research) becomes imperative.
Chapter 5 discusses the actual exposure. Chapter 6 describes the methods and procedures used, discusses the standard values as formulated and presents recommendations for additions to and improvement of methods.

2 DESIGN OF THE FURTHER RESEARCH

2.1 Introduction

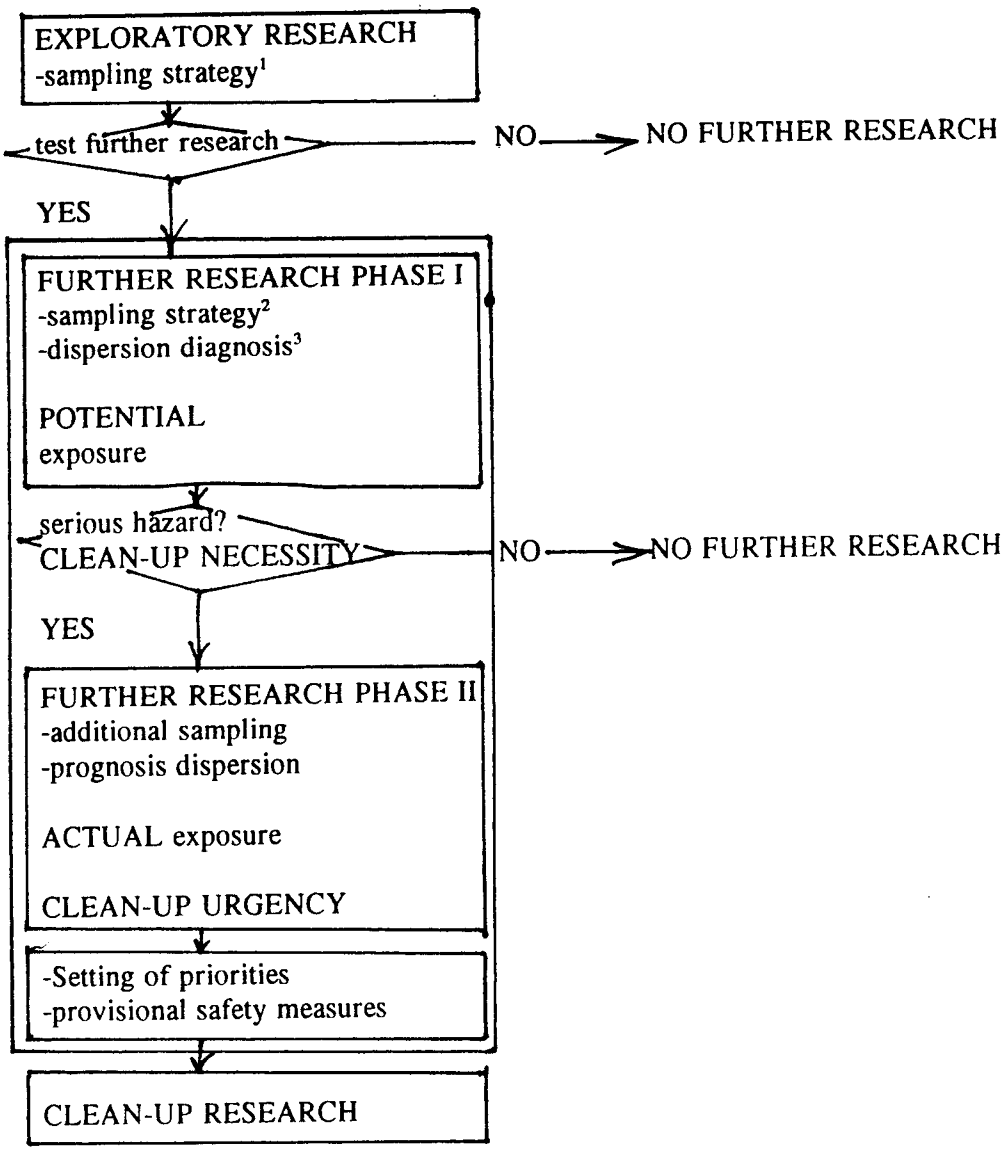
The design of the Further Research and the presentation of the results must provide insight into the soil contamination (both the terrestrial soil and the aqueous soil), so that it may be determined whether or not the concept "serious hazard to public health or the environment" applies and so that an analysis can be made of the present (and future) extent of the exposure and the related risks if this is indeed the case.

In the implementation of the Further Research, two phases are distinguished (see Figure 1). In the first phase, the nature, concentration and extent of the contamination are diagnosed, based on the appropriate sampling strategy, and the dispersion of contaminants is analysed. The results are tested by comparison with the criterion "serious hazard to public health or the environment". If this is the case, the second phase of the Further Research focuses on the actual exposure and the risk of damage to human health or the environment due to the exposure, taking into account the possibility of further dispersion of the contamination (prognosis on the dispersion of the contamination).

2.2 Phase 1: diagnosis of the contamination and test by comparison with the criterion "serious hazard to public health or the environment" (Analysis of "potential exposure")

Based on the results from the Exploratory Research (Lamé and Bosman, 1991a), a sampling strategy is developed, taking into account the spatial distribution of the contamination (homogeneously or heterogeneously distributed contamination). Lamé and Bosman (1991b) described the procedure to be followed in great detail. The boring-, sampling- and analysis programme may be optimized on the basis of estimations of the mobility of the contaminants and the flow velocity and direction of the groundwater. Lagas et al. (1990) discuss substance mobility in greater detail and provide simple instruments for the description of groundwater flow. In cases where the estimated dispersion strongly deviates from the measurements, more sophisticated calculation models are to be recommended. In this way, the necessary refinement of the schematic representation of the subsoil, like soil layer composition, the presence of ditches, drainage systems and groundwater drawoff can be incorporated in the calculations.
The SAMWAT (Samenwerking onderzoek waterbeheer: Cooperation Water Management research) bureau, affiliated with the TNO Commission for Hydrologic Research, operates a database of models available in the Netherlands.
Most models may be used for the schematic representation of the subsoil and the calculation method. For reasons of uniformity and in order to be able to compare data, this report recommends that the models are reduced to a limited number of generally available and applicable models.

Figure 1 Design of the Further Research and the positioning in the overall research chain in the context of soil clean-up.



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[1] See Lamé and Bosman (1991a). Exploratory Research into the nature, concentration and extent of the soil contamination.

[2] See Lamé and Bosman (1991b). Further Research into the nature, concentration and the extent of the soil contamination

[3] Lagas et al. (1990). Dispersion of substances in cases of soil contamination.

Note that the quality of the calculations is determined not so much by the calculation method, as by the quality of the schematic representation of the subsoil and the choice of model parameters (see also the observations on a phased groundwater research design in the TCB advisory report with reference to additional groundwater protection against diffuse sources of soil contamination; TCB, 1991a).
The understanding of the contamination circumstances and the local soil constitution and hydrogeological circumstances is increasingly refined by means of a cyclic sampling strategy and a systematic comparison of measurement results and estimations based on hypotheses and calculation models. On concluding the first phase of the Further Research, the nature and extent of the contamination must be known. The location of the main sources of the contamination must have been identified. Furthermore, the sites at which the concentrations of contaminants in the soil and the (uppermost and when necessary the deeper) groundwater exceed the (local) standard value, or the C-standard value respectively, must be known.

Important factors in the assessment of whether or not there is a "serious hazard to public health or the environment" are the extent of the contamination (of environmental compartments) and the potential effects this may have for man and the environment, and the possibility of dispersion or contact. Chapter 3 outlines the starting-points and backgrounds of this assessment. Chapter 4 presents the C-standard values based on which the assessment is performed.
In addition to the above, it is proposed to speak of "serious hazard to public health or the environment" when it becomes clear from the results of the first phase of the Further Research that the average concentration for individual substances in the soil or the groundwater respectively (over a depth of at least 0.5 meter or 1 meter respectively) exceeds the C-standard values, measured on a sampling-scale with a distance between measurement points of 5 to 10 meter respectively (see also 4.3 for simultaneous exposure to different contaminants). Instituting a follow-up research and implementing clean-up measures in the framework of the Provisional Law on Soil Clean-up (which will become the section on clean-up of the Soil Protection Act in due time) will then be justified. These follow-up activities are provided with an implementation schedule in the provincial soil clean-up programmes on a case-by-case basis. Naturally, the law continues to govern cases of impending soil contamination as well (see Guideline Soil Clean-up, part I, pp.1-2).

If the level or volume of the contamination in the soil does not exceed the criterion defined above, this does not mean that there is no contamination or that the situation as such is acceptable. Depending on soil-use, measures outside the scope of the Provisional Law on soil Clean-up may be called for, for instance prohibiting the growing of sensitive crops or taking agricultural measures like liming of plots).

If the contamination level or volume in a terrestrial or aqueous soil does exceed the criterion described above, the second phase of the Further Research should be started as soon as possible. This is based not only on the mapped extent of the contamination, but also on the knowledge of the nature of the contamination, the ecochemical behaviour of the contaminants (the distribution of the substances over the solid, liquid and gaseous soil-phases and the physicochemical characteristics, such as soil-water and air-water distribution coefficients, water-solubility, vapour pressure, the permeation coefficient and the local soil and hydrogeological conditions (for instance quantification of lutum- and organic substance rate and the acidity of soil (layers) and hydrogeological parameters considered relevant to the site). Depending on the results of this second phase, the timing of the Clean-up (Research) may be determined.

2.3 Phase 2: Estimation of the extent of the exposure and related risks (analysis of "actual exposure")

Based on the nature and extent of the contamination and the local contamination circumstances, the exposure is assessed. Current soil-use is a determining factor for the extent of exposure and the related risk of damage to human health or other organisms.
On the basis of these calculations, the urgency of a Clean-up (research) or taking temporary safety

measures directed to preventing dispersion of the contamination and the harmful effects related to this, may be motivated and justified.

In the implementation of the second phase of the Further Research, a number of steps should be distinguished:

a. determination of the routes via which man may be exposed to the contamination of the terrestrial or the aqueous soil;

b. determination of the contribution via these separate routes to the total human exposure and determination of the relevant exposure routes;

c. supplementary measurements of concentrations of contaminants in indoor and creeping-space air of houses, drinking water, food crops (and other relevant food products);

d. calculation of the present human exposure to soil contamination and where necessary, an estimation of the future human exposure, based on a prognosis of the dispersion of the contamination. The CSOIL model (Van den Berg, 1991) is used for steps a to d; Linders (1990) may be used for calculating the exposure due to consumption of food products other than crops contaminated by soil pollution and for calculating the direct exposure via polluted surface water;

e. determination of the human exposure risk by means of an evaluation of the total human exposure based on toxicological recommended values (included in Vermeire et al., 1991);

f. estimation of the effects of soil contamination on soil organisms (for some contaminations or test organisms this must be done by means of separate tests; for others on the basis of the available literature);

g. estimation of the future dispersion of the contamination (contamination dispersion prognosis) based on model calculations; calculation of the potential soil- and groundwater contamination volume (see Lagas et al., 1990);

h. interpretation of the present exposure of man and the environment to soil contamination taking into account as far as possible the background exposure and combination toxicity.

The second phase of the Further Research must consider each of the 8 steps, which must be reflected in the reporting stage, particularly in the case of an integrated implementation of the steps.

Chapter 5 discusses the individual steps in greater detail.

2.4 Setting of priorities

The Provisional Law on Soil Clean-up (and the section on clean-up in the forthcoming Soil Protection Act) is based on a programmed and phased approach of cases of soil contamination. Consequentially, there is a strong emphasis on instigating research and decontaminating the soil according to an urgency priority sequence.

The urgency priority is primarily determined by the criterion "serious hazard to public health or the environment". If this is the case, soil-use in relation to the nature and extent of the contamination constitutes an important factor in the decision at which moment Clean-up (research) becomes imperative.

In determining the urgency for Clean-up (research) based on the results of the calculations described in section 2.3, attention should be given both to the public health and environment aspect and to the possibility of dispersion of the contamination. None of these three aspects in themselves can be considered to be of overriding importance. The presence of residential, work areas or other living areas, water-catchment areas and nature reserves has been taken into account in the three aspects and so constitute an integral part of the primary urgency assessment. In contrast to the relatively high level of abstraction with reference to these three areas, the 8 steps mentioned above allow for a more concrete assessment of the priority sequence. In principle, the determination of the urgency should not be influenced by aspects other than those mentioned above.

Temporary measures, which serve to postpone the urgency of definitive provisions and to control the situation, in practice will have to be limited to two of these three aspects: public health and the dispersion of the contamination.

It should be noted, however, that the safety measures must have a temporary character and should not be allowed to obstruct definitive clean-up.

The Guideline Soil Clean-up (pp. 11-19) also specifies other measures which contribute to preventing the consequences of the contamination, but that are not considered temporary safety measures in the sense of the Provisional Law on Soil Clean-up. The latter measures are likewise primarily directed to public health (for instance discontinuing the provision of drinking water or replacing plastic water pipes by metal pipes).

3 SERIOUS HAZARD TO PUBLIC HEALTH OF THE ENVIRONMENT

3.1 Introduction

This chapter discusses the definition and quantitative elaboration of the criterion "serious hazard to public health or the environment". Furthermore, the methods and procedures generating the toxicological data are described and the derivation of the proposed C-standard values from these data is explained.

In clarifying the "serious hazard" test criterion, the Hazard Leaflet to the National Environmental Policy Plan, *Dealing with Hazards* (VROM, 1988c), has been followed. Referrences are moreover made to the *1991-1994 Environmental Programme, Part III, Conceptual Framework of Environmental Policy* (VROM, 1990b). In this context, the C-standard value is introduced as an intervention value indicating an environmental quality level at which the concentrations derived from the "maximum tolerable risk" have been exceeded to such an extent that intervention is necessary.

3.2 Serious hazard to the environment

3.2.1 Introduction

The ecological function of the soil constitutes the starting-point for the definition of the criterion "serious hazard to the environment".

The ecological function of the soil is based on a complex interaction of biological and physico-chemical processes. The essential consideration in this context is the capacity of the soil to provide for the primary production and to serve as "substrate" for decomposition and mineralization processes. Moreover, the soil plays an important part in the geochemical cycles and the hydrological cycle.

A soil-ecosystem is characterized by structural (individuals, populations, communities) and by functional characteristics of the ecosystem, which are embodied by groups of organisms - "functional entities"). In order to be able to extrapolate from the effects on the structure of ecosystems to the functioning of ecosystems, knowledge of the relation between these two is required. Generally speaking there is no simple and straightforward relation between "functional entities" in ecosystems and a particular species composition. Different species may perform the same function. The same species may perform different functions. Effects on the population of a certain species from a "functional entity" need not automatically entail consequences for this function, as another species from this entity may take over the role. The capacity to compensate future disturbances has however been impaired. Some species also have a specific function. Effects on these species may have direct consequences for the functioning of the ecosystem. In short, a general extrapolation from effects on populations or communities to effects on the functioning of ecosystems cannot be made directly (TCB, 1990a). However, it is assumed by the *Gezondheidsraad* (Public Health Council), among others, that protecting the structure provides guarantees for the functioning of the ecosystem. It is furthermore assumed that the structure is endangered when the composition of the species is affected (VROM, 1989a).

3.2.2 The criterion "serious hazard"

The criterion "serous hazard" has been defined as the occurrence or immanent occurrence of irreversible and irrepairable damage to the composition of species. This definition has been based on the finding by Moriarty (1983) and the *Voorlopige Technische Commissie Bodemsanering (VTCB*, 1986) (Provisional Technical Committee Soil Protection) that toxic substances may be dangerous as a result of their effects on communities, but that they also act on individual organisms, so that effects become visible here first. Therefore, the question to be answered is 'when should we speak of serious impairment of the composition

of species in an ecosystem and when is recovery, or re-colonization of an area by soil organisms no longer reasonably to be expected as a result of soil contamination'. The answer to this question must be based on the subsistence of species and communities. Apart from mortality, reproduction and growth are mentioned as the main population-dynamic parameters (Ma, 1983; Joosse and Verhoef, 1983; Van Capelleveen, 1987). This does not mean that other sublethal parameters do not affect the subsistence of organisms. Negative effects on other important parameters, however, will generally lead to reduced reproduction (Kooijman and Metz, 1984).
Based on the essential parameters mentioned above, Denneman and Van Gestel (1990) observed that there is a case of **"serious hazard" when 50% of the species in a system are adversely affected by the presence of one or more contaminants in concentrations exceeding the NOEC (No Observed Adverse Effect Concentration) for these species**. The dimensions of the adverse effects given such a concentration are different for each species and vary from almost negligible to highly substantial.
This means that sensitive species are not explicitly protected. Should protection of these species be considered relevant, the actual risk analysis must pay attention to this (see Chapter 5). If required, measures may be taken to protect sensitive species. The *Interimwet bodemsanering (IBS, 1982)* (Provisional Law on Soil Clean-up) is not the appropriate framework to this purpose.
It must be noted in this context that the qualification "no serious hazard to the ecological function of the soil" has a completely different meaning not entailing a full-proof and certain protection of the ecological function as aimed for by means of target values or limit values. Concentrations below the "serious hazard" level, but exceeding the target value level, may impair the ecological function. This damage, however, is not such as to justify the qualification "serious danger".

A literature study has been performed in order to provide the numerical values for the test criterion. This literature study comprised the available soil-ecotoxicological data with reference to plants, soil animal life and micro-organisms for all substances presently included in the test table. It was decided to gather NOEC-, LC50- and EC50-values for preferably non-lethal effects (particularly reproduction and growth) for (individual) species in case of chronic exposure. In order to derive an ecotoxicologically-based C-standard value (Denneman and Van Gestel, 1990) from these toxicity data, the RAB-method (*Risico-Analyse-Bodemverontreiniging*; Van Straalen and Denneman, 1989: Risk Analysis Soil Contamination) has been used. This is in accordance with the 1988 *Gezondheidsraad* (Health Council) advice and the current derivation practice of maximum tolerable risk levels (Van de Meent et al., 1990). Application of the RAB method to the selected criterion, full protection of 50% of the species in an ecosystem, results in reducing the method to the determination of the median of the frequency distribution of the sensitivity of species, estimated on the basis of the geometric average of the effect data.
In general, it became clear that there were insufficient data for application of the RAB method, so that alternative procedures had to be used. By comparing these methods in a number of cases to the RAB method, it was found that a similar result may be achieved in this way.

In deriving the proposals for ecotoxicological C-standard values, the influence of (the variation of) soil properties on the availability of substances, and so on the risk for soil organisms) has been taken into account. For this reason, C-standard values have been proposed that depend on the organic substance- and lutum rate (Denneman and Van Gestel, 1990). The method used for this purpose has been based on the procedure described for the soil quality reference values (VROM, 1987).

Particularly the risks for those organisms which are in direct contact with the soil have been considered. The risks of continued poisoning have not been explicitly considered. It is assumed that these organisms on the whole do not run greater risks than organisms directly related to the soil. There are two reasons for this assumption. In the first place, the exposure of such organisms is limited to food, whereas the organisms in the soil are also exposed via direct dermal contact. Secondly, the foraging area of such organisms is generally much larger than the site where the C-standard values have been exceeded. This means that they get their food only partly from a contaminated area. For some substances which are known to accumulate strongly in food chains, this assumption may be too unspecified, however, and intolerable risks will occur particularly for higher organisms. This will have to be worked out further in due time.

The results of the research, the C-standard values, also apply to the aqueous soil. This is in accordance with the multifunctionality principle, which states that each soil-type must potentially be capable of continuing to perform all functions, taking into consideration its natural properties. In this way, terrestrial soils may become aqueous soils and vice versa. In addition, it is also in accordance with a report by the Technical Commission for Soil Protection: "Standardization of Aqueous Soils" (TCB, 1989b). Although this report states that the structure of aquatic systems differs from that of terrestrial systems and that effects in aqueous soils may operate at different concentration levels than in terrestrial soils (these differences become manifest in the assessment of actual exposure risks), it may not be automatically concluded from this that separate standardization is called for. In view of the importance of guaranteeing the exchangeability of soils, the report argues against such a separate standardization as being ineffective and suggests that joint reference points must be aimed for. This strategy, which is tailored to establishing target values, is also considered to be valid for intervention values in this report.

For aqueous soil organisms, even more so than for terrestrial soil organisms, it is true that there are very few data on which a risk assessment can be based. The procedure and data used do not imply that aqueous soil organisms are less protected than terrestrial soil organisms.
TCB (1989b) conducted a risk assessment for some substances, based on effects related to aqueous soil organisms. This showed that, given the same concentrations, aqueous soil organisms are definitely not more sensitive than terrestrial soil organisms.
Consequentially it may be assumed that the C-standard values proposed by Denneman and Van Gestel (1990), which are primarily based on effects on terrestrial soil organisms, represent a comparable risk level for aqueous soil organisms.

3.2.3 Additions and adjustments

Following an external round of consultations (including reactions by representatives from the Association of Provincial Authorities (IPO), the Ministry of Agriculture, Nature Management and Fisheries (LNV), the Ministry of Transport, Public Works and Water Management (V&W) and a number of individual experts), Denneman and Van Gestel wrote an addendum to report no. 725201001. Report no. 725201008 (Denneman and Van Gestel, 1991) compares the results from report no. 725201001 to aquatic toxicity data and data

Table 1. Adjusted proposals for ecotoxicologically-based C-standard values [mg.kg^{-1} ds], based on data from report no. 725201008 (Denneman and Van Gestel, 1991). The values refer to a standard soil type.

substance name	ecotoxicogically-based C-standard value standard soil [mg.kg^{-1} ds]
DDT/DDD/DDE	4
phtalates	60
PAH	40
chlorinated aliphatic hydrocarbons	60
chlorophenols,	10
excl. pentacholorophenol	5
cholorobenzenes	30
atrazine	6

derived by means of quantitative structure-activity relations (QSARs). The use of aquatic toxicity data is relevant, as the sensitivity of water- and aqueous foil organisms is considered comparable by experts. These data have been used on the one hand to improve the basis of the previously proposed values. On the other hand, this also allows for effects to aqueous soil organisms and water organisms as a result of exposure to a contaminated aqueous soil or to contaminated surface water (due to transport of substances from the aqueous soil). This has led to a modification of the originally proposed C-standard values for some substances.

Table 1 shows the adjustments to the original report no. 725201001.

Table 2 gives the proposed ecotoxicologically-based C-standard value for a selected group of substances presently included in the Guideline.
Denneman and Van Gestel (1990,1991) furthermore collected data for a large number of other substances, mainly pesticides, and determined ecotoxicologically-based C-values for these substances. When other contaminants than those mentioned in the test table are found to operate in cases of soil contamination, report nos. 725201001 and 725201008 may be consulted.

3.2.4 Reliability score

Table 3 represents the reliability score of the derived ecotoxicologically-based C-standard value for a number of selected substances. Reliability depends on the derivation procedure for the standard values (Denneman and Van Gestel, 1990). Table 4 presents a detailed elaboration of the final score system.

3.3 Serious hazard to public health

3.3.1 Introduction

For the purpose of deriving human toxicologically-based C-standard values, a generally applicable procedure has been described in order to assess the risks for man in the event of exposure to heightened concentrations in the environment (Linders, 1990). This procedure may be used for two purposes:

1. calculating the potential exposure and the toxicological evaluation of the exposure;
2. calculating the actual exposure and the toxicological evaluation of this exposure in the context of the Further Research (following determination of the necessity of soil clean-up).

Table 2. Proposals for human toxicologically-based (Van den Berg, 1991) and ecotoxicologically-based (Denneman and Van Gestel, 1990 and 1991) C-standard values in mg.kg^{-1}, standardized for the standard soil-type (10% organic substance and 25% lutum).
The present C-standard value (VROM, 1990a) has been given as reference value.

A. Metals, arsenic and inorganic compounds.

Substance name	C-soil (human-tox.) [mg.kg^{-1}]	C-soil (ecotox.) [mg.kg^{-1}]	present C-soil [mg.kg^{-1}]
I Metals			
arsenic	300	40	50
barium	698	650	2000
cadmium	1,8	12	20
chromium (III)	2140	230	800
chromium (VI)	0,1	--	--
Cobalt	400	120	300
copper	5020	190	500
mercury	87	10	10
lead	162	290	600
molybdeen	37	<480	200
nickel	179	210	500
tin	571000	--	300
zinc	1840	720	3000
II Inorganic compounds			
cyanides free	18,5	--	100
cyanides complex	7,2	--	500
thiocyanates	2,0	--	--
sulfides	0,18	--	200

(Table 2 ctnd. Proposals for human toxicologically-based (Van den Berg, 1991) and ecotoxicologically-based (Denneman and Van Gestel, 1990 and 1991) C-standard values in mg.kg^{-1}, standardized for the standard soil-type (10% organic substance and 25% lutum).
The present C-standard value (VROM, 1990a) has been given as reference value.

B. Organic substances.

Substance name	C-soil (human-tox.) [mg.kg^{-1}]	C-soil (ecotox.) [mg.kg^{-1}]	present C-soil [mg.kg^{-1}]
III Aromatic compounds			
benzene	48	25	5
ethylbenzene	149	--	50
phenol	41	40	5
cresol	4,6	50	5
toluene	368	150	30
xylene	73	--	50
catechol	32	--	--
resorcinol	14	--	--
hydroquinol	12	--	--
IV Polycyclic Aromatic Hydrocarbons			
anthracene**	86900	40	100
benzo(a)anthracene**	10100	40	50
benzo(k)fluoranthene**	10900	40	50
benzo(a)pyrene**	996	40	10
chrysene	88	40	50
phenanthrene	319	40	100
fluoranthene	496	40	100
indene(1,2,3cd)pyrene**	11400	40	50
benzo(ghi)perylene**	11800	40	100
napthalene	977	40	50
V chlorinated hydrocarbons			
aliphatic chlorohydrocarbons			
1,2-dichloroethane	3,5	60	50
dichloromethane	39	60	50
tetrachloromethane	2,2	60	50
tetrachloroethene	14	60	50
trichloromethane	27	60	50
trichloroethene	666	60	50
vinyl chloride	0,09	--	--

clorobenzenes

monochlorobenzene	716	30	10
dichlorobenzene	1754	30	10
trichlorobenzene	10	30	10
tetrachlorobenzene	17	30	10
pentachlorobenzene	23	30	10
hexachlorobenzene	25	30	10

cholorophenols

monochlorophenol	16	10	5
dichlorophenol	34	10	5
trichlorophenol	57	10	5
tetrachlorophenol	25	10	5
pentachlorophenol	76	5	5

** exceeding water-solubility at given C-soil value

(Table 2 ctnd. Proposals for human toxicologically-based (Van den Berg, 1991) and ecotoxicologically-based (Denneman and Van Gestel, 1990 and 1991) C-standard values in mg.kg^{-1}, standardized for the standard soil-type (10% organic substance and 25% lutum). The present C-standard value (VROM, 1990a) has been given as reference value.

B. Organic substances.

Substance name	C-soil (human-tox.) [mg.kg^{-1}]	C-soil (ecotox.) [mg.kg^{-1}]	present C-soil [mg.kg^{-1}]
<u>V chlorinated hydrocarbons ctnd.</u>			
chloropolycyclic aromatic hydrocarbons			
chloronaphthalene	4.8	-	5
polychlorobiphenyles			
trichlorobiphenyl	7.6[1]	70[2]	5
hexachlorobiphenyl	12[1]	70[2]	5
<u>VI Pesticides</u>			
organo-chlorine pesticides			
*DDT***	10400	4	5
*DDE***	3450	4	5
Aldrin	12	0.35	5
dieldrin	5.0	4	5
endrin	4.1	0.06	5
α-HCH	21	2	5
β-HCH	0.63	2	5
γ-HCH	18	2	5
δ-HCH	24	2	5
other pesticides			
carbaryl	473	5	10
carbofuran	409	1.5	10
propoxur	650	0.75	10
maneb**	58800	30	10
atrazine	21	6	10

VII Other contaminants

heptane	10	-	-
octane	7.2	-	-
cyclohexanone	770	-	60
butylbenzylphtalic acid	754	60	250
di(2-ethylhexyl)phtalic acid**	8890	60	250
pyridine	3.5	150	20
styrene	102	-	50
tetrahydrofuran	1.2	-	40
tetrahydrothiophene	5.4	-	50

** exceeding water-solubility at given C-soil value

1 Aroclor 1248, 48% chlorine - mean 4 Cl-substituents

2 Aroclor 1254, 54% chlorine - mean 6 Cl-substituents

Table 3. Overview of the reliability score accompanying the proposed human toxicologically-based and ecotoxicologically-based C-standard values.

Substance name	reliability score human tox recom.value	exposure	total score human-tox.	applied ecotox. procedure	reliability score ecotox.
I Metals					
arsenic	3	1	3	2a(2)	6
barium	3	1	3	3a	2
cadmium	3	1	3	1a	10
chromium (III)	2	1	2	2a(2)/1b	7
chromium (VI)	1	1	1	-	-
cobalt	3	1	3	3a	2
copper	3	1	3	1a	10
mercury	3	1	3	2a(2)/2a(3)	6
lead	3	1	3	1a	10
molybdenum	3	1	3	3d	2
nickel	3	1	3	3a/1a	6
tin	3	1	3	-	-
zinc	3	1	3	2a(2)/2b(4)	6
II Inorganic compounds					
cyanides free	3	1	3	-	-
cyanides complex	2	1	2	-	-
thiocyanates	3	1	3	-	-
sulfides	1	1	1	-	-
III Aromatic compounds					
benzene	3	2	6	4	2
ethylbenzene	2	2	4	-	-
phenol	3	2	6	3c	2
cresol	2	2	4	(3a)	2
toluene	3	2	6	2a(2)	6
xylene	2	2	4	-	-
catechol	2	2	4	-	-
resorcinol	2	2	4	-	-
hydroquinol	2	2	4	-	-
IV Polycyclic Aromatic Hydrocarbons					
anthracene	1	3	3	(4)/add.	5
benzo(a)anthracene	2	3	6	(4)/add.	5
benzo(k)fluoranthene	2	3	6	(4)/add.	5
benzo(a)pyrene**	2	3	6	(4)/add.	5
chrysene	2	2	4	(4)/add.	5
phenanthrene	2	2	4	4/add.	5
fluoranthene	2	2	4	(4)/add.	5
indene(1,2,3cd)pyrene**	2	3	6	(4)/add.	5
benzo(ghi)perylene**	2	3	6	(4)/add.	5
napthalene	2	2	4	4/add.	5

V chlorinated hydrocarbons

aliphatic chlorohydrocarbons

1,2-dichloroethane	3	2	6	(3a)/add.	5
dichloromethane	3	2	6	(3a)/add.	5
tetrachloromethane	2	2	4	(3a)/add.	5
tetrachloroethene	2	2	4	3a/add.	5
trichloromethane	2	2	4	(3a)/add.	5
trichloroethene	3	2	6	(3a)/add.	5
vinyl chloride	3	2	6	(3a)/add.	5

chlorobenzenes

monochlorobenzene	3	2	6	(3a)/add.	5
dichlorobenzenes (ind.)	3	2	6	3a/add.	5
trichlorobenzenes (ind.)	2	2	4	2a(2)/add.	5
tetrachlorobenzenes (ind)	2	2	4	(3a)/add.	5
pentachlorobenzene	2	2	4	3a/add.	5
hexachlorobenzene	3	2	6	3a/add.	5

chlorophenols

monochlorophenols (ind.)	2	2	4	3b/add.	5
dichlorophenols (ind.)	3	2	6	3b/add.	5
trichlorophenols (ind.)	2	2	4	3a/add.	5
tetrachlorophenols (ind.)	2	2	4	3c/add.	5
pentachlorophenol	3	2	6	2a(3)/add.	5

chloropolycyclic aromatic hydrocarbons

chloronaphthalene	1	2	2	-	-

polychlorobiphenyles

trichlorobiphenyl	2	2	4	(4)	2
hexachlorobiphenyl	2	2	4	(4)	2

VI Pesticides

organo-chlorine pesticides

DDT**	3	3	9	3c/add.	5
DDE**	3	2	6	(3c)/add.	5
aldrin	2	2	4	2a(2)	6
dieldrin	2	2	4	3a/2a(4)	5
endrin	2	2	4	3a	2
α-, γ- and δ-HCH	3	2	6	2a(2)	6
β-HCH	2	2	4	-	-

other pesticides

carbaryl	3	2	6	2a(3)	7
carbofuran	3	2	6	1a	10
propoxur	3	2	6	3c	2
maneb**	2	3	6	3c/2a(3)	4
atrazine	3	2	6	2a(3)/1a/add.	7

VII Other contaminants

heptane	1	2	2	-	-
octane	1	2	2	-	-
cyclohexanone	2[1]	2	6	-	-
butylbenzylphatalic acid	3	2	6	(3a)/add.	5
di(2-ethylhexyl)phatalic acid**	3	2	6	(3a)/add.	5
pyridine	2	2	4	3a	2
styrene	3	2	6	-	-
tetrahydrofuran	2	2	4	-	-
tetrahydrothiophene	1	2	2	-	-

** exceeding water-solubility at given C-soil value

see reports 725201001 and 725201008 (add.)

[1] adjusted on the basis of the TCL-value

Table 4. Reliability score of the methods used to derive C-standard values based on ecotoxicological criteria (according to Denneman and Van Gestel, 1990, 1991).

procedure	definition (method, criterion data)	reliability	reliability score @
1a	RAB, geom. average >4 Noe (L) C's	excellent	10
1b-c	RAB, geom. average >4 E (L) C50's divided by 5 or 10	good	9
2a	geom. average <5 NOE (L) C's	number of data	
		4	8
		3	7
		2	6
2b-c	geom. average <5 E (L) C50's	number of data	
		4	5
		3	4
		2	3
3	1 toxicity data	insufficient	2
4	only limited aquatic tox. data	insufficient	2
report 725201008	combination of terrestrial and aquatic toxicity data	reasonable	5

@ If, in addition to the method used, a control procedure has been used, the average of the relevant reliability scores has been taken as the final score.

Vermeire et al. (1991) have furthermore explained the toxicological assessment framework. To this purpose, maximum tolerable risk levels have been specified for carcinogenic and non-carcinogenic substances, based on toxicological information. Combining these two reports with the model for the exposure for individual routes (Van den Berg, 1991) yielded proposals for human toxicologically-based C-standard values.

3.3.2 Determination recommended value for non-genotoxic carcinogens and non-carcinogens

It is suposed that there is a threshold value for these substances, that is to say a dose or concentration below which adverse effects will not occur. **Exceeding the maximum tolerable risk (MTR)** has been taken as the criterion for "serious hazard to public health". In the Memorandum "Dealing with Risks (*Tweede Kamer*, 1988-1989, 21 137, no. 5; VROM, 1988) this level has been set at the level of the **toxicologically tolerable daily intake (TDI)**. This has been defined as the amount of a substance, expressed on the basis of body weight for oral exposure, which can be taken in by man during his entire life without anticipating adverse effects to his health, given present knowledge. TDI, to a certain extent, takes account of extra sensitive sub-populations, like children. This definition was modelled on the acceptable daily intake (ADI) for food additives formulated by the Joint FAO/WHO Expert Committee on Food Additives (JECFA) of the World Health Organization WHO (IPCS, 1987). For many substances from the Guideline, no ADI in the sense of the WHO terminology can be formulated, as they are not denoted as food additives or food contaminants. Other reasons are that these substances have been insufficiently researched in toxicological respect, or that they are considered as genotoxically carcinogenic to man.
The toxicologically tolerable concentration in air (TCL) has been introduced for some, relatively volatile substances. TCL is considered comparable to TDI in this report.

The toxicologically tolerable daily intake (TDI) is to be estimated preferably on the basis of reliable human toxicological data. When these data are not available -and this is mostly the case- the assessment must be based on animal test data. The animal model most relevant to man is chosen on the basis of biological considerations, like a similar metabolism, kinetics or effects. The analysis of the observed effects and dose-response relations in the -preferably long-term- study considered most reliable by the experts, yields the highest dose or concentration level at which adverse effects do not occur, the so-called NOAEL ("No Observed Adverse Effect Level"). NOAEL is the dose or concentration which does not give a statistically or biologically significant increase in the frequency or severity of adverse effects to the exposed population in comparison with a control population. When NOAEL cannot be determined, the lowest dose or concentration still giving adverse effects, LOAEL (Lowest Observed Adverse Effect Level") is determined instead.

TDI is derived from the NOAEL (or LOAEL) value by application of a safety factor, which must take into account uncertainties as a result of inter-species extrapolations, intra-species differences in sensitivity to the relevant substance, as well as differences in the quality of the toxicological data. Generally, the safety factor is a product of uncertainty factors. The standard factors are:

- a factor 10 for the uncertainty arising from differences in sensitivity in the population (intraspecies);
- a factor 10 for the uncertainty in extrapolating from the test animal to man (interspecies).

When the quality of the toxicological data decreases, additional safety factors may be applied. On the other hand, there may be reasons for applying a lower safety factor than usual, for instance when accurate dose-response data for man are available or when man turns out to be less sensitive than the test animal.

Vermeire et al. (1991) introduced the parameter N for the derivation of the intervention values, or the level at which "serious hazard" occurs. In contrast to the derivation of intervention values for the "preventive approach of the ADI or TDI, it is to be recommended to gain an insight into the margin between the extrapolated TDI and the actual lowest effect-level for man in deriving the intervention values in this case. Overestimation of the safety factors (which might occur when more uncertainty factors are included) must be avoided. This has been expressed in the parameter N, which indicates the intervention value level when multiplied by the TDI. The parameter N has a value of 1 (human data), 2 ($VF \leq 250$), 3 ($VF > 250$, but ≤ 1000) or 4 ($VF > 1000$) for non-carcinogenic compounds.

3.3.3 Determination recommended value for genotoxic carcinogens

It is assumed to be impossible to establish a dose or concentration for these substances below which there is no risk of adverse effects, in this case genotoxicity and carcinogenicity. The criterion for "serious hazard to public health" has been determined at **exceeding the maximum tolerable risk (MTR)**. In the memorandum "Dealing with Risks" (VROM, 1988) the level has been set at **that amount of a substance, expressed in terms of body weight in case of oral intake and on the basis of air volume for the intake via the lung (inhalatory), which gives a risk of 1 additional case of cancer per 1,000 individuals exposed during a life-time.**

The dose or concentration with a maximum tolerable risk of 1 : 10^4 is to be estimated preferably on the basis of human-epidemiological data. As these data are usually not available, chronic carcinogenicity studies on test animals must be used instead. In the ideal case, the animal model most relevant to man is chosen on the basis of biological considerations, like similarities in metabolism, kinetics or tumour type. When this approach cannot be chosen, which is mostly the case, it is assumed that man is at least as sensitive as the test animal, rats in most cases. The MTR is estimated by means of a linear extrapolation from the relatively high doses or concentrations in the animal test to the lower exposure level giving an acceptable tumour incidence for man. The extrapolation is carried out from the lowest dose or concentration still giving a significant increase in tumour incidence (see for instance Vermeire et al., 1988). This extrapolation model tends to be conservative, that is to say, the calculated dose or concentration with an acceptable risk is higher rather than lower.

Vermeire et al. (1991) described the toxicity data for most substances mentioned in the Guideline Soil Protection. The recommended values have been derived from these so-called substance profiles. The addendum to report no. 725201005 contains a list of supplementary recommended values for a limited number of substances.

3.3.4 Human exposure

Van den Berg (1991) described one complete exposure model (CSOIL), based on an earlier discussion by Linders (1990) and on studies of other exposure models: SOILRISK (Ten Berge, 1990) and HESP (ECETOC, 1990). With this model, the exposure for adults and children (up to six years old) can be calculated, based on concentrations of contaminants in the soil. Report 725201006 gave a justification of the choices that led to this model. This concerns the routes considered and the quantification of the exposure, including the quantification of potential preliminary stages, like calculations of concentrations in indoor air or in drinking water after permeation of water pipes.

Van den Berg (1991) furthermore gave an insight into the relative contributions of all exposure routes for the substances considered. Inhalation of soil particles, inhalation during showering, dermal contact with soil, intake of drinking water and dermal contact during showering turned out to contribute only marginally with a maximum of 9% for all routes mentioned together for the substances considered here, given that all potential exposure routes are operational. Soil ingestion, crop consumption and inhalation of (particularly indoor) air together constitute at least 90% of the dose. This is probably true for a very wide range of substances.
Soil ingestion is an important factor for some substances (metals, PAH, DDT), as the water and air routes contribute to the total dose only to a limited extent, because of low water-solubility or low volatility. Inhalation of polluted air and consumption of contaminated crops constitute the major contributions for most of the substances.

In addition to the routes mentioned above, the following routes should be considered as well for human exposure to contaminated aqueous soil: dermal contact with water (soil), drinking of water that is in contact with the contaminated aqueous soil and indirect exposure through consumption of fish. Linders (1990) indicated how the exposure via these routes can be calculated. From this report it can be concluded that exposure via certain routes (for instance consumption of contaminated surface water) shows analogy with exposure via the routes described above (consumption of contaminated groundwater as drinking water - see 3.3.6.4), so that they have been implicitly considered. Lastly, it may be concluded from Linders (1990) that the exposure via the remaining routes contributes to the total potential exposure to such an insignificant extent in absolute terms, that they have not been included in the calculation of the C-standard value.

The human-toxicological C-standard value is constituted by the concentration of contaminants in the soil at which the lifetime mean daily dose calculated by means of the CSOIL model, equals the dose pertaining to (exceeding of) the product of the maximum tolerable risk and the parameter N.
The calculated value has been adjusted in a limited number of cases. This was done in cases where for instance the odour threshold value or TCL value (indoor air concentrations) were exceeded.

Table 2 shows the proposed human toxicologically-based C-standard values for the test table substances.

The human-toxicological C-standard values for each substance are not only determined by the human-toxicological recommended value (MTR or MT*R): there is no one-to-one relation between MTR and the C-soil standard value. However, the dose resulting from exposure to an instance of soil contamination with a concentration level in the solid phase equal to the C-soil standard value, does have this one-to-one relation. The way in which exposure takes place, therefore constitutes an important factor. This exposure is mainly determined by the eco-chemical behaviour of the contaminants (the physicochemical parameters). The extent to which the C-soil standard value and the MTR are related, then, is an expression of the most important routes: a very close link for soil-ingestion as the dominant route (low volatility, low water-solubility), a weak link for inhalation as the dominant route (high volatility and/or high water-solubility), and somewhere in between for the routes where crop consumption is the dominant factor.

Each of the parameters, transfer processes or exposure routes used in the CSOIL model is characterized by a degree of uncertainty, sometimes up to a factor 10 or more.
The CSOIL model has been based on the best available knowledge of the different fields. Moreover, it was decided to use simple descriptions in cases where input data were not available. In choosing parameters in the middle ranges and average situtations, a worst-case scenario approach was definitely avoided.
Also with reference to the actual exposure analysis in Chapter 5, the uncertainties connected with the exposure assessment should be clearly stated at the outset. Although it is possible to use local parameter values and exposure routes in the assessment of actual risks, this by no means guarantees the reliability of the results of the model in absolute terms. It is therefore recommended to use measurements, particularly in those cases where the largest uncertainties are anticipated given the assumptions in the present model (see 5.2, steps b and c). The results of estimating the acutal exposure to soil contamination can be readily used for a contrastive analysis.

3.3.5 Reliability score

In order to arrive at a meaningful integration of ecotoxicological and human toxicologically-based C-standard values, these values must be provided with a reliability score.
Table 5 illustrates this reliability score by analogy with the procedure described in 3.2.4. The score is formed by the product of a score for the human toxicological recommended value (score 1 to 3) and a score for the exposure route (likewise 1 to 3).
Table 3 gives all scores, including the resulting overall score for the substances considered. Note that in contrast to current ecotoxicological procedure, a number of scores cannot occur (for instance final scores of 7 to 8), which reduces the discriminating value of the human toxicological standard value. This should be taken into consideration when integrating the proposed C-standard values.

Table 5. Reliability score for the methods used to derive C-standard values based on human toxicological criteria (Van den Berg, 1991).

The final reliability score for the C-standard value is the product of the reliability score for the derived human toxicological recommended values (Vermeire et al., 1991) and the reliability score for the estimated exposure (lower score for an increasing influence of the ecochemical behaviour of the contaminants, resulting in a larger degree of uncertainty in the model).

reliability score derivation human toxicological recommended value

3: ADI or TDI with safety factor ≤ 100 (non-genotoxic carcinogens and non-carcinogens)
2: ADI or TDI with safety factor > 100
1: derived values (particularly substances mentioned in the addendum to Vermeire et al., 1991, see also 3.3.3)

In the case of genotoxic carcinogens a comparable approximation of the reliability score has been chosen, depending on the quality of the information collected in the so-called substance profiles.

reliability score estimation of the exposure

3: relatively small influence of ecochemical behaviour (organic compounds; ingestion exposure route dominant)
2: relatively large influence of ecochemical behaviour (organic compounds; remaining exposure routes)
1: accurate quantification of soil-chemical behavour only to a limited extent (metals and other inorganic compounds)

3.3.6 Additional considerations

Following the publication of the Van den Berg report (1991), individual experts questioned the equilibrium relations between soil and groundwater and the observations concerning permeation of drinking water pipes. This necessitates a closer examination of the facts.

3.3.6.1 Soil-water partition

In addition to the proposed human toxicological standard values for the solid phase, the Van den Berg report (1991) also provided such values for the liquid phase. The latter values were not derived on the basis of an exposure analysis, but calculated from the former values assuming an equilibrium between the solid and liquid phases.
The porewater concentrations determined in this way are substantially higher than the standard values for groundwater included in the present Guideline (milligrammes instead of microgrammes per liter). This has led to a limited study into the soil-water distribution coefficients used by Van de Berg (1991) and in more recent studies.

Tables 6A (metals and arsenic) and 6B (organic substances) compare soil-water distribution coefficients (Kd-values) calculated or measured in related studies to the Van den Berg report (1991). In the study "Streven naar Waarden" (Van de Meent et al., 1990: "Aiming for Values") both empirically and theoretically calculated soil-water distribution coefficients are given. This study has been used for the determination of the Environmental Quality Targets Soil and Water (VROM, 1991). The Provisional Inspection Guideline (VROM, 1989b) gives soil- and groundwater concentrations, that should be interrelated through the soil-water distribution coefficient. For this reason, distribution coefficients have been calculated on the basis of these data. In a similar way, Kd-values were calculated, based on the reference values for soil and groundwater for a standard soil (test table, VROM, 1990a). Baes et al. (1984) proposed distribution coefficients for a large number of metals in order to model the transport of radionuclides through the terrestrial system. These proposals were based on observed Kd-values (Table 6 indicates the range as well as the average) and on calculations. Based on observed BCF-values (for plant-soil accumulation) and Kd values, a relation was derived in this study for the purpose of derving a Kd-value for substances for which no experimental Kd-values were available. Empirical values had already been given in a previous study (Baes, 1982). The same BCF-Kd relation has been used in order to calculate Kd-values based on BCF-data by Sauerbeck (1988). Lagas et al. (1990) gave Kd-values which were calculated on the basis of empirical relations derived from observed distribution coefficients (DHV, 1988). CSOIL (Van den Berg, 1991) uses the Lagas et al. (1990) data for a particular set of parameters (pH: 4, foc: 0,01, CEC: $\mathrm{mmol^{+}.kg^{-1}}$).

Tabel 6A. Vergelijking van Kd-waarden voor metalen en arseen [$dm^3.kg^{-1}$]. De gegevens zijn zowel uit empirische als theoretische studies afkomstig, en waar niet gegeven, is aangenomen dat ze van toepassing zijn op een standaardbodem met 10 % organische stof en 25 % lutum. Voorgesteld wordt als Kd-waarde voor afleiding van grondwater C-toetsingswaarden het geometrisch gemiddelde van al deze studies te nemen.

stofnaam	empirische waarden	range in empirische waarden	geometrisch gemidd. range	berekend met Kd-form.#	berekend met Kd-form.#	berekende waarden DHV-formule	empirische waarden	berekende waarden	geometrisch gemiddelde	CSOIL
	Baes (1982)	Baes et al. (1984)			Sauerbeck (1988)	Lagas et al. (1990) DHV (1988)	Van de Meent (1989)	VROM (1990-a)	van kolom 3,7,8, gg 4+5, gg 6	Van den Berg (1991)
kolom	1	2	3	4	5	6	7	8		
arseen				190	245	60 - 880	6500	2900	**980**	59
barium				58					**58**	50
cadmium	6	1.3- 26.8	6.4	18	4.1	23 - 387	85000	530	**187**	14
cobalt	60	0.2-3800	47	351	245				**117**	100
chroom	5000			841	245		190000	100000	**14400**	100
koper		1.4- 330	35	24	32	198 - 1570	35000	2400	**538**	581
kwik				12	245		110000	6000	**3300**	100
molybdeen		0.4- 400	18	37	11				**19**	50
nikkel				132	245	6 - 350	5300	2330	**565**	22
lood	400	4.5-7640	400	171	245	0.2-730000	430000	5670	**2380**	96
tin				245					**245**	50
zink		0.1-8000	38	8	4.1	70	75000	930	**250**	96

\# Kd-formule: ln Kd = 2.38 - 0.89 * ln BCF
In het eerste geval zijn de BCF-gegevens van Baes et al. (1984) gebruikt, in het tweede die van Sauerbeck (1988).

Please, see next page for the translation

Table 6A. Comparison of Kd-values for metals and arsenic $[dm^3.kg^{-1}]$.
The data have been taken from both empirical and theoretical studies. When not given, it has been assumed that the data apply to a standard soil consisting of 10% organic substance and 25% lutum. It is suggested to take the geometric avarage of all these studies as the Kd-value for the derivation of groundwater C-standard values.

HORIZONTAL
substance name
empirical values
range of empirical values
geometric average range
calculated with Kd.- form.[#]
calculated with Kd.-form.[#]
calculated DHV-formula
empirical values
calculated values
geometric average
CSOIL

VERTICAL
arsenic
barium
cadmium
cobalt
copper
mercury
molubdenum
nickel
lead
tin
zinc

of column 3,7,8, gg 4+5, gg 6

***For the rest of this table (figures) please
see page 28 of the Dutch text!!!!!!

[#] Kd-formula: In Kd = 2.38 - 0.89 * In BCF
In the first case the BCF-data by Baes et al. (1984) were used, in the second case the Sauerbeck (1988) BCF-data.

Table 6B. Comparison of soil-water distribution coefficients (Kd-values) [$dm^3.kg^{-1}$] for a number of organic substances given in or to be calculated from the different studies. Calculated for a standard soil (10% organic substance), pH = 6.

substance name	SNW[1] empir.	SNW[1] calc.	VIR[2]	present test table 3	CSOIL[4]
atrazine	6.9	0			3.1
lindane	250	141	862	1724	108
dieldrin	37500	39622	862	1724	4700
monochlorophenol	9	7	1.1	2870	5
dichlorophenol	22	34		2870	29
trichlorophenol	40	214	123	2870	156
tetrachlorophenol	86	297		2870	46
pentachlorophenol	88	73	431	2870	2775
napthalene	129	79	8620	1440	47
anthracene	2630	791	17240	8620	635
phenanthrene	2291	791	17240	8620	712
fluoranthene		3147		8620	4400
benzo(a)anthracene		9953		8620	8375
chrysene		9953		8620	89750
benzo(k)fluoranthene		25000		8620	142250
benzo(a)pyrene		25000		8620	46000
benzo(ghi)perylene		99527		8620	20550
indene(1,2,3-cd)pyrene		62797		8620	552500
phenol			1.7	172	0.6
benzene			8.6	860	2.8
toluene			14.4	520	10
xylenes			8.6	720	32
trichloroethene			0.9	860	10.6
tetrachloroethene			3.5	860	8.2
tetrachloromethane			43	860	9
1,2-dichloroethane			8.6	860	0.6
monochlorobenzene			52	4310	13.3
octane			862		565
aldrin			862	4310	515000
DDT			862	4310	31000

1 SNW: "Streven naar Waarden" (Van de Meent et al., 1990: "Aiming for Values")

2 VIR: "Voorlopige Inspectierichtlijn Blootstellingsrisico bij bodemverontreiniging: Provisional Inspection Guideline Exposure Risk in cases of soil contamination (VROM, 1989b)

3 VROM (1990a)

4 Van den Berg (1991)

From the information presented in Table 6A (calculated for a standard soil), it becomes clear that widely varying soil-water distribution coefficients are given for metals. The values indicated in "Streven naar Waarden" (Van de Meent et al., 1990) are the highest. These values have been derived from field measurements of concentrations of contaminants in suspended sludge compared to those in surface water. The Van de Meent report indicates that these values will be lower for terrestrial soil. There is no (equilibrium) relation between the final target values formulated for soil and groundwater (VROM, 1991). The target values for soil have been chosen at the level of the reference values. The target value for groundwater was based on observations by the national measuring system groundwater quality.
The values given by Baes et al. (1984) have been based partly on calculations on the basis of the observed BCF-Kd relation and partly on experimental research. The Kd-values calculated on the basis of data by Sauerbeck (1988) virtually match those by Baes et al. (1984), as they are based on similar BCF-values. The Kd-values calculated by Lagas et al. (1990) are based on empirical relations and reflect the substantial influence of soil factors.
As the given Kd-values vary widely and it is impossible to indicate directly which Kd-value should be used in this report to carry out the calculations, it is proposed to approximate the Kd-value for the standard soil in this report (which means adjusting CSOIL) by taking the geometric average of the reported values. In so doing, a number of observations were associated because of similarity or overlapping of data. Table 6A shows the final proposal in bold print. These data were used to derive C-standard values for groundwater. (see also 3.3.7).
With reference to the dependency of the Kd-values on organic substances and lutum, it is suggested to use the soil-type correction factors of the reference values, as they were used by Denneman and Van Gestel (1990) (see also 4.2). For two metals, molydenum and tin, such a relation has not been given. It is suggested to use a direct dependency on the CEC-rate for these substances, as had been suggested before in the Discussion Paper Soil Quality (VROM, 1986) with reference to the metals discussed in that context. The CSOIL model will have to be modified in this respect as well.
Diagram 1 on the next page indicates how this dependency can be calculated, assuming that the C-standard value for groundwater does not depend on the rate of lutum and organic substance (see 4.3). Table 7 shows these relations.
Additional research into the influence of acidity on the equilibrium between the solid and liquid soil-phases, as well as the organic substance rate and other soil parameters is advisable (see also VTCB, 1986).

The approximation for organic substances in "Streven naar Waarden" (Van de Meent et al., 1990) and CSOIL (Van den Berg, 1991) has been based on the same starting-points. The figures given in the two studies do differ, however, which is due in part to the choice of the octanol-water distribution coefficients.

Diagram 1. **Derivation soil-type corrections for C-standard values soil and soil-water distribution coefficients.**

$$R(L,H) = F(L,H)$$

R (L,H) : reference value for
L % lutum and
H % organic substance

F (L,H) : soil-type correction formula for the reference value

$$Cgr(L,H) = \frac{Car(25,10)}{Kd(25,10)} * F(L,H)$$

Cgr(L,H) : C-soil standard value as function of L and H
Cgr(25,10) : C-soil standard value for standard soil
standard soil: 10% organic substance, 25% lutum
R(25,10) : reference value standard soil

$$Cgw(25,10) : \frac{Car(25,10)}{Kd(25,10)}$$

Cgw(25,10) : C-groundwater standard value for the standard soil
Kd(25,10) : soil-water distribution coefficient for the standard soil as represented in Table 6A

assumption : Cgw(L,H) is constant; Cgw(L,H) = Cgw(25,10) (see 4.3)

$$Kd(L,H) = \frac{Car(L,H)}{Cgw(L,H)} = \frac{Car(L,H)}{Cgw(25,10)} = \frac{Kd(25,10) * F(L,H)}{R(25,10)}$$

Kd(L,H) : soil-water distribution coefficient as function of L and H

Cgw(L,H) : C-groundwater standard value as function of L and H

The main difference in comparison with "Streven naar Waarden" is that CSOIL does not use a correction factor for dissociating compounds. This approximation needs to be incorporated in the CSOIL model (see annex 1.1). For the derivation of the human toxicological C-standard values, such a correction is only relevant for the higher chlorophenols (more than two chlorine substituents). The results of this correction are given in 3.3.8.

In deriving the target values, calculated values have only been used when experimental values were not available. It can be concluded from Table 6B that the experimental values generally correspond with the calculated values.

The values calculated on the basis of data from the Provisional Inspection Guideline (1989b), are slightly comparable to the previously mentioned data. The relations between the present C-standard values for soil and groundwater laid down in Kd-values, however, deviate strongly from the other values in the Table. The values are mostly too high, and the concentrations in the groundwater consequentially too low. This cannot be explained on a theoretical basis, nor on the basis of data presently available. For this reason it is proposed to maintain the values for organic substances used in the CSOIL model (Van den Berg, 1991) unchanged.

Tabel 7. Overzicht van de bodemtypecorrectieformules voor de referentiewaarden, grond-waterverdelingscoëfficiënten en C-toetsingswaarden voor grond voor metalen en arseen.

	stofnaam	F(L,H)[1]	R(25,10)	Kd(25,10)	Kd(L,H)	Cgr(25,10)	Cgr(L,H)
1	arseen	15+0.4(L+H)	29	980	500+13.5(L+H)	55	28+0.8(L+H)
2	barium	300+3.9(L)[2]	400[2]	60	45+0.6(L)	650	490+6.3(L)
3	cadmium	0.4+0.007(L+3H)	0.8	190	95+1.7(L+3H)	12	6+0.1(L+3H)
4	chroom	50+2(L)	100	14400	7200+290(L)	380	190+7.6(L)
5	cobalt	10+0.17(L)[2]	14[2]	120	85+1.5(L)	120	85+1.5(L)
6	koper	15+0.6(L+H)	36	540	225+9(L+H)	190	80+3.2(L+H)
7	kwik	0.2+0.0017(L+H)	0.3	3300	2200+19(L+H)	10	6.7+0.06(L+H)
8	lood	50+(L+H)	85	2400	1400+28(L+H)	530	310+6.2(L+H)
9	molybdeen	0.18(L+3H)[3]	10	20	0.36(L+3H)	37	1.3(L+3H)
10	nikkel	10+(L)	35	560	160+16(L)	210	60+6(L)
11	tin	0.36(L+3H)[3]	20	240	4.3(L+3H)	300	65(L+3H)
12	zink	50+1.5(2L+H)	140	250	90+2.7(2L+H)	720	260+7.7(2L+H)

1 VROM (1987) Milieuprogramma 1988-1991 Voortgangsrapportage
2 TCB (1990) Advies bouwstoffen
3 dit rapport, op basis VROM (1986) Discussienotitie bodemkwaliteit

F(L,H) : bodemtypecorrectieformule voor de referentiewaarde [mg.kg^{-1}]
L : percentage lutum
H : percentage organische stof
R(25,10) : referentiewaarde standaardbodem [mg.kg^{-1}]
standaardbodem: 10 % organische stof, 25 % lutum, dus: H=10, L=25
Kd(25,10) : grond-waterverdelingscoëfficiënt voor de standaardbodem [dm^3.kg^{-1}], zoals voorgesteld in tabel 6A
Kd(L,H) : grond-waterverdelingscoëfficiënt als functie van L en H [dm^3.kg^{-1}], berekend volgens schema 1
Cgr(L,H) : C-toetsingswaarde grond als functie van L en H [mg.kg^{-1}]; berekend volgens schema 1
Cgr(25,10) : C-toetsingswaarde grond standaardbodem [mg.kg^{-1}]; voorstel voor de geïntegreerde waarde (zie 4.1.1)

Please, see next page for the translation

Table 7. **Survey of the soil-type correction formulas for the reference values, soil-water distribution coefficients and C-soil standard values for metals and arsenic.**

substance name
1 arsenic
2 barium
3 cadmium
4 chromium
5 cobalt
6 copper
7 mercury
8 lead
9 molybdenum
10 nickel
11 tin
12 zinc

1 VROM (1987) Envrionmental Programme 1988-1991, Progress Report
2 TCB (1990) Advice Building Materials
3 this report, based on VROM (1986) Discussion Paper Soil Quality

F(L,H) : soil-type correction formula for the reference value [mg.kg^{-1}]
L : lutum percentage
H : organic substance percentage
R(25,10) : reference value standard soil [mg.kg^{-1}]
standard soil: 10% organic substance, 25% lutum, so: H=10, L=25
Kd(25,10) : soil-water distribution coefficient for the standard soil [dm.kg^{-1}], as represented in Table 6A
Kd(L,H) : soil-water distribution coefficient as function of L and H [dm.kg^{-1}], calculated according to diagram 1
Cgr(l,H) : C-soil standard value as function of L and H [mg.kg^{-1}]; calculated according to diagram 1
Cgr(25,10) : C-standard soil value [mg.kg^{-1}]; proposal for the integrated value (see 4.1.1)

The actual exposure calculations (see Chapter 5) must not be based on one distribution coefficient per substance. The distribution of the Kd-values must be taken into consideration.

3.3.6.2 *Toxicity test groundwater organisms*

The distribution of contaminants over the solid and liquid soil-phases raises the question whether the concentrations calculated in the pore water (Cpw), as well as groundwater concentrations derived from this, may have effects on groundwater organisms. As there are hardly any toxicological data on groundwater organisms, this issue may be approached from the proposition that the sensitivity of these organisms to substances will be similar to that of aquatic organisms. Van Gestel and Ma (1988, 1990) showed that the concentration in the soil moisture determines the toxicity of substances in the soil for worms. It became clear that soil moisture concentrations are readily assessable by means of the soil-water distribution coefficients. Moreover, for a large number of substances, a strong correlation was found between the sensitivity of worms (on the basis of soil mositure concentrations) and the sensitivity of fishes. Such experiences have also been described in the literature for sediment-organisms, with the result that in the United States the risk assessment of contaminations of sediments is presently based primarily on aquatic toxicity data (Giesy and Hoke, 1989) by means of the so-called equilibrium partition method.
This theory has been used in order to generate more reliable ecotoxicological C-standard values for soil (see also 3.2).
This theory therefore also works the other way round. A quantitative approach is not useful for the time being, as the "serious hazard" criterion for terrestrial organisms also covers groundwater organisms.

3.3.6.3 *Drinking water pipe permeation*

For the permeation of drinking water pipes, a distinction must be made between permeation from the vapour phase (when the drinking water pipe is located in the water-unsaturated zone) and the aqueous phase (when the pipe is located beneath the groundwater table. The calculations in the CSOIL model were based on the permeation from the aqueous phase only. Although Van den Berg discussed this choice in greater detail in section 4.3.3 (1991), a number of additional observations should be made.

The same formula is used both for the calculation of permeation from the vapour phase and for permeation from the aqueous phase (Vonk, 1985), while the concentration and the permeation coefficients were related to the vapour phase and aqueous phases respectively. The relation between the concentration in the soil air (Csa) and the pore water (or the higher groundwater) (Cpw) is given by the dimensionless Henry-constant (H = Csa/Cpw). Veenendaal et al. (1985) state that the permeation coefficients for vapour and water are to each other as the inverse of the maximum concentrations in these phases, which corresponds to the inverse of the Henry-constant (Dpe vapour/Dpe water = 1/H). This means that identical concentrations in the drinking water are calculated (in the given equilibrium situation) - independent of the phase from which permeation takes place. Therefore, making further distinctions in the model is not considered relevant.

Table 8. Comparison of the concentrations in drinking water after permeation in the drinking water pipe from the vapour- and aqueous phase.

permeation aqueous phase

compound	Cs [mg.kg^{-1}]	Cpw [g.m^{-3}]	Dpe [m^2.d^{-1}] *10^{-7}	Cdw w [g.m^{-3}]
toluene	132	31.6	12	0.173
trichloroethene	247	56.2	16	0.41
tetrachloroethene	5.13	1.46	7.7	0.0051
chlorobenzene	0.59	0.108	35	0.0017

permeation vapour phase

compound	Cs [mg.kg^{-1}]	Cpw [g.m^{-3}]	Dpe [m^2.d^{-1}] *10^{-7}	Cdw d [g.m^{-3}]	**Cdw w/ Cdw d** [-]
toluene	132	6.89	76	0.239	0.72
trichloroethene	247	22.9	82	0.86	0.48
tetrachloroethene	5.13	1.28	35	0.020	0.25
chlorobenzene	0.59	0.0121	140	0.00077	2.23

Cs : content in the soil
Cpw : concentration in the pore water
Csa : concentration in the soil air
Dpe : permeation coefficient
Cdw : drinking water concentration;
w: aqueous; d: vapour

In Table 8 a comparative analysis has been made for a number of substances for which concentrations in the drinking water were calculated (by means of <u>measured</u> permeation coefficients) after permeation from the aqueous- and vapour phases. From this it becomes clear that the relation of the concentrations in the drinking water after permeation from the aqueous and vapour phases deviates from the ratio by a factor 4 maximally. Modification of the calculation method described in Van den Berg (1991) is not considered necessary, given the distribution of the model's application.

A Rotterdam Municipal Drinking Water Company study was taken as the starting-point for the assessment of the permeation coefficients (Van der Heijden and Hofman, 1987). For a number of compounds, estimations have been made and used for the exposure calculations. In view of the results from the drinking water permeation calculations, the permeation coefficient for cyclohexanone has been critically evaluated. Data by Veenendaal et al (1985) show that the estimated $20*10^{-7}m^2.d^{-1}$ is far too high and should be set at less than $1*10^{-7}m^2.dm^{-1}$. This is corroborated both by comparison with other ketones (measurement methyl-ethyl ketone: $0.058*10^{-7}m^2.d^{-1}$) and by calculations based on the diffusion and partition coefficient. The partition coefficient is proportional to the solubility of the substance in LDPE (Low-Density polyethene),

which is relatively low for ketones, and inversely proportional to the water-solubility, which is high for cycolohexanone.
This consideration has led to a renewed calculation for cyclohexanone. The result has been incorporated in Table 11. It does not differ from the value proposed earlier, as in both cases adjustment on the basis of the TCL-value was required.

3.3.6.4 Testing against drinking water standards

Van den Berg (1991) discussed possible adjustments to the proposed human-toxicological C-standard values following comparison with other criteria, such as drinking water norms. There are two forms of testing.
In the first place, concentrations of contaminants in the pore water (or the groundwater) can be tested directly against the drinking water norms. In this way it can be ascertained in how far consumption of unpurified drinking water withdrawn from groundwater (for instance in private water catchment) leads to exposure. It should be made clear at the outset that, in agreement with the Provisional Inspection Guideline (VROM, 1989b), drinking water norms in this context (as a basis for the derivation of intervention values) need not directly apply to the calculated pore water concentrations or groundwater concentrations derived from them.
However, it may be argued that it must be possible to use the contaminated groundwater as drinking water without additional structural purification. This means that the consumption of drinking water must not lead to cases where the human-toxicological recommended value are exceeded.
Table 9 indicates at which concentration in the groundwater (C-max) the human-toxicological recommended value (the factor N having been taken into consideration) is reached (excluding all remaining exposure routes), if the groundwater is used directly as drinking water. Higher concentrations in the groundwater used for drinking water, then, lead to exceeding of the human-toxicological recommended value. Anticipating the integration of the proposals for ecotoxicological and human-toxicological C-standard values (see Chapter 4), Table 9 also gives the pore water concentration given the integrated C-soil standard value. For approximately one third of the substances (indicated by an *), the calculated maximimum tolerable concentration in groundwater (C-max) is lower than the pore water concentration in equilibrium with the integrated C-soil standard value. With the exception of a few readily soluble substances, like dihydroxybenzenes, thiocyanates and free and complex cyanide) as well as two metals (barium and cobalt), C-max is exceeded by less than a factor 10.

Table 9. Comparison of the maximum tolerable concentration in groundwater (C-max), based on the consumption of this groundwater as drinking water, with the concentration in the pore water (Cpw) in equilibrium with a content in the soil equal to the integrated C-soil standard value.
The concentrations are given in mg.m^{-3} [= μg.l^{-1}].
When C-max is exceeded by Cpw, this is indicated by an *

	C-max [mg.m^{-3}]	Cpw [mg.m^{-3}]	
arsenic	66	41	
barium	629	11000	*
cadmium	31	63	*
chromium	472	16	
cobalt	88	1000	*
copper	8810	350	
mercury	19	3	
lead	54	68	*
molybdenum	629	1850	*
nickel	3140	375	
tin	126000	1250	
zinc	31400	2880	
cyanides free	3140	139000	*
cyanides complex	1230	54000	*
thiocyanates	346	15200	*
benzene	5350	14200	*
ethylbenzene	12800	4400	
phenol	5660	49900	*
cresol	4720	1830	
toluene	13500	12700	
xylene	943	1920	*
catechol	3770	102000	*
resorcinol	1890	50500	*
hydroquinol	2360	55100	*
anthracene	75	54	
benzo(a)anthracene	10	4.1	
benzo(k)fluoranthene	0.6	0.24	
benzo(a)pyrene	0.3	0.3	
chrysene	1.5	0.38	
phenanthrene	629	48	
fluoranthene	265	7.8	
indene(1,2,3cd)pyrene	0.1	0.06	
benzo(ghi)perylene	0.26	0.26	
napthalene	4720	731	

Table 9 ctnd. Comparison of the maximum tolerable concentration in groundwater (C-max), based on the consumption of this groundwater as drinking water, with the concentration in the pore water (Cpw) in equilibrium with a content in the soil equal to the integrated C-soil standard value.
The concentrations are given in mg.m^{-3} [= μg.l^{-1}].
When C-max is exceeded by Cpw, this is indicated by an *

	C-max [mg.m^{-3}]	Cpw [mg.m^{-3}]	
1,2-dichloroethane	440	4320	*
dichloromethane	3770	20400	*
tetrachloromethane	252	206	
tetrachloroethene	1510	1440	
trichloromethane	2830	11400	*
trichloroethene	34000	4830	
vinyl chloride	110	6.7	
monochlorobenzene	18900	1930	
dichlorobenzene	11900	500	
trichlorobenzene	31	36	*
tetrachlorobenzene	31	14	
pentachlorobenzene	31	6.5	
hexachlorobenzene	31	4.8	
monochlorophenol	189	1670	*
dichlorophenol	189	296	*
trichlorophenol	189	55	
tetrachlorophenol	189	187	
pentachlorophenol	1890	1.6	
chloronaphthalene	16	28	*
trichlorobiphenyl	8.5	0.8	
hexachlorobiphenyl	8.5	0.14	
DDT	3.1	0.11	
DDE	40	0.31	
aldrin	6.3	0.0006	
dieldrin	6.3	0.73	
endrin	6.3	1.9	
HCH	63	16	
carbaryl	120	0.9	
carbofuran	500	0.71	
propoxur	1260	768	
maneb	0.1	0.047	
atrazine	314	1600	*

heptane	9500	50	
octane	660	11	
cyclohexanone	289000	452000	*
butylbenzylphtalic acid	1570	91	
di(2-ethylhexyl)phtalic acid	285	19	
pyridine	94	101	*
styrene	4840	3460	
tetrahydrofuran	943	32	
tetrahydrothiophene	943	20	

As groundwater instead of pore water is used for the collection of private water for drinking water production, so that dilution may be assumed to take place, and in view of what has been said before with reference to the relations between concentrations in the solid and aqueous phases, particularly in the case of metals, it is not considered necessary to adjust the C-standard values in the light of these figures.
Section 4.1 argues against basing the C-standard values for groundwater directly on the pore water concentrations acquired through equilibrium calculations. When these standard values for groundwater are not exceeded, this means that given direct lifelong consumption of the groundwater referred to, adverse effects are not anticipated. It should be emphasized, however, that soil contamination does operate under these circumstances; a situation such as this cannot be referred to as acceptable in itself.
When the C-standard values for groundwater are exceeded, a further research can lawfully be instigated into the actual exposure and the accompanying risk of health damage related to possible private water collection (see Chapter 5).

In a letter by the Minister of Housing, Spatial Planning and the Environment to the chairman of the *Tweede Kamer* (Second Chamber) (VROM, 1989c), further details were given on the lines of conduct concerning the disclosure of cases in which drinking water norms have been exceeded and the approach to be followed by the Public Health Inspector for Environmental Protection when this is the case. If the drinking water norms are exceeded as a consequence of a (too) high content of a substance in the source from which water is withdrawn for drinking water, it goes without saying that measures directed to eliminating the contamination will be sought first of all. Regardless of the nature of the measures, time will be consumed in determining the necessity of and implementing the measures per se. The inspector must decide between continuing the drinking water supply, while the norms for that particular substance are exceeded and discontinuing the drinking water supply, or changing over to an alternative drinking water supply. In this consideration, a distinction is made between exceeding norms by substance categories for which an exemption may be issued and those for which this is not the case. The procedure for granting exemption and the criteria used are clearly specified in the *Waterleidingbesluit* (Water Supply Decree) (*Staatsblad, 1984)*, based on EU regulations. Relevant to the purposes of the present report are those situations in which an exemption cannot be granted or situations in which the exemption criteria are not met. If no toxicological norms formulated by the World Health Organization are available for that particular substance and if the contents found do not exceed that particular norm, it may be tolerated that the *Wateleidingbesluit* norm is temporarily exceeded. When there are no international norms for that particular substance, an assessment will be made of the medically tolerable daily intake based on expert advice. If this value is not exceeded, the drinking water supply may be continued. When the value is exceeded, which is mostly based on the tolerable lifelong daily intake of the individual substance, further considerations must be made, taking into account the reliability of the toxicological information and the significance for sensitive groups. If such an evaluation indicates an intolerable medical risk, the consumption of mains supply water will be advised against, or the supply will be discontinued until such time as the necessary measures will have been taken. A similar line of conduct is followed when substances are found in the drinking water for which no norms have been formulated in the *Waterleidingbesluit*. The situation as well as possible measures is therefore assessed on a case-by-case basis.

Secondly, the concentrations found in tap water after permeation of the drinking water pipeline may be tested by comparison with drinking water norms.
In analogy with the procedure described above, the drinking water concentrations (after permeation in the drinking water pipeline) may be tested by comparison with exceeding of the MTR-level, given an exposure only via this route. In this case it can be calculated that the MTR-level is not exceeded by more than 10% for any of the substances. This reference point is used for instance in the Provisional Inspection Guideline (VROM, 1989b) in cases for which no drinking water norms exist. In view of the above, the conclusion already drawn by Van den Berg (1991) not to adjust the C-standard values on the basis of possible permeation, including exceeding of the drinking water norms, is borne out by the following arguments:

1. concentrations of more than 10% of the MTR-level are not found in the drinking water as a result of permeation;
2. the permeation calculation method chosen, is a clear case of a "worst-case scenario" calculation;
3. in the case of a large number of micro-contaminations, the drinking water norm has been based on the starting-point that these substances ought not to be found in the drinking water; the drinking water norm has therefore been set at the analytical detection limit and does not constitute an effect-related norm;
4. continously adjusting of the C-standard values to changing or new drinking water norms is not expedient.

In the case of aromatics and volatile chlorinated aliphatic hydrocarbons, a pore water concentration in the soil may be found that, without exceeding the C-soil standard value, may lead to exceeding of the drinking water norm as a result of the permeation process, if the drinking water pipeline consists of a type of material sensitive to permeation. Given the stipulations in the *Waterleidingwet* (Water Supply Act), the owner of a Water Company is obliged to guarantee the supply of sound drinking water to the consumer. Under this Act the owner is also bound to see to it to the best of his ability that (in addition to the means of water catchment, treatment and storage), the means of transport and distribution are such and are used and maintained in such a way that the drinking water is not contaminated.

Even if the C-standard value has been exceeded to such an extent that there is "serious hazard to public health or the environment", the situation with reference to the owner of a Water Company still applies. The *Gedeputeerde Staten* (Provincial Executive) may order a soil contamination research in these cases or take temporary safety measures (see also Chapter 2) and subsequently take clean-up measures in the framework of the Provisional Soil Clean-up Act. The Water Supply Act and the Provisional Soil Clean-up Act overlap in this case, pertaining to the responsibilities and the resulting financial consequences for the Water Company or the Provincial Executive with respect to the implementation of research and measures. Thus, a situation may arise where one measure saves the need of another. These responsibilities have meanwhile been coordinated. The Guideline specifies the activities that fall under the responsibility of the Water Company (e.g. tap water research, pipeline material, the surrounding soil, replacing of pipelines).

Over the past few years a great deal of field experience has been gained with respect to situations of soil contamination. Some Water Companies (for instance the Rotterdam Municipal Drinking Water Company - Van der Heijden and Hofman, 1988) have formulated assessment procedures. It was found that in contrast to what was expected, based on model calculations, relatively high concentrations of aromatics and phenol do not always lead to permeation. Likewise, expectations regarding the permeation sensitivity of pipeline material (polyethylene, polyvinylchloride) were not always corroborated in actual practice. Finally, the lingering period of contaminants in the soil may influence permeation. When a substance lingers in the soil for a long time, this may lead to a relatively limited permeation in relation to model estimations.

Further verification of model calculations in field tests given varying contamination situations, is required in order to be able to improve the formulas. Although both the Inspection Guideline for the assessment of contaminations of drinking water as a result of permeation (VROM, 1985) and the Provisional Inspection Guideline Exposure Risk of Soil Contamination (VROM, 1989b) offer good indications of the risk of permeation, it may be considered advisable or even necessary to start research or take measures in cases where concentrations of contaminants deviate from the so-called signal-values.

3.3.6.5 *The deeper groundwater*

Although the pore water concentration may be taken to be representative of the undeep groundwater concentration, the relation between the deeper groundwater (with lower organic substance contents in the soil and completely water-saturated) and the pore water or upper groundwater has not yet been described. From calculations and theoretical considerations, it may be concluded that when all pores are water-saturated, volatilization (via the vapour phase) will no longer occur. Given a like concentration of the contamination in the groundwater, the dose will therefore decrease or remain the same, when volatilization plays an important part in the exposure. In 4.2 it is indicated that the C-standard value for groundwater is independent of soil parameters (and so of differences between organic substance contents in the top- and sub-soil). Depending on the contribution of volatilization to the exposure, the dose pertaining to the C-standard value for groundwater in an unsaturated top-soil will be equal to or higher than that for the same concentration in the water-saturated sub-soil.

3.3.7 Additions to and adjustments of human-toxicological C-standard values

Compared to the Vermeire et al (1991) and Van den Berg (1991) reports, additions and adjustments have been proposed for a number of substances. These changes will be briefly discussed in this report. More detailed information will be made available through the revised reports (725201005 and 725201006) in due time.

Vermeire (1991) derived TDI values for a number of substances, however without including them in the overview. In many cases the Vermeire report confined itself to group values. These individual data have therefore not been incorporated into the proposals for human-toxicological C-standard values in a number of cases (Van den Berg, 1991). Following further consultations on this matter, it has been decided to use these data as yet. The basic information and the results of the calculations have been given in Tables 10 and 11 respectively. Substances for which no human-toxicological recommended value were available on publication of the Van den Berg report (1991), are also given in these tables (these substances have been included in the addendum to report 725201005.

In the light of the deviating TDI for ß-HCH, an exposure calculation has been performed for all HCH-isomers and C-values have been proposed. The results of the calculation have been included in Table 11.

Given the influence of soil-acidity on the degree of dissociation, modified human-toxicological C-standard values have been derived for the higher cholorinated chlorophenols (> 2 Cl-substituents), given a pH-value of 6. These values are also shown in Table 10.
Annex 1.1 discusses the calculation of the pH-correction in connection with the calculation of the Kd-values for organic substances.

In checking the proposals for human-toxicological C-standard values derived by Van den Berg (1991), it was found that an incorrect calculation procedure had been used for zinc. This has been corrected in this report (see adjusted values in Table 11).

Table 10. Survey of the toxicological recommended values, the extrapolation method applied and the daily exposure in a non-contaminated situation.

substance name	toxicological recommended value $\mu g.kg\ lg^{-1}.d^{-1}$ or $\mu g.m^{-3}$	type[1]	safety factor/ extrapolation method[2]	daily exposure $\mu g.kg\ lg^{-1}.d^{-1}$
monochlorobenzene	300	TDI	100 (d)	< 0.07
	1000	TCL	1 (h)	
dichlorobenzene	190	TDI	100 (d)	< 0.07
	600	TCL	100 (d)	
atrazine	5	TDI	100 (d)	0
	1	TDI	1 (h)	0
sulfides	50	TDI	(s)	0.0004
anthracene	50	TDI	1000 (d)	0
napthalene	0.5	TDI	(s)	0
chlorinePAH	10	TDI	(s)	0
tetrahydrothiophene	35	TCL	(s)	0

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1 TDI: toxicologically tolerable daily intake [$\mu g.kg\ lg^{-1}.d^{-1}$]
TCL: toxicologically tolerable concentration in air [$\mu g.m^{-3}$]

2 (d): extrapolation from NOAEL or LOAEL determined on the basis of animal tests
(h): extrapolation form human odour or taste threshold level
(s): derived from structure-related compound(s)

Table 11. Proposal for human-toxicological C-standard values for soil and groundwater (standard soil: foc = 0.058

Substance name	N[1]	proposals human-toxicological C-standard values soil [$mg.kg^{-1}$]	groundwater[3] [$g.m^{-3}$]
monochlorobenzene	2	716[1]	46
dichlorobenzene	2	1754	30
atrazine	2	21.5	5.95
sulfides	1	0.18	1.38
anthracene**	3	86900	0.075
napthalene	3	977	18.5
chloronapthalene	1	4.8	0.0286
tetrahydrothiophene	3	5.4[2]	0.020
zinc		1840	19.1
trichlorophenol		56.8	0.34
tetrachlorophenol		24.9	0.75
pentachlorophenol		76.0	0.47

α-HCH	21.0	0.117
β-HCH	0.63	0.038
γ-HCH	18.3	0.153
δ-HCH	24.3	0.086
cyclohexanone	770[2]	464

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[1] parameter N (Vermeire et al., 1991)

[2] adjusted on the basis of the TCL-value

[3] directly derived from the C-standard value for soil by means of equilibrium relations

** water-solubility exceeded at this C-soil standard value

4 INTEGRATED PROPOSALS TOWARDS A SINGLE C-STANDARD VALUE

4.1 The Integration

In this chapter a proposal will be made for the integration of the proposed human-toxicological and ecotoxicological C-standard values into one single C-standard value. Apart from the numerical content of the proposed human-toxicological and ecotoxicological standard values (see Table 2), the reliability of these individual values (see Table 3) is an equally important factor in the integration. As is to be expected, the more stringent of the two values will normally be chosen, unless this value has a substantially reduced reliability. In such cases, the relatively less stringent value is chosen as starting-point. As already stated in 3.3.5, the human-toxicological reliability score is less distinguishing than the ecotoxicological reliability score. This has been taken into account in the integration. Otherwise, the two scores are considered of equal value.

4.1.1 C-soil standard values

Table 12 shows the result of the integration process in the form of the final proposals for integrated toxicologically-based C-soil standard values. This Table also supplies the information used in this context: the ecotoxicological and human-toxicological C-standard values with their reliability scores.

In the case of two compounds, DDT and maneb, it was decided not to use the less stringent (but relatively more reliable) human-toxicological values, but the more stringent ecotoxicological values with a lower reliability. The arguments for this choice are:

1. these compounds are characterized by low water-solubility, with the result that ingestion constitutes the main exposure route, so that:
   a. a less stringent C-standard value is calculated;
   b. a higher reliability score is given;
   in fact, the reliability score is overrated by this combination;
2. all other pesticides have been given similarly stringent C-soil standard values.

4.1.2 C-groundwater standard values

As was explained in 3.3.6.1, the C-standard value for groundwater is normally derived from the C-soil standard value based on the equilibrium theory. The relation between the two values is determined by the soil-water distribution coefficient. In cases of contamination, it is found that due to problems of a measurement-technical nature, the high degree of heterogeneity of the soil and related to this, of the ecochemical behaviour of the contaminants, equilibrium situations can be shown only to a limited extent, or equilibium is only found between the solid phase (soil) and the liquid phase (the pore water between the soil particles) over a limited distance.

Table 12. Integration table.

Proposal for integrated C-soil standard values, for a standard soil (10% organic substance, 25% lutum).

Based on the proposed human-toxicological and ecotoxicological C-standard values and the matching reliability scores (BS).

Substance name	C-hum.-tox [mg.kg^{-1}]	BS	C-ecotox. [mg.kg^{-1}]	BS	C-integration [mg.kg^{-1}]
I Metals					
arsenic	300	3	40	6	40
barium	698	3	650	2	650
cadmium	1.8	3	12	10	12
chromium (III)	2140	2	230	7	230
chromium (VI)	0.1	1	-	-	
cobalt	400	3	120	2	120
copper	5020	3	190	10	190
mercury	87	3	10	6	10
lead	162	3	200	10	290
molybdenum	37	3	<480	-	37
nickel	179	3	210	6	210
tin	571000	3	-	-	300
zinc	1840	3	720	6	720
II Inorganic compounds					
cyanides free	18.5	3	-	-	18.5
cyanides complex	7.2	2	-	-	7.2
thiocyanates	2.0	3	-	-	2.0
sulfides	0.18	1	-	-	0.18
III Aromatic compounds					
benzene	48	6	25	2	48
ethylbenzene	149	4	-	-	149
phenol	41	6	40	2	41
cresol	4.6	4	50	2	4.6
toluene	368	6	150	6	150
xylene	73	4	-	-	73
catechol	32	4	-	-	32
resorcinol	14	4	-	-	14
hydroquinol	12	4	-	-	12

** Water-solubility exceeded at this C-soil value

Table 12 (ctnd.) **Integration Table.**
Proposal for integrated C-soil standard values, for a standard soil (10% organic substance, 25% lutum).

Based on the proposed human-toxicological and ecotoxicological C-standard values and the matching reliability scores (BS).

Substance name	C-hum.-tox [mg.kg^{-1}]	BS	C-ecotox. [mg.kg^{-1}]	BS	C-integration [mg.kg^{-1}]
IV Polycyclic Aromatic Hydrocarbons					
anthracene**	86900	3	40	5	40
benzo(a)anthracene**	10100	6	40	5	40
benzo(k)fluoranthene**	10900	6	40	5	40
benzo(a)pyrene**	996	6	40	5	40
chrysene	88	4	40	5	40
phenanthrene	319	4	40	5	40
fluoranthene	496	4	40	5	40
indene(1,2,3cd)pyrene**	11400	6	40	5	40
benzo(ghi)perylene**	11800	6	40	5	40
napthalene	977	2	40	5	40
V chlorinated hydrocarbons					
aliphatic chlorohydrocarbons					
1,2-dichloroethane	3.5	6	60	5	3.5
dichloromethane	39	6	60	5	39
tetrachloromethane	2.2	4	60	5	2.2
tetrachloroethene	14	4	60	5	14
trichloromethane	27	4	60	5	27
trichloroethene	666	6	60	5	60
vinyl chloride	0.09	6	60	5	0.09
chlorobenzenes					
monochlorobenzene	716	6	30	5	30
dichlorobenzenes (ind.)	1754	6	30	5	30
trichlorobenzenes (ind.)	10	4	30	5	10
tetrachlorobenzenes (ind)	17	4	30	5	17
pentachlorobenzene	23	4	30	5	23
hexachlorobenzene	25	6	30	5	25
chlorophenols					
monochlorophenols (ind.)	16	4	10	5	10
dichlorophenols (ind.)	34	6	10	5	10
trichlorophenols (ind.)	57	4	10	5	10
tetrachlorophenols (ind.)	25	4	10	5	10

pentachlorophenol	76	6	5	5	5
chloropolycyclic aromatic hydrocarbons					
chloronaphthalene	4.8	2	-	-	4.8
polychlorobiphenyles					
trichlorobiphenyl	7.6[1]	4	70[2]	2	7.6
hexachlorobiphenyl	12[1]	4	70[2]	2	12

** water-solubility exceeded at this C-soil value

[1] Aroclor 1248, 48% chlorine - mean 4 Cl-substituents

[2] Aroclor 1254, 54% chlorine - mean 6 Cl-substituents

Table 12 (ctnd.) **Integration Table.**
Proposal for integrated C-soil standard values, for a standard soil (10% organic substance, 25% lutum).

Based on the proposed human-toxicological and ecotoxicological C-standard values and the matching reliability scores (BS).

Substance name	C-hum.-tox [mg.kg^{-1}]	BS	C-ecotox. [mg.kg^{-1}]	BS	C-integration [mg.kg^{-1}]
VI Pesticides					
organo-chlorine pesticides					
DDT**	10400	9	4	5	4
DDE**	3450	6	4	5	4
aldrin	12	4	0.35	6	0.35
dieldrin	5.0	4	4	5	4
endrin	4.1	4	0.06	2	4.1
α-HCH	21	6	2	6	2
β-HCH	0.63	4	-	-	0.63
γ-HCH	18	6	2	6	2
δ-HCH	24	6	2	6	2
other pesticides					
carbaryl	473	6	5	7	5
carbofuran	409	6	1.5	10	1.5
propoxur	650	6	0.75	2	650
maneb**	58800	6	30	4	30
atrazine	21	6	6	7	6
VII Other contaminants					
heptane	10	2	-	-	10
octane	7.2	-	-	-	7.2
cyclohexanone	770	4	-	-	770
butylbenzylphtalic acid	754	6	60	5	60
di(2-ethylhexyl)phtalic acid**	8890	6	60	5	60
pyridine	3.5	4	150	2	3.5
styrene	102	6	-	-	102
tetrahydrofuran	1.2	4	-	-	1.2
tetrahydrothiophene	5.4	2	-	-	5.4

** water-solubility exceeded at this C-soil value

From the empirically determined and calculated distribution coefficients and the variation in these values (for different studies) presented in 3.3.6.1, it may moreover may be concluded that explicit standard values for pore water cannot be directly derived merely on the basis of the equilibrium theory, since the concentrations in the (deeper) groundwater will deviate from pore water concentrations, for instance due to effects from a non-contaminated area (dilution).

In view of the above, there are plausible reasons for lowering the C-standard value for groundwater as compared to the pore water concentrations derived from the equilibrium calculations and consequentially base the legitimization of a further research and the implementation of clean-up measures in the context of the current legislative framework on groundwater observations.

It is proposed to lower the pore water concentrations, determined on the basis of a theoretical equilibrium, by a factor 10 for organic substances and inorganic substances other than metals and arsenic. This also implies that the test criterion discussed in 3.3.6.4 (exceeding of the human-toxicological recommended value as a result of consuming groundwater as drinking water) is met, with the exception of the highly water-soluble dihydroxybenzenes.

Concerning metals, it is considered advisable to test by comparison with two further data: the target values from the Memorandum Environmental Quality Targets (VROM, 1991) and the data on incidence in the groundwater by the *Landelijk Meetnet Grondwaterkwaliteit* -LM GWK - (Prins, 1991): National Measurement System Groundwater Quality. This comparison has been made in Table 13.

From this Table it becomes clear that the reduction of the groundwater concentration by a factor 10 leads to values below the target values or the 90%-values from the LM GWK for a number of metals. The more stringent of the two calculated groundwater concentrations was applied to these substances.

The C-standard values for groundwater must be such that they can be pointed out by present analytical-chemical techniques. For this reason, the proposals have been tested by comparison with the current detection limits (WAM/CCRX, 1991; Heusinkveld et al., 1991). Table 14 indicates the substances that have been modified for this reason.

4.1.3 Test by comparison with environmental quality objectives

It is advisable to see to it that the new C-standard values for soil and groundwater are not below the levels mentioned in the Memorandum Environmental Quality Targets soil and water (VROM, 1991).

This test was already discussed in 4.1.2 with reference to the groundwater and metals and arsenic and has been taken into account. The proposed C-standard value for groundwater is not lower than the target value for any of the organic substances discussed here.

Another conclusion that may be drawn from the comparison is that the proposed C-soil standard value is not lower than the target value for soil/sediment in any situation.

Table 13. Proposal for C-standard value groundwater for metals and arsenic.
Comparison of calculated groundwater concentrations [$mg.m^{-3}$ = $\mu g.l^{-1}$] with target values and data from the *Landelijk Meetnet Grondwaterkwaliteit* (National Measurement System Groundwater Quality).

substance name	**Cpw**[1]	C-max[2] Table 9	target-value[3]	Land. Meetnet Grondwaterkwal. 50%	90%[4]	**proposal C-value groundwater**
arsenic	41	66	10	2	7.5	40
barium	11000	629	50	73	220	600
cadmium	63	31	1.5	0.11	0.4	6
cobalt	1000	88	20	0.16	0.7	90
chromium	16	472	1	1.09	3.5	15
copper	350	8810	15	0.6	4.1	35
mercury	3	19	0.05	0.04	0.04	0.3
molybdenum	1850	629	5	0.16	1.02	185
nickel	375	3140	15	5	18	40
lead	68	54	15	2.1	5.0	50
tin	1250	126000	10			100
zinc	2880	31400	150	10	59	300

1. concentrations in the pore water in equilibrium with the proposed integrated C-soil standard value
2. maximum tolerable concentration in groundwater used for drinking water (on the basis of the human-toxicological recommended value)
3. target value for groundwater (VROM, 1991)
4. data from the Landelijk Meetnet Grondwaterkwaliteit (Prins, 1991); the concentrations given are concentrations below which 50% and 90% respectively of the measurement points are situated.

The limit values for newly-formed sediment (derived from the limit value for water and the equilibrium partition theory), however, are higher than the proposed C-standard values in the case of arsenic, lead and chromium. It is therefore proposed to raise the C-standard values for these three substances to the level of the limit value for newly-formed sediment referred to above. The C-standard values for groundwater are not adjusted to this level.

4.1.4 The test table

Table 14 gives the final proposals for standard values for (soil and) groundwater at the level above which Clean-up (research) is necessary for the substances from the test table of the Guideline Soil Protection.

Table 14. The test table.
Proposals for C-standard values for soil [mg.kg^{-1}] and groundwater [mg.m^{-3} = μg.l^{-1}] for a standard soil (10% organic substance, 25% lutum).

Substance name	C-value soil [mg.kg^{-1}]	C-value groundwater [mg.m^{-3}]
I Metals		
arsenic	55	40
barium	650	600
cadmium	12	6
chromium (III)	380	15
cobalt	120	90
copper	190	35
mercury	10	0.3
lead	530	50
molybdenum	40	185
nickel	210	40
zinc	720	300
II Inorganic compounds		
cyanides free	12	8000
cyanides complex	12	8000
thiocyanates	2	1500
sulfides	0.2	140
III Aromatic compounds		
benzene	50	1300
ethylbenzene	150	440
phenol	40	3800
cresol	5	160
toluene	150	1200
xylene	75	190
catechol	30	5600
resorcinol	14	2600
hydroquinol	12	2600
IV Polycyclic Aromatic Hydrocarbons		
anthracene	40	5
benzo(a)anthracene	40	0.4
benzo(k)fluoranthene	40	0.02
benzo(a)pyrene	40	0.03
chrysene	40	0.04

phenanthrene	40	5
fluoranthene	40	0.8
indene(1,2,3cd)pyrene	40	0.01[dl]
benzo(ghi)perylene	40	0.03
napthalene	40	75

dl : adjusted in connection with the detection limit

Table 14. The test table
(ctnd) Proposals for C-standard values for soil [mg.kg^{-1}] and groundwater [mg.m^{-3} = µg.l^{-1}] for a standard soil (10% organic substance, 25% lutum).

Substance name	C-value soil [mg.kg^{-1}]	C-value groundwater [mg.m^{-3}]
<u>V chlorinated hydrocarbons</u>		
aliphatic hydrocarbons		
1,2-dichloroethane	3.5	330
dichloromethane	40	1800
tetrachloromethane	2	20
tetrachloroethene	14	14
trichloromethane	25	1000
trichloroethene	60	470
vinyl chloride	0.1	0.6
chlorobenzenes		
monochlorobenzene	30	190
dichlorobenzenes (ind.)	30	50
trichlorobenzenes (ind.)	10	4
tetrachlorobenzenes (ind.)	17	1.5
pentachlorobenzene	25	6
hexachlorobenzene	25	5
chlorophenols		
monochloropnenols (ind.)	10	160
dichlorophenols (ind.)	10	30
trichlorophenols (ind.)	10	6
tetrachlorophenols (ind.)	10	20
pentachlorophenol	5	0.15
chloropolycyclic aromatic hydrocarbons		
chloronaphthalene	5	3
polychlorobiphenyles		
Σ polychlorobiphenyles	10	0.03

Table 14. The test table
(ctnd) Proposals for C-standard values for soil [mg.kg^{-1}] and groundwater [mg.m^{-3} = μg.l^{-1}] for a standard soil (10% organic substance, 25% lutum).

Substance name	C-value soil [mg.kg^{-1}]	C-value groundwater [mg.m^{-3}]
<u>VI Pesticides</u>		
organo-chlorine pesticides		
DDT	4	0.01
DDE	4	0.03
aldrin	0.4	0.005^{dl}
dieldrin	4	0.07
endrin	4	0.2
α-HCH	2	1.5
β-HCH	0.6	0.5
γ-HCH	2	1.5
δ-HCH	2	1.5
other pesticides		
carbaryl	5	0.09
carbofuran	1.5	0.07
propoxur	650	80
maneb	30	0.1^{dl}
atrazine	6	150
<u>VII Other contaminants</u>		
heptane	10	5
octane	7	1
cyclohexanone	770	40000
butylbenzylphtalic acid	60	9
di(2-ethylhexyl)phtalic acid	60	2
pyridine	4	10
styrene	102	340
tetrahydrofuran	1	3
tetrahydrothiophene	5	2

dl : adjusted in connection with the detection limit

4.2 Correction for soil-type

The proposed C-standard values for soil may be considered soil-type-dependent within the preliminary conditions for the derivation method (in this context there are differences between human exposure and the exposure of soil-organisms; the ecochemical behaviour of substances is another factor). The following starting-points have been used för the different contaminant categories.
The C-soil standard values for metals may vary as a function of the soil parameters organic substance content and lutum content. The dependency on these soil parameters has not yet been explicitly quantified.
The toxicity of a substance is partly dependent on a number of physico-chemical soil factors, while assuming that there is a relation with the availability of the substance to the organism. The physico-chemical soil factors may vary strongly in the experiments based on which the toxicity data have been determined. This obstructs good comparability. For this reason, a correction for soil-type has already been performed in order to normalize the body of toxicity data. Denneman and Van Gestel (1990) show that the correction for soil-type leads to a significant improvement of the comparability of toxicity data. The soil-type correction indicated for the soil reference values in the Environmental Programme Progress Report 1988-1991 (VROM, 1987) has been used. Although this soil-type correction entails uncertainties, this method must be considered as the best available method for the present moment. This method can only be improved by specific research in this field. Various TCB-reports (1989a,b,c, 1990a,b) discuss the calculation method used for correction for the availability of substances. It has been pointed out that the availability of substances need not be the same for all organisms and that availability depends on other factors beside the amount of organic substance and lutum in the soil. TCB likewise recommends aiming for incorporating as many determining factors as possible, as well as collecting additional data.
It is proposed in this report to apply the soil-type correction for metals and arsenic, as derived in diagram 1 (on the basis of the soil reference values) and given in Table 7, to the C-soil standard values.

For the inorganic compounds, no dependency on any soil parameter can be pointed out, given the present calculations.

The C-soil standard value for organic substances has been made dependent on the organic substance content. In this case, too, the correction factors as formulated for the soil reference values (VROM, 1987; see also Denneman and Van Gestel, 1990) have been taken as starting-point for the time being.

It is stated that the C-standard value for groundwater does not depend on physico-chemical properties of the soil for any of the substances considered. This is related to the availability concept. It is assumed that the availability of a substance, or in other words the concentration in the aqueous phase, decreases with the increase in the lutum content and/or organic substance content. This corresponds with the proposition that the effect a substance has is determined by the concentration of that substance in the aqueous phase.
This holds both for soil organisms and man. Human exposure to most organic substances primarily takes place via crop consumption and inhalation and is therfore actually determined by the concentration in the pore water and the soil air, which is related to the pore water concentration through the Henry-constant.
The concentration in the groundwater at which effects pertaining to "serious hazard to man or the environment" occurs, will therfore likewise be independent of the physico-chemical properties of the soil.

4.3 Simultaneous exposure to different contaminations

In the case of a simultaneous exposure to different contaminants, a "serious hazard to public health or the environment" may arise as a result of this combined exposure, even if the individual contamination in itself does not cause concentration levels exceeding the C-standard value. This reasoning is based on the assumption that each contamination to some extent adds to the adverse effects on the organism exposed.

The combined impact of the toxic effect of different contaminants has only been studied to a very limited extent. The fact that numerous combinations of contaminants may occur, constitutes a complicating factor. Moreover, one may deal with substances with similar or different effect mechanisms and proporties. For substances with a similar effect mechanism, an additive effect may be assumed for both man and other organisms. For substances with different effect mechanisms, different types of effects will operate at the same time. Denneman and Van Gestel (1990) have concincingly argued that different types of effects have a cumulative impact on (soil) organisms, as each effect causes the vitality of that particular organism to be impaired. TCB (1990a) also considers simultaneous exposure to more than one substance. The TCB report concludes that as an effect parameter relates to a higher biological level, the sensitivity of this parameter to specific substances decreases, while the cumulative effect of substances increases.

In the following paragraphs, proposals will be outlined towards an approach of combined exposure in the light of the issue at hand (the definition of the test criterion "serious hazard to public health or the environment"). No distinction is made with reference to the exposed organism (man or other organisms). Substances are divided into substance groups with comparable physico-chemical properties. Although the homogeneity concerning the toxicological effects of the different substance groups is open to discussion, a distinction has primarily been made between contaminants belonging to or not belonging to a particular substance group. Examples of substance groups for organic compounds are: PAHs, chlorophenols, chlorobenzenes, chlorinated aliphatic hydrocarbons, aromatics and hydroxy aromatics. The metals included in the test table may be classed with one substance group, as the exposure of soil organisms determines the derivation of C-soil standard values.

It is assumed that complete addivity may occur within a substance group, whereas only partial addivity is assumed for the incidence of substances from different substance groups.

It is furthermore assumed that (in accordance with the approach by Denneman and Van Gestel, 1990) a substance can only cause a vitality decrease when it occurs in a concentration of at least the A- or reference value. At lower concentration levels, it is assumed that no vitality decrease occurs. A correction is therefore made for these concentration levels. In comparing measured concentration levels of contaminants with the C-standard values, identical organic substance contents must of course be taken as starting-point (if necessary, the C-standard values should first be corrected before it is substituted in the comparison).

Experts have called for caution in using the approach described above for metals and inorganic compounds. A proposal has now been defined, which still assumes complete addivity, but does not consider concentrations below [A+C]/2. It is supposed that the vitality decrease operating below this value is very limited and that the importance of concentrations just below the C-standard values is emphasized by the non-linear dose-response curve (in the range between 5 and 50% of the species which are affected).

Table 15 gives an overview of the formulas to be used. These formulas are based on the assumption that there is a linear relation between the concentration of a substance and the effect caused by that substance. This assumption must be considered a broad generalization; it is not a proven relation valid for all substances. Some substances will remain under the risk threshold, whereas others will exceed it. For most substances, the dose-effect relations are unknown.

It is assumed that as the result of the summation increases, the risks resulting from the combined exposure increase as well. An absolute quantification cannot be given as yet. It has to be noted, moreover, that the C-standard values determined for the individual substances are characterized by a varying degree of reliability, depending on the available data. This approach which has been based on theoretical grounds, has not been validated in practice.

As the reliability of the sum formulas is unclear in quantitative terms, application is open to discussion. Nevertheless, based on the additive effect of many contaminants, of which the individual concentrations do not exceed the C-standard values, it can be argued that further research and clean-up measures are legitimate in the context of present legislative procedure.

4.4 Additional remarks

Cyanides

In view of the lack of differentiating data, Vermeire (verbal statement) suggested to consider the human-toxicological C-standard values for complex and free cyanide equivalent and to take the average of the individual values presented by Vermeire et al. (1991).

Chromium

In proposing human-toxicological C-standard values, a distinction has been made between chromium (III) and chromium (VI) in connection with the toxic properties. Concerning the oral uptake, the main exposure route, it is a known fact that chromium (VI) is partially converted into chromium (III) in the stomach and that in urine only chromium (III) is found. Moreover, chromium (VI) is not a generally found speciation in the (Dutch) soil, as chromium (VI) is reduced to chromium (III) relatively fast in the soil (Slooff et al., 1989).

As for the toxicity affecting soil organisms, chromium (VI), moreover, is more toxic than chromium (III), by approximately a factor 10 (Molnar, 1989). However, this study found that after 4 weeks, the difference in toxicity between the two species of chromium can no longer be shown. This is probably caused by the reduction of chromium (VI) in the soil.

It is proposed that only chromium (III) be included in the test table for use in the general soil quality assessment. If there are indications for a chromium (VI)-contamination (for instance in tanning works), this contamination must be given specific attention.

Tin

No ecotoxicological data are available for a risk-assessment in the case of tin. Moreover, tin only becomes dangerous for man in relatively high doses, as becomes clear from the calculations (see Table 2; Van den Berg, 1991). For this reason it is poposed to exclude tin from the test table. An exception should be made for the organo-tin compounds, which are mainly used as pesticides. The toxicological data and exposure for this group of compounds will have to be evaluated in the short term, so that they can be included in the test table under the category pesticides.

Polychlorobiphenyles

In connection with the fact that the toxicity data for both man and the environment have been derived from Aroclor-mixtures (mixtures of a number of congeners), it is proposed to use one figure for polychlorobiphenyles for the C-standard value. This figure represents the sum of the polychlorobiphenyles as no data have been derived for individual congeners. The human exposure was based on the behaviour of a tri- and hexachlorobiphenyl congener, which were considered representative of the range in the average composition of the Aroclor-mixtures. The geometric average of the two calculated values for the individual congeners has been proposed as the C-soil standard value.

Table 15. Formulas for the calculation of combination toxicity.

For **organic substances** which occur in combination:

a. within a group, complete addivity;

that is to say $\Sigma\ (i=1..n)\ \frac{\text{conc. } i}{C_i - A_i} \geq 1$ means: "serious hazard"

b. outside a group, partial addivity;

that is to say $\Sigma\ (i=1..n)\ \frac{\text{conc. } i}{C_i - A_i} \geq \sqrt{n}$ means: "serious hazard"

For **metals** and **inorganic compounds**

$$\Sigma\ (i=1..n)\ \frac{\text{conc. } i - [A_i+C_i]/2}{[C_i-A_i]/2} \geq 1$$ means "serious hazard"

or:

$$\Sigma\ (i=1..n)\ \frac{2^*\text{conc. } i - A_i-C_i}{[C_i-A_i]} \geq 1$$ means "serious hazard"

If there is a case of a **combination of organic substances** and **metals**, the sum of the partial toxicities for these two groups should be determined first; after summation it should subsequently be decided if there is "serious hazard" according to the partial addivity.

5 ACTUAL EXPOSURE TO SOIL CONTAMINATION

5.1 Introduction

In case of "serious hazard to public health or the environment", as described in Chapter 2, and follow-up researches and clean-up measures may be taken under the Provisional Soil Clean-up Act, the actual exposure to soil contamination and the corresponding risk of damage to human health and the environment is assessed in the second phase of the Further Research, based on nature and extent of the contamination, the local contamination situation and soil-use.
When the essential test criterion defined in the Provisional Act (in due time, the clean-up section of the Soil Protection Act) has been met, in fact the first step in the determination of the urgency, this must be followed by a more specific analysis of the urgency of research and clean-up in the second phase. If the "serious hazard" criterion indicates an intolerable risk for man or the environment, assuming that all potential exposure routes are operational, this by no means automatically implies an acute threat to the public health or the environment (see Guideline Soil Protection).

The scale of the soil contamination problem in the Netherlands (see the Government stance on the advisory report by the Steering Group Ten-Year Scenario Soil Clean-up; VROM, 1990d) warrants different categories of priorities. The following sets can be distinguished:
- prevention→inventory→protection→clean-up;
- regulation→logistics→organization→time→money;
- polluter→consumer→public authorities.

In accordance with the Steering Group advice, social interests outstripping environmental protection should be avoided. Both priority determination and implementation of research and clean-up require careful proceeding.

5.2 Analysis of the extent of exposure and related risks

The assessment of actual exposure to soil contamination differs from that of potential exposure. For the actual human expposure, there are two important factors:
- in contrast to the assessment of potential exposure, it is no longer assumed that all routes might be operational;
- the specific local soil conditions are taken as starting-point.

  With reference to the exposure of soil organisms, only the latter aspect applies, as there are no alternative exposure routes in this case.

In section 2.3, eight steps have been mentioned, which are used in the analysis of exposure to soil contamination and related risks in the second phase of the Further Research. In the following paragraphs these eight steps will be discussed in greater detail.

Step a: determination exposure routes (human exposure)

Linders (1990) among others inventorized the direct and indirect routes via which man may be exposed to (the consequences of) soil contaminants. After the contaminants have been distributed over the soil phases and different transfer processes have taken place (transport to the soil surface, ground- and surface water, uptake by crops, fish and cattle, permeation and drinking water collection), direct or indirect exposure may occur.

Depending on the nature and extent of the contamination, the local contamination situation and soil-use, the number of relevant (human) exposure routes may be determined. Van den Berg (1991) discusses both the ecochemical properties of substances (see also 3.3.4) and the transfer processes. The number of relevant exposure routes may be determined on the basis of this information.

Step b: **determination of the relevant exposure routes, assessment of the relative exposure contribution per route**

In order to determine the contribution via the routes defined in step a, the individual contribution to the total exposure may be determined on the basis of rough, followed by more elaborate estimates. In this way the final number of relevant exposure routes can be determined.
Van den Berg (1991) indicated the relative contribution (in percentages) for the substances from the (new) test table, assuming that the exposure routes as included in the CSOIL model and the assumptions on which the model is based (for instance with reference to crop consumption) are valid for a particular site.
Annex 1 gives the formula sets for all direct and indirect exposure routes.

In order to calculate the human exposure, the normative concentration in soil and groundwater must be determined for each route.
In addition to the measurement data gathered in assessing the nature and content of the contamination and depending on the local situation, it may be necessary to collect additional information concerning the contents of contaminants in the soil and the groundwater in an area that is more representative of the specific exposure route. A more accurate estimation of the normative concentration in soil or groundwater (for instance as a preparation of step c) may be acquired by collecting additional information focused for instance on the range of action of a vulnerable target group (children for the ingestion exposure route) or additional sampling in allotment gardens (crop consumption exposure route). Recent publications emphasize the importance of reliable and exposure-representative data as input for exposure calculations (TCB, 1990c and 1991c; Leenaers et al., 1991).
The concentrations of contaminants in soil and groundwater that are to be filled in in the formulas are average values (since the C-standard values should be considered average values as well). The average of the measurement data can be determined per physical unit (if necessary in relation to the surface area). Moreover, a normative soil depth is selected for each exposure route. For the short-term determination of the urgency in the case of soil ingestion, the data on the soil layer from 0-20 cm minus the ground level provide the most relevant information. Depending on the current soil-use, the upper 50-100 cm of the soil profile may, however, be affected by this form of exposure. This should therefore be taken into account in determining the urgency in the longer run. For crop consumption the concentration of contaminants in the soil layer from 0-75 cm minus the ground level must be considered; for consumption of contaminated drinking water as a result of permeation, this is the soil layer reaching to a depth of 50 cm. below the drinking water pipe; for inhalation of indoor and creeping-space air, the depth of the boundary surface water - air must be considered. With the help of the formulas presented by Kliest (1990), the concentration level of contaminants in the indoor- or creeping-space air may be estimated, based on local data on soil and hydrogeology and the derived normative concentrations in soil and groundwater.
In the case of private water collection (see also 3.3.6.4), the use of surface water and prevention of contamination in relatively undiluted form (floating layers), specific sampling and analysis are indispensable.
By using the formulas given in annex 1, it is possible to determine which of the exposure

routes selected in step a. must be considered relevant in quantitative terms.
If exposure as a result of inhalation of (indoor) air, consumption of drinking water, crops, fish and other animal products, is considered relevant given the calculations performed in this step, or if the uncertainties with reference to the estimates are too high, a supplementary measurement programme is required. In view of the limited validation of the calculations/models, measurements will be required without exception.

Step c: supplementary measurements in other media than soil and groundwater

Kliest (1990) gives indications for circumstances under which an air research in the living- and creeping- space can be conducted, as well as guidelines for the implementation of such a research and the evaluation of the measurement data acquired.
In addition to the reasons mentioned in step b., a crop research may be considered on the basis of knowledge of crop-specific uptake, the growing rate of the crop, the consumption pattern of the local population, or the acidity of the soil. In this context, the allotment and vegetable gardens and the local possible contamination of vegetables, potatoes and fruit, are particularly relevant. Cereals will not be liable to this situation, as there is no local consumption. This contamination may be the result of soil contamination (uptake via the liquid and gaseous phase; contact with near-by contaminated soil) and air pollution (dry and wet deposition; particularly for lead and PAHs). Performing additional measurements and taking measures should be preceded by verification of the cause of the crop contamination. Van den Berg (1991) gives an overview of the importance of uptake by crops for common contaminants. These data have been derived partly from research by Sauerbeck (1988), who also gives data with reference to accumulation in different crop types.
High contents of heavy metals and organic contaminants in the soil may have a phytotoxic effect on the crop.
Phytotoxic effects, however, may also be caused by air pollution and the application of pesticides. "Sound agricultural practice" is not always found in allotment and vegetable gardens. The acidity of the soil may be (too) low, which increases the uptake of heavy metals by crops. Owners of allotment and vegetable gardens may consume relatively much privately-grown food. The burdening by certain substances from the food parcel may therefore deviate from that of the average Duchman. Cases in which the signal values formulated by the Agricultural Advisory Body Environmentally Critical Substances (LNV, 1986) are exceeded, as well as crop norms specified in the context of the *Warenwet* (Pure Food Act) (Staatsblad, 1985) and Pesticides Act (Staatscourant, 1984), may lead to crop analyses being conducted.
In the interest of carrying out research into crop contamination, the regulations to the Pure Food Act give an extensive description of sampling, sample treatment and analysis methods.
Research into the occurrence of contaminants in fish, meat- and dairy products should preferably be conducted in cooperation with the regional inspector of public health and environmental protection.

Permeation of plastic water pipes by soil-contaminants has been discussed by Van den Berg (1991) and section 3.3.6.3 in this report. Based on the formulas presented by Van den Berg, it can be estimated at which concentrations in soil or groundwater, relevant substances (aromatics, chlorinated hydrocarbons, oil, phenols) cause permeation (depending on the pipe material) to such an extent that drinking water norms are exceeded. The regional inspector or the drinking water company may require additional sampling of the drinking water in this case, in view of the maximum tolerable

concentration in drinking water.
The drinking water research should be conducted by or in consultation with the the relevant drinking water company.

Based on the measurements referred to above, mean concentration levels in different media may be determined for each application area (drinking water, crops, animal food products, air in houses) and directly applied to exposure calculations.

Step d: **calculation of the present human exposure to soil contamination**

With the help of the concentrations determined in the previous steps and the formulas presented, the total human exposure may be calculated. Particularly in the case of indirect exposure, the formulas may be deviated from on the basis of additional information. The exposure can be calculated on the basis of the total amount of the product consumed and the concentrations measured in it.

Based on the calculations of the dispersion of the contamination (see also Chapter 2; Lagas et al. (1990) and Kliest (1990), an indication can be given in how far exposure of objects (for instance houses, drinking water pipes, consumption crops) in the vicinity of the contamination should be allowed for as a result of dispersion of the contamination.

Step e: **assessment of the risk of damage to human health (comparison with toxicological recommended values)**

In this step, the total human exposure is compared to the toxicological recommended values presented by Vermeire et al. (1991). In this step the human exposure must moreover be compared to the (daily) exposure in a non-contaminated situation (background exposure), for which Vermeire et al. (1991) have already given estimations based on literature research. Additional data focused on a particular site are collected.

Step f: **assessment of the effects of soil contamination on soil-organisms**

In the case of contaminants for which relatively many toxicological data are available (all substances for which test procedures 1a, 1b, 1c and 2 (4 data) have been used; see Denneman and Van Gestel (1990)) based on the Risk Analysis Soil Contamination method, the percentage of the species adversely affected can be assessed, given the concentration found. The formulas in Table 16 can be used for this purpose (according to Van Straalen and Denneman, 1989).
The concentrations to be substituted in these formulas must have been corrected for the values for a standard soil. In view of the fact that laboratory data (NOEC-values) serve as the input for these formulas, the formulas are more suitable to comparing contamination situations than to indicating risks in absolute terms.
The formulas have an added value (in relation to the estimation of the potential exposure) particularly in estimating the actual exposure to soil contamination in cases of contamination by different substances (see also TCB, 1990a for the estimation of the percentage of species affected by simultaneously applied pesticides). In Table 16 the Aldenberg and Slob improved correction for assessment factors has been used, depending on the number of test species (d_m, orginally formulated by Kooijman, 1987). The Aldenberg and Slob d_m-Table cannot be used directly for the present purposes. For this reason some modifications have been made to this report. The estimation has little significance in absolute terms. As has already been explained above, the difference

between actual and potential exposure is mainly determined by the application of data concerning the local soil conditions, which in turn determine the availability of contaminants for soil organisms. The combination of substance properties, properties of the soil environment and properties of the organisms determine the bioavailability.
The TCB report on ecological risks of a heightened availability of cadmium and lead by soil acidification (TCB, 1991b), addresses this issue. There is no protocol as yet for the standardized description of actual exposure (and the related effects).

In relevant cases, use can be made of bioassays, in which the exposure of organisms to a contaminated soil is studied under laboratory conditions. Earth worms and plants are already available as test organisms for terrestrial soils. For a discussion of bioassay applications, the reader is referred to Adema et al. (1987) and Kamerman and Van Gestel (1991). Chironomid larvae constitute relevant test organisms for aqueous soils. For a discussion of bioassays in aqueous soils, the reader is referred to Maas-Diepeveen and Van der Guchte (1990). Finally, the microtox test might be used for the assessment of both terrestrial and aqueous soils, as proposed by Slooff and De Zwart (1991).
The Health Council published a number of reports on Ecological Norms for Water Management. One of these reports discusses parameters that may play a role in ecological standardization (Health Council, 1989). The Health Council is currently preparing a similar report on terrestrial environments. The parameters may also be useful in assessing actual risks of contaminations. On publication of the Health Council report, its relevance should be further studied.

In using bioassays for the purpose of indicating the actual risks of exposure to soil contamination for soil-organisms, the following observations should be made:

- soil samples must be relevant to the organisms with which the bioassay is conducted. For the organisms studied, the upper soil-layers are relevant; the tests, then, should involve these layers (mostly the top 20 cm.);
- toxicity data used towards proposing C-standard values, are based on mean concentrations. Correspondingly, bioassays should also be conducted with mean concentrations. This may be done by means of mixed samples. In addition, samples may be collected for higher concentration levels and for the (local) background concentration. The results of the bioassays at these concentration levels may be used in step h.
- the nature and extent of the contamination situation has become clear from the results of the first phase of the Further Research. Bioassays give an indication of the combined effect of the substances present, so that a comparison can be made between the individual sites. A quantitative, absolute evaluation of the results of bioassays, particularly in relation to the present purpose, cannot be given as yet.

In addition to bioassays, as well as for use in step h, it may be important to conduct an (quantitative and qualitative) inventory of species involved in that particular case of soil contamination. The inventory should then be compared to the incidence of species in non-contaminated situations (to be derived from literature research or databases of specialized institutions). Aiming for the highest possible degree of correspondence between the sites to be compared, is a prerequisite for this comparison (apart from the contamination).
In conducting the inventory, it should be noted that it is virtually impossible and not very useful to inventorize all species occurring. Choices should be made; the following observations may serve as a guideline:

1. groups of species of which the incidence in contamination situations is known, must be emphasized. Nematodes are an example of this. A method has been

developed, which relates the composition of the nematode population to the disturbance of the balance of the soil (Bongers, 1990). Although this method was orginally devised as a general disturbance index, it also seems to be possible to use it in order to gain an understanding of the nature and extent of the contamination;

2. groups performing an essential function in the ecosystem (for instance the primary decomposition of organic material) should be emphasized as well. Particularly if such a group is (virutally) absent, the actual risks for soil organisms will be considerable.

Listing species that are characteristic of an ecosystem and inventorizing on the basis of this list in analogy to the AMOEBA-approach developed for aquatic systems, also seems viable for terrestrial ecosystems and has already been worked out for two pilot areas for vegetation and higher organisms (Latour and Groen, 1991). In the short term, however, this approach cannot be used for soil organisms, as insufficient data are available at the present moment.

Table 16. Formulas for the assessment of risks of exposure to soil contamination for soil organisms.

$$\delta = \left\{ 1 + \exp \frac{(\pi^2(X_m - \ln\ C))}{3.d_m.S_m} \right\}^{-1}$$

in which:

δ = fraction of the species at which adverse effects may operate (δ * 100) indicates the percentage of species for which NOEC is exceeded)

X_m = mean ln NOEC

d_m = correction for estimation errors depending on the number of test species (see also Table below)

s_m = standard deviation of the ln NOEC

C = measured concentration in the contaminated soil

Correction factor (d_m) for the possibility of estimation errors (s) in relation to the number of test organisms (m). (according to Aldenberg and Slob, 1991)

	d_m for	
m	s=5%	s=50%
3	9.09	2.29
4	6.13	2.15
5	4.99	2.07
6	4.39	2.02
7	4.01	1.99
8	3.77	1.97
9	3.56	1.96
10	3.42	1.93
20	2.78	1.88
infinite	1.81	1.81

Step g: estimation of future dispersion of contamination (model calculations)

Based on calculations by means of models considered most relevant to the local contamination situation, an estimation can be made of the dispersion(velocity) of the contamination.

Lagas et al. (1990) envisage the contamination of the soil as a reservoir containing the amount of contamination that might become available for dispersion in the course of time. In their view, the urgency of measures depends on the size of the reservoir and the speed at which it is emptied. As a first step, the approach assumes complete dispersion of the contamination via the groundwater (the so-called groundwater contamination potential). Sites contaminated by different contaminations, may be readily compared in this way, too. When a contamination is immobile, the comparison of sites may be made by means of the groundwater contamination potential.

Step h: interpretation of present and future exposure for man and the environment

Based on the previous steps, the total human exposure has been determined and an estimation has been made of the adverse effects to which soil organisms are subjected as a result of the presence of soil contamination at a certain site.

In interpreting this exposure for man and the environment in the interest of considerations concerning measures, the site-related background exposure and the possibility of simultaneous exposure to different contaminants (see 4.3) should be taken into consideration as well. Vermeire et al. (1991) includes data (in as far as they are available) on the daily exposure in non-contaminated situtations for all substances considered in the so-called substance profiles. Local point sources for extra exposure have not been considered, nor a possible extra exposure caused by smoking, drinking, or occupational exposure. It has not been the intention to quantify the total health risk.

5.3 Other systems for the determination of actual risks of exposure to and dispersion of soil contamination in order to set priorities

In the following paragraphs recently published systems for the determination of actual risks of exposure to and dispersion of soil contamination will be briefly discussed in the context of setting priorities for conducting research and taking measures. It is important to discuss these systems as they are aimed at estimating actual risks of soil contamination in a structured and standardized manner.
In view of the purpose of the present report, a methodological comparison is in place. Moreover, a number of recommendations are made.

5.3.1 Final report Commission for Soil Clean-up of industrial estates in use

In her final report, the Commission for Soil Clean-up of Industrial Estates in Use (BSB, 1991), deals with the issue of setting priorities in the approach of soil clean-up under direct management of industrial estates in use. A distinction is made between different phases, in which priorities concerning research and measures are determined on the basis of increasingly accurate data.
Particularly in the first phases (PR 1 up to PR 3), in which data are used collected before the Further Research has been conducted, priorities are set rather schematically. Further refinement and modification of the categorization of substance groups in the light of toxicity data is to be recommended. Using Vermeire et al. (1991), modifications can be made relatively simply. The division of substance groups into categories based on the mobility of substances should be further analyzed as well. The mobility assessment of substances as described in Lagas et al. (1990) and the background literature referred to in the latter report, as well as in this report, may be helpful in this context. Based on the data of the Further Research (phase PR-4) and following the determination of whether or not there is "serious hazard to public health or the environment", an assessment is made of the actual exposure and the risk of dispersion of the contamination, as discussed in Chapter 2 and in this chapter. The BSB Commission recommends that these two aspects be judged jointly. Van den Berg (1991) reported on the extensive and detailed study of the HESP (ECETOC/Veerkamp) and SOILRISK (Ten Berge, 1990) models, in addition to RIVM (Linders, 1990) and accompanying background literature. After carful consideration of the correspondences and differences between the models mentioned, the CSOIL model was introduced. In view of the fact that both the public authorities and the industrial sectors involved require uniformity of the assessment of cases of soil contaminiation, and given the coordination that has taken place with referrence to the modelling, it is assumed that the CSOIL model will be used in the context of voluntary clean-up operations of the said industrial estates.
The final assessment of both the actual exposure risk (for man and soil organisms) and the risk of dispersion is expressed in scores. The definition of these scores in relation to the chosen terminology (which to some extent differs from the conceptual framework used in the Guideline Soil Protection) requires clarification. This can be done partly on the basis of knowledge and experience gained in field situations and the information in this report. It is recommended to be very precise and specific in setting priorities,

which is possible on the basis of the results of the Further Research.

5.3.2 Provisional inspection guideline exposure risks in case of soil contamination

The (Chief) Inspectorate for Public Health and Environmental Protection has issued a working document (VROM, 1989b) consisting of a number of guidelines allowing a more accurate assessment of the actual exposure risks for man to cases of soil contamination in actual practice. The main exposure routes (see 3.3.4) have been considered in the guidelines. The risk of dispersion has also been taken into account.
According to the authors of the working document, the priority (sequence) of different cases of soil contamination and the need of (timely) measures may thus be better assessed.

The working document contains useful practical information. The provisional inspection guideline analyses the exposure and the related risk per exposure route and tests this risk in comparison with an evaluation framework focused on particular routes. These evaluation frameworks have not been sufficiently coordinated. According to the authors of the working document, inconsistencies arise as a consequence of differences between the various normative evaluation frameworks used and differences in the availability of data per exposure route. Another important disadvantage of the document is the fact that it does not allow an indication of the total exposure which could then be compared to a general toxicologically-based evaluation framework.
The assessment of the exposure with reference to the individual routes has been both implicitly and explicitly discussed in chapters 3 and 4 and in the reports 725201002, 725201005 and particularly 725201006. In report 725201004, Kliest (1990) added considerably to the description of the backgrounds of the calculations and the aim and interpretation of measurements in the context of the assessment of risks for man as a result of exposure to soil contamination via air.

The working document was advantageous on the other hand in the sense that the data contained in it have been widely used, thus allowing improvements and additions to the models to be made.
By means of the present report, the assessment of the exposure to soil contamination per exposure route has been improved allowing an assessment of the total exposure. It should be emphasized here that although this report makes for an indication of the total exposure, it does not allow determination of the total health risk. Moreover, it should be said that although the introduction of the CSOIL model meant an important step forward compared to other proposals, one should still be careful in using it. The authors of the present report and previous reports as well, have repeatedly warned against the uncertainties in relation to the calculations and the assumptions on which the assessments are based.

6 GENERAL RECOMMENDATIONS

In the previous chapters, the toxicological basis of the intervention value for the different substances from the test table has been presented. In this context, uncertainties have been pointed out and proposals made for the required adjustments to the procedure. In the following paragraphs, general recommendations are given in brief outline, mainly focused on improving the methods for the assessment of exposure to soil contamination and the risk of damage to human health and the environment. The implementation of the improved methods should be given priority.

Stepping up the coordination with international developments (for instance in the US, Canada and Germany) in the field of risk assessment (in cases of soil contamination) is recommended. This will not only promote (international) standardization of the methods used, it will also lead to an increase in the availablitity of data.

Both in relation to the ecotoxicological and human toxicological basis of the intervention value, a lack of reliable data is felt. Research focused on filling up the gaps in toxicological basis information (particularly data on the chronic toxicity of substances (NOEC)) should be stepped up and conducted on a structural basis. In the meantime, extrapolation from other data (on the acute toxicity of a substance, for instance E(L)C50), is required. However, no standardized extrapolation procedures are available at present.
There is a shortage of data particularly on the toxicity with reference to soil organisms. Moreover, the number of test species is limited. Test species for laboratory expermiments should be selected in such a way that they are representative of the species found in actual (field) conditions. Both the number and the diversity of the test species will therefore have to be increased. The statistical consequences of the selection of the sensitivity of (soil)organisms (related to the choice of test species) should be evaluated and where necessary lead to methodological adjustments. In view of the issue of exposure of higher organisms to soil contamination as a result of continued poisoning and the risk of damage to the health of these organisms, toxicity data for higher organisms, as well as an extrapolation method for the quantification of risks to higher organisms are required.

The simultaneous exposure of organisms to contaminants should be given explicit attention in the light of the ecotoxicological and the human-toxicological basis of reference values. A distinction must be made between the toxicity of substances with similar and dissimilar effect mechanisms. Combination toxicity of mixed substances constitutes an area on which hardly any research has been done as yet. The risk of exposure of an organism to contamination should be taken into account on a more systematic basis.
Finally, methods must be developed for the prognosis of bioavailability. Uptake of substances by man or (soil) organisms and the related effects cannot be predicted merely on the basis of the total concentration of a substance in the soil. In view of the frequent exposure of man to substances like lead and polycyclic aromatic hydrocarbons and experimental possibilities for the relevant substances, further study into bioavailablity is definitely required.

The description of the ecochemical behaviour of substances is highly incomplete. This has unwanted consequences for the description of human exposure to contaminants in the soil, particularly in the case of metals and other inorganic compounds. Research should be started in the short term into the relations between contents of metals (where possible species-related) in soil and pore water, and subsequently related to acidity, organic substance-, lutum and CEC-content. Partly as a result of the defective description of the ecochemical behaviour of contaminants, there is still a great deal of uncertainty, particularly concerning the exposure routes most critical to man. Testing and validation of the CSOIL model must take place. This should preferably be done in field situations. Contradictory results from field situations (as found for instance in permeation of drinking water pipes) should lead to intensification of measurements and testing in comparison with model calculations.

The representativeness of the ecotoxicity data should be validated as well. It must be determined whether similar effects operate in laboratory and field sitations, given the same concentration of a toxic substance. The validation issue has been mistakenly neglected so far.
The toxicity of a substance partly depends on a number of physico-chemical soil factors, while a relation with the availability of the substance to an organism is assumed. These physico-chemical soil factors vary both in laboratory experiments (for instance those in which the toxicity of a substance is determined) and in field situations. This obstructs the comparability of data. Soil type correction will lead to a substantial improvement of the comparability of toxicity data. The soil type correction used in this report constitutes an interim solution. Further research is required in order to develop a soil type correction specifically designed for intervention values. Attention should also be given to the incorporation of other determining factors (like acidity).

Concerning the (toxicological) basis of the intervention values, choices have been made in this report, based on presently available data. In this respect, the values may be considered to reflect the present scientific
possibilities. However, the scientific basis of the methodology should be further improved. In the future, this might lead to an adjustment of the presently proposed intervention values, provided that sufficient knowledge has been gained.

The number of substances in the test table must be increased. Apart from scientific development and improvement of the risk assessment methods, a statistical analysis of input data and the results of this method is required. Optimizing the risk assessment method is impossible without this analysis. Moreover, such an analysis also supports policies carried out in the framework of the intervention value.

Coordinating the development of intervention values and the development of target and limit values is a matter requiring contant attention. If this coordination does not lead to uniformity of methods, the differences between the approaches chosen should be made explicit.

In this report the design of the Further Research has been structured and the reduction of potential and actual exposure has been placed on the map. In order to be able to assess the actual exposure for man and soil organisms and the related effects, additional protocols are required. The methods formulated in this report may serve as a basis.

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Glossary

actual exposure: site-specific exposure as it actually operates given the present soil use

ADI Acceptable Daily Intake; the amount of a substance, expressed in terms of body weight for oral exposure and in terms of air volume for inhalatory exposure, that can be taken up by man on a daily basis during his entire life without causing substantial risk of adverse effects to health.

bioconcentration factor (BCF): Relation between the concentrations of a compound in one compartment compared to another- in this report generally content in crop as compared to content in soil or pore water

exposure route: the route via which humans (or animals) may come into contact with a substance: by ingestion (oral), breathing (inhalatory) and/or via the skin (dermal)

C-standard value: soil-quality criterion; the content of a contaminant in the soil which indicates "serious hazard to the public health or the environment"; when this is the case, soil clean-up is required and actual exposure analyses as well as clean-up (research) must be conducted

limit value: the lowest concentration of a substance still causing changes in the physiological activity of (parts of) an organism

odour threshold value (OT-value): content of a substance in air, which induces odour nuisance when this limit is exceeded

human-toxicological recommended value TDI, TCL or 1:10^4/life risk-level

LOAEL Lowest-Observed-Adverse-Effect-Level; the lowest dose or concentration of a substance at which a statistically or biologically significant increase is observed in the frequence or seriousness of adverse effects in the exposed population in comparison with a control population

maximum tolerable risk: by the maximum tolerable risk level of a substance we mean the concentration of a substance in the environment that leads to intolerable effects for man or the environment;
for substances with a **threshold value** this has been set at the same level as the ADI, while for substances **without threshold value** it has been equated with a risk of one extra cancer case per 10,000 life-long exposed individuals (1:10^4 per life)

maximum tolerable risk for carcinogens: the amount of a substance, expressed in terms of body weight for oral exposure and in terms of air volume for inhalatory exposure, giving a risk of one extra case of cancer per 10,000 life-long exposed individuals

NOAEL No-Observed-Adverse-Effect-Level; the highest dose or concentration of a substance at which no statistically or biologically significant increase is observed in the frequence or seriousness of adverse effects in the exposed population in comparison with a control population

NOEC No-Observed-Adverse-Effect-Concentration; the highest of a series of test concentrations at which no significant effect operates on non-lethal parameters

parameter N: a multiplication factor applied to the TDI in the derivation of proposals for human toxicological C-standard values in order to express the uncertainties in the safety factors used in the derivation of the TDI, in the relation TDI- "serious danger for the public health"

potential exposure: the exposure that might occur, assuming that all exposure routes operate

risk: unwanted effects of a certain activity in relation to the possibility of this occurring

sublethal effects effects on organisms other than mortality: e.g. growth inhibitance, reproduction

toxicologically tolerated concentration in air (TCL-value): the volume of a substance expressed in terms of body weight which may be breathed in by man on a daily basis during his entire life, without damaging effects to health

toxicologically tolerable daily intake (TDI): the volume of a substance expressed in terms of body weight which may be taken up by man on a daily basis during his entire life, without damaging effects to health (equivalent of the ADI)

transfer processes: processes by which a substance may move from one compartment to the other

tumour incidence the number of animals with a tumour compared to the total number of exposed animals

Annex 1. Formulas for the calculation of actual exposure.

Annex 1.0.	Users' guideline.
Annex 1.1	Formulas for the calculation of the distribution over the soil phases.
Annex 1.2	Formulas air flux calculations.
Annex 1.3	Formulas dilution air flux outdoor.
Annex 1.4	Formulas air concentration calculation .
Annex 1.5	Formulas crop content.
Annex 1.6	Formulas for the calculation of the concentration in drinking water after permeation.
Annex 1.7	Formulas for the calculation of the air concentration in the bathroom during showering.
Annex 1.8	Formulas for the quantification of the exposure.
Annex 1.9	Primary set of data.
Annex 1.10	Conversion formulas.

Annex 1.0. Users' guideline

In report 725201006 and annex 1 to this report, tools are given for the calculation of potential and actual exposure.
In annex 1.8 formulas are given for the calculation of the exposure via all exposure routes considered, based on the contents measured or calculated in the different compartments (soil, groundwater, pore water, crops, indoor and outdoor air, drinking water). The different exposure routes are discussed consecutively.

For the calculation of the potential exposure, a nuber of substance properties (see annex 1.9) and a fixed set of soil-data, the site, and the exposure routes (contact amounts and surfaces, etc) are taken as starting-point. The default data have been included in report 725201006 (Tables 5-8).

For the calculation of the actual exposure, all parameters used may be adjusted to the actual value, provided these are known. If the concentration in one of the compartments is not known, it can be calculated by means of the formulas presented in annexes 1.1 up to 1.7. The schematic representation below indicates which parameters are relevant for which route and in which annex the formulas for the calculation of that particular parameter may be found.

exposure route	parameter	annex	parameter	annex	parameter	annex
soil ingestion					Cs	1.1
dermal contact soil					Cs	1.1
inhalation soil					Cs	1.1
inhalation indoor air	Cia	1.4	Vf	1.4		
			J	1.2	Cs Cpw Csa	1.1
indoor outdoor air	Coa	1.4	J	1.2		
			Vf	1.3	Cs	1.1
crop consumption			Cpt'	1.5	Cpw	1.1
drinking water consumption					Cpw	1.1
inhalation vapour showering	Cbk	1.7	Cdw	1.6	Cpw	1.1
dermal contact showering			Cdw	1.6	Cpw	

Annex 1.1. Formulas for the calculation of the distribution over the soil phases (solid, liquid and gaseous).

organic substances

Za = 1 / (R*T)

Za : fugacity capacity constant air [mol.m^{-3}.Pa^{-1}]
R : gas constant [8.3143 Pa.m^{3}.mol^{-1}.K^{-1}]
T : temperature [K]

Zw = S / Vp

Zw : fugacity constant water mol.m^{-3}.Pa^{-1}]
S : water-solubility [mol.m^{-3}]
Vp : vapour pressure pure substance [Pa]

Zs = Kd * SD * Zw / Vs

Zs : fugacity capacity constant soil [mol.m^{-3}.Pa^{-1}]
Kd : distribution coefficient soil-water [(mol.kg^{-1} dry soil)/(mol.dm^{-3})]
SD : mass volume of the dry soil [kg dry soil.dm^{-3} humid soil]
Vs : volume fraction solid phase [-]

Vs = 1 - porosity
= 1 - Va- Vw

Va : volume fraction air [-]
Vw : volume fraction water [-]

Kd = Koc * foc

Koc : distribution coefficient soil-water corrected for organic carbon [(mol.kg^{-1} org. C)/(mol.dm^{-3})]

foc : fraction organic carbon [kg org.C.kg^{-1} dry soil]

Koc = 0.411 * Kow or: log Koc = 0.989 * log Kow - 0.346

Kow : octanol-water distribution coefficient [(mol.dm^{-3})/(mol.dm^{-3})]

H = Za / Zw
= Vp / (S * R * T)

H : dimensionless Henry-constant [-]
[(mol.m^{-3} air)/(mol.m^{-3} water)]

for dissociating substances

Kd = Kd * fnd
fnd : fraction non-dissociated substance

fnd $= 1 / (1 + 10^{[pH - pKa]})$

pH : acidity of the soil (4-8, as default: 6)
pKa : acidity dissociation constant of the substance

Calculation mass fractions

Pa $= (Za * Va) / (Za * Va + Zw * Vw + Zs * Vs)$
Pw $= (Zw * Vw) / (Za * Va + Zw * Vw + Zs * Vs)$
Ps $= (Zs * Vs) / (Za * Va + Zw * Vw + Zs * Vs)$

Pa : mass fraction in soil air [-]
Pw : mass fraction in soil moisture [-]
Ps : mass fraction in solid phase soil [-]

<u>inorganic substances</u>

$Za = 0; Csa = 0; Pa = 0$

$Pw = 1; Ps = 0$

<u>metals and arsenic</u>

$Za = 0; Csa = 0; Pa = 0$

$Pw = Vw / (Vw + Kd * SD)$

$Ps = 1 - Pw$

calculation concentrations

Csa $= Cs * SD * Pa / Va$

Csa : soil air concentration [mol.dm^{-3} or g.m^{-3}]
Cs : initial soil content [mol.kg^{-1} dry soil or mg.kg^{-1}]
Cpw $= Cs * SD * Pw / Vw$

if : $Cpw > S$ then: $Cpw = S$
$Csa = S * Vw * Pa / (Pw * Va)$

Cpw : soil moisture concentration [mol.dm^{-3} or g.m^{-3}]

Annex 1.2. Formulas air flux calculations.

Dsa $= Va^{10/3} * Da / (1-Vs)^2$ with $Da = .036 * (76/M)^{1/2}$

Dsa : diffusion coefficient in the soil-gas phase $[m^2.h^{-1}]$
Da : diffusion coefficient in free air $[m^2.h^{-1}]$
M : molecular mass $[g.mol^{-1}]$

Dsw $= Vw^{10/3} * Dw / (1-Vs)^2$ with $Dw = 3.6 * 10^{-6} * (76/M)^{1/2}$

Dsw : diffusion coefficient in the soil-water phase $[m^2.h^{-1}]$
Dw : diffusion coefficient in free water $[m^2.h^{-1}]$

Du $= (Pa * Dsa / Va) + (Pw * Dsw / Vw)$

Du : diffusion coefficient in the soil $[m^2.h^{-1}]$

J2 $= Da * Csa/d$
J3 $= Cpw * Ev / 24$
J4 $= Du * Cs * SD / dp$

J2 : boundary layer flux $[g.m^{-2}.h^{-1}]$
J3 : water-evaporation flux $[g.m^{-2}.h^{-1}]$
J4 : diffusion flux water-soil $[g.m^{-2}.h^{-1}]$
Ev : flux of evaporating water [Fast et al., 1987: 0.0001 $m^3.m^{-2}.d^{-1}$]
d : thickness boundary layer [Jury et al., 1985: 0.005 m]
dp : mean depth of the contamination [0.75 m]

if : $J3+J4 < J2$ then: $J = J4 + J3$
if : $J3+J4 >= J2$ then: $J = J2$

J : total soil flux $[g.m^{-2}.h^{-1}]$

Annex 1.3. Formulas dilution air flux outdoor.

SOILRISK (applied in CSOIL)

$Vf = Vg * Sz / Lp$

Vf : dilution velocity [$m.h^{-1}$]
Vg : mean wind velocity [$m.h^{-1}$]
Sz : vertical Pasquill dispersion coefficient, related to Pasquill weather stability class D [m]
Lp : diameter contaminated soil [100 m]

$Vg = (Vx + V')/2$

Vx : wind velocity at x m altitude [$m.h^{-1}$]
V' : friction velocity [$m.h^{-1}$]

$Vx = \ln [Z/Zo] * V' / k$

Z : breathing altitude [x m]
Zo : roughness of the surface area (residential area) [1.0 -]
k : Karman constant [0.4 -]

$V' = k * V10 / \ln (Z10/Zo)$

Z10 : altitude [10 m]
V10 : wind velocity at 10 m altitude [18,000 $m.h^{-1}$]

$Sz = Co * 0.2 * Lp^{0.76}$

$Co = (10 * Zo)$ ^ $(0.53 * Lp^{-0.22})$

Co : correction factor for the roughness length [-]

model calculation for Lp = 100 m:

		children	adults	
Z	=	1.0	1.75	[m]
V'	=	3,127	3,127	[$m.h^{-1}$]
V1	=	0	4,375	[$m.h^{-1}$]
Vg	=	1,564	3,751	[$m.h^{-1}$]
Co	=	1.56	1.56	[-]
Sz	=	10.31	10.31	[m]
Vf	=	161.3	386.9	[$m.h^{-1}$]

HESP

$$Vf = 1 / [1/Kg - 1/Kgs - \ln(Y/Xa) / (Prc * V')]$$

Vf : dilution velocity $[m.h^{-1}]$
J : flux from the soil $[g.m^{-2}.h^{-1}]$
Kg : gas phase mass transport coefficient $[m.h^{-1}]$
Kgs : mass transport coefficient for diffuse substrata $[m.h^{-1}]$
Y : breathing zone altitude above soil surface [1.5 m]
Xa : thickness of the air boundary layer [m]
Prc : Prandtl constant [0.4 -]
V' : friction velocity air $[m.h^{-1}]$

$$Kg = 0.029 * V10^{0.78} * L^{-0.11} * Sc^{-0.67}$$

V10 : wind velocity at 10 m altitude $[27000\ m.h^{-1}]$
L : length of the intestinal surface [300 m]
Sc : Schmidt-figure for the gas phase [-]

$$Sc = u / (P * Da)$$

u : air viscosity $[65.8\ g.m^{-1}.h^{-1}]$
P : air density $[1280\ g.m^{-3}]$
Da : air density coefficient $[m^2.h^{-1}]$

$$Kgs = Da / Xa$$

$$Xa = 26 * Sc^{1/3} * Vk / V'$$

Vk : kinetic viscosity of air $[0.05137\ m^2.h\text{-}^{-1}]$

$$V' = (V10 * k) / \ln[(h+sr)/sr]$$

k : Karman constant [0.4 -]
h : altitude [10 m]
sr : surface roughness [0.28 -]

calculation for Da = $0.0295\ m^2.h^{-1}$:

Sc = 1.74 [-]
Kg = 30.53 $[m.h^{-1}]$
Xa = 0.00037 [m]
Kgs = 79.67 $[m.h^{-1}]$
V' = 2,997 $[m.h^{-1}]$
Vf = 75.4 $[m.h^{-1}]$

Annex 1.4. Formulas air concentration calculation.

$Coa = J / Vf$

Coa : concentration in the outdoor air [g.m^{-3}]
Vf : dilution velocity [m.h^{-1}]

$Cba = J * Bo / (Bv * Vv)$

Cba : concentration in the creeping-space air [g.m^{-3}]
Bo : surface area of the creeping-space [50 m^{2}]
Bv : volume of the creeping-space [25 m^{3}]
Vv : ventilation multiplication [1.25 h^{-1}]
(experimental data (Fast et al., 1987): min. 0.03 - max. 7.4 - av. 1.25)

$Cia = fbi * Cba$

Cia : concentration in the indoor air [g.m^{-3}]
fbi : contribution of the creeping-space to the indoor air as fraction [0.1 -]
(experimental data (Fast et al., 1987): min. 0 - max. 0.68 - av. 0.11 - 95 perc. 0.39)

if : $Cia < Coa$, then: $Cia = Coa$

Annex 1.5. Formulas crop content.

Deposition

$$Cdp = TSPo * DRo * frs * Cs * [\, fin / (Yv * fEi) \,] * \{ 1 - (1 - \exp[- fEi * te]) / (fEi * te) \}$$

Cdp : crop content compared to deposition [mg subst.kg^{-1} dw crop]
fin : fraction interception by crop [0.4 -]
Yv : crop yield [0.28 kg dw.m^{-2}]
fEi : efflorescence constant [0.033 d^{-1}]
TSPo : concentration particles in outdoor air [0.05 mg.m^{-3}]
frs : fraction soil in particles [0.5 -]
DRo : deposition velocity [1 m.s^{-1} = 864 m.d^{-1}]
te : growing period crop [180 d]

$$Cdp = 0.778 * 10^{-3} * Cs$$ [mg.kg^{-1}]

Metals

$$\ln BCFpl = 2.67 - 1.12 * \ln Kd$$

BCFpl : bioconcentration factor plant [(mg subst.kg^{-1} dw crop)/(mg subst.kg^{-1} dry soil)]

$$Cpl = BCFpl * Cs$$

Cpl : content in crop [mg.kg^{-1} dw]

Other inorganic substances

$BCFpl = 4$
$Cpl = BCFpl * Cpw$
Cpl : content in crop [mg.kg^{-1} dw]

Organic substances

root : $BCFr = 10^{(0.77 * \log Kow - 1.52)} + 0.82$

BCFr : bioconcentration factor root [(mg subst.kg^{-1} fresh crop)/(mg subst.dm^{-3} soil moisture)]

stem : BCFs: $[10^{(0.95 * \log Kow - 2.05)} + 0.82] *$
$[0.784 * 10\hat{}\ (-0.434 * (\log Kow - 1.78)^2/2.44)]$

BCFs : bioconcentration factor stem [(mg subst.kg^{-1} fresh crop)/(mg subst.dm^{-3} soil moisture)]

$$Cpl = 0.5 * (BCFr + BCFs) * Cpw / fdw$$

fdw = ratio dry weight -fresh weight of the crop [0.2 kg dw.kg^{-1} fresh]

Total

Cpt = Cpl + 0.5 * Cdp

Cpt : content of the crop [mg.subst.kg^{-1} dw crop]

or : Cpt' = Cpt * fdw [mg.subst.kg^{-1} fresh weight crop]

Annex 1.6. Formulas calculation concentration in drinking water after permeation.

$$Cmax = Cpw * 2 * Dpe * t / (r * d)$$

Cmax : maximum concentration in the drinking water after t days stagnation [$mg.dm^{-3}$]
Dpe : permeation coefficient [$m^2.d^{-1}$]
t : time period of water stagnation [Vonk, 1985: 8 h = 1/3 d]
r : radius of the pipe [0.0098 m]
d : thickness of pipe wall [0.0027 m]

24 hrs permeation:

$$Cdw = Cmax * 3 * pi * r^2 * L / Qwd$$

Cdw : 24-hrs mean drinking water concentration [$mg.dm^{-3}$]
t : in this case 24 h = 1 d, hence correction factor 3
L : length of the pipe along which permeation can occur = size location [100 m]
Qwd : mean daily water consumption [500 dm^3]

$$Cdw = 4.56 * 10^3 * Dpe * Cpw$$ [$mg.dm^{-3}$]

Annex 1.7. Formulas for the calculation of air concentration in the bathroom during showering.

kwa $= [(H/RT) * kL * kG]/[(H/RT) * kG + kL)] * [Ad/Vd] * tf$

kwa : degree of evaporation of the contaminant [-]
(H/RT) : diminesionless Henry-constant at a temperature of 313 K [-]
kL : water mass transport coefficient [$m.s^{-1}$]
kG : vapour mass transport coefficient [$m.s^{-1}$]
Ad : surface area water drop [m^2]
Vd : volume water drop [m^3]
tf : dropping time drop [1 s]

Ad $= 4 * pi * r^2$
Vd $= 4/3 * pi * r^3$
Ad/Vd $= 3 / r$

r : radius of the drop [0.5 mm = 0.0005 m]

kL $= Kl * (44/M)^{1/2} / 3600$ [$m.s^{-1}$]
kG $= Kg * (18/M)^{1/2} / 3600$ [$m.s^{-1}$]

Kl : liquid phase exchange velocity [$0.2\ m.h^{-1}$]
Kg : gas phase mass transport coefficient [$29.88\ m.h^{-1}$]

Cbk $= kwa * Vwb * Cdw / (2 * Vbk)$

Cbk : concentration in the bathroom air [g $subst.m^{-3}$]
Vwb : volume water consumption [$0.15\ m^3$]
Vbk : volume bathroom [$l*b*h = 3*2*2.5 = 15\ m^3$]

Cbk $= kwa * Cdw * 5 * 10^{-3}$ [$g.m^{-3}$]

Annex 1.8. Formulas quantification exposure.

Soil ingestion

DI = AID * Cs * fa/W

DI : uptake via soil-ingestion [mg subst.kg^{-1}.d^{-1}]
AID : daily intake amount soil [kg dry subst.d^{-1}]
adults : $50 * 10^{-6}$
children : $150 * 10^{-6}$
W : body weight (Table 3)
fa : absorption factor, set at 1 [-]
Cs : content in the soil [mg.kg^{-1}]

DIa = $50 * 10^{-6}$ * Cs / Wa
DIc = $150 * 10^{-6}$ * Cs / Wc

Dermal contact soil

DA = A exp * fm * DAE * DAR * Cs * t * ft / W

DA : uptake via dermal contact with soil [mg.kg^{-1}.d^{-1}]
A exp : exposed surface area [m^2]

	indoor	outdoor
adult	0.09	0.17
children	0.05	0.28

fm : matrix factor [0.15 -]
DAE : degree of coverage [mg soil.cm^{-2} = * 0.01 kg.m^{-2}]

	indoor	outdoor
adults	0.056	3.75
children	0.056	0.51

DAR : absorption velocity [h^{-1}]

adults	0.005
children	0.01

t : duration of exposure [h d^{-1}]

	indoor	outdoor
adults	12	8
children	12	12

tf : time fraction exposure [-]
correction from daily to annual average exposure

	indoor	outdoor
adults	1.0	0.12
children	1.0	0.21

DAa = $5.1 * 10^{-6}$ * Cs / Wa
DAc = $5.9 * 10^{-6}$ * Cs /Wc

Inhalation of soil (particles)

IP = ITSP * Cs * fr * fa/W

IP : uptake of inhaled soil particles [$mg.kg^{-1}.d^{-1}$]
ITSP : amount of inhaled soil particles [$kg.d^{-1}$]
fr : retention factor particles in lung [0.75 -]

ITSP = TSP * frs * AV * t * ft

TSP : amount of air-suspended particles [$mg.m^{-3}$]
TSPo : outdoor: 70 $ug.m^{-3}$
TSPi : indoor : 0.75 * 70 = 52.5 $ug.m^{-3}$
frs : soil-fraction in particles in the air [-]
frso: outdoor: 0.5
frsi: indoor : 0.8
AV : breathing volume [m^3d^{-1}]
AVa: adults : 20 = 0.83 $m^3.h^{-1}$
AVc: children : 7.6 = 0.32 $m^3.h^{-1}$
t : duration of exposure [h]

	indoor	outdoor
adults	8	8
children	16	8

tf : time fraction exposure [-]
correction from daily to annual average exposure

	indoor	outdoor
adults	2.14	0.14
children	1.32	0.36

IPa = 0.40 * 10^{-6} * Cs / Wa
IPc = 0.20 * 10^{-6} * Cs / Wc

Inhalation of air

IV = [AV * Cia * tfi * ti + AV * Coa * tfo * to] * fa * 10^3/ W

IV : uptake of contamination via vapours $mg.kg^{-1}.d^{-1}$]
Cia : concentration in the indoor air [$g.m^{-3}$]
Coa : concentration in the outdoor air [$g.m^{-3}$]
ti/to : daily duration of exposure indoor resp. outdoor (see Table under inhalation soil particles [h]
tfi/tfo : annual average time fractions for stay indoor resp. outdoor (see Table under inhalation of soil particles) [-]

IVa = (17.12 * Cia + 1.12 * Coa) * 0.83 * 1000 / Wa
IVc = (21.12 * Cia + 2.88 * Coa) * 0.32 * 1000 / Wc

Ingestion of crops: vegetables and fruit

$VI = (Qfv * Cpt' * fv * fa / W$

VI : uptake of contamination via crop [mg.kg^{-1}.d^{-1}]
Qfv : daily consumption of crop [kg dry weight.d^{-1}]
 adults : 0.558
 children : 0.295
fv : fraction contaminated food [0.1 -]
Cpt' : content of the crop [mg.subst.kg^{-1} fresh weight]

$VIa = 0.0558 * Cpt' * Cpt' / Wa$
$VIc = 0.0295 * Cpt' * Cpt' / Wc$

Ingestion of drinking water

$DIw = Qdw * Cdw * fa / W$

DIw : intake of contaminated drinking water by drinking [mg.kg^{-1}.d^{-1}]
Qdw : daily drinking water intake [m^3.d^{-1}]
 adults : 0.002 = 2 dm^3.d^{-1}
 children : 0.001 = 1 dm^3.d^{-1}
Cdw : drinking water concentration [mg dm^{-3} = g.m^{-3}]

$DIwa = 2 * Cdw / Wa$
$DIwc = 1 * Cdw / Wc$

Inhalation of vapours during showering

$IVw = Cbk * AV * td * fa * 1000 / W$

IVw : intake contamination by inhaling vapour during showering [mg.kg^{-1}.d^{-1}]
Cbk : concentration in the bathroom air [g.m^{-3}]
td : duration of stay in the bathroom [h]
 15 min. showering + 15 min. drying/dressing = [0.5 h]

$IVwa = 2.08 * Cdw * kwa / Wa$
$IVwc = 0.792 * Cdw * kwa / Wc$

Dermal contact during showering

$DAw = A\ tot * f\ exp * DAR * tdc * (l-kwa) * Cdw * fa / W$

DAw : uptake as a result of dermal contact during showering [mg.kg^{-1}.d^{-1}]
A tot : exposure surface area = entire body [m^2]
 adults : 1.8
 children : 0.95
f exp : fraction exposed skin [0.4 -]
DAR : dermal absorption velocity [(mg.m^{-2})/(mg.dm^{-3}).h^{-1}]

tdc : contact time = showering time [0.25 $h.d^{-1}$]
(1-kwa) : substance remaining in water after evaporation [-]

DAR = P1 * P2 according to Fiserova-Bergerova and Pierce (1989), with:

P1 = (0.038 + 0.153 * Kow)
P1 = 5000 * P1 / (5000 + P1), maximation according to Braun
P2 = exp (-0.016 * M) / 1.5

DAwa = 0.18 * DAR * (1-kwa) * Cdw / Wa
DAwc = 0.095 * DAR * (1-kwa) * Cdw / Wc

<u>(Dermal contact during bathing)</u>

DAw = A tot * DAR * tb * Cdw * fa / W

DAR : dermal absorption velocity [10 $(mg.m^{-2})/(mg.dm^{-3}.h^{-1}$]
tb : contact time = duration of bathing [0.5 $h.d^{-1}$]

Total exposure

TAD = DIa + DAa + IPa + IVa + VIa + DIWa + IVwa + DAwa
TCH = DIc + DAc + IPc + IVc + VIc + DIWc + IVwc + DAwc

TAD : daily exposure adult [$\mathrm{mg.kg^{-1}.d^{-1}}$]
TCH : daily exposure child [$\mathrm{mg.kg^{-1}.d^{-1}}$]

DOSE = (6 * TCH + 64 * TAD) / 70

DOSE : life-time mean daily exposure [$\mathrm{mg.kg^{-1}.d^{-1}}$]

Annex 1.9 Primary set of data.

Necessary set of data in order to be able to calculate potential exposure.

S	: water-solubility	[$mol.m^{-3}$] or [$mg.l^{-1}$]
Vp	: vapour pressure pure substance	[Pa] or [atm] or [mm Hg]
	or	
H	: dimensionless Henry-constant (this may also be calculated from the other data, see annex 1.10)	[-]
M	: molecular mass	[$g.mol^{-1}$]
Kow	: octanol-water distribution coefficient (for organic substances)	
	or	
Kd	: soil-water distribution coefficient (for metals)	[$dm^3.kg^{-1}$]
Dpe	: permeation coefficient	[$m^2.d^{-1}$]
pKa	: acidity dissociation constant (if relevant!)	[-]

Annex 1.10 Conversion formulas

H = $Vp / (S * R * T)$

Vp = $H * S * R * T$

R1 = 8.3144 $Pa.m^3.mol^{-1}.K^{-1}$ or: $8.206*10^{-5}$ $m^3.atm.mol^{-1}.K^{-1}$

if H is given as [$Pa.m^3.mol^{-1}$] then: $H = H / (R * T)$
if H is given as [$atm.m^3.mol^{-1}$] then: $H = H * 9.87*10^6 / (R*T)$

1 mm Hg-pressure = 133.33 Pa = $1.316*10^{-3}$ atm

if S is given in [$mg.l^{-1}$] then: $S = S / M$ [$mol.m^{-3}$]

M : molecular mass [$g.mol^{-1}$]

foc = 0.58 * fom or fom = 1.724 * foc or OS = 172.4 * foc

fom : fraction organic substance mass
OS : percentage organic substance

Locher and Bakker (1987)

Annex 2 **Revised proposals* for the integrated intervention values for a standard soil (10% organic matter, 25% lutum), based on ecotoxicological and human-toxicological intervention values, and compared with the current C-values. The uncertainty scores have also been given, as well as the human-toxicological MTR (maximum tolerable risk level)**

Compound	Ecotox. int.-value $[mg.kg^{-1}]$	US[1]	Human MTR value $[\mu g.kg^{-1}.d^{-1}]$	Human-tox. int.-value $[mg.kg^{-1}]$	US[1]	Integrated int.-value soil $[mg.kg^{-1}]$	Current C-value soil $[mg.kg^{-1}]$	Proposed int.-value soil $[mg.kg^{-1}]$	Proposed int.-value groundwater $[\mu g.l^{-1}]$
I Metals and trace elements									
arsenic	40	6	2.1	680	3	40	50	55	55
barium	625	2	20	4300	3	625	2000	625	605
cadmium	12	10	1	35	3	12	20	12	6
chromium	230	7	5	2250	2	230	800	380	26
cobalt	240	2	1.4	450	1	240	300	240	44
copper	190	10	140	16000	3	190	500	190	35
mercury	10	6	0.61	200	3	10	10	10	0.3
lead	290	10	3.6	300[2]	3	290	600	530	54
molybdenum	<480	2	10	910	3	480	200	200	310
nickel	210	6	50	6600	3	210	500	210	37
tin	-	-	2000	650000	3	300	300	-	-
zinc	720	6	1000	56000	3	720	3000	720	290
II Inorganic compounds									
cyanides (free)	-	-	50	18	3	18	100	20	1500
cyanides (complex, pH<5)	-	-	13	4.6	2	5	500	650	1500
cyanides (complex, pH≥5)	-	-	13	4.6	2	5	500	50	1500
thiocyanates	-	-	11	3.9	3	3.9	200	20	1500
III Aromatic compounds									
benzene	25	2	4.3	1.1	6	1.1	5	1	30
ethyl benzene	-	-	136	50	4	50	50	50	150
phenol	40	2	60	46	6	40	5	40	1900
cresol	50	2	50	4.7	4	5	5	5	200
toluene	130	64	430	340	6	130	30	130	1040
xylene	-	-	10	26	4	26	50	25	66
catechol	-	-	40	23	4	23	-	23	1300
resorcinol	-	-	20	10	4	10	-	10	630
hydroquinone	-	-	25	11	4	11	-	11	780
IV Polycyclic aromatic hydrocarbons									
anthracene	40	5	50	29000	3	40	100	-	5
benzo(a)anthracene	40	5	20	11000	6	40	50	-	0.4
benzo(k)fluoranthene	40	5	20	11000	6	40	50	-	0.02
benzo(a)pyrene	40	5	2	1000	6	40	10	-	0.03
chrysene	40	5	2	360	4	40	50	-	0.04
phenanthrene	40	5	20	630	4	40	100	-	5
fluoranthene	40	5	20	1000	4	40	100	-	0.8
indeno(1,2,3cd)pyrene	40	5	20	12000	6	40	50	-	0.01[dl]
benzo(*ghi*)perylene	40	5	20	12000	6	40	100	-	0.03
naphthalene	40	5	50	590	4	40	50	-	75
total PAHs (10)								40	-

dl = adjusted for the detection limit; - = no data;

[1] uncertainty score (the higher the score the lower the uncertainty); [2] TDI for a child

* The original proposals of Van den Berg and Roels (1991) have been revised, and will be published in a revised report (Van den Berg and Roels, 1993). The Dutch parliament will decide on the final values to be used.

Annex 2 continued **Revised proposals* for the integrated intervention values for a standard soil (10% organic matter, 25% lutum), based on ecotoxicological and human-toxicological intervention values, and compared with the current C-values. The uncertainty scores have also been given, as well as the human-toxicological MTR (maximum tolerable risk level)**

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V Chlorinated hydrocarbons									
Aliphatic chlorohydrocarbons									
1,2-dichloroethane	60	5	14	3.9	6	4	50	3.5	430
dichloromethane	60	5	60	19	6	19	50	20	1000
tetrachloromethane	60	5	4	0.92	4	1	50	1	10
tetrachloroethene	60	5	16	3.9	4	4	50	4	41
trichloromethane	60	5	30	8.9	4	9	50	10	380
trichloroethene	60	5	540	300	6	60	50	60	480
vinyl chloride	60	5	3.5	0.078	6	0.1	-	0.1	0.7
Chlorobenzenes									
monochlorobenzene	30	5	300	530	6	30	10	-	190
dichlorobenzene (ind.)	30	5	190	1100	6	30	10	-	50
trichlorobenzene (ind.)	30	5	0.5	8.8	4	9	10	-	3
tetrachlorobenzene (ind.)	30	5	0.5	17	4	17	10	-	1.4
pentachlorobenzene	30	5	0.5	23	4	23	10	-	0.6
hexachlorobenzene	30	5	0.5	25	6	25	10	-	0.5
total chlorobenzenes								30	-
Chlorophenols									
monochlorophenol (ind.)	10	5	3	14	4	10	5	-	95
dichlorophenol (ind.)	10	5	3	32	6	10	5	-	30
trichlorophenol (ind.)	10	5	3	54	4	10	5	-	6
tetrachlorophenol (ind.)	10	5	3	18	4	10	5	-	54
pentachlorophenol	5	5	30	76	6	5	5	5	3
total chlorophenols (except penta)								10	-
Chlorinated polycyclic aromatic hydrocarbons									
chloronaphthalene	-	-	0.5	8.8	2	9	5	9	5
Polychlorobiphenyls									
trichlorobiphenyl	70	2	0.09	5.2	4	1[2]	5		
hexachlorobiphenyl	70	2	0.09	8.3	4	1[2]	5		
total chlorinated biphenyls (7)								1	0.01

dl = adjusted for the detection limit; - = no data

[1] uncertainty score (the higher the score the lower the uncertainty)

* The original proposals of Van den Berg and Roels (1991) have been revised, and will be published in a revised report (Van den Berg and Roels, 1993). The Dutch parliament will decide on the final values to be used.

Annex 2 continued **Revised proposals* for the integrated intervention values for a standard soil (10% organic matter, 25% lutum), based on ecotoxicological and human-toxicological intervention values, and compared with the current C-values. The uncertainty scores have also been given, as well as the human-toxicological MTR (maximum tolerable risk level).**

Compound	Ecotox. int.-value [mg.kg^{-1}]	US[1]	Human MTR value [μg.kg^{-1}.d^{-1}]	Human-tox. int.-value [mg.kg^{-1}]	US[1]	Integrated int.-value soil [mg.kg^{-1}]	Current C-value soil [mg.kg^{-1}]	Proposed int.-value soil [mg.kg^{-1}]	Proposed int.-value groundwater [μg.l^{-1}]
VI Pesticides									
Organochlorine pesticides									
DDT	4	5	20	11000	9	4	5	-	0.01
DDE	4	5	20	750	6	4	5	-	0.03
total DDT/DDD/DDE								4	0.01
aldrin	0.35	6	0.1	13	4	0.4	5	-	0.005^{dl}
dieldrin	4	5	0.1	5.2	4	4	5	-	0.07
endrin	0.06	2	0.1	4.1	4	4	5	-	0.2
total drins								4	0.1
α-HCH	2	6	1	20	6	2	5	-	1.1
β-HCH	-	-	0.2	0.4	4	0.4	5	-	0.2
γ-HCH	2	6	1	20	6	2	5	-	1.1
δ-HCH	2	6	1	24	6	2	5	-	0.7
total HCHs								2	1
Other pesticides									
carbaryl	5	7	10	440	6	5	10	5	0.09
carbofuran	1.5	10	10	410	6	1.5	10	1.5	0.07
propoxur	0.75	2	20	650	6	22	10	22	2.6
maneb	35	4	50	29000	6	35	10	35	0.1^{dl}
atrazine	6	7	5	21	6	6	10	6	160
VII Other pollutants									
heptane	-	-	3100	3.5	2	3.5	-		
octane	-	-	3100	2.5	2	2.5	-		
mineral oil							5000	5000	600
cyclohexanone	-	-	4600	270	6	270	60	270	15500
butyl benzylphthalate	60	5	25	740	6	60	250	-	9.1
di(2-ethylhexyl)phthalate	60	5	25	4000	6	60	250	-	1.9
total phthalates								60	5
pyridine	150	2	1	1.1	4	1	20	1	2.9
styrene	-	-	77	100	6	100	50	102	350
tetrahydrofuran	-	-	10	0.4	4	0.4	40	0.4	1.0
tetrahydrothiophene	-	-	180	94	4	94	50	94	34

dl = adjusted for the detection limit; - = no data

[1] uncertainty score (the higher the score the lower the uncertainty)

* The original proposals of Van den Berg and Roels (1991) have been revised, and will be published in a revised report (Van den Berg and Roels, 1993). The Dutch parliament will decide on the final values to be used.